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ORIGINAL ARTICLE

The impact of climate change under different thinning regimes on carbon sequestration in a German forest district

A. Borys · F. Suckow · C. Reyer · M. Gutsch · P. Lasch-Born

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Abstract The objective of this paper is to assess how much carbon (C) is currently stored in a forest district in Thuringia, Germany, and how the carbon stocks will develop up to the year 2099 with a changing climate and under various management regimes (including no management), with different assumptions about carbon dioxide (CO_2) fertilization effects. We applied the process-based model 4C and a wood product model to a forest district in Germany and evaluated both models for the period from 2002 to 2010, based on forest inventory data for the stands in the district. Then, we simulated the growth of the stands in the forest district under three different realizations of a climate change scenario, combined with different management regimes. Our simulations show that in 2099, between 630 and 1149 t C ha^{-1} will be stored in this district. The simulations also showed that climate change affects carbon sequestration. The no management strategy sequestered the highest amount of carbon (8.7 t C ha^{-1} year^{-1}), which was greater than the management regimes. In the model, the possible fertilization effect of CO_2 is an important factor. However, forest management remains the determining factor in this forest district.

Keywords 4C · CO_2 fertilization · Forest growth and harvest · Thinning from above and below · Unmanaged · Thuringia · Process-based modeling · Wood product model

1 Introduction

Forests are the world's largest terrestrial carbon (C) stock with 1640 Pg C stored in plants and soils to a depth of 3 m (Sabine et al. 2004). Globally, forests store around 300 Pg C in living biomass (Mackey et al. 2013). Preserving and increasing forest C-stocks are crucial, since forests contribute to climate change mitigation globally and, at the same time, provide important other ecosystem services to society at a more regional level. Climate change mitigation is a key global strategy for combating the ever increasing concentration of atmospheric carbon dioxide (CO_2), which is causing anthropogenic climate change. The CO_2

A. Borys (✉) · F. Suckow · C. Reyer · M. Gutsch · P. Lasch-Born
Potsdam Institute for Climate Impact Research, P.O. Box 60 12 03, 14412 Potsdam, Germany
e-mail: Alexander.Borys@gmx.de

concentration in the earth's atmosphere has increased by almost 100 parts per million (ppm) over its preindustrial level, reaching 379 ppm in 2005 (IPCC 2007) and breaching the record daily concentration of more than 400 ppm in 2013, a value not reached in several million years (Monastersky 2013). Therefore, the study of the C-storage potential of forests has been at the center of scientific interest in the past decade in very different forest ecosystems around the world. Ter-Mikaelian et al. (2014) indicate, for example, that short-term changes in C-storage depend on initial forest condition and that there are limits on how these changes can be manipulated by altering harvest and disturbance rates. Ruiz-Peinado et al. (2014) analyzed the effect of thinning on C-stocks in a long-term experiment in Burgos, Spain. Alvarez et al. (2014) considered how forest management impacts on C-stocks in Mediterranean mountain forests based on various combinations of site index, tree species composition, and thinning intensity under different climate scenarios. Wirth et al. (2004) examined the dynamics of forest C-stocks in the Thuringia region in Germany and quantified C-stocks at the stand level for the entire state. They construed C-fluxes from forest inventory data and quantified them by means of the biogeochemical model BIOME-BGC.

At the same, it is also clear that extant climate change has altered the site conditions of forests, which in turn affects their C-sequestration potential (IPCC 2014). However, forest carbon is not only influenced by the climate but also by forest management. Mund and Schulze (2006) analyzed the C-stocks of beech forests under different silvicultural systems. They found that the successive removal of trees in a shelterwood system and a selection system reduces the amount of carbon stored in the tree biomass by, on average, about 30 % compared to an unmanaged forest. Schulze et al. (2006) dealt with the actual C-stocks of the Thuringian Forest by explicitly considering the wood products originating from the forest and the amount of carbon stored in the forest soil. They described how in Thuringian-managed forests the carbon stocks in the living aboveground biomass increased by up to 1.0 t C ha^{-1} $year^{-1}$ between the beginning and the end of the 1990s.

Alvarez et al. (2014) found a strong effect of tree species composition and a negligible effect of thinning intensity. In the study of Ruiz-Peinado et al. (2014), different thinning intensities (moderate, heavy, and not managed) had been applied over the last 30 years in a Scots pine (*Pinus sylvestris* L.) stand, with a thinning rotation period of 10 years. They found that the mitigation capacity of Scots pine stands is slightly modified by thinning, suggesting the long-term sustainability of these interventions in terms of C-stocks. Wördehoff et al. (2011) concluded that C-sequestration in the forest soil, as well as in living and dead biomass and in wood products, is strongly influenced by forest management.

All these studies, however, focus on current conditions and do not consider the projection of possible future forest C-stocks under global change. However, for planning the adaptation to and mitigation of climate change, it is crucial to address the following questions: (1) How do C-stocks develop under already changing site conditions (such as soil, forest management, and climate) and those projected for the future? (2) How much can silvicultural measures potentially influence C-sequestration in the forest and forest products under a changing climate?

Process-based forest models are suitable tools for examining these questions in detail (Mäkelä et al. 2000; Landsberg 2003). Such models particularly need to satisfy the following requirements: to be climate sensitive, able to be initialized with actual forest stand data, and capable of describing different forest management types. Furthermore, it is important that the models are able to represent different soil types in order that they can reflect possible changes in the soil carbon balance. Moreover, to ensure the comparability of managed forests with unmanaged protected areas, the models should possess not only a decomposition model for the deadwood remaining in the stand, but also a model to track the carbon in wood products resulting from timber harvest.

The process-based forest model 4C (Schaber et al. 1999; Lasch et al. 2005) fulfils these requirements: 4C describes complex forest growth processes under changing environmental conditions and forest management (Reyer et al. 2010; Gutsch et al. 2011; Borys et al. 2013), and it is driven by different forest management strategies as well as stand, soil, and climate data and atmospheric CO_2 concentrations. The amount of harvested timber simulated by the model 4C can be taken up in the wood product model (WPM; Eggers 2002) to assess the degradation rates and average time of use (storage period) of different wood products.

The objective of this paper is to assess how much carbon is currently stored in the forest district of Buchfart in Thuringia and how the C-stocks in the forest and in wood products will develop in a changing climate and under different approaches to forest management. Furthermore, we aim to delimit the extent to which different forest management regimes can influence C-sequestration under changing site conditions. In addition, we present a validation of 4C using forest growth data based on forest inventories for the Buchfart public forest.

We take the forest district of Buchfart in Thuringia, Germany, as an example to illustrate how combing a process-based forest model, which is climate sensitive and responsive to different management strategies, with a WPM allows to assess carbon sequestration potential in managed forests. This approach allows to assess forest mitigation strategies in all kinds of forests and does not stop at the biological level—hence, how much carbon is in the forest—but accounts for what is relevant outside of the forest and where society and nature interact, namely the timber products and the influence of different management strategies.

