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IMPACT OF SILVICULTURAL PRACTICES ON SOIL CARBON: A REVIEW

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The paper provides a review of Russian and foreign articles that study the impact of silvicultural practices on the soil carbon pool to assess the effectiveness of forest carbon projects. Analyzing the works allowed us to conclude that silvicultural practices affect the content of soil carbon through a change in the rate of influx and decomposition of organic matter and, as a result, the redistribution of carbon in the soil profile. High-intensity felling, including clear felling, removal of logging residues, damage to the ground cover when planting forest crops, and development of monocultures may have a negative impact on the soil carbon pool. On the contrary, selective and low-intensity thinning, leaving logging residues, and planting mixed forest stands, especially on abandoned agricultural lands, proved to be promising forest management practices that contribute to the accumulation and conservation of soil carbon.

Keywords: *carbon, soil, forest carbon projects, silvicultural practices*

In 2019, the Russian Federation ratified the Paris Agreement, which, in turn, replaced the Kyoto Protocol. Currently, the Agreement is the main regulatory tool to reduce the content of carbon dioxide (CO_2) in the atmosphere. It recognizes the forest as one of the most important sinks and reservoirs of greenhouse gases, which requires special focus and protection by the member states (Paris Agreement, 2015). To achieve the goals under the Paris Agreement, the Strategy of socio-economic development of Russia with a low level of greenhouse gas emissions until 2050 was approved in 2021. The Strategy claims to aim at achieving carbon neutrality along with sustainable economic growth. This goal is planned to be implemented, inter alia, through an increase in forest absorptive capacity that may result from improving the practices of administration, forest management, reforestation, and afforestation. In this regard, forest carbon projects aimed at strengthening the carbon depositing function of forest ecosystems are becoming very popular. Moreover, reforestation and other types of biological carbon sequestration are among the most cost-efficient approaches to reducing greenhouse gas emissions (Gillingham, Stock, 2018).

Forest carbon projects constitute a set of practices that may reduce (prevent) greenhouse gas emissions or increase their absorption, taking into account the absorptive capacity in forests and other ecosystems (Zakonoproekt N 1116605-7). According to KPMG International Limited and considering the data of the Institute of Global Climate and Ecology, the potential effect of forest carbon projects to reduce greenhouse gas emissions in Russia is up to 40–45% among all other areas of greenhouse gas sequestration (360–420 million tCO2e/year^{-1}) (Special'nye lesoklimaticheskie proekty ..., 2021).

The idea that forests reduce CO_2 concentration can be supported by the argument that living trees absorb CO_2 from the atmosphere, forming organic compounds during photosynthesis and releasing oxygen into the environment. Forests store carbon in vegetation and soils for a long time, which gives them an advantage over agrocenoses that exhibit a rapid return of carbon to biological cycles. Each ton of carbon stored in wood corresponds to the removal of 3.67 tCO_2 from the atmosphere (Oliver et al., 2014).

It has been shown that tree growth decreases with age, and the forest deadwood releases CO_2 while decomposing, which is emitted back into the atmosphere (Oneil, Lippke, 2010). Therefore, it was found that CO_2 absorption rate slows down in aging forests. For example, Western Australia's forest inventory data show that tree growth rates have begun to slow down by the fifth decade with little subsequent growth by the ninth decade, as the mortality of some trees is offset by the growth of others (FIA, 2020; Lippke et al., 2020). However, provided that carbon is preserved in long-lived wood products, reasonable felling modes may contribute to continuous carbon uptake. Such modes include felling in a period when the stands are close to their maximum growth rate and the felling volume does not exceed the annual growth value. A typical example of sustainably managed forests in Western Australia shows that, after 23 years, each hectare of trees planted in 2000 has absorbed approximately 45 tons of carbon from the atmosphere, and 165 tons of CO_2 have been removed from the atmosphere by the middle stage of the first felling cycle. By the end of the first cycle at age 45, 180 tons of carbon have been absorbed in the wood and 660 tons of CO_2 have been removed from the atmosphere as a result of tree growth. Of 180 tons of carbon, 75 tons were stored in long-lived wood products and related products, 30 tons in short-lived paper products, 60 tons in logging residues, and 15 tons in biofuels.

As a result, total emissions from fossil fuel combustion from all sources decreased from 20.4 to 5.4 tons of carbon (Lippke et al., 2021).

