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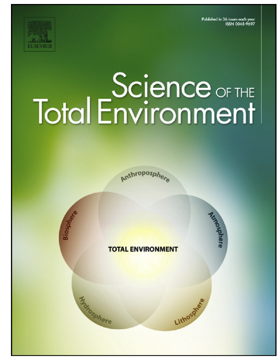

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Comparing soil inventory with modelling: Carbon balance in Central European forest soils varies among forest types

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Abstract

Forest soils represent a large carbon pool and already small changes in this pool may have an important effect on the global carbon cycle. To predict the future development of the soil organic carbon (SOC) pool, well-validated models are needed. We applied the litter and soil carbon model Yasso15 to 1838 plots of the German national forest soil inventory (NFSI) for the period between 1985 and 2014 to enables a direct comparison to the NFSI measurements. In addition, to provide data for the German Greenhouse Gas Inventory, we simulated the development of SOC with Yasso15 applying a climate projection based on the RCP8.5 scenario. The initial model-calculated SOC stocks were adjusted to the measured ones in the NFSI.

On average, there were no significant differences between the simulated SOC changes (0.25 ± 0.10 Mg C ha^{-1} a^{-1}) and the NFSI data (0.39 ± 0.11 Mg C ha^{-1} a^{-1}). Comparing regional soil-unit-specific aggregates of the SOC changes, the correlation between both methods was significant (r^2=0.49) although the NFSI values had a wider range and more negative values. In the majority of forest types, representing 75 % of plots, both methods produced similar estimates of the SOC balance. Opposite trends were found in mountainous coniferous forests on acidic soils. These soils had lost carbon according to the NFSI (-0.89 ± 0.30 Mg C ha^{-1} a^{-1}) whereas they had gained it according to Yasso15 (0.21 ± 0.10 Mg C ha^{-1} a^{-1}). In oligotrophic pine forests, the NFSI indicated high SOC gains (1.36 ± 0.17 Mg C ha^{-1} a^{-1}) and Yasso15 much smaller (0.29 ± 0.10 Mg C ha^{-1} a^{-1}).

According to our results, German forest soils are a large carbon sink. The application of the Yasso15 model supports the results of the NFSI. The sink strength differs between forest types possibly because of differences in organic matter stabilisation.

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Keywords: Soil carbon changes, soil organic carbon, climate, soil inventory, Yasso15, litter and soil carbon model, temperate forests

1. Introduction

Globally, forests are a carbon sink (Pan et al., 2011) that needs to be maintained and strengthened to compensate for anthropogenic greenhouse gas emissions and to mitigate climate change. Forest soils have an important role in the global carbon cycle and they contain more carbon than the terrestrial biomass pool or the atmosphere (Batjes, 1996). Consequently, even small changes in the soil carbon pool can have a substantial impact on the carbon sink (Paustian et al., 2016).

In the United Nations Convention on Climate Change (UNFCCC), countries have agreed to report the annual carbon stock changes of soils under different land uses and under land-use change as a part of their greenhouse gas inventories (IPCC, 2006). A straightforward method to fulfil this commitment of the UNFCCC is a repeated large-scale soil inventory. This method was applied in Germany, where carbon stock changes were estimated by the National Forest Soil Inventory (NFSI) (Wellbrock et al., 2016). Repetition of the NFSI in Germany revealed that forest soils act with 0.39 Mg C ha^{-1} a^{-1} as a considerable carbon sink between 1990 and 2008 (Grüneberg et al., 2014). Earlier studies deliver results from soil inventories for other European countries. The measured soil carbon stock changes may vary from carbon losses in England and Wales (Bellamy et al., 2005) to no significant changes in Denmark (Callesen et al., 2015) or with 0.12 Mg C ha^{-1} a^{-1} slight increases in Finland (Rantakari et al., 2012), and with 0.35 Mg C ha^{-1} a^{-1} up to large soil carbon accumulation in France (Jonard et al., 2017).

The NFSI measurements provide information on the current soil carbon stocks but are not able to predict soil carbon changes under changing climate. In the context of the German greenhouse gas reporting system a near-future scenario of soil carbon stock development is necessary to calculate the Forest Management Reference Level (IPCC, 2014). This serves as baseline, against which the net emissions and removals reported for Forest Management, will be compared for accounting purposes. On the basis of the NFSI data, soil carbon changes can be reported only by extrapolating until the NFSI is repeated again, which is planned for 2022. The repetition of the NFSI shows changes in soil carbon stocks only in longer time periods due to the slow turnover rate of soil carbon stocks and measurement uncertainties. Calculating soil carbon stocks requires the determination of soil carbon concentrations, bulk densities, stone contents and soil depth, which all vary depending on site and have associated different measurement errors (Schrumpf et al., 2011). Due to the uncertainties of a soil inventory, the annual change rates need to be relatively high to be detected; for example, the Danish inventory reveals that annual changes must exceed 0.15 Mg C ha^{-1} a^{-1} to be detected by the inventory (Callesen et al., 2015). Furthermore, inventories allow the impact of environmental conditions as well as different management practices to be addressed only if these were surveyed during the inventory. Consequently, investigations that go beyond the conditions of the inventory or projections of soil carbon development can be made only by extrapolation. In contrast, soil carbon models allow the investigation of the effect of changing climate in combination with different management practices on soil carbon dynamics under various scenarios.

Upscaling soil organic matter (SOM) cycling in a model from fine-scale processes to soil profile or landscape level is challenging (Schmidt et al., 2011). As models which are applied on higher spatial scales usually operate with average environmental conditions, their results could be biased due to fine-scale heterogeneity. In practice, a model should be validated with measured data to assess whether it is able to predict soil carbon stocks and their changes across the entire region of interest. On a global scale, comparisons of soil organic carbon (SOC) stock predictions by various earth-system models have resulted in high variation and deviations from global SOC databases (Todd-Brown et al., 2013). On a regional scale, a comparison among models revealed that soil fertility affects the relationship between modelled SOC stocks and measured SOC stocks (Tupek et al., 2016). The lack of conformance on nutrient rich sites in that study was attributed to the lack of mechanisms that model mycorrhizal organic uptake and organo-mineral stabilization processes. There are even fewer studies that compare modelled SOC stock changes with measurements. Studies from Finland and Sweden comparing modelled SOC stock changes and large scale inventories show SOC changes of similar order of magnitude (Rantakari et al., 2012, Ortiz et al., 2013). However, previous studies comparing model and measured estimates of soil carbon stock changes have not investigated the model performance for different forest types in detail. Soil carbon stocks and their changes are related to forest types because on the one hand the distribution of forest types depends on site properties like climate, soil texture and soil fertility, which influence soil carbon balance (Guckland et al., 2009). On the other hand tree species composition and diversity influence soil carbon balance in various ways. Species-specific properties in rooting patterns (Finér et al., 2017), litter-chemistry (Vesterdal et al., 2008) and tree architecture (Joly et al., 2017) influence litter input, litter decomposition and carbon stabilisation (Schleuß et al., 2014). Species diversity may lead to increased ecosystem productivity by niche partitioning, which increase litter input, e.g. inter-specific competition increases fine root biomass (Bolte & Villanueva, 2006). Consequently, SOC stocks are related to tree diversity (Gamfeldt et al., 2013). Therefore, the stratification of a large scale soil inventory into forest types and the comparison of measured and modelled SOC balances will provide a reference for future research of soil carbon dynamics.

We used the Yasso litter and soil carbon model in our study. This model is a widely used litter and soil carbon model (Tuomi et al., 2008, Tuomi et al., 2009, Tuomi et al., 2011a, Järvenpää et al., 2018). On average, Yasso has proven to perform well in various comparisons among models (Lehtonen et al., 2016) or in litter decomposition studies (Didion et al., 2016). It has been applied in different European countries (Thürig et al., 2005, Liski et al., 2006, Rantakari et al., 2012, Ortiz et al., 2013, Dalsgaard et al., 2016, Hernandez et al., 2017), used in several national greenhouse gas inventories (www.en.ilmatieteenlaitos.fi/yasso) and has been implemented in the land surface scheme of the MPI Earth System Model (Goll et al., 2015, 2017). The Yasso15 is a new version of the model that is parameterized over a wide range of site conditions and was recently validated with litter-bag decomposition studies (Didion et al., 2016).

The aim of our study is to compare modelled and measured soil carbon balances based on a large scale inventory. The comparison is conducted for the whole of Germany and stratified into forest types or soil units. Furthermore, this study provides data for Forest Management Reference Level for the German Greenhouse Gas inventory. We hypothesized that i) forest types dominated by coniferous trees and/or stocking on base poor soils show higher soil carbon stock changes compared to forest types dominated by broadleaved trees and stocking on base rich sites, ii) Yasso15-results differ most from NFSI-results on sites where NFSI indicates high soil carbon stock changes.

2. Material and methods

2.1. The National Forest Soil Inventory (NFSI)

In Germany the National Forest Soil Inventory (NFSI) is a nationwide survey on a systematic 8 km x 8 km grid on approximately 1900 plots carried out from 1987-1994 (NFSI I) and repeated between 2004 and 2008 (Fig. 1). With the NFSI II 1341 plots were resampled (= paired sample) whereas app. 600 were replaced by new established plots (= unpaired sample). A third inventory is planned for 2022. Within the NFSI, all parameters were investigated on a 30 m diameter circular plot with a soil profile in the centre. During the NFSI II, vegetation was also assessed on 1838 plots. In addition, in a forest inventory collected forest mensuration data in 2011 / 2012 on all NFSI II plots. In the forest inventory, tree and stand parameters, forest regeneration and amounts of deadwood were recorded.