2 Material and methods

2.1 Study area: the forest district Buchfart

The Buchfart forest district is located in the Federal State of Thuringia, Germany (50.90 N/11.34 E). In the period 1980 to 2010, the mean annual mean temperature ranged from 7.0 to 8.5 °C and the annual precipitation ranged from 550 to 770 mm (German Weather Service 2010). The forest area of the district covers approximately 1500 ha, of which 881 ha was state forest in 2011. The private and state forest in the district is managed in the same manner. Since the data is, however, only available for the state forest, we can only present a quantitative analysis of this part of the forest. Hence, our analysis only refers to the 881 ha of the state forest in the district. However, qualitatively, our results are valid for the whole forest district and even for similar forests in temperate Europe.

The last forest management plan from 2002 identified a growing stock of 285,000 m^3 in this area, which is divided into 521 forest stands. The annual prescribed yield was, on average, 11 m^3 ha^{-1} $year^{-1}$ between 2002 and 2010. In 2010, a new inventory (according to Thuringian Forest management order: FA 2010) was carried out showing that the state forest stock includes 65 % beech (*Fagus sylvatica* L.), 16 % Norway spruce (*Picea abies* L. Karst.), 5 % Scots pine (*Pinus sylvestris* L.), and 2 % of oak (*Quercus petraea* Liebl. and *Quercus robur* L.).

2.2 Soil data

The main soil types in the study area were defined by site mapping in the forest management plan (2002). The chemical and physical properties of these soil types were derived from soils of the Buchfarter catena (Fiedler and Hofman 1991) and from soil samples from forest experimental sites (Borys et al. 2013). The nine soils used in this study are described in

Table 1. Soils 1 to 3 are soils from the experimental sites, and soils 4 to 9 are from the Catena. The nine soils describe the variability of the local soil conditions in the forest district: soils 1 to 3 are Cambisols/Chromic Luvisols (Borys et al. 2013); soils 4 to 9 consist of one Orthic Luvisol (soil 4), two Chromic Cambisols (soils 5 and 6), two Calcic Cambisols (soils 7 and 8), and one Eutric Cambisol/Calcic Cambisol (soil 9) (Fiedler and Hofman 1991). The classification of soil types was conducted in accordance with the World Reference Base for Soil Resources (IUSS Working Group WRB 2006).

The carbon content of the four soils of the experimental plots was examined in 2009. In order to validate the C-storage rate in the model, the carbon content of the four soils was reexamined in 2014. The annual storage rate was calculated as the difference between these two studies (2014 and 2009). The measured storage rates were compared with the modeled annual storage rates (2002 to 2010).

2.3 Climate data

This study used daily weather data from the Knau meteorological station (DWD number 21109) for the period 2002 to 2010. The station is located about 36 km from the district and has a mean annual temperature of 7.3 °C and a mean annual precipitation of 651 mm. From 2011 onward, we used climate scenario data following the Representative Concentration Pathway 8.5 (hereafter referred to as RCP 8.5). RCPs are the climate change scenarios developed for the Fifth Assessment Report of the United Nations Intergovernmental Panel on Climate Change (IPCC). Their corresponding Extended Concentration Pathways (ECPs), important for deriving atmospheric CO_2 levels, were developed depending on demographic development, energy production, food production, and land use (Meinshausen et al. 2011; van Vuuren et al. 2011). The RCP 8.5 scenario is marked by a strong radiative forcing reaching 8.5 W m^{-2} in 2100 and resulting in an increase of Germany's mean annual temperature by 3.6–4.1 °C, depending on the general circulation model (GCM). In this study, the results of the GCM ECHAM6-OM, driven by RCP 8.5, were regionalized by the STAtistical Resampling Scheme (STARS; Orlowsky et al. 2008) model. For a given temperature trend derived from a

Table 1 Soil properties of the three experimental sites (soil 1–soil 3) and soils of the Buchfarter Catena (soil 4–soil 9)

Soils	pH-Oh	pH-Ah	Clay, Ah	Silt, Ah	Sand, Ah	C/N-Oh	C/N-Ah	C org., Oh	C org., Ah	C-soil
	($CaCl_2$)	($CaCl_2$)	(%)	(%)	(%)	–	–	(%)	(%)	(t C ha^{-1})
Soil 1	5.4	6.7	45.6	51.7	2.7	27.4	13.5	37.3	4.1	89
Soil 2	5.5	6.4	43.5	53.7	2.8	29.4	13.0	39.2	4.4	121
Soil 3	5.3	5.8	44.9	53.3	1.8	29.1	15.5	37.8	5.3	106
Soil 4	5.0	4.5	13.3	80.8	5.9	25.1	17.0	39.7	7.1	214
Soil 5	4.8	4.7	19.0	71.6	9.4	24.0	17.0	35.3	7.7	88
Soil 6	5.5	6.2	20.0	76.5	3.5	28.5	17.3	33.0	6.9	124
Soil 7	5.7	5.8	29.9	48.7	5.0	27.6	14.2	35.3	15.4	256
Soil 8	5.3	6.5	37.3	51.5	5.0	29.5	15.3	41.6	11.9	215
Soil 9	4.9	6.7	21.4	63.0	5.0	22.6	15.8	32.7	13.0	368
Mean[a]	5.3	5.9	30.5	61.2	4.6	27.0	15.4	36.9	8.4	176

[a] Any differences are due to rounding

GCM, STARS generates 100 different realizations, which follow the temperature trend of the GCM but differ mainly in precipitation and are sorted according to the weighted trends of the climate water balance for some selected meteorological stations (Orlowsky et al. 2008). Here, we used three of these realizations, namely the 5th, 50th, and 95th percentiles (referred to henceforth as CS1, CS2, and CS3; for more details, see Table 2). To capture the uncertainty regarding the persistence of CO_2 fertilization effects on forest productivity (Körner 2006; Norby et al. 2010), we ran 4C with the three climate realizations twice–once with a constant CO_2 concentration at 380 ppm until 2099 and once with an increasing CO_2 (up to 932 ppm CO_2 in 2099) according to the ECP of RCP 8.5.

2.4 Forest management

The management of the 521 stands was derived separately according to tree species and age classes. We considered the age classes for each tree species separately, starting with age class 1 (0 to 20 years old), age class 2 (21 to 40 years old), and so on. The number of age groups differed because not all age classes were available for all tree species (see Table 3). Birch stands, for example, were only represented by two age classes (age class 1 and age class 6). The stands' management was based upon prescribed removals by yield tables for the corresponding age groups. The yield rates per tree species are based on mean site index, as defined by the 2002 Forest Management Plan. Thinning intervals correspond to the requirements of the respective yield tables. The grouping of stands into age classes was not changed during the simulations. We simulated three different management regimes: heavy thinning from above, referred to as {H} in the following, weak thinning from above, referred to as {W} in the following, medium thinning from below, referred to as {M}, and no management, referred to as {N}.