It is important to understand the ways in which wood can be used to assess carbon transformation during forest felling. In the production of long-lived wood materials, carbon does not return to the atmosphere; this results in no CO_2 emissions. Materials with a short life cycle, for example, biofuels, wood chips, pulp and paper products, contribute to rapid CO_2 emissions. According to B. G. Fedorov (2017), forest felling should be considered as not CO_2 emissions but rejuvenation of the felling area, an increase in net ecosystem products (NEP), and the oxidation of wood and wood products should be considered as they are used by consumers.

The climate regulating role of forests is not limited to carbon deposition in wood. Total soil carbon stocks (including forest floor) account for approximately 70% of ecosystem carbon stocks in boreal forests, approximately 60% in temperate forests, and approximately 30% in tropical forests, respectively (Pan et al., 2011). In old-growth forests, this value reaches 90% (Johnson et al., 2010). In total for Russia, soil carbon reserves in the layers 0–30 cm, 0–50 cm, and 0–100 cm deep are estimated at 128.4×10^9 tons, 166.5×10^9 tons, and 215.8×10^9 tons.

For the layer 0–100 cm deep, average reserves are 162 ± 23 $tC \cdot ha^{-1}$ (Chestnyh et al., 2022). On the one hand, land management activities aimed at obtaining raw wood materials to replace carbon-intensive production may increase carbon losses by damaging soils, lowering the strength of aggregates, increasing wildfires, and reducing the vegetation cover. On the other hand, carbon uptake increases if certain forest management practices are used that add more biomass to the soil, reduce soil damage, conserve moisture, improve soil structure, and increase soil biota activity. Despite the size of soil carbon stocks, the role of soil carbon has often been ignored or underestimated in many climate change initiatives in the past.

Increased demand for high-quality environmental carbon credits may outpace their supply, as evidenced by a 60% increase in the average price of carbon offsetting associated with natural solutions in 2021 compared to the previous year, reaching a global average price of USD 4 per 1 ton (Donofrio et al., 2022). A bottleneck in carbon project implementation is the uncertainties related to the response of soils to project activities, which can be crucial for supporting and justifying investment decisions to both protect existing carbon stocks (e. g., by conserving intact forests) and develop new carbon stocks (e. g., by reforestation). If the objective of forest management is to significantly reduce the impact of climate change, more in-depth research is required on the effects of various types of silvicultural practices on soil carbon levels, as well as recommendations for the most efficient practices in forest carbon projects.

The purpose of this paper is to consider how various silvicultural practices affect the soil carbon pool from the point of view of their effectiveness in forest carbon projects.

1. FUNDAMENTAL REQUIREMENTS TO FOREST CARBON PROJECTS

To assess the efficiency of carbon sequestration practices, it is necessary to analyze the requirements to forest carbon projects. Complementarity, sustainability, and no-leakage policy are the most important requirements to forest carbon projects (The Greenhouse Gas Protocol ..., 2006; Verified Carbon Standard, 2022, Shanin et al., 2022).

The principle of complementarity assumes that a project would result in a greater greenhouse gas absorption than if it were not implemented. To assess this "project complementarity," we need to develop a baseline (a scenario of the usual development of events, or a reference scenario) which would most often contain compulsory requirements of the existing laws, and the project scenario. Emission reduction units resulting from the project should be calculated as the difference between the project outcome and the baseline. Units that do not reflect actual emission reduction are called false. They may occur, for example, if the project does not take into account the baseline or defines it incorrectly. Complementarity is key to ensuring that false carbon units do not jeopardize global carbon markets (Michaelowa et al., 2019). If countries or any other entity use false units to meet their emission offsetting obligations, this could result in an overall increase in emissions rather than a decrease. Thus, complementarity is a guarantee of environmental integrity.

The principle of sustainability requires that the results of a project to remove greenhouse gases last for a long time (at least 100 years). To demonstrate sustainability, it is necessary to conduct third-party monitoring of project outcome, at a frequency that depends on the project type (every 5 years, every year).

