Bioenergy Systems in Sweden – Climate impacts, market implications, and overall sustainability

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Förord

Energimyndighetens forskningsprogram Bränsleprogrammet hållbarhet pågick mellan 2011 och 2017. Resultaten från programmet till och med 2016 redovisas i syntesrapporter för programmets delområden. Syftet med syntesrapporterna är att sammanställa kunskap, att identifiera kunskapsluckor som behöver belysas vidare samt att placera och diskutera de sammanvägda forskningsresultaten i ett större energi- och samhällsperspektiv, bland annat med koppling till miljökvalitetsmål och skogspolitiska miljö- och produktionsmål.

I produktionen av bioenergi behöver man ta hänsyn till flera faktorer för att den ska anses vara hållbar. En av de viktigaste är klimataspekten, dvs hur bioenergisystemet påverkat nettoutsläppen av växthusgaser vid utvinning, produktion, transport och energiomvandling samt vilka kolförändringar som sker i landskapet till följd av ett förändrat brukande. Frågan är komplex och det är därför angeläget att kunna bedöma olika bioenergisystem ur klimatperspektiv utifrån fakta och vetenskapliga analysmetoder för både kort och lång sikt.

Denna rapport fokuserar på klimatpåverkan av bioenergisystem i Norden och metoder för att utvärdera dessa effekter. Rapporten behandlar projekt inom Bränsleprogrammet hållbarhet, näraliggande enskilda projekt som Energimyndigheten finansierar, samt annan nationell och internationell näraliggande verksamhet. Målgruppen är forskare, myndigheter, företag och branschorganisationer inom bioenergisektorn samt övriga med verksamhet som berörs av bioenergin. Rapporten skrivs på engelska för att möjliggöra en mer omfattande resultatspridning och nå en bredare publik.

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Rapporten har granskats av Energimyndigheten. En referensgrupp har lämnat värdefulla synpunkter på arbetet. Författarna står för analys och slutsatser. Det är vår förhoppning att denna syntesrapport ska ge läsaren en inblick i kunskapsläget på detta område.

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Innehåll

1 Summary

The aim of this report is to put findings from selected research projects in a research program, "*The Biomass Fuel Program*", 2011–2015, financed by the Swedish Energy Agency, into context and, if possible, to synthesize the results. Focus is on climate impacts of bioenergy systems in the Nordic countries and methods to evaluate these effects. Bioenergy is already today one of the largest contributors in the Swedish energy system (23% of supplied energy in 2014) and a major part of the biomass behind that input originates from our managed forests. This has been made possible as a result of good forest policy put in place by the end of the 19th century, when it was realised that the Swedish forests were low in growing stock due to over logging at the same time as a growing forest industry demanded more feedstock. Since then the growing stock has doubled. This shows that *with appropriate forest policies in place it is possible to meet an increased demand for forest biomass without compromising future woody biomass supply and carbon sequestration in forests – rather the opposite.*

A substantial part of terrestrial C is found in the soil. Therefore, effects of biomass production systems on the soil C pool has been given attention in the climate change mitigation research and debate. *The potential to increase the soil C pool by cultivation of perennial crops is large in many agricultural soils, whereas the challenge in most forest soils is to maintain or slightly increase the C pool.*

Studies within the program show that mechanical soil disturbances (mechanical site preparation, stump harvest) do not increase net soil CO_2 emissions on upland forest soils in Sweden. A short- to medium term "loss" in soil C, when also logging residues (slash, stumps, small-diameter trees) are harvested can be observed, however, this cannot be used as a basis for final conclusion about the climate impact.

To determine climate impacts of forest systems, it is the long-term impacts on soil C that must be evaluated. Model approaches suggests that the long-term effect of more intense harvest on soil C is moderate with similar results for two common soil C models used (Q model, Yasso model). However, the pathways for long-term carbon in forest soils is still uncertain and more process-based knowledge is needed to understand soil C dynamics and thus, to model the long-term development.

A bit of a "soil C dilemma" is that decomposition of organic material in the soil and thereby mineralisation of essential nutrients is what drives the production systems together with water, CO_2 and photosynthesis. Indeed, studies within the program show that decreased soil C pools as a result of intense mechanical site preparation were fully compensated for by increased tree growth. Consequently, *a significant short- to medium-term loss in soil C cannot be used as a basis for conclusion on the climate impact without including effects on the growing crop.* This applies to both agricultural and forestry systems.

