Chapter 13
Carbon Exchange, Storage and Sequestration

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Abstract

- Boreal forests sequester and store large amounts of carbon both above and below ground.
- Forest management could influence carbon storage.
- Differences between upland soils and peatlands are important. In peatlands, large amounts of carbon are stored in the peat, making them more susceptible to differences in forest management.
- On peatlands, carbon balance is mostly determined by groundwater levels.
- Carbon storage on both upland and peat soils depends on harvest intensity since most carbon losses, apart from harvested forest products, come from decomposition of roots and logging residues.
- Scales in both space and time are both important considerations when estimating the effect on carbon balances.

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13.1 Forestry's Role in Mitigating Climate Change

Understanding how carbon (C) cycling in forest soils is affected by forest management has long been an important research topic (e.g. Hesselman 1926). Large quantities of C are exchanged annually between forests and the atmosphere. Established forests are estimated to offset about 30% of global fossil fuel emissions (Birdsey and Pan 2015). The offset could be increased by expanding forested areas, improving C management in existing forests, and using wood for products and energy that substitute for fossil fuel emissions (McKinley et al. 2011). These approaches are interconnected and influenced by forest-management practices. For instance, reducing harvest rates immediately increases C storage in the forest, but it also decreases the influx of new C into harvested wood products.

The boreal biomes account for approximately 17% of the worlds land area, but more than 30% of the total terrestrial C stock, with the majority found in the soil (Bradshaw and Warkentin 2015). Lal (2005) estimated that in the boreal zone around 90% of the ecosystem C is in soils. A large part of this C is stored in peatlands. Still, in life-cycle analyses of forest C balances, soils are typically neglected.

To provide various ecosystem services, it is essential to optimise synergies and possibly minimise related trade-offs. Thus, we need to better understand forest-based mitigation potential and identify areas of potential conflict among different ecosystem services.

13.1.1 Comparing Carbon Impacts of Forest-Management Methods: A Challenging Task

Productive Fennoscandian forests (production >1 m^3/ha/yr) cover about 55 million ha. The climate varies enormously along the 2000 km latitudinal gradient, from low-arctic climate in the boreal north to the nemoral climate zone in southern Sweden. The west-east gradient changes from an oceanic climate and boreal rainforests on the Norwegian Atlantic coast, to the continental, dry forests of eastern Finland. Today's forest-production landscape is largely a twentieth century creation of active management and governance, aiming to optimise economic value and increase standing forest biomass. Today's forest landscape is a mosaic of different stand ages, dominant tree species, productivity classes, and patch sizes (< 1 to >100 ha), intermixed with semi-open and open land with low or no forest productivity (e.g. mires, bogs, and areas above the treeline and with thin soil). In addition, there is a large diversity among private forest owners' goals for their holdings. All these factors influence how forests are managed.

The dominant silvicultural method is rotation forestry (RF) or even-aged forestry. There is an ongoing debate regarding whether transitioning to continuous cover forestry (CCF) or uneven-aged forestry would enhance climate benefits. However, there are several management strategies that, depending on definitions, fall into one of these groups (see Chap. 2 for definitions). This variety of methods complicates the interpretation of available data. Further, contradictory results arise as the timeframe (present, 20, or 100 years), spatial scale (individual trees, stands, landscapes), and the choice of system boundaries (e.g., biological cycles within forests, industrial cycles, and the value of substitution for fossil C) vary among studies.

Additionally, several other factors should be considered when comparing the carbon balance of CCF and RF:

- What serves as the reference case for silviculture? What are we comparing CCF or RF against?
- How will forests and forestry develop in the future, considering legislation, forest certification and climate change?
- How quickly can fossil fuels be phased out and what is the future energy portfolio like?
- How will the value of substitution for fossil C change over time?

Considering these factors, comparing results from different studies is not generally straightforward. Differences in assumptions made in modelling and different studies could have led to sometimes contrasting results.