Leakage in forest carbon projects means a decrease or increase in greenhouse gas sequestration outside its geographical boundaries, which is directly or indirectly related to project implementation (Atmadja, Verchot, 2012; Streck, 2021). Leakages can be either positive or negative. In the first case, they intensify the reduction of greenhouse gas emissions thanks to the positive effects of displacement. In mountainous regions, for example, afforestation in the project area contributed to reducing erosion in the adjacent downhill area, resulting in a greater deposition of soil carbon. While the effects of positive leakages can be substantial, they are understandably not the subject of any debate. In the second case, leakages reduce emissions sequestration, making a project ineffective with emissions moving elsewhere with no decrease. A negative leakage sends the load from one area to another. Among other things, it can be caused by movement of people, technologies, or capital. E. g., a ban on felling in a certain area may lead to increased felling in another area. Depending on whether the events that result in the movement effects are direct or indirect, we can distinguish between a primary or a secondary leakage (Aukland et al., 2003). For a forest carbon project to be considered effective, it is necessary to fulfil all the requirements at the same time.

2. EFFECT OF SILVICULTURAL PRACTICES ON THE FOREST SOIL CARBON POOL

In forest ecosystems, aspects of management (felling, removal of logging residues, thinning, reforestation, fertilization, forest conservation and protection, etc.) may impact soil carbon storage. Such practices affect soil carbon stocks by altering the rate of influx and decomposition of organic matter. Below is a brief overview of the effect of major silvicultural practices, which may be part of forest carbon projects, on the soil carbon pool.

2.1. Forest felling

Forest felling practices are one of the main anthropogenic factors that change forest soils (Dymov, 2017). Clear felling is the most common felling practice worldwide, which has a generally negative impact on soil carbon stocks. Soil carbon losses after clear felling can be associated mainly with a decrease in carbon influx (i. e. litter) and/or an increase in the decomposition rate, and, as a result, the renewal type of ground cover and tree layer. It has been suggested that higher insolation and warmer, more favorable microclimatic soil conditions may stimulate microbial respiration after tree canopy removal (Pumpanen et al., 2004; Kulmala et al., 2014; Mayer et al., 2017). Soils in felling areas are warmer than soils in coniferous forests, but soils in felling areas show higher rates in daily amplitude of temperature fluctuations (Dymov, Startsev, 2016). However, it has also been demonstrated that enzymatic processes involved in the breakdown of organic matter and forest floor decomposition may remain unchanged or decrease after clear felling practices (Cardenas et al., 2015; Kohout et al., 2018). Lower enzymatic activity is associated with a decrease in root litter, whose bulk is thin roots, and with changes in the soil microbial community (Kohout et al., 2018). According to a meta-analysis (Holden, Treseder, 2013), clear felling reduces microbial and fungal biomass by 14–33% and 20–40%, respectively, with smaller impact than wildfire but greater impact than insect outbreaks. In addition, a decrease in moisture in surface layers of the soil reduces the forest floor decomposition rate (Prescott et al., 2000). On the other hand, due to a decrease in transpiration resulting from forest stand felling, the moisture content increases in the lower layers of the soil. It has been shown that felling practices in the middle and northern taiga forests of the Komi Republic result in a change in the ratio of surface and underground runoff, as well as the modes of constant watercourses and yearly runoff redistribution (Transformacija ..., 1996). Accelerated erosion, leaching and avalanches also may contribute to a decrease in soil carbon stocks after felling (Katzensteiner, 2003). Displaced carbon is deposited elsewhere (Hoffmann et al., 2013), which contradicts the no-leakage policy. Intensity of manifestation in individual elementary soil-forming processes depends primarily on the forest type, ground cover and tree layer plants, type and granulometric composition of soils, and climatic conditions in a given area (Dymov, 2017).

Clear felling reduces soil carbon storage by up to 10% across the entire soil profile with the greatest losses in the forest floor (Johnson, Curtis, 2001; Achat et al., 2015). A meta-analysis (Nave et al., 2010) of studies exploring temperate forests showed that felling reduced soil carbon stocks by an average of 8%: forest floor carbon stocks decreased by 30%, while mineral soil horizons showed no significant overall changes. Carbon losses in the forest floor were higher in deciduous forests (–36%) than in coniferous or mixed forests (–20%). According to another meta-analysis (James, Harrison, 2016), forest harvesting reduces soil carbon stocks by an average of 11.2% with the greatest losses occurring in organic soil horizons (–30.2%). Losses are smaller in the upper mineral layer of the soil (0–15 cm deep; –3.3%) and deeper layers of the soil (60–100 cm; –17.7%). Timeline studies and meta-analysis show that soil carbon stocks in the forest floor and mineral soil begin to recover 10 to 50 years after felling (Tang et al., 2009; Nave et al., 2010; Achat et al., 2015; James, Harrison, 2016).