Soil C dynamics on farmland is however somewhat different. For accurate estimates of the GHG-balance, studies within the program highlights the importance of including the increase in soil and biomass C when changing from fallow to dedicated perennial bioenergy crops such as willow or poplar. Crediting such C sinks could also

significantly increase the incentives for new dedicated biomass plantations. Policy challenges for such C-sink credits include permanence and leakage of C sequestered in soils and crops together with possible indirect land-use change (iLUC).

To fully capture the temporal or the spatial dimensions of forest C pool changes, relevant temporal or spatial scales, i.e. a full rotation period or the whole production landscape, have to be considered. Studies within the program have shown that, depending on the initial forest conditions, a managed forest landscape may experience initial C loss (a lot of mature forests) or C gain (a lot of recent plantations and young forests) as well as a stable C-stock with an even distribution of forests in different development stages. Therefore, also short-term responses at the landscape level has to be interpreted with care.

Studies within the program show that if nutrient rich slash is harvested a moderate negative effect on forest production can be expected unless the production loss is compensated for by more ambitious silviculture promoting forest growth, whereas stump-harvest appears to have no such effect. This slash effect together with forest owner response to a new market potential, potentially with more ambitious measures to increase growth, are not often addressed when analysing effects of using forest biomass for energy on the total forest C pool.

In the Nordic countries biomass for energy is an integrated assortment that interacts with saw timber and pulpwood in forests managed over long rotations. Therefore, *it is necessary also to include the C stored in durable forest products in analyses of the climate impact of managed forests.* But analyses including C stock changes in soil, trees and forest products still do not capture the full impact on the GHG balance. *Input energy used in forest management and the substitution effect of using biomass for energy or wood instead of other materials with their C footprint have to be included as well.*

This is exemplified in a modelling study within the program. The study used national forest inventory data for the managed forests in Sweden and modelled its development over 100 years. A business as usual scenario (BAU) was compared with a production scenario including silvicultural measures to increase forest production and a set-aside scenario where the area of set-aside productive forestland was doubledas as compared to the current area. Based on the C-stocks in soil, trees and forest products the set-aside scenario falls out as the best strategy in mitigating climate change followed by the production scenario. This is in line with other studies limited to C-stocks suggesting that leaving the forest to grow and sequester C could be a good climate mitigation strategy. However, when also the management emissions and the substitution effect of using the harvested biomass instead of other carbon intensive materials or fossil fuels were included in the analysis the production scenario was the best climate mitigation strategy. After 90 years also the BAU scenario fell out as a better option than the set-aside scenario. The superiority of the production scenario was further enhanced when also logging residues (slash and stumps) were harvested and used for energy purposes.

An important message from the above described study is that *the effect of using wood to substitute materials and fossil fuels continues to deliver climate mitigation over time, whereas the positive effect of setting aside forestland diminishes over time.* Furthermore, long-term carbon storage in forests is a risky business due to snow, wind, fire and forests pests – not the least in a changing climate. Hence, *a management strategy that maintains or increases the forest C stocks while supplying the market with a large annual yield will generate a positive and sustained GHG mitigation over time.*

The fact that material substitution often has a higher climate mitigation efficiency per unit of wood used than direct fossil fuel substitution is one reason why cascading use of wood has been suggested as a valid strategy with biomass for energy as the last wood-use option. But *due to the extensive use of fossil fuels in the transport sector the climate mitigation potential is much higher for direct fossil fuel substitution than for material substitution.* Furthermore, in managed forests a lot of low quality and low-priced woody biomass fall out as residues, also including some stemwood, that currently is not suitable for any other products. This is one important reason why substantially increased demand for woody biomass is unlikely to lead to substantial increase in forest harvest levels. On the contrary, *decisions on harvest of trees in managed forests are driven by other, more valuable, assortments than biomass for energy.*

Life cycle assessment (LCA) is a common tool when comparing climate mitigation strategies in managed landscapes. There are still many uncertainties and thus challenges involved in LCA studies. E.g. *too much focus on near term GHG emission reduction targets can be counterproductive in combating climate change.* Studies that use a stand-level perspective have been questioned as potentially misleading due to that the outcome is strongly influenced by how the carbon accounting is made. *A larger landscape perspective, rather than the stand/field level, would in general be the appropriate scope for studies that intend to inform policy development.* This is particularly the case for managed forests where growth and thereby carbon sequestration takes place in stands of different ages in the landscape allowing sustainable annual harvests in some of the stands. Policy makers also have to consider regional and local conditions for biomass production and energy system conversion. *Common policies over large areas are unlikely to be optimal.*