Currently, there are only a few comparisons of the C impacts of RF and CCF, and no clear conclusions can be drawn from these studies. Conclusions are affected by factors such as the examined C stocks, tree growth and decomposition models, and the baseline used in the comparison. Studies have examined the development of C stocks in trees and soil (e.g. Peura et al. 2018), while some have also included substitution calculations in the assessment (e.g. Pukkala et al. 2011; Pukkala 2014; Lundmark et al. 2016; Table 13.1). When comparing different studies, the examination period selected also has an impact on conclusions. In a Norwegian study by Nilsen and Strand (2013), the C stock of trees in RF was found to be three times higher than in CCF at the time of the measurement. However, as both CCF and RF can be implemented using a range of harvesting methods and intensities, making direct comparisons between the management methods is challenging. Hence highlighting the significance of the timing of field measurements for comparisons.

13.1.2 Carbon Cycling in Forest Ecosystems

In this chapter, we review the carbon cycle in forest ecosystems. For an in-depth look at the cycle and the processes involved, see, for example, Ågren and Andersson (2012) or Gower (2003).

The biogenic C cycle (Fig. 13.1) in forest ecosystems is dominated by two prominent fluxes: the uptake of carbon dioxide (CO_2) via photosynthesis (gross primary production), and the release of C through respiration. This can be autotrophic, originating from living biomass, like root exudates, or heterotrophic respiration, such as decomposition of organic matter including roots, stumps, and soil organic matter (SOM). Most assimilated C is lost via autotrophic respiration, leaving only a minor part of the assimilated C to accumulate in biomass and soil as net ecosystem production. Forest logging operations reduce the uptake via photosynthesis at least temporarily, regardless of management system, since both even- and uneven-aged forestry will lower standing biomass, although with substantial differences.

The north-south climatic gradient is also reflected in C storage. In Finland and Sweden, C storage in biomass and the soil are much higher in the south than the north (Stendahl 2017; Merilä et al. 2023).

Carbon cycling in boreal forests is closely linked with nitrogen (N) cycling. Nitrogen is the mineral nutrient that limits growth rates in boreal conditions. Detailed descriptions of how the nitrogen cycle interacts with decomposition are found in a long list of publications (e.g. Tamm 1990). Low N availability enhances decomposition while N addition via deposition or fertilisation slows decomposition (Mayer et al. 2020), especially on nutrient-poor soils, leading to an accumulation of soil organic matter. Stability of SOM is also controlled by soil pH. At high pH, the

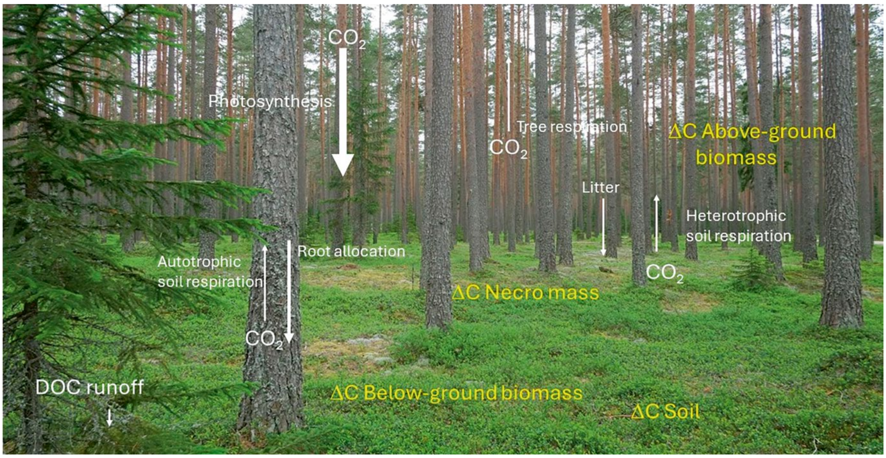

Fig. 13.1 Details of forest carbon cycling. Note that the length of the arrows does not reflect the size of the fluxes. Forest management can change the rate of these processes. Necro mass refers to dead organic material including woody debris. (Photo Lars Högbom)

solubility of the SOM increases, making more C available for soil microorganisms.