Selective thinning, which preserves living trees, may reduce soil carbon losses associated with forest harvesting. Strukelj et al. (Strukelj et al., 2015) showed that, under the conditions of boreal forests in Canada, 9 years after clear felling of aspen trees, the area became a carbon source, whereas selectively thinned areas became a carbon sink. In areas with predominant European spruce (*Picea abies*) in Austria, selective thinning resulted in an 11% increase in soil carbon reserves in the upper layer of the mineral soil compared to the traditional clear felling when trees reached the age of ripeness (Pötzelsberger, Hasenauer, 2015). Selective thinning in lenga beech (*Nothofagus pumilio*) forests in Chilean Patagonia resulted in only short-term soil carbon losses (Klein et al., 2008). Reduced soil carbon stocks in mineral soils and neutral effects on the forest floor were reported after harvesting in an oak forest in New England (Warren, Ashton, 2014). However, other researchers found little to no difference between the effects of selective thinning and clear felling on soil carbon stocks (Hoover, 2011; Christophel et al., 2015; Puhlick et al., 2016). Still, it should be noted that change in the soil carbon pool after felling may occur over decades and even centuries (Achat et al., 2015; James, Harrison, 2016), but most studies explore the change in soil carbon within 15 years after felling. To fill this gap, it is necessary to create and maintain long-term research sites in various types of forest ecosystems (Clarke et al., 2015). Moreover, it is possible to fill the gaps in long-term estimates of soil carbon stocks using mathematical modeling methods.

The Romul_Hum model (Komarov et al., 2017, Komarov et al., 2017; Chertov et al., 2017a, 2017b) describes the transformation (mineralization and humification) of soil organic matter depending on its chemical properties and soil weather conditions. The corresponding simulation results are presented in a range of studies (Kalinina et al., 2018; Shanin et al., 2021; Shanin et al., 2022). At the moment, most simulation experiments using the model have been conducted for objects in the boreal zone, but it can be parameterized for coniferous/broadleaved forests too. Thanks to the Romul_Hum model being integrated with the soil climate statistical simulator (SCLISS) (Byhovec, Komarov, 2002), it is possible to use simulations under various climate change scenarios. Also, both of the above models, along with the dynamic forest stand model FORRUS-S (Chumachenko, 1993; Chumachenko et al., 2003), BioCalc model (Khanina et al., 2006; Khanina et al., 2007, 2014) used to predict the dynamics of the species diversity in the living soil cover of forest areas, forest area evaporation model EVAPROF (Karpechko, 2016; Kondrat'ev et al., 2019), hydrological model for runoff formation ILHM (Kondrat'ev, Shmakova, 2005; Kondrat'ev, 2007), and nutrient load formation model for water bodies ILLM (Kondrat'ev, 2007; Kondrat'ev et al., 2011; Behrendt, Dannowski, 2007) are combined into the unified model system RUFOSS (Chumachenko et al., 2020) to assess compromise and synergy of forest ecosystem services.

2.2 Removal of logging residues

Logging residues are a mix of tree components with low commodity value that remain in the felling area after the felling process. They consist of leaves/needles, twigs, poor-quality trunks or trunks of smaller diameter, bark, dry stands, stumps, and roots. Logging residues contain carbon and may impact carbon accumulation/loss indirectly, for example, by affecting microbial communities (Mushinski et al., 2019) and soil microclimate (Devine, Harrington, 2007). Modern demand for renewable energy sources (such as fuelwood) increased the interest in using logging residues. Biomass is currently the largest renewable energy source, and most IPCC pathways that limit global warming to 1.5 °C include greater use of biomass for energy production (De Coninck et al., 2018). Removal of logging residues from the area affects soil carbon stocks, so actual carbon balance when replacing fossil fuels with logging residues for energy production should consider these possible changes.

Literature sources offer ambiguous conclusions regarding how removal of logging residues may affect soil carbon stocks, most often reporting a decrease. For example, it was found that removal of logging residues leads to a decrease (–6%) in soil carbon stocks, while conservation of logging residues increases soil carbon stocks (Johnson, Curtis, 2001). A meta-analysis (Achat et al., 2015) shows that removal of logging residues may result in significant losses of the soil carbon pool in the forest floor (10–45%) and even in soil layers deeper than 20 cm (10%), and that it may have a greater impact in temperate forests than in cold (boreal) forests. Carbon stocks in the forest floor also decrease as harvesting intensity increases (–24% with removal of logging residues). Thus, removal of logging residues results in an average loss of 11% of the carbon in the entire soil profile.