2.1.2. Carbon stocks of organic layer and mineral soil

The NFSI I was carried out according to a common sample plot protocol. Based on this, adjustments and extensions in new findings and standards were implemented in a new common sample plot protocol applied for the NFSI II (Wellbrock et al., 2016). In both inventories eight satellites were sampled located 10 m from the plot centre in cardinal and inter-cardinal directions on a 30 m diameter plot. Central soil profiles were used to designate soil horizons and to classify soil types according to the German manual for soil sampling (Ad-hoc-Arbeitsgruppe Boden, 2005). The organic layer incl. branches and cones was collected with metal frames and combined to a mixed sample from satellites and subsequently partitioned into a fine and a coarse fraction set at a diameter of >20 mm.

The mineral soil was sampled at fixed depth increments of 0-5, 5-10, 10-30, 30-60 and 60-90 cm. During the NFSI I a large amount of plots was sampled only to a mineral soil depth of 30 cm (~25 %) or 60 cm (~10 %). Fixed volume samples were taken at the satellites and mixed within depths increments, which allowed the estimation of fine earth stocks and bulk densities based on mass and weight of fine and coarse soil fractions. Depending on stone content, volume based mineral soil samples were taken with cylindrical core or cap cutter, with an AMS Core sampler or a motor driven auger. Fine earth stocks were derived from bulk density and coarse soil fractions that were estimated or measured with several different methods. If the proportion of the coarse soil fraction was <5 % bulk density of the fine earth was equated with bulk density of the bulk soil. If the proportion of the coarse soil fraction was >5 % fine earth was separated from the coarse soil by sieving. Subsequently, weight, specific density, and volume of the coarse soil fraction were determined. If the proportion of coarse soil fraction was >20 % an additional shovel or spade sample or a coarse soil volume estimation on the soil profile was required. If volume based soil sampling was not practical or feasible due to a high stoniness, onsite estimates of the bulk density were permitted. The stocks of the fine-earth fraction were calculated as a function of dry bulk density of the fine-earth fraction and the volume of the coarse-soil fraction, or the stocks of the coarse-soil fraction and the dry bulk density of the gross soil (Grüneberg et al. 2014).

Laboratory analyses for both inventories were provided by the forest research stations of the federal states and conducted according to a common analytical protocol (Wellbrock et al., 2016). Organic C for NFSI I was determined by elemental analysis with elemental analysers (79 % of the samples), by dry combustion with subsequent conductometric CO_2 determination according to Wösthoff; by indirect estimation via loss of ignition at 550 °C and factor correction for mineral soil and organic layers; by wet combustion with potassium dichromate and sulphuric acid with subsequent photometric chromium (III) determination. The methods applied are verified as comparable to NFSI II where exclusively elemental analysers were used (Grüneberg et al., 2014). Carbonate determination was carried out gas-volumetrically according to Scheibler on samples of both inventories. Comparability between the inventories was tested by several preliminary studies and comparability among various laboratories was by ring analysis (Wellbrock et al., 2016). Soil carbon stocks were calculated by multiplying fine-earth stocks with carbon concentrations.

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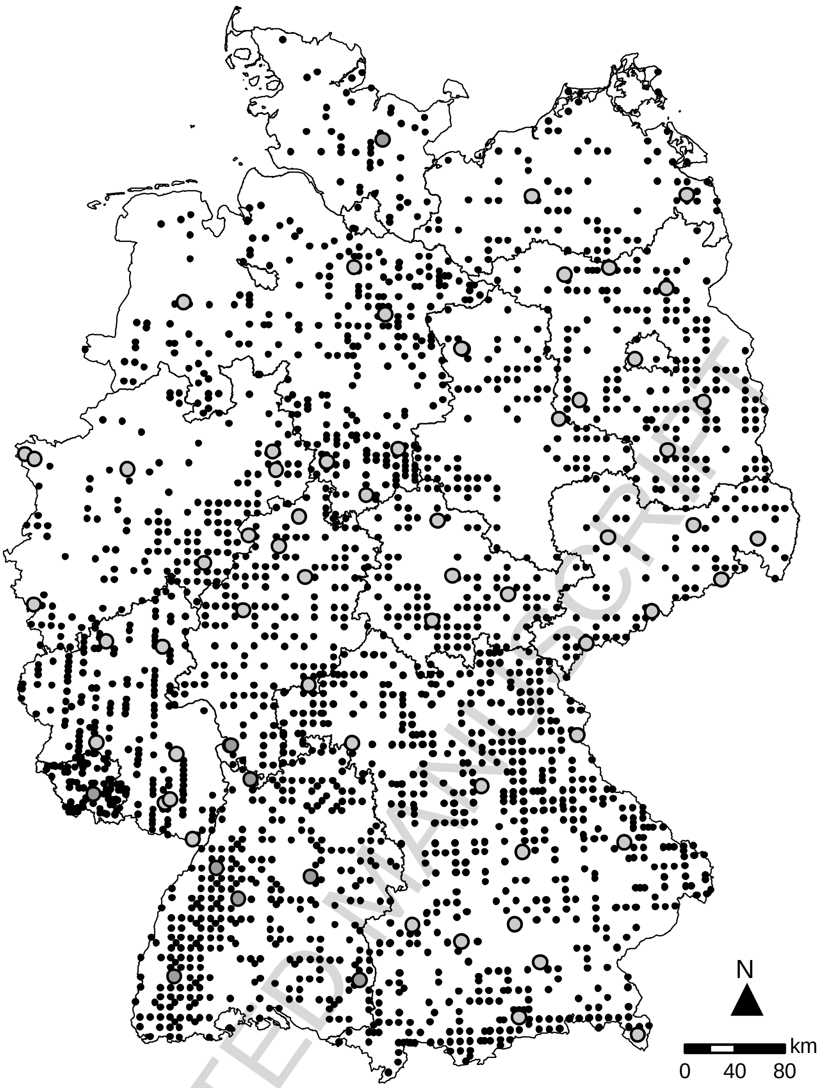

Fig. 1) Spatial distribution of the second National Forest Soil Inventory plots (NFSI II, black dots) and the intensive forest monitoring sites of the ICP-Forests network Level II (grey circles).

2.1.2. Carbon stocks of biomass

The forest inventory in 2011/2012 provided information on tree species, tree diameters, heights and crown length for individual trees on the plots. All trees with a diameter at breast height (dbh) >30 cm were measured on a circular area of 0.1 ha; trees with dbh of 10-30 cm were measured on an area of 0.05 ha and trees with dbh of 7-10 cm were measured on an area of 0.01 ha. For the German State Bavaria, data from the third National Forest Inventory (NFI) were used (Thünen-Institut, 2014), because the NFSI grid was a sub-grid of the NFI. For individual trees, the biomass of tree compartments was estimated using allometrical functions: For foliage, we used the functions of Perruchoud et al. (1999, oak), Wirth et al. (2004, spruce), Wutzler et al. (2008, beech), Pretzsch et al. (2014, pine and Douglas fir) and Rubatscher et al. (2006, larch). Woody above ground compartments were calculated using revised functions based on Pretzsch et al. (2014), which provided functions for coarse-wood and non-coarse wood compartments for spruce, pine, Douglas fir, beech and oak. Coarse roots were calculated using functions from Bolte et al. (2003, spruce, beech, oak), Johansson & Hjelm (2012, poplar) and Neubauer et al. (2015, pine). Tree species for which no functions were available were attributed to another species from the same genus or family or to beech-functions in case of hardwoods or spruce-functions in case of softwoods.

Trees with dbh less than 7 cm were assessed in the Forest Inventory by counting and classifying them into height and dbh classes on 4 circular plots with a radius of 5 m. Their above-ground biomass was estimated using biomass equations from Annighöfer et al. (2016), and the biomass of leaves, stems and roots was calculated using size-dependent allometric relationships (Poorter et al., 2012). The biomass of stems with a height below 1 m was estimated using the vegetation survey (see below).

In the vegetation survey of the NFSI II, all plant species and their cover were recorded for the different vegetation layers on a 400 m² sample area. The biomass of ground vegetation was assessed using the PhytoCalc-model (Bolte, 1999, Bolte et al., 2002, 2009). The PhytoCalc-model estimates biomass per area based on plant cover and shoot height. For vegetation data from clearings, we used correction factors from Heinrichs et al. (2010). While plant cover was assessed within the vegetation survey, shoot height was not measured. Therefore, we derived species-specific plant heights as averages from the ranges listed in Jäger & Werner (2000). The minimum and maximum values were used as a quantity in the calculation of the uncertainties of the biomass. The same data were used to attribute the species to a growth class in the PhytoCalc model. Mosses were attributed to a PhytoCalc growth class according to the description in Düll & Düll-Wunder (2012) and Frahm & Frey (2004). The PhytoCalc-model was parametrized with data from species collected in lowlands and lower mountain ranges of Germany.