13.1.3 Soil Organic Matter (SOM)

Soil organic matter (SOM) is the largest C pool in boreal forest ecosystems. Carbon enters the SOM pool as dead organic litter from both above- and belowground. Over time, a significant portion of this is decomposed by fungi, soil fauna and bacteria and returned to the atmosphere. Forest fires also deplete SOM. The balance between C input and output determines whether there is a net increase or decrease in the soil's C stock.

Soil organic matter is highly variable and contains an array of organic substances with different degrees of recalcitrance and longevity. It is also important to differentiate between labile and stable soil C (Jandl et al. 2007). For technical reasons, fine roots (diameter < 2 mm) are usually included in the soil organic matter pool.

Quantifying changes in SOM and soil C content following various forest measures is notoriously difficult because of large within-stand variability and relatively small changes in very large soil C pools (Peltoniemi et al. 2004). Much of the C stored in SOM is highly stable, with a turnover time from hundreds to thousands of years. Only a minor fraction of soil C is actively cycled on monthly to yearly scales. Schmidt et al. (2011) estimated a mean SOM residence time of around 50 years. In addition to SOM, other C pools in the soil include stumps and coarse roots (> 2 mm).

Apart from the large CO_2 fluxes mentioned earlier, there are some other natural fluxes of greenhouse gases (GHG) in forested ecosystems. Leaching of dissolved organic carbon (DOC) constitutes a small fraction of the overall gross primary production. However, during heavy rain, the losses can be quite substantial. Methane (CH_4) fluxes, originating primarily from the soil surface, are minor under aerobic conditions. On the other hand, under anaerobic conditions in organic soils, CH_4 emissions can be substantial, for example from ditches (Rissanen et al. 2023). Since CH_4 is a potent GHG, it impacts climate change considerably. Another important GHG is nitrous oxide (N_2O) which can be produced in certain waterlogged conditions in nutrient-rich soils.

The litter derived from biomass, harvest residues and natural mortality constitutes an input into the soil C stock. Over time, a significant portion of this is decomposed by fungi, soil fauna and bacteria, and returns to the atmosphere. Additionally, soil C can be depleted when burnt by forest fires. This balance between C input and output determines whether soil C stocks grow or shrink.

Since the end of the last ice age, about 73 Mg C/ha has accumulated in the organic layer and the uppermost 50 cm of mineral soil of Swedish forests. Presently, 40–190 kg C/ha/yr accumulates within forests on mineral soils in Fennoscandia (Högberg et al. 2021). This increase is due to higher forest productivity and active suppression of forest fires. Notably, more C is stored in soil than in living tree biomass.

The carbon stored in peat soils (i.e. mire ecosystems) significantly surpasses that found in mineral soils, implying that peat soils have sequestered C faster since the last ice age. In contrast to most mineral soils, however, many organic soils are currently C sources (Jauhiainen et al. 2019). The total emissions of CO_2, N_2O and CH_4 from organic soils correspond to a loss of 1400 kg CO_2 eq. /ha/yr according to Swedish forestry-related climate reporting (Högberg et al. 2021). Drained peatlands emit the most, as aerobic conditions enhance peat decomposition, estimated to be around 5500 kg CO_2 eq./ha/yr.

Beyond C sequestered within biomass and soil, another important C stock is coarse dead wood, including standing or fallen dead trees and stumps following harvesting, although this stock is smaller compared to soils and live trees.