It is also often reported that removal of logging residues has no effect at all. Reviews

(Johnson, Curtis, 2001; Thiffault et al., 2011; Clarke et al., 2015) found no clear evidence of an overall reduction in soil carbon content following removal of logging residues. In addition, fourteen pilot sites that are part of the long-term soil productivity monitoring network in Canada (Morris et al., 2019) and Europe (Walmsley et al., 2009) showed no clear impact of removal of logging residues on soil carbon stocks within 20 years after felling.

The use of stumps is widespread in Scandinavian countries. Field studies of stump harvesting in these countries revealed either no decrease in soil carbon stocks in mineral soils (Strömgren et al., 2013; Jurevics et al., 2016) or only a slight decrease (Hyvönen et al., 2016; Vanguelova et al., 2017). However, removal of stumps resulted in a 24% reduction in soil carbon stocks in a temperate forest in Washington, D.C., US (Zabowski et al., 2008). In central Sweden, areas where stumps had been harvested were found to have significantly lower soil carbon content in the humus layer compared to areas where scarification had been performed (Persson et al., 2017). The impact of stump removal on the carbon pool is higher for organic soils than for mineral soils. For example, the peatlands of Wales with a high content of organic matter lost up to 50% of the total carbon in layers 0–80 cm deep within the first four years after stump harvesting (Vanguelova et al., 2017).

Felling is characterized by a volley flow of residual wood in the form of logging residues and large wood residues, including stumps, which persist for a sufficiently long time and form the soil organic matter. It is shown that, under middle taiga conditions in the Republic of Komi, there is little to no downed deadwood in 40-year-old secondary birch stands (Dymov, 2017).

2.3. Planting of forest stands

As of today, both reforestation and afforestation, including planting of forest stands, serve as generally accepted approaches to carbon sequestration. The analysis of how forest planting affects soil carbon shows very diverse results, since soils may accumulate carbon, stay unchanged, or even lose carbon after afforestation (Guo, Gifford, 2002; Vesterdal et al., 2002). However, most reviews present initial carbon losses followed by a slight increase. Carbon losses may occur within a short period of time after afforestation, when there is a lack of balance between carbon losses from soil microbial respiration and carbon influx from litter. Planting results in soil disturbance and can stimulate the mineralization of soil organic matter. Several studies comparing different site preparation methods have shown that soil carbon losses increase as soil disturbance intensity increases (Schmidt et al., 1996; Mallik, Hu, 1997). Sandy soils are particularly sensitive to disturbance (Carlyle, 1993). These losses are not necessarily offset by the influx of litter carbon in young planted forest stands due to its low volume. Experimental evidence supports this theory. The increased carbon influx in the upper mineral soil of planted forests can be compensated for by losses of old carbon from deeper parts of the soil (Paul et al., 2002; Vesterdal et al., 2002). Experiments in South Carolina with frankincense pine (*Pinus taeda*) showed that 80% of the carbon accumulation occurred in biomass, while some accumulation was found in the forest floor and only a small amount was accumulated in mineral soil (Richter et al., 1999). Carbon accumulation initially occurs in the forest floor. Conditions that are not conducive to microbial processes in the soil, such as sandy soils, unavailable nutrients and low pH, may result in the formation of a powerful layer of the forest floor (Vesterdal et al., 1995; Vesterdal, Raulund-Rasmussen, 1998). Its power and chemical properties may also vary depending on the species (Vesterdal, Raulund-Rasmussen, 1998).

Of particular importance is the selection of tree species for reforestation. According to a meta-analysis (Laganiere et al., 2010), the average increase in soil carbon stocks 20-30 years after afforestation by broadleaf species was 25%. For comparison, the increase over the same period is 2% for coniferous stands. Soil carbon stocks in the forest floor are usually larger under coniferous trees than under broadleaf trees (Vesterdal et al., 2013; Boča et al., 2014), while in mineral soil layers, it is the other way around (Vesterdal et al., 2013). The results of mathematical modeling (Shanin et al., 2022) showed that development of Scots pine (*Pinus sylvestris*) monocultures was less efficient compared to natural overgrowth of small-leaved stands, which contradicts the principle of complementarity. The greatest efficiency in carbon storage is predicted for mixed cultures of Scots pine with an admixture of 2-3 small-leaved species like birch and aspen (*Betula* spp. and *Populus tremula*).

