

Review

Carbon sequestration and forest management

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Abstract

Forest management has the potential to increase the terrestrial C pool. According to the rules of the Kyoto Protocol and of the United Nations Framework Convention on Climate Change, forestry can generate a sink for greenhouse gases that can contribute to meeting the national commitment to emissions reductions. Afforestation is a common strategy that over the course of decades leads to the incorporation of carbon dioxide (CO₂) in plant biomass. However, site types such as wetlands and peatlands may even be a source of greenhouse gases when they are afforested. Adapted management of existing forests may have a less obvious or slower effect on the terrestrial C pool. It is mainly relevant in countries that already have a large forest cover. We analysed the effects of harvesting, rotation length, thinning, fertilizer application and tree-species selection. All these treatments have an impact on the forest productivity and consequently on C sequestration in the ecosystem. Many forest treatments are already an integral part of sustainable forestry practice. In the context of C sequestration and its accounting in national greenhouse-gas budgets, ecosystem stability is highly rated. Forests that are robust against disturbances up to a certain degree of severity are better suited for political commitments than stands of maximum productivity with a high risk of damages.

Keywords: Forest management, Ecosystem dynamics, C sequestration

Introduction

Forests comprise the largest C pool of all terrestrial ecosystems and the annual gross exchange of CO₂ between forests and the atmosphere exceeds the anthropogenic release of CO₂ due to combustion of fossil fuels more than seven times. Obviously, forest C dynamics cannot be ignored when ways to mitigate climate change are sought. Changes in land use necessarily have a strong effect on the terrestrial C pool. Deforestation can mobilize large quantities of C, and afforestation can bind CO₂ in new biomass and dead organic matter. These changes in land use are regionally of different relevance. Deforestation is ongoing at high rates mostly in tropical regions, where forests are converted to agricultural land and valuable timber species are exploited. Afforestation is commonly dominant in regions where

incentives for agriculture are weak and where land owners resort to the less intensive forestry. Large-scale afforestation projects can therefore indicate a societal change. Examples are afforestations in Mediterranean European countries, the UK, in South America, and in China.

When a change in the land use is not an issue, adapted ways of managing existing forests are required. Strategies are sought that increase the C pool in forest ecosystems and also provide the traditional services of forests such as timber production, biodiversity and protection against natural hazards such as landslides and avalanches.

In order to be relevant for the mitigation of climatic change, the C pool of the land and in forest products needs to be increased sustainably and the change in the C pool needs to be verifiable. The role of forests in the mitigation of climate change is acknowledged in the

United Nations Framework Convention on Climate Change (UNFCCC), which committed all industrialized countries to adopt measures for the reduction in greenhouse-gas emissions. A complex set of rules for applicable measures and verification is part of the Kyoto Protocol, which came into force in 2005.

The accounting rules specify that the mere existence of large terrestrial C pools in forest ecosystems represents no advantage for countries. Relevant for the mitigation of climate change are only *changes* in the terrestrial C pool. Countries are required to maintain their forest cover, and the increase in the C pool by specific forms of forest management is important. In countries with a low forest cover, the major opportunity is afforestation of former arable land. The potential for afforestation within Europe has been estimated at about 20% of the agricultural land area [1].

Increasing the terrestrial C pools is a low-tech option and is certainly limited by the availability of land surface and the extent of forest management effects on C pools. Neither the establishment of new forests nor the increase in the C density in existing forests is a long-term solution. They only lead to a one-time sequestration of C realized mainly during one rotation period. Future accounting schemes may put more emphasis on the C storage in harvested wood products.

Forests of the Northern Hemisphere have been identified as a C sink [2]. European forests absorb 7–12% of European emissions [3]. Forest encroachment, i.e. the succession after active farming from bushland to early successional forests, comprises a net C sink. It is controversial whether or not the conversion of grassland to forest is indeed a C sink [4].

Carbon in terrestrial ecosystems is present in forms that have a wide range of turnover times. Biomass can store C during one rotation period. Thereafter, the fate of C depends on the utilization of the biomass. The use of biomass as a source of energy returns C to the atmosphere immediately. Different types of wood products have life spans of decades and in exceptional cases even of centuries (paper, construction material and furniture). Overall, the storage of C in the biomass is relevant in the time range of decades. A promising field of the forest industry is to substitute products of the chemical industry with goods that are based on renewable resources. The ideal situation is to utilize wood products along a cascade of products that are finally used as fuel material.

Soils globally store four times as much C as the aboveground biomass [5]. Soil organic matter (SOM) contains a rapidly decomposing fraction that is less relevant for long-term C storage and a fraction of extremely long turnover times. It remains to be shown that this recalcitrant soil C pool can be influenced by forest management practices [6].

The effects of forest management strategies have been treated in a number of reviews that elaborated on effects of land-use change (L), rotation length (R), thinning

regimes (T), harvesting methods (H), site preparation (S), nitrogen fertilization (N) and nitrogen fixers (F). The purpose of the current paper is to extract some general conclusions from these reviews. For more detailed reviews refer to the following papers: Johnson [7] on S, N, and F, de Wit and Kvindesland [8] on R, T, H and N, Post and Kwon [9] on L, Johnson and Curtis [10] on F, N and H, Guo and Gifford [11] on L, Freeman *et al.* [12] on L, R, T, H, S, N and F and Jandl *et al.* [13] on L, R, T, H, S, N and F.

This review covers two aspects of forest management:

- the design of forests that fulfil the assigned tasks (afforestation, selection of tree species, and peatland management);
- management of existing forests (thinning, fertilization, and specific forest management strategies).