13.2 How Does Continuous Cover Forestry Affect Carbon Cycling in Forests?

13.2.1 Variability within Cycles and among Sites

Forest stands under RF and CCF differ considerably over time in C sequestration and stocks. Forests under RF begin their rotation period as a significant GHG source during clearcutting. Clearcuts on mineral soil lose around 16–20 Mg CO_2 ha/yr. (Grelle et al. 2012; Vestin et al. 2022; Grelle et al. 2023). For drained peat soils, this loss is larger. Tong et al. (2022a, 2022b) estimated the first-year loss in Sweden at 48 Mg CO_2/ha in a boreal stand and 26 Mg CO_2/ha in a hemi-boreal stand. In the second year, the losses had decreased to 26 and 7 Mg CO_2/ha. Korkiakoski et al. (2023) estimated the C loss at 30 and 22 Mg CO_2/ha for years one and two, respectively, in a nutrient-rich drained peatland in southern Finland.

As vegetation establishes and trees begin to grow, stands become C sinks, often within 10 years on mineral soils (Peichl et al. 2023; Grelle et al. 2023). The peak CO_2 uptake and most robust sink phase occur when the forest reaches about 40–50% of the normal length of a rotation period (e.g. Grelle et al. 2023; cf. Magnani et al. 2007). Subsequently, the sink slows as the forest ages (Odum 1969; Pregitzer and Euskirchen, 2004; Besnard et al. 2018; Repo et al. 2021). C stocks in dead wood and soil vary, influenced by factors like litter input and decomposition. Sequestration rates and stocks also vary among CCF methods, primarily triggered by harvesting. Since CCF forests retain standing trees, the variation is reduced. However, CCF forests require sufficient openness to facilitate regrowth, which lowers their sink strength and C stock in biomass compared to RF forests.

The variation in C sequestration and stocks depends on the specific CCF method used. For instance, CCF with gap cutting probably shows similar patterns to RF, particularly when evaluated at the scale of a gap.

Large broad-scale emissions at clearcutting are compensated by the high uptake of CO_2 in growing forest stands. Any annual variation due to active RF will be levelled out. However, a simulation study by Lehtonen et al. (2023) showed that at a national level, with equal harvesting levels, sinks are stronger when clearcuts are avoided on fertile drained peatlands. From a climate-mitigation standpoint, an important question is which forest-management approach stores most C in soil and biomass and reduces peat-soil-related emissions. The answer depends on underlying assumptions. For instance, biomass C stocks increase with extended RF rotation lengths.

How much continuous biomass storage can occur in CCF without impeding regrowth remains uncertain. Notably, ecosystem productivity emerges as a pivotal factor. A more productive management approach can sustain significant harvest potential and support litter production contributions to C stocks in woody debris and soil. Existing studies provide equivocal conclusions on productivity differences between RF and CCF, suggesting a small difference or large variation due to stand characteristics (see Chap. 4). However, the effect on peatlands, particularly nutrient-rich drained peatlands, differs from upland soils. On peatlands, climate benefits following CCF have been reported (Korkiakoski et al. 2023; Lehtonen et al. 2023). It is vital to note that ground vegetation and moss also contribute to overall ecosystem productivity, and that litter quality and local climate influence decomposition.

An alternative perspective is to begin with the present state of the forest (e.g., Pukkala et al. 2011) and to seek the optimal forest-management strategy for climate-change mitigation. In this perspective, the response may vary based on the forest's status, and therefore diverge among stands. Is it a 5-year old stand established after clearcutting? Does the analysed area encompass a forest with a well-distributed array of tree ages and sizes? Is it an old-growth spruce forest on fertile peat soil with elevated N_2O emissions? Is it an aging spruce forest where the sink strength has begun to wane? Can a present-day loss be offset by higher future uptake or vice versa? This final query underscores the importance of the timeframe considered.

13.2.2 Comparing CCF and RF

In this section, we discuss how the choice of RF or CCF may influence different elements of the forest C cycle. Direct comparisons between these two management systems are scarce in Fennoscandia and they differ in approaches, methods, and system boundaries. Effects of CCF and RF on growth and yield are discussed in Chap. 4.

Differences in forest soil C stocks between management regimes are the result of changes in organic material inputs to the soil and changes in decomposition rates.