Development of mixed stands instead of monocultures can contribute to higher soil carbon stocks thanks to the complementarity of above-ground and underground niches (Pretzsch, 2014). Firstly, mixed stands show higher biomass production and, therefore, litter influx than monoculture stands (Resh et al., 2002); secondly, their roots use soil more efficiently, allowing for an increased influx of root litter (Finér et al., 2017).

At national temperate and boreal forest inventory sites throughout Sweden, a consistent positive association between species diversity and soil carbon stocks was found (Gamfeldt et al., 2013). The same patterns were revealed in subtropical forests of China (Li et al., 2019). The study (Akkumuljacija ugleroda ..., 2018) describes the example of forests in Moscow Oblast, Bryansk Oblast, and the North-Western Caucasus with a close linear positive relationship between carbon stocks in the mineral soil horizons and species diversity, reflecting the increase in diversity (different ratios between nutrients and secondary metabolites) of litter produced by plants of different species. For the same objects, it was later shown that mixed litter is a predictor of high activity of earthworms belonging to different functional groups and, as a consequence, higher soil carbon stocks in coniferous/deciduous forests (Kuznetsova et al., 2021). The study (Kuznetsova et al., 2020) shows that an increase in the proportion of undergrowth of deciduous trees and grasses that produce high-quality litter contributes to an increase in the rate of litter decomposition, a decrease in forest floor reserves, and an increase in carbon reserves in the mineral profile. In Central Europe, mixed stands of beech and spruce are the best option, even if monoculture spruce stands have a higher growth rate (Pretzsch, 2005). This may be due to the contribution of different woody plant species to the intra-profile soil carbon distribution. The paper by Fischer et al. (Fischer et al., 2002) showed that, when beech trees are planted among pine stands, more carbon accumulates in the deeper mineral soil layers, since beech roots penetrate deeper into the soil than pine roots. At the same time, planting of spruce forests to replace beech forests results in release of carbon from the mineral soil horizons which are no longer permeated with roots (Kreutzer et al., 1986). It has also been shown that a small addition of rapidly degradable deciduous tree litter results in transfer of carbon to mineral soil horizons (Cotrufo et al., 2013; Córdova et al., 2018) due to the increased influx of dissolved organic matter from the forest floor (Fröberg et al., 2011). Researchers are currently developing a functional forest classification to assess the efficiency of their carbon cycle regulation function, which is based on ecosystem processes of litter decomposition (Lukina et al., 2021).

Species composition affects soil carbon stability. Studies in Germany show that coniferous forests store about 35% of the total amount of soil carbon in a labile organic layer, which is often subject to anthropogenic impact, forest wildfires, and temperature change (Wiesmeier et al., 2013). Mixed forests contain more carbon in the mineral strata, so carbon is less susceptible to temperature changes. The stability of soil carbon in mineral soil is higher in mixed forests than in coniferous forests, in particular due to the symbiotic relationship of deciduous species and arbuscular mycorrhiza (Craig et al., 2018; Keller et al., 2021).

Differences in carbon stocks in soils of monoculture and mixed forests cab also be explained by the influence of woody plants on the supply of dissolved organic carbon and its removal with soil waters. It is believed that, because of the high leaf surface index, coniferous stands contribute to a greater degree than deciduous ones to the interception and transpiration of precipitation (Achat et al., 2015), which affects the volume of soil carbon removal and its redistribution in the profile. However, the study by A. I. Kuznetsova et al. (2022) shows that in old-growth polydominant coniferous/deciduous forests, carbon influx with precipitation is less than in young pine trees of shrub-green moss and complex pine trees. This is due to the denser canopy of the stand, which intercepts precipitation more effectively, and to the water being less carbon-enriched during the passage of precipitation through the canopy. Therefore, carbon removal from organogenic soil horizons in older polydominant deciduous forests is on average 4.8 times lower than in relatively young pine forests.