Under {N}, all timber removals ceased and the natural mortality simulated in 4C occurred. The mortality of members of the tree cohorts in 4C is determined depending on the carbon budget of the cohort (stress mortality). The stress mortality appears if the carbon balance of a cohort is negative over a defined period, which leads to a lower foliage biomass in the current year compared with the previous year. In this case, a stress counter is used and a mortality rate is calculated depending on the stress counter and a tolerance class of stress using a Weibull distribution (Keane et al. 1996). Under {H}, {W}, and {M}, no natural mortality is simulated, because in Germany's managed forests, the C-sequestration in dead wood is less than 1 % and very low (Federal Forest Report 2009).

The simulations started in 2002 with no initial C-stock in the wood product {H}, {W}, and {M} and the deadwood pool {N}. For the managed version {H}, the removal percentages were applied to dominant trees and codominant trees, which hindered the good crown

Table 2 Characteristics of the recent climate (baseline) and the three realizations of the applied climate change scenarios (CS1–CS3)

Climate scenarios	Period	Precipitation mean (mm yr^{-1})	Precipitation max (mm yr^{-1})	Precipitation min (mm yr^{-1})	Temperature mean (°C)
Baseline	2002–2010	763	978	569	8.1
CS1	2011–2099	676	972	425	9.9
CS2	2011–2099	716	959	503	9.9
CS3	2011–2099	696	941	378	9.9

Table 3 Composition of age classes of the 521 stands at the forest district for the initialization of 4C

Tree species	Oak	Spruce	Pine	Beech	Birch	Larch
Number of stands	72	93	48	240	7	61
Number of age classes	6	7	6	7	2	5
Area per tree species (ha)	14	142	43	570	81	31

formation of the target trees. The average removal percentage for all tree species was 4.5 % of biomass per year. For {W}, the removal percentages were applied mainly to suppressed trees, and for {M}, the dominant trees were mostly removed. For {W}, the average removal percentage over all tree species was 1.5 % of the biomass per year, and for {M}, the average removal percentage for all tree species was 2.5 % of the biomass per year.

Prescribed yield rates for the forest management from 2002 to 2099 were taken from Dittmar et al. (1983) yield tables for beech for stocking level 1.0 for {H} and stocking level 1.2 for {W}. The stocking level is the quotient of the basal area taken from the yield table and the real basal area of the trees in the forest. The removal percentages for {M} were taken from Schober (1975). Yield rates for Scots pine were taken from Lembcke et al. (1975), with stocking levels of 0.8 for {H}, 1.2 for {W}, and 1.0 for {M}. The removal percentages for Norway spruce were taken from Wenk et al. (1985), for oak from Ertelt (1961), for larch from Schober (1975), and for birch from Tjurin and Naumenko (1956). Since 4C is not parameterized for larch, we relied here on the Douglas fir parameters to simulate the small proportion of larch in the forest in the district, since the two species have shown similar growth behavior in Germany's forests.

2.5 Model 4C

The process-based forest growth model 4C (Lasch et al. 2005) was developed to analyze the growth of forest stands under different climatic and silvicultural conditions. Ecological experiments, tree growth studies, long-term monitoring of stands, and analyses of physiological processes were used for the development of the model. The submodels describe the water and nutrient balance (resources available from atmosphere and soil), the assimilation and allocation of net primary production to categories (coarse and fine roots, trunk, branches, and leaves), species-specific phenology (leafing according to Schaber and Badeck 2005), mortality (stress-induced and age-related), regeneration, and management (setting of management methods). In the model, the trees of a forest stand are classified into cohorts (trees of the same species, age, and dimensions), which compete for light, water, and nutrients. Growth, mortality, and regeneration for each cohort result from the balance of photosynthesis and respiration. These processes are calculated in different timescales: water and heat fluxes as well as phenology in daily cycles; soil carbon and nitrogen dynamics as well as photosynthetic production in daily to weekly cycles; and allocation, growth, mortality, and regeneration in annual cycles. The model simulates stand and soil characteristics as well as water, carbon, and nitrogen fluxes within the ecosystem (Suckow et al. 2001). Amongst other species, 4C is parameterized for beech (*F. sylvatica* L.), Norway spruce (*Picea abies* L. Karst.), Scots pine (*Pinus sylvestris* L.), Douglas fir (*Pseudotsuga menziesii* [Mirb.] Franco), oak (*Q. petraea* Liebl and *Q. robur* L.), birch (*Betula pendula* Roth), and aspen (*Populus tremula* L.), and it has previously been applied to simulate climate change impacts on forest productivity across Europe (Reyer et al. 2014).

2.6 Wood product model

The wood product model (WPM) in 4C (Eggers 2002; Fürstenau et al. 2007; Borys et al. 2013) estimates the C-stock in wood products. It calculates carbon fluxes after timber harvest and simulates wood distribution along the chain of wood products (see Fig. 1). As an input, WPM uses the amount of harvested timber estimated by 4C. Our simulations started in 2002 with no initial C-stock in wood products. The harvested timber is sorted on the basis of the German Rules for Quality Grading of Merchantable Round Wood (HKS), according to middle diameter and top diameter. Possible downgrading caused by wood defects is considered by the allocation of fixed downgrading percentages within each of the grading classes. The wood within different grading classes is assigned to different wood product groups. In doing so, different carbon losses (e.g., sawmill losses) are subtracted in percentages from the respective group. The carbon flux within these product groups is calculated over the entire simulation period. Each product group possesses a special mean durability $f(u^i)$ (a). This mean durability is defined with a life span function (Eq. E1) according to Row and Phelps (1991).

$$f\left(u^i\right) = d - \frac{a}{1 + b\,e^{-ct}} \qquad \text{(E1)}$$

with

u^i	The proportion of products which are in use
a, b, d, e	Parameter according to Row and Phelps (1991)
c	The reciprocal of the halved durability time of the products (a^{-1})

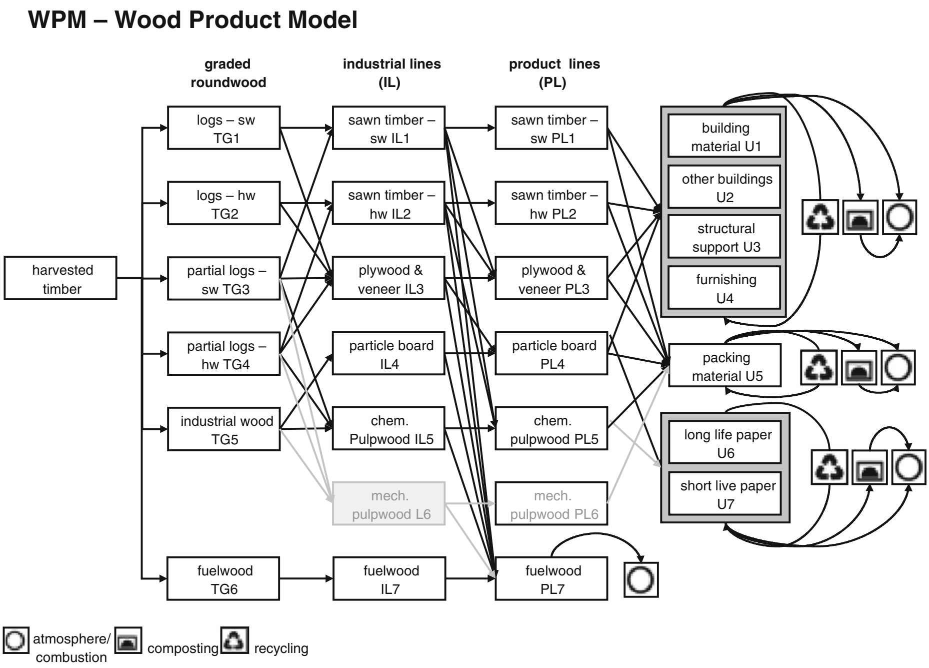

Fig. 1 Carbon flow diagram of the wood product model (Fürstenau 2008)

t	Time (a)
i	Product group (U1 up to U7).