The Role of SOM

The size of the soil C pool is determined by the balance between C input from litterfall, belowground litter production, rhizodeposition, the release of C during decomposition and leaching of dissolved organic C (DOC). The relevance of the soil C dynamics for C sequestration in terrestrial ecosystems depends on the temporal scale of the question. In experiments of a few years, the change in the soil C pool is negligible. The major C fluxes in the soil are the C emission by heterotrophic and autotrophic respiration and the C influx by aboveground litterfall and root litter. In short-term experiments it is a challenge to estimate the total belowground C allocation in order to quantify the amount of C transferred to the soil via the root system. Experimental approaches to assess the short-term C fluxes are available [14].

Over decadal or longer time scales the dynamics of the pool of SOM cannot be ignored. One approach is to measure all processes that lead to a change in the soil C pool. The required workload is substantial, because C needs to be captured as a gas (CO₂), as a solid (biomass), and in the aqueous phase (DOC). Whereas the CO₂ emissions have a long history of measurements [15], the fluxes of DOC have only recently received more attention [16, 17]. DOC as an anion adsorbs strongly to the mineral soil. However, in shallow soils, soils that are rich in organic C and at rapid water fluxes with short contact times to the soil the adsorption is incomplete and DOC leaches through the soil. Accordingly, the export of dissolved organic matter has been identified as an important contribution to the global C flux [18].

In contrast, the change of the solid phase can be measured without obtaining knowledge of the controlling processes. The challenge of the latter approach is to measure small changes of a large pool that is known to have a considerable spatial variability. The high interest in

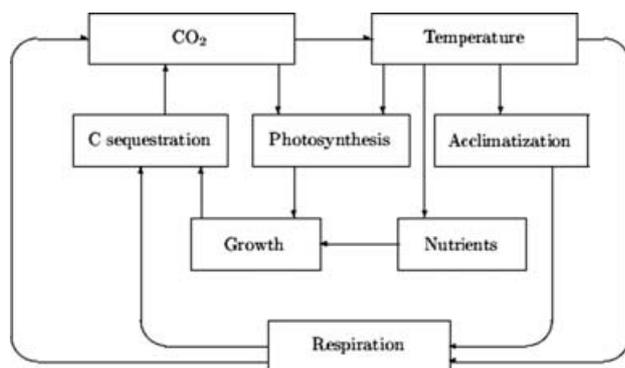


Figure 1 Feedbacks in a climate–C cycle system. CO₂ accelerates the growth rate and C sequestration. The ecosystem will acclimatize and respiration will reach a new level that will either lead to C sequestration or to the formation of CO₂ [23]

soil C arises because C stored in the soil can occur in several forms with different degrees of protection from decomposition. The most stable parts of SOM have turnover times of hundreds of years. Operationally, soil C is classified into a labile and a recalcitrant (stable) pools. Both pools are decomposable and release C into the atmosphere and into the groundwater. Increasing the size of the stable C pool is an appealing management option but methods to achieve this in forest ecosystems and the extent to which stable soil C pools can be increased are both poorly understood. Management options to increase C in agricultural soils have been most successful where soils have previously been depleted of C through ploughing and other management practices. Such C-depleted soils are, however, not common in forest ecosystems.

The soil C pool does respond to climate change (Figure 1). A prominent example is the substantial change in C-rich soils in the southwest parts of the UK [19]. The alarming result of a long-term soil monitoring exercise was that many soils are already sources of C, thereby not playing the desired role as terrestrial C sinks. These results are, however, not generally valid. In the longest available time series of soil data (150 years) from Russian grasslands, only insignificant changes in the soil pool were found. A comparison of recent data with archived soil material from the Russian steppe shows minimal changes over a century. Despite cultivation and global warming the recalcitrant C stock remained unchanged [20]. However, the C pool of forest soils responds more strongly to soil warming than soils under other forms of land use [21].

Given the spatial variability of soil properties, very large numbers of soil samples are required to statistically support changes in the soil C pool [22]. Economic constraints are imposed and the required sample size is not reached. As an alternative, soil models can be used. But calibration and validation of sophisticated soil process models and

the representation of climate change responses within such model remain a significant challenge.

Afforestation

Forests have a higher C density than other types of ecosystems [24]. The terrestrial C pool has been greatly reduced by human activities such as conversion of forests into agricultural land and urban areas. Among the consequences was a reduction of the soil C pool. In many temperate ecosystems, the currently observed C sink in afforested sites is a reversal of past C losses that were caused by the agricultural use of the land [25, 26].

The afforestation of former agricultural land increases the C pool in the aboveground biomass and replenishes the dead organic matter (litter and woody debris) and soil C pools. Accumulation occurs until the trees mature and the soil reaches a new equilibrium between C input (litterfall and rhizodeposition) and C output (respiration and leaching). Recent reviews report that the average rate of soil C sequestration was 0.3 t C/ha/yr (range 0–3 t C/ha/yr) across different climatic zones [9]. On average afforestation increases total soil C stocks by 18% over a variable number of years [11]. The initial soil C accumulation occurs in the forest floor. Its thickness and chemical properties vary with tree species [27, 28].

Changes in soil C storage have been reported from a number of studies based on stand chronosequences, paired plots and repeated sampling. Results are quite diverse as soils may gain C, experience no change, or even lose C following afforestation [11, 29]. Carbon can be lost from soils in a brief period following afforestation, when there is an imbalance between C loss by soil microbial respiration and C gain by litterfall and rhizodeposition. Site preparation and mechanical weed control can lead to soil disturbance and can stimulate the mineralization of SOM in the first years following afforestation. These losses are not necessarily offset by the low C input by litterfall in a young plantation as C gains in the upper mineral soil of plantation forests can be offset by losses of old C from deeper parts of the soil [14, 29–32]. Radiocarbon analyses and ¹³C tracer experiments have shown that litter-derived C was moved into the mineral soil, but it remained unstabilized and was lost rapidly by decomposition [33, 34]. The available long-term experiments found that several decades after afforestation more C is moved to the mineral soil [9, 35–42].