2.1.3. Carbon stocks of dead wood

To determine carbon stocks in dead wood, we used the amounts of dead wood assessed in the forest inventory in 2011/2012. The size and decay class of deadwood pieces were estimated according to the manual of the BioSoil-Programme. For Bavaria, data from the third national forest inventory were used (Thünen-Institut, 2014). Deadwood volume was calculated according to Meyer (1999), transformed to mass according to Fraver et al. (2002) and Müller-Using & Bartsch (2009) and converted to mass of carbon by multiplying with 0.5.

2.2. Modelling soil carbon stock changes

We used Yasso15 litter decomposition and soil carbon model (Järvenpää et al. 2018) in this study to model organic carbon in soils above the depth of 1 m in mineral soils. Yasso15 is the latest version of the model following Yasso (Liski et al., 2003, 2005) and Yasso07 (Tuomi et al., 2008, 2009, 2011a, 2011b).

Yasso15 is based on a larger dataset than any of the earlier versions. The new dataset contains 16596 records of non-woody or woody litter decomposition and 4169 soil carbon stock measurements from different parts of the world. The most important new data is the global soil organic carbon dataset (Zinke et al., 1986) that has been linked to global datasets on gross primary production (Beer et al., 2010) and to the ratio between net and gross primary production (Zhang et al., 2009) to estimate soil carbon input at the sites of taking the soil carbon measurements. The new data combined with more efficient mathematical modelling enabled distinguishing differences in climate sensitivity between carbon fractions in Yasso15. In addition, they changed the decomposition patterns of woody litter to some extend compared to the earlier Yasso version somewhat (Stendahl et al., 2017).

Yasso15 model divides litter and soil carbon into five chemically different fractions. These fractions are a polar-solvent-soluble fraction (sugars etc., denoted with W), a non-polar-solvent-soluble fraction (waxes etc., denoted with E), an acid-hydrolysable fraction (celluloses etc., denoted with A) and a non-soluble-non-hydrolysable residue (lignin etc., denoted with N). In addition, there is a still more recalcitrant fraction (humus etc., denoted with H). All fractions have their own decomposition rate that is independent of the origin of the litter or soil carbon but dependent on temperature and precipitation. The decomposition rates of woody litter in the first four fractions depend additionally on the size of woody litter.

Litter entering soil is divided into the four first fractions according to its chemical quality. The decomposition products of the fractions are transferred to the other fractions (microbial catabolism or anabolism) or out of the system (respiration). A small fraction of the decomposition of products of these four fractions is transferred to the fifth, more recalcitrant fraction.

All parameter values of Yasso15 and their uncertainties were quantified simultaneously using all data, Bayesian methods and Markov chain Monte Carlo sampling. The results of the model are accompanied with uncertainty distributions representing uncertainty about the parameter values.

Input data of Yasso15 consists of information on climate (temperature and precipitation) and litter input to soil (quantity, chemical composition in terms of the above four fractions, and diameter in case of woody litter).

2.2.1. Climate data. We obtained climate information for the years 1961-2014 for each NFSI-plot by geostatistical modelling based on climate data provided by the German weather service (DWD). For temperature we used regression-kriging as described in Ziche & Seidling (2010) and for precipitation we used ordinary kriging. The DWD dataset consists of daily temperature measurements from 200 weather stations and precipitation measurements from 1200 stations from 01 January 1961 to 31 December 2014. To provide data for the Forest Management Reference Level on the future development of soil carbon stocks for the period from 2015 to 2030, we used the central STARS Projection (Gerstengarbe et al., 2015), which was also available for the weather stations of the DWD as daily time series and was attributed to the NFSI-plots using the same methods as described above. The STARS model is a statistical forecast based on temperature conditioned resampling of measured time series in order to project changes of climate variables for a given temperature increase. The temperature increase is derived from the general circulation model ECHAM6 run for the RCP 8.5 scenario, which represents a high-emission business as usual scenario and thus the most extreme climate change scenario. We choose that scenario because the Forest Reference Management Level requires a business as usual scenario.

2.2.2. Litter input from living trees. As we had data only from one forest inventory, we assumed a constant, annual litter input from living trees for the simulation period. To estimate annual litter production, the biomasses of the distinct plant compartments were multiplied with their turn-over rates. We used turn-over rates assembled in Peltoniemi et al. (2004), Thürig et al. (2005) and Wutzler & Mund (2007) for different tree species and compartments (Supplement 1). The fine root litter input was calculated according to data assembled by Wutzler & Mund (2007). For ground vegetation, the turnover rates for different plant groups in Peltoniemi et al. (2004) were used.

2.2.3. Litter input from natural mortality and forestry operations. We estimated carbon input from harvest and thinning operations in three steps:

i) First, at plot specific-level each strata of a stand (species, age class) was attributed to digital yield tables (Nagel & von Gadow, 2000) by species, and its actual age and mean height or top height. For the respective age and yield class, the ratio of removed wood volume to standing stock volume was calculated using the yield table data. Biomasses of the different woody compartments were then multiplied by that ratio, which resulted in the harvested biomass. It was assumed that wood accruing from natural mortality was included within the yield table data for thinnings. The net increase of standing stock biomass was estimated in the same way and the ratio of harvested biomass to net increase was calculated as a plot specific harvest level which served as baseline ratio.

ii) Then the interannual variability of the plot-specific harvest level was estimated according to Germany-wide averages (Röhling et al., 2015) to obtain annual values of harvested biomass. Based on harvest statistics and NFI-results Röhling et al. (2015) estimated German-wide average annual values of harvested carbon and net-growth between 1990 and 2013. In relation to that ratio, we calculated the annual harvest level on each individual plot relative to its baseline ratio.

iii) In the last step the share of harvest residues of the total harvested biomass was estimated to be 15 % using the results of the National Forest Inventory (Thünen-Institut, 2014) in combination with forestry products statistics (Jochem et al., 2015). According to Jochem et al. (2015), changes in the share of non-used coarse wood residuals were not significant during the investigated period 1995-2013, while the share of harvested non-coarse wood increased. Based on their data, we estimated a decrease in the share of residual non-coarse wood from 86% in 1990 to 65% in 2012. Finally, the share was multiplied with the harvested biomass. The share in 1990 served as baseline for model initialisation.

For the future projection, we used plot-wise the average input of the years 2009-2013.

2.2.4. Litter size and quality. For the model input, we divided the litter into 3 fractions: a fine fraction (leaf/needle litter and fineroot litter), a non-coarse wood fraction (wood < 7 cm diameter) and a coarse wood fraction (>= 7 cm diameter). The values for chemical quality of different plant species, genera or groups and plant compartments were adopted from the Yasso manual (Liski et al., 2009) and complemented with values from Strakova et al. (2010) and Liski et al. (2013). Missing values for tree species were filled using means for plant genera or softwood and hardwoods. Means for herbs, ferns, shrubs and mosses were applied for ground vegetation.

2.2.5. Initialization. For each plot plot-specific carbon stock change rates and soil carbon stocks from the NFSI (Grüneberg et al., 2014) were used to calculate carbon stocks back to 1985, the year before the first NFSI-plot was sampled. This value was set as total carbon stock at the beginning of the modelling period. To obtain values for the five carbon fractions for each NFSI plot, the model was run plot-wise for 10,000 years with constant climate (the averages of the time-period 1961 – 1990) and constant plot-specific litter input per litter fraction. For the initialization of the model, we used the baseline harvest level and the baseline share of harvest residuals. The carbon fractions from the 10,000 year time-series were then selected at that point in time when their sum matched the carbon stocks in 1985. In the event that the sum after 10,000 years was still smaller than the measured carbon stocks, the excess carbon was allocated to the most recalcitrant carbon pool.

2.2.6. Uncertainties. The uncertainties in carbon input were calculated using error propagation with the uncertainty data of the used biomass functions and turn-over rates. For ground vegetation, the information of minimum and maximum growth height of each species was included in the calculation. To obtain estimate the uncertainty for the share of harvested wood, we simulated the age and height of each stratum 2000 times according to their uncertainties and took the respective values for harvested wood from the yield tables. Data on uncertainties were not available for some of the foliage biomass functions. Therefore, we used data from the intensive forest monitoring program (ICP forests level II) to estimate the uncertainties of foliage biomass. This database contains litterfall data of 57 monitoring sites distributed throughout Germany (Fig.1). Furthermore, forest mensuration data for these plots were also available in the database. We used the forest mensuration data to estimate litter fall as described above. Details of methods used in the intensive forest monitoring program to measure litterfall and to sample forest inventory data are described in Ukonmaanaho et al. (2016) and Dobbertin & Neumann (2016). We compared the estimates with the measured values using regression analyses. The average mass of foliage litter amounted to 2.78 ± 0.13 Mg ha^{-1} a^{-1} for estimated litter and 2.95 ± 0.10 Mg ha^{-1} a^{-1} for measured litter input, with r^2=0.9 and CV=30% (based on RMSE). We used this CV as the quantity for uncertainties of foliage litter input. Data on the uncertainties of turn-over rates were acquired from Wutzler & Mund (2007).

Apart from the amounts of litter input, the uncertainties of the model results depend on the uncertainties of the litter quality, initial values and model parameters. We estimated the uncertainties of AWEN fractions for each litter component by setting their coefficient of variation to 10%.

To estimate the uncertainty in the modelling results we simulated 100 input datasets per litter fraction, year and NFSI plot. The same number of Yasso15 parameter sets was selected. Finally, to derive the uncertainty of the model output, we ran the model 100 times per litter fraction, year and NFSI-plot. The initialization was done in the same way.