Finally, the wood products leave the product group and can be recycled, composted, or released to the atmosphere by combustion. WPM distinguishes seven different use categories with seven different average lengths of stay and redistribution rates. The mean durability is staggered from short live paper with a mean durability of 1 year (U7) up to building material with a mean durability of 50 years (U1). The in-wood product-stored carbon C_{wpm} (t C ha^{-1}) is defined by Eq. E2.

$$C_{\mathrm{wpm}} = \sum_{i=1}^{7} f\left(u^i\right) \tag{E2}$$

A calculation of CO_2 emission savings in the form of material and energy substitution would be possible using WPM, but it was not considered in this study because the substitution effects are not considered in the Kyoto protocol.

To validate the WPM, the actual harvest data of the state forest from 2002 to 2010 were compared with the logging data from the WPM. The timber harvest volumes from 2002 to 2010 were surveyed by the forestry office for all tree species as a total and not for each individual tree species.

2.7 Model validation and projections until 2099

In a pilot study, we validated 4C and WPM based on four long-term experimental sites (0.25 ha for each experimental sites) of the Technical University of Dresden, which were also located in the forest district "Buchfart" for the time period 1959–2009. 4C and WPM successfully simulated the past growth of four study sites independent of their thinning regime (heavy thinning from above, weak thinning from above, and medium thinning from below and no management). Comparing the measured and simulated stem biomass (dry mass) for the four experimental sites with different management regimes resulted in a linear regression between measured and simulated stem biomass for the timespan 1959–2009, with a correlation coefficient of r=0.94 (for more details, see Borys et al. 2013).

In the context of this study, 4C has been validated also at the forest district level on the basis of forest inventory data collected in 2002 and 2010. The model initialization is based on the inventory data from 2002, which describe tree species, age, height, basal area, and diameter for 521 stands on the area of 881 ha. These data are used to calculate cohort-specific starting values such as diameter or height depending on distribution functions. The data measured in 2010 describes the same variables as the 2002 data, but the stands have been regrouped resulting in 458 stands on the 881 ha.

To validate 4C, we compared the initialized data of 4C for 2002 with the inventory data from 2002 and the end value of the simulation of 4C from 2002 to 2010 (hence, the 2010 value) with inventory data from 2010 (see Table 4). The carbon in biomass C_{BM} was calculated from the inventory data on basal area G and height h_g in 2002 (n=521 stands, A=881 ha) and in 2010 (n=458 stands, A=881 ha) following Eqs. E3 to E5:

$$C_{\mathrm{BM}} = \frac{\sum_{i=1}^{n} C^i A^i}{A} \tag{E3}$$

with

$$C^i = V^i K^i_E \tag{E4}$$

$$V^i = G^i h^i_g f^i \tag{E5}$$

and

C_{BM}	Average carbon stock of aboveground and belowground biomass of the forest district (t C ha^{-1})
C^i	Carbon stock (aboveground and belowground biomass) per stand (t C ha^{-1})
V^i	Volume per stand (m^3 ha^{-1})
G^i	Basal area (m^2)
h^i_g	Height of mean basal area stem (m)
f^i	Form factor for stands (by Kramer and Akça 1985), which depends on tree species and diameter at breast height
K^i_E	Expansion and conversion factor (Wirth et al. 2004, p. 68), which depends on tree species and age, for example, for beech between 1.11 (age 1 up to age 20) and 0.38 (age 81 and older)
A^i	Area per stand (ha)
i	Number of the stand

The total carbon sequestration C_{seq} (t C ha^{-1}) (referred to as C-sequestration in the following) for the time period from year t_1 to t_n (n=98) is calculated as the sum of the change in the carbon stock of the aboveground and belowground biomass C_{BM} (t C ha^{-1}), the change in soil carbon including the organic layer C_{soil} (t C ha^{-1}), carbon sequestered in deadwood C_{DW} (t C ha^{-1}), and the carbon content of all pools of WPM, the end of the time period C_{WPM} (t C ha^{-1}):

$$C_{\mathrm{seq}}(t_n) = C_{\mathrm{BM}}(t_n) - C_{\mathrm{BM}}(t_1) + C_{\mathrm{soil}}(t_n) - C_{\mathrm{soil}}(t_1) + C_{\mathrm{DW}}(t_n) + C_{\mathrm{wpm}}(t_n) \tag{E6}$$

The total carbon stock (referred to as C-stock in the following) is the sum of carbon in the aboveground and belowground biomass C_{BM} (referred to as biomass in the following), the sum of carbon in the organic layer and mineral soil C_{soil} (referred to as soil in the following), the carbon content of the total dead stem wood C_{DW} (referred to as deadwood in the following), and the carbon in wood product pools C_{WPM} (referred to as wood product in the following).

The model projections over a period of 98 years up to the year 2099 were carried out with the model's initialization at 2002. The model simulations were run with a combination of each of the three management regimes and with the no management strategy, with each of the three different realizations of the climate scenario (CS1, CS2, CS3), once with a constant CO_2 concentration and once with increasing CO_2 according to the ECP of RCP 8.5. We present our results as the average for the climate realizations CS1 to CS3 with constant CO_2 (henceforth referred as R380) and as the average for the realizations CS1 to CS3 with increasing CO_2 (henceforth referred as R932).

