The greenhouse gas mitigation potential of wood-based jet-biofuels – A case study for Western Norway

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Master Thesis in Climate Change Management

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Preface

Writing this thesis revealed to me how difficult, frustrating, intriguing and mostly frustrating the process of science can be. I want to thank Carlo Aall for introducing me to this important subject. Thank you, Bart Holtsmark, for providing me your own work, which I absolutely depended upon. At last, a big thanks to Valeria Jana Schwanitz for your valuable knowledge and patience. This thesis would have ended up a gigantic mess without your effort.

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Abstract

The Norwegian aviation industry has proposed substituting jet-fuels with biofuels based on forest biomass. This can potentially contribute to mitigating the expected increase in greenhouse gas emissions from aviation activity toward 2030. However, retrieving the biofuel feedstocks requires harvesting of forest, and thus the climate system is affected through changes in forests biogeophysical and biogeochemical climate functions. These effects have not been accounted for in assessments of wood-based biofuels in Norwegian aviation, although climate change mitigation is the underlying reason for the proposed strategy. This thesis assesses how the inclusion of one of the forests most important climate functions, the role as a carbon pool, can affect the climate change mitigation potential of wood-based biofuels in Western Norwegian aviation. For this purpose, a simple forest model for Western Norwegian spruce forest is developed, and different scenarios for fuel-use in the aviation sector are explored. When 30 % of jet-fuel sales are gradually replaced by biofuels between 2019 and 2030 (BIO-JET scenario), this leads to 0.6 Mt less of cumulated carbon dioxide equivalent emissions over the period, compared to business as usual. However, considerable temporary reductions in the carbon stored by the forest results in overall 133 % higher temperature response from GHG emissions in 2050 when biofuels are utilized. The warming effect is still 40 % higher in 2100 and will approach 14 % lower towards 2200. In comparison, reducing fuel-use by 30 % between 2019 and 2030 (AVI-RED scenario) leads to 2 Mt less of cumulated carbon dioxide equivalent emissions over the period, and 32 % lower temperature response compared to business as usual. The scenario analysis indicate the potential for mal-mitigation by introduction of wood-based biofuels in the aviation sector, and shows that policy measures targeting demand reduction are very important. This reveals a need for more comprehensive research on the effects of wood-based biofuels, and for the potential options in Norway for utilizing forest management to cope with climate change.

Sammendrag på norsk

Den norske luftfartsindustrien har foreslått å erstatte jetdrivstoff med biodrivstoff basert skogbiomasse. Dette kan potensielt bidra til å begrense den forventede økningen i klimagassutslipp fra flytrafikk det kommende tiåret. Å hente ut biomassen vil dog innebære høsting av skog, og fører dermed til konsekvenser for klimaet gjennom endringer i skogens biofysiske og biokjemiske påvirkninger på klimasystemet. Disse effektene har ikke blitt medregnet i foreløpige analyser av innfasing av skogbasert biodrivstoff i flysektoren, selv om den underliggende begrunnelsen for det foreslåtte tiltaket er å hindre klimaendringer. Denne masteroppgaven undersøker hvordan inkludering av en av skogens viktigste funksjoner i klimasystemet, rollen som karbonlager, påvirker klimapotensialet for skogbasert biodrivstoff i den Vestnorske flysektoren. I denne hensikt ble en enkel modell for Vestnorsk granskog utviklet, og scenarier for drivstofforbruk i flysektoren ble utforsket. Når 30 % av omsatt jetdrivstoff blir erstattet gradvis av biodrivstoff mellom 2019 og 2030 (BIO-JET scenario), fører dette til 0.6 Mt lavere kumulerte karbondioksid utslipp over perioden, sammenlignet med normal virksomhet. Tiltaket vil derimot medføre betydelig midlertidig redusert karbonlagring i Vestnorsk granskog, og dette resulterer i totalt 133 % høyere temperatur respons i 2050, når biodrivstoff blir benyttet. Oppvarmingen er fortsatt 40 % høyere i 2100, og nærmer seg 14 % lavere i 2200. Til sammenligning vil gradvis redusert bruk av jetdrivstoff, med opptil 30 % mellom 2019 og 2030 (AVI-RED scenario), føre til 2 Mt lavere kumulerte karbondioksid utslipp over perioden, og 32 % lavere temperatur respons, sammenlignet med normal virksomhet. Scenario analysen indikerer at innfasing av skogbasert biodrivstoff i luftfart potensielt kan virke mot sin hensikt, og peker på viktigheten av tiltak rettet mot å redusere etterspørsel. Dette viser et behov for mer omfattende forskning på effektene av skogbasert biodrivstoff, og på potensialet for å benytte skogforvaltning for å håndtere klimaendringene.

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1. Introduction

Emissions from Norwegian aviation could increase in the coming decades despite ambitious mitigation goals in the transport sector. Strong growth in inland and international air traffic from Norway is expected toward 2040, and no specific goals for emission reductions in the aviation sector are yet implemented in national policy. To combat emission growth, the state-owned aviation company Avinor has proposed introducing sales-share requirements for biofuels in aviation. The share of biofuels in jet-fuel sales are sought to reach 30% within 2030. Biofuels are considered carbon-neutral in the energy sector, and they are therefore projected to lower emissions in inland flight traffic by 0.34 Mt carbon dioxide equivalents between 2021 and 2030.

The biofuels are proposed to be based on woody biomass feedstock and harvesting of forest is required for producing the fuels. However, forests play an important role in the climate system by influencing the greenhouse effect and the surface reflectivity of the planet. The benefits of replacing fossil fuels by biofuels can therefore not be calculated straightforward, but the complex interlinkages need to be accounted for. The effect that harvesting of wood could have for the forest-climate functions is not considered with traditional greenhouse gas accounting methods, but they are important for the climate impacts of wood-based biofuels.

Of the most important climate functions are the forests role as a carbon pool. Forests sequester carbon dioxide from the atmosphere, by locking the carbon in living biomass and organic soil. Burning biofuel releases this carbon to the atmosphere, thereby increasing the atmospheric carbon dioxide concentration temporarily, and changing the radiative forcing, until the carbon is re-sequestered by the growth of new biomass. In this context, it is important to note that it can take decades to over a century before the pre-harvest carbon levels are restored in boreal forest. Disagreement regarding methodological approaches between studies that include temporary carbon loss from the forest, leads to widely different conclusions on the mitigation potential of wood-based biofuels. However, harvesting forest is generally found to lead to increased radiative forcing short-term, and this breaks the equivalency between carbon-neutrality and climate neutrality for wood-based biofuels.

The aim of this thesis is to assess the greenhouse gas mitigation potential of wood-based jet-biofuels in Western Norwegian aviation. The assessment accounts for emissions from biofuel production, as well as the loss of forest carbon, compared to emissions from conventional aviation fuel. The forest carbon loss is accounted for with the carbon sequestration parity approach, using a dynamic model of the Western Norwegian spruce forest. Different scenarios for fuel-use in the aviation sector are explored. The results of the scenario analysis are discussed in the context of the current policy plans and compared with other studies.

1.1. Structure of the report

Chapter 2 outlines the role of the Norwegian aviation for mitigating climate change by introducing sectoral plans for the deployment of bioenergy. Furthermore, the literature on the role of forests in the climate system with a focus on Norway is reviewed. The chapter concludes with an overview on how methodological choices lead to conflicting conclusions on the mitigation potential of wood-based biofuels of comparable bioenergy systems.

Chapter 3 explains the forest model used for calculating temporal carbon loss, and climate impacts of emissions.

Chapter 4 introduces the aviation fuel scenarios. In addition to the baseline, two explorative scenarios are developed.

Chapter 5 presents the results from the scenario analysis. The results are discussed with a focus on result-driving assumptions and choices. The results are furthermore compared with other studies.

Chapter 6 concludes and suggest on how this study can be taken further.

2. Literature review

2.1. Biofuels in the Norwegian aviation sector

Aviation constitute a relatively large share of Norwegians climate footprint. Air travel in Norway was responsible for about 2.5 % of greenhouse gas (GHG) emissions in 2017 (1.3 of 52.7 Mt CO_2e) (SSB 2017). However, this estimate only includes flights within Norway's borders. According to a study by Avinor, a state-owned limited company under the Norwegian Ministry of Transport and Communications, GHG emissions from all flights carried out by Norwegians sum up to 3.6 Mt CO_2e (Avinor 2017). Civil aviation emitted 859 Mt CO_2 globally in 2017 (IATA 2018). Hence, Norwegians per capita aviation emissions are today around 6 times higher than the global average.

In addition to carbon dioxide, air-plane exhaust cause other types of emissions and lead to indirect effects in the atmosphere that are not accounted for in the national inventory (Miljødirektoratet 2019). Among those are water vapor, nitrogen oxide (NOx) and particulate matter (e.g. sulfur dioxide). Emissions of water vapor induce direct radiative forcing by itself, and additionally form contrails and cirrus clouds that induce substantial extra forcing. Emissions of NOx also play a role, because its effect on the atmosphere leads to reduction in methane, but concurrently increases ozone which is also a potent climate gas. The net effect of NOx emissions is increased warming in the short term, followed by cooling before even out after about 70 years (Lund et al. 2016, Lund et al. 2017). Small emissions of soot also contribute to warming by absorbing radiation, while sulfur dioxide and organic carbon reflect solar radiation and leads to cooling. Particle and sulfur dioxide emissions can also serve as condensation nuclei's and increase formation of large cirrus clouds, but the climate impact of this process is currently too uncertain to quantify, though it might be substantial (Lund et al. 2016).

The extra climate impacts of air travel are short lived compared to CO_2, and the regional potential can vary starkly (Lund et al. 2017). However, they are generally found to contribute a large extra warming potential on short timescales. An extra warming factor between 1.2 and 1.8 has been recommended added to CO_2 equivalents to account for the additional climate effects, significantly increasing the climate impact of aviation emissions even on a 100 year timescale (Lund et al. 2016). A publication by Peters et al. (2011) shows that long distance air travel constitute a large share of Norwegians climate impact from transportation when short-lived climate forcers and cirrus cloud formation is taken into account.

Aviation is in rapid growth and emissions from Norwegian inland trips rose by 55 % between 1990 and 2008 (Lund et al. 2016). The number of flight travels within Norway is expected to grow by 33 %, while trips to Europe and international trips from Norway are expected to grow by 107 % and 147 %, respectively, toward 2040 (Larsen 2015, Avinor 2017). Fuel-use is a major cost in the aviation industry, and therefore aviation companies are consistently working toward increasing airplane fuel efficiency. However, efficiency gains have been decreasing sharply since the 60s, and the trend indicates the emission efficiency gains will be below 1 % in the future, because technical efficiency limits are being approach (Peeters et al. 2016). Considering the underestimated climate impacts and the future growth projections, decarbonizing aviation should be prioritized in national climate change mitigation strategies.

Climate change mitigation in aviation has both international and national dimensions. International air travel emissions are mainly addressed by the International Civil Aviation Organization (ICAO), a specialized agency under the UN. ICAO adopted a global Carbon Offsetting and Reduction Scheme in International Aviation (CORSIA), with the aim of securing carbon-neutral growth in global aviation from 2020. The scheme requires operators to monitor and report their emissions, and purchase emission units to offset emission growth, from 2019 (IATA 2018). According to Warnecke et al. (2019), aviation might become the largest purchaser of offsets after 2020, but if the CORSIA program will lead to actual emission reductions is uncertain and will depend on robust eligibility criteria's for carbon offset credits. Norway is currently included in the European cap and trade scheme which targets international air travel within EU's borders. 80% of international flights are covered by the scheme (Miljø-direktoratet 2018). Norway is also a member of the European Civil Aviation Conference (ECAC). In this way, Norway is to varying degree, together with the other member states, involved in European research and emission reduction programs aimed at increasing aircraft fuel efficiencies and finding alternative fuels(Norway-state-action-plan 2016). However, the effectiveness of these measures is uncertain.

No specific emission goals have been introduced for Norwegian aviation. However, within 2030 Norway is committed to reduce the national inventory emissions by 40% from 1990 level and transport, being the sector with the largest GHG footprint, has to take the lion's share of the cuts (Gullberg and Aakre 2015, miljødepartementet 2018). The four state owned transportation companies Avinor, Jernbaneverket, Kystverket and Vegvesenet together proposed reducing emissions in the sector by 50% from 2014 to 2030 to contribute to the National Transport Plan (NTP) (Avinor 2016, Samferdselsdepartementet 2017). While the proposed amount of reductions in the aviation sector has not been specified, the document proposes the following targets.

Two significant targets in the NTP are 1) the phasing-in of biofuels, and 2) the phasing-in of electric airplanes. Additionally, measures like improved air traffic management and more efficient flight operations, are also mentioned in the state action plan for ICAO (Norway-state-action-plan 2016). Electric airplanes can potentially be feasible for short domestic trips first within 10-15 years, according to Avinor, so the short-term emission reduction will mainly happen with biofuels. Biofuels are proposed to be phased in with a sales-share requirement (Avinor 2016, Samferdselsdepartementet 2017). The target share of biofuels in the fuel market shall gradually increase, starting with 1% of fuel sales in 2019, reaching up to 30% in 2030. This sales-share requirement for biofuels in inland flight traffic is projected to reduce emissions by 0.34 Mt CO_2e between 2021 and 2030 (Miljødirektoratet 2018).

The biofuels are sought to be of advanced types, as the EU is moving away from first generation biofuels (Creutzig et al. 2015, Samferdselsdepartementet 2017). However, many challenges exist for the different alternatives, such as technological challenges for alga-based-, and feedstock availability for waste-based biofuels. Biofuels based on woody biomass were therefore deemed the most promising option for Norwegian aviation through studies on behalf of Avinor and the Norwegian aviation industry (Trømborg et al. 2012, Killingland 2013, Rambøll 2013, 2017). Particularly the use of biomass from slash and other residuals, which commonly are not used as building materials, are emphasized. These studies also contributed to the knowledge base in an impact assessment conducted by the Norwegian Environment Agency, investigating the phasing-in of 1 % aviation biofuels in 2019 (Miljø-direktoratet 2018).

Killingland (2013) explores two potential technologies most suitable for wood-based biofuel production in Norway. One is based on the Fischer-Tropsch process and the other on biochemical conversion via the alcohol to jet (ATJ) pathway. The former relies on thermochemical conversion by gasification of wood into syngas. This mixture of carbon monoxide and hydrogen is then converted with the Fisher-Tropsch technological process into hydrocarbons applicable as aviation fuel. A blend-in of 50 % in regular jet-fuel has been approved for this fuel type, but the production process is difficult due to high capital investment costs and technological challenges (Karatzos et al. 2014). For the ATJ technological process, biomass is first treated with enzymes to break down the wood, before fermentation into ethanol proceeds. Due to the low energy and high oxygen content of ethanol compared to jet-fuel, the ethanol must further go through dehydration, oligomerization and hydrotreatment to be considered a drop-in biofuel (Rambøll 2013, Han et al. 2017). This fuel type was certified for 50 % blend-in in conventional jet-fuel in 2018 (Kunkel 2018). All in all, Rambøll (2013) concluded that forest-based jet-biofuels have the potential to replace 30 % of fossil jet-fuel sales in Norway by 2030 (Rambøll 2013, 2017).

2.2. The role of forests in the climate system and the Norwegian forest

Trees sequester and store carbon from the atmosphere, giving forests a crucial role in regulating the atmospheric CO_2 concentration. Terrestrial sinks sequester close to 30 % of anthropogenic CO_2 emissions, most of which is due to forest growth (Sandro Federici 2017, Scott et al. 2018). The carbon stored in wood typically constitutes 50 % of the dry weigh, and the world's forests store around 420 billion metric tons carbon in the living biomass (Lamlom and Savidge 2003, Cunningham and Cunningham 2010). Additionally, large amounts of carbon are stored in litter that ends up on the forest floor and in the soil. The litter can further be broken down by fungi or get bound to minerals, thereby forming stable carbon materials that can last for several thousand years as soil organic content (Fontaine et al. 2007, Clemmensen et al. 2013) The combined storage in tropical, temperate and boreal forest soils amounts to 704 Gt C (Lal 2005).

Forestry has a large influence on the carbon storage potential of forests. Deforestation is a major reason for land-use changes climate impact, by permanently reducing the carbon in the living biomass as well as soil carbon by up to 25 – 50 % (Scharlemann et al. 2014). Indeed, from deforestation and land-use change originated about one third of anthropogenic CO_2 emissions since 1750, contributing about 12 % of annual emissions today (Smith 2014). Regular harvesting does also reduce the carbon in living biomass, but only temporarily, and can also affect the organic soil carbon (Achat et al. 2015).

Norway is part of the boreal forest belt, which stretches around the Northern hemisphere through Russia, Scandinavia and Canada. Around 38 % of Norway's main land is forest covered, of which 42 % is deciduous broadleaf and 58 % conifer forest (Majasalmi et al. 2018). The land area stretches from 57 to 81 degrees latitude North, with coldest climate in the north, wet milder climate in the south west and cold drier climate in the east. The height of the forested areas varies by up to 1000 m at the most, and much of the forest areas are tilted in different directions and angles, which affects the incoming sunlight and precipitation. The total productive forest volume constitutes about 784 Mt dry -matter (Løken et al. 2012), equivalent to approximately 392 Mt carbon. Recalculated to CO_2 equivalents, the carbon currently locked in the above-ground biomass in productive forest amounts to around 27 times the emissions ascribed to Norway each year. However, the carbon storage in forests also encompasses the soil. Boreal forests are generally less productive than temporal and tropical types, and the cold climate in areas where boreal forests are located leads to slow decomposition of dead wood and litter (Clemmensen et al. 2013). As a consequence, around 1 800 Mt C is bound in Norwegian forest soils, thus forest soils contain around 80 % of the total forest carbon in Norway (Grønlund et al. 2010, Strand et al. 2016).

Deforestation changes around 58 square kilometers of forest land in Norway every year. The resulting loss of forest carbon leads to emissions of 2.6 Mt CO_2, almost 5 % compared to the national inventory emissions, when uncertain soil carbon loss estimates are included (Breidenbach et al. 2017). The annual harvest volume amounts to around 11 million cubic meter wood, leading to temporal reductions in forest carbon, but also storage of carbon in building materials and increased growth in new forest (Trømborg et al. 2012). The Norwegian forests are gaining volume due to a combination of regrowth of old farmed land following urbanization, heavy spruce planting in the 60s, warmer climate and CO_2 fertilization (Amundsen 2014). Overall the forest act as a strong carbon sink, sequestering carbon at a level near half the national inventory emissions every year, and this is expected to continue for several decades (Finansdepartementet 2017).