System boundaries are critical for the outcome of LCA-analyses, and the trend for research-based LCAs is to make studies more complete and complex by expanding the system boundaries. For bioenergy systems, extended boundaries that include land use change have been widely discussed. Indirect land use change (iLUC) estimates tend to yield more uncertain results than direct land use change (dLUC) estimates, effecting the outcome of the LCA studies. Although recent focus has been put on dLUC and iLUC, the reasoning is relevant also for indirect effects on markets for forest products (iWUC) and energy (iFUC). *Many studies take for granted that iLUC and other indirect effects are per definition associated with increases in net emissions, which is not always true as land and forest products also can sequester carbon.*

To be useful for policy, analyses of the GHG-balance of biomass production systems have to include all C-stock changes, GHG emissions, and the substitution effect of the full suite of products produced. *Ideally a combination of biophysical, climate and socio-economic models is used to capture the full climate effects of bioenergy, including effects on parallel industries (wood products, agriculture and energy).* As the complexity of the GHG cause and effects of emerging bioenergy markets increases additional assumptions and uncertainties have to be introduced in the modelling. It is therefore important to first look carefully into these assumptions before accepting the results of the study and to be aware of that different methodological approaches and metrics capture the C dynamics in contrasting ways. Hence, it is important that the choice of LCA-methodology and choice of data are in line with the research question and the intended use of the results.

The choice of counterfactual without an evolving bioenergy market often explains why studies end up with opposite results concerning the climate benefits of biomass for energy. Not the least in forestry where the counterfactual "leaving the trees uncut" have been used in several studies despite the fact that *"leaving the trees uncut" is an unlikely counterfactual in managed Nordic forests where forest fuel is an integrated assortment and decisions to cut is driven by other more valuable assortments.* Lack of empirical data on how the C stock develops as set aside forest stands grow older or when uneven-aged forestry systems are introduced into the studies adds uncertainty to the modelling.

Within their limits, LCA studies in *The Biomass Fuel Program* have shown that many wood pellet value chains on the Swedish market will be able to meet the requirements on net GHG savings according to the renewable energy directive (RED, 2009/28/EC). Studies also show that independent of type of forest feedstock (slash, small-diameter trees, stumps) combined heat and electricity production contributes positively to climate change mitigation in the long-term perspective. The choice of soil C model and feedstock transport distance had limited impact on these results. Studies also suggest that Swedish biofuel production from agricultural or forest raw material have a large GHG reduction compared to fossil fuels.

While focus here has been on GHGs, climate is also influenced by changes in the atmospheric concentration of aerosols, solar irradiation (cloudiness), and land surface albedo. Ideally these effects should also be included in the analyses making them more complete and complex.

There are also other environmental sustainability issues than climate to consider when deploying large-scale bioenergy systems. This includes e.g. long-term site productivity, biodiversity, acidification, eutrophication and mobility of toxic substances (e.g. methyl mercury), all of which have been studied within *The Biomass Fuel Program*. These are environmental issues that also needs to be kept in mind in the operational and policy planning processes as biomass markets develop.

2 Sammanfattning

Syftet med denna rapport är att sätta forskningsresultat från utvalda forskningsprojekt finansierade av Energimyndighetens forskningsprogram *Bränsleprogrammet* (2011–2015) i sitt sammanhang och om möjligt syntetisera resultaten. Focus ligger på klimatpåverkan av bioenergisystem i Norden och metoder för att utvärdera dessa effekter. Bioenergi bidrar redan idag med en betydande andel i det svenska energisystemet och en stor andel av denna bioenergi har sitt ursprung i biomassa från våra skogar. Detta bidrag har möjliggjorts genom en genomtänkt skogspolicy som sjösattes i slutet av 1800-talet, då virkesförrådet i våra skogar var lågt på grund av tidigare års överavverkning samtidigt som en växande skogsindustri efterfrågade mer råvara. Sedan dess har virkesförrådet fördubblats. *Detta visar att med en genomtänkt skogspolicy går det att möta en ökad efterfrågan på vedråvara utan att äventyra framtida tillgång.* I det svenska fallet har råvarutillgången istället ökat.

En betydande del av kolet i våra skogar återfinns i marken. Därför har en hel del uppmärksamhet inom såväl forskning som i debatten riktats mot effekter av produktionssystem för biomassa på kolmängder i marken, vilka påverkar produktionssystemets potential att motverka klimatförändringar. *Generellt kan sägas att potentialen att öka kolförrådet i marken är stor i många jordbruksmarker, medan utmaningen på skogsmarker är att vidmakthålla eller måttligt öka mängden kol i marken.*

Studier inom *Bränsleprogrammet* ger inte stöd för den uppfattning som ofta framförs, att omrörning av skogbevuxna fastmarker (markberedning, stubbskörd) leder till ökad kolomsättning och därmed CO_2-avgång till atmosfären. Andra studier visar att markkolsförluster på kort (10 år) till medellång (30–50 år) sikt orsakade av skogsbränsleuttag (grot, stubbar, klena stammar) i samband med avverkning inte kan användas som grund för slutgiltiga slutsatser om effekten på det framtida klimatet. Den slutgiltiga klimateffekten beror istället på den långsiktiga förändringen av markens kolförråd.