Conditions that are not conducive to soil microbial activity such as sandy texture, low nutrient availability, and low pH can lead to the formation of a thick forest floor layer [27, 43, 44]. It is less certain how C sequestration in the mineral soil is affected by the soil type. In some cases, fertile and clayey soils stored more C, because the production of above- and belowground litter is high and because the formation of organo-mineral complexes protect SOM from decomposition [45, 46]. In other cases,

Table 1 Rates of total C sequestration and the relative contribution of soil and biomass components in different chronosequences of afforestation. Rates of C sequestration may not add up to ecosystem total due to different treatment of non-significant regressions and unlisted pools (e.g. woody debris). Modified from Vesterdal *et al.* [27]

Forest type	Time (yr)	C sequestration (t/ha/yr)				Ecosystem total	Soil contribution to ecosystem C sequestration (%)	Mineral soil contribution to the sum of mineral soil + forest floor C sequestration (%)	Source
		Plant biomass	Forest floor	Mineral soil (sample depth in cm)					
Former cropland									
Oak, Denmark	31	2.72	0.08	-0.47 (25)	2.36	~0	~0	Vesterdal <i>et al.</i> [47]	
Oak and Norway spruce, the Netherlands	29	4.55 ¹	0.34	1.14 (25)	7.14 ¹	20	77	Vesterdal <i>et al.</i> [47]	
Norway spruce, rich soil, Denmark	32	4.61	0.35	-0.47 (25)	4.46	~0	~0	Vesterdal <i>et al.</i> [47]	
Norway spruce, poor soil, Denmark	41	3.76	0.43	0.66 (25)	4.84	23	61	Vesterdal <i>et al.</i> [47]	
Norway spruce, southwest Sweden	92	1.20	0.65	0.09 (25)	2.36	31	14	Vesterdal <i>et al.</i> [47]	
White pine, Rhode Island, USA	115	1.53	0.37	0.15 (70)	2.10	24	29	Hooker and Compton [38]	
Mixed deciduous/coniferous, New Hampshire, USA	65	1.67	0.40	-0.05 (50)	1.95	18	~0	Hamburg [52] in Hooker and Compton [38]	
Loblolly pine, South Carolina, USA	35	4.26	0.95	0.04 (60)	5.24	19	4	Richter <i>et al.</i> [42]	
Oak, Minnesota, USA	39	0.96	0.12	0.13 (25)	1.21	21	52	Johnston <i>et al.</i> [53]	
Loblolly pine, Virginia, USA	47	3.03	0.23	0.20 (100)	3.52	12	47	Schiffman and Johnson [54]	
Red pine, Michigan, USA	38	1.91 ²	0.28	~0 (10)	2.18	13	~0	Pregitzer and Palik [55]	
Larch, NE China	33	2.11	0.34	0.59 (100) ⁴	3.04	31	63	Wang <i>et al.</i> [56]	
White spruce, Quebec, Canada	50	1.86 ²	0.06	-0.44 (30)	1.50	- ⁵	- ⁵	Tremblay <i>et al.</i> [51]	
Former grassland									
Norway spruce, N Italy	62	2.73 ³	0.36	~0 (A horizon)	3.18	13	~0	Thuille <i>et al.</i> [57]	
Norway spruce, Germany and Italy	93-112	2.75 ²	0.34	-0.25-0.08 (50)	2.75	0-15	- ⁵	Thuille and Schulze [50]	

¹Biomass and ecosystem rates apply to 18-year period only.²Aboveground biomass only.³Stem biomass only.⁴Based on status at 33 yr, initial decrease in soil C until age 12 years.⁵The soil compartment was a source for C.

Table 2 Specific wood density of European tree species [62]

Deciduous trees	(kg/m ³)	Coniferous trees	(kg/m ³)
<i>Salix</i>	330	<i>Thuja</i>	400
<i>Populus</i>	410	<i>Abies</i>	410
<i>Tilia</i>	490	<i>Picea</i>	430
<i>Alnus</i>	510	<i>Pseudotsuga</i>	470
<i>Acer</i>	590–640	<i>Pinus</i>	490
<i>Fagus</i>	680	<i>Larix</i>	550
<i>Carpinus</i>	790		

poor mineral soils were reported to store more C, which was attributed to the slow decomposition and complex formation between organic molecules and metal ions [47]. In an assessment of soil C stocks in pure Norway spruce and mixed spruce–broad-leaved stands on poor soils, the C stocks were positively related to soil aluminium pools in an area with relatively poor soils [48], because decomposition of SOM is slow in acidic soils. However, the question of how the C stock of different soil types responds to afforestation is not yet resolved [49].

Tree growth is of major importance for changes in ecosystem C content [9, 32], but the relative contribution of soils remains uncertain. During the last few years, new studies have quantified the total C sequestration and the relative contribution of soil and biomass components in several chronosequences of afforestation (e.g. [47, 50, 51]). Table 1 lists rates of C sequestration in forest floors, mineral soils and biomass and the relative contribution of soils for a selected number of chronosequences in mainly temperate regions.

A contribution of soils of 15–20% of total ecosystem C sequestration appears to be a common picture for afforestation of former cropland (Table 1). However, some chronosequences had lower relative total soil contribution because there were no changes or even a decrease in mineral soil C stock. Carbon sequestration rates in the forest floor are fairly similar among many chronosequences (0.3–0.4 t C/ha/yr). The contribution of the mineral soil to total soil C sequestration varies tremendously among the studied chronosequences and ranges from nil to 80%. The relative contribution of soils and also the partitioning of C between forest floor and mineral soil are probably strongly influenced by factors such as climate, soil type, tree species, former agricultural land use, length of the chronosequence, and sampling methodology. These factors all vary between studies and, with this in mind, the relatively comparable contributions of the total soil to ecosystem C sequestration are remarkable.