3 Results

3.1 Model evaluation

The C-stocks and sequestration were estimated by the approach of Wirth et al. (2004) (Eqs. E3–E5) on the basis of inventory data for the forest district for the years 2002 and 2010 and the real harvest data for 2002 to 2010, and these were compared with the values simulated by 4C for the three management regimes and the no management strategy (initialized at 2002 and simulated up to 2010). For the 881 ha of the state forest area of the Buchfart district, a C-stock in the biomass of 130 t C ha^{-1} was estimated in 2002. A C-sequestration rate of 22 t C ha^{-1} from 2002 to 2010 led to an estimated C-stock of 152 t C ha^{-1} in 2010 (see Table 4). In 4C, the initialization of the forest district in 2002 featured 151 t C ha^{-1} of biomass. Figure 2 characterizes the species-specific mean C-stocks in 2002 and indicates that beech stands contained the highest C-stocks (173 t C ha^{-1} in biomass and 145 t C ha^{-1} in soil). In 2010, 4C simulated a minimum of 137 t C ha^{-1} in the biomass in the case of management

Table 4 Carbon stocks of the forest district 2002 and 2010, estimated according to the approach of Wirth et al. 2004 (measurements) and simulated with 4C ({H}, {W}, {M}, {N})

Carbon stock	2002 (t C ha^{-1})	2010 (t C ha^{-1})	Carbon increment per year[a] (t C ha^{-1} year^{-1})
Measurements			
Soil	100	116	1.8
Biomass	130	152	2.4
Real harvest (from 2002 until 2010)	–	44	4.9
Carbon sequestration[a]	–	–	9.2
{H}			
Soil	149	174	2.8
Biomass	151	137	−1.6
Harvest (from 2002 until 2010)	–	55	6.2
Carbon sequestration[a]	–	–	7.4
{W}			
Soil	149	175	2.9
Biomass	151	159	0.9
Harvest (from 2002 until 2010)	–	26	2.8
Carbon sequestration[a]	–	–	6.6
{M}			
Soil	149	174	2.8
Biomass	151	162	1.2
Harvest (from 2002 until 2010)	–	38	4.2
Carbon sequestration[a]	–	–	8.2
{N}			
Soil	149	157	0.9
Biomass	151	186	4.0
Dead wood (from 2002 until 2010)	–	9	1.0
Carbon sequestration[a]	–	–	5.8

[a] Any differences are due to rounding

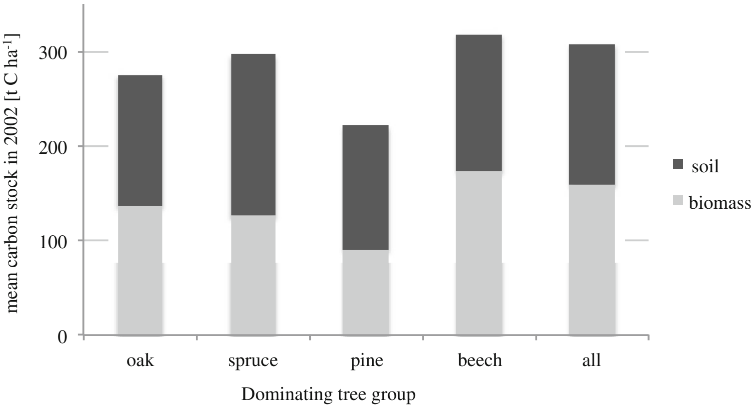

Fig. 2 Mean carbon stock of the year 2002 (used for 4C initialization) pooled for the stands of the four tree species in the forest district Buchfart

regime {H} and a maximum of 186 t C ha^{-1} in biomass for regime {N}. The measured timber harvest (44 t C ha^{-1}) was higher than the simulated harvest in the evaluation period for the {W} (26 t C ha^{-1}) and {M} management regimes (38 t C ha^{-1}). Only the simulated harvest for management regime {H} was 11 t C ha^{-1} higher than the measured timber harvest.

3.2 Carbon stock of the forest district in 2099 under constant carbon concentration (R380)

The highest amount of carbon in the soil was stored with 288 t C ha^{-1} under {M}, which means 22 t C ha^{-1} more than that under realization {H} (see Fig. 3). The highest stock in the biomass was found under {N} with 564 t C ha^{-1}. The lowest stock in the biomass (251 t C ha^{-1}) was simulated with {M}. The highest stock in the wood product was simulated with 110 t C ha^{-1} for {H}, 19 t C ha^{-1} more than that under the same simulations with {M} and 22 t C ha^{-1} more than that under {W}. The deadwood in the case of {N} only stored

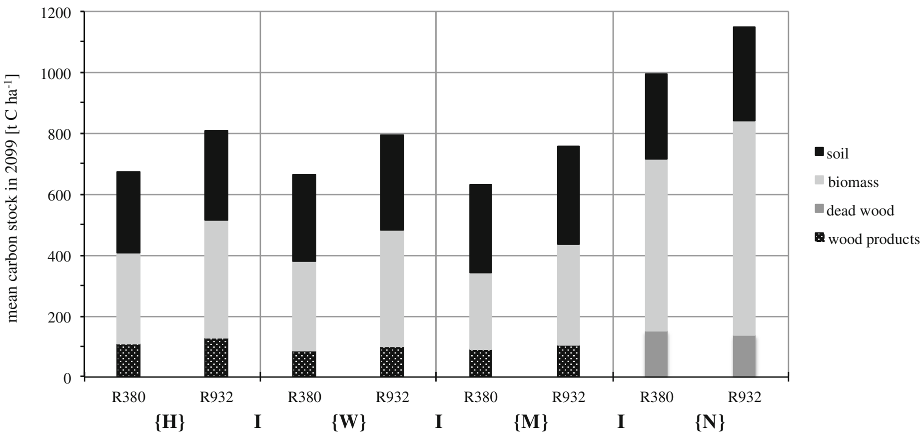

Fig. 3 Simulated mean carbon stock in 2099 of the forest district Buchfart for three management (*H* heavy thinning from above, *W* weak thinning from above, *M* medium thinning from below) strategies and no management (*N*) and the RCP 8.5 climate scenario with constant CO_2 (R380) and increasing CO_2 (R932)

149 t C ha^{-1}. For {N}, 4C simulated the highest C-stock, with 994 t C ha^{-1} in 2099, while under {M}, the lowest C-stock was simulated, with 630 t C ha^{-1}.

3.3 Carbon stock of the forest district in 2099 under increasing carbon concentration (R932)

For the final simulation year (2099), the model simulated the highest amount in the soil under {M}, with 322 t C ha^{-1}. The lowest stock in the soil was found in the case of {H}, with 294 t C ha^{-1}. The highest amount in biomass of 704 t C ha^{-1} was simulated with {N}, and the lowest amount of 332 t C ha^{-1} with {M}, 53 t C ha^{-1} less than with {H}, and 52 t C ha^{-1} less than with {W}. The wood products stored 128 t C ha^{-1} in the case of {H}, which is 28 t C ha^{-1} more than that with {W} and 24 t C ha^{-1} more than that with {M}. The deadwood stored 136 t C ha^{-1} in the {N} strategy only, 13 t C ha^{-1} less than that under constant CO_2 concentration. 4C simulated the highest C-stock (1149 t C ha^{-1}) under the {N} strategy, which is 155 t C ha^{-1} more than that under constant CO_2 concentration. The lowest C-tock was simulated in case of {M}, with 758 t C ha^{-1}.

3.4 Carbon sequestration of the forest district from 2002 to 2099 under constant carbon (R380)

In the soil, 135 and 140 t C ha^{-1} were sequestered with {W} and {M} management regimes, respectively, and 132 t C ha^{-1} was sequestered with {N} and 117 t C ha^{-1} with {H}, respectively (see Table 5). The annual C-sequestration rates varied from 1.2 to 1.4 t C ha^{-1} year^{-1} (see Table 6). The highest amount in the biomass 414 t C ha^{-1} was simulated with {N}. This is 268 t C ha^{-1} more than that with {H}, 271 t C ha^{-1} more than that with {W}, and 314 t C ha^{-1} more than that with {M}. The highest C-sequestration was 695 t C ha^{-1}, simulated for {N}. If the C-storage is considered by tree species, then the larch with {H} management and −2 t C ha^{-1} was the least amount, and the largest amount sequestered was spruce in case of {N}.