Modelleringsansatser inom *Bränsleprogrammet* indikerar att den långsiktiga effekten av skogsbränsleskörd på markens kolförråd är måttlig och att liknade resultat erhålls oavsett vilken markkolmodell som används (Q-modellen, Yasso modellen). Men fortfarande saknas fullständig kunskap om hur mer stabila och långsiktiga markkolförråd uppstår. Därför behövs mer processförståelse som förklarar markkolsdynamiken och därmed kan användas för att förbättra markkolsmodellerna.

Ett markkolsdilemma är att nedbrytning av organiskt material och därmed frigörelsen av såväl kol som de näringsämnen som finns bundet i det organiska materialet är vad som driver biomassaproduktionen tillsammans med vatten, CO_2 och fotosyntesen. Detta samband stärks av studier inom *Bränsleprogrammet* som visar att minskningar i markkolförrådet orsakade av att mer biomassa skördas samtidigt som marken bearbetas intensivt fullt ut kompenseras av ökad kolinlagring i växande träd. En slutsats av detta är att *signifikanta markkolsförluster inte kan utgöra grund för slutsatser om klimatpåverkan.* För sådana slutsatser krävs åtminstone att också kolinlagringen i den växande grödan ingår.

Studier inom *Bränsleprogrammet* visar att förutsättningarna att öka mängden markkol på jordbruksmark ofta är goda. Det är därför viktigt att inkludera förändringen i kolförråd i mark och gröda vid skattningar av klimateffekten av att till exempel gå från träda till odling av fleråriga energigrödor såsom Salix eller poppel. Att ge en ersättning för sådana kolsänkor skulle kunna vara ett sätt att få fart på biomassaproduktionen på jordbruksmark. Policyutmaningen här ligger i varaktigheten av sådana kolsänkor tillsammans med eventuella indirekta markanvändningsförändringar (iLUC).

För att korrekt åskådliggöra kolförrådsförändringen i skog krävs att relevanta temporala och spatiala skalor såsom hela omloppstider eller hela landskap används. Studier inom *Bränsleprogrammet* har visat att dagens skogstillstånd påverkar kolförrådet i brukade skogslandskap på kort och medellång sikt. Kolförrådet kan minska (mycket äldre och avverkningsmogen skog), öka (mycket yngre och medelålders skog) eller vara i stort sett opåverkat (jämn åldersklassfördelning). Därför bör också kortsiktiga till medellånga effekter på kolförrådet i ett skogslandskap tolkas med försiktighet.

Studier inom *Bränsleprogrammet* visar att om näringsrik grot skördas kan man förvänta sig en måttlig negativ effekt på skogsproduktionen, såvida inte produktionsförlusten kompenseras genom mer ambitiös skogsskötsel inriktad mot ökad produktion, medan stubbskörd inte verkar ha någon sådan effekt. Denna tillväxteffekt och eventuella effekter på skogsägarnas vilja att investera i åtgärder för att öka skogsproduktionen då biobränslemarknaden (bioekonomimarknaden) växer till ingår sällan i analyser av skogsbränsleskördens effekter på kolförrådet i skogen.

I brukade skogar i Norden är skogsbränsle ett integrerat sortiment tillsammans med sågtimmer och massaved. *Det är därför nödvändigt att också inkludera kolförrådet i skogsprodukter vid analyser av brukade skogars klimatpåverkan.*

Men inte heller analyser som omfattar kolförrådsförändringar i mark, träd och skogsprodukter är tillräckligt för att fånga den totala effekten på växthusgasbalansen. *För att fånga hela växthusgasbalansen krävs att även insatsenergin vid brukandet av skogen och substitutionseffekten av att biomassa från skogen ersätter fossila bränslen och material såsom stål och betong ingår i analysen.*