Forests have many additional functions in the climate system to storing carbon. For example, the presence of a forest can increase evaporation in an area through the process of evapotranspiration. Evaporation requires uptake of latent heat, resulting in a local cooling effect. The latent heat is released when the water vapor precipitates again, causing local warming. Additionally, forests emit biogenic volatile organic compounds (BVOC). These are oxidized in the troposphere, thereby changing the oxidative capacity of the atmosphere, in turn leading to changes in the concentration of the GHGs methane and ozone. Several effects arise, but the strongest impact is a strengthening of the greenhouse effect, leading to increased warming from boreal forest (Scott et al. 2018). However, the BVOCs also form aerosols that can reflect incoming solar radiation, which leads to cooling. Aerosol production, and the transpiration process, are both also important for weather and cloud formation. Spracklen et al. (2008) found that boreal forests can double the concentration of cloud condensation nuclei's, leading to increased cloud cover that reflects short wave radiation and leads to cooling. These effects, in addition to changes in wind patterns, are also part of the reason why forests are important in the hydrological cycle. Many indirect effects can arise from changes in hydrology, one being nutrient runoff into water which can lead to algae bloom and uptake of CO_2 in the ocean.

Forests also directly affects the albedo of the earth surface. Forests normally have darker color than the ground or more sparse types of vegetation, thus they cause warming by taking up a larger share of the incoming solar radiation than the alternative surface cover. The albedo effect is very important in Norway because the forest can mask the seasonal snow cover, drastically reducing the reflectivity of the surface in the winter and spring. Generally, deforestation in boreal forest is found to cause cooling because the increased reflectivity of the land surface outweighs the radiative forcing from increased CO_2

concentration (Bonan 2008, Hallgren et al. 2013, Scott et al. 2018). All the effects mentioned here are very variable depending on local geographical conditions. Figure 1 illustrates some of forests important functions in the climate system.

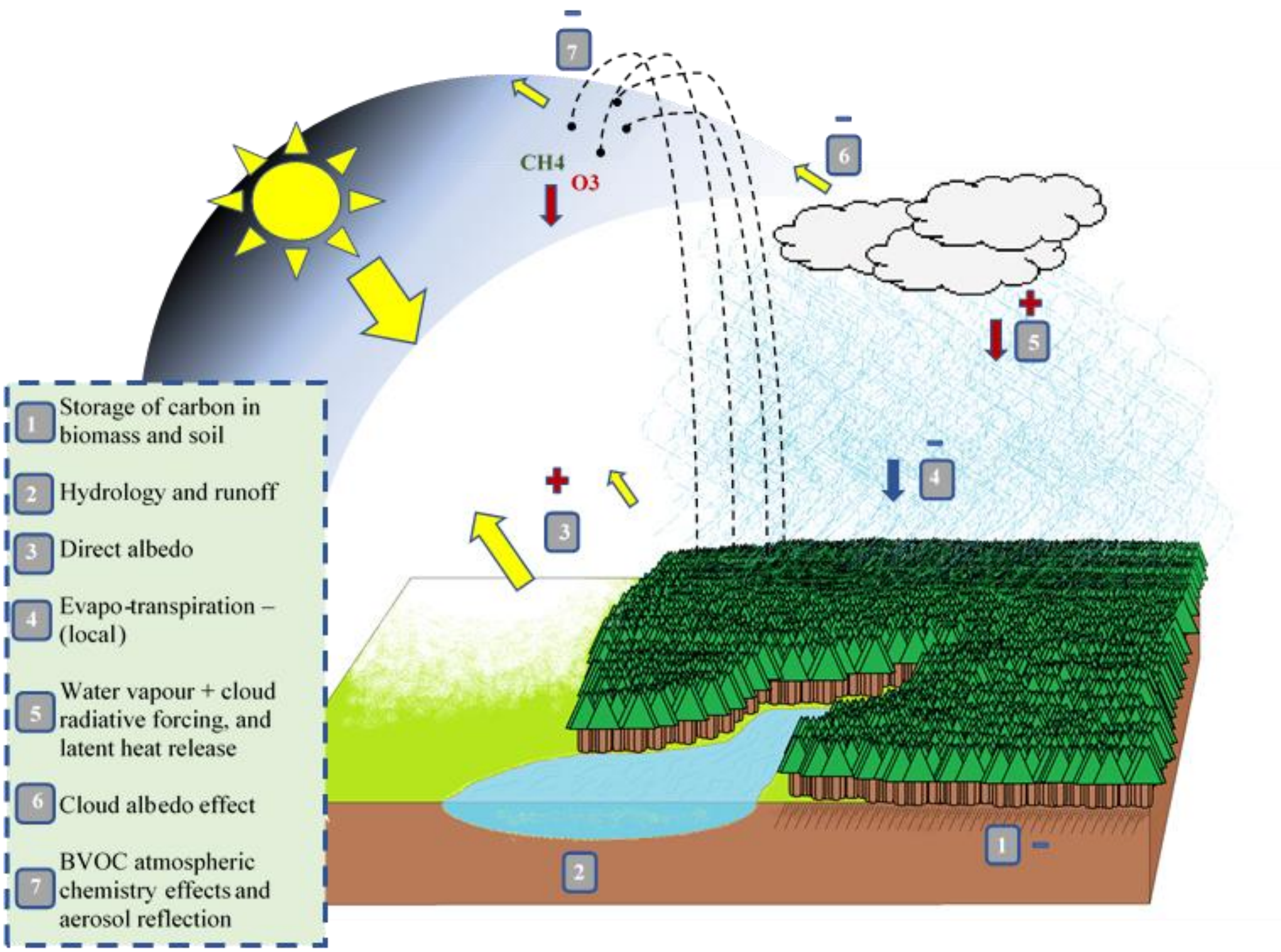


Figure 1: Illustration of a selection of forests functions in the climate system, with plus / minus sign depending on if the forests impact on the specific effect generally induce warming or cooling (local and global temperature effect) – Based on Strand et al. (2016), Scott et al. (2018), Spracklen et al. (2008), Cunningham and Cunningham (2010), Ellison et al. (2017).

Understanding the climate change mitigation potential of introducing wood-based biofuels in aviation requires comparing the benefits of fossil fuel substitution with alternative management options for the forest (Koponen et al. 2018). However, this is a challenging task since the forest's climate functions are not yet fully understood, difficult to quantify and, thus, connected with large uncertainties. Studies considering the climate effects of harvesting forest have largely only focused on the changes in the forest carbon pool. Moreover, assumptions and methodological choices vary strongly between studies which has led to different conclusions on the mitigation potential of wood-based bioenergy. In the following section the most important assumptions and methodological differences causing disagreement between studies of harvesting forest and wood-based-biofuels are discussed.

2.3. Assessment of wood-based biofuels

Bioenergy systems can be very complex, and therefore Life Cycle Assessment (LCA) is the common method for assessing their climate footprint. The goal of LCA is to sum up all the environmental impacts involved in a products life cycle from cradle to grave (ISO 2006). The impacts are then compared with a similar assessment of a fossil reference system. In this way, the emission savings from replacing fossil fuel with biofuel can be calculated.

LCA was conducted for both technological processes proposed for aviation biofuel production in Norway. The results showed that both fuels have the potential to achieve the limit of 60 % emission reductions compared to fossil jet-fuel, allowing them to be certified as a biofuel in the EU (Rambøll 2013). Biofuels are counted as zero emissions in the national inventory, and thus replacing 30 % of fossil jet-fuels will count as 30 % reduction in emissions from aviation as well. However, biogenic CO_2 emissions were not included in the life cycle of the biofuels. This practice has been common in LCA literature and the reason stems from carbon accounting in national emission inventories, where bioenergy emissions are counted in the land-use sector instead of the transport sector to avoid double counting (Cherubini et al. 2011a, Cintas 2018).

Counting biogenic CO_2 emissions as carbon-neutral can be reasonable when considering biofuels based on annual crops, but this simplification is not justified in the case of slow-growing forest (Searchinger et al. 2009, Holtsmark 2012, Lamers and Junginger 2013, Ter-Mikaelian et al. 2015). Harvesting forest for bioenergy involves a loss of carbon from the tree biomass and potentially from the forest soil. This carbon is temporarily transferred from the biosphere to the atmosphere. As long as the forest is allowed to grow back, a similar amount of CO_2 will in time be re-sequestered into new biomass. However, in boreal forest this can take several decades to over a century. During that time period, the biogenic CO_2 resides in the atmosphere, increasing the radiative forcing and thus contributing to climate change (Ter-Mikaelian et al. 2015, Liu et al. 2017). The system might eventually be deemed carbon-neutral when carbon has been sequestered back into new biomass. Nevertheless, the period of increased radiative forcing due to the temporary excess carbon in the atmosphere breaks the equivalency between carbon-neutrality and climate neutrality (Bright et al. 2012a).

The inclusion of temporal CO_2 emissions in bioenergy accounting led to the term carbon debt, and the climate impact wood-based bioenergy is strongly affected by how this concept is approached in studies (Ter-Mikaelian et al. 2015). Two main ways exist for estimating carbon debt. The term usually refers to the timeframe from harvest until the pre-harvest carbon levels have been restored (Lamers and Junginger 2013) This time period is called the carbon debt payback time. The alternative way of calculating carbon debt is to compare harvesting for biofuel against an alternative land-use scenario. The reference case can be forest protection, use for agriculture, etc. In this case, the carbon debt is calculated as the time required before new forest in the harvested area reaches the same carbon levels as the reference case. This is commonly referred to as the carbon offset parity time (Mitchell et al. 2012, Lamers and Junginger 2013). The choice of carbon debt approach can greatly affect the conclusions in studies because the carbon debt payback time only accounts for the carbon in the forest before harvest, while the carbon parity approach also accounts for avoided sequestration in the case of forest protection.

The two approaches have been central for scientific disagreement, and this is partly due to studies asking slightly different questions. For instance, Cherubini et al. (2013) argues that there is no direct causal relationship between the biofuel and the avoided sequestration in the forest, and therefore, they choose the carbon debt payback method. However, since this approach does not account for additionality, others argue that when assessing the climate impact of bioenergy as a component of total human activity, then an alternative reference scenario for the forest is needed (Murray et al. 2007, EPA 2014, Laganière et al. 2017, Koponen et al. 2018). Holtsmark (2015b) demonstrates how important the choice of approach is for identical systems. Bright et al. (2012b) argues that reference scenarios can provide useful insights, but that the scenario-based carbon assessments still neglect important biogeophysical considerations and are therefore of limited usefulness when considering climate impacts.

Accounting for carbon debt requires modelling of how the forest is affected by harvest, and in the case of carbon parity approach, how it would continue to grow in absence of harvesting. The most common method is to model a fixed forest stand, often 1 ha of forest. This simple model type can be expanded to include several forest stands, and can be adapted to actual forest inventory data (Lamers and Junginger 2013). However, the level of detail varies between studies, for example in which forest carbon pools that are included. Pingoud et al. (2012) only models the living biomass, while Guest et al. (2013) goes one step further by including residues. Laganière et al. (2017) further includes dead wood, but neglects soil carbon effects, which are taken into account by Holtsmark (2015b) and Repo et al. (2015). Assessments taking into account material production from wood usually also counts parts of the trees used in infrastructure as an additional carbon pool, but many neglect storage in landfill after the lifetime of the products (Lamers and Junginger 2013).

The results from models are strongly influenced by parameterization, and many parameters are uncertain or case specific. For example, the growth parameters of the forest stand can vary based on forest type and productiveness, and slow growth typically leads to longer carbon payback/parity times. Soil carbon loss can also be important for the post-harvest emissions, but lack of empirical data renders soil carbon loss highly uncertain, which is why many studies don't include it (Lamers and Junginger 2013, Laganière et al. 2017). Regional differences can often be a reason for differing parameters between models. Decomposition time, for instance, will typically be longer in northern regions, and residues and dead trees will therefore decay slower, storing carbon for a longer period, increasing the carbon debt for residues (Federici, S., Lee, D. and Herold, M. 2017). Uncertainty in each of the many parameters that goes into a forest model can add up, affecting the carbon debt and consequently the climate impact of the bioenergy system.

Forest management practices also vary between studies based on local custom and forest conditions. For instance, in Canada, large areas of trees are dying due to natural disturbances. This allows for harvesting standing dead trees, a practice which is more uncommon other places (Lamers et al. 2014, Laganière et al. 2017). In Norway, residues have traditionally been left in the forest, while in Sweden both slash and stump extraction are getting more common (Gustavsson et al. 2015). For both dead trees and residues, the only reasonable alternative to bioenergy production is for the biomass to rot in the forest (Bernier and Paré 2013). This would result in temporary carbon storage until the biomass has decayed, but the carbon debt becomes much shorter. Thus, different combinations of living trees, dead trees and residues leads to different climate impacts. Due to such regional variations in forest conditions and forestry practices, general conclusions on the mitigation potential of forest bioenergy becomes problematic (Junginger 2013).

Comparison of wood-based energy systems

In addition to the biogenic CO_2, fossils fuel emissions are usually involved in harvesting, processing, distribution and use of bioenergy. To account for these emissions, assumptions regarding the type and efficiency of conversion processes, harvest practices, storage capacity, inclusion of waste heat for fossil fuel substitution etc. must be formulated. Individual assumptions can have large effects on the results of studies. For instance, the storage of wood can lead to methane emissions, and Röder et al. (2015) found that alternative facility storage capacities alone can change the mitigation potential of the bioenergy system. Thus, the system boundaries are detrimental for the results. They are set by the researcher conducting the LCA, and can vary based on the subject and the intended goal of the study (ISO 2006). Differing system boundaries between assessment of the same products can lead to differing conclusions between studies.

For calculating the relative climate benefit of the bioenergy system, the biogenic carbon emissions must be compared to a baseline. This is usually a fossil energy system and the emissions involved in gathering, processing, distribution and burning of the fossil fuel are accounted for by LCA in the same way as for the bioenergy system. The major factor influencing the results between studies in the fossil reference system is the chosen fossil energy type to be substituted. Coal substitution usually leads to the best results for the bioenergy system, because coal with 90 g CO_2e MJ^{-1} in electricity generation has the highest emission intensity among the fossil fuels considered (Peake et al. 2018). Moreover, the technological adjustments necessary to replace coal with biomass-based fuels are relatively small. Liquid biofuels require more processing and replace liquid fossil fuels which have lower emissions per unit energy. Therefore, liquid biofuels usually perform worse compared to the reference system.

Impact indicators and time frame for forest bioenergy accounting

After all emissions in the bioenergy and the fossil reference system are accounted for, the systems must be compared, and the results presented. Due to the temporal dynamics of biogenic CO_2 emissions, with fast release and slow sequestration, the chosen impact indicator and time frame becomes highly influential for how the results are interpreted. Overall, snapshot indicators fall short, because the dynamic evolution has to be considered, but a standardized approach accounting for dynamic aspects is currently not existing (Brandão and Levasseur 2011).

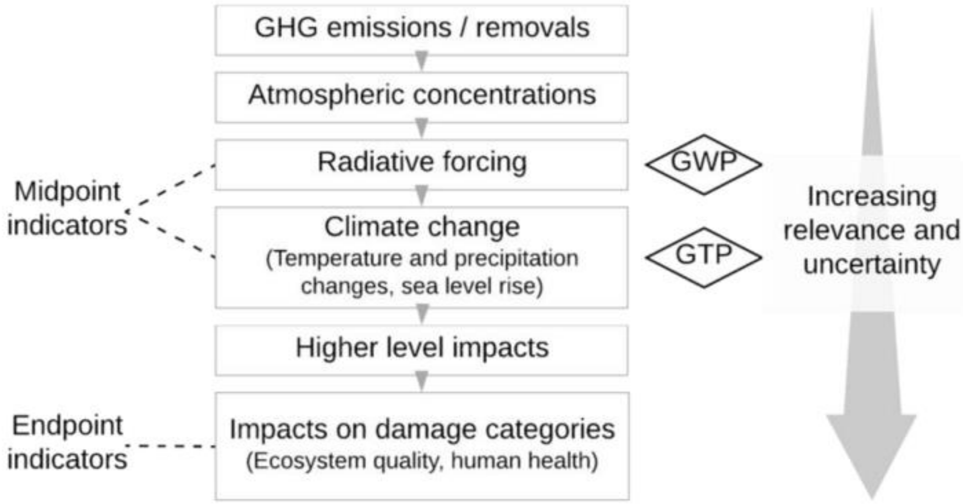


Figure 2: Shows the cause-effect chain of emissions in the climate system – Taken from Breton et al. (2018).

Figure 2 shows the cause-effect chain of GHG emissions. Many temporal carbon studies only estimates the cumulative GHG emissions and sequestration in the forest (Lamers and Junginger 2013). This approach is simplistic, because it does not explicitly display important earth system feedbacks which can be included by using atmospheric concentration. Radiative forcing is often applied in LCA, making the results also more suitable for comparison with other studies. The radiative forcing can be further calculated into temperature changes, enabling the inclusion of additional earth system feedbacks (e.g. the absorption of heat into the ocean). The optimal knowledge for decision-making would be summarized in the endpoint indicators (Withers et al. 2015). However, every step down the cause-effect chain involves increased complexity of assessments and higher uncertainty in the results (Fuglestvedt et al. 2003).

The impact indicators can be modelled over time, but for being able to provide simpler conclusions and basis for comparison, standardized climate change metrics are usually applied. The most used metric in carbon accounting and LCA is the Global Warming Potential (GWP), which is based on radiative forcing over a specified timeframe (Breton et al. 2018). Another metric, the Global Temperature Change Potential (GTP) is also gaining interest, because it estimates the relative temperature change at a specific point in time, which is often the target used in global policy initiatives (Frischknecht and Jolliet 2016). Both metrics are strongly influenced by the time frame / horizon chosen by the researcher, which in the case of wood-based bioenergy is highly important. Since the choice of time frame is a value laden decision, it is causing a fundamental problem for forest biofuel and temporary carbon storage accounting (Brandão and Levasseur 2011).

Many other metrics have been proposed specifically to account for bioenergy and temporary carbon storage due to the temporal profile of the climate impacts. Notably, the GWP-bio metric is aimed to provide a transparent indicator for including biogenic CO_2 emissions in LCA (Cherubini et al. 2011b). Also, the Climate-Tipping-Potential metric addresses important aspects of short-term emission spikes (Jørgensen et al. 2014). Still, it is impossible to identify one indicator that can show a balanced representation of the climate impact when comparing greenhouse gasses with different lifetimes (Cherubini and Tanaka 2016, Frischknecht and Jolliet 2016). An indicator that can represent the short-term impacts from emissions, overlooks the long-term consequences. Therefore, Frischknecht and Jolliet (2016) recommend applying two indicators in studies. GTP100 is suggested for longer term impacts, and GWP100 for short term impacts because this metric tends to underestimate long term effects, and usually presents values similar to GTP40.

Other aspects

As previously discussed, forests have many additional functions in the climate system to that of storing carbon. The climate impacts of changes in these functions, due to harvesting, are seldom included in bioenergy assessments. Especially the albedo effect is considered very important in the case of deforestation. Albedo change was found to be of slightly less importance for radiative forcing than the loss of forest carbon from harvesting in Norway, when using carbon debt payback approach in Cherubini et al. (2012). When using the sequestration parity approach and including several carbon pools, Holtsmark (2015a) found albedo change, estimated with the same albedo method as by Cherubini et al. (2012), to be clearly less impactful than loss of forest carbon. The studies applied albedo data for the Norwegian region Hedmark, which has relatively long-lasting snow cover. On the contrary, the indirect albedo effect induced by forests contribution to cloud formation might be significant in the other direction. Spracklen et al. (2008) found that boreal forests increase the cloud cover so much that it might counteract albedo change from deforestation in boreal areas during climatically warm periods.