CCF can create a more open stand structure, which may lead a to higher live-crown ratio in spruce forests (Bianchi et al. 2020; Kumpu et al. 2020). For Scots pine, 10-year-old clearcuts were shown to have lower above- and below-ground litter input than the partially cut stands (Roth et al. 2023). The highest litter input, however, occurs in uncut mature forest at the end phase of RF. Moreover, CCF has continuity of living roots and thus more below-ground litter input than RF (Prescott and Grayston 2023; Roth et al. 2023).

Decomposition is the processes of chemical and physical breakdown of organic material. It consists of leaching of organic compounds, fragmentation by animals and chemical alteration mainly by microbes, but light is also important. These processes are further regulated by the physical environment. Temperature, humidity, soil texture, soil disturbance, litter quality (chemistry, texture, etc.) and the decomposer community's composition and abundance all play roles.

The CO_2 emissions in the first stages after clearcutting in RF are probably the largest difference in stand-level C storage compared to CCF. During this period, an RF stand is subjected to highly variable sunlight, temperature, and humidity, compared to the more stable climate in a closed-canopy forest. At the same time, cutting large trees causes a flush of litter input and a total change in the decomposer community. Early-stage RF forests are dominated by saprotrophs, while mycorrhizal fungi that break down the most recalcitrant organic compounds largely disappear for some years after final felling (Wallander et al. 2010). There is also a transient period of litter inputs from a wider array of species, as the clearcut is first occupied by pioneer species.

During this stage, any differences in decomposition will depend heavily on the local climate, and also on decomposer communities. The question is whether the large-scale disturbance and successional setback changes full-rotation decomposition rates compared with a corresponding period in a forest with more frequent, but much-smaller-scale disturbances. Not surprisingly, we find no studies on this subject. There are also very few studies comparing mature, previously clearcut forests with CCF or other management regimes. However, in a comparison of Scots pine stands, Roth et al. (2023) found that fresh litter decomposed slower in a 10-year old clearcut than in a retention cut or an uncut stand. The clearcut had similar decomposition rates to a gap cut. They attributed the slower decomposition to a cooler microclimate and faster litter input in the retention and uncut stands. The clearcut and uncut mature stands represent the different extremes of RF, while gap and retention cuttings are two different types of CCF. Further, Purahong et al. (2015) compared decomposition of fresh leaf litter among even-aged spruce and beech forests, beech forests under CCF-like "near-to-nature" forest management, and an unmanaged beech forest in Germany. They found significantly faster general decomposition, lignin decomposition, and mineralisation of key elements in both RF forests compared with the CCF and natural forests.

Clearcutting has long-term effects on mycorrhizal-community composition (Kyaschenko et al. 2017; Hasby 2022) compared to never-clearcut forests. As some ectomycorrhizal fungi specialise on decomposing lignin and other recalcitrant litter components, such a change in the decomposer community might lead to differences in SOC. A recent meta-analysis by Latterini et al. (2023) concludes that a legacy of previous clearcutting decreases decomposition compared with unharvested controls, while retention-tree systems show increased decomposition. However, neither unharvested controls nor retention-tree systems are comparable with CCF and none of the included studies were performed in European boreal forests. In summary, existing knowledge suggests that potential long-term changes in the decomposer community might be the most important difference between CCF and RF, but it remains unclear to what degree this causes SOC differences.

13.3 Continuous Cover Forestry and Carbon Balance on Mineral Soil

Continuous cover forestry changes the timing of C fluxes compared to RF. After felling, RF forests are a C source until biomass growth and litter production exceeds the C released through the decomposition of soil C and logging residues (Kolari et al. 2004; Schulze et al. 2021). CCF may have C benefits since the source phase is shorter or non-existent compared to RF, leading to greater long-term average C sequestration. However, this might not be true if the C release in RF is later offset by larger uptake in RF than in CCF (see Chap. 4 for growth and yield comparisons). Nevertheless, for climate change mitigation, the timing of the emissions or uptake matters because of residence times in the atmosphere.