The former land use affects the C sequestration potential of afforested sites. Two studies of afforestation chronosequences in former grassland found little change in soil C (Table 1) because of C loss in the mineral soil. Pasture soils already have high C stocks and high root densities in the upper part of the mineral soil, so

afforestation has a small effect [11, 50, 58, 59]. Other chronosequence studies from New Zealand on former pastures and northern England on peatland found that soils initially lost, but later gained C [60, 61]. Thuille and Schulze [50] found that at least 80 years were necessary to restore the C stock levels of the original, historic forest in afforested former pasture soils in the Italian Alps. Croplands are often more depleted in soil C, and therefore have a greater potential for sequestration of soil C following afforestation.

In conclusion, the rate of soil C sequestration is slower than changes in biomass C, and it may take decades until the C pool in former arable mineral soils is replenished. Forest floors accumulate C quickly and often offset temporary C losses in the mineral soil, but most of the forest floor C is in a labile form and may be lost following a change in management.

Influence of Tree Species

In the future the flexibility of forest owners with respect to the choice of tree species will increase. In mountain regions the warming will lead to a potential upward migration of tree species that formerly did not grow in these regions. The effect will be partly balanced by 'losing' certain tree species due to an upward or northward migration. However, not only trees will respond to climate changes, but potential pests will also be affected. The ability to cope with site conditions therefore needs to be considered in the context with possible ecosystem disturbances.

The choice of tree species is relevant for the terrestrial C pool for the following reasons:

- different growth patterns over time (see yield tables);
- specific achievable stand density;
- different rooting depths and rooting patterns;
- different effect on soil C pool;
- specific wood densities;
- different life spans;
- different vulnerability to disturbance.

Broad-leaved trees usually have a higher wood density than coniferous trees (Table 2). The wide variability of the fraction of stem wood to the total biomass is shown in Figure 2.

In countries with a highly developed forestry sector, these factors are taken into account in the management for stem-wood production. The market of C credits may generate new aspects and may favour tree species or species combinations that are currently not considered optimal. European forests are already intensively managed. The present tree species composition already favours high stem production and options for adjustments of stand management for the sake of high C accumulation in the biomass are limited. In a few cases only, man-made

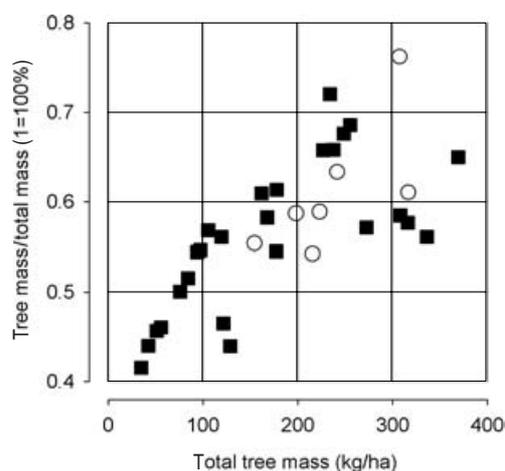


Figure 2 Stem mass fraction in function of total tree mass (t dry mass/ha) measured (root masses are estimates in some cases) in spruce stands. Data from Cannell [63] (rectangles) and Schulze [157] (circles)

forests are not exploiting the full site potential, e.g. Scots pine (*Pinus sylvestris* L.) stands in eastern Germany that have replaced less profitable native broad-leaved forests.

Table 3 shows the C pools in selected European forest types, and the differences are obviously large. The table does not only reflect differences in the stand structure and the specific C dynamics, but also the dominance of the respective species on certain site types. For instance, pine forests often grow on poor sites with shallow and dry soils whereas beech (*Fagus sylvatica* L.) would be found on more fertile soils [65].

The influence of tree species on site conditions has always attracted the attention of forest scientists [66]. Mixed species forest stands are often seen as a remedy for the establishment of stable forests. The central issue is that the benefits of single tree species can be utilized and the production risk of the entire forest can be minimized. Mixed species are superior to single species stands when the individual species exploit different resources at the same site, but can also lead to a competition that reduces the overall productivity of a stand [67, 68]. These effects influence the C sequestration potential of forests.

Figure 3 shows a commonly made comparison between beech and spruce. This species comparison is especially relevant, because spruce is a dominant tree species in central Europe, but is believed to have negative effects on the site quality because of the slow decomposition rate of spruce litter and the species' shallow rooting depth. Spruce is, however, substantially more productive than beech [69]. For C sequestration, both the volume productivity and the weight of the produced wood needs to be taken into account. Figure 3 shows in the upper panel that, at a comparable site, spruce is more productive than beech in terms of stem volume production. When the higher density of beech wood is taken into account, the difference is almost compensated (Figure 3; Tables 2 and

Table 3 Median of C pools in European forests. The data are based on country reports for biomass and soil carbon pools [62]

	Tree C (t/ha)	Soil C (t/ha)	Sum C (t/ha)
<i>Pinus sylvestris</i> (Scots pine)	60	62	122
<i>Picea abies</i> (Norway spruce)	74	140	214
<i>Abies alba</i> (Silver fir)	100	128	228
<i>Fagus sylvatica</i> (beech)	119	147	266
<i>Quercus</i> sp. (oak)	83	102	185

3). Another view is that a conversion from spruce to beech may be justified by ecological arguments, but based on the comparison of biomass C pools, the advantage of beech is not convincing.