Bioenergy can also alter fuel- and other products markets, which can potentially be important for the full climate impact of wood-based bioenergy. For example, biofuels can create a market for residuals, making harvesting more profitable, which could increase harvesting for building materials. Wood can substitute steel and concrete, materials involving high emissions in production, leading to emission reductions in the building sector (Gustavsson et al. 2017). On the contrary, according to Schulze et al. (2012) biofuel substitution raised the price of timber in Germany, which would favour the use of concrete. Additionally, it could potentially lead to increase of less environmentally concerned forestry operations, and deforestation in other places to cover the demand in the market (Schulze et al. 2012). Plevin (2017) argues that integrated assessment models should be utilized to shed light on these interlinked aspects of bioenergy systems. However, these types of assessments are complex and still involves large uncertainties, so market effects are not within the scope of most studies, and a full-fledged integrated assessment model for forest-based biosystems is still in its infancy.

The approach in this thesis

From above elaboration, it became clear that assessments of wood-based biofuels are strongly affected by methodological choices and assumptions. For the results to provide any meaningful insights, the methodology must be clearly presented. Box 1 gives a brief overview of the important methodological choices applied in this thesis, incorporating the points raised above. The method is described in detail in chapter 3 and 4, and in the supplementary material.

Box 1 | Overview of important methodological choices and assumptions in this thesis

Biogenic carbon:

- Model type: very coarse fixed landscape
- Forest type: Western Norwegian spruce
- Carbon debt approach: Scenario based / Carbon sequestration parity
- Carbon pools: Living biomass, dead wood, residues, organic soil carbon

Bioenergy system

- Biofuel type: Drop-in jet-biofuels based on the alcohol-to-jet pathway
- Important parameters: Storage capacity of wood, Nordic power mix,
- Emissions ignored in ETJ conversion, high ethanol-to-jet-fuel mass conversion efficiency

Fossil reference system

- Liquid jet-fuel produced in Norway

Impact indicators

- Atmospheric CO_2, radiative forcing, temperature change
- Timeframe: Year 2010 - 2209, 1-year timesteps

Other biogeochemical / physical effects and market effects

- Not included

3. Forest model for Western Norway

In summary, a forest model for Western Norway's spruce forest, based on Bjart Holtsmark's forest stand model, was developed to estimate the changes in forest carbon due to biofuel production. National forest inventory data was used for the forest area and distribution of age. Carbon dynamics for each hectare in the forest were based on a 1-hectare forest stand model. This chapter describes how the forest model was developed, and how the climate impacts from CO_2 emissions were calculated.

Box 2 | Important forestry terms

Site quality (SQ): Is a measure of the productiveness of forest. In Norwegian forestry, the site quality is classified using the H40 system, by a number usually between 6 and 26, which represents how high the trees in an area become 40 years after they reach breast height (1.3 m).

Development class (DC): Is a grouping system dividing forest into 5 classes. Development class 1 is areas with sparse forest that's going to be prepared for forestry. Development class 2, 3, and 4 consists of further development stages of forest, and finally, development class 5 is forest ready to be harvested. The best forestry practice is to harvest trees first when they reach development class 5, but still, forest stands are sometimes harvested in less developed state. Forest with higher site quality reach development class 5 faster than low site quality forest, and vice versa.

3.1. Holtsmark's forest stand model

The stand model laying the foundation for the carbon dynamics in each hectare was developed by Bjart Holtsmark to estimate the carbon debt from harvesting wood (Holtsmark 2015b, a). The carbon pools in the model encompass living biomass, dead wood, soil carbon and residues. I was provided from Holtsmark an excel version of this model. Its formal description is found in Holtsmark (2015b). In the following is a description of how the model project carbon dynamics in a 1-ha-spruce-stand after harvest, as shown in Figure 3.

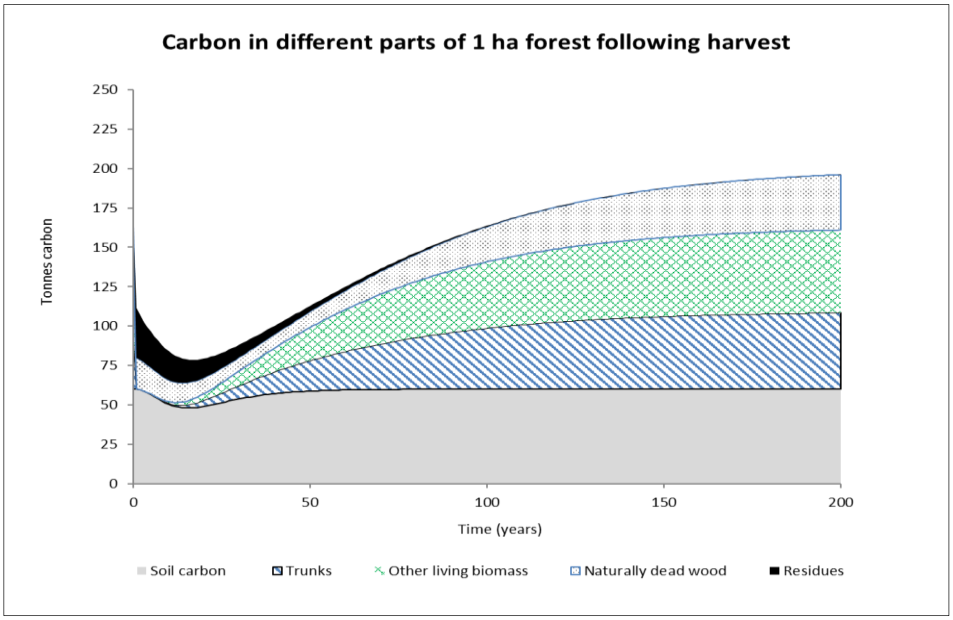


Figure 3: Shows the carbon development in Holtsmark's forest stand model following harvest.

The main equations in the model describe the content of carbon in living biomass, dead biomass, organic soil and residues. The carbon contained in living biomass is the combined carbon in "trunks" and "other living biomass". With tau representing the time, the carbon in living biomass $B(\tau)$ is simulated on a yearly basis with the Richard Chapmans function:

$$B(\tau) = v1(1 - e^{-v2\tau})^{v3}, \tag{1}$$

where $v1$ is a scale parameter related to the volume of trees, calibrated so that the volume of stems in the model matches the typical value in 1 ha of 100-year-old Norwegian spruce forest, i.e.194 m3 (Holtsmark 2015b). This roughly corresponds with the average stem-volume density found in Majasalmi et al. (2018). It should be noted that stems typically accounts for 48 % of the biomass in Norwegian spruce trees (based on Eastern Norway region from Løken et al. (2012)). Furthermore, the relationship between living biomass and carbon content is ca. $0.2\ tC\ m^{-3}$ wood (Lamlom and Savidge 2003, Asante and Armstrong 2012). Therefore, the value of $v1$ in is the maximum carbon content that can be contained in living biomass in the model, given in tons. $v2$ is an empirical growth parameter scaling the growthrate of living biomass, and $v3$ is an empirical parameter based on the proportionality between an organsim's mass and catabolism (Pienaar and Turnbull 1973). The values of $v2$ and $v3$ are based on Holtsmark (2013) and Cherubini et al. (2011b). The following parameter values applies to the Holtsmark model: $v1 = 103.067$, $v2 = 0.0245$ and $v3 = 2.6925$.

The carbon in the naturally dead wood is dependent on how much dead wood is accumulated as a portion of living biomass, and its rate of decay. The carbon in the dead wood pool $D_N(t)$ develops over time as follows:

$$D'_N(t) = \beta B(t) - \omega D_N(t), \tag{2}$$

where β is a parameter representing litterfall based on the living biomass $B(t)$, and $\omega D_N(t)$ describes decomposition. Therefore, the amount of dead wood emerging at time k, that is left at time t, is $e^{-(t-k)\omega}\ \beta B(k)$. Thus, the carbon dynamic in the dead wood pool is described by the formula:

$$D_N(t) = e^{-t\omega} D_0 + e^{-t\omega} \int_0^t e^{k\omega}\ \beta B(k) dk, \tag{3}$$

where D_0 is the carbon contained in the dead wood present in the forest stand before the harvest. The value of D_0 is set to 21.5 t C when 100-year-old forest is harvested. $\beta = 0.01357$, and was taken from Asante and Armstrong (2012) and Asante et al. (2011), where it was estimated based on the typical dead organic matter dynamics for pine in the Carbon Budget Model of the Canadian Forest Sector (CBM – CFS3) (Kurz et al. 2009).

Soil carbon in the stand model originates from the organic soil layer. The soil carbon content $S(t)$ follows the formula:

$$S(t) = S_0 + s_1 e^{s2t}(1 - e^{s2t})^{s3}, \tag{4}$$

where S_0 represents the stable organic soil carbon content in the fully grown forest (De Wit and Kvindesland 1999). The other parameters were calibrated so that the soil carbon loss after harvest would reach 12 t C, 15 years after harvest. At this point the soil carbon starts to increase again. Values of parameters are: S_0 = 60 t C, S_1 = -113.5, S_2 = -0.09, S_3 = 3.003

When forest is harvested, a specified portion σ of the living biomass is extracted from the forest. The carbon in the residue pool $D_R(t, \tau_h, \sigma)$ is the remaining biomass, which is left to decay on the forest floor, described by the following formula:

$$D_R(t, \tau_h, \sigma) = e^{-t\omega}(1 - \sigma) B_{(\tau_h)}, \tag{5}$$

where ω is the decay rate for organic matter, and $B_{(\tau_h)}$ is the living biomass at the time of harvest. *The value of ω is set to 0.04.*

3.2. Adapting the model for Western Norway

Data on the size and productiveness of the Western Norwegian spruce forest (exemplary 2010) was collected from reports of the Norwegian institute for forestry and landscape, for the counties Møre og Romsdal, Sogn og Fjordane, Hordaland and Rogaland (referenced in Appendix Table 1). They provide data for each region based on permanent sample planes. The 250 m^2 planes are located in a 3x3 km net. Additionally, smaller temporary sample planes are added within each permanent sample plane to increase the regional accuracy. The data for Møre og Romsdal is older than for the other counties. However, no correction for this was done, which might slightly underestimate the real spruce forest volume and area in Western Norway in 2010.

Tables with data for development class 2 to 5 classified as spruce forest in each county was merged and used as a basis for developing the Western Norwegian forest model. A more detailed description of data collection process is found in the Appendix, page 3 - 5, including tables of forest area, volume, volume density and age, based on the forest inventory. In summary, the main data assumptions are shown in Box 3, and the estimated average age and area distribution of development class 2 to 5 is shown in Table 1. The area and age of the development classes were used for dividing the forest into four age classes, which lay the foundation for a forest model of 166 289 ha.

Box 3 | Main data assumptions from forest inventory

Total stem volume of spruce forest in Western Norway: 36.7 Mm3

Average site quality of the spruce forest (by area): SQ 19

Stem volume density of SQ 19 spruce forest in development class 5: 429 $\mathrm{m}^3\mathrm{ha}^{-1}$

Table 1: Area and starting age of age classes in model

Development class	*Total area*	*Average age*	*Age class*
	Area (ha)	Age (y)	*Class name*
DC2	41200	17	*AC1*
DC3	72361	36	*AC2*
DC4	42875	53	*AC3*
DC5	9853	77	*AC4*

According to the forest inventory data, site quality 19 was found to best represent the average productivity of Western Norwegian spruce forest. The average stem volume density of a development class 5 spruce stand of site quality 19 is estimated to 429 $m3\ ha^{-1}$ (Development class 5 is average 77 years old). However, since Holstmark's model simulates a forest stand with a stem volume density at 194 $m3\ ha^{-1}$ for a 100-year-old forest, the original growth rate parameters need to be adjusted to depict growth in a typical hectare in Western Norway.

Following Holtsmark (2015b) and Cherubini et al. (2011b), parameter $v1$ was adjusted by fitting the growth to a pre-determined volume. However, this neglects that denser forest can also grow at a different phase, and the growth tend to reduce faster (Pienaar and Turnbull 1973, Bergseng and Dale 2015). Uncertainties are high, but assumptions on forest growth rates strongly influence overall model results. Thus, to account for the sensitivity on growth assumptions and parameter fitting, four different growth rate versions are applied to test the forest model in this thesis. Table 2 lists the growth rate parameters (GRP) used in the different cases. Each set of parameters are given a scenario identifier referencing model version. Note that "**GRP original"** is the growth rate parameters as used in Holstmark's stand model.

The adjustment procedure for growth parameters is in brief as follows. The total living biomass volume of Western Norwegian (WN) spruce forest (77-year-old site quality 19) is ~ 894 $m3\ ha^{-1}$, based on the stem volume. The carbon contained in this biomass is ca. 178 t following Lamlom and Savidge (2003). In the parameter set "**GRP low**" parameter $v1$ was changed so that the modelled carbon in living biomass in 100-year-old forest matches the average productiveness in development class 5. This likely underestimates the carbon content compared to the actual carbon in an average hectare of the same age. For parameter set "**GRP mid**", parameter $v1$ was changed so that the modelled carbon in living biomass after 77 years of growth matches the estimated "real" carbon in living biomass of 178 t. For parameter set "**GRP high**" $v1$ is the same as in "GRP mid", but parameter $v2$ was also changed to depict a case where the forest has a greater maximum growth, followed by less growth (compared to the other growth rates) as it gets older. The carbon accumulation in living biomass under each parameter set is shown in Figure 3. All new growth rate versions have a much higher total carbon accumulation in living biomass than the "GRP original", and the productivity between the new sets differ considerably as it takes "GRP low" around 150 years to reach the same carbon level as "GRP high" at 60 years.

Table 2: Growth rate parameters (GRP)

PARAMETER	ORIGINAL	LOW	MID	HIGH
$v1$	103.067	227.73	278	278
$v2$	0.0245	0.0245	0.0245	0.04
$v3$	2.6925	2.6925	2.6925	2.6925

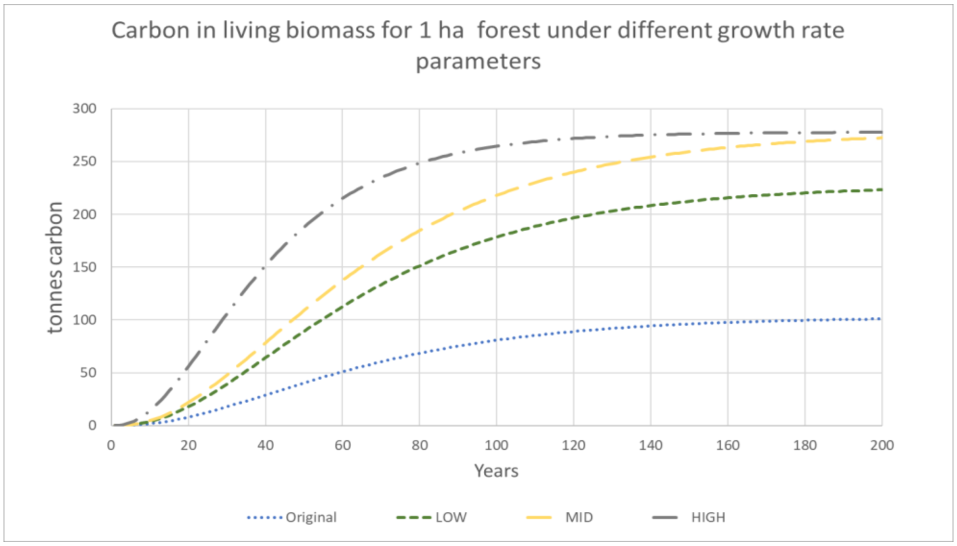


Figure 4: Shows the carbon accumulation in living biomass in 1 ha forest under the different growth rate parameters.

Adjustments in the model equations were carried out for adapting the dead wood (see equation 6), residues (see equation 7) and soil carbon (see equation 8), to the forest model and the relevant harvesting regime. In detail they are the following.

Dead wood and residue dynamics in the original stand model version I was provided differ from Holtsmark (2015b). Instead of applying one decay rate for wood, the dead wood and residues are divided into into stems, stumps, branches, roots and barnacles based, on the proportions given in a report by Landbruks- og matdepartementet (2009), each biomass type having individual decay rates. Additionally, the improved dead wood and residue functions had to to be adjusted to simplify the excel calculations in the Western Norwegian forest model. In this model, the carbon in dead wood $D_{N(t)}$ is calculated as:

$$D_{N(t)} = \sum_{n=0}^{t} \beta o B_{(t)} \sum_{i=1}^{5} e^{-t\omega_i} Pd_i + DW_H \sum_{i=1}^{5} e^{-t\omega_i} Pd_i \,, \qquad (6)$$

where $\beta oB_{(t)}$ represents the litterfall as portion of living biomass with the "GRP original" parameter set. The choice of using the original growth rate was due to uncertainty in how the litterfall accumulation is affected by increased forest productivity, and since the original growth rate is lower, this represents a more conservative approach. ω_i is the specific decay rate for- and Pd_i is the share of- each tree-part (stumps, branches, barnacles, roots, stem) in intact trees. DW_H represents the dead wood left in the forest at the previous harvest. Following from this, the first term describes litterfall, which is accumulating and decaying simultaneously, while the second term describes old dead wood which is only decaying. The assumption that all the forested areas in the model were previously harvested at 100-years of age was adopted, and therefore $DW_H = D_0 = 21.5$ in undisturbed areas. $\beta = 0.01357$ and values for ω_{1-5} and Pd_{1-5} are listed in Table 3. Figure 4 in the Appendix shows how the dead wood pool develops over time in undisturbed forest.

Due to the assumption that all the forest areas have been harvested previously, two residue pools are needed in the forest model. The old residue pool describes the residues that are left from when the age class started growing after the previous harvest, while the new residue pool is the residues generated when the forest is harvested once more in the case of biofuel production. The carbon contained in the old residue pool, $oD_{R(y)}$, is described as:

$$oD_{R(y)} = 0.52B_{(100)} \sum_{i=1}^{4} e^{-y \cdot \omega i} Po_i\,, \tag{7}$$

where $0.52B_{(100)}$ represents the amount of carbon left in the forest as residues when the forest was harvested (at age 100 years). Common harvesting practice in Norway has been to take only the stem and leave the other biomass as residues. Therefore, 52 % of the living biomass is left, since stems account for 48 % following Løken et al. (2012). y is a time variable representing the number of years since the old residue pool was generated, which is also the age of the specific age class where the residue pool belongs. ω_i is the decay rate for- and Po_i is the share of, each tree-part, when only stems are removed. Values for ω_{1-5} and Po_{1-5} are listed in Table 3.