2.3. Data evaluation. On plot-level the simulated annual carbon balance was calculated from the Yasso15 outcome as the difference of carbon stocks of the current year and the previous year. Averages were calculated for the periods 1990-2008 (n=1756). Carbon stock changes of the paired sample of the inventory were estimated plot-wise by the difference of carbon stocks divided by the elapsed years between inventories. Soil carbon stock changes of the unpaired sample were derived by regionally averaged carbon stocks and elapsed time between the inventories.

The soil carbon stocks and their changes were stratified according to Grüneberg et al. (2014). Stratification was based on a German soil map (BGR, 1998) with mapping units summarized to 16 groups. In a first step site-specific soil information was attributed to the German soil map. Then, regional averages classified in soil units and German States were calculated. These values were pooled to an area weighted mean of Germany.

For calculating stratified NFSI averages we used two approaches: To calculate a German-wide average for purpose of comparing the German-wide mean carbon balance we used the data of organic layer and mineral soil down to 30 cm depth, because during the NFSI I only data down to 30 cm was completely available for whole Germany. These values originate from Grüneberg et al. (2014). For calculating the regional averages to correlate both methods we used a subsample with data of organic layer and mineral soil down to 90 cm depth. Both approaches include the paired and unpaired samples.

Furthermore, as Yasso15 is mainly driven by litter input, we classified the sample into different forest types (Table 1) and compared the inventory results of organic layer and mineral soil down to 30 cm depth with the Yasso15 estimates. Classification was achieved using the vegetation survey (Ziche et al., 2016). The first group of forest types is based on phytosociological syntaxa and their tree species composition is near-nature like or they are a result of pre-industrial forest use. These classifications are hardwood floodplain forests, beech forests on intermediate or moist base-rich soils, acidophilous beech forests, thermophilous oak-hornbeam forests, moist oak-hornbeam forests, acidophilous oak and moist acidophilous oak forests. The classification into forest types contains also the mountainous forest types mesotrophic beech/fir forests and acidophilous spruce/fir forests. Therefore it captures the main climatic gradient in Germany, which is in conjunction with the elevational gradient. The sites of both forest types are characterised by distinct lower temperature and higher precipitation compared to other forest types (Table 1). The second group consists of forest types in which tree layer composition deviates from near-natural composition. They are classified according to the composition of the ground vegetation into eu-, meso- and oligotrophic. These classifications include pure Scots pine and Norway spruce stands, deciduous forests, mixed deciduous/coniferous and coniferous forests others than pure Scots pine and Norway spruce stands. The group of oligotrophic Scots pine forests also contain near-natural Scots pine forests. Furthermore, clearings of various plant species compositions are segregated. Clearings are sites in which the tree layer had been removed by harvest or natural disturbances. In the case that different forest types occurred within a plot, only one unit was assigned. As vegetation data were available only for the NFSI II, only plots which belong to the paired sample could be recognized in this evaluation (1274 plots). Only forest types with n >= 9 were recognized in the evaluation. Documentation of the range of climate, stand and soil properties for the forest types can be found in Table 1 and in Ziche et al. (2016).

Tab.1) Forest types with climate, soil and stand characteristics, with n = number of plots in paired sample, T = temperature, Pr = precipitation, Age_{Bamax} = Age of tree layer with maximum basal area, C_{conc} = organic carbon concentration, pH_{CaCl2} = pH measured in suspension with 0.01 m $CaCl_2$, BS = base saturation, BD = bulk density, CF = coarse fragment content, SOM= organic carbon of mineral soil (0-90cm) + organic layer + deadwood, C-Input = estimated annual carbon input (baseline). The soil properties C_{conc}, pH_{CaCl2}, BS, BD, CF are for mineral soil 0 – 30 cm. In () = standard deviation.

	n	T [°C]	Pr [mm]	Age (Ba_{max}) [a]	C_{conc} [%]	$pH_{(CaCl2)}$	BS [%]	BD [g/ccm²]	CF [%]	SOC [MgC ha^{-1}]	C-Input [MgC ha^{-1} a^{-1}]
Hardwood floodplain	11	9.1 (0.9)	767 (111)	62 (32)	2.5 (1.2)	6.1 (1.6)	84 (27)	1.16 (0.24)	2.7 (3.4)	124.1 (39.2)	5.52 (1.62)
Beech on intermediate or moist base-rich soils	155	8.2 (0.7)	813 (176)	108 (44)	2.4 (1.5)	4.7 (1.2)	55 (36)	1.13 (0.34)	13 (16.2)	99.8 (38.3)	4.43 (1.22)
Acidophilous beech	189	8.1 (1.0)	892 (239)	106 (43)	2.5 (1.6)	3.9 (0.3)	15 (12)	1.11 (0.24)	16.7 (17.4)	104.4 (42.2)	3.91 (1.17)
Thermophilous oak-hornbeam	33	8.7 (0.7)	776 (114)	103 (42)	2.7 (1.2)	4.9 (1.2)	61 (37)	1.06 (0.25)	17.4 (20.6)	92.7 (39.8)	4.92 (1.43)
Oak-hornbeam (moist)	21	8.9 (0.5)	748 (151)	90 (39)	2.3 (1.0)	4.5 (1.1)	58 (32)	1.43 (0.70)	3.3 (5.1)	163.2 (159)	5.07 (1.45)
Acidophilous oak	31	8.5 (0.5)	765 (172)	85 (41)	2.4 (1.3)	3.8 (0.2)	12 (8)	1.18 (0.25)	18.5 (17.3)	90.8 (29.3)	3.99 (0.94)
Acidophilous oak (moist)	12	9.0 (0.7)	765 (157)	87 (33)	2.2 (0.8)	3.7 (0.3)	14 (9)	1.27 (0.14)	4.5 (8.5)	137.5 (46.4)	4.59 (1.04)
Eutrophic deciduous	27	8.7 (0.8)	733 (148)	66 (39)	2.7 (1.5)	5.5 (1.3)	81 (30)	1.15 (0.23)	4.9 (6.6)	136.7 (72.0)	4.89 (1.74)
Mesotrophic deciduous	23	8.6 (0.9)	776 (156)	58 (34)	2.1 (1.0)	4.2 (1.1)	32 (31)	1.12 (0.18)	8.3 (12.2)	107.1 (40.7)	4.84 (1.24)
Mesotrophic pine	55	8.6 (0.4)	625 (98)	65 (24)	1.3 (0.8)	3.9 (0.6)	21 (21)	1.40 (0.11)	3.4 (5.2)	100.8 (37.6)	3.59 (0.71)
Oligotrophic pine	144	8.6 (0.4)	600 (63)	64 (29)	1.3 (0.7)	3.8 (0.2)	12 (10)	1.41 (0.1)	2.8 (6.3)	105.7 (43.1)	3.34 (0.70)
Eutrophic spruce	14	7.6 (0.9)	862 (119)	67 (31)	3.5 (1.7)	5.6 (1.5)	81 (28)	0.97 (0.22)	21.4 (21.0)	112.1 (32.0)	4.11 (2.17)
Mesotrophic spruce	102	7.6 (0.7)	928 (199)	70 (29)	2.8 (1.3)	3.9 (0.4)	21 (22)	1.05 (0.22)	19 (16.9)	117.4 (40.4)	4.23 (1.97)
Oligotrophic spruce	78	7.7 (0.9)	893 (168)	73 (33)	2.7 (1.3)	3.7 (0.2)	9 (7)	1.11 (0.25)	19.3 (17.8)	123.7 (43.3)	3.74 (1.58)
Montaneous mesotrophic beech/fir	9	6.7 (0.9)	1183 (299)	100 (39)	6.2 (3.1)	5.6 (1.5)	66 (41)	0.71 (0.17)	30.2 (17.4)	142.3 (53.8)	3.94 (1.00)
Montaneous acidophilous spruce/fir	47	6.1 (0.8)	1291 (279)	95 (39)	3.7 (1.7)	3.7 (0.2)	8 (8)	0.76 (0.23)	26.6 (17.4)	124.5 (43.2)	4.59 (1.53)
Eutrohic mixed deciduous/coniferous	19	8.0 (1.1)	831 (192)	92 (40)	3.0 (1.6)	5.6 (1.6)	75 (36)	1.08 (0.25)	16.5 (22.6)	111.1 (44.1)	4.61 (1.74)
Mesotrohic mixed deciduous/coniferous	66	8.2 (0.9)	845 (228)	79 (38)	2.2 (1.3)	4.3 (1)	37 (33)	1.14 (0.22)	10.3 (13.3)	102.5 (36.7)	4.20 (1.31)
Oligotrophic mixed deciduous/coniferous	39	8.3 (0.7)	825 (255)	79 (37)	1.8 (0.8)	3.8 (0.3)	12 (10)	1.22 (0.20)	9.8 (13.4)	110.4 (41.7)	3.97 (1.20)
Mesotrophic coniferous	32	8.2 (0.8)	843 (224)	66 (35)	2.0 (1.1)	3.8 (0.3)	20 (17)	1.18 (0.30)	14.4 (19.5)	101.7 (37.7)	4.09 (1.55)
Oligotrophic coniferous	55	8.1 (0.7)	911 (234)	75 (35)	2.5 (1.5)	3.7 (0.4)	11 (11)	1.16 (0.28)	9.6 (14.3)	141.0 (66.6)	3.97 (1.32)
Clearings	13	7.8 (0.7)	881 (201)	-	3.3 (2.1)	4.5 (1.1)	41 (39)	1.03 (0.37)	27.3 (21.6)	108.2 (31.9)	2.21 (1.02)

