Comparing living and dead tree carbon stocks between burnt and managed forests

– A chronosequence analysis between burnt and managed forests

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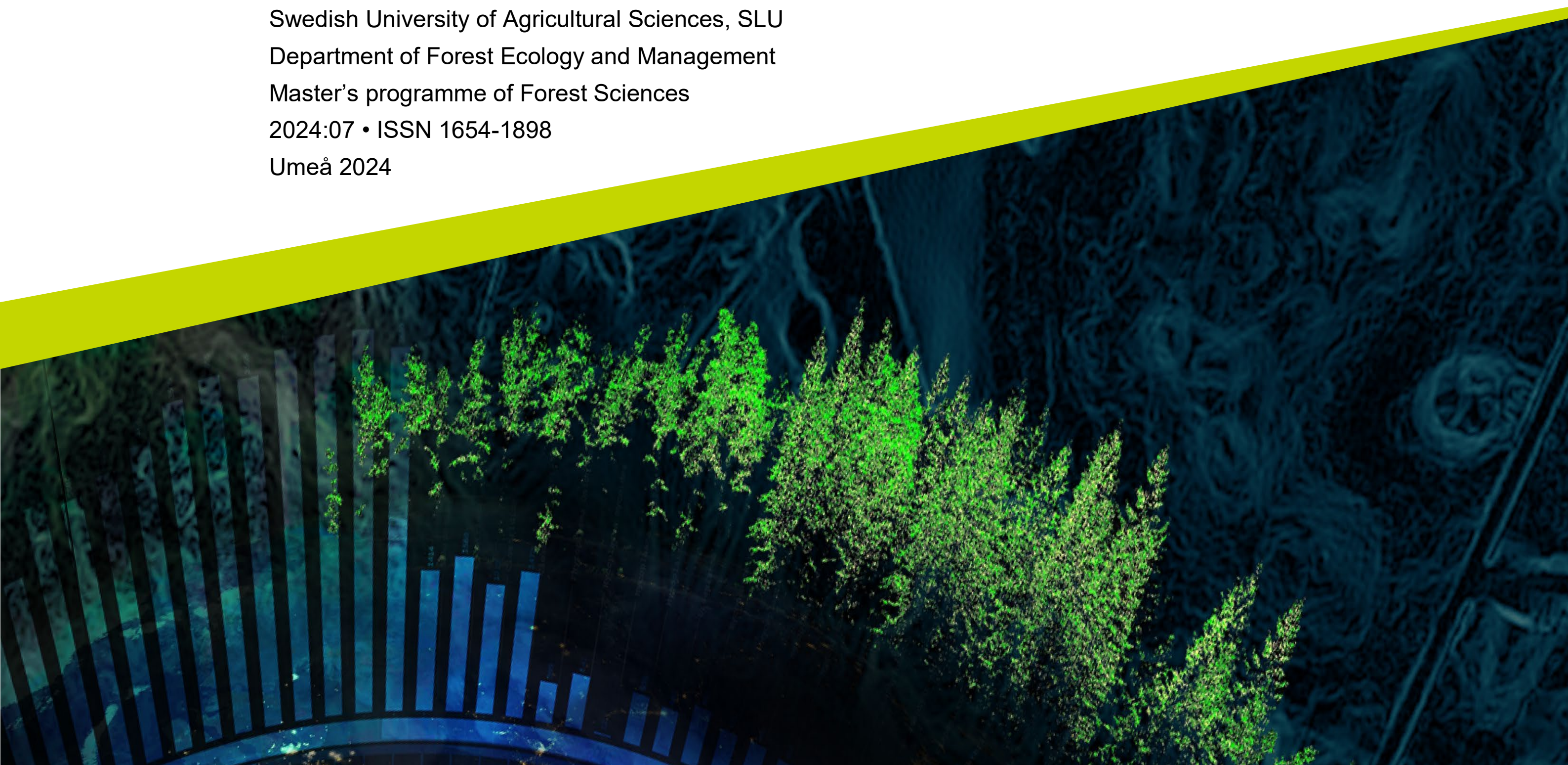

Comparing living and dead tree carbon stocks between burnt and managed forests – A chronosequence analysis between burnt and managed forests

Jämförandet av kollagren från levande och döda träd i brända och skötta skogar – En kronosekvensanalys mellan brända och skötta skogar

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Abstract

Boreal forests store about 32% of the global forest carbon stocks, second only to tropical forests that store around 55% of the global forest carbon stocks. However, with the changing climates, it is unsure whether managed or unmanaged forest store more carbon, especially after disturbances such as forest fires. Therefore, carbon stock analyses are important to help prevent climate change. This thesis aimed to compare the carbon stocks of living and dead trees in the boreal zone between two chronosequences: one of managed forest stands and one of previously burnt forest stands. The two chronosequences were created in Västerbotten and Norrbotten counties in northern Sweden and were based off similar site characteristics. Each stand in each chronosequence was inventoried and living and dead tree data was collected. The wood density was calculated for the living trees in each stand. Then the carbon stocks were calculated for all living and standing dead trees above 5 cm in diameter at breast height and for all lying dead wood above 1 cm at the smallest end. Finally, regression analyses were performed to compare the average carbon stocks over time for both chronosequences. The results showed that burnt forests had on average larger carbon stocks over 100-year time periods, but that the managed chronosequence had a larger and more linear carbon stock increase over time. This study did not include carbon stocks for trees smaller than 5 cm dbh, shrubs or the soil and did not account for the impact of soil nutrient availability on total carbon stocks. The study also did not include any calculation of the substitution effect of carbon stocks extracted from the managed stands. Overall, the study showed that 100 years after a forest fire or final harvest, burnt forests store more carbon than managed forest, but this topic could be expanded further.

Keywords: Carbon stocks, boreal forests, forest fire, managed forests, chronosequence, dead wood

Sammanfattning

Boreala skogar lagrar ungefär 32% av allt kol från världens skogar och lagrar bara mindre än tropiska skogar, som lagrar 55% av allt kol från världens skogar. Med dagens klimatförändringar så är det osäkert om skötta eller ej skötta skogar lagrar mer kol, framför allt efter stora störningar som skogsbränder. Kollageranalyser är därför viktiga för att hjälpa till att motverka klimatförändringar. I det här masterarbetet jämförs kollagren av levande och döda träd i den boreala zonen mellan två skilda kronosekvenser: en med skötta bestånd och en med före detta brända bestånd. Kronosekvenserna skapades i Västerbotten och Norrbotten län i norra Sverige och baserades på liknande beståndskaraktärisitik. Alla bestånd inventerades och data på levande och döda träd samlades in. Ved-densiteten beräknades för de levande träd i alla bestånd. Därefter beräknades kollagren för alla levande och stående döda träd med en brösthöjdsdiameter över 5 cm och för all liggande död ved med en minsta diameter över 1 cm. Slutligen utfördes regressionsanalyser för att jämföra det genomsnittliga kollagret över tid för båda kronosekvenserna. Resultaten visade att kollagren för de brända skogarna var i genomsnitt högre efter 100 år, men att den ökningen i kollagren var högre och mer linjär över tid för de skötta bestånden. Studien inkluderade inte kollagren för träd som var mindre än 5 cm i brösthöjdsdiameter, markvegetationen eller markkolet och redogjorde inte för hur skogarnas näringstillgång påverkade skogarnas kollager. Studien inkluderade inte heller någon uträkning av substitionseffekten av kolet som utvanns från de skötta skogarna. Den här studien visade att 100 år efter en skogsbrand eller slutavverkning så lagrade brända skogar mer kol än skötta skogar, men det här forskningsområdet kan utökas ytterligare.

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1. Introduction

1.1. Carbon sequestration and source-sink dynamics

The Earth's climate is changing due to an increase in greenhouse gas emissions, including rising atmospheric carbon dioxide (CO_2) concentrations throughout the 20^{th} and 21^{st} centuries. Trees sequester CO_2 from the atmosphere through photosynthesis as they grow, and when trees are harvested many wood products keep the carbon stored for a long time, helping to subsidise the use of non-renewable products and fuels (Bellassen & Luyssaert 2014). The rate at which trees sequester carbon changes over time as the forest undergoes succession. The terrestrial ecosystems of Earth sequester a lot of atmospheric CO_2 through biomass production, which helps offset the impact of fossil fuel emission on global climate change (Canadell *et al.* 2007).

Boreal forests store about 32% of the global forest carbon stocks and stores the second most carbon globally among forest types, with tropical forests storing around 55%, and temperate forests storing approximately 14% (Pan *et al.* 2011; Astrup *et al.* 2018). Boreal forests therefore play an important role in sequestering carbon dioxide (CO_2) and storing carbon in order to prevent climate change, especially since only around 11% of the terrestrial land area is covered by boreal forests (Bruckman & Pumpanen 2019; Peichl *et al.* 2022). When large-scale studies focusing on boreal C cycling have been conducted, they have been skewed towards the larger land areas within Canada and Russia, where a smaller percentage of the forest is actively managed compared to in Scandinavia, and the largest portion of forest succession is driven by natural disturbances (Peichl *et al.* 2022).

In boreal forests, trees grow for long periods. The managed boreal forests usually utilise intensive rotational forestry, which create distinct C cycles in each forest depending on which growth and management phase the forest currently is in (Bellassen & Luyssaert 2014; Peichl *et al.* 2022). Carbon uptake and sequestration is usually highly dependent on the ecosystem's gross primary productivity (GPP), which is the net total CO_2 uptake via photosynthesis. Net ecosystem respiration (NEP) is what you get when you subtract both heterotrophic and autotrophic respiration from the ecosystem, which releases carbon back into the atmosphere and therefore has a negative impact on the total carbon stocks in the ecosystem.