It is essential to expand the evaluation beyond the standing tree biomass to the soil C pool and to stand stability. Both factors are of great importance for the long-term sequestration of C in forests. With respect to soil C, beech leads to a deeper distribution of C within the soil profile because it builds roots much deeper in the mineral soil [71]. In contrast, spruce leads to the formation of a thick humus layer on top of the mineral soil. Data that fully account for the soil C pool are sparse, because the soil C pool is often assessed only to a shallow depth, thereby ignoring C deeper in the mineral soil [48]. The disturbance regime of spruce versus beech may also be quite important. Catastrophic storms have repeatedly destroyed large areas in central and northern Europe and spruce forests are especially vulnerable. It is not only the immediate loss of standing biomass but also the subsequent pressure by bark beetles on the remaining stands that calls for effective counter-measures. Among the forest strategies is the conversion of secondary Norway spruce forests to mixed species stands in which beech is often playing an important role [72].

The influence of tree species composition on forest soil development and on soil C stocks has been discussed for more than a century (cf. [73, 74]), indicating that a higher proportion of deciduous trees in stands dominated by coniferous trees exerts a positive influence on the decay of plant litter material and the humus form, but not necessarily on total C stocks [75, 76]. This observation is generally attributed to the fact that deciduous litter is often more easily decomposable than coniferous litter, as the latter often has lower nutrient contents and wider C:N and C:P ratios [28, 77–79]. This increase in soil C sequestration may result in a decrease of the amount of C stored in the forest floor litter layer. However, most findings come from studies on spruce and beech.

Experiments on soil-related influences of forest conversion showed significant effects on soil C dynamics, especially C storage in mineral soil and the depth gradient

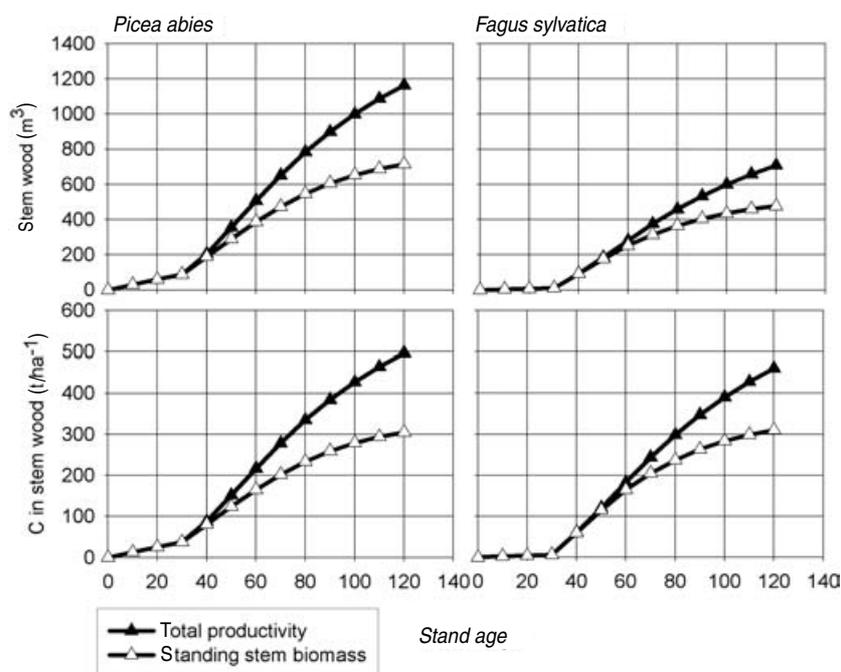


Figure 3. Stem volume (upper panel) and C stocks in the stem wood (lower panel) of beech and spruce, growing at the same site. Data are taken from the Austrian Yield Tables [70]. 'Total productivity' is the sum of the standing and the extracted stem biomass.

in soil C [80, 81]. In most forest regions of central Europe the dominating coniferous forests have been subject to criticism for decades. Forest conversion pursues the goal of establishing more natural, stable, and site-adapted forests. In this context, an important goal is to create forests appropriate for the site and rich in species through ecologically oriented silvicultural measures [82]. In Germany, for example, transforming large Scots pine (*P. sylvestris* L.) forests into mixed stands of Scots pine and European beech by underplanting beech in pure pine stands or finally into pure beech stands is actually in important forestry practice. Several studies have demonstrated that forest conversion has a distinct impact upon soil C storage and general SOM properties [80, 83, 84].

A 180-year chronosequence of typical stages of forest transformation from pure pine stands to pure beech stands of different ages has been studied in the German northeastern lowlands. At the end of the first transformation period, the conversion of tree species composition from pine to beech resulted in a clear shift in humus forms towards humus forms with increased soil faunal and decomposer activity, higher litter turnover rates, increasing nutrient availability and decreasing C:N ratios [80, 81, 85]. Moreover, in mixed pine-beech stands, the depth gradient of soil C was changed and more C accumulated in deeper parts of the mineral soil. This was attributed to beech roots reaching deeper into the mineral soil [86]. It remains to be seen if this C will be shifted into a stable pool. Despite these changes, the total soil C gain after conversion from pine to beech seems to be low within the first decades [85].

Many ideas about influence of tree species on soils have been based on evidence from comparative studies that confounded influences of current vegetation with prior differences in soil conditions, such as parent material or land use. Common garden experiments provide an opportunity to minimize such confounding effects as the same tree species are planted in adjacent blocks so that climate, parent material, time, hydrology and previous land use are almost the same. Common garden experiments are, however, rare [87] and sometimes without replication [88].