Vikten av att inkludera insatsenergin och inte minst substitutionseffekten visas i en modellstudie finansierad av *Bränsleprogrammet*. Studien utgick från riksskogstaxeringens data för den brukade skogen i Sverige och modellerade dess utveckling över 100 år givet några olika Scenarier. I studien jämfördes ett ”business as usual” scenario (BAU, avverkningsnivån = tillväxten) med ett produktionsscenario, där flera insatser för att öka skogsproduktionen ingick, och ett BAU-scenario där arealen avsatt och därmed skyddad skogsmark fördubblades vilket minskade den möjliga årliga avverkningsnivån. Baserat enbart på kolförrådsförändringar i mark, träd och produkter föll scenariot med ökade avsättningar ut som det bästa alternativet följt av scenariot med insatser för ökad skogsproduktion. Dessa resultat ligger helt i linje med andra studier begränsade till enbart kolförrådsförändringar, vilka visar att det bästa alternativet ur klimatsynpunkt är att lämna skogen orörd. Men då även substitutionseffekten inkluderades i analysen föll produktionsscenariot ut som det bästa och efter ca 90 år var även BAU-scenariot överlägset scenariot med ökade avsättningar. Överlägsenheten för produktionsscenariot förstärktes ytterligare då också skogsbränsle i form av grot och stubbar togs ut i samband med avverkningarna.

Ett viktigt budskap från denna studie är att *substitutionseffekten av att ersätta material och fossila bränslen fortsätter att motverka klimatförändringar över tid, medan effekten av att avsätta mer mark eller att helt lämna skogen för att lagra kol klingar av över tid.* Det är också viktigt att tänka på att långsiktig lagring av kol i skog är riskfyllt på grund av snö, vind, brand och skadegörare – inte minst i samverkan med klimatförändringar. Därav följer att *en förvaltningsstrategi som vidmakthåller eller ökar kolförrådet i skog samtidigt som den förser marknaden med en stor årlig skörd uthålligt kommer att motverka klimatförändringen.*

Det faktum att materialsubstitution per enhet ved som används ofta motverkar klimatförändringar mer effektivt än då veden direkt substituerar fossila bränslen är ett skäl till att kaskadanvändning av vedråvara har framförts som en bra klimatstrategi. Det vill säga att vedråvaran i första hand ska användas där den gör mest klimatnytta. Men *på grund av den stora användningen av fossila drivmedel är potentialen att motverka klimatförändringar avsevärt större inom drivmedelssektorn än inom materialsektorn.* Därtill så faller det ut stora volymer lågkvalitativ vedråvara från brukade skogar, även en viss volym stamved, som idag inte har något annat användningsområde än som råvara för energiproduktion. Detta är också ett av skälen till att en ökad efterfrågan på biobränslen inte direkt påverkar avverkningsnivåerna, vilket ofta antas. Ett annat är att *skogsbränslen är relativt lågt prissatta varför skogsägares beslut om avverkning i första hand styrs av mer värdefulla sortiment såsom sågtimmer*.

Livscykelanalys (LCA) är ett vanligt verktyg som används för att jämföra olika klimatstrategier i brukade landskap. Det finns fortfarande många osäkerheter och utmaningar att jobba med vid LCA analyser. För mycket fokus på kortsiktiga minskningar i utsläpp av växthusgaser kan verka kontraproduktivt på det långsiktiga målet att bidra till minskad påverkan på klimatet. Studier på beståndsnivån har ifrågasatts som potentiellt missvisande eftersom resultaten påverkas starkt av antaganden och hur beräkningarna görs. *Studier på landskapsnivån förordas istället, framförallt om ambitionen är att ge stöd för policyutveckling*.

Policyansvariga måste också ta hänsyn lokala förutsättningar för biomassaproduktion och förändringar i energisystemet. *En gemensam markanvändningspolicy över stora områden är troligen inte optimalt.* Dragna systemgränser och gjorda antaganden är ofta avgörande för resultaten i LCA-studier. Ett exempel är markanvändningsförändringar där antaganden om indirekta markanvändningsförändringar är mer osäkra än antaganden om direkta markanvändningsförändringar. *Många studier förutsätter att indirekta markanvändningsförändringar och andra indirekta effekter per definition resulterar i ökade emissioner av växthusgaser vilket inte stämmer*. Fokus i många studier har lagts på markanvändningsförändringar medan resonemanget också är relevant för förändringar på marknader för skogsråvaror och energi.

För att vara användbara i policysammanhang bör studier av växthusgasbalans för produktionssystem för biomassa inkludera alla förändringar i kolförråd, växthusgasemissioner längs produktionskedjan och substitutionseffekten av alla produkter producerade och inte bara från den del som används för bioenergi. *I idealfallet kombineras biofysiologiska modeller med klimat- och socio-ekonomiska modeller där även effekter på parallella industrier ingår (skogsprodukter, jordbruksprodukter och energi) för att fånga den totala klimateffekten av bioenergisystemet.* Samtidigt bör man vara medveten om att ju komplexare analyssystemet blir desto fler osäkerheter och antaganden introduceras i modellen.