Both respiration and GPP will be impacted by disturbances such as fires and forest management. Parro *et al.* (2019) found that both soil respiration and soil carbon were lower in burned stands compared to unburned stands, where reduction was greater in younger than older stands. The Swedish forest landscape has changed from previously mostly fire-influenced before the 19th century, to being production-forest dominated with rotation periods varying between approximately 50-150 years before harvest (Östlund *et al.* 1997; Skytt *et al.* 2021). While trees in boreal forests generally achieve very stable and high growth rates between the ages of 50 and 100 years (Stokland 2021), it remains unclear how the average carbon balance of such forests over the full rotation period compares to unmanaged forests subjected to wildfire. Depending on the timescale, certain management strategies might be preferable regarding carbon sequestration (Bellassen & Luyssaert 2014). Swedish forests are sought to be a solution to many things: societal and economic welfare, climate change, biodiversity preservation and other ecosystem services. However, many of the uses and goals for Swedish forests are conflicting, and there is not always a scientific or social consensus on how to reach certain goals, such as counteracting climate change and preserving biodiversity. Sweden has set a goal to become a fossil-free society by 2045, and Swedish forests and forestry is proposed to be a big contributor to this goal (Berg 2018).

There are plenty of disagreements and debates on whether managed forests or unmanaged forests store more carbon and whether they act as carbon sinks, sources or are carbon neutral (Bellassen & Luyssaert 2014). A forest can be a carbon source or a carbon sink depending on how much carbon is sequestered through photosynthesis versus how much carbon is lost from the ecosystem through respiration or other sources of carbon losses. A carbon source is when more carbon is lost from the forest, and a carbon sink is when more carbon is sequestered into the forest than lost. The strength of the carbon source-sink dynamics varies depending on the age of the stand and how much C stored in living biomass and soils remains. Ecosystem respiration (ER) does not change as rapidly as photosynthesis after a disturbance, and thus forests become stronger carbon sources immediately after stand-replacing disturbances. Many articles have found that most forest ecosystems become carbon sources up to 10-20 years following stand-replacing disturbance like wildfire or clear-cutting, thereafter becoming net carbon sinks (Amiro *et al.* 2010; Matkala 2020). Canadian forests have been shown to lose around 3.17 ± 1.8 Mg C ha^{-1} and 60.3 ± 2.5 Mg C ha^{-1} for the first 9-17 years after a stand-replacing disturbance depending on the species/ecosystem types present in each forest, only starting to be offset at stand age of 19-47 years old (Coursolle *et al.* 2012). Boreal ecosystems have been found to be relatively time-invariant in terms of carbon sequestration and storage 20 years after disturbances due to GPP and ER being relatively age-invariant (Matkala 2020). The carbon sink strength in older forests can start decreasing and taper off over time as tree mortality starts increasing (Luyssaert *et al.* 2008). The research is a bit conflicting however regarding the strength of the carbon source-sink dynamics at different forest ages. Some papers suggest that reductions in harvests can yield short-term benefits in terms of carbon sequestration, whereas others argue that subsiding more fossil fuels with renewable products through increased harvest may provide more climatic benefits (Skytt *et al.* 2021). Skytt *et al.* (2021) concluded that reduced harvest levels provide more short-term climatic benefits compared to increased harvest levels, which would lead to reduced carbon uptake in the short term but potentially greater long-term carbon uptake, particularly in high productivity forests and given suitable substitution effects of replacing non-renewable products with biobased products. Some suggest that old-growth forests continue to act as strong carbon sinks, with Luyssaert *et al.* (2008) finding that forest ecosystems of ages between 15 and 800 years old have net positive carbon balances, with undisturbed old-growth forests continuing to accumulate carbon over long time-periods. Luyssaert *et al.* (2008) found that there typically is a decline in NEP with increased stand-age at the age of 200 and that there is a potential upper limit of biomass accumulation depending on the forest type and stand structure. Part of the reason why older forests continue to accumulate carbon is due to dead wood usually decomposing slowly over time, thus losing carbon to the atmosphere at a slower rate than in managed forests (Luyssaert *et al.* 2008). During this time, any CO_2 released through decomposition is usually evened out by ingrowth of seedlings in the ecosystem (Luyssaert *et al.* 2008). Therefore, it becomes important to include carbon stored within different types of dead wood when discussing carbon sequestration and climate benefits.

Slow-growing boreal forests are very important for export of timber and wood products (Astrup *et al.* 2018). In the context of carbon sequestration, carbon storage and climatic benefits, however, the commonly used long rotation periods for managed forestry may be too short in the context of trying to reverse current climate change. It has been found that volume increments in older forests do not substantially decrease in stands older than the typical rotation age adopted in Scandinavian boreal forests, and that extending the rotation periods 20-30 years may sequester the most carbon, even when considering the substitution effect from wood products (Stokland 2021). There is also the issue of forests sequestering carbon at different rates depending on the age of the trees, as the net primary production (NPP) of ecosystems differs more compared to net ecosystem respiration. NPP at forest stand-level has been shown to decrease with forest age, particularly in the boreal region (Bond-Lamberty *et al.* 2004). This decrease in NPP is often attributed to tree mortality and species shift. NPP at the forest stand-level has been shown to decrease with forest age, particularly in the boreal region (Bond-Lamberty *et al.* 2004). NEP however is not always highly affected by stand age, as Fernandez-Martinez *et al.* (2014) found that stand age had a smaller impact on NEP compared to GPP. Different tree species sequester carbon at different rates, with dead trees of different species also storing varying amounts of carbon and decomposing at different rates. Some tree species also have different carbon densities that can shift over time. For example, Norway Spruce (*P. abies*) has been shown to have a lower carbon concentration in decaying wood compared to Scots Pine (*P. sylvestris*; Sandström *et al.* 2007). Carbon density in dead wood also increases at higher levels of decay according to the Swedish National Forest Inventory (NFI) system of decay classes, when measuring dead wood decay, due to the loss of certain carbon compounds and increased lignin abundance in more decayed wood.

It is sometimes uncertain how much carbon unmanaged forests store compared to managed forests, and whether managing forests in favour of biodiversity preservation could be a feasible strategy for combatting climate change. Natural disturbances like wildfires and insect outbreaks may also throw a wrench into the conversation, as it is less studied how natural succession regimes of different forms impact carbon budgets in unmanaged forests. Carbon and biomass analyses of both managed and unmanaged stands of different ages are required to provide better support for management decisions regarding how to best balance forest management, biodiversity preservation, and climate change mitigation in boreal forests. Carbon analyses of specifically burnt forests provide different angles to the carbon balance debate as natural disturbances like fires that are more common in unmanaged forests may create discrepancies in carbon budgets.

1.2. Boreal disturbances: Forest fires and harvests

Natural and anthropogenic disturbances affect both carbon uptake and carbon stocks (Coursolle *et al.* 2012). Disturbances are key components for forest regeneration and succession and can include forest fires, harvests, storms and insect infestations (Matkala 2020). Disturbances therefore impact the carbon balance, and cause vegetational succession, which reduces the net carbon uptake of the vegetation through the death of photosynthesising tissue, decreased biomass growth and biomass removal (Amiro *et al.* 2010; Matkala 2020). Disturbances changes net ecosystem CO_2 exchange, sometimes by changing ecosystem heterotrophic respiration, and always by influencing post-disturbance vegetation growth. Forest fires remove fine materials, and harvests remove coarse woody material, which may impact respiration patterns, with fires also mineralising nutrients more quickly (Matkala 2020).

Natural disturbances are increasing in frequency in the boreal zone because of a rapidly changing climate, in particular wildfires are increasing in prevalence (Astrup *et al.* 2018). Boreal forests also experience the largest increase in temperatures amongst forested regions, which increase the frequency and impact of disturbance regimes, and may alter the long-term carbon-sequestration dynamics

(Bond-Lamberty *et al.* 2004). The CO_2 emissions from wildfires can in some situations almost equal the total net CO_2 ecosystem uptake in certain regions, as wildfires in the Arctic Basin 1997-2006 equalled almost 80% of the total CO_2 sequestrations in the affected region (Astrup *et al.* 2018). Bond-Lamberty *et al.* (2004) found in their study that approximately 9% of the NPP in their studied chronosequence of burnt black spruce stands in Canada was consumed in the fires, but that higher losses could have occurred due to carbon losses through errors in combustion measurements, erosion, and dissolved organic carbons.

Different disturbances can impact the NEP and carbon stocks of forests differently. The type, severity, and effect of the disturbance on the ecosystem plays a major role on how the carbon source-sink dynamics play out following the disturbance, controlling the amount of photosynthetic biomass, the respiratory biomass (or both), as well as other ecosystem functions. For sites that had burned, stands younger than 10 years post fire became carbon sources on average, becoming carbon sinks afterwards with high interannual variability in NEP (Luyssaert *et al.* 2008). GPP increased with stand age 20-30 years after the disturbance, where ER did not vary much with age (Matkala 2020). Harvested sites had similar changes in NEP as burnt sites, though it was more climate and site-dependent, with the greatest C loss being in warmer sites compared to cold. GPP recovered within 20 years of the initial harvest, though ER did not vary much over time (Matkala 2020). Large and intense disturbances such as wildfires heavily affect forest structures, species composition, biogeochemical cycle, energy flows, landscape diversity and carbon fluxes and storages within the boreal region (Amiro *et al.* 2010). This can result in patchy forest mosaics within secondary succession. Although the initial losses of carbon are usually great, the carbon stocks tend to recover over time.

1.3. Research questions and hypotheses

When you intend to investigate the effects of disturbances on forest stands over time, a practical approach is through chronosequence analyses instead of using long-term experiments that can span several decades. Chronosequence analyses aims to study sites with varying ages and of as similar characteristics and disturbance histories as possible, to evaluate the effects and changes over time (Amiro *et al.* 2010). When comparing different disturbances and their effects on forests, it is common to use a chronosequence for each disturbance of interest. This thesis aims to analyse and compare the aboveground carbon stocks between two chronosequences; one of managed forest stands and one of previously burnt forests of similar site characteristics in the boreal zone. The carbon stock analyses will provide evidence for whether managed forest stands store more or less carbon over a rotation age than burnt unmanaged stands, and to what degree these properties change with forest stand age. The thesis will also help analyse whether there are climatic benefits to harvesting forests compared to letting natural disturbances like wildfires be the main successional determinant in boreal forests. The main questions for this thesis are the following:

1. How does wood carbon density differ between managed and burnt stands, and how does this influence the carbon stocks?
2. Do managed forest stands or unmanaged burnt forests store more carbon, and how does it vary over time?
3. How does dead wood affect the development of carbon stocks between managed and burnt stands?