3. Results

Our simulation indicates that forest soils in Germany accumulate carbon. Overall, the simulated carbon stocks increased from 113 Mg C ha^{-1} in the year 1990 to 119 Mg C ha^{-1} in 2030 (Fig. 2a). The simulated carbon balance between 1990–2030 exhibits a relatively large variance (Fig. 2b). Most striking is a peak in 1990, which results from a high litter input. The average litter input amounts to 5.30 ± 0.05 Mg C ha^{-1} a^{-1} in that year (fig. 2c). This was due to a series of heavy storms and resulting wind throws. In the subsequent years, the carbon balance declined due to the decomposition of the 1990 deadwood peak. In 1994, forest soils became a carbon source. Within the period of the NFSI's this is the only year with significant average carbon losses. In 1996 carbon stocks increased again. In both years, the changes in carbon accumulation were due to climatic conditions as the carbon input varies only slightly (Fig. 2d). The year 1994 was with 9.6 °C and 924 mm comparably warm and rich in precipitation, which increased the decomposition of organic material; in contrast, the year 1996 was with 7.0 °C on average the coolest year in the simulation period (Fig. 2d). All years within the period 2003-2008 exhibit carbon accumulation, which coincides with high levels of carbon input during that period. Between 2002 and 2008, litter input increased due to increased harvest and a consequent increase in harvest residues. The highest litter input within this period was in the year 2007, with 5.04 ± 0.05 Mg C ha^{-1} a^{-1} on average. Additionally, the year 2003 was the driest year, with 633 mm precipitation on average (Fig. 2d), which led to reduced decomposition rates.

At the beginning of the simulation period, the estimated average litter input was 4.15 ± 0.04 Mg C ha^{-1} a^{-1}. The litter input consisted of 1.47 Mg C ha^{-1} a^{-1} (35.5 %) litter from the above ground compartments (Fig. 2c) and 1.61 Mg C ha^{-1} a^{-1} (38.9 %) from below ground compartments. The input from natural mortality and forestry operations was 0.49 Mg C ha^{-1} a^{-1} (11.7 %), and from the understory vegetation (trees < 7cm dbh, shrub, herb and moss layer) 0.58 Mg C ha^{-1} a^{-1} (14.9%).

When investigating the development of the carbon fractions, it becomes apparent that the most recalcitrant fraction did not change over time and contained 61 Mg C ha^{-1} (Fig. 2a). At the beginning of the modelling period, the most recalcitrant fraction (H) had a share of 55% of the total soil carbon pool while at the end of the modelling period it dropped to 51%. In contrast, the non-soluble-non-hydrolysable residue fraction (lignin etc., N) increased from

31% or 35 Mg C ha^{-1} to 34% or 40 Mg C ha^{-1}. The other carbon fractions (A, W, E) increased proportionately with total carbon stocks.

From 2014 onwards, the model runs with constant carbon inputs (± uncertainties). Consequently, the estimated changes in soil carbon balance are only driven by climatic variation and legacy effects. Between 2021 and 2030, 2 years with carbon accumulation and losses are predicted while the other years show no significant deviation from 0. The area weighted averages of carbon stock changes of the period 1990-2008, the period of the NFSI, amounted to 0.25 ± 0.10 Mg C ha^{-1} a^{-1}. At 0.26 ± 0.10 Mg C ha^{-1} a^{-1}, accumulation rates in the period 2013-2020 are in the same order of magnitude. For the period 2021-2025, the predicted rate dropped to -0.10 ± 0.10 Mg C ha^{-1} a^{-1} and in the period 2026-2030 the prediction amounted to 0.13 ± 0.10 Mg C ha^{-1} a^{-1}.

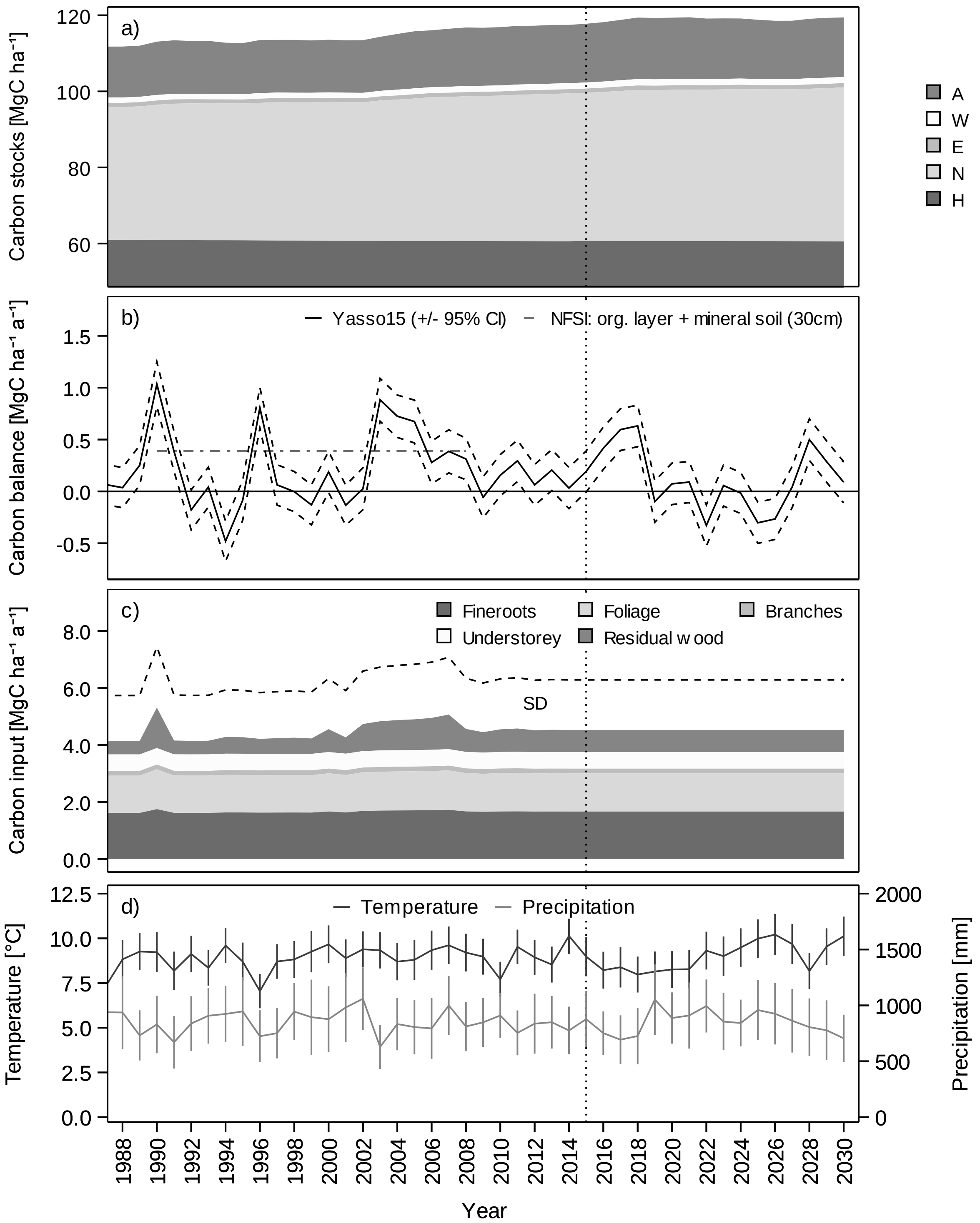

Fig. 2 a) Development of carbon stocks within the different carbon fractions (AWENH) of Yasso15 (n=1756), with A = acid-hydrolysable fraction (celluloses etc.), W = polar-solvent-soluble fraction (sugars etc.), E = non-polar-solvent-soluble fraction (waxes etc.), N = non-soluble-non-hydrolysable residue (lignin etc.) and H = recalcitrant fraction (humus etc.). **b)**

Time course of the simulated carbon balance of organic layer and total soil between 1986 and 2030 (n=1756). For comparison, the results of the NFSI for organic layer and mineral soil down to 30 cm soil depth (Grüneberg et al. 2014) were added. **c)** Average annual litter input into the soil (+SD, n=1756). Prior to 1990, the values were set as constant on the level of model initialization; from 2014 onwards the values are on the level of the mean values from 2009-2013. **d)** Average annual temperature and precipitation (+SD, n=1756). The vertical dotted lines in c) & d) represent the beginning of the projection.

Both modelled and measured changes in soil carbon stock indicated an increase in German soil carbon stocks in 1990-2008. The comparison of the modelled carbon balance with the measured value of the NFSI for organic layer and mineral soil down to a depth of 30 cm reveals a lower level of the modelled values compared to the inventory (Fig. 3). Nevertheless, at 0.25+/-0.10 Mg C ha^{-1} a^{-1} and 0.39+/-0.11 Mg C ha^{-1} a^{-1}, the values are within their confidence intervals. Consequently, the difference between these average values was not statistically significant.