Det är viktigt att först bilda sig en uppfattning om introducerade osäkerheter och antaganden gjorda innan resultaten från en studie används som policygrund. Man bör också vara medveten om att olika metodologiska angreppssätt och enheter beskriver koldynamiken på olika sätt. Det är därför viktigt att vald LCA-metodik är anpassad för den fråga som är ställd och den tänkta användningen av resultatet.

Valet av referensscenario utan en växande bioenergimarknad förklarar ofta varför studier kan få diamet motsatta resultat rörande klimateffekten av bioenergisystem. Inte minst studier där träden i referensscenariot får stå kvar och växa istället för att avverkas pekar mot negativa klimateffekter. *Ett referensscenario där skogen, i avsaknad av en bioenergimarknad, får stå kvar och växa är osannolikt i nordiskt skogsbruk där skogsbränslet är ett integrerat sortiment och där beslut om avverkning främst styrs av mer värdefulla sortiment såsom sågtimmer.* Brist på data rörande kolförrådets utveckling i bestånd som tillåts bli gamla bidrar också med osäkerhet i modelleringen.

Inom sina begränsningar visar LCA-studier inom *Bränsleprogrammet* att många träpelletsvärdekedjor på den svenska marknaden möter de krav rörande minskningar av växhusgasutsläpp som ställs i förnybartdirektivet (RED, 2009/28/EC). Studier visar också att oberoende av vilken typ av skogsbränsle som används (grot, klena stammar, stubbar) i kraft-värmeproduktion så motverkar dessa system framtida klimatförändringar. Valet av markkolsmodell och transportavstånd har begränsad påverkan på resultaten. LCA-studier pekar också på att svensk biodrivmedelsproduktion baserad på biomassa från jord- och skogsbruk reducerar växthusgasutsläppen avsevärt jämfört med fossila bränslen.

Rapporten har fokus på växthusgasernas effekt på klimatet. Klimatet påverkas också av andra faktorer såsom aerosoler, solinstrålning (molnbildning) och markens albedo, vilka alla kan påverkas av markanvändningsstrategier. Dessa faktorer borde också tas med i analyserna, vilket skulle göra dem än mer komplexa. *Det finns också andra miljöaspekter än klimatet att ta hänsyn till då storskaliga bioenergisystem växer fram*. Här kan nämnas långsiktig produktionsförmåga, biodiversitet, försurning, övergödning och mobilitet av toxiska föreningar såsom till exempel metylkvicksilver, vilka alla har studerats i andra projekt inom *Bränsleprogrammet*. Dessa och andra miljöaspekter måste också vägas in i planeringen och policyprocessen då storskaliga marknader för biomassa utvecklas och växer.

3 Introduction

One of the main reasons for paying attention to climate change is the impact that it might have on human society, e.g., economic damages, health impacts, reduced security of food and energy supply (IPCC, 2014). The different scenarios analysed in the AR5 report from the United Nation's Intergovernmental Panel on Climate Change (IPCC, 2014) suggest that bioenergy will play a significant role in mitigating climate change during the coming decades. The mitigation potential from using certain biomass for energy purposes is however questioned from both science and different interest groups. A case in point is biomass originating from northern forests that due to the harsh climate have a slow growth rate and therefore long rotation periods. With the simple assumption that carbon (C) emitted from forest biomass used for energy today has to be compensated for by carbon sequestered in regrowing trees at the same site before it can be considered fully carbon neutral, it will take time before neutrality and the full mitigation potential is reached for biomass from northern forests. This could be used to argue against such biomass – not the least within time frames common for greenhouse gas emission targets agreed on by policy makers.

Sweden is a country with a substantial share of its energy use based on biomass, with the major part originating from our forests (Figure 1). A vast proportion of that forest biomass consists of industrial residue streams from our large and for the country important forest industry. To satisfy the market demand, also primary residues from harvest operations like branches and tops (slash), small diameter trees, damaged trees, and stumps have been on the market as well as some round wood in competition with the pulp and paper industry. A minor contribution comes from dedicated energy crops (*Salix*).