The hypotheses for each research question are:

1. The wood carbon density in the living trees will increase with age in both managed and burnt stands, but the trees in burnt stands will have higher carbon density than the trees in the managed stands.
2. Managed forests will have higher carbon stocks than burnt forests at around 100 years of stand age. However, the carbon stocks will vary more over time in the managed forest stands compared to the burnt stands across similar time periods, such that unmanaged forests will have a higher average carbon stock.
3. Dead wood will play a bigger role in the carbon budget in burnt stands due to a higher abundance and less dead wood remaining for long in managed forests.

2. Materials and methods

2.1. Area of study

Two chronosequences of forest stands with similar sites were created, with the forests being located around Arvidsjaur and Vuollerim in Västerbotten and Norrbotten counties in Sweden. One chronosequence contained managed stands and one contained unmanaged stands with known fire history. Each stand in the chronosequences was searched for in a GIS framework, following the criteria that each site was pine dominated, mesic, and had dwarf shrub and feathermoss ground vegetation communities. When possible, the stands were selected in clusters, with unmanaged and managed stands being in close proximity to each other (Fig 1). Each stand in either chronosequence was given a name based on if it was a managed or burnt stand, as well as a number relating the average stand age for the managed stands or years since the last fire in that stand for the unmanaged stands. In each stand, five subplots were marked using GPS coordinates in each cardinal direction with one centre-plot. Living tree data, dead wood data and tree cores were collected from Scots Pine trees in these subplots.

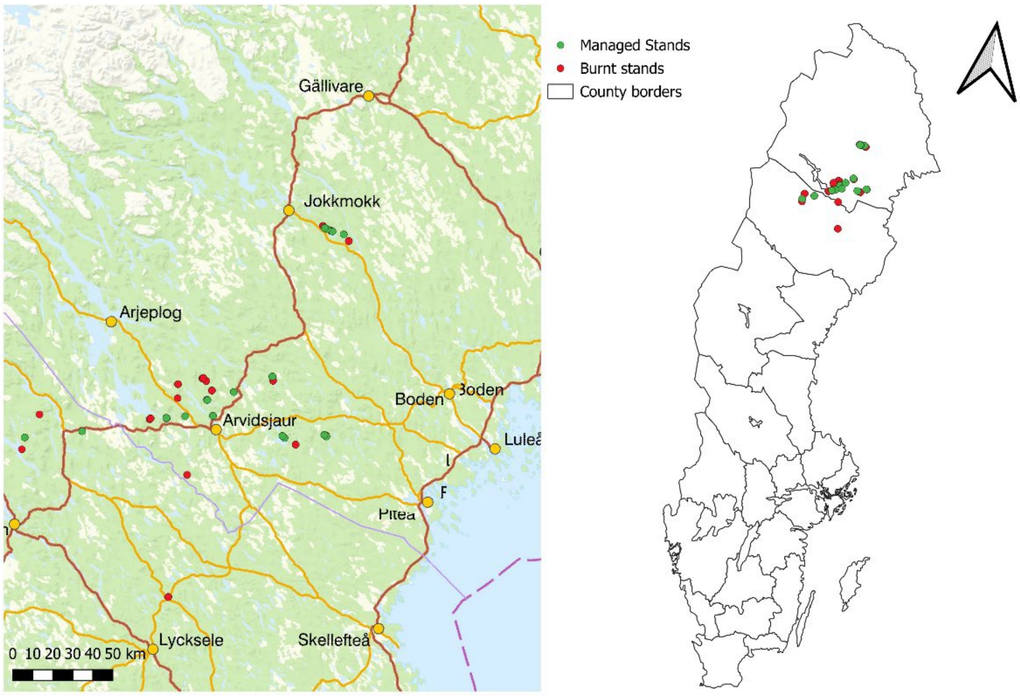

Figure 1. The study area of the two chronosequences of managed and burnt forest stands in the counties of Västerbotten and Norrbotten in Sweden. Green dots represent the managed chronosequence and red dots represent the burnt chronosequence. The topographic map background map to the left originates from Statistics Sweden in the Sweref99 TM coordinate system. The map of Sweden to the right originates from the Swedish Land Survey service (2022).

2.2. Lab work

The tree cores collected from each subplot were first put into marked 15 ml tubes, filled with deionized water, and then sealed. Then a scientific balance-scale was filled with deionized water covering the inverted basket by about 2 cm. The scale was turned on and water temperature was measured with a fluke thermometer and set to 21 °C. The cores were removed from the tubes one at a time and gently dried on paper towels. The scale was balanced without the core pieces and then the core pieces were put into the water of the scale so that they floated and pushed into the inverted basket. The scale door was closed, and after weight stabilisation, the volume and weight of each sample were recorded. After this The core pieces were then gently removed and put into pergamyn bags before weighing the next sample. Once all cores had been weighed and put into separate pergamyn bags, they were put in an oven and dried at 60 °C for 24 hours. Once dried, each core sample were weighed again on a dry scale to obtain the dry weight, after which the wood density of the tree core was calculated (mass/volume). Then, each tree core was ground twice, once roughly using a manual coffee bean grinder and once thoroughly afterwards using metal grinding cups with a grinding ball inside. When grinding and transferring the ground wood cores from the coffee grinder to the grinding cups using metal spoons, gloves were worn to avoid contaminating the sample with wood powder from another core. The grinding cups with ground wood cores inside were then attached tightly to a lab grinder and were ground at a frequency of 23 rpm for 6 minutes per sample. Once a sample had been ground into very fine wood powder, the powder was extracted into 2 ml beakers, trying to fill 2-3 beakers per wood core sample, depending on the initial size of the tree core. The coffee grinders, grinding cups and spoons were thoroughly cleaned using ethanol between each sample. The beakers were then sent to another lab in order to conduct a carbon-nitrogen analysis. The C and N in each sample was converted to CO_2 and N_2 through combustion and the dry mass was obtained by oven drying the content at 70 °C for 18 hours. Then the average carbon content in each sample was estimated using mass spectrometric measurements on the CO_2 and N_2. Finally, the carbon content in the trees of each subplot were compared to see if they substantially differed depending on if the stand was burnt or managed.

2.3. Biomass calculations and carbon conversions

2.3.1. Living trees

In each stand, five subplots with a diameter or 20 meters each were marked using a GPS, one subplot being in the middle of the stand and the rest located in the halfway point from the middle of the stand and each corner. If there were many trees in each stand, the subplot diameter was adjusted to 10 meters instead. Diameter at breast height (DBH) was recorded for each tree in each subplot, whereas tree height was also recorded for 4 trees per tree species per subplot, each with a DBH greater than 5 cm. For the trees where both height and DBH was measured, total tree biomass for each tree was calculated for each tree using biomass functions for different parts of the tree by Marklund (1988) and root biomass functions by Petersson and Ståhl (2007). After estimating tree biomass for each part of the tree (stem + bark, dead branches, living branches, needles, stump and roots) and for each tree species, the height for the remaining trees had to be modelled in order to apply the biomass functions to those trees as well. A linear mixed effect model was created in R-Studio to predict the height of the trees based on the species and DBH of the validation trees, using the stand of each tree and the subplot where it was measured as the random effect variable for the model (*RStudio: Integrated Development Environment for R* 2023). The model was built in R-Studio using the lme4 package v1.1-34 (Douglas Bates 2015) .The model looked like this in R:

lme.height.prediction←lmer(Height.m~log(DBH.cm)*Species+(1|Stand/Subplot), data=validation.data)

After height for the main trees was predicted, biomass for the remaining trees was calculated using the modelled height. The biomass results from the trees with modelled height were compared to the biomass of the trees with measured height. Then the biomass calculations for each part of the tree (stem + bark, dead branches, living branches, needles, stump and roots) were summed together to get total tree biomass. The total tree biomass evaluations were divided by the sampling area for the subplot where they belonged to get biomass estimates for each tree in kg/m^2. Then the biomasses for all trees belong to the same stands were added together to get stand biomass in kg/m^2. Then the stand biomass was recalculated to Mg/ha by multiplying it with 1000/10 000. Finally, the stand biomass calculations were converted into total carbon pools by multiplying the average carbon content percentage of 0,5232. This was the average C content value of all tree cores from all stands. Graphs were then created to illustrate the change in carbon pools over time and to compare the differences between burnt and managed stands.

2.3.2. Lying dead wood

Lying dead wood was recorded in each stand if it was lying inside the subplots and were wider than 10 cm in diameter at the large end. The length of the logs and their diameter at each end were measured. Tree species and dead wood decay class were recorded in the same manner as inventories by the Swedish National Forest Inventory (NFI, classes: 0-4; Fridman & Walheim 2000). From the measurements, volume was calculated for each log using the equation for a conical frustrum shape through the website made by Furey (2023-10-04), as a conical frustrum shape is commonly used when estimating the volume of coarse woody debris (Fraver *et al.* 2007).

After calculating the tree volumes, the tree mass was calculated by multiplying the tree volume with its estimated wood density. Sandström *et al.* (2007) calculated the dead wood density (g/cm^3), as presented in Table 1 for Scots Pine and Norway Spruce logs from both preserved and managed forests based on their decay class. They also calculated dead wood density (g/cm^3) for Birch trees in managed forests based on the decay class. The wood densities were recalculated from g/cm^3 to kg/m^3 by multiplying the density by 1000. Then, log mass was estimated for the different tree species by multiplying the log volume by tree density for that tree species, decay class and forest type. Then the carbon content for each log was calculated using a carbon content conversion factor for each species and decay class. The average carbon content (%) used for lying dead wood was taken from Sandström *et al.* (2007) for Scots Pine and Norway Spruce, and from Mäkinen *et al.* (2006) for Birch, though their calculations used different albeit similar decay class system, with decay classes of 1-5. Despite the carbon content for birch being calculated for a different decay class system (class 1-5), each system had the same amount of decay classes and the descriptions of each were similar enough at each corresponding level in both systems. Thus the carbon content for birch from Mäkinen *et al.* (2006) in decay class 1 was used as if the log was categorized as decay class 0 using the Swedish National Forest Inventory system, and so forth for each decay class. For some logs, the tree species could not be detected, and therefore the wood density and carbon content factors used were the same as for Norway Spruce in the corresponding forest type and decay class. For the non-birch deciduous logs, the mass was calculated using the same dead wood densities as for birch, as Sandström *et al.* (2007) had done the same in their article.