Common garden experiments have shown very different forest floor C content among tree species. In Denmark, a 30-year-old common garden experiment of seven species replicated at seven different sites along a soil fertility gradient focused on the forest floor C stock [28]. Lodgepole pine (*Pinus contorta*), Sitka spruce (*Picea sitchensis*) and Norway spruce had much higher C stocks than beech and oak (*Quercus robur*). Other common garden and single-tree studies have indicated that species-induced differences in soil C are clearly seen in forest floors but not in mineral soil [89–91]. However, in a non-replicated common garden experiment, Oostrá *et al.* [90] found differences in the C stock of the upper 20 cm of the mineral soil after 67 years. Elm (*Ulmus glabra*), oak and ash (*Fraxinus excelsior*) had more C in the mineral soil than spruce and beech, whereas the trend for the forest floor was almost reverse. Results from a Danish common garden experiment with Norway spruce and a range of native broad-leaved species replicated at six sites found similar differences in C distribution between spruce and

beech compared to other broad-leaved species (Vesterdal *et al.* in preparation). Such difference in soil C distribution could be due to species' influence on earthworm activity. Earthworms affect organic matter dynamics by incorporating material from the forest floor into deeper soil horizons and by processing the litter [93]. A recent study of 14 tree species in a Polish common garden experiment revealed that maple (*Acer pseudoplatanus*) and lime (*Tilia cordata*) had high and Norway spruce had low earthworm abundances, which was attributed to differences in litter Ca [14]. These recent findings suggest that high forest floor C sequestration in a tree species is not necessarily indicative of high mineral soil C sequestration. In fact, there might be a negative correlation between the size of these two pools. In conclusion, common garden experiments have provided strong evidence for an effect of tree species on forest floor C stocks, but so far only little support for generalizations regarding the influence of tree species on total soil C stocks, including the mineral soil.

Adapted Stand Management

Forestry did develop from the intention to maximize forest productivity and yield. Given the long production cycles of forests, management strategies had to take into consideration issues of forest stability. Disturbances had to be minimized. Many pitfalls and experiences led to a refined framework of recommendations for appropriate forest management. The management history is especially long in Central and Northern Europe and the European forestry tradition was influential in other parts of the world as well. In other regions, forestry sciences either developed later or forestry was never pursued at a high level of intensity. Based on these experiences from management for forest productivity, many conclusions can be derived for the management for C sequestration in forest ecosystems. The specific intention to increase the terrestrial C sink is a novel question for which answers cannot be derived only from textbook knowledge.

The density of forest stands during the life cycle of a forest needs to be actively modified by forest managers in order to reduce competition-induced tree mortality and to avoid natural disturbances such as storm damage and bark beetle infestation. Stand thinning has a long history in practical forest management. In the context of C sequestration, thinning reduces the amount of C sequestered in biomass and dead organic matter for the sake of improved stand stability and stand longevity. Thinned stands contain fewer trees with larger diameters and therefore higher value, in markets that favour larger trees. Thinning not only removes biomass but also stimulates microbial soil processes by exposing the forest floor to solar radiation and more throughfall precipitation. Thinned stands therefore never hold the maximum C density in the biomass, dead organic matter and the soil [44, 95–97]. It can be expected, however, that thinned

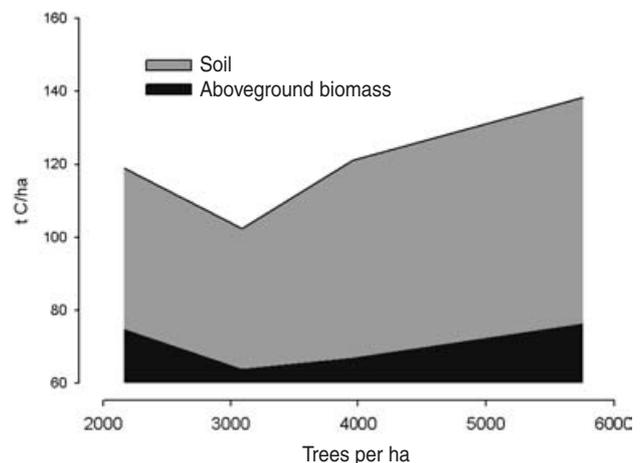


Figure 4. Carbon in the aboveground biomass (stem + branches) and the soil (mineral soil 0–50 cm + forest litter layer) in a Norway spruce thinning experiment eight years after the intervention (data: Hager [98]).

stands are less vulnerable to disturbances and therefore represent more stable C pools than unmanaged forests. Figure 4 shows how thinning reduces the C pool for a young Norway spruce stand. A recent timber market driven trend is the increased production of stems with medium to low diameters instead of larger diameter stems.

The effect of different silvicultural systems in comparison to totally unmanaged beech forests was investigated in Germany. Moderate differences in silvicultural practices have the expected impact on the biomass but only a small effect on the C pool in the forest floor and the mineral soil. These slight effects can hardly be verified, given that the soil spatial variability is large [99].

Extending the Rotation Period

Old forests have a high C density and young stands have a large C sink capacity. Short rotation lengths maximize aboveground biomass production, but not C storage. Support for long rotation periods comes from the observation of very high C stocks in mature forest ecosystems, where the sum of C in the biomass and the soil peaks [64, 100, 101]. A crucial question that calls for a site-specific answer is whether or not old forests are desirable at individual sites.

- Protection forests in high elevations are examples of forests that are purposefully held at a certain level of maturity, where they are considered to optimally meet protection objectives.
- Spruce forests are understood to become increasingly unstable at old age.

Research opportunities on the effect of very long rotation periods on the C pool in forest ecosystems still exist.

Temperate European forests are always actively managed and accordingly the measurable standing biomass is always affected by thinning. A compilation of biomass data from unmanaged temperate forests supports a steady increase of biomass C pools with age [102]. However, specific site conditions might prohibit the prolongation of rotation periods because of increased risks of disturbances with increased age or dimension of trees (e.g. wind-throw on peat soils).

Apart from ecological considerations, the question remains to be answered whether forests fulfil their climate change mitigation potential best when they store a large quantity of C or when they provide wood products that substitute goods produced from non-renewable resources. Moreover, mature forests represent a large, but saturated C pool that has little potential for future additional C sequestration.