The inclusion of an old residue pool based on current growth likely overestimates the residues in the forest, because much of today's spruce area was previously less productive forest types (Roll-Hansen 2015). This could overestimate the carbon in the forest in early years and underestimate the carbon accumulation in the forest model for the first decades since large parts of the forest is young and would have a significant decaying residue pool. However, this has no impact on the scenario results because new harvest simulation does not affect the old residue pool. The new residue pool is only simulated when the forest is harvested and is described further down.

Table 3: Parameters in decay functions

Part	Pd_i *whole trees*	Po_i *no slash harvest*	Pr_i *70 % slash harvest*	ω_i *Decay rate*	*Term*
Stump	0.06	0.12	0.15	0.03	1
Branches	0.18	0.35	0.14	0.06	2
Barnacles	0.06	0.12	0.15	0.5	3
Roots	0.22	0.42	0.56	0.04	4
Stem	0.48	0	0	0.03	5

The soil carbon formula was changed to:

$$S(t) = (S_0 + s_1 e^{s2t}(1 - e^{s2t})^{s3}) - (S_0 - sC_t) + (S_0 - sC_t)(1 - e^{-sv1t})^{sv2}, \tag{8}$$

where sCt represents the soil carbon value in the harvested area at the time of harvest, and $sv1$ and $sv2$ are parameters. The first term is identical to the earlier presented soil carbon function from Holtsmark (2015b). The second and third term were added to adapt the soil carbon for the forest model. The second term ensure that if the forest is harvested before the soil carbon is fully restored, then the soil carbon will start declining from the current level after the harvest, instead of starting at 60 t C. The third term causes the soil carbon to gradually increase to the maximum value of 60 t C again. The parameters were fitted in order to let the soil carbon increase at a similar rate as in the original formula. The values of $sv1$ and $sv2$ are 0.04 and 2, respectively. See the soil carbon description on page 7 in the Appendix for further details.

With the adjusted soil carbon formula, the soil carbon can sink lower than 12 t C, which was maximum soil carbon loss in the original model. In a model study by Repo et al. (2015) the soil carbon level is also further reduced following several harvests, but if the increased soil carbon loss following shorter rotation cycles is realistic, is unclear. However, the additional soil carbon loss has no significant impact on the results because the lowest soil carbon content is around 58 t C per ha, in AC3, at the time of harvest, in the scenarios investigated in this thesis.

The adjusted stand model was used for all the Western Norwegian spruce forest area. Thus, all the hectares in the forest are assumed to have the same productivity, depending on the parameter set for growth rates applied. However, the age differences cause the four age classes to accumulate carbon at different phase. For example, age class 1 was on average 17 years of age in 2010. Therefore, the accumulation of biomass starts at t = 17 for the 41 200 ha of forest in this class. Age class 2 starts accumulating from t=36 and so on. This results in different carbon accumulation rates and biomass densities, (see Appendix page 6 for graphs of carbon accumulation and biomass density over time for the individual age classes). The sum of the carbon within each class is the total carbon in the Western Norwegian spruce forest, as shown in Figure 6. Based on the carbon content in living biomass, the forest model estimates the stem volume in the forest in 2010 under different growth rate parameter sets as shown in Figure 5. Evidently, the parameter set "GRP high" overestimates the biomass by 45 %, while "GRP low" underestimates the biomass by about 35 %. The "GRP Original" is 70 % lower than the forest inventory estimates.

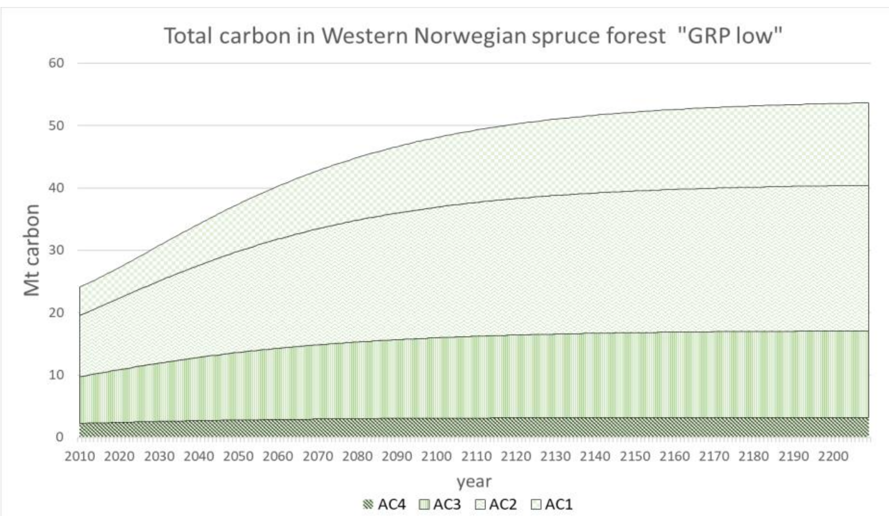


Figure 6: Shows the total carbon in modelled Western Norwegian spruce forest (GRP low).

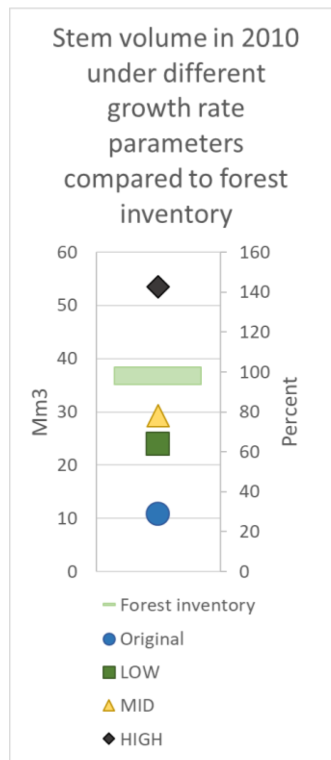


Figure 5: Shows the total stem-volume in modelled forest under different growth rate parameter sets, compared to forest inventory, in 2010.

When forest is harvested in the model, the number of harvested hectares is removed from the relevant age class, and the same area is added to a new class named harvest, as illustrated with Figure 7. The harvest class consists of all the harvested areas, of different age, depending on which year harvesting took place. The carbon in the harvest class is the sum of carbon in all the harvested areas of different age. For instance, if one ha of age class 4 is harvested in, 2020 and every subsequent year until 2030, the area of age class 4 will then be 10 ha smaller than in 2019. This area is continually moved to the harvest class, which in 2030 is composed of 10 hectares of different age, from 1 to 10-year-old forest, each following the same carbon dynamics as the other age classes.

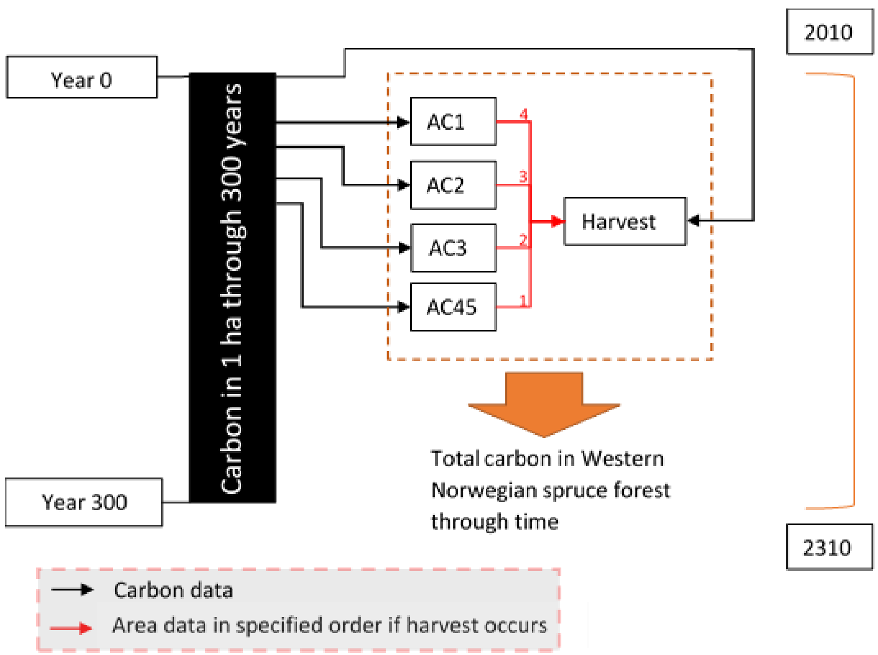


Figure 7: Illustration of model logic.

The area requirement for a given amount of wood depends on the biomass density in the age class where harvest takes place, as well as how much residues that are extracted. In this thesis, 70 % of slash is harvested together with the stems. Under this harvest regime, the volume received per harvested hectare amounts to 61 % of the living biomass. The oldest age class is always harvested first in the model. Only clear cutting is simulated, so harvesting does not affect the biomass density within in the age classes. Figure 6 in the Appendix shows how the biomass density in each class develops over time during the relevant period.

As 61 % of the living biomass is harvested, 39 % is left in the forest. This biomass is the residues, which consist of stumps, roots, needles and 30 % of the slash. This is modelled as a new residue pool in the relevant harvest class area. The new residue pool, $nD_{R(t,ageH)}$, is described by the formula:

$$nD_{R(t,ageH)} = 0.39B_{(ageH)} \sum_{i=1}^{5} e^{-t\omega_i} Pr_i, \tag{9}$$

where $0.39B_{(ageH)}$ represents the share of the living biomass left in the forest after the harvest, which depends on the age of the forest at the time of harvest $(ageH)$. ω_i is the specific decay rate for- and Pr_i is the share of, each tree-part, when the stem and 30 % of slash is removed. Values for ω_{1-5} and Pr_{1-5} are listed in Table 3.

Summed up: After harvest all the carbon pools are transferred from the age class to the harvest class, and additionally, a new residue pool is generated in this class. The time variable t starts at 1 for all new areas in the harvest class. The living biomass pool starts accumulating carbon following Equation 1. The old residue pool continues decaying at the same rate as before the recent harvest, following Equation 7. The carbon in the dead wood pool becomes the starting value for the rotting dead wood in the new harvest class ($D_N = DW_H$) , while the litterfall part of the dead wood pool now starts accumulating as a portion of the living biomass (based on “GRP original”), following Equation 6 (This might lead to a slight overestimation of the decomposition of the dead wood right after harvest, because the decay starts from rates corresponding to full trees, while in the dead wood pool in the age class, the more slowly decaying tree parts would make up an increasing portion of the dead wood). The soil carbon starts decreasing following Equation 8. Figure 8 illustrates how harvesting is simulated in the forest model.

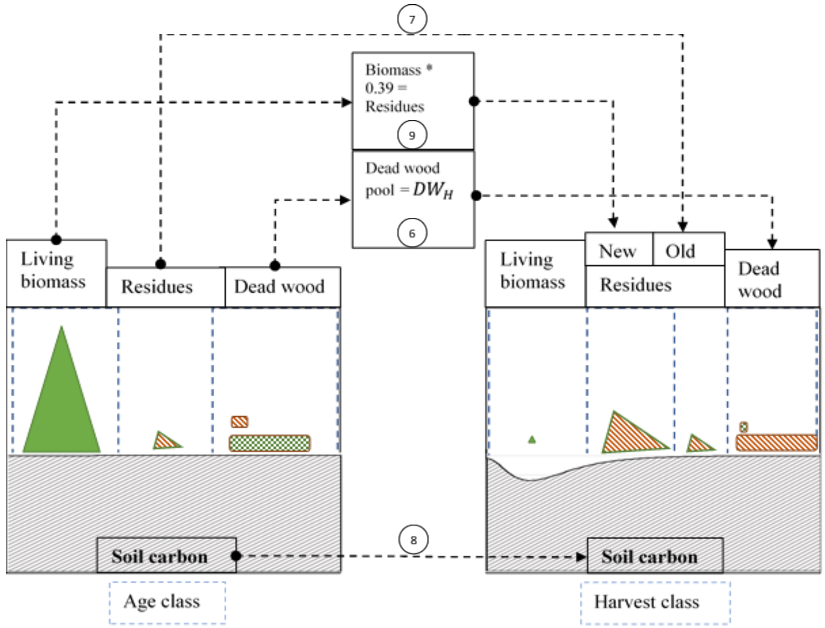


Figure 8: Illustration of harvesting in forest model, numbers refer to functions.

Harvesting can occur in two age classes in the same year if the oldest age class is nearing reaped. If this happens, then the residues and the dead wood generated, and the soil carbon content in the harvested area, are treated as if all the harvest took place in the younger age class that year. As a consequence of this calculation method, the dead wood and residue pools are overestimated, the old residue pool is slightly overestimated, and the soil carbon loss is slightly overestimated in this specific year. By how much, will vary based on the relative portion of the total harvest taken in each age class. However, the effect is not considered to have significant impact on the results since AC 4 and 3 are the only age classes harvested in the scenario in this thesis.

A more practical explanation of how the harvest calculations were carried out in excel, is provided in the Appendix, page 14 – 18.

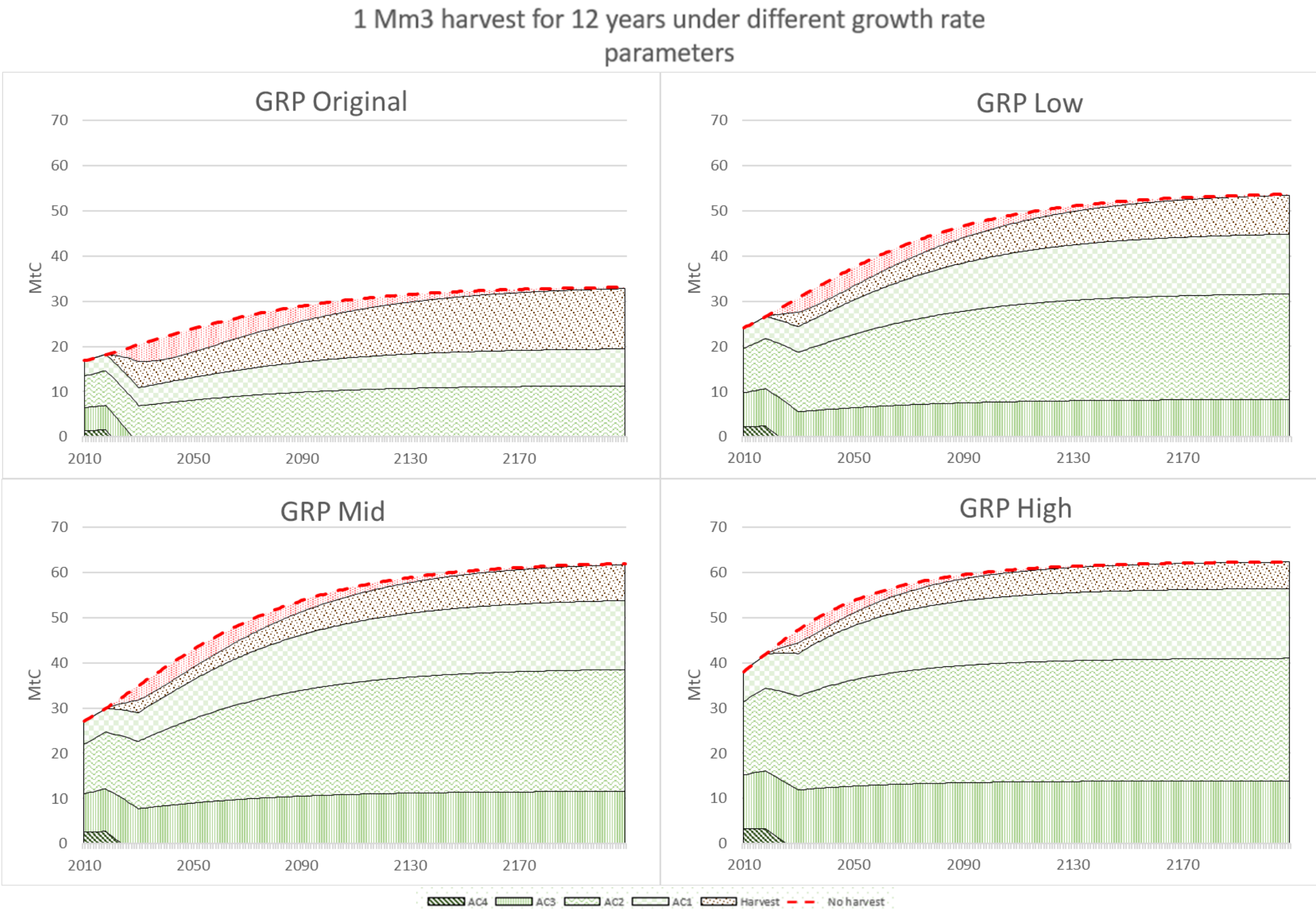


Figure 9: Shows a simulation of 1 Mm3 annual harvest rate for 12 years from 2019, under different growth rate parameters (70 % slash extracted).

Figure 9 shows the modelled carbon in the forest after harvest, under the different growth rate parameter sets. An annual harvest rate of 1 Mm3 is simulated over 12 years from 2019. The impact of the harvesting can be seen in the graph as a reduction in forest carbon and the emergence of the harvest class, which has higher growth than the other age classes through the timeframe of the simulation. The red-dotted line represents the alternative carbon level in the forest had it not been harvested. Thus, the area between the red-dotted line and the forest carbon in the harvest case represents the alteration in the forest carbon pool due to harvesting. When no fossil CO_2 emissions are substituted, this can be considered as the carbon debt, with the sequestration parity approach. The sequestration parity time is then the number of years from harvest until the red area is covered due to increased growth in the forest again. As Figure 9 illustrates, the carbon debt is much larger over a longer period in slow-growing forest. The sequestration parity time can be based on climate impact indicators, such as the atmospheric concentration of CO_2 resulting from the change in forest carbon, as well.

In this thesis, forest is harvested for the purpose of producing biofuels. Therefore, the substitution effect of replacing fossil fuels with biofuels is also considered, which shortens the carbon parity time. To account for the substitution effect, a bioenergy system and a fossil reference system are needed. The following paragraphs describes how the biofuels and fossil fuels are accounted for in this thesis.

For the biofuel production, the stem and 70 % the of slash from trees are utilized as feedstock. The woody biomass is transformed into drop-in biofuels via the alcohol-to-Jet pathway. This first involves harvesting, transportation, and biochemical fermentation of wood into ethanol. The energy efficiency and emission intensity in this part of the production chain is based on a study by Bright and Stromman (2009). Bioethanol then further goes through dehydration, olefin oligomerization and α-olefin hydrogenation, to become drop-in jet-biofuels. No scientific assessments on this process for Norwegian conditions were found. Therefore, emissions in this production step is excluded, and only the conversion efficiency, based on Han et al. (2017), is taken into account.