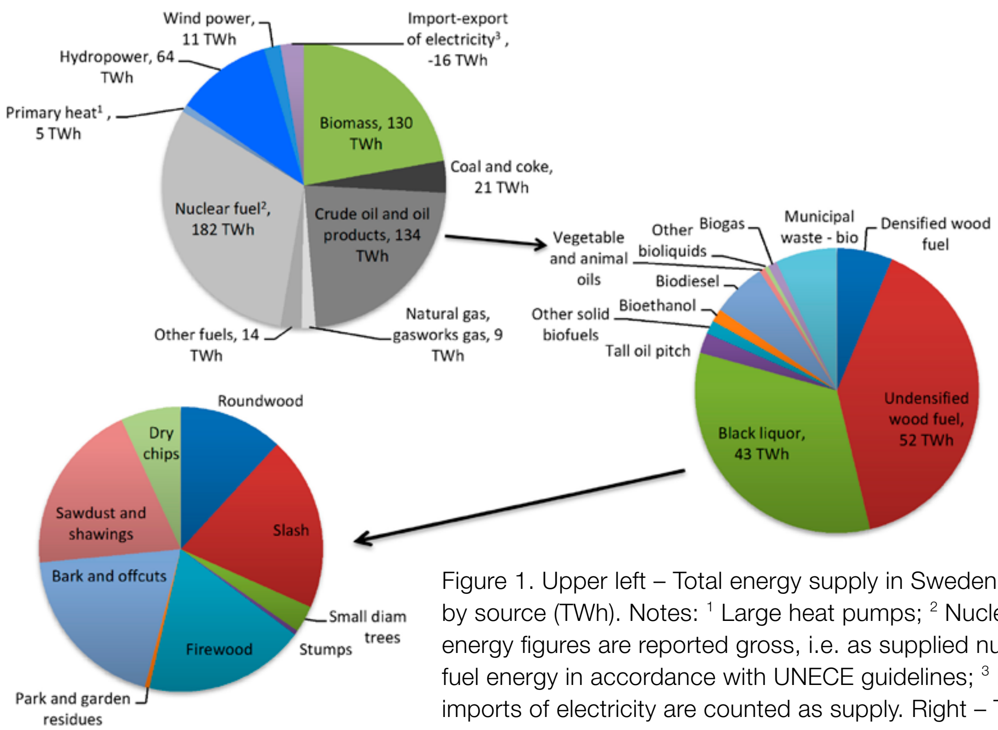

Figure 1. Upper left – Total energy supply in Sweden 2014 by source (TWh). Notes: [1] Large heat pumps; [2] Nuclear energy figures are reported gross, i.e. as supplied nuclear fuel energy in accordance with UNECE guidelines; [3] Net imports of electricity are counted as supply. Right – Total biomass supply 2014 (130 TWh) divided into different sources. Lower left – Supply of undensified wood fuels 2014 (52 TWh) divided into different sources. Source: The Swedish Energy Agency (2016).

The global bioenergy market is potentially large and it could easily deplete our forest resources (Egnell et al. 2011). This has resulted in studies that concludes that large-scale bioenergy from increased forest harvest levels is not sustainable and not greenhouse gas (GHG) neutral (e.g. Schulze et al. 2012). The forest resource in Sweden was to a large extent wiped out during the 18th and 19th centuries. The need for feedstock to the forest industries that started to emerge in the country from the mid-19th century resulted in forest policies that included two important components. First, annual harvest levels should never exceed annual growth and secondly, action to regenerate the forest after harvest was mandatory. On top of that afforestation was conducted on large areas of extensively used degenerated land. This has resulted in a continuous increase in growing stock and thereby annual growth at the same time as harvest levels have increased (Figure 2).

With appropriate forest policies in place it is possible to meet an increased demand for forest biomass without compromising future woody biomass supply and carbon sequestration in forests – rather the opposite.

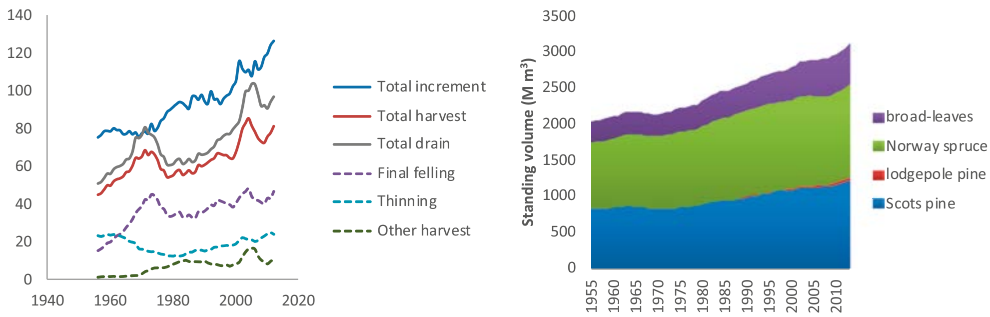

Figure 2. Left- Annual volume increment, harvest, and drain (harvest + mortality) on productive forestland (production ≥ 1 m^3 ha^{-1} and $year^{-1}$) in Sweden 1956–2014 (million m^3). Moving 5-year averages, i.e. 2012 show average value for 2010–2014. Right – Standing volume by species on productive forestland in Sweden 1955–2015 (million m^3). National parks, nature reserves and nature protection areas that are protected from forestry activities as of 2015 are excluded. Source: National Forest Inventory, Swedish University of Agricultural Sciences.