Table 5 Carbon sequestration (t C ha^{-1}) of the forest district from 2002 until 2099

2002–2099	R380	(CO_2 constant)			R932	(CO_2 increasing)		
	{H}	{W}	{M}	{N}	{H}	{W}	{M}	{N}
Soil	117	135	140	132	145	163	173	161
Biomass	146	143	100	414	235	233	182	554
Wood product	110	88	91	0	128	100	104	0
Dead wood	0	0	0	149	0	0	0	136
Total carbon sequestration	374	366	331	695	508	497	459	850
Biomass per tree species[a]								
Oak	22	112	165	425	86	195	210	568
Spruce	244	276	257	620	365	399	372	749
Pine	304	333	341	598	398	434	391	777
Beech	103	73	37	480	182	144	68	632
Birch	69	229	176	306	99	315	295	509
Larch	−2	309	155	405	27	407	240	533

[a] Not area-weighted

Table 6 Carbon sequestration per year (t C ha^{-1} $year^{-1}$) of the forest district from 2002 until 2099

2002–2099	R380	(CO_2 constant)			R932	(CO_2 increasing)		
	{H}	{W}	{M}	{N}	{H}	{W}	{M}	{N}
Soil	1.2	1.4	1.4	1.3	1.5	1.7	1.8	1.6
Biomass	1.5	1.5	1.0	4.2	2.4	2.4	1.9	5.7
Wood product	1.1	0.9	0.9	0.0	1.3	1.0	1.1	0.0
Dead wood	0.0	0.0	0.0	1.5	0.0	0.0	0.0	1.4
Total carbon sequestration	3.8	3.7	3.4	7.1	5.2	5.1	4.7	8.7
Biomass per tree species[a]								
Oak	0.2	1.1	1.7	4.3	0.9	2.0	2.1	5.8
Spruce	2.5	2.8	2.6	6.3	3.7	4.1	3.8	7.6
Pine	3.1	3.4	3.5	6.1	4.1	4.4	4.0	7.9
Beech	1.0	0.7	0.4	4.9	1.9	1.5	0.7	6.5
Birch	0.7	2.3	1.8	3.1	1.0	3.2	3.0	5.2
Larch	0.0	3.2	1.6	4.1	0.3	4.2	2.4	5.4

[a] Not area-weighted

3.5 Carbon sequestration of the forest district from 2002 to 2099 under increasing carbon (R932)

Under increasing CO_2 conditions, the soil sequestered 173 t C ha^{-1} with {M}, 163 t C ha^{-1} with {W}, 161 t C ha^{-1} with {N}, and 145 t C ha^{-1} with {H}. The annual C-sequestration rates varied from 1.6 to 1.8 t C ha^{-1} $year^{-1}$ and were therefore higher than under constant CO_2 (see Table 6). The highest amount sequestered in the biomass was with {N}, namely 554 t C ha^{-1}, which is 319 t C ha^{-1} more than that with {H}, 321 t C ha^{-1} more than that with {W}, and 372 t C ha^{-1} more than that with {M}. Also, under increasing CO_2 conditions, the highest amount of C-sequestration was simulated for the realizations with {N}, with a C-sequestration of 850 t C ha^{-1}. The lowest amount of C-sequestration was simulated for {M}, with 459 t C ha^{-1}. If the carbon sink is broken down by tree species, in the case of {H} management regime, larch sequestered the least amount, 0.3 t C ha^{-1} $year^{-1}$. Pines saved the largest amount of carbon, 7.9 t C ha^{-1} $year^{-1}$ in {N}.

4 Discussion

4.1 Model evaluation

4C overestimated the initialized C-stock (2002) and underestimated the C-sequestrations (2002 to 2010) of biomass, except for the case of C-sequestrations under {N}. This is consistent with the results of the pilot study by Borys et al. (2013). In the pilot study, 4C was validated for the period from 1959 to 2009 for the same management strategies at the four long-term experimental sites in the forest district. In the present study, we validated 4C for the actual forest management (2002 to 2010) and with our three management strategies and the no management strategy for the same time span to see how sensitive 4C was to the various management strategies and how these differed from the actual forest management. However, 4C underestimated the C-sequestration in all cases. The C-sequestration simulated with 4C was between 63 and 90 % of the measured C-sequestration as a function of the management systems for all 521 forest stands in the period from 2002 to 2010 (see Table 4).

The model's evaluation for 2002 and 2010 shows that 4C overestimates the absolute C-stock in the biomass in comparison to the estimates based on inventory data and the approach of Wirth et al. (2004). For the period from 2002 to 2010, 4C underestimates sequestration in the biomass and in the harvest, except for the harvest with {H} and the C-sequestration in the biomass in {N}. There is a similar change in the C-stock from 2002 to 2010 for the actual measurement and the simulation, and this shows clearly that the model reacts sensitively to fluctuations in the harvest. The difference between the harvested timber in reality and in the model can be explained by the simplified management approach used for 4C in connection with using yield table specifications, as the model does not allow the thinning measures applied in reality to be simulated in detail. For example, it is not possible to calculate stem wood assortments in 4C because the WPM uses only long timber sections of stem wood.

This is one reason why the initial values of 4C (2002) were already 14 % higher than the measured values (biomass). One other potential reason for this difference is that, in order to generate cohorts in 4C, the forest data used for initialization (mean height, mean diameter, stand density, basal area) were determined by attributing assumed diameter distributions and diameter height relations, thereby producing virtual individual trees in the cohorts. Whilst this averaging process must inevitably lead to deviations, this is not the case if single tree data are available for stand initializations, as was the case in the pilot study by Borys et al. (2013).

Based on the approach of Wirth et al. (2004), the deviation between the 4C results and the estimates of carbon in the biomass in 2010 was 5 % (average over all realizations 161 t C ha^{-1}). This difference partly originates from the mismatch between the initialization of 4C and the measured data in 2002, which is in turn due to the unknown diameter and height distribution of the forest stands. Moreover, Wirth et al. (2004) identified an error of the expansion and conversion factor, which, depending on tree species, varied from 8.6 to 17.4 %.