The *conservation of forests* can be seen as an active decision to increase the terrestrial C pool. In central and northern Europe the harvest rates are lower than the annual growth rate. Reasons are the fertilization effect of nitrogen deposition and CO₂ and the rising average age of forest stands as a consequence of insufficient incentives to supply the timber market [103, 104]. This implies that the C pool in forests is at present in an aggradation phase and not in equilibrium. It remains to be seen if conservation of forests is indeed suitable as strategy to increase the terrestrial C sinks.

Harvesting is a major disturbance for the soil. In order to ensure minimum soil disturbance, cautious harvesting techniques are recommended. Nevertheless, the pressure to reduce operational costs in forest management makes a highly mechanized forest harvest a necessity that usually increases soil disturbance. Experimental evidence shows that soils are indeed extremely stable parts of the ecosystem and only grave disturbances of the soil lead to relevant C losses [10, 99].

Liming mobilizes nutrients from the organic matter of the forest floor material. That activity has numerous benefits for a forest ecosystem, but leads to C loss instead of C sequestration [105, 106].

Nitrogen Fertilization

Nitrogen fertilization of forests has several aspects: where climatic change increases the productivity of forests (e.g. as a result of prolongation of the growing season) this will lead to a higher N demand. Forests that are currently limited by suboptimal N supply will experience even more accentuated N shortages. Some of the required additional N will be supplied by atmospheric deposition, but deposition is unlikely to cover the N demand at the appropriate rate. Thus, elevated levels of N deposition are harmful to forests, but climate change may create an increased demand for N uptake. Recent publications indicate that N is the main driver

of the increased growth rates in European forests [103, 107].

Whether or not N-enriched soils are a C sink is the subject of ongoing discussions. Scandinavian studies have shown that N-fertilized forests have the capacity to sequester more C than N-deficient forests [108]. The mass loss rate for high-lignin materials may be reduced by high levels of external N [109–111]. Thus, a decreased decomposition rate determines soil C accumulation rates rather than increased litter production [110–113].

Nitrogen stimulates the initial decomposition of fresh litter, but suppresses humus decay in later stages [114]. In a meta-analysis of experiments a significant increase in soil C was found in the upper mineral soil and in the total soil C pool [10]. The effects of N fertilization on the soil C pool vary widely and depend on the interaction with other soil processes. A decrease in the soil C:N ratio is often observed, indicating that the N retention effect of the soils is stronger than the C sequestration [115]. Site-specific responses of soil C sequestration to N fertilization have been observed in the Pacific Northwest. Douglas-fir stands fertilized with nitrogen have generally responded to an extent that fertilization has been recommended as a measure to increase C sequestration [116]. However, the response was very low on sandy soils and strong on loamy soils.

A different perspective is provided by van Miegroet and Jandl [117] who focus on the carbon use efficiency of soil microbes. Adding N increases the N uptake by the stand but also adds N to the soil. The incorporation of N in SOM is a biological process and the overall effect on the C budget depends on the efficiency of microbes. At low efficiencies, microbes respire a lot of CO₂ in order to incorporate the incoming N in labile SOM [118]. In that case the overall effect of N fertilization is a C release from the ecosystem. At a high carbon use efficiency, the effect is C sequestration. From field observations it is known that high dosages of N administered over a short period have a different effect than long-term continuous additions of small N doses [108]. The current knowledge gaps call for a closer cooperation of soil microbiologists with biogeochemists [119].

Nitrogen fertilization stimulates biomass production. However, the subsequent emission of nitrogen oxides can partially offset the effect of C sequestration in the aboveground biomass because the global warming potential of nitrous oxide (N₂O) is 320 times higher than that of CO₂.

Atmospheric N deposition affects N oxide emissions from soils. Especially in N-limited forest ecosystems, the atmospheric N input leads to an increase in N oxide emissions from soils, suggesting that – under increased N-input – such ecosystems can potentially function as strong sources of N trace gases in the future [120]. However, it is expected that N-limited forests gradually adapt to higher N availability. Due to the increase in productivity the release of N oxides is expected to decline.

Water Management – Peatlands

Peatland soils have an organic layer of at least 40 cm depth where C often has accumulated over long time due to excess water and suppressed rates of decomposition [121]. However, the present accumulation rate may be lower than the long-term rate [122]. Zicheng *et al.* [123] showed large cyclic changes in moisture and C accumulation. These soils are characterized by a high water table and an organic layer with a C concentration exceeding a minimum value and by a certain minimum depth, e.g. 40 cm [124]. Peatlands are categorized as bog, fen, swamp, marsh and shallow water [125]. In this synthesis, only the first three categories are considered. Peatlands with anoxic conditions emit methane (CH₄) [126, 127] while N₂O emissions are insignificant [128]. In the Nordic countries, approximately 15 million ha of peatlands are under forest management [129] and some of these have been drained to enable forest establishment and to increase production [130, 131]. Improved aeration as a result of drainage or global warming increases peat oxidation and could turn peatlands from C sinks to C sources [132–138]. On the other hand, CH₄ emissions would decrease after drainage [135–137, 139]. Alternating oxidation and reduction may increase N₂O production significantly [136, 137, 140]. The amount of emitted N₂O has been shown to be related to the peat C:N ratio [141] and forest conditions [142]. The increased decomposition of organic matter following drainage is at least partly compensated by the higher inflow of C into the system through primary production [130, 131, 143, 144]. Leaching of DOC increases immediately after digging the drainage network, but returns later to pre-drainage levels [145, 146]. Direct measurements of C balances in peatlands are rare, but both decreases and increases following drainage have been reported [143, 147–156]. The most recent results from Sweden and Finland [138] suggest that the cumulative emissions of CO₂ and N₂O are of such a magnitude that they can hardly be compensated by increased uptake in forest biomass. In conclusion, forest drainage decreases CH₄ emissions but increases N₂O and CO₂ emissions from peat and C sequestration in the vegetation. Recent results from Sweden and Finland suggest that the drained forest ecosystems on peatland may be net emitters of greenhouse gases.