The fossil reference system is based on Killingland (2013) and involves extraction and refining of oil from the North sea, and distribution and burning of jet-fuel in Norway. Burning accounts for 90 % of the total emissions in this case. All parameters which the biofuel production process is based on, and the relevant sources, are provided in Table 4 and Table 5. Next, the climate impacts of CO_2 emissions resulting from production and burning of fuel, and from alterations in the forest carbon pool, are calculated as in the following part.

Table 4: Applied heat values and energy densities

Substance	Heat value	Density	Reference
Dry wood	-	400 kg $m3^{-1}$ *	(Norsk ved 2018)
Ethanol	29.7 MJ kg^{-1}	0.789 kg l^{-1}	(Engineering Toolbox 2003)
Jet-A1	43.15 MJ kg^{-1}	0.804 kg l^{-1}	(AirAir BP 2000)

* Density of wood is rounded up from 380 kg $m3^{-1}$ to simplify calculations

Table 5: Applied conversion efficiencies and production emissions

Process	Conversion efficiency	GHG emissions	Reference
Wood to ethanol	261 l ethanol per t dry wood	21.7 g CO_2e per MJ Ethanol	(Bright and Stromman 2009)
Ethanol to jet-fuel	18 MJ jet-fuel per kg ethanol	0 (chosen value)	(Han et al. 2017)
Fossil jet-fuel	-	84 g CO_2e per MJ (includes production and burning)	(Rambøll 2013)

3.3. Calculations of climate impacts from emissions

Biogenic CO_2 emissions

When forest biomass is harvested in the model, it is assumed to be processed into biofuels and burned during the same year. Therefore, all the resulting carbon loss from the forest is counted as direct CO_2 emissions to the atmosphere. When the forest grows and gains carbon, it counts as negative CO_2 emissions. The sequestered or released carbon from the forest is converted into CO_2 emissions (Em_{CO2}) with the following formula:

$$Em_{CO2} = -\Delta FC \cdot \frac{w\ CO2}{w\ C}, \tag{10}$$

where ΔFC represents the change in forest carbon, given in mass. $w\ CO2$ and $w\ C$ represents the molecular weight of carbon dioxide and carbon, with respective values of $44\ g\ mol^{-1}$ and $12\ g\ mol^{-1}$.

Atmospheric CO_2

A large fraction of CO_2 that is released to the atmosphere, is over time absorbed by the oceans and the biosphere. Since the carbon in the atmosphere, ocean and biosphere is working towards balance, a negative emission induces an opposite effect. This can be modelled with an impulse response function based on carbon cycle-climate models, using a constant background CO_2 concentration (Cherubini et al. 2011c). This method can sufficiently model the fraction of CO_2 remaining in the atmosphere for emissions smaller than 100 Gt C (Joos et al. 2013). The following impulse response function, where $C(t)$ represents the fraction of CO_2 that remain in the atmosphere, is used for both positive and negative emissions:

$$C(t) = a_0 + \sum_{i=1}^{3} a_i \cdot e^{-t/\tau i}. \tag{11}$$

The amplitude $a0$ represents the fraction of the initial CO_2 emission that stays in the atmosphere over geologic timescales, while the amplitudes ai represent the relative sink capacity of the oceans and the biosphere. The movement toward balance between the atmosphere, the oceans and the biosphere, happens over time, corresponding to the relaxation time scales τi (Cherubini et al. 2011a). The parameters were taken from Levasseur et al. (2010) and has the following values: $a0 = 0.217, a1 = 0.259, a2 = 0.338, a3 = 0.186, \tau 1 = 172.9\ years, \tau 2 = 18.51\ years, \tau 3 = 1.186\ years.$

Radiative forcing

For the time CO_2 remains in the atmosphere, it will increase the radiative forcing on the earth. Radiative forcing from a change in the atmospheric concentration of CO_2 can be parameterized with an expression based on radiative transfer models:

$$RF_{(\Delta C \rightarrow 0)} = \alpha \, ln\left(\frac{C_0 + \Delta C}{C_0}\right), \tag{12}$$

where C_0 is the reference burden of CO_2 as of spring 2019 and ΔC is the CO_2 perturbation size. $\alpha = 5.35\ W\ m^{-2}$ (Myhre et al. 1998, Hodnebrog et al. 2013).

Temperature change

The change in earth's energy balance expressed with radiative forcing do not have a linear effect on the surface temperature. The large oceans, having much higher heat capacity than the atmosphere, absorbs most of the extra warming due to radiative forcing perturbations. Heat is rapidly transferred into the upper layer the water column. Thereafter, the mixing between the shallow and deeper layers in the ocean leads to a slow heat uptake that lasts for long periods of time. The temperature impact of an instantaneous change in radiative forcing is best modelled in complex climate models containing many heat pools and feedback mechanisms. However, in recent literature and in the IPCC assessment report (AR5), authors argue that useful approximations are a sum of exponentials with 2 or 3 response terms. The following formula, where $R_T(t)$ is the surface temperature response to a unit forcing, was used for the temperature change calculations in this thesis:

$$R_T(t) = \sum_{j=1}^{2} \frac{c_j}{d_j} exp\left(-\frac{t}{d_j}\right). \tag{13}$$

The c_j terms are given in $(K(W\ m^{-2})^{-1})$ and are related to climate sensitivity. The d_j terms are given in $years$ and are associated with the heat uptake in the upper and deeper layers of the oceans. The formula is based on the Hadley Centre Coupled Model version 3. The climate sensitivity in this model is 1.06 K per W forcing, or 3.9 K for a doubling of the atmospheric CO_2 concentration. Values of parameters are: c_1 = 0.631, c_2 = 0.429. $d_1 = 8.4, d_2 = 409.5$ (Shine et al. 2005, Myhre et al. 2013a, Myhre et al. 2013b).

Several tests were conducted to confirm that the climate impact calculations can replicate results from literature. This involved comparison of CO_2 pulse emission effects on atmospheric CO_2, radiative forcing and temperature response, and a test of the temperature response following 1-watt increased forcing. The tests show varied results, but all were within the ranges of the results found in literature. The tests are shown in the attached excel workbook: climate response calculation test, and some of the test results are presented in the Appendix, page 9.

4. Fuel scenarios for the Western Norwegian aviation sector

Scenarios are plausible, simplified descriptions of future development. Scenarios are often used in environmental science when assessing policy measures or emission targets. The advantage is that scenarios enable to cope with decision-making under high uncertainty through investigating different plausible outcomes (Rosentrater 2010). Scenarios can be divided into normative and exploratory types. The former starts with a future, and tries to answer what policies, trends etc. can lead to the envisioned target. Exploratory scenarios, on the other hand, explores future states induced from different policies, trends etc. (European-Commission 2005). Alternative scenarios can be compared, and their strengths and weaknesses can be weighed against each other.

In this chapter, three simple scenarios are developed, which investigates fuel sales in Western Norwegian aviation between 2019 and 2030. They explore the mitigation potential in the aviation sector by 1) phasing-in of wood-based jet-biofuels, and 2) reducing jet-fuel sales. The baseline scenario (BAU) involves increase in fuel sales in the aviation sector, in accordance with projections from literature. In this scenario the forest is not harvested and keeps growing as projected by the forest model when no harvesting takes place. In explorative scenario 1 (BIO-JET), the total fuel sales in the aviation sector grow at the same phase as in BAU. However, an increasing portion of the jet-fuels are substituted by wood-based biofuels throughout the period. This also involves processing of biofuels and harvesting of forest. Finally, explorative scenario 2 (AVI-RED) involves a gradual reduction of aviation fuel sales over the period. In this case the forest is not harvested and continues growing as in BAU. The scenarios are illustrated in Figure 10.

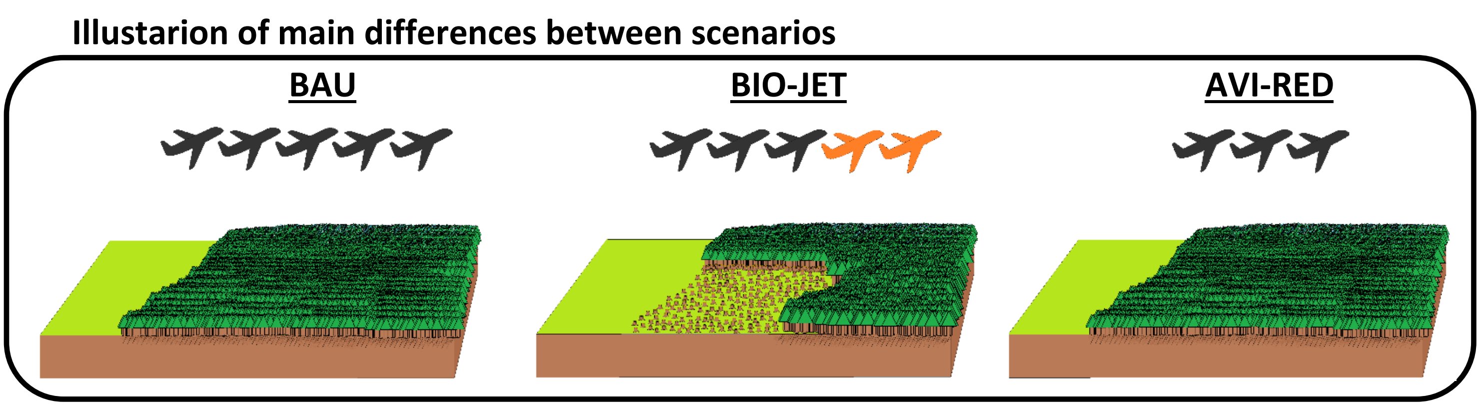


Figure 10: Illustrates the three scenarios in the thesis. The baseline (BAU) scenario involves increased aviation activity and no harvesting of forest. Explorative scenario 1 (BIO-JET) involves equal increase in aviation activity, but most of the rise in fuel sales is covered by wood-based biofuels. Explorative scenario 2 (AVI-RED) involves reduced aviation activity, and no harvesting of forest.

Here follows a more detailed description of the scenarios

Business as usual scenario

The business as usual – short: BAU scenario – is commonly a continuation of current trends. Thus, it represents a future where no additional measures are implemented in the Norwegian aviation sector. The scenario is grounded in predictions of fuel sales taken from Avinor (2017). The assumptions which their projections are based upon were not fully provided. Therefore, it is uncertain if a high annual fuel efficiency gain of 1.6 %, which is the target over the period, is included (Avinor 2016).

According to Henriksen (2018) 1 130-million-liter jet-fuel was sold in Norway in 2017, of which 185-million-liter (16.4 %) was sold in Western Norway. Of the national jet-fuel sales, 900-million-liter was registered to aircrafts, and because the statistics don't distinguish between sectors within counties, 147-million-liter (16.4 % of 900 Ml) is taken as the reference for 2019 fuel sales to Western Norwegian aviation.

The target of 30 % biofuels in total fuel sales amounts to 400-million-liter in 2030 (Rambøll 2017). This translates into a total of 1 333-million-liter jet-fuel sold to aviation. Therefore, 218-million-liter (16.4 % of 1 333 Ml) is used as the projected fossil fuel sales in 2030 in the BAU scenario. Thus, the BAU scenario depicts a linear increase in jet-fuel sales from 147-million-liter in 2019, to 218-million-liter in 2030, as shown in Figure 11.

Figure 11: Jet-fuel sales in Western Norway under the three scenarios.

Explorative scenario 1: National Transport Plan Jet-biofuel scenario

In the NTP Jet-biofuel – short: BIO-JET scenario - the target share of biofuels is reached in the Western Norwegian part of the aviation sector, and the biofuels are based on spruce forest from the region. The biofuels are gradually phased in from 1 % of fuel sales in 2019 to 30 % in 2030, as proposed in the National Transport Plan (Samferdselsdepartementet 2017). Overall, the total volume of fuel sales is the same as in BAU, but a linearly growing portion consists of jet-biofuels. Therefore, the growth in jet-biofuel sales over the period starts at 1.5 million liter in 2019, ending at 65.5 million liter in 2030, as shown in Figure 11.

The phasing-in of biofuels leads to lower fossil jet-fuel-use compared to BAU, without compromising aviation activity. However, this scenario additionally involves reductions in the carbon levels in the Western Norwegian spruce forest, due to harvesting for biofuel feedstock. This scenario therefore leads to lower GHG emissions from upstream processing and burning of fossil jet-fuel, but additionally includes upstream emissions from biofuel processing, and reduced carbon levels in the forest, as estimated with the Western Norwegian spruce forest model.

Explorative scenario 2: Aviation reduction scenario

In the Aviation reduction – short: AVI-RED scenario – jet-fuel sales are gradually reduced by 30 % between 2019 and 2030 due to restrictions on air travel. Since BAU involves growth over the period, the 30 % reduction from 2019 level leads to over 50 % lower fuel sales in 2030 relative to BAU, as shown in Figure 11. The AVI-RED scenario does not involve processing of biofuels, and the forest continues growing as in BAU. Table 6 shows the fuel-use in the three scenarios.

Table 6: Fuel-use in each scenario over the period 2019 – 2030

	Total fuel-use (Ml)	Fossil fuels (Ml)	Biofuels (Ml)
BAU	2 194	2 194	0
BIO-JET	2 193.3	1 829	364.3
AVI-RED	1 495	1 495	0

5. Results from scenarios for Western Norwegian aviation

This chapter presents the climate impacts resulting from GHG emissions and changes in the forest carbon in the scenarios. The estimates do not take into account indirect atmospheric effects arising from airplane exhaust, nor additional biogeochemical and biogeophysical effects potentially induced by harvesting of forest. Therefore, the climate impact estimates given in radiative forcing and temperature changes should not be considered as conclusive for the climate effect of the scenarios. Overall, they provide rough reference points for magnitudes of climate impacts.

Table 7 shows the main impacts and differences between scenarios for: cumulated GHG emissions (biogenic CO_2 ignored), carbon storage in the forest and temperature response under the three scenarios. Δ is defined as to change from BAU. The lower fossil jet-fuel-use in the BIO-JET scenario leads to 0.6 Mt lower cumulated CO_2e emissions between 2019 and 2030 compared to BAU. However, in this scenario, changes in carbon storage potential of the forest leads to higher temperature impacts by 2050 and 2100. The AVI-RED scenario has overall lower jet-fuel-use, resulting 2 Mt less of cumulated CO_2e emissions over the period compared to BAU. This leads to the lowest temperature impacts of the three scenarios over 200 years. The results indicate that growth in aviation activity, together with increased emissions from fuel-production and temporary reductions in the carbon storage in the forest, can counteract the mitigation potential for the phasing-in of biofuels in the aviation sector. The results also indicate that limiting traffic volume can be more effective for mitigating GHG emissions than phasing-in of biofuels. Next, a more detailed presentation of the scenario results follows.

Table 7: Main results from scenario analysis (Mid-range between GRP low and GRP high)

	BAU	BIO-JET	Δ BAU	AVI-RED	Δ BAU
Cumulated GHG emissions (2019 – 2030) (Mt CO_2E)	6.4	5.8	- 0.6	4.4	- 2
Carbon stored in forest in 2030 (Mt)	39	37	- 2	39	0
Carbon stored in forest in 2100 (Mt)	54.1	53.1	-1	54.1	0
Carbon stored in forest in 2150 (Mt)	57	56.7	-0.3	57	0
Temp-response 2050 (K)	3.7 E-06	8.5 E-06	+133 %	2.5 E-06	-32 %
Temp- response 2100 (K)	2.9 E-06	4.1 E-06	+40 %	2 E-06	-32 %
Temp- response 2150 (K)	2.7 E-06	2.7 E-06	-3 %	1.9 E-06	-32 %

5.1. Greenhouse gas emissions from aviation fuels

Figure 12 shows the GHG emission intensity of the fuels. The production emissions for biofuels in this thesis are similar to the estimates for alcohol-to-jet biofuels from the report by Rambøll, conducted on behalf of the Norwegian aviation industry (Killingland 2013). However, the biofuel in this thesis just exceed the limit of 60 % reductions, compared to the fossil fuel alternative, and would therefore not qualify as a biofuel in the EU (Killingland 2013). The discrepancy in the emission intensities between the two biofuels is influenced by highly different conversion efficiencies in the wood-to-ethanol conversion process. This is balanced by a higher ethanol-to-jet conversion efficiency in this this thesis, as shown in Table 8.

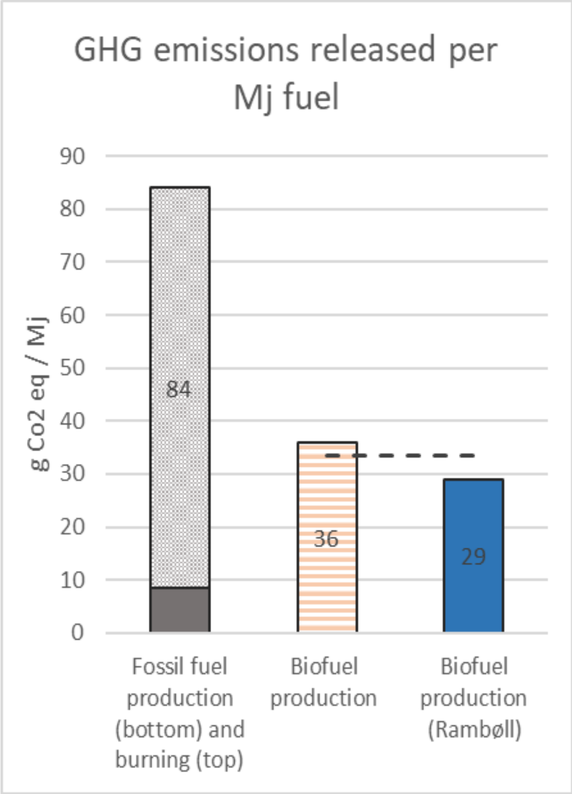


Figure 12: Shows the GHG emissions per unit energy for the fossil jet-fuel (dark grey – production, light grey - burning), the biofuel in this thesis (orange striped), and the ATJ biofuel from Rambølls assessment (blue), when biogenic CO_2 is ignored (Killingland 2013). The striped line marks the upper limit for 60 % emission reductions through substitution.

Table 8: Mass Conversion Efficiency rates

Process step	This thesis	Rambøll LCA
Wood to ethanol	0.21	0.51
Ethanol to bio-jet	0.42	0.27
Wood to bio-jet	0.09	0.14

Western Norway
University of
Applied Sciences

Figure 13 shows the annual GHG emissions resulting from aviation fuel-use in the three scenarios, when biogenic CO_2 is ignored. Replacing 30 % of the jet-fuel between 2019 and 2030 leads to 18 % reduced emissions in 2030, and 10 % reductions in cumulated emissions over the period. The AVI-RED scenario is more effective for mitigating GHG emissions, as the emissions are halved in 2030 compared to the BAU. The cumulated emissions over the period is 32 % lower than BAU in this scenario. The reason why BIO-JET shows lower mitigation potential than AVI-RED is due to 1) the growth in the overall fuel-use over the period, and 2) the increased processing emissions from biofuel production compared to conventional jet-fuels. The results indicate that just limiting the growth in the aviation sector can be more effective for mitigating GHG emissions than the phasing-in of biofuels, even when biogenic CO_2 emissions from burning of biofuels are ignored.