However, research in the US shows that reforestation, which is currently carried out on more than 500 million hectares of land, increases carbon stocks in the upper soil layer in the long term, and that reforested lands will absorb cumulatively 1.3–2.110^9 tons of carbon over a century (13–21 × 106 tC·ha^{-1}). Each year, these carbon gains account for 10% of carbon uptake in the US forest sector (Nave et al., 2018). From 1950 to 2012, planted forests in China occupying an area of 79.5 million hectares sequestered 1.686 × 10^9 tons of carbon. The carbon stock on modern Chinese planted forests is 7.894 × 10^9 tons, including 21.4% of the total uptake as forest biomass and 78.6% as soil organic matter (Huang et al., 2012).

According to a number of experts, including those from the Scientific Council of the Russian Academy of Forest Sciences (Rezoljucija ..., 2021), one of the most promising types of forest carbon projects for Russia is afforestation, including forest planting, on abandoned agricultural lands. According to R. M. Ritter and L. Ritter (2020), planted forests of five tree species (hybrids of aspen, poplar, hanging birch, European spruce, and larch) on the former agricultural lands of Sweden, showed the carbon sequestration rate after almost a decade ranging from 0 to 2.3–4.9 tC·ha^{-1}/year^{-1}, whereas the average rates of soil carbon sequestration ranged from –3.0 to 0.78 tC·ha^{-1}/year^{-1} during the first 8–9 years. According to a field experiment (Rytter, 2012) in southern and central Sweden, the accumulation of carbon in woody biomass by planted forest stands of poplar and willow on arable land is estimated at 76.6–80.1 tC·ha^{-1}, and the soil carbon accumulation is estimated at 9.0–10.3 tC·ha^{-1} for the first 20–22 years of cultivation. The average carbon sequestration rates were 3.5–4.0 tC·ha^{-1}/year^{-1} in woody biomass and 0.4–0.5 tC·ha^{-1}/year^{-1} in soil. According to other data in Iowa, US, when growing willow and poplar for biofuel, the rate of carbon accumulation in biomass is 3.4 and 4.3 tC·ha^{-1}/year^{-1}, in soil — 0.9 and 1.9, respectively (Lemus, Lal, 2005). It also was reported that soil carbon stocks under 24-year-old planted willow and poplar stands are 1.5 times greater than in treeless areas (Georgiadis et al., 2017). The results of model assessment (Priputina et al., 2016) for planted stands of fast-growing forms of aspen in the Republic of Mari El showed that, over 30 years, soil carbon reserves vary from –06.6 to 19.5 tC·ha^{-1}/year^{-1}, depending on the planting scheme. It should be noted that most published field estimates of the annual net change in soil carbon stocks on planted forest energy stands exceed the minimum requirements (0.25 tC·ha^{-1}/year^{-1}, Volk et al., 2004) for recognizing such stands from fast-growing species of woody plants as carbon neutral.

2.4 Thinning

The mechanism of thinning impact on soil carbon is the same as that of selective thinning. Partial removal of the canopy is expected to reduce soil carbon stocks due to reduced litter influx and/or increased decomposition rate due to increased soil temperature and moisture, especially in the early years after thinning, when the canopy is open. Thinning has been shown to increase soil temperature by 8.7% and soil respiration by 29.4%, thereby reducing forest floor stocks by 23.7% (Zhang et al., 2018). Thinning may result in a significant decrease in large tree residues (Achat et al., 2015), as longevity of leaves and tree branches increases due to the increased distance between trees. However, thinning reduces competition between trees, increasing the carbon stock gain in tree stand biomass. Experimental data (Lee et al., 2023) showed that net carbon uptake (difference between the yearly increase in the wood carbon pool and carbon emissions due to microbial respiration) in planted stands of blunt cypress (Chamaecyparis obtusa (Siebold and Zucc.) Endl.) increased after 30% stand removal compared to non-thinned control objects.