Regarding soil carbon, Wäldchen et al. (2013) found (for the Hainich-Dün region, also located in Thuringia, Germany) an average C-stock in mineral soil of 147±60 t C ha^{-1} and an average C-stock of the organic layer of 5±3 t C ha^{-1}. For the initialization of 4C, we derived from the soil data an area-weighted average total soil C-stock of 147 t C ha^{-1} (see Fig. 2) and a non-area-weighted average total soil C-stock of 176 t C ha^{-1}, as described in Table 1. In 2009, we found an area-weighted average total soil C-stock of 101 t C ha^{-1} at the four experimental sites and, in 2014, an area-weighted average total soil C-stock of 110 t C ha^{-1}. Consequently, we measured the C-sequestration rate as 1.8 t C ha^{-1} year^{-1} at the mineral soil and organic layer over a time span of 5 years. Our model simulates carbon rates between 2.9 t C ha^{-1} year^{-1} for {W} and 0.9 t C ha^{-1} year^{-1} in case of {N} (see Table 4). Thus, 4C also simulates soil carbon (organic layer included) of the same order of magnitude as that found at the four experimental sites and in observational studies in the region.

The mean C-stocks in the biomass for the initialization of 2002 in 4C are quite similar to the results of Wutzler et al. (2007) and Klein et al. (2013). Wutzler et al. (2007) found 100± 6 t C ha^{-1} in the biomass. For the main tree species in Bavaria (Germany), Klein et al. (2013) found between 163 t C ha^{-1} (for pine) and 202 t C ha^{-1} (for beech) in the biomass. In this paper, we derived from the stand data a non-area-weighted mean C-stock in biomass (2002) of 137 t C ha^{-1} (see Fig. 2). Our result for beech (173 t C ha^{-1}) agrees with results of Mund and Schulze (2006). They found that the carbon content of the biomass in an unmanaged beech forest was 238 t C ha^{-1} and in a managed beech forest (managed by the shelterwood system) was 155 t C ha^{-1}. Rademacher et al. (2009) also calculated biomasses of between 196 and 248 t C ha^{-1} for managed beech stands in the Solling and the Göttingen forests.

Heinsdorf and Kraus (1990) conducted extensive studies on the biomasses of pine stands in eastern Germany. Here, an 85-year-old pine stand with an average height of 24 m had a C-stock in biomass of about 99 t C ha^{-1}. We found, similar to this, an average of 91 t C ha^{-1} in biomass for the pine stands in the forest district (a mean age of 85 years for pine in the forest district at 2002, see Fig. 2).

4.2 Projections of carbon stock and sequestration of the forest district

In 2099, the C-stock of the forest district simulated with 4C was clearly higher than that in 2002. A rough estimation shows that 297 t C ha^{-1} in biomass (mean for R380 with {H} management in 2099) would mean a growing stock of about 800 m^3 ha^{-1}. Today, we have about 320 m^3 ha^{-1} in Germany (Federal Forest Report 2009). The reason for this doubling of the future growing stocks could be that today, the age class structure of the forests in Germany is characterized by a high fraction of the area (21 %) in the second to fourth age group (21–100 years). The high percentage in the third age group is a consequence of overexploitation before, during, and after the Second World War, followed by bark beetle calamities in the 1950s and 1960s and the subsequent reforestation. Due to this, the high age and high growth stock forest areas have decreased markedly today (Federal Forest Report 2009). However, the harvesting systems have been changed from clear cutting and shelterwood systems to a single stem harvesting with substantially longer rotation periods. Because of these changes, there will definitely be more carbon stored in managed forests in the next 100 years than there is today. Therefore, the shift in the age–class distribution is mapped very well by 4C, but the model has deficits in describing complex management regimes. Probably, the management strategies are implemented in 4C in too simple a way, so that the different thinning regimes do not lead to changes as strong as those we would expect in reality. This should be improved in future studies.

The C-stock increased from 2002 until 2099 by about 3.4 t C ha^{-1} year^{-1} under climate realization R380 managed by {M} and by about 8.7 t C ha^{-1} year^{-1} under climate realization R932 with {N}. It seems that the availability of water for forest stands, despite the declining precipitation in the forest district (see Table 2), will not be a factor limiting growth. Generally, the largest amount of carbon was sequestered was under {N}, specifically 554 t C ha^{-1} in biomass (R932). In 2099, the carbon content of the simulated biomass varied between 100 t C ha^{-1} ({M} R380) and 554 t C ha^{-1} ({N} with R932). Moreover, our results show that a change of management has an impact on the C-stock in the soil, but it most severely affects the C-stocks in biomass, deadwood, and wood products (see Table 5).

Mund and Schulze (2006) concluded that the absence of forest management leads to an increase in C-stock in beech forests. A direct effect of forest management on C-sequestration in soils could not be proven by our study. These findings are consistent with previous studies arguing that harvesting does not significantly affect soil organic carbon (Johnson and Curtis 2001; Yanai et al. 2003).

The first support for this result was from Kahl (2008), who found that increased deadwood in beech forests would not increase the soil C-stock. Probably, the lack of free binding sites on mineral surfaces prevented a detectable additional to storage of carbon by deadwood in the mineral soil.

For Lower Saxony, Wördehoff et al. (2011) derived a C-stock of 133 t C ha^{-1} (for 2006) in the biomass of beech stands. In their projections up to the year 2036, the amount of living biomass of beech stands managed in a “nature conservation-oriented” way will increase by about 24 % up to 165 t C ha^{-1}, at an annual rate of 1.1 t C ha^{-1} year^{-1}. Our growth rates for biomass are consistent with this rate since they are between 1.0 t C ha^{-1} year^{-1} for {M} R380 and 5.7 t C ha^{-1} year^{-1} for {N} R932 (see Table 6). The C-sequestration in the soil varies from

117 t C ha^{-1} (1.2 t C ha^{-1} year^{-1}) with {H} R380 and up to 173 t C ha^{-1} (1.8 t C ha^{-1} year^{-1}) when managed under {M} R932. Wutzler (2008) indicated a higher possible carbon sink for mineral soil and an organic layer of 5.7±1.5 t C ha^{-1} year^{-1} in the next 100 years. Schulze et al. (2000) reported an average carbon immobilization in soils of 1.44±0.92 t C ha^{-1} year^{-1} along a gradient of forest sites in Europe, which corresponds well with simulated C-sequestration in the soil with 4C.

Under {N}, between 12 % (R380) and 15 % (R932) of the carbon was stored in the deadwood. This is the only case in our simulations in which a carbon storage pool under constant CO_2 conditions has a higher sequestration rate (1.5 t C ha^{-1} year^{-1}) than under increasing CO_2 conditions (1.4 t C ha^{-1} year^{-1}). The modeled deadwood stocks all contain deadwood biomass, including amounts with advanced decay, which usually cannot be measured in the forest. Our modeled deadwood C-stocks were higher than for most results in the literature. The reasons for this might be that the decay rates applied in this study are too low or that the definition of deadwood differs in the studies from the literature. All kinds of deadwood are considered in the model, whereas often, only deadwood with a diameter at breast height (DBH) of greater than 20 cm is measured, for example, in the National German Forest Inventory (BWI) from 2002. Klein et al. (2013) found that in unmanaged forests, 10 to 12 % of the C-stock is stored in deadwood. They considered all deadwoods with a DBH greater than 7 cm. Still, compared to the deadwood stocks measured for European beech forests, the model results seem to be high, although within the range of various studies.