After calculating all log biomass and carbon mass, all log biomass values of the same tree species within the same subplots in each stand were added together and divided by the area of the subplot where the logs were recorded to get mass per area (in kg/m^2). The mass per area values were averaged for the 5 subplots for each stand to obtain a single mass per area value per stand for both log biomass and carbon mass for each tree species. Finally, each stand biomass and carbon estimate were recalculated to tonnes per ha (Mg/ha) by multiplying the kg/m^2 measurements by 10 000.

2.3.3. Standing dead wood

Calculating the standing dead wood was different compared to calculating the lying dead wood. The standing dead trees were recorded with no decay class associated to them. DBH and height of the dead trees were recorded, however some standing dead trees were missing height (i.e. in stands recently burned by stand replacing fire), thus a linear mixed effect model was created in R-Studio to predict the height of the dead standing trees with missing height measurements to fill in the gaps, as was done for live trees described earlier. The model was created using the lme4 package (v1.1-26; Bates et al., 2015), was using and looked like this in R-Studio:

height.dead.trees🡐lmer(Height.dead~DBH.cm+Species+(1|Stand/Subplot), data=standing.dead.tree.data).

As standing dead trees decay and lose height and primarily top-end volume, a volume assuming a different geometric shape than a truncated frustrum or cylinder had to be applied to the volumetric calculations (Woodall & Westfall 2008; Ducey & Fraver 2018). Ducey and Fraver (2018) developed multiple dead wood volume equations to take the change in dead wood mass and geometric shape into consideration. The diameter for the standing dead trees in this study was only measured at breast height (1,3 m height) and thus, one equation in Ducey and Fraver's article fitted the data the best: a volume equation for a conic-paraboloid, assuming the top-end basal area of the tree being 0. The equation follows: $V_{cp} = \frac{5*A_0*L}{12}$, with V_{cp} being volume of conic-paraboloid, A_0 being basal area at the bottom of the tree and L being the length of the tree. This equation was applied to the standing dead trees to calculate their volume, however, since the diameter was for the trees was measured at breast height and not at the bottom of each tree, the total volume would not be accounted for using this equation alone. Therefore, each standing dead tree higher than 1.3m was complimented with a simple cylindrical volume equation of $V = \pi * r^2 * h$, with r being the radius of the tree at 1.3m and h being the height of 1,3m.

After having calculated the total volume of each standing dead tree, the dead wood biomass was calculated by multiplying the volume with the snag wood density values calculated by Mäkinen *et al.* (2006) for Pine, Spruce and Birch respectively. Mäkinen calculated the wood density for each decay class of the standing dead wood separately, and as the decay class of each standing dead tree was not registered, the wood density values used to calculate the standing dead tree biomass was obtained by averaging the wood densities of each decay class for each species (table 2). After calculating the standing dead wood mass, it was converted to carbon stocks by multiplied by the standing dead tree carbon content percentage calculated by Mäkinen *et al.* (2006). Just like with the wood density, each carbon content factor had been calculated for each decay class individually, and therefore the carbon content was averaged across all decay classes (Table 3) and multiplied with the dead tree biomass to get total standing dead wood carbon stocks.

After calculating all standing dead wood biomass and carbon mass, all snag biomass values of the same tree species within the same subplots in each stand were added together and divided by the area of the subplot where the logs were recorded to get mass per area (in kg/m^2). The mass per area values were averaged for the 5 subplots for each stand to obtain a single mass per area value per stand for both log biomass and carbon mass for each tree species. Each stand biomass and carbon estimate were recalculated to tonnes per ha (Mg/ha) by multiplying the kg/m^2 measurements by 10 000. Finally, 100-year estimates were created for the carbon stocks of each chronosequence by calculating average carbon stocks over 100-year timeframes using regression equations.

2.4. Statistical analyses

After aggregating and calculating the carbon stocks for living trees, lying dead wood and standing dead wood, statistical analyses were conducted with a significance level of 0.05 on the total carbon tocks in Mg/ha per stand in R-Studio and applied to the total calculated carbon stocks and each individual carbon stock in order to determine if forest type (managed vs burnt), stand age/time since fire, the interaction effect between forest type and stand age/time since fire and tree species significantly affected the carbon stocks. Linear regression models were created for each carbon stock and the total carbon stocks and the results are presented in Table 6. For all models, the total carbon stocks had to be either log or square-root-transformed and the age of the stand had to be square-root-transformed. Each statistical model created in R-Studio looked as follows:

Total carbon stocks:

```
lm.carbon.total<-lm(log(carbon.ton.ha+0.1)~stand.type*sqrt(age),
data=stats.carbon)
```

Living trees:

```
lm.carbon.living<-lm(log(carbon.ton.ha+0.1)~stand.type*sqrt(age),
data=living.stats.carbon)
```

Lying dead trees:

```
lm.carbon.logs<-lm(log(carbon.ton.ha+0.1)~stand.type*sqrt(age),
data=logs.stats.carbon)
```

Standing dead trees:

```
lm.carbon.snags<-lm(sqrt(carbon.ton.ha)~stand.type*sqrt(age),
data=snags.stats.carbon)
```

3. Results

3.1. Wood densities and carbon percentages

The average living tree wood density from the cored trees was 0.453 g/cm^3, 0.42 g/cm^3 and 0.437 g/cm^3 for the burnt, managed and all forests combined respectively (Table 1). The carbon content for the living trees in each stand type was 52.42%, 52.22% and 52.32% for burnt, managed and all forests combined respectively (Table 1). The tree carbon-content that was calculated from the tree cores did not significantly differ between the trees and therefore, an average carbon content percentage was chosen to represent each tree from each stand, no matter if it was managed or burnt. The average carbon percentage of living trees was calculated to 52.32% and the rest of the carbon percentages used for the dead wood is presented in Table 2. Table 3 presents the wood densities used for each tree species, divided by decay class for the lying dead wood and average wood density for the standing dead wood.

Table 1. Average wood density (g/cm^3) and carbon content (%) for the living trees in each stand.

Stand type	***Average wood density (g/cm^3)***	***Carbon content (%)***
Burnt forests	0.453	52.42
Managed forests	0.42	52.22
All forests	0.437	52.32

Table 2. The biomass to carbon mass conversion factors used to calculate the results for each type of tree mass (living trees, lying dead wood in different decay stages and standing dead wood). The conversion factors for the dead wood were obtained from Sandström et al. (2007) and Mäkinen et al. (2006).

Carbon percentage conversion factor per mass type	***Pine***	***Spruce (including unknown coniferous tree)***	***Birch (including known/unknown deciduous trees)***
Living trees	0.5232	0.5232	0.5232
Lying dead wood (decay class 0)	0.5032	0.4922	0.4923
Lying dead wood (decay class 1)	0.5052	0.4917	0.4978
Lying dead wood (decay class 2)	0.5146	0.4968	0.5014
Lying dead wood (decay class 3)	0.5146	0.5081	0.5105
Lying dead wood (decay class 4)	0.5223	0.5127	0.4967
Standing dead wood	0.5127	0.5092	0.4962

Table 3. Standing dead wood density (kg/m^3) calculated by Mäkinen et al. (2006) for Pine, Spruce and birch for each decay class and averaged across all decay classes.

Tree Species	***Wood density decay class 1 (kg/m^3)***	***Wood density decay class 2 (kg/m^3)***	***Wood density decay class 3 (kg/m^3)***	***Wood density decay class 4 (kg/m^3)***	***Average wood density (kg/m^3)***
Pine	413.51	326.79	289.77	246.35	319.11
Spruce	376.65	290.34	261.58	179.84	268.52
Birch	413.94	308.66	238.08	214.19	293.72

3.2. Carbon pools

All carbon pools included tree species of at least 5 cm in diameter at breast height and above for living and standing dead trees, with lying dead wood including trees with a minimum diameter of 1 cm at the shortest end. Living trees accounted for the largest carbon pools in most of the stands (Table 4, Fig. 2 and 5), particularly with increasing time after the disturbance (Fig. 5). The standing dead wood made up a larger portion of the total carbon stocks than the lying dead wood in most stands, with some exceptions of younger unmanaged stands where many dead trees remained standing following fire (Fig. 3 and 4.). For all parts of the carbon stocks, coniferous trees, particularly living Scots Pine, made up the largest portion of the total stocks. The abundance of different tree species differed depending on the time after disturbance, and Birch trees appeared much more common among all carbon stocks in the burnt stands compared to the managed stand. Spruce C generally increased through time in both chronosequences, but no clear trends for lying and standing dead tree carbon were observed. The regression analysis showed that carbon stocks on average increased by 0.0674 Mg C ha^{-1} year^{-1} and 0.96 Mg C ha^{-1} year^{-1} for burnt and managed forests respectively (Fig. 6 and 7, Table 5). The results from the regression equations showed that average carbon stocks after 100 years were 88.4 Mg C ha^{-1} year^{-1} and 53.1 Mg C ha^{-1} year^{-1} for burnt and managed forests respectively (Table 5).

Table 4. Biomass and carbon pools of living trees, lying and standing dead wood, all with a diameter of at least 5 cm.