Reporting C Pools in Forests

The accounting rules of the Kyoto Protocol for greenhouse gas emissions and removals are an instrument of the UNFCCC. Several industrialized countries have supported the inclusion of terrestrial C sinks in order to reduce the burden of reduction of greenhouse-gas emissions and to create incentives to reduce deforestation rates. Through Article 3.3 of the Kyoto Protocol, countries are obliged to account for C pool changes arising

from afforestation, reforestation and deforestation. The inclusion of land-use change in the greenhouse gas accounting under the Kyoto Protocol reflects the fact that since 1750 about 35% of the human-induced carbon emissions into the atmosphere originate from deforestation.

Article 3.4 of the Kyoto Protocol provides countries with the option to elect the reporting of greenhouse-gas emissions and removals resulting from the 'direct human-induced' land management activities. The political willingness to enact Article 3.4 is diverse, because several disadvantages have been identified.

- The C stocks have to be maintained in the future.
- Maintaining large terrestrial C stocks compromises efforts to deliver wood products and decreases incentives for the wood industry.
- The accountable C sink due to forest management is limited and is often small, compared to the overall national fluxes of greenhouse gases.
- The required monitoring efforts are considerable and are not necessarily justified by their impact on the national greenhouse-gas budget.
- Ecosystem disturbances can compromise the expected effect and can lead to large national C debts, virtually overnight.
- Climate change compromises the currently observed C sequestration function of forests – forests may turn into C sources.
- Factoring out the effect of management effects that are not directly human-induced is difficult. The remedy in the currently valid accounting rules was the adoption of a negotiated cap on the maximum accountable C sequestration ascribed to forest management. Opting for Article 3.4 under the uncertainty of future changes in accounting rules may create a substantial risk of unpredictable disadvantages.

The main data source for reporting the C pool and pool changes in terrestrial ecosystems are National Forest Inventories (NFI) that typically have to be augmented with additional information to fully meet the requirements of Kyoto Protocol reporting. Their main objective is the periodic assessment of stem volume rather than the accounting for C pool changes. Emphasis is placed upon tree species composition, stem volume, yield class, forest cover and age structure. Forests with mixed species or a heterogeneous age structure were historically less effectively inventoried, but the design of the inventories has evolved. The sampling design varies with the parameter of interest. The development of international reporting requirements, such as the Montreal Process Criteria and Indicators of Sustainable Forest Management and Biodiversity, lead to the inclusion of additional parameters.

The obtained estimates of stem wood volumes need to be transformed to estimates of pool sizes of the total aboveground and the belowground tree biomass. Further

Table 4 Summary of the effects of specific forest management actions on ecosystem C stocks ('+', increases C stock; '-', decreases C stock; '+/-', neutral with respect to C stock) and unintended side effects of the management strategy

Afforestation	
+	Accumulation of aboveground biomass
+	Formation of a C-rich forest floor and slow buildup of the C pool in the mineral soil
+/-	Stand stability depends on the mixture of tree species
-	Monotone landscape, in the case of large even-aged mono-species plantations. Afforestation can also provide variability to a landscape.
Tree species	
+	Affects stand stability and resilience against disturbances, effect applies for entire rotation period, positive sideeffect on landscape diversity when mixed-species stands are established
+/-	Effect on C storage in stable soil pools controversial and so far insufficiently proven
Stand management	
+	Long rotation period ensures less disturbance due to harvesting
+	Many forest operations aim at increased stand stability
+	Every measure that increases ecosystem stability against disturbance
+/-	Different impact, depending on whether or not harvest residues are left on-site
-	Forests are already C-rich ecosystems – small increase in C possible, thinning increases stand stability at the expense of the C pool size, harvesting invariably exports C
Disturbance	
+	Effects such as pest infestation and fire can be controlled to a certain extent
+/-	Low-intensity fires limit the risk of catastrophic events
-	Catastrophic (singular) events cannot be controlled, probability of disturbance can rise under changed climatic conditions, when stands are poorly adapted
Site improvement	
+	N fertilization affects aboveground biomass, effect on soil C depends on interaction of litter production by trees and carbon use efficiency of soil microbes
+/-	Drainage of peatland enables the establishment of forests (increased C storage in the biomass) and decreases CH ₄ emissions from soil, but is linked to the increased release of CO ₂ and N ₂ O from the soil
-	Liming and site preparation always stimulate soil microbial activity. The intended effect of activating the nutrient cycle is adverse to C sequestration, N fertilization leads to emission of potent greenhouse gases from soils

key C pools such as dead wood on the ground and soil C require separate monitoring schemes and can usually not be derived from stem volume data.

With respect to monitoring for the requirements of Article 3.3 of the Kyoto Protocol, some regional aspects of afforestation projects require attention. In central Europe, most afforestations occur on a small scale. These projects are not part of a major national or international policy, but are instead the decision of the owners of small tracts of land, who may choose to afforest their marginal agricultural fields. For the national reporting requirements, the monitoring requirements are expensive, because a large number of sites each contributing only a small amount to the greenhouse-gas budget, need to be monitored. Another example is the land-use change in high elevation ecosystems when pastures are increasingly abandoned and overgrown with trees. In order to monitor when and where these former agricultural lands meet the definition of forests, substantial efforts in land-based assessments are required.

Conclusions

A compilation of forest management activities indicates that few practices are clearly good or bad with respect to C sequestration (Table 4). The verdict on their impact

depends on their effect on soil C and the degree of stability against disintegration of the stand structure. Optimized forest management with regard to soil C sequestration should aim to secure a high productivity of the forest on the input side, and avoid soil disturbances as much as possible on the output side.

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