WPM calculates a C-sequestration rate between 0.9 and 1.3 t C ha^{-1} year^{-1} in all managed realizations up to 2099 (see Table 6). Profft et al. (2009) found an average of about 1.1 t C ha^{-1} year^{-1} in the forest enterprise of Hummelshain in Thuringia, Germany, for the decade 1999–2009. For carbon storage in the harvested wood product pool, Klein et al. (2013) found a stock of 115 t C ha^{-1} after 115 years for spruce in Bavaria. After a simulation period of 180 years, the wood product pool of Klein et al. (2013) varies between 72.8 t C ha^{-1} (spruce) and 52.7 t C ha^{-1} (oak). The WPM was stored between 88 t C ha^{-1} ({W} R380) and 128 t C ha^{-1} ({H} R932) after 99 years.

4.3 Effects of CO_2 and forest management

A transient CO_2 fertilization effect from 2011 to 2099 caused the largest change in C-sequestration due to environmental factors of about 155 t C ha^{-1} ({N} R380 compared to {N} R932). For the same site conditions, a maximum change in C-sequestration of about 392 t C ha^{-1} was simulated by changing forest management ({M} R932 compared with {N} R932). This highlights that a change in forest management has a much greater impact on the future C-sequestration than our assumptions about the possible fertilization effect of increasing CO_2 concentration. Also, Wutzler (2008) and our pilot study (Borys et al. 2013) have anticipated that the effects of climate change in the next 100 years will be to a large extent overruled by forest management. However, it is important to note that our assumptions about CO_2 fertilization are very strong and express a very large uncertainty about the impact of CO_2 compared to the well-known differences between management systems. Reyer et al. (2014) discussed in detail how the current photosynthesis formulation in 4C may be overly optimistic about the positive effects of CO_2 if other environmental factors are limited.

4.4 Managed vs. unmanaged stands

During the projection period, the managed stands sequestered less in the C-stock than the unmanaged stands. This applies to all climate change scenarios. Compared with this, the difference between the C-stocks simulated by the three management methods was much smaller. It was less than 12 % of the C-stock or 43 t C ha^{-1} if the fertilization effect of increasing CO_2 was not considered.

All the applied thinning methods are based on a special level of intervention, which depends on the yield table data and the kind of intervention, that is, the type of thinning. Firstly, changing the kind of intervention from thinning from above to thinning from below had a small impact (around 10 %) on C-sequestration, which is demonstrated, for example, by the C-sequestration for {W} R380 (366 t C ha^{-1}) and for {M} R380 (331 t C ha^{-1}). While both management regimes involve a quite similar level of intervention, it is a different kind of intervention. Secondly, changing the level of intervention from {H} to {W} also had a small impact (around 2 %) on C-sequestration, which is demonstrated by the C-sequestration for {H} R380 (374 t C ha^{-1}) and for {W} R380 (366 t C ha^{-1}). Also, Alvarez et al. (2014) found only a negligible effect of thinning intensity on C-stock, but a strong effect of composition of tree species for Scots pine (*Pinus sylvestris* L.) and Pyrenean oak (*Quercus pyrenaica* Willd.) mixed forests in Central Spain. In line with this, we have also observed large differences in C-sequestration for the different tree species.

Only the presence of interventions strongly influenced the C-sequestration; for example, the C-sequestration of {H} R380 (374 t C ha^{-1}) differs from that of {N} R380 (695 t C ha^{-1}) by about 86 % or 121 t C ha^{-1}. These findings are consistent with previous studies arguing that mean annual net ecosystem productivity is not changed by commercial thinning (Wang et al. 2013). At a first glance, it looks that the no management strategy is best for the maximization of C-sequestration in this area, without considering carbon losses from disturbances and without considering material and energy substitution.

Globally, disturbances and extreme events, however, have been shown to influence carbon sequestration substantially and threatening the mitigation potential of forests. Ma et al. (2012) reported that the aboveground living biomass might be reduced owing to increased tree mortality in boreal forests. In response to climate warming, Peng et al. (2011) found a massive increase in tree mortality in unmanaged Canadian boreal forests in response to climate warming. Zhao and Running 2010 have concluded that droughts from 2000 to 2009 on a global scale resulted in a decline of forest productivity. In the light of this, it is increasingly apparent that careful forest management is necessary under changing global climate conditions to enhance the mitigation potential of forests, even if payments for ecosystem services such as C-sequestration are becoming reality. But, also taking heavily managed forests out of wood production could be part of a portfolio of sustainable forest management strategies and forest mitigation strategies in response to climate change that have substantial cobenefits with nature conservation. These is consistent with previous studies arguing that forest management should maximize increments, not stocks, to be more efficient in sense of climate change mitigation (Kindermann et al. 2013).

Although we considered several future climate projections, a limitation of our study is that the climate scenarios do not cover extreme events such as increasing storms or fire. Furthermore, the model 4C does not account for the effects of storms, forest fires, and pest infestations on forest stands. The results of this study are only valid for the assumed initial conditions of sites and stands and only under the applied climate projections. We do not assign any probabilities to the occurrence of the future climatic conditions used here. The long-term trend of global CO_2 emissions, which affects the development of the future climate, is still unknown. Given these limitations, and depending on the selected climate change scenarios, the study shows a range of possible impacts of climate change on C-sequestration at forest district level and the interrelation between climate projections and management scenarios.

For prospective estimations of C-sequestration in similar forests, either intervention or non-intervention (rather than the level or the kind of intervention) could be considered another important control variable. Moreover, it would be interesting to test different management approaches in the model that focus, for example, on single tree selection. In general, our approach of combining projections from a process-based model with an empirically based model of forest products can be used as an example to assess forest mitigation potential in forest all over the globe if data on forest growth changes under climate change, effects of forest management strategies, and empirical information on different forest products is available. For tropical forests, for example, simulations from dynamic global vegetation models could be coupled with regional data of forest products to study the mitigation potential of different ways of using the forest. Such studies are crucial to bring highly regional forest management strategies into global mitigation strategies.

5 Conclusions

This study presents an application of a process-based forest model and a WPM at forest district level in Thuringia, Germany. Our results are qualitatively valid for the whole forest district and even for similar forests in temperate Europe. The study shows that the forest model 4C and the WPM together can depict the development of forest stands and harvests at forest district level and that a variety of management measures and climate scenarios can be studied.

Our simulations show clearly that climate change affects C-sequestration. In general, they also show that the influence of forest management on C-sequestration is still much stronger than the influence of a possible climate change.

The study highlights that only the presence of interventions strongly influenced the C-sequestration. Changing the level of intervention (intensity of thinning) or kind of intervention (type of thinning) had, in best case, a small impact on C-sequestration.

In the model, the most uncertain factors remain the possible CO_2 fertilization effect and the lack of disturbances.

Moreover, this study provides an example on how different forest management strategies can be assessed up to the wood product level under different climate change scenarios to inform forest mitigation strategies. Our approach can be adopted to assess forest mitigation strategies in different world regions, and regional studies such as this one are needed to reflect regional climate and management changes in global mitigation strategies.

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