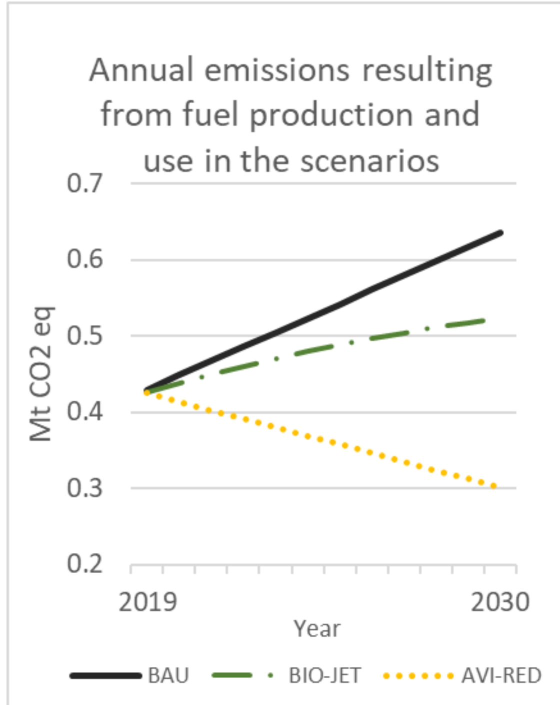


Figure 13: Shows the GHG emissions involved in production and burning of fuel in the scenarios, when biogenic CO_2 is ignored.

For all the scenarios, about 10 % of the emissions from fossil fuels are process emissions that would mostly not be ascribed to aviation, while this is true for all the biofuel emissions. Thus, for the aviation sector alone, the biofuel alternative would result in larger emission reductions, but the processing emissions would increase in other sectors in Norway when the biofuels are produced domestically. If these emissions are subtracted due to accounting, then the effectiveness of phasing in biofuels for reaching the overarching national emission targets can appear higher than what is realistic.

Table 9: GHG emissions in scenarios when biogenic CO_2 is ignored

	BAU	BIO-JET	Δ BAU	AVI-RED	Δ BAU
Emissions in 2030 (Mt CO_2e)	0.64	0.52	-18 %	0.3	-53 %
Cumulated emissions (2019-2030) (Mt CO_2e)	6.4	5.8	-10 %	4.4	-32 %

5.2. Climate impacts from emissions in scenarios

When considering the climate impacts resulting from emissions in the scenarios, then the temporary changes in forest carbon in the BIO-JET scenario must be considered as well. Figure 14 shows the change in atmospheric CO_2, over 200 years, resulting from fossil fuel production and use, biofuel production and changes in the forest carbon pool, under the three scenarios.

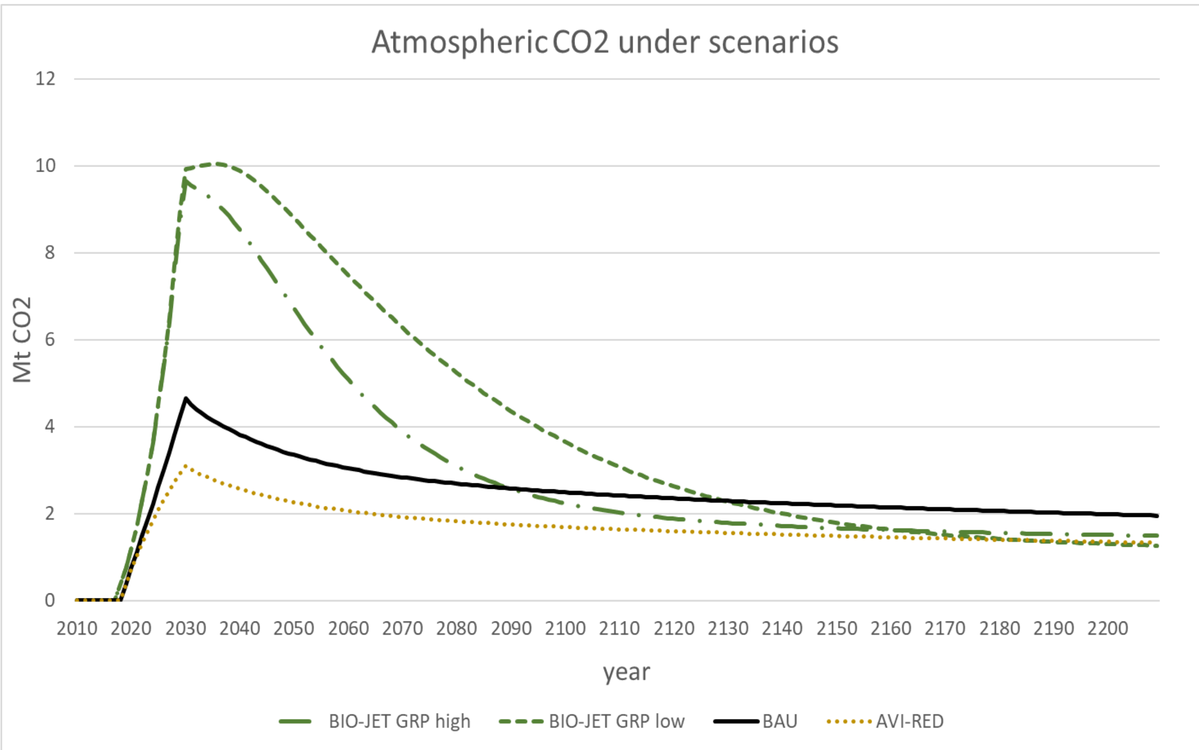


Figure 14: Shows the atmospheric CO_2 impacts under the scenarios. Two sets of growth rate parameters are applied in the BIO-JET scenario.

The fuel production and utilization only occur from 2019 to 2030, while the resulting changes in forest carbon ensues over the whole timeframe. The drop in the atmospheric CO_2 content after 2030 for the BAU and AVI-RED scenarios is due to CO_2 being taken up by earth's oceans and biosphere following Equation 11. The same is true for both the fossil and the biogenic CO_2 in the BIO-JET scenario, but in this case the biogenic CO_2 is also taken up by the growth of new forest in the harvested areas.

Since great uncertainty is linked to the growth of the forest, two growth rate parameter sets, "GRP low" and "GRP high", are applied in all the presented climate impact estimates for the BIO-JET scenario. These were the parameter sets (disregarding "GRP original") that consistently provided the largest range of climate impacts. The "GRP mid" set provided very similar results as "GRP low". For specific results on the climate impacts from biogenic CO_2 under the different growth rates, see the Appendix page 10 – 11.

Compared to BAU, the BIO-JET scenario has much higher short-term atmospheric CO_2 impacts. The atmospheric CO_2 in BIO-JET reaches equal levels as BAU between 2090 and 2130 depending on the forest growth, and approach similar levels as AVI-RED in the long term. The atmospheric CO_2 impact in the AVI-RED scenario is 30 % lower compared to BAU throughout the studied timeframe.

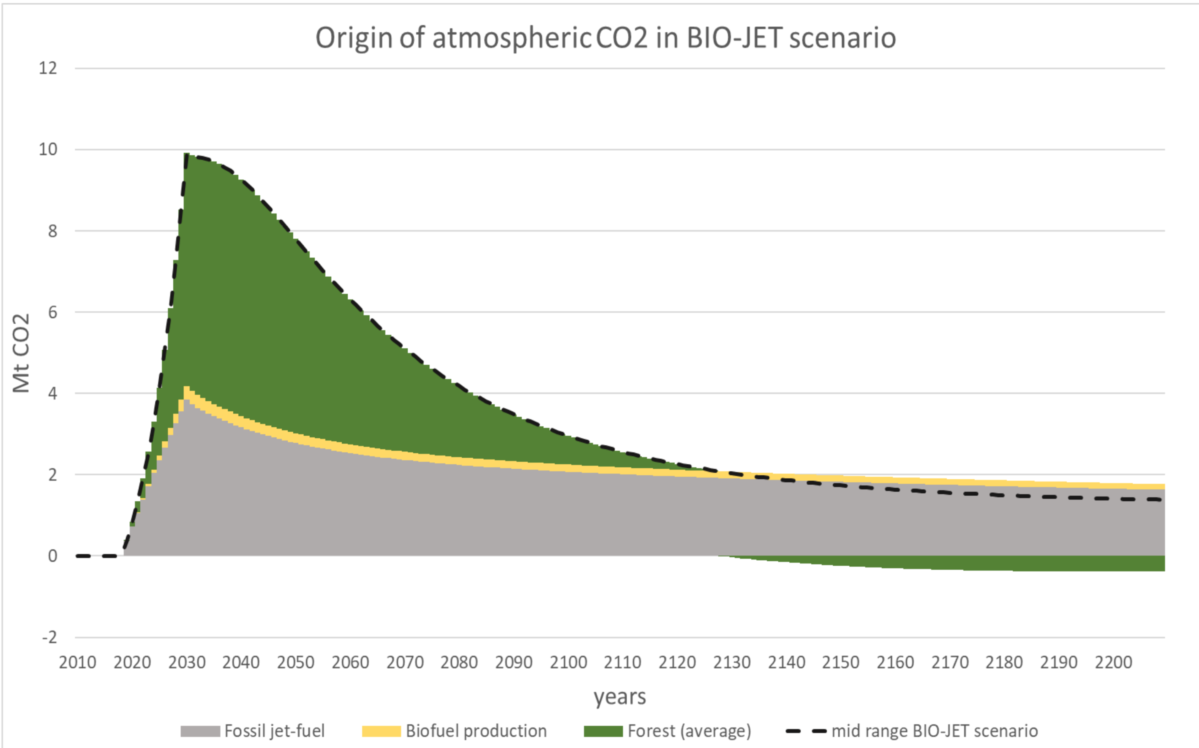


Figure 15: Shows the origin of the atmospheric CO_2 in the BIO-JET scenario. Mean values between GRP high and GRP low were used for "Forest", and for the dashed line representing "mid-range BIO-JET scenario".

Figure 15 shows the origin of the atmospheric CO_2 ascribed to the BIO-JET scenario. The release of carbon from forest-biomass and soil is the source of a substantial part of the atmospheric CO_2 impact, for around a century after the end of the scenario period. Biofuel production emissions constitute a relatively small share of the atmospheric CO_2 impact. In the long-term, the changes in forest-carbon contributes to reducing the atmospheric CO_2 impact in the scenario. The reason for this, is because the harvested forest is younger and can act as a stronger carbon sink over the timeframe, compared to when the forest is not harvested. However, even further into the future this effect will decline together with the forest growth.

It should be noted that all the fossil fuel emissions, which were given in CO_2 equivalents, are treated as CO_2 in these estimates. Since a significant portion of the biofuel production emissions in Bright and Stromman (2009) was methane, the atmospheric CO_2 from this process is overestimated. A case was tested where 40 % of the CO_2 equivalent production emissions was simulated with similar atmospheric decay and radiative efficiency as methane. This had negligible little effect on the climate impact estimates, because production emissions constitute a very small portion of the overall emissions. The test is described in the Appendix, page 20.

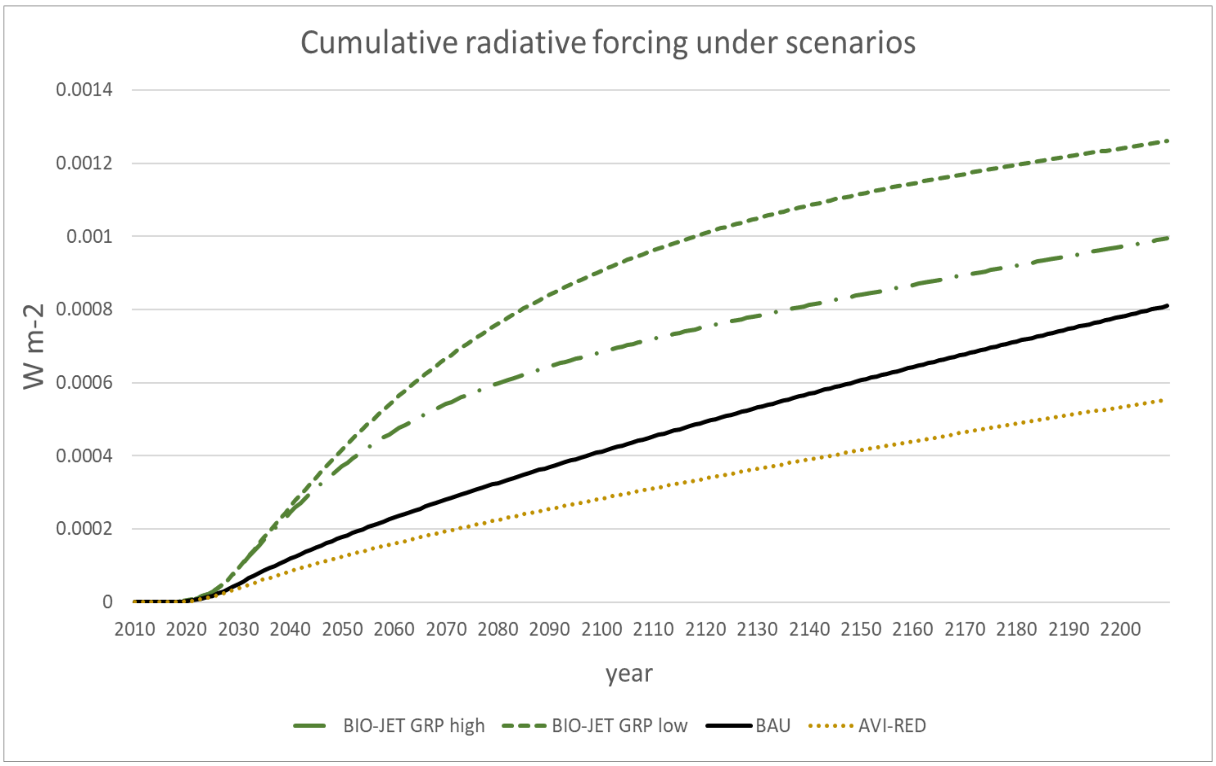


Figure 16: Shows the cumulative radiative forcing from atmospheric CO_2 under the scenarios.

Figure 16 shows the cumulative radiative forcing impact over 200 years for the three scenarios, with a CO_2 radiative efficiency of $1.67 \cdot 10^{-15}\ W\ \mathrm{m2}^{-1}\ \mathrm{kg}^{-1}$. The BIO-JET scenario has around 120 % higher impact on radiative forcing compared to BAU in 2050, approaching 40 % higher radiative forcing in 2200. The AVI-RED scenario has around 30 % lower radiative forcing over the timeframe. The reason why the cumulative radiative forcing continues increasing in the long term is due to the part of the fossil CO_2 emissions released in each scenario that stays in the atmosphere for long timeframes (a0 in Equation 11). This CO_2 will continue affecting the energy balance of the earth. Therefore, over even longer time periods than presented here, the BAU and BIO-JET scenarios will cross, while AVI-RED will continue to have the smallest impact because it has the lowest share of fossil CO_2 emissions.

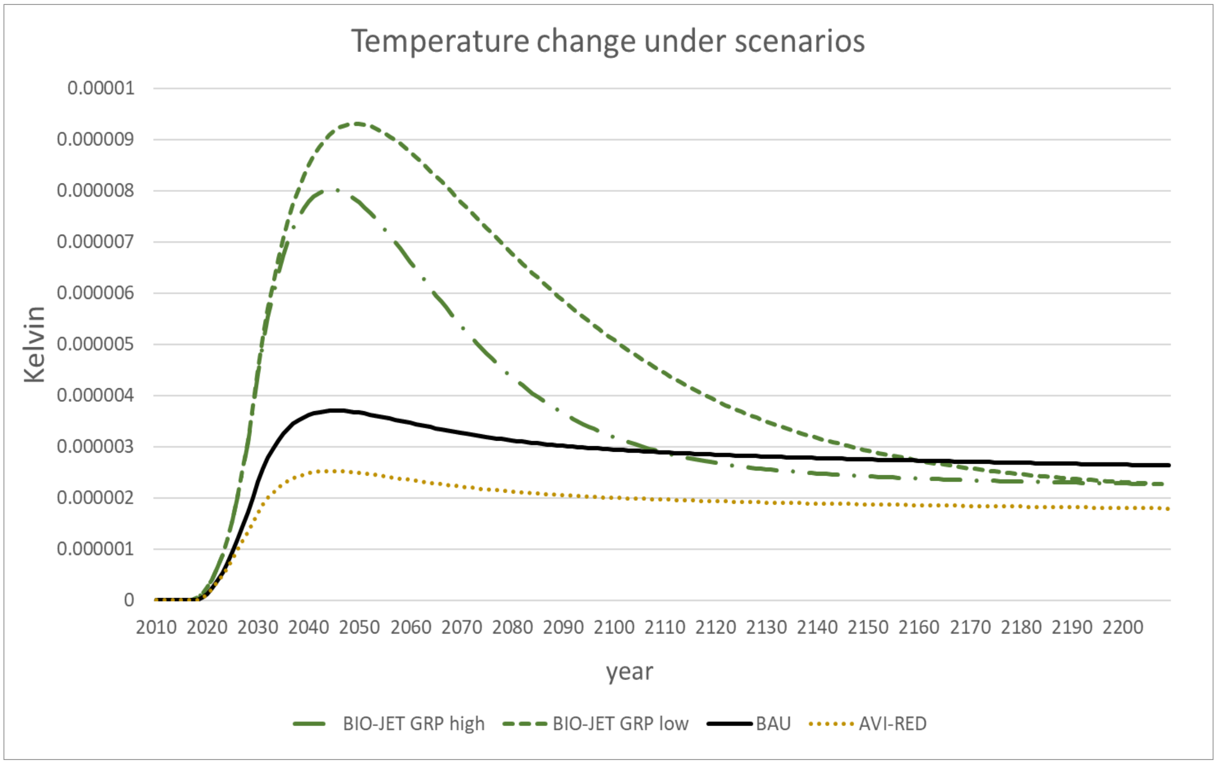


Figure 17: Shows the temperature response resulting from atmospheric CO_2 under the scenarios.

Figure 17 shows the surface temperature response under the scenarios. The temperature change estimates involve high uncertainty but provides extra insights into the climate effects resulting from the different development paths. The BIO-JET scenario leads to 130 % higher temperature response in the short-term. The temperature impacts rapidly decline after 2050 but continue to be significantly larger than in BAU for around a century after the simulation period ends. In the long-term, the temperature impacts approach lower than BAU. The shape of the temperature response curve is very similar, but has a slight delay, compared to atmospheric CO_2. This results in equal temperature response between the BIO-JET and BAU between year 2110 and 2160, depending on forest growth. As for atmospheric CO_2 and radiative forcing, the temperature impact is about 30 % smaller in the AVI-RED scenario compared to BAU throughout the studied timeframe. Although the atmospheric CO_2 at the end of the assessed timeframe is almost equal between BIO-JET and AVI-RED, the temperature response is clearly lower in the latter.