Stand	***Stand Type***	***Living tree biomass (Mg/ha)***	***Living tree carbon stocks (Mg C/ha)***	***Lying dead wood biomass (Mg/ha)***	***Lying dead wood carbon stocks (Mg C/ha)***	***Standing dead wood biomass (Mg/ha)***	***Standing dead wood carbon stocks (Mg C/ha)***	***Total stand living/dead tree biomass (Mg/ha)***	***Total stand living/dead tree carbon (Mg C/ha)***
F4a	Burnt	0	0	24.9	12.7	116.2	59.1	141.2	71.8
F4b	Burnt	133	69.0	4.7	2.4	49.9	25.6	186.6	97.0
F5	Burnt	71.2	37.2	23.4	11.7	41.5	21.0	136.1	69.9
F8	Burnt	79.6	41.7	15.3	7.9	31.7	16.1	126.7	65.7
F28	Burnt	217.1	113.6	17.3	8.8	26.3	13.5	260.8	135.9
F51	Burnt	89.5	46.8	3.7	1.9	0	0	93.2	48.7
F56a	Burnt	24.9	13.0	24.9	12.8	32.4	16.6	82.2	42.5
F56b	Burnt	59.4	31.1	28	14.354	5.9	3.0	93.3	48.5
F98	Burnt	192.4	100.7	7.7	3.9211	25.3	12.8	225.4	117.3
F121	Burnt	166.8	87.3	7.2	3.6283	16.9	8.6	190.8	99.5
F137	Burnt	187.2	98	3.1	1.5957	20.9	10.6	211.3	110.2
F197	Burnt	163.6	85.6	12.0	6.0101	26.3	13.3	202	104.9
F208	Burnt	172.5	90.3	2.4	1.2441	6.2	3.2	181.2	94.7
F229	Burnt	308.4	161.3	15.9	7.9839	28.1	14.4	352.3	183.7
F263	Burnt	168.3	88.1	16.9	8.6079	26.1	13.3	211.4	109.9
F288	Burnt	149.0	78	14.2	7.1543	18.9	9.7	182.1	94.8
F310	Burnt	94.7	49.6	18.6	9.3764	27.4	14.0	140.7	72.9
F375	Burnt	116.1	60.7	21.2	10.602	21.2	10.8	158.5	82.2
M1a	Managed	0	0	3.1	1.5411	0	0	3.1	1.5
M1b	Managed	38.2	20	5.2	2.6636	10.7	5.4	54.1	28.0
M2	Managed	0	0	7.3	3.6832	0	0	7.3	3.7
M13	Managed	2.1	1.1	18.3	9.4025	0	0	20.4	10.5
M18	Managed	20.0	10.5	4.4	2.2698	8.5	4.3	32.9	17.1

M24	Managed	17.4	9.1	13.3	6.7587	0	0	30.7	15.9
M32	Managed	66.3	34.7	3.5	1.8065	4.5	2.3	74.3	38.8
M36	Managed	56.4	29.5	0.3	0.1518	0	0	56.7	29.7
M39	Managed	62.7	32.8	2.1	1.0907	0	0	64.8	33.9
M42	Managed	113.6	59.4	1.5	0.7913	0	0	115.1	60.2
M61	Managed	82.1	43	3.2	1.6639	4.0	2.1	89.4	46.7
M65	Managed	222.5	116.4	3.1	1.5577	4.3	2.2	229.9	120.2
M71	Managed	150.9	79	5.7	2.8773	0	0	156.6	81.8
M80	Managed	131.8	68.9	7.9	3.9989	7.4	3.8	147.1	76.8
M94	Managed	222.8	116.6	2.3	1.1955	5.1	2.6	230.1	120.4
M100	Managed	148.4	77.6	12.8	6.4736	4.1	2.1	165.2	86.2
M102	Managed	122.4	64.1	67	3.4906	14.9	7.6	144.3	75.2
M109	Managed	192.8	100.9	4.7	2.343	9.7	5	207.2	108.2

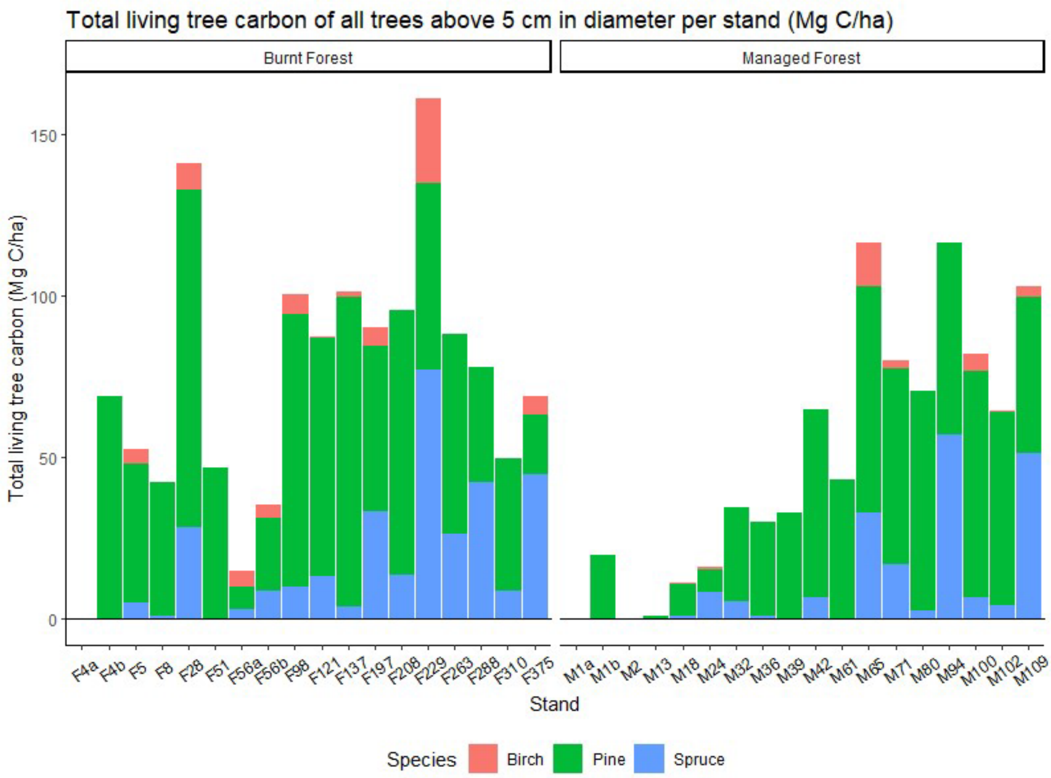

Figure 2. Total living tree carbon per stand in each chronosequence for all trees larger than 5 cm in diameter at breast height. The stand names in the X-axis refers to stand age for each chronosequence as the name for each stand was given based on the type of chronosequence (F = Fire, M = Managed) and the stand age since the last major disturbance. The colours represent different tree species present in each stand.

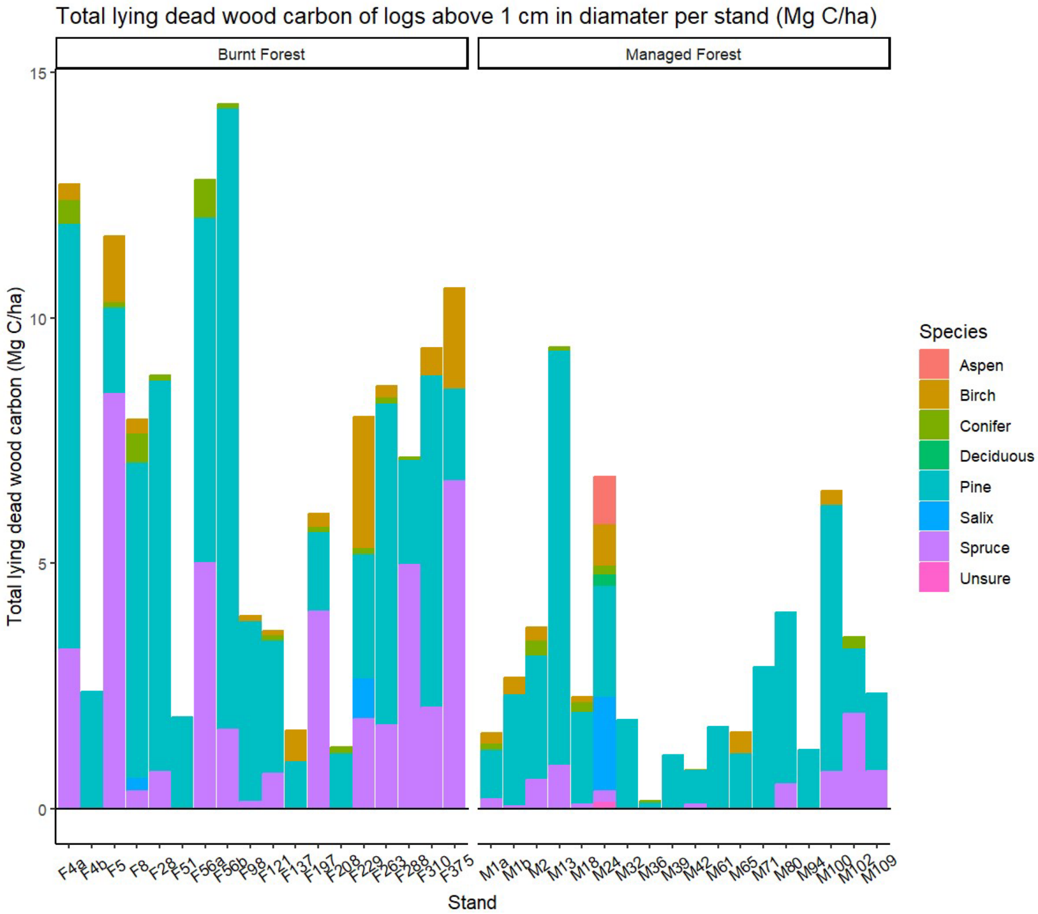

Figure 3. Total lying dead wood carbon per stand in each chronosequence for all lying dead wood larger than 1 cm in diameter at the small end. The stand names in the X-axis refers to stand age for each chronosequence as the name for each stand was given based on the type of chronosequence (F = Fire, M = Managed) and the stand age since the last major disturbance. The colours represent different tree species present in each stand.