The initial carbon stocks after 10,000 yr. spin-off runs amounted on average to 109.6 ± 1.1 Mg C ha^{-1} and are slightly lower on average as the measured carbon stocks of mineral soil down to soil depth 90 cm + organic layer + deadwood which amounted to 116.9 ± 1.3 Mg C ha^{-1} (Fig. 3). The measured carbon stocks of mineral soil down to soil depth 30 cm + organic layer + deadwood amounted to 89.2 ± 1.0 Mg C ha^{-1}.

A more detailed comparison of the results revealed other differences between the methods. Comparing the individual soil carbon change rates of the NFSI-plots, a correlation revealed a significant but low correlation (r^2=0.15) between measured and modelled values. Aggregating the NFSI-plots regionally to soil units and German states decreased the variability and strengthened the correlation (r^2=0.49, Fig. 4a). The most distinct differences between the methods are a larger variability and a higher number of negative values in the inventory results. Only three simulated regional averages are below 0 (Fig. 4a). The model-calculated and the measured soil carbon stocks are strongly correlated with each other in 2008 (Fig. 4b). The model-calculated values were adjusted to match the measured ones in the beginning of the simulations in 1986. Thus, the high correlation between the values 22 years later indicates, on the one hand, consistency of the methods and, on the other hand, the fact that the changes in the simulated values were relatively small compared to the initial soil carbon values.

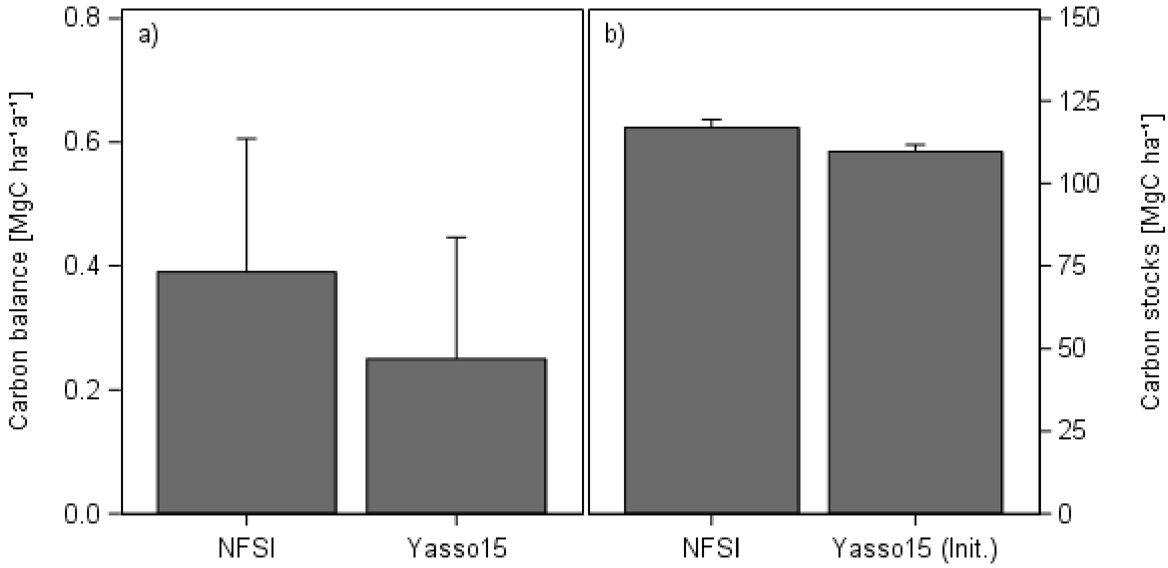

Fig. 3 a) Germany-wide average (± 95 % CI) of modelled soil carbon balance (Yasso15) and measured carbon stock changes (organic layer + mineral soil down to a depth of 30 cm) between 1990 and 2008. The plot-values were pooled to an area weighted mean according to Grüneberg et al. (2014). The results of Yasso15 include beside the sample variance also the model uncertainties.

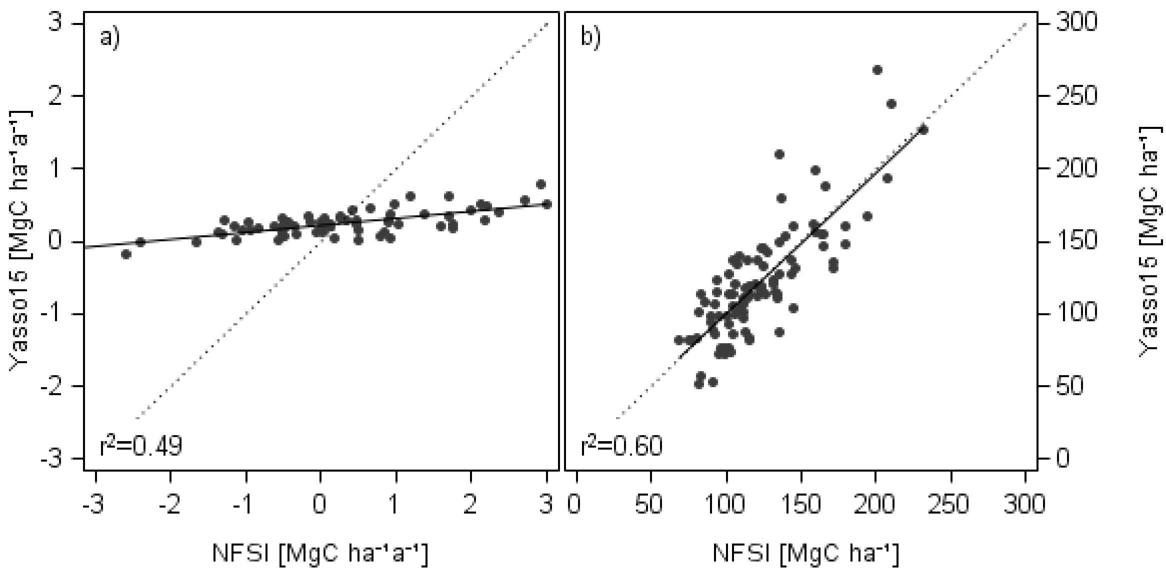

Fig. 4) Comparison of measured (NFSI) and modelled (Yasso15) soil carbon balance and soil carbon stocks. The values of the NFSI-plots were pooled regionally according to soil unit and German state, with a) comparison of SOC changes and b) soil carbon stocks in 2008 (NFSI II). The measured values in a) are carbon balance of organic layer and mineral soil down to 90 cm soil depth and contain only the subsample, where the estimation of carbon balance of organic layer and mineral soil down to 90 cm soil depths was possible (n=947).

The classification of the modelled and measured values into forest types demonstrates that the sample size needs to be relatively large to detect changes (Fig. 5). In the inventory, significant positive change is apparent in oligotrophic coniferous and mixed deciduous/coniferous, meso- and oligotrophic pine and spruce forests and eutrophic deciduous forests. In oligotrophic forests, the measured changes were high and reached the highest values in pine forests, at 1.36 ± 0.13 Mg C ha^{-1} a^{-1}. The soils of the oligotrophic forests are characterised by low values of group means of pH (< 3.9) and base saturation (< 15 %, Table 1). At 0.12 ± 0.15 Mg C ha^{-1} a^{-1} and 0.11 ± 0.15 Mg C ha^{-1} a^{-1}, the large groups of beech forests show no significant changes in the inventory. High carbon losses are detected in mountainous acidophilous spruce and fir forests with -0.89 ± 0.30 Mg C ha^{-1} a^{-1}. Also, hardwood floodplain forests, at -0.43 ± 0.44 Mg

C ha^{-1} a^{-1} on average, have negative values, which due to the small sample size are not significant.

The Yasso15 estimates indicated in most groups a significant positive change. However, the increase in the soil carbon stock was smaller than the carbon stock changes in the inventory. Exceptions included the beech forests, where the carbon accumulation rate at 0.34 ± 0.11 Mg C ha^{-1} a^{-1} and 0.25 ± 0.10 Mg C ha^{-1} a^{-1} is notably higher compared to the inventory results. Within groups of forests with comparable tree layer, Yasso15 averages increase with increasing nutrient availability, for example within spruce forests from 0.13 ± 0.10 Mg C ha^{-1} a^{-1} in oligotrophic to 0.22 ± 0.10 Mg C ha^{-1} a^{-1} in mesotrophic and 0.43 ± 0.23 Mg C ha^{-1} a^{-1} in eutrophic forests. This trend was also observed within groups of beech forests and when comparing acidophilous oak forest and oak-hornbeam forests. Differences in carbon balance between groups of forest according to the nutrient availability were accompanied by differences in carbon input (Table 1), e.g. the litter input in spruce forests from mesotrophic sites (4.23 ± 0.20 Mg C ha^{-1} a^{-1}) is higher compared to oligotrophic sites (3.74 ± 0.18 Mg C ha^{-1} a^{-1}).

Significant differences between Yasso15 and NFSI occurred in the groups mountainous acidophilous spruce and fir, oligotrophic pine, oligotrophic coniferous, oligotrophic mixed deciduous / coniferous and mesotrophic mixed deciduous / coniferous forests. These groups represent 25 % of the plots of the examined forest types.