Only a few studies have compared the impact of RF and CCF on C sequestration in forests on mineral soils in Fennoscandia. There is, to our knowledge, only one published field study. This showed decreased C sequestration because of lower biomass production in CCF (Nilsen and Strand 2013). Studies based on models or simulations have found different results. Lagergren and Jönsson (2017) observed no difference in C sequestration between RF and CCF management in their ecosystem model analysis. Their model, however, involved a shift to more shade-tolerant tree species after CCF. A study by Peura et al. (2018) simulates a transition from RF to CCF forestry. The average annual C sequestration at a landscape level over a 100-year period was 0.68 Mg C/ha/yr, in CCF when harvested at 15-year intervals, reducing the basal area density to 10–12 m^2/ha, while in RF, C sequestration was 0.23 Mg C/ha/yr. However, the forest landscape shifted to CCF during this study's simulation period, and the total harvest was roughly 15% smaller in CCF than in RF, which may partly explain the difference. Both C-sequestration values are of the same order of magnitude as is typical of forests on mineral soils. These ranged from 0.45–0.51 Mg C/ha/yr from 1990–2017 when estimated using the methods of Finland's national greenhouse-gas inventory (EU NIR 2019). In another simulation study, harvesting at 10–30-year intervals in CCF reduced tree basal area to 8–15 m^2/ha (Shanin et al. 2016). In this study, net ecosystem production, excluding harvested biomass, ranged from −3 to +2 Mg C/ha/yr in various CCF methods. While CCF lacks a clearcut phase, this study shows that forest stands managed with CCF are not always C sinks but can be either a sink or source depending on harvesting intensity.

How will CCF affect soil C stocks in comparison to RF? In a Norwegian study based on field measurements, soil C stocks were higher in CCF than in even-aged forests, but the difference was not statistically significant (Nilsen and Strand 2013). A simulation study by Peura et al. (2018) found a somewhat higher average soil C stock in CCF than RF over a 100-year period. Since the harvested volume from the CCF simulation was lower than from the RF, and a decreased harvest led to higher soil C (cf. Mäkipää et al. 2023), it is difficult to distinguish whether the result is due to the management method or the decreased harvest. The hypothesis of colder microclimates and continuous litter input causing higher SOC stocks in CCF than on clearcuts was tested in a 10-year Finnish field experiment (Roth et al. 2023). No statistically significant differences were found in the SOC density or stocks between treatments despite the measured warmer microclimate and estimated lower litter inputs on the clearcut plots. Roth et al. 2023 view 10 years as too short of a period to detect measurable changes in stocks. It is well known that soils lose C the first year after clearcutting (e.g. Peichl et al. 2023; Grelle et al. 2023). However, the effect on soil C over the whole rotation period is still unclear. The microclimate, for example, can be less favourable for decomposition in a dense mature RF stand compared to a more open CCF stand (Roth et al. 2023). In addition, the large variation in CO_2 uptake within a rotation period makes the comparison more difficult. Some simulation studies, accounting for the whole rotation period, show that RF in comparison to CCF either harbours larger soil C stocks (the no biofuel alternative in Pukkala 2014), similar C stocks (Lagergren and Jönsson 2017) or smaller C stocks (Lundmark et al. 2016; Kellomäki et al. 2021; see Table 13.1).

However, a study by Roth et al. (2023) detected changes in the processes controlling organic matter accumulation and decomposition. In-situ decomposition was lower in retention cuts and in a mature uncut stand, where forest cover caused a cooler microclimate and higher litter input. Decomposition rates were equally high on clearcut sites and in canopy gaps of gap-cut stands, also indicating differences between CCF methods. In addition, the study found differences in litter quality between treatments. The study concludes that the accumulation of labile compounds in retention cuts together with decreased decomposition rates indicate a higher soil-C-accumulation potential in this CCF method.