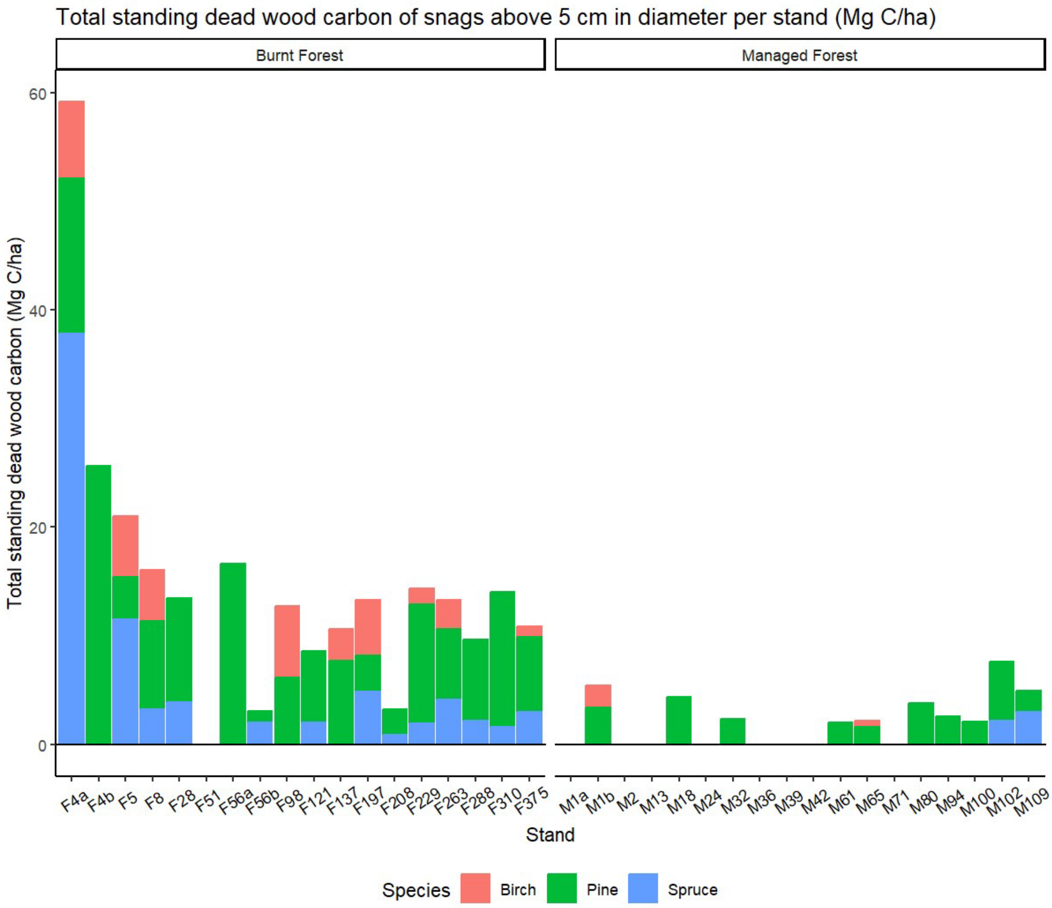

Figure 4. Total standing dead wood carbon mass per stand in each chronosequence for all dead trees larger than 5 cm in diameter at breast height. The stand names in the X-axis refers to stand age for each chronosequence as the name for each stand was given based on the type of chronosequence (F = Fire, M = Managed) and the stand age since the last major disturbance. The colours represent different tree species present in each stand.

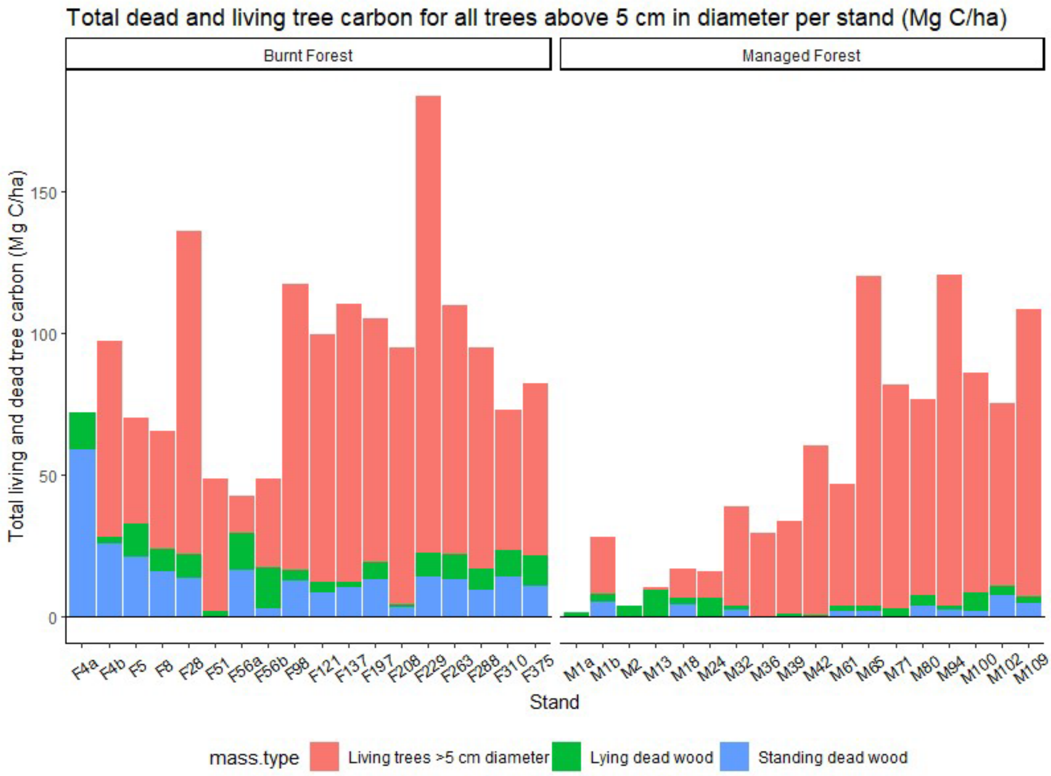

Figure 5. Total dead and living tree carbon stocks per stand in each chronosequence for all standing trees larger than 5 cm in diameter at breast height and lying dead wood larger than 1 cm at the small end. The stand names in the X-axis refers to stand age for each chronosequence as the name for each stand was given based on the type of chronosequence (F = Fire, M = Managed) and the stand age since the last major disturbance. The colours represent the different carbon stocks making up the total carbon stocks in each stand.

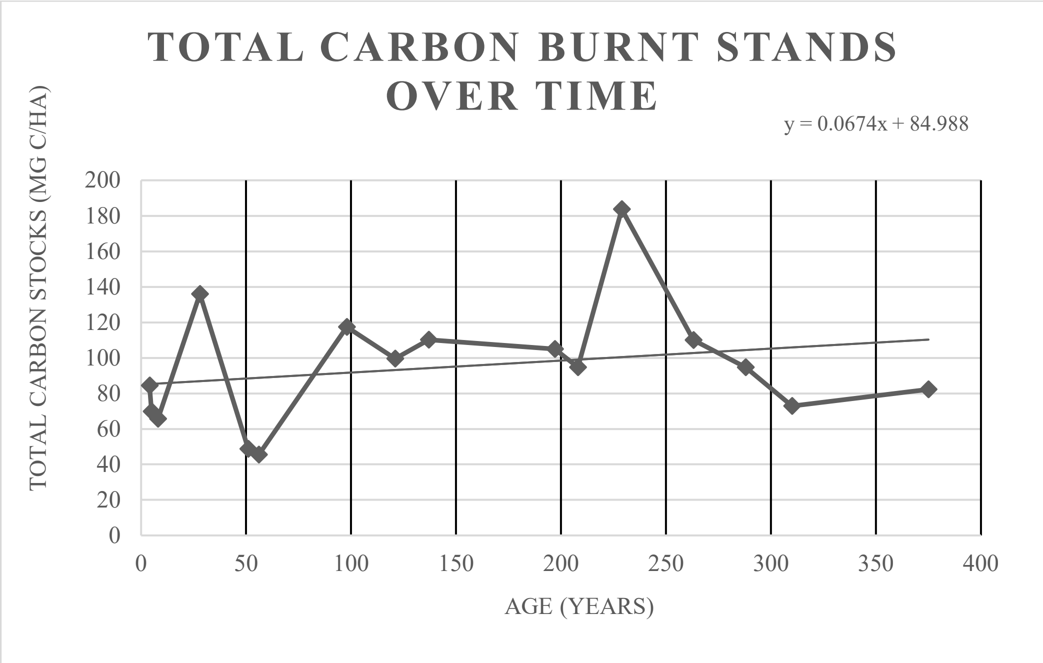

*Figure 6. Calculated average carbon stocks over time for the entire burnt stand chronosequence. The average carbon stocks over time were calculated using the regression line Y = 0.0674 * X + 84.988.*

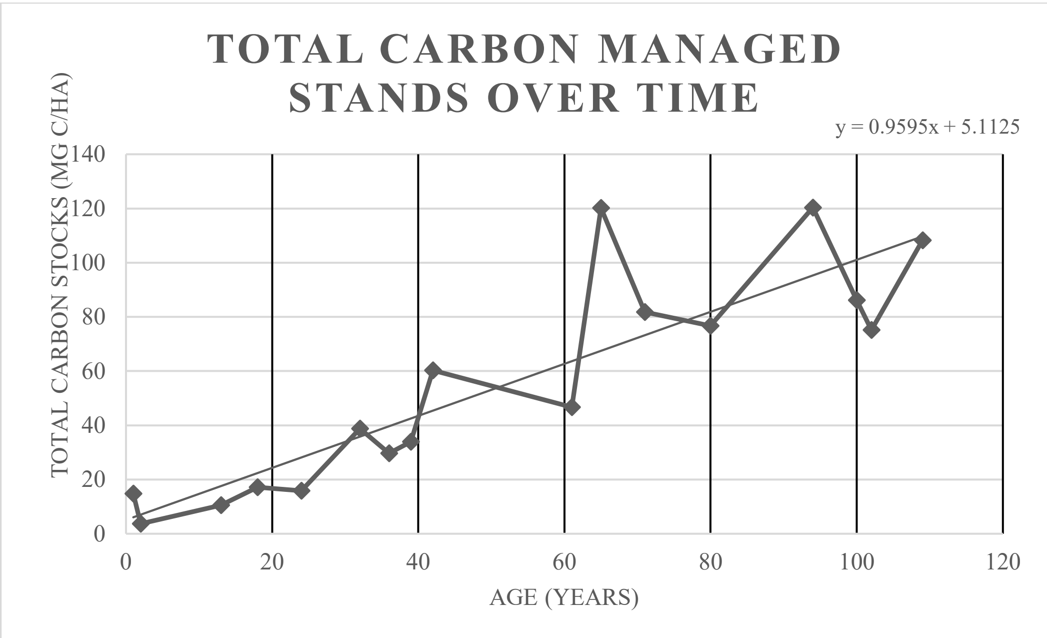

*Figure 7. Calculated average carbon stocks over time for the entire managed stand chronosequence. The average carbon stocks over time were calculated using the regression line Y = 0.9595 * X + 5.1125.*

Table 5 shows the results of the linear regression equations used to estimate the average total caron stocks for each chronosequence at different times. At age 100, the managed stands had sequestered an average of 53.1 Mg C /ha, with the burnt stands having sequestered an average of 88.4 Mg C/ha. At age 200, the managed stands would have sequestered an average of 149 Mg C /ha following the linear regression, with the burnt stands sequestering an average of 95.1 Mg C/ha. At age 300, the managed stands would have sequestered an average of 245 Mg C /ha following the linear regression, with the burnt stands sequestering an average of 101.8 Mg C/ha.