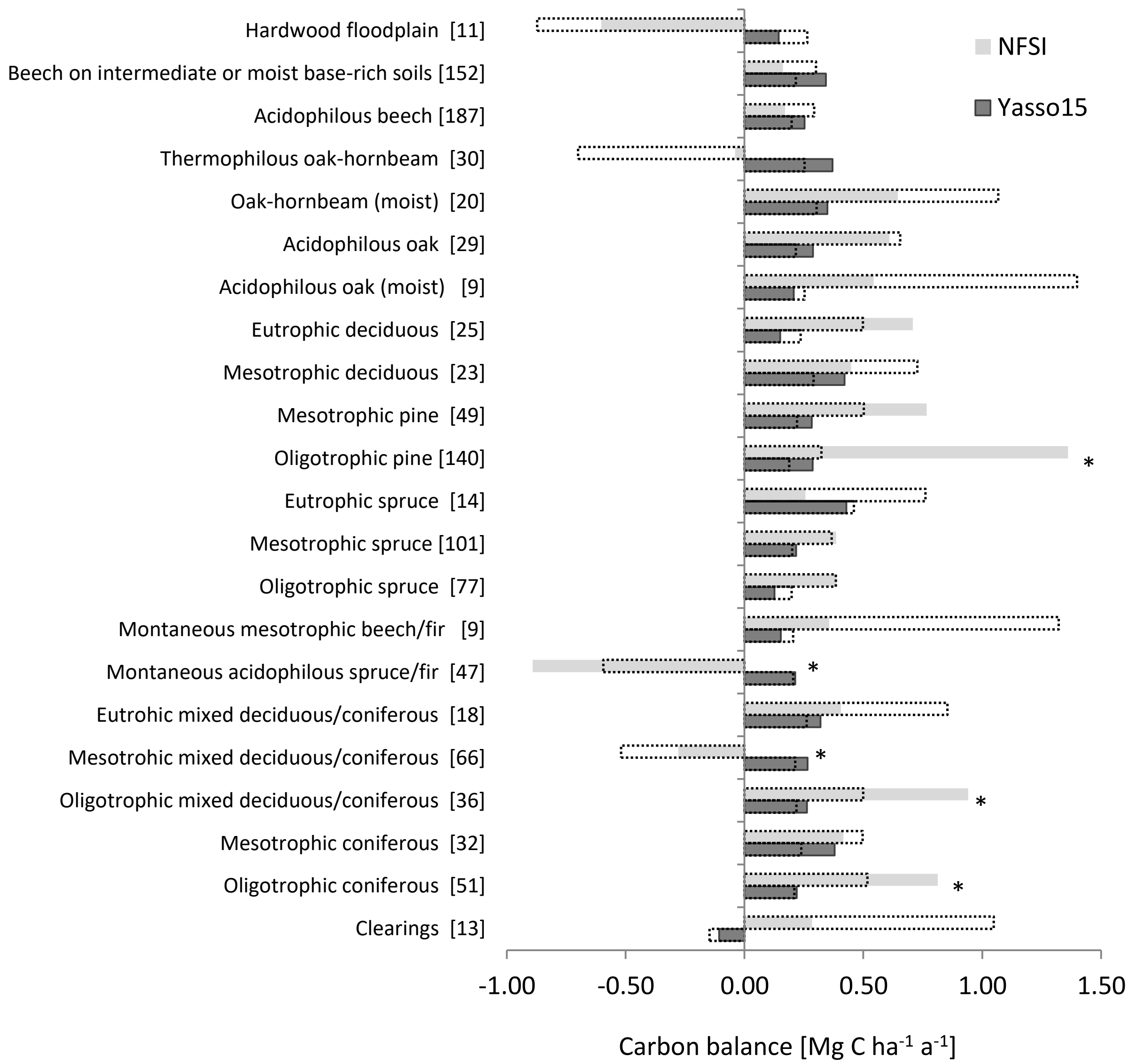

Fig. 5) Comparison of the results of the Yasso15 model with the NFSI results for average carbon stock changes of organic layer and mineral soil down to 30 cm soil depths in the forest types during the period 1990-2008. The unfilled bars represent the 95%-confidence limit: when the filled bars exceed them, the respective change rate deviates significantly from 0, with values in [..] = number of observations, * = indicates if Yasso15 and NFSI- results deviate significantly from each other.

4. Discussion

The simulation as well as the repeated soil inventory indicate that under current conditions, Central European forest soils accumulate considerable amounts of carbon. The results of our modelling approach indicate a high accumulation rate equal to 0.25+/-0.10 Mg C ha^{-1} a^{-1}. This rate is somewhat lower, although not statistically different, from the 0.39+/-0.11 Mg C ha^{-1} a^{-1} found in the German NFSI and reported in Grüneberg et al. (2014). Carbon accumulation in forest soils was also found in other European countries, estimated with repeated inventories (Jonard et al., 2017), modelling with Yasso07 (Ortiz et al., 2013, Dalsgaard et al., 2016), other models (e.g. de Vries et al., 2017, Tipping et al., 2017) or both Yasso07 and soil inventories (Rantakari et al., 2012, Lehtonen et al., 2016).

The modelled carbon stock changes of this study are high compared to results from other European countries where Yasso07 was used. For Finland and Norway, the reported change rates are below 0.15 Mg C ha^{-1} a^{-1} (Rantakari et al., 2012, Dalsgaard et al., 2016). The differences compared to Germany can be explained in part by the lower NPP and lower carbon input to the soil along the latitudinal gradient. On the other hand, case studies from other European countries also obtained lower rates with Yasso07 (Hernandez et al., 2017). The comparable high carbon accumulation during the period of the NFSI's can also be explained by an increase of carbon input during the modelling period. This was due to the storm events in 1990 and the increasing amounts of litter input from harvest residues between 2002 and 2008. The difference in simulated carbon accumulation between this and the earlier studies may also be attributed to the methodology. Our approach differs from other modelling studies: the initial carbon stocks were adjusted to measured carbon stocks. Therefore, the carbon accumulation is also the result of initial conditions which coincide with measured carbon stocks but which were not in a steady-state. Our process led to closer correlation between modelled and measured carbon stock changes than is usually observed. However, it has to be taken into account that adjusting the initial values to measured carbon drives on long-term the stock development towards steady-state stocks.

In our future scenario the litter input was held constant, which is not a realistic assumption for long-term projections. In reality many factors will influence harvest levels and forest development leading to variations in the annual litter input. We choose that approach because the context of Greenhous Gas Reporting the Forest Reference Management Level requires a business as usual scenario. The results of the projection indicate that for the period 2012-2020 compared to NFI's period no changes in soil carbon accumulation can be assumed as reference for greenhouse gas accounting. An interesting feature of our simulations is that in this time perspective C accumulation takes place mostly in the labile pools and that this accumulated carbon thus exhibits a higher sensitivity to decomposition. These findings are in line with a model study by Tipping et al. (2017), which revealed that much of the modelled SOC increase takes place in the rapidly decomposing pool in their model.

The classification of the modelled and measured values by forest types gives new insights about patterns of carbon accumulation in Central European forests. For many classes, comparison between Yasso15 and the inventory among forest types revealed the same trend as the Germany-wide average. In most cases, the inventory measured on average higher carbon stock accumulation than was modelled by Yasso15. In some classes, representing 25 % of the plots, the outcomes of the two methods differed significantly. Our study revealed that mainly oligotrophic forests showed the highest carbon accumulation in the inventory with highest changes in oligotrophic pine forests. This may be attributed to land use history. Many pine forests are reforestations or were exposed in the past to management practices such as clearcutting and litter raking. Consequently, their soils are depleted in carbon, as it was demonstrated for Suisse forest soils (Gimmi et al., 2013). Now, with increasing stand age and changed management practices, the carbon stocks of these soils are recovering. One indication of this development is the disappearance of lichen pine forests in Germany (Fischer et al., 2015), which stocked on the nutrient-poorest mineral forest soils.

This trend is not only restricted to pine forests as a comparison of historical forest maps indicates that large parts of German forests are affected by land use conversion in the past (McGrath et al., 2015). Based on forest use practices of the past, such as litter raking and clearcutting, it can be assumed that forest soils are also on average depleted in carbon stocks and still recovering as has been modelled for conversion of arable land to grassland (Lugato et al., 2014). Our study revealed that mainly oligotrophic forests showed the highest carbon accumulation in the inventory. In contrast to oligotrophic coniferous forests the large groups of beech forests showed no significant changes. The enhanced soil carbon accumulation can be explained by an increased formation of acid compounds in pine compared to beech litter as pine litter contains more slowly decomposing components (Berg, 2000). Their soils are characterized by low pH-values and base saturation. Also soil fauna is less active in a more acidic environment, decreasing the amount of humus mixing through mineral soil and leaving more material in the forest floor (Thuille & Schulze, 2006).

Another striking difference between the Yasso15 and inventory results is the diverging trend in soils from mountainous acidophilous spruce/fir forests. Yasso15 showed an increase of carbon stocks while the inventory revealed carbon losses on these sites. Carbon losses were also reported for mountainous forest soils from the German alps and have been attributed to climate warming (Prietzel et al., 2016). These authors demonstrated that carbon losses are mainly restricted to calcareous soils, while acidic soils show no changes, which is in contrast to our NFSI result. The NFSI results indicate that acidic mountainous soils exhibit the highest carbon losses. Mountainous regions belong to the areas which were affected most heavily by acidifying sulphuric deposition, which declines during the last decades (EEA, 2017). This coincides with increasing DOC concentrations of water basins in many lower mountain ranges in Germany (Musolff et al., 2017). Increasing DOC concentrations in surface waters have been attributed to decreasing sulphuric deposition, due to interplay of its solubility with Al in soil solution (Monteith et al., 2007). Therefore, possible explanations for SOC losses in base-poor mountainous soils include beside climate change also the decline of sulphuric deposition.