5.1. Discussion of results

The results presented in this thesis depend on specific assumptions, uncertain parameters, and show strongly time-dependent impacts in the BIO-JET scenario. Therefore, the first part of the discussion encompasses sensitivity analysis and evaluation of parameters and important assumptions, while the second part discusses how to interpret the results. From this, advice on potential future research priorities in the field, and policy recommendations, can be drawn. Figure 18 provides an overview of how the different parameters and calculations are interconnected.

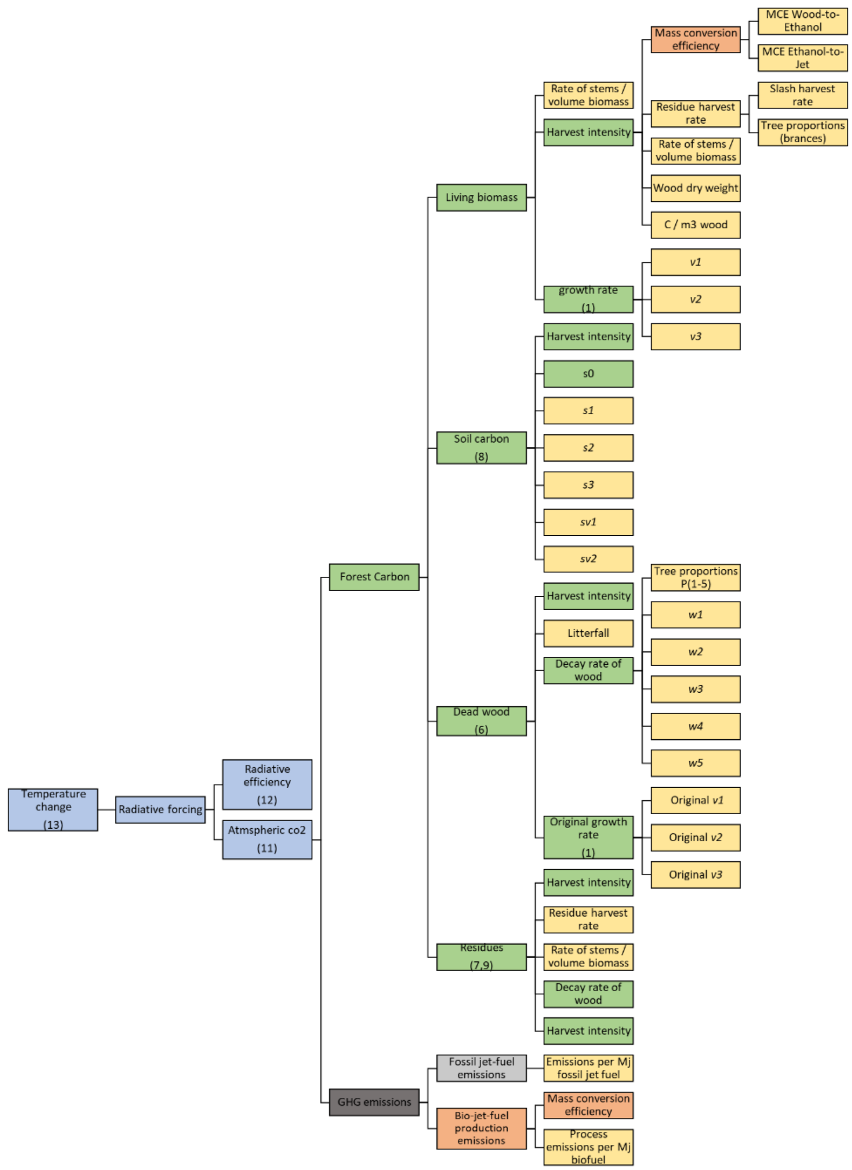
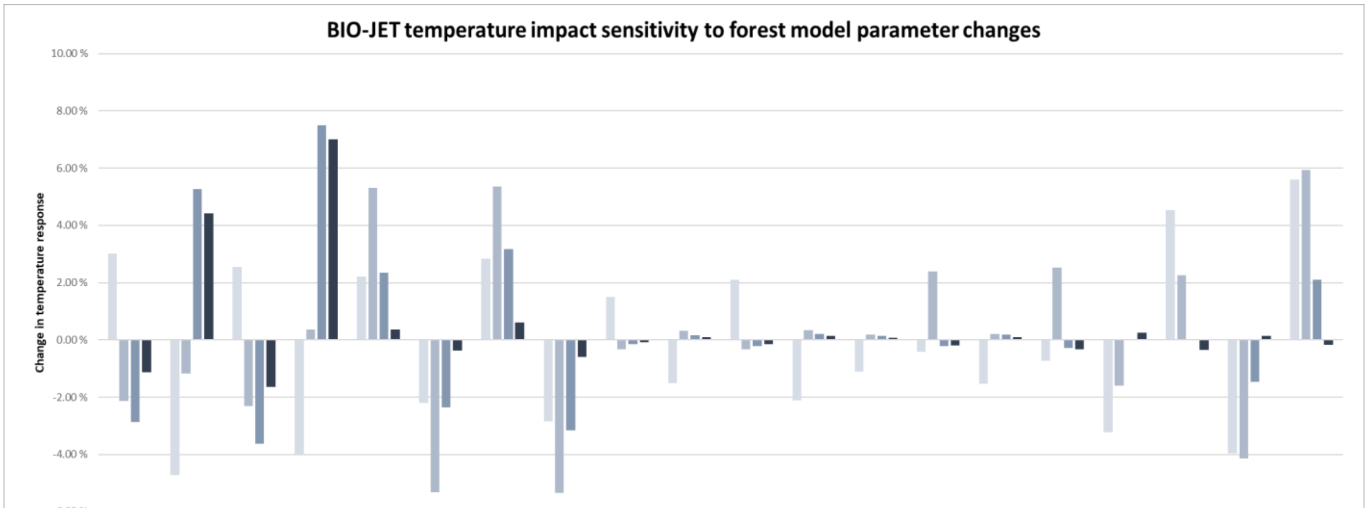


Figure 18: System overview showing which part in the calculations are affected by which parameters. The number of the relevant equations are referenced in the bottom of the boxes. Parameters are marked yellow.

Sensitivity analysis

Sensitivity analysis was carried out for most of the parameters, testing the effect of parameter changes for cumulated GHG emissions and temperature response. A more detailed explanation of how the parameter changes was carried out can be found in the Appendix, page 19 - 21. The sensitivity analysis is presented as changes in the results at several points in time, due to changes in the given parameters. Synergetic effects between the parameter changes were not explored but can be important when considering the uncertainties in the results.

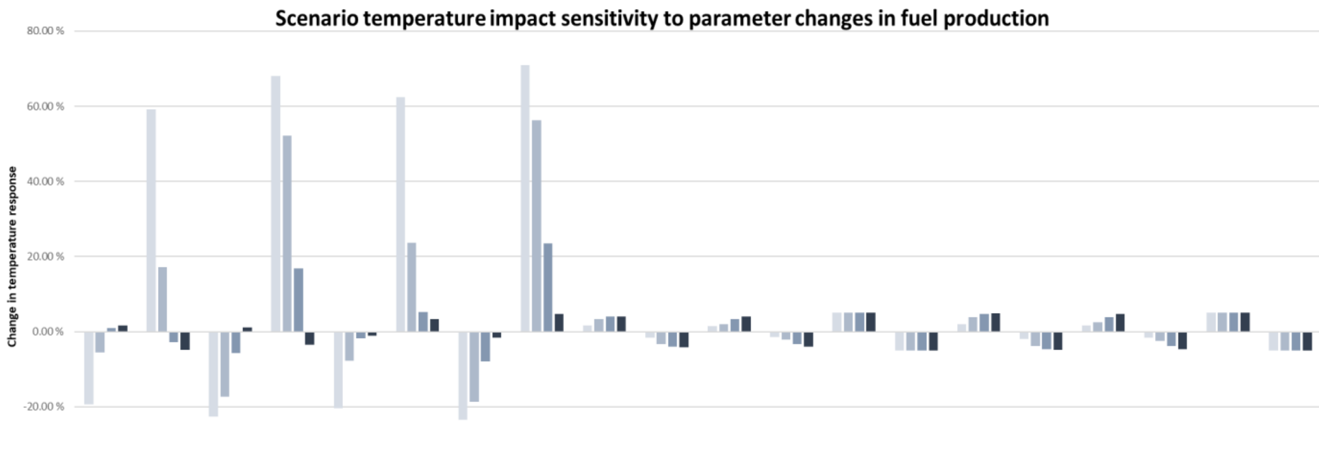


	Decay rate				Litterfall				Soil carbon s1 (maximum loss)				Soil carbon s2 (duration of loss)				Slash harvest rate			
	GRP high		GRP low		GRP high		GRP low		GRP high		GRP low		GRP high		GRP low		GRP high		GRP low	
	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	100 %	40 %	100 %	40 %
2050	3.02 %	-4.72 %	2.55 %	-4.00 %	2.20 %	-2.20 %	2.84 %	-2.84 %	1.51 %	-1.51 %	2.11 %	-2.11 %	-1.11 %	-0.42 %	-1.53 %	-0.74 %	-3.22 %	4.54 %	-3.96 %	5.60 %
2100	-2.13 %	-1.17 %	-2.32 %	0.36 %	5.32 %	-5.32 %	5.35 %	-5.35 %	-0.32 %	0.32 %	-0.34 %	0.34 %	0.19 %	2.38 %	0.20 %	2.51 %	-1.61 %	2.26 %	-4.15 %	5.93 %
2150	-2.86 %	5.28 %	-3.63 %	7.49 %	2.35 %	-2.35 %	3.16 %	-3.16 %	-0.15 %	0.15 %	-0.21 %	0.21 %	0.13 %	-0.21 %	0.18 %	-0.28 %	0.02 %	-0.03 %	-1.46 %	2.10 %
2200	-1.14 %	4.42 %	-1.65 %	7.01 %	0.36 %	-0.36 %	0.60 %	-0.60 %	-0.09 %	0.09 %	-0.14 %	0.14 %	0.06 %	-0.20 %	0.11 %	-0.32 %	0.25 %	-0.35 %	0.13 %	-0.17 %

Figure 19: Shows the BIO-JET scenario temperature sensitivity to parameter alterations in the forest model.

Figure 19 shows the change in temperature response from forest model parameter alterations in the BIO-JET scenario. The parameter test that produced the most significant results was the – 50 % decay rate with the «GRP low» growth rate. This leads to lower temperature response in 2050, but almost 8 % higher temperature response in 2150 and 2200. The main reason for the higher long-term values is because more dead wood accumulates in the unharvested forest in the BAU scenario, leading to longer carbon parity time in the BIO-JET scenario. Higher decay rates produce more significant short-term warming, but lower long-term warming. Changes in soil carbon parameter s1, which regulates how low the soil carbon can sink after harvest, had a modest effect in the short term, but negligible effects for long-term warming. Parameter s2 regulates the duration of the soil carbon loss, and a – 50 % change in this parameter prolong the duration, leading to the largest soil carbon loss from harvesting appearing in

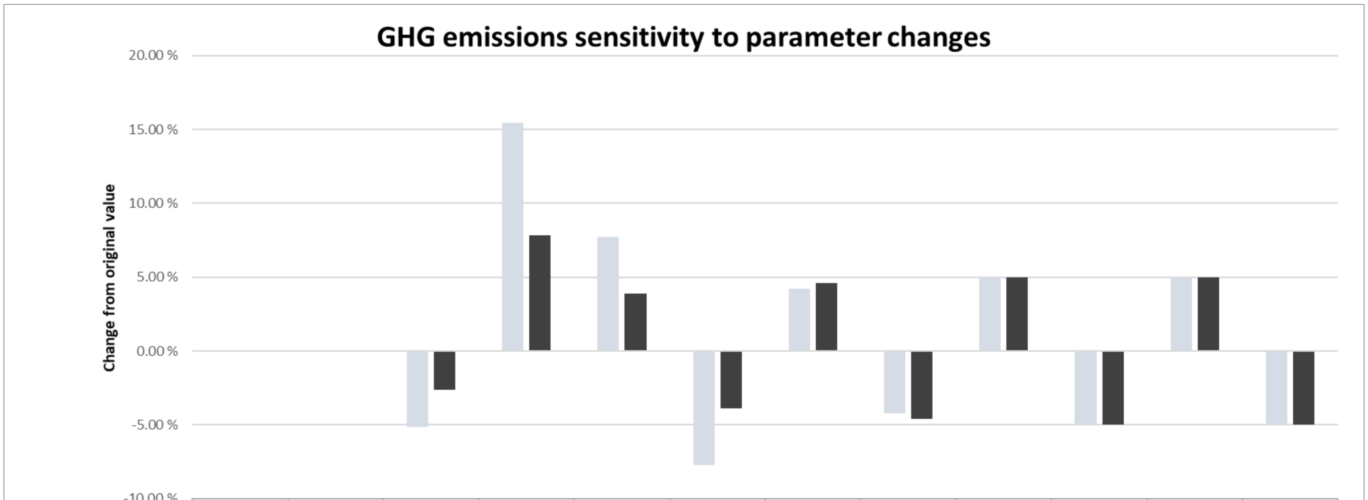
2100 instead of 2050. The figure also shows that 100 % slash harvest rate could lower the climate impacts in the BIO-JET scenario in year 2050 and 2100, but the effect is lower in the longer term.



	MCE Wood-to-Ethanol				MCE Ethanol-to-Jet				Biofuel production emisisons				Fossil fuel production emisisons		Fossil fuel production emisisons		Fossil fuel production emisisons		Fossil fuel production emisisons	
	BIO-JET GRP high		BIO-JET GRP low		BIO-JET GRP high		BIO-JETGRP low		BIO-JET GRP high		BIO-JET GRP low		BAU		BIO-JET GRP high		BIO-JET GRP low		AVI-RED	
	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%	(+) 50%	(-) 50%
2050	-19.34 %	59.07 %	-22.54 %	68.08 %	-20.46 %	62.42 %	-23.47 %	70.88 %	1.68 %	-1.68 %	1.40 %	-1.40 %	5.00 %	-5.00 %	1.96 %	-1.96 %	1.63 %	-1.63 %	5.00 %	-5.00 %
2100	-5.61 %	17.16 %	-17.35 %	52.11 %	-7.79 %	23.70 %	-18.72 %	56.21 %	3.27 %	-3.27 %	2.05 %	-2.05 %	5.00 %	-5.00 %	3.83 %	-3.83 %	2.41 %	-2.41 %	5.00 %	-5.00 %
2150	0.87 %	-2.77 %	-5.63 %	16.86 %	-1.81 %	5.27 %	-7.85 %	23.52 %	4.02 %	-4.02 %	3.33 %	-3.33 %	5.00 %	-5.00 %	4.72 %	-4.72 %	3.91 %	-3.91 %	5.00 %	-5.00 %
2200	1.56 %	-4.79 %	1.13 %	-3.44 %	-1.17 %	3.41 %	-1.57 %	4.67 %	4.10 %	-4.10 %	4.05 %	-4.05 %	5.00 %	-5.00 %	4.82 %	-4.82 %	4.77 %	-4.77 %	5.00 %	-5.00 %

Figure 20: Shows the change in temperature response due to parameter changes in fuel production (MCE = Mass Conversion Efficiency).

Figure 20 shows the sensitivity of temperature response to parameter alterations in the fuel production process. The wood-to-jet mass conversion efficiency is of great importance for the short-term impacts in the BIO-JET scenario, since it impacts both forest carbon and the biofuel production emissions, as shown in Figure 18. Negative changes in the conversion efficiency leads to greater changes in the forest carbon compared to enhanced conversion efficiency, and this effect is amplified because younger forest is harvested to cover the increased feedstock requirements. A reduction in the wood-to-ethanol mass conversion efficiency of 50 % increases the temperature response by around 70 % in 2050, and just below 60 % in 2100. In 2200 it leads to 4 % lower warming. The ethanol-to-jet mass conversion efficiency has very similar effect, but the warming is slightly higher. This is because production emissions were ascribed to the ethanol at the product gate, so increased demand for ethanol leads to higher production emissions, while increased demand for wood for ethanol production, although unrealistic, does not lead to higher biofuel production emissions. Changes in the production emissions of jet-biofuels has a much lower impact for the temperature response than mass conversion efficiency. The fossil jet-fuel production emissions are of modest importance for the BAU and AVI-RED scenarios, and are less important for the short-term effects in the BIO-JET scenario, compared to long-term effects due to the high share of biogenic atmospheric CO_2 in the short-term.



	BIO-JET MCE Wood-to-Ethanol (+) 50%	BIO-JET MCE Wood-to-Ethanol (-) 50%	BIO-JET MCE Ethanol-to-Jet (+) 50%	BIO-JET MCE Ethanol-to-Jet (-) 50%	BIO-JET Biofuel production emisisons (+) 50%	BIO-JET Biofuel production emisisons (-) 50%	BIO-JET Fossil fuel production emisisons (+) 50%	BIO-JET Fossil fuel production emisisons (-) 50%	BAU Fossil fuel production emisisons (+) 50%	BAU Fossil fuel production emisisons (-) 50%	AVI-RED Fossil fuel production emisisons (+) 50%	AVI-RED Fossil fuel production emisisons (-) 50%
Emissions in 2030	0.00 %	0.00 %	-5.15 %	15.45 %	7.72 %	-7.72 %	4.23 %	-4.23 %	5.00 %	-5.00 %	5.00 %	-5.00 %
Cumulated (2019 - 2030)	0.00 %	0.00 %	-2.61 %	7.82 %	3.91 %	-3.91 %	4.61 %	-4.61 %	5.00 %	-5.00 %	5.00 %	-5.00 %

Figure 21: Shows sensitivity of GHG emission results to parameter alterations.

Figure 21 shows the change in GHG emissions resulting from parameter alterations. Ethanol-to-jet mass conversion efficiency is also the most impactful parameter in this category. Overall, mass conversion efficiencies are the most important parameters for the results, with especially negative mass conversion efficiency alterations having great impacts for the BIO-JET scenario results. The other parameters tested here had modest effects on the results, while changes in soil carbon parameters are of little importance for the results. The sensitivity analysis is summed in Table 10.