Table 5. Results of the linear regression equations to estimate average total carbon stocks for each chronosequence at different times. X denotes the halfway point at which the average total carbon stock was calculated from.

Stand type:	***Burnt stands***	***Managed stands***
Linear regression equation:	Y = 0.0674 * X + 84.988	Y = 0.9595 * X + 5.1125
Stand age (years)	**Average carbon stock (Mg C /ha)**	**Average carbon stock (Mg C /ha)**
100 (X = 50)	88.4	53.1
200 (X=150)	95.1	149
300 (X = 250)	101.8	245

3.3. Model comparisons of carbon stocks

Table 6 shows the residual distribution and input variables responses of the linear regression models for the total carbon stocks and each separate part of the total carbon stocks individually. The significance level of each linear regression model was 0.05. Regarding the input variables, stand type refers to whether the stand was a managed or burnt stand. Age refers to the naming of each stand and time since its last major successional disturbance (clearcut harvest or forest fire). Species refers to the different tree species found in each stand. The interactive effect between the stand type and time since disturbance was also analysed.

Table 6. The statistical models applied to each carbon stocks and the P-values of each input parameter. Stand type refers to whether the stand was a managed or burnt stand. Age refers to the naming of each stand and time since its last major successional disturbance (clearcut harvest or forest fire). Species refers to the different tree species found. Stand.type:sqrt(age) refers to the interaction effect between the type of stand it was and the time since its last major successional disturbance.

	Response variables and their statistical significance		
Total carbon stocks Anova Table	lm = log(carbon.ton.ha + 0.1) ~ stand type * sqrt(age)		
	Response: log(carbon.ton.ha + 0.1)		
		F value	Pr(>F)
	Stand type	5.1723	0.02382
	Sqrt(age)	6.5033	0.01138
	Stand type*sqrt(age)	16.2045	7.592e-05
Living trees Anova Table	lm = log(carbon.ton.ha + 0.1) ~ stand.type * sqrt(age)		
	Response: log(carbon.ton.ha + 0.1)		
		F value	Pr(>F)
	Stand type	0.0000	0.997141
	Sqrt(age)	11.6520	0.001026
	Stand type*sqrt(age)	5.4371	0.022329
Lying dead wood Anova Table	lm = log(carbon.ton.ha + 0.1) ~ stand.type * sqrt(age)		
	Response: log(carbon.ton.ha + 0.1)		
		F value	Pr(>F)
	Stand type	3.1305	0.07986
	Sqrt(age)	0.6125	0.43566
	Stand type*sqrt(age)	2.1445	0.14619
Standing dead wood Anova Table	lm = log(carbon.ton.ha + 0.1) ~ stand.type * sqrt(age)		
	Response: log(carbon.ton.ha + 0.1)		
		F value	Pr(>F)
	Stand type	28.9978	1.371e-06
	Sqrt(age)	4.4916	0.03835
	Stand type*sqrt(age)	5.7847	0.01938

4. Discussion

The goal with the study was to compare the carbon stocks of living and dead wood in managed and burnt forest stands of different ages since the last final harvest or fire. The study made use of two chronosequences for the burnt and managed forests and the results pointed towards different trends in carbon stocks for each chronosequence. This thesis therefore helps fill a knowledge gap regarding the boreal forest C cycle in Scandinavia. The results from the study will be discussed under the following subheadings, each of which relates to each stated hypothesis.

4.1. How does wood carbon density differ between managed and burnt stands, and how does this influence the carbon stocks?

The wood density for living trees differed by 0.033 g/cm^3 between managed and burnt stands, with the wood density being slightly higher for the burnt stands at 0.453 g/cm^3 compared to 0.42 g/cm^3 for the managed stands. There was no major difference in wood density and the carbon-nitrogen analysis performed on the tree cores showed no significant pattern in terms of wood carbon density for either stand type or tree age. Thus, the decision was made to use the average carbon content of all analysed tree cores, being 52.%, to convert the total living tree biomass results to total carbon. The living tree wood densities were higher than the dead wood densities presented by Sandström *et al.* (2007) and Mäkinen *et al.* (2006), however it is common for wood density to decrease as the wood decays more over time (Fraver *et al.* 2007). The dead wood densities were borrowed from the literature of Sandström *et al.* (2007) and Mäkinen *et al.* (2006) in order to help convert the dead wood biomasses to dead wood carbon stocks. The literature highlighted slight differences in wood carbon densities between tree species and dead wood decay classes. Howevery, every tree core from this thesis had carbon percentages between 49.22% and 52.23% carbon. Also, the wood density did not impact the total biomass calculations as the biomass functions by Marklund (1988) already takes wood density into consideration.

Sandström *et al.* (2007) did show however that although total wood density decreased with increasing decomposition of the wood, the carbon concentrations increased slightly with increasing decomposition. The differences in carbon content were very small however when dead wood of different decay stages were compared. Both this thesis and Sandström *et al.* (2007) found slight differences in wood density depending on the stand age, however the difference in wood density with stand age was not large for either study and the pattern was not completely uniform with stand age. Sandström *et al.* (2007) did find species-specific differences in wood density and carbon content, with Scots Pine wood density and carbon concentrations being higher than that of Norway Spruce. This indicates Scots Pine is more resistant to microbial decomposition than Norway Spruce across decay classes. This thesis only looked tree cores from living Scots Pine when it came to living tree wood density and carbon concentration. Therefore, to make better estimations of wood density and carbon content, tree cores would need to be sampled from all species present in each stand, as well as from both living and dead trees. Better estimations of dead wood decay classes would also be needed to be able to better correct for wood decay and calculate the wood densities and carbon concentrations more thoroughly.

4.2. Do managed forest stands or unmanaged burnt forests store more carbon, and how does it vary over time?

The results of the regression equations in table 5 show that burnt stands have higher average carbon stocks compared to managed stands over a 100-year frame. The burnt stands had stored 88.4 Mg C/ha on average compared to 53.1 Mg C/ha in the managed stands. This shows that after a stand-replacing disturbance over a 100-year time-period, the burnt stand store on average greater amounts of carbon compared to the managed stands. This is in line with studies such as Law *et al.* (2004) and Sharma *et al.* (2013), where both studies found that C stocks were on average larger in forests that had not been managed for timber production. Regarding carbon accumulation, the trajectory differs a lot between managed and burnt forest chronologies, with the managed forest chronosequence increasing by almost 1 Mg C ha^{-1} $year^{-1}$, whereas the burnt chronosequence remains more time-invariant and increases on average by 0.067 Mg C ha^{-1} $year^{-1}$ instead. This shows that managed forests will store a lot of carbon over time, however it would most likely not be as linear of an increase at older stand ages. Although the managed chronosequence in this study does not extend beyond 109 years, there are no direct signs of the carbon stores plateauing over this time period (Fig. 6). Goulden *et al.* (2011) found that that NPP decreases in older forest stands due to increased ecosystem respiration, which supports the idea that the carbon stocks in managed forests would continue to grow as linearly as the linear regression suggested.

There was variation with stand age within the living tree carbon stocks for both chronosequences, though the carbon stocks seem to follow a more linear and stable pattern for the managed forests. Stand type on its own was not a significant explanatory factor for living tree carbon. However, the stand age and the interaction between stand type and stand age did significantly help explain the living tree carbon. As for the total combined carbon stocks, most of the patterns are dictated by the largest pool, i.e. the living tree carbon stocks, with the remaining dead wood carbon stocks having a greater impact on the burnt forests than the managed forests, as that is where dead wood is more prevalent. However, just because the living trees make up a larger portion of the total carbon stocks does not mean that the contribution of dead wood of any kind is insignificant, as carbon stored in dead wood is retained within forest ecosystems far longer in compared to in living biomass that gets removed through logging (Sharma *et al.* 2013). Decomposition of dead wood can take several decades depending on the site conditions and therefore the dead wood carbon continues to make a up a large portion of the total carbon stocks for long even after any mortality-inducing disturbance like a harvest or forest wildfire (Luyssaert *et al.* 2008).

There are great fluctuations in the carbon stocks for the managed chronosequence as well as the burnt chronosequence, which can indicate that the average carbon stocks for either chronosequence can be either over- or underestimated. When you compare different 100-year time-periods (either 100-200 or 200-300 years), the managed chronosequence carbon sequestration over time can no longer be trusted, due to lacking data from stands of those ages and the fact that ecosystem NEP decreases over time in stands older than 100 years old (Pregitzer & Euskirchen 2004; Peichl *et al.* 2022). Luyssaert *et al.* (2008) explains that despite most forests up to the age of 800 having positive carbon balances, the rates at which a forest sequesters carbon from the atmosphere heavily depends on stand age and structure, with old and unmanaged forests usually sequestering less carbon over time (Gundersen *et al.* 2021). Peichl *et al.* (2022) also found that actively managed forests with longer rotation periods of up to 138 years was the sweet spot for serving serve as strong carbon sinks, which could provide greater climatic benefits compared to other management styles. This can support that longer rotation periods in managed forests have the potential to store more carbon than burnt forests over timeframes longer than 100 years, though that is not supported in this thesis. Carbon stocks only showed a decreasing trend for stands older than 250+ years in the burnt chronosequence. This could suggest that previously disturbed stands can plateau and turn into net carbon sources over longer timespans without reoccurring disturbances, however this is not a conclusion that can be drawn from this study, and needs to be studied further. Pregitzer and

Euskirchen (2004) also showed that although living biomass started to plateau at higher stand ages for both managed and unmanaged boreal forests, there can be major stand-specific differences and fluctuations and overall carbon stocks do increase with stand age.