However, most studies have not reported any significant effects of thinning on soil carbon stocks in mineral soil horizons (Noormets et al., 2015; Zhou et al., 2013; Kim et al., 2016), although some researchers report carbon losses (e. g., Mushinski et al., 2019) even in deeper soil horizons (up to 1 m) (Gross et al., 2018). It is apparent that the degree of change in carbon stocks depends on felling intensity. Reserves of organic carbon in the forest floor may be reduced in case of high-intensity felling, for example, with an up to 50% decrease in a cross-sectional area compared to the control objects (Vesterdal et al., 1995; Achat et al., 2015; Bravo-Oviedo et al., 2015). A meta-analysis (Zhang et al., 2018) showed that removal of less than 33% of the stem wood stock contributes to a 17.2% increase in soil carbon stocks; removal of 33–65% of the stem wood stock does not affect soil carbon stocks; removal of more than 65% of stem wood stock reduces soil carbon stocks by 7.6%. At the same time, total soil carbon content increases by 29.5% during the first two years after harvesting, regardless of intensity. The authors (Zhang et al., 2018) attribute this fact to the influx of a large amount of organic matter after thinning. In addition, increased soil temperature and solar radiation contribute to the development of undergrowth and remaining trees. In addition, Pang et al. (2016) found that soil temperature and moisture had a positive effect on the growth of smaller roots after thinning.

Long-term field experiments are required to study soil carbon reserves at different thinning intensity (Zhang et al., 2018). Such experiments would be necessary to link soil carbon stocks to thinning intensity and to define thresholds for the number of trunks to be removed. In addition, it is necessary to take into account abiotic (climatic conditions, soil-forming rocks, terrain) and biotic (vegetation, soil biota) factors of functioning in existing and new forest ecosystems.

CONCLUSION

Forest carbon projects are an effective way to achieve the goals of the Paris Agreement, an integral part of such projects being the aim to increase the productivity of existing forest ecosystems, in particular by means of silvicultural practices. However, when assessing the effectiveness, changes in the soil carbon pool that may affect the compliance with forest carbon project requirements, namely complementarity, sustainability and no-leakage policy, are often ignored.

Analysis of the current state of the issue suggests that silvicultural practices may impact soil carbon content through changes in the organic matter influx and decomposition rate and, as a result, affect the redistribution of carbon in the soil profile. Forest felling is the leading factor in changing the soil carbon pool. Most often, studies report a decrease in soil carbon stocks after clear felling and an increase in soil carbon stocks after selective thinning and thinning of low and moderate intensity. Another argument in favor of moderate-intensity thinning is the maintenance of structural diversity in communities with the creation of mosaic forest stands and age-heterogeneous forests. Forests with high biodiversity store carbon more efficiently. Removal of logging residues reduces soil carbon stocks by decreasing organic residue influx as well as impacting microbial communities and soil microclimate. Most forest planting surveys present initial carbon losses followed by a slight increase. Carbon losses may occur within a short period of time after afforestation, when there is a lack of balance between carbon losses from soil microbial respiration and carbon influx from litter. Planting mixed forest stands instead of developing monocultures may result in higher soil carbon stocks due to the complementarity of above-ground and underground niches, higher influx of litter into the soil due to canopy compaction, higher productivity of mixed stands, greater input of root litter due to the maximum efficient soil use by roots, effective regulation of carbon influx with precipitation and its carryover with soil waters. One of the most promising types of forest carbon projects for Russia is afforestation on abandoned agricultural lands because of the low project baseline, significantly large land areas, and high accessibility of abandoned lands. Published field studies show a high potential for the use of clones of fast-growing woody plants in forest carbon projects. However, long-term field experiments are required to assess the effect of silvicultural practices on soil carbon stocks for specific soils and specific climatic conditions.

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ВЛИЯНИЕ ЛЕСОВОДСТВЕННЫХ МЕРОПРИЯТИЙ НА ПОЧВЕННЫЙ УГЛЕРОД: ОБЗОР

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В работе приводится обзор российских и зарубежных статей, посвященных изучению влияния лесоводственных мероприятий на пул почвенного углерода для оценки эффективности лесоклиматических проектов. Проведенный анализ работ позволяет заключить, что лесоводственные мероприятия влияют на содержание почвенного углерода через изменение скорости поступления и разложения органического вещества и как следствие влияют на перераспределение углерода в профиле почв. Рубка высокой интенсивности, в том числе сплошная, уборка порубочных остатков, повреждение напочвенного покрова при посадке лесных культур, создание монокультур могут отрицательно влиять на пул углерода почв. Напротив, выборочные рубки и рубки ухода слабой интенсивности, оставление порубочных остатков, создание смешанных лесных плантаций, особенно на заброшенных сельскохозяйственных землях, являются перспективными лесохозяйственными приемами, которые способствуют накоплению и сохранению углерода почв.

Ключевые слова: *углерод, почва, лесные климатические проекты, лесоводственные мероприятия*

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