Even small changes in soil C stocks can significantly impact boreal forest C budgets as two-thirds of the total C stock is below ground on mineral soils and even more on peatlands. Harvest timing and intensity determine the effects of CCF on soil-C stocks. Shanin et al. (2016) used the EFIMOD and ROMUL models to estimate that thinning from above every 10–20 years to lower tree basal area below 12 m^2/ha led to reduced soil C stocks compared to forests with an initially larger basal area. If harvesting was less intensive (basal area 16 m^2/ha after harvest), soil C stocks increased regardless of thinning frequency.

Soil preparation in RF is suggested to increase C losses from mineral soils. Consequently, the lack of soil preparation could be seen as a C benefit for CCF. However, a recent review by Mäkipää et al. (2023) concludes that the effect of post-clearcutting soil preparation on CO_2 emissions is minor. The potential C benefits depend on the effects on both soil and biomass. The simulation study of Kellomäki et al. (2021) concluded that in spruce forests of central Finland over a long period (401–1000 years), soil C stocks were significantly higher under CCF than RF, whereas C stocks in CCF trees were roughly twice those in even-aged forests. However, the ecosystem C benefit varies among studies (see Table 13.1). In

Table 13.1 Ecosystem-level C sequestration of continuous cover forestry (CCF) compared with rotation forestry (RF) on mineral soils. Parentheses () indicate a weak effect. HWP stands for harvested wood product and n.d. for no difference. Blank cells indicate that the topic was not covered by the study

Biomass C	Soil C	Ecosyst C	HWP C	Comment	Reference
–	n.d.	–	–	Field study, 81 years, no replicate	Nilsen and Strand (2013)
+	–	+		Model. RF low thinning no biofuel vs. CCF	Pukkala (2014)
–	–	–		Model. RF high thinning biofuel vs. CCF	Pukkala (2014)
n.d.	+	+	–	Model. Assumed same biomass production in CCF and RF	Lundmark et al. (2016)
–	+		–	Model. Assumed CCF had 80% of mean annual increment vs. RF	Lundmark et al. (2016)
n.d.	n.d.	n.d.	+	Model. Broadleaf fraction was 45% in RF and 13–20% in CCF	Lagergren and Jönsson (2017)
		(+)	–	Model. Decreased harvest in CCF	Peura et al. (2018)
–	+	–	+	Model. Increased biomass production in CCF	Kellomäki et al. (2021)

a simulation study comparing different management options in a changing climate with the LPJ-GUESS model, Lagergren and Jönsson (2017) found only minor differences in ecosystem C stocks and C sequestration between RF and CCF.

13.4 Continuous Cover Forestry and Greenhouse Gas Exchange on Peatlands

CCF is suggested as a feasible way to decrease C emissions and water runoff relative to even-aged management of drained nutrient-rich spruce peatland forests (Nieminen et al. 2018). CCF relies on continuously maintaining a tree stand with significant transpiration and interception capacity to moderate the water table, thereby avoiding large openings whose C sinks are temporarily weak due to having only surface vegetation.

Carbon emissions vary greatly between peatland sites and even within sites (e.g. Jauhiainen et al. 2019). Furthermore, there are significant uncertainties whether individual drainage locations are C sources or sinks. However, two key site characteristics increase peat decomposition and thus net C emissions: fertile soil and deep water tables (Minkkinen et al. 2020; Ojanen et al. 2013; Ojanen and Minkkinen 2019). Emissions particularly seem to increase if the water level drops below 30 cm (Ojanen et al. 2013). On the other hand, water levels approaching the ground surface (0–20 cm) after regeneration felling increase methane (CH_4) emissions (Ojanen et al. 2010; Korkiakoski et al. 2020).