Both simulations and measurements produced estimates of changes in SOC stocks with no statistical difference. The modelled SOC changes tend to be lower than the measured carbon stock changes, both on average across Germany and in most forest types. One potential explanation is that our estimate for litter input is lower than the estimates from two different forest models for Germany (Repo et al., 2015). However, in our study using ICP Forest data the leaf and needle litter input estimate was in line with the measured values. Furthermore, we added ground vegetation litter as suggested by Lehtonen et al. (2016) and demonstrated that it contributes with ~15% to the total litter input. Nevertheless, an underestimation of litter input may have resulted lower modelled initial carbon stock compared to measured values.

When comparing the Yasso15 results with the results from the NFSI, it should be borne in mind that Yasso15 models the aggregated changes in the deadwood, organic layer and soil carbon pool to 1 m depths, while the NFSI reports changes for whole Germany in the carbon pool of organic layer and mineral soil to 30 cm depths. This implies that differences between both methods could be caused by changes in the carbon pools of deadwood or mineral soil below 30 cm depths, whose changes could not be reported for whole Germany by the NFSI. Results from the German forest inventory revealed that deadwood pools increased by 0.06 Mg C ha^{-1} a^{-1} during 1990-2008 (Dunger et al., 2016). This suggests, that changes in the deadwood pool could not explain the differences between both methods.

Another possible explanation for the differences is carbon dynamics below 30 cm soil depth. Carbon balance below 30 cm soil depths is influenced by fine root dynamics, as root litter and exudates were the main resources contributing to deeper soil carbon input (Rasse et al., 2005). This could cause a bias when recognising only the measured carbon balance down to 30 cm soil depth. 20 to 50 % of total fine root mass in forests occurs below 30 cm (Jackson et al., 1996, Meinen et al., 2009). The vertical fine-root distribution is modulated by a variety of abiotic and biotic factors. Therefore, it can be expected that if deviations between NFSI and Yasso15 are caused by carbon and fine-root dynamics below 30 cm soil depth, these differ between forest types e.g. the decline of soil acidity in spruce stand could lead to deeper rooting systems (Jentschke et al., 2001). On the other hand, forest soils store more than 50 % of total carbon in the first 1 m within the first 30 cm of the mineral soil (Jobaggy & Jackson, 2000). Including deadwood and organic layer, our study reveals that this share is even higher as carbon stocks of German forest soils down to a depth of 30 cm were estimated at 89.2 Mg C ha^{-1} and down to a depth of 90 cm at 116.9 Mg C ha^{-1}. Various studies showed that deep soil carbon has older radiocarbon compared with surface soils, suggesting that deep soil OC is stable on longer time scales (Scharpenseel et al., 1989, Lorenz & Lal, 2005, Kögel-Knabner et al., 2008). Therefore, the most significant component of soil carbon dynamics is expected to occur in the top 30 cm of the soil, which justifies the comparison of NFSI results with Yasso15 output.

The lower variability and the smaller number of negative values in the Yasso15 outcome compared to the inventory can be attributed mainly to the inaccuracy of the input data and other site-specific variability that were not accounted for in our simulations, as discussed also previously (Lehtonen et al., 2016, Hernandez et al., 2017) and to the uncertainties associated with repeated soil sampling. In our study, the estimation of annual variations in biomass input depends on yield tables and harvest statistics. Deviations from these average trends on individual plots, for example plots with no harvest over the whole period or wind throws, were not modelled. Consequently, on a plot-level, deviations of model output and inventory results are expected. A repetition of the forest inventory would increase the accuracy of the carbon input estimates for the individual plots and would therefore improve the relationship between the model and inventory data. Furthermore, when comparing the model output with the measurements on individual plots, it is important to acknowledge that the measurements of SOC changes taken using repeated soil sampling are also quite uncertain due to various factors, such as spatial variability of soil carbon, the estimated mass of fine earth fraction and analytical accuracy (Goidts et al., 2009, Schrumpf et al., 2011, Grüneberg et al., 2014). Grüneberg et al. (2014) estimated that the methodological uncertainties contribute to total uncertainties of the SOC changes with 0.15 Mg C ha^{-1} a^{-1}. Therefore, the reliability of the simulated results could be improved by using more detailed input data of the model and improving soil carbon measurements for testing the validity of the simulations.

Regarding historical land-use effects on carbon stocks, it seems justifiable to adjust in the model the initial carbon stocks to the measured carbon stocks in the model. This method requires the availability of soil inventory data or knowledge of site history, as spin-up runs until equilibrium during model initialization tend to overestimate initial carbon stocks (Wutzler & Reichstein, 2007). Our method of initialization considers that the soil carbon stocks of Central European forests are below their soil carbon storage potential.

General mechanisms that increase soil carbon accumulation in industrialized countries are feedbacks with the N-cycle due to the impact of nitrogen deposition on decomposition rates (Janssens et al., 2010). The effect is related to litter quality and thus to tree species because mainly lignin degradation is negatively affected. This suggests that carbon sequestration could be more favoured under trees with high lignin content. The application of a coupled C:N:P ecosystem model demonstrated that the effect of N-deposition on soil carbon accumulation (Tipping et al., 2017) amounts to 10% of the total SOC change, which was modelled by positive effects of nitrogen deposition on litter production. Feedbacks with the N-cycle are not included in Yasso15. Therefore, the impact of N-deposition on decomposition of organic matter cannot be indicated, which may contribute to the differences between the NFSI and Yasso15. Nevertheless, the parameterization of Yasso indirectly reflects present-day nutrient effects on decomposition, as it is based on data from leaf litter experiments that were conducted over a wide range of site conditions (Goll et al., 2017). Consequently, an inter-comparison study revealed that models that include nitrogen coupling do not perform better compared with Yasso (Tupek et al., 2016).

5. Conclusions

The results of this study indicate that German forest soils act on average as a carbon sink. Application of the Yasso15 model confirms the results from the NFSI. On average, the simulated carbon balance is lower than the results of the inventory but the differences are not significant. Accumulated carbon is expected to be sensitive to decomposition, because the most stable SOM-fraction in Yasso15 does not change over time. Adjusting initial values to measured soil carbon stocks yields a close relationship between modelled and measured values. When using this initialization method for long-term projections, it is important to note that the initial values are not in steady state and are therefore subject to a trend. On plot-level comparison it is apparent that the model fit to the overall mean dynamics but not the extremes. Especially, conifer-dominated stands on base-poor soils deviate in both directions which pointed to differences in decomposability of litter and recalcitrance of soil organic matter compared to deciduous stands or base-rich soils. A repeated forest inventory would improve the predictions for individual plots, as it would improve the reliability of the estimates of litter input. A third soil inventory would substantially improve the estimates of soil carbon balance as it would provide German-wide data down to 90 cm and a third point in time. The fact that modelled rates are generally lower may be attributed to effects not captured by the Yasso15 model and the used approach for litter input estimation; for example, nitrogen emissions or soil carbon depletion due to land-use changes or intense forest management practices in the past may have an effect. Therefore, the reliability of the simulated results could be improved by repeating collection of stand data, adding site-specific features of the soil carbon cycle to the Yasso model and improving soil carbon measurements for testing the validity of the simulations.

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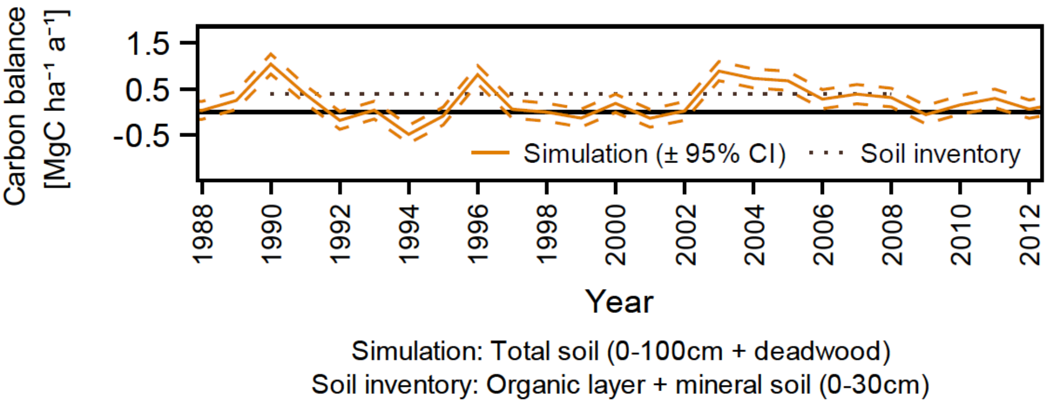

Graphical abstract

Highlights

- Soil organic carbon balances were examined for German forest soil inventory (NFSI)
- Results from the litter and soil carbon model Yasso15 were compared with NFSI-values
- On average soil organic carbon balances are comparable between Yasso15 and the NFSI
- In coniferous forests on base-poor soils results from both methods deviate
- Temperate forest soils are a large carbon sink, which differs between forest types