Table 10: Summary of results from sensitivity analysis

	Temperature			GHG emissions		
Parameters	BAU	BIO-JET	AVI-RED	BAU	BIO-JET	AVI-RED
Decay rates	-	**	-	-	-	-
Litterfall	-	**	-	-	-	-
Soil carbon	-	*	-	-	-	-
Slash harvest rate	-	**	-	-	-	-
Mass conversion efficiency	-	****	-	-	**	-
Biofuel production emissions	-	**	-	-	**	-
Fossil fuel production emissions	**	**	**	**	**	**

* Low sensitivity ** Modest sensitivity *** High sensitivity **** Very high sensitivity

Evaluation of assumptions and parameters

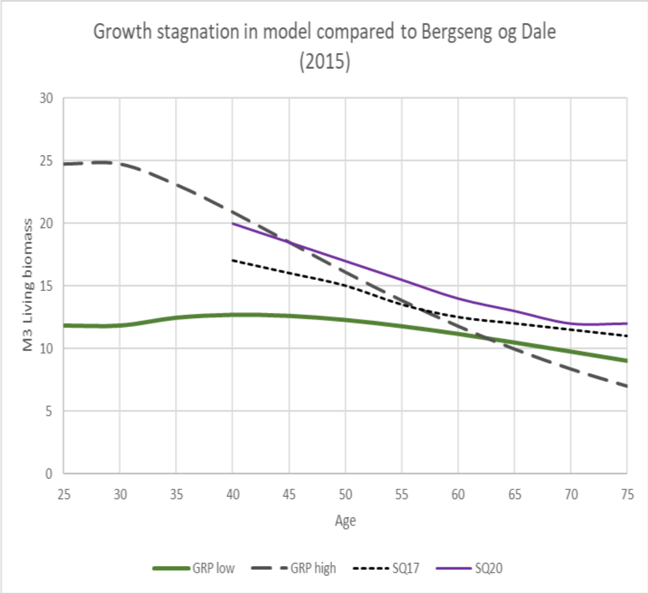


Figure 22: Shows the stagnation of biomass accumulation under the two growth rate versions, compared to values for site quality 17 and 20 Western Norwegian spruce forest plots (values taken from figure 1 in Bergseng and Dale (2015).

The atmospheric CO_2 from changes in the forest carbon pool dwarfed the production emissions and was even greater than the fossil CO_2 emissions from aviation, for the first decades in the BIO-JET scenario, (Figure 15) showing that the accuracy of the forest model is of high importance for the results. Here follows an evaluation of the forest model, with emphasis on how different parameters and assumptions represent reality, and how achieving a more realistic approach can affect the results.

The future growth of the forest is uncertain, but very important for the results. Two different sets of growth rate parameters for the carbon accumulation in living biomass are used in the presented results. One was adjusted to show a lower carbon accumulation than the measured in harvest-ready forest, while the other is based on the actual estimated average carbon in living biomass from the national forest inventory, but has faster growth and decline. These were considered to represent two extremes, since low carbon density and slow growth leads to relatively higher and longer-lasting impacts, while high biomass density, fast growth and fast decline leads to lower climate impacts. However, in comparison to Bergseng and Dale (2015), the biomass accumulation between age 55 and 75 is lower for both growth rates compared to estimates for SQ 17 and SQ 20 forest in Western Norway. If this is true further on with age, then the long-term climate impacts from changes in forest carbon might be underestimated under both growth rates. I did not find the original source which the referenced figure is based on, and other estimates from literature were difficult to translate into comparable values.

The dead wood pool is regulated by the litterfall, decay rates, and the carbon accumulation in living biomass from «GRP original». This way the dead wood pool in the model is decoupled from changes in living-biomass accumulation. The pool constitutes a maximum of between 13 and 16 percent of total aboveground (living plus dead wood) carbon, depending on the growth rate parameters. The latest estimate of dead wood based on the forest inventory found values at around 45 m3 per ha in old-growth high site quality spruce forest (Storaunet et al. 2011). In comparison, the forest stand model has a dead wood pool at 21.5 t C after 100 years of growth, and this corresponds to over 108 m3 dead wood per hectare as dead wood has slightly lower carbon per volume than fresh wood (Dalsgaard et al. 2015). However, the dead wood estimates from the forest inventory only includes pieces of wood larger than 10 cm in diameter, while the litterfall in the model encompass all detritus, including roots, needles and small branches (Kurz et al. 2009, Asante and Armstrong 2012). These components have much higher turnover rates, but also faster decay (Kurz et al. 2009). Results from several other studies indicate that dead wood constitutes up to 22 to 37 percent of total aboveground biomass in old growth spruce forest in Scandinavia, even when roots and needles are neglected (Dalsgaard et al. 2015). Furthermore, a global study of forest carbon that includes foliage and roots found that carbon stored in dead wood can increase from around 20 % for 100-year-old forest, to over 40 % of above-ground carbon in 300-year-old forest (Liu et al. 2014). This indicates that the size of the dead wood pool, and the accumulation of dead wood in old forest, might be underestimated in the model. Especially the latter point will lead to underestimation of long-term climate impacts, as shown in the sensitivity analysis for -50 % decay rate, which leads to higher accumulation of dead wood in old forest.

Otherwise, the results from Storaunet et al. (2011) indicates that the general profile of the dead wood curve is suitable, since they reported a decreasing dead wood pool after harvest until around 20 to 50 years of growth before accumulation starts again. However, they also reveal that dead wood varies considerably between sites, as 8 % of the old growth spruce forest had 80 m3 dead wood or more, constituting 40 % of total dead wood in old growth spruce forest in their estimates. The large variability shows that local differences can be important for dead wood pool modelling.

Large uncertainties are tied to how harvesting affects the soil carbon. The soil carbon loss function in the model was originally constructed to reproduce the results from a study by Olsson et al. (1996). Later, a literature review was conducted by Dalsgaard et al. (2015) looking into carbon accumulation in living biomass, dead wood and soil, in Scandinavian forest. Based on results from the available relevant studies they conclude that after clear cutting, the total soil carbon can be expected to decrease by up to 7 to 22 percent. The loss will last for decades before starting to accumulate again, and it is uncertain whether it is restored within the timeframe of a rotation period. They operate with soil carbon values of around 140 t C per ha, leading to losses between 9.8 to 30.8 t C. Therefore, the soil carbon loss in this forest model, at 12 t C, is within the lower range of their estimates. The soil carbon loss in the BIO-JET scenario has a modest effect on the magnitude of the short-term climate impacts, but insignificant effect in the long-term. However, while the model simulates a separate soil carbon pool that flattens after reaching 60 t C, much of the soil carbon in old forest systems comes from decaying dead wood, which is generated as a share of living biomass. Therefore, since both the living biomass and the proportion of biomass in dead wood are increasing with age in old forests, it is reasonable to assume the soil carbon would continue to increase over longer periods (Dalsgaard et al. 2015). This is also indicated by findings of several thousand years old dead organic matter in forest soils (Grønlund et al. 2010). Following from the points mentioned above, the magnitude of the soil carbon loss could be underestimated, and the duration of decreased soil carbon might also be underestimated.

The age distribution of the forest is very coarse, and this might lead to more biogenic emissions than if the resolution were higher. This mostly affects the «GRP low» results, because 20 % of age class 3 was harvested in the BIO-JET scenario with this model. The way the forest model is constructed now, the average density for the whole class applies to age class 3. However, if the age classes were further divided in say 4 age groups, then only the oldest and most dense age group in age class 3 would have been harvested with the "GRP low" set, leading to less harvesting and lower climate impacts. For the «GRP high» results, the effect would be smaller because all of age class 4, and only a tiny portion of age class 3 was harvested. However, the harvest regime in the model is not based in realistic practice, and this likely has a much greater impact on the biogenic carbon emissions.

The assumption that the oldest forest is always harvested first is flawed since the current driver of harvesting is usually economic considerations. Although the biomass density is important, the economic aspects of forestry are additionally affected by, amongst others, timber price, accessibility of the forestry site to roads or docks and the steepness of the site. Since much of the old forest in Western Norway is difficult to harvest due to steep terrain and limited accessibility from roads, harvesting in younger forest might be the most economic option in many cases (Roll-Hansen 2015). Bergseng and Dale (2015) found that the average age of harvested spruce forest in Western Norway from 2011 to 2015 was between 51 to 58 years-old, for spruce with site quality 17 to 26. This is younger than age class 3 throughout the biofuel production period in the forest model. Following from this, the harvesting regime used in this thesis can lead to significantly underestimated climate impacts since harvesting in age class 3 is likely to occur simultaneously as age class 4, leading to larger changes in the forest carbon. In this context it should be noted that the reason for harvesting young forest is linked to price development of building materials (Bergseng and Dale 2015). Therefore, the larger biomass volume in older forest could potentially be more important if biofuel production is driving harvesting.

When considering the economic aspect of biofuel production, Rambøll (2017) emphasize that using the entire biomass of trees for biofuel production is unlikely, because parts of the stem are much more profitable when traded as building materials. They suggest using 20 % of the wood for this purpose, which could lead to storage of carbon in buildings for decades, and potentially substitute emission intensive steel or concrete production. A meta-analysis by Sathre et al. (2010) found an average substitution factor of 3.9 t CO_2e per ton dry wood utilized in long lived products. If such a case were considered in this thesis, then the climate impact from the BIO-JET scenario could be positively affected. However, using 20 % of the biomass for building materials would require a proportionally increased harvest volume for reaching the biofuel goals. This would lead to more harvest in young forest, which involves higher biogenic CO_2 emissions as demonstrated by the sensitivity analysis for wood-to-jet-fuel conversion efficiency. Further research would be necessary to estimate how beneficial wood-product substitution would be in the BIO-JET scenario. Wood-product substitution of emission intensive materials will certainly contribute to preferable temperature response when timeframes of several centuries are considered, with the method used in this thesis, because fossil CO_2 emissions stays in the atmosphere while the carbon balance in the forest is restored.

The emission values for wood-to-ethanol conversion is within the range of what has been estimated in literature previously. Bright and Stromman (2009) lists several studies with values ranging from 9.3 to 21 g CO_2e MJ^{-1} for well to product gate for biochemical systems, most of which is in the upper range. For the second step, ethanol-to-jet-fuel, finding emission estimates proved difficult. This process also involves loss of energy in the conversion process, since deoxygenation either requires partial combustion of the biomass, or addition of hydrogen (Karatzos et al. 2017). In the case of Han et al. (2017), steam reforming of natural gas was used for producing the hydrogen. Therefore, additional emissions in this production step could be significant. In the results presented here, production emissions constitute a small share of the total emissions when forest carbon is taken into account, and the potential extra emissions from ethanol conversion is therefore unlikely to have an important effect on the temperature impacts. However, they might contribute for further lower the substitution potential in conventional bioenergy LCA.

The mass conversion efficiencies in wood-to-Jet-fuel production process is the most important parameters in the sensitivity analysis. The wood-to-ethanol step is based on the best-case biochemical system in Bright and Stromman (2009), where 261 l ethanol can be produced from 1 tonne wood, translating into a mass conversion factor of 0.21 g g^{-1}. Their worst-case conversion efficiency is around 10 % lower. According to Singh et al. (2010) the maximum theoretical yield of ethanol is 0.32 g g^{-1} wood, so there is limited potential for improvements in this process. In this context, it is worth noting that

Killingland (2013) used an ethanol yield of 0.51 g g^{-1} wood, according to table 6-5 in their report, which is the estimated theoretical yield of sugar-to-ethanol (Singh et al. 2010). The ethanol-to-jet-fuel conversion step is based on Han et al. (2017), where a jet-biofuel yield of 0.42 g g^{-1} ethanol was used. This is in the upper-range between 0.12 to 0.44 g g^{-1} mass conversion efficiencies for ethanol to jet-fuel reported by Killingland (2013), and thus might represent a very optimistic case. Lower conversion efficiencies could render the BIO-JET scenario much more impactful, as shown by the sensitivity analysis.

The climate impacts assessed here are only calculated for the fuel scenarios between 2019 and 2030. If the share of 30 % biofuels were maintained for a longer period, it would take longer for the climate benefits to occur because the high initial biogenic emissions by far outweighs the fossil GHG emission savings in BIO-JET. At the same time, the higher fossil CO_2 emissions in the BAU scenario would also be maintained for a longer period and accumulate in the atmosphere, so when the benefits of using bioenergy first appear, they would be considerably larger. The production of biofuel from forest resources would require large investments and installation of infrastructure, so presumably the biofuel production will continue longer than 2030. If such a case were considered in this thesis, the estimated long-term climate impacts would be larger in 2100 but might become lower at the end of the assessed timeframe (2150 – 2200). However, with the forest model and the conversion efficiency applied here, maintaining the same share of biofuels could only be possible for a few years after 2030, because more wood is required than what is replaced by new growth.

Interpretation of results

The results presented in this thesis confirms that the GHG mitigation potential for wood-based jet-biofuels largely depends on the inclusion of changes in the forest-climate functions, and the time perspective of the evaluator. If only short-term impacts were considered in this thesis, the long-term increased sequestration in the forest would be ignored, leading to business as usual appearing highly preferable to the biofuel option. In the same way, considering only the long-term effects would omit the consequences of the substantial reductions in forest carbon due to biofuel production, which could lead to BIO-JET seeming clearly preferable to BAU for climate change mitigation. This underlines the statement by Breton et al. (2018), that considering climate impacts only with a single time-frame / point or climate metric can lead to incomplete or potentially counterproductive advice for decision-making, which is the reason why several impact indicators at different points in time are presented in this thesis.

Which point in time or which impact indicator that should be given highest priority is related to value choices, and this is a central concern in ongoing scientific discussions regarding bioenergy, temporary carbon storage, climate metrics comparing greenhouse gasses, and climate discount rates (Brandão and Levasseur 2011, Myhre et al. 2013b, Cherubini and Tanaka 2016). Giving high priority to short-term warming might arguably undervalue the importance of a "permanent" change in the earth's energy balance caused by fossil CO_2 emissions (Bright et al. 2012b). However, only including long-term effects might disregard the potential contribution of short-term warming to early strengthening of climate feedback mechanisms, such as the release of carbon dioxide and methane from permafrost (Jørgensen et al. 2014). Additionally, short-term warming could contribute to the early rate of temperature change, which might leave human and ecological systems with shorter time to adapt. Knowledge regarding feedback mechanisms in the climate system, and their incorporation in climate models, are still a big factor of uncertainty in climate projections (Flato et al. 2014). Topics like species migration rates and rapid evolution are also highly uncertain factors in ecological models. Following from this, future advances in climate science might shed light on the relative importance of short-term climate change mitigation, and thus short-term impacts should not be neglected in bioenergy assessments.

It should be noted that the temperature impacts assessed here only relates to surface temperatures, and thus do not reflect the warming effect on the oceans. Additionally, parts of the CO_2 emissions to the atmosphere are taken up by the ocean due to the impulse response function (equation 11). Thus, this part of the CO_2 emissions do not contribute much to climate change, but will still contribute to ocean acidification which is also an important environmental concern (Ripple et al. 2017). Withers et al. (2015) assessed the climate benefits from liquid biofuels based on woody biomass, applying several impact indicators. They also found that temperature break-even for biofuels lags behind atmospheric CO_2, but more notably, they show that economic damages significantly lag behind temperature impacts, especially when applying modest damage discount rates. This indicates that economic damage indicators, which tries to incorporate all harm to humans, might provide important insights when assessing wood-based biofuels. Furthermore, it indicates that the large short-term temperature impacts in the BIO-JET scenario presented here have the potential for increasing climate change induced harm to humans, although the long-term impacts are lower than in BAU.

It is very important to emphasize that the climate impacts presented here do overlook many biogeophysical-and-chemical effects induced by harvesting of forest. Albedo change is the additional effect that is most frequently mentioned in literature, and it should be included when assessing the full climate impact of forest bioenergy (Bright et al. 2012b). Holtsmark (2015a) applied albedo change modelling to the original forest stand model and found that the albedo effect could not outweigh the radiative forcing from increased atmospheric CO_2 from harvesting. However, the growth rate parameters applied in the study were based on the average biomass density for spruce forest in Norway. The growth rates used here are adjusted to Western Norwegian conditions, leading to decreased area effect per harvested volume and faster regeneration of carbon levels in the forest, as illustrated in Figure 9. On one hand, this leads to lower forest carbon impacts in this thesis, which could increase the relative importance of albedo. On the other hand, the larger biomass density leads to decreased area-use per harvested volume, which would result in less albedo-change per unit biomass harvested, which again decreases the relative importance of albedo change. Furthermore, Holtsmark simulated harvest in 100-year-old forest, while the forest in this thesis is younger, which leads to higher CO_2 impacts. The albedo values applied in the study were based on a site in Hedmark county. Hedmark has a longer-lasting seasonal snow-cover than the coastal areas of Western Norway (Hanssen-Bauer et al. 2009), and the current snow cover is projected to shrink within the timeframe of this study (Iversen et al. 2005, Scott et al. 2019). These factors could render albedo change less impactful if applied to the Western Norwegian forest model. However, further research would be necessary to gain insight into how albedo change would affect the estimated climate impacts from biofuel production in this thesis.

The omissions of many important forest-climate functions in assessments of wood-based jet-biofuels, strongly suggest that further research is necessary for promoting good decisions on how to manage forests in a climate change perspective. From the findings in this thesis, three main suggestions for further research can be drawn. Firstly, further efforts into parameterizing forest-climate functions in forest models are needed for being able to sufficiently account for the full climate impact of harvesting forest. Second, forests climate impacts can vary strongly based on local conditions, such as the growth rate of trees and snow-cover patterns. Therefore, researchers should use forest models that are specified for local conditions. Thirdly, presenting results with several time perspectives and impact indicators, and quantification of uncertainty, are necessary for sufficiently informing decisionmakers, even though this might add complexity to assessments. In a broader perspective, further knowledge is needed on the importance of emission timing. This could enhance our understanding regarding the mitigation potential of temporary carbon storage in forests or building materials, and the consequences of short-term warming involved with wood-based biofuels. Research on how forests will be to be affected by climate change, such as mapping of which areas are at high risk for forest fires and insect outbreaks, is highly important for future managing of forest. If a certain forest area is at high risk for disturbance, this might also affect how the carbon debt from harvesting in this area should be accounted for. Knowledge on future disturbance risk and the incorporation in forest models could therefore be highly important both for climate change mitigation, and adaptation.

6. Conclusion and outlook

The aim of this thesis is to assess the GHG mitigation potential of introducing wood-based biofuels in the Western Norwegian aviation sector, taking loss of forest-carbon and avoided sequestration in the forest into account. The general findings are that phasing-in of wood-based jet-biofuels can mitigate fossil GHG emissions in the aviation sector, but the reductions are partly offset by increased processing emissions in other sectors. The loss of forest carbon and the avoided sequestration in the forest leads to higher atmospheric CO_2 concentration for several decades to over a century, compared to if no mitigation efforts are introduced. Although the CO_2 is eventually sequestered by new forest, the reduction in forest-carbon leads to climate warming. These results, produced with a bioenergy system based on relatively fast-growing forest with high biomass density, indicates that the mitigation potential is unlikely to be higher in other parts of Norway, the potential exception being be if albedo change is included in areas with long-lasting seasonal snow cover. Mitigation strategies aimed at restricting air-travel can be much more effective for mitigating climate change due to the expected emission growth under business as usual, and because no additional processing emissions or release of forest carbon is involved.

The results depend on many uncertain parameters, and forests have other functions in the climate system which are not included in this assessment. Therefore, an urgent need exists for comprehensive forest models able to capture local variations in forest-climate functions, allowing for proper scenario assessments of forest-based climate change mitigation strategies. Future research should focus on comparing different strategies, so that decisionmakers can recognize the range of options for climate change mitigation through forest management.

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