Drained peatlands continue to lose C until the peat layer is fully decomposed. The amount of C sequestered in trees may not fully compensate for the loss from peat, and the end use of the harvested wood determines the ultimate contribution of forestry to climate change mitigation (Ojanen 2014). If CCF methods can maintain peatland water levels within a range that minimises C emissions, they may offer opportunities, at least in peatland areas, to control peat decomposition and reduce soil GHG emissions. The first findings of CCF impacts on C emissions from peatlands support this hypothesis. Studies conducted by Korkiakoski et al. (2020, 2023) found that a 13-ha test area subjected to felling based on CCF (selection harvesting, roughly 70% of total volume removed) was a minor source of CO_2, while net CO_2 emissions were five times higher in a clearcut. Correspondingly, the CCF site remained a CH_4 sink, whereas the clearcut became a minor source of CH_4 (Korkiakoski et al. 2020). Mäkiranta et al. (2010) and Korkiakoski et al. (2020, 2023) also measured N_2O emissions from clearcut sites that were many times higher than from otherwise-similar stocked peatlands. According to a simulation by Shanin et al. (2021), a nutrient-rich spruce-dominated peatland forest ecosystem managed by selection harvesting remained a C sink over the examined 240-year period, regardless of felling, provided the stand basal area remained above 6 m^2/ha (Fig. 13.2).

A modelling study found that prohibiting clearcuts on fertile drained peatlands would increase Finland's C sink by 1 Tg CO_2 eq./yr compared to business-as-usual management (Lehtonen et al. 2023). This simulation coupled the MELA simulator with the SpaFHy-peat hydrological model (Launiainen et al. 2019), the Yasso07 soil model (Tuomi et al. 2011) and empirical emissions models (Ojanen and Minkkinen 2019). In this work the major reasons for CCF's larger climate benefits were avoiding clearcuts and the period of seedling stands with marginal tree growth.

After clearcutting, emissions can be up to 30 Mg CO_2/ha/yr. In the 5 years following felling, a stand can lose an amount of C from peat equivalent to the timber harvested in the final felling (Korkiakoski et al. 2023). In the control area managed by partial harvesting (CCF), the net emissions were significantly smaller. The large emissions from the clearcut area seem to be due to the increase in the decomposition of surface peat, and especially the decrease in new C input as litterfall drops precipitously after cutting (Korkiakoski et al. 2023). After partial harvesting, the water table rises, reducing the fraction of the peat layer susceptible to rapid decomposition. However, on drained peatlands, even with CCF the water table may remain so low that the peat respiration rate will not necessarily decline significantly. There is therefore a risk that a change in management method alone will not be enough to reduce peat-soil respiration. However, more research is needed to verify this (Table 13.2).

Fig. 13.2 A spruce-dominated peatland forest after regeneration felling (upper photo, rotation forestry) and selection harvesting (lower photo, continuous cover forestry) in a drained, thick-peated site. Photos: Sakari Sarkkola

Table 13.2 Effects of CCF vs. RF on carbon sequestration in peatland forest ecosystems. HWP stands for harvested wood product and n.d. for no difference. An empty cell indicates that the topic was not addressed in the study

Biomass C	Soil C	Ecosyst C	HWP C	Comment	Reference
		+		Field study, partial harvest compared to clearcut	Korkiakoski (2023)
	n.d.			Field study, uncut compared to clearcut	Mäkiranta et al. (2010)
+	+	+		MELA simulations (Finland) with equal harvesting. CCF on fertile peat soils	Lehtonen et al. (2023)

13.5 Challenges and Future Research Directions

Several potential areas for further inquiry and research will help improve our understanding of CCF's carbon-sequestration potential. Carbon balance is closely linked with forest production and harvesting, so there is a need to incorporate prediction of biomass and soil-C development under CCF into simulation models like MELA, Heureka and others. MELA- and Heureka-type models should be coupled with soil models that can incorporate both mineral soils and drained peatlands in a way that allows users to optimise forestry for climate change mitigation and adaptation. The current MELA and Heureka models allow calculation of tree-biomass carbon, but soils are excluded. More data and information are needed about net ecosystem exchange after clearcutting on mineral soils and drained peatlands to estimate the full carbon impact of different forest practices. More studies are also needed on different harvesting regimes' effects on peatland groundwater levels.

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