There are however differences in the carbon stock starting points, particularly for the burnt forest stands. Whilst the managed forest stands had been established using traditional rotational forestry and started from a clearcut state, the fire disturbance does not indicate a starting point for stand development for the burned stands as they were established since before the disturbance. This indicates that most of the forest fires in the burnt chronosequence were not stand-replacing and did not have as big of an impact on total living biomass and stand NPP as the final harvests did. Goulden *et al.* (2011) showed that carbon stocks in older stands were generally less impacted by fires than in younger stands and that recovery after the fire was relatively rapid. There is a knowledge gap regarding the stand history, prior to silvicultural measure or general state of each burnt stand before the disturbance, as well as the intensity of the fire or resulting biomass loss for this study. Therefore, there are many unknown factors when predicting carbon stocks over time for either chronosequence. If the burnt stand carbon stocks followed more of a linear pattern, a linear regression prediction as seen in Fig. 7 would be more suitable. However, the flat slope from Fig. 7 shows that the regression equation has an insignificant increase over time for the burnt stands. There is a lot of variability over time in both chronosequences, however, some results are difficult to compare. For example, there was no knowledge about the fire patterns and intensities in the burnt stands, making it difficult to know how much of the total woody biomass has burnt and decayed over time, and what successional patterns were created by the fires. This was especially so for the burnt stands varying a lot in terms of how much standing and lying dead wood there was, as seen in figures.

4.3. How does dead wood affect the development of carbon stocks between managed and burnt stands?

There was more dead wood in the burnt stands compared to the managed stands, but the dead wood carbon stocks also varied depending on the specific stand and the age of the stand and stand age. There was not much standing dead wood recorded in the managed forests, with highest quantities occurring in the older stands. As for the burnt forests, the standing dead wood is evenly abundant for most stands, with one of the more recently burnt stands standing out with most of its total carbon stocks consisting of standing dead wood. The stand type and age of the forests are significant explanatory variables for the standing dead wood. The stand type seems to significantly affect the total carbon stocks for the lying dead trees, but neither stand age nor the interactive effect between stand age and stand type help significantly explain the total lying dead wood carbon stocks. As for the lying dead wood, it is not as prevalent as the standing dead wood in the burnt stands, though there is more lying than standing dead wood in managed stands. Some reasons as to why dead wood was more prevalent in the burnt stands could be due to the there being no active removal of dead wood in burnt or unmanaged stands compared to managed forests. Studies have found that total dead wood amounts are heavily impacted by forestry operations such as thinnings and final harvests, resulting in less dead wood in managed forests and much more dead wood in unmanaged forests (Fridman & Walheim 2000). This can be seen in this thesis, as primarily standing dead wood is uncommon in the managed chronosequence compared to the burnt chronosequence. This can also be a result of the forest fires removing much less biomass from the burnt chronosequence compared to the harvests in the managed chronosequence. Instead of strictly removing and extracting the carbon of the living trees, much of that carbon has instead changed carbon pool and become part of the dead carbon stocks instead. This is clear when looking at a stand like F4a from the burnt chronosequence, where there was no living carbon, and the absolute majority of the carbon stocks came from the standing dead trees. In comparison, most of the dead carbon resulting from harvesting operations find other uses in wood-based products and help store the carbon outside the forest ecosystems.

Ultimately, the burnt stands did have larger stocks of both standing and lying dead wood compared to the managed stands, where the standing dead wood makes up a much larger portion of the total carbon stocks in the burnt stands compared to the managed stand (Fig. 5). Halme *et al.* (2019) found that coarse woody debris was much more prevalent in natural forests compared to heavily managed and denoted that birch was the dominant species in terms of dead wood volume despite their study area being spruce-dominant. From our chronosequences, both spruce and pine carbon stocks were higher for either type of dead wood compared to birch dead wood (Fig. 3 and 4), though Halme *et al.* (2019) explains that the lack of spruce dead wood volume could be due to their study area's previous management methods of slash-and-burn. Hagemann *et al.* (2009) found that final harvest operations do generate a large amount of lying dead wood, but that was most of it decompose 34-36 years after the harvesting operation. The results from our study show a similar, albeit weak pattern, as standing dead wood was high in some managed stands until after 32 years of age, but then decreased at older stand ages (Fig. 4). This increase in standing dead wood in our managed forest stands could be due to an increase in self-thinning or simply due to site-to-site variation. The study by Hagemann *et al.* (2009) also found that burnt stands produced a much larger amount of standing dead wood, which can also be seen in our study, with most burnt stands having higher carbon stocks of standing dead wood compared managed stands. One stand in particular, F4a, had a total carbon stock of 71.8 Mg C/ha, with standing dead wood carbon stocks constituting the absolute majority of it, being 59.1 Mg C/ha standing dead wood carbon (Table 4). Although the F4a stand was an outlier in terms of how much dead wood there was compared to all of the other stands, it highlights the impact that large forest fires can have on the total carbon stocks in a forest.

4.4. Sources of error and further work

This study did not include all carbon pools present in the boreal forest ecosystem and cannot therefore classify as a complete carbon stock study. Carbon pools that were excluded from this study due to missing data and time constraints were soil carbon, trees smaller than 5 cm in DBH, mosses and brushes. As this thesis focused on the mature forest stands and dead wood, the analyses give only explain the above ground carbon in the wood and do not explain managed vs burnt forest ecosystems in total. This study did also not account for how much of the dead carbon that was extracted from forestry operations would have contributed to the substitution effect of replacing fossil fuels with bio-based fuels and products. Leskinen *et al.* (2018) did suggest however that wood-based were able to reduce average emission levels by approximately 1.2 kg C per 1 kg C from wood products. Therefore, a big gap in knowledge that this thesis did not address was the substitution effect that the carbon from the extracted harvested wood in the managed chronosequence would have provided.

Another key factor not researched in this thesis is the soil nutrient availability and how that might have impacted the total carbon stocks. Fernandez-Martinez *et al.* (2014) found that nutrient availability in forests is a large controlling factor for GPP and NEP and needs be considered when calculating carbon balances. This thesis does not consider or research forest carbon fluxes or carbon balances either and factors like NEP and respiration has not been taken into account. Carbon stocks are a snapshot of the total amount of forest carbon at that given moment, but if you want to make long-term predictions regarding how carbon stocks develops and what kinds of forest sequester more carbon over time, a more thorough analysis utilising trends in NEP and carbon fluxes would help give better estimates. The regression equations from Figures 6 and 7 that predict average carbon stocks over time across the chronosequences could be improve by correcting for changes in NEP and carbon sequestration over time. There are also improvements and corrections that can be made based on climatic differences such as temperature and different weather factors, as those heavily impact the productivity and respiration of forest stands, which in turn impacts carbon sequestration and total carbon stocks. It was difficult finding tapering-equations for standing dead wood given the collected in the field work. The most applicable tapering-conversion for standing dead wood was made by Ducey and Fraver (2018) and assumes a conic-paraboloid volume with the top-stem basal area (A_L) equalling 0. This is a large source of error as there could be better optimised volume equations that and that more accurately corrects the volume for the standing dead wood based on their form-factors.

There was no prior knowledge about the history of either the managed or the burnt stands. This was most important for the burnt stands since the stand structures and total carbon stocks varied a lot between the stands and did not follow a linear trend in increasing carbon stocks. The burnt forest stands peaked in total carbon stocks around 200 years with many fluctuations both before and after that time-period, and therefore it might be more logical to fit non-linear equations to the data to predict total carbon stocks over time.

Future studies on this topic should include most if not all carbon stocks found in the ecosystem. Accounting for the the substitution effect of wood products from managed forests stands and the impact that soil fertility has on the total carbon stocks is also important to get a complete picture of the boreal forest carbon stocks. Prior management and fire history should be accounted for, especially for comparing the tree-mortality induced by the forest fire compared to that of a final harvest. Better calculations of dead wood carbon stocks could also be implemented by more accurately noting down decay states, taking measurements of both large-end and small-end diameters and by applying more robust tapering equations for the standing dead wood. Finally, more direct comparisons between different management systems and different forest types could also be made to give a global comparison and estimation of carbon stocks.

5. Conclusions

The conclusions of the thesis are that burnt forests store more carbon than managed forest stands after a major disturbance when looking at comparable time-spans of approximately 100 years, with most of that stored carbon coming from the living tree biomass in both chronosequences. This shows that burnt forests have on average a higher potential to store more carbon over 100 years after a major disturbance. However, managed forest stands have more linear increases in carbon stocks over time compared to burnt forests which are more time-invariant, which indicates the potential for managed forests to store more carbon over time periods longer than 100 years. The wood density and carbon content for living trees was slightly higher in burnt forests compared to managed forests, but the difference was so small that the average carbon content was used for calculating the living tree carbon stocks. Both standing and lying dead wood made up a much larger portion of the total carbon stocks in burnt forest stands compared to managed forests, however there were no direct trends over time for those results as the total amount of both standing and lying dead wood varied depending on the state of each stand. Topics that could be expanded more upon in the future to improve upon these results are differences in management and fire history, influences of soil nutrient availability and inclusions of soil and shrub carbon into the carbon stock calculations. The equations for calculating the total carbon stocks could probably be improved further as well, particularly for the standing dead wood. The substitution effect of wood-based products should also be included in future analyses in order to get a complete comparison of total carbon pools and climate change mitigation.

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