The two spikes in harvest and drain in the late 1960s and early 2000s in figure 2 are the result of two storms in 1967 and 1969 (10 million m^3 storm-felled in each of them) and a major storm in 2005 (75 million m^3 storm-felled) followed by bark beetle outbreaks that killed even more trees. This should be viewed in the light of a documented increasing trend in disturbances from wind, insects, and wild fires in European forests with a potential to offset management strategies to increase the forest carbon sink (Seidl et al. 2014).

4 Aim of the report

The aim of this report is to put findings from selected research projects in a research program, "*The Biomass Fuel Program*", 2011–2015, financed by the Swedish Energy Agency, into context and, if possible, to synthesize the results. Selected research projects focus on climate impacts of bioenergy use and energy systems from primary biomass production to the final end use. Although publications and findings based on research within *The Biomass Fuel Program* have been given special attention, findings from other relevant studies from primarily the peer-reviewed literature are also included. The ambition has not been to give a full overview of the published literature. Focus is on Swedish (Nordic) conditions with a large biomass resource in our forests that are managed over long rotation periods with an even-aged forestry system primarily to produce more valuable forest products than biomass for energy. Based on this future research challenges are also suggested.

5 Framing the issue

Reducing and eventually eliminating the accumulation of fossil carbon in the atmosphere is one of today's most demanding challenges. Simultaneously meeting this and other energy objectives requires a focus on the energy system as a whole to ensure correct conditions for a large-scale energy transition. Several projects within *The Biomass Fuel Program,* a Swedish research program (2011–2015) financed by the Swedish Energy Agency, have addressed climate impacts of biomass use for energy and for biomaterial production. The climate change mitigation potential of bioenergy depends on the biomass feedstock used and the alternative energy source/system that would otherwise have been used. Comparison with fossil fuels is always relevant since more than 80% of the global energy use currently is based on that source. But if, over time, the use of fossil fuels will be reduced the relevant comparison may be other renewable sources. Estimates of the mitigation potential also depends on the system boundaries within which it is analysed including spatial and temporal scales, market situation, direct and indirect effects etc. As the complexity of the system and thereby the analyses increase, empirical evidence together with different assumptions form the base for single or multiple model analysis approaches. For forest biomass originating from northern forests with long rotation periods, primarily managed for other forest products than wood fuels (saw timber, pulpwood), this has proven to be a difficult task, resulting in different results and conclusions from different studies. Here this is described with increasing complexity starting with a simple carbon pool change approach in soil and biomass.

6 Biomass primary production and greenhouse gases

6.1 GHG-emissions in primary production – The soil carbon pool

While soil fertility normally is not limited by the soil C content in forest soils that is often the case in agricultural soil. Thus, there is a large potential to increase the soil C pool in agricultural soils globally and at the same time increase crop production on those soils. Strategies to achieve that include afforestation, no-till farming, cover crops, fertilization, biochar, manuring and sludge application, agroforestry practices, and growing perennial energy crops on spare land (Lal 2004). This has also been demonstrated in a project within *The Biomass Fuel Program,* where Porsö and Hansson (2014) showed significant soil C pool increases when former fallow land was used for dedicated energy crop production (willow and poplar, c.f. figure 5).

> The potential to increase the soil C pool by cultivation of perennial crops is large in many agricultural soils, whereas the challenge in most forest soils is to maintain or slightly increase the C pool.

Forests are significant C pools with two third of the C stored in forest soils and one third in the growing biomass (Dixon et al. 1994). Therefore, the fate of soil C following forest management decisions has received a lot of attention. There are several ways in which the soil C pool may be affected by increased harvest intensity as a response to a bioenergy market, i.e. harvest also of logging residues including slash, stumps, small diameter trees or other non-merchantable trees. Suggested impacts of intensive biomass harvest on soil C dynamics include:

(i) Direct C-loss with the harvested biomass that otherwise would decompose slowly over time.

(ii) C-loss or gain as an effect of changed decomposition rates of soil organic material – an effect of increased soil disturbance often reported following increased harvest intensity, particularly following stump harvest.

(iii) C-loss or gain as an effect of changed vegetation and tree growth and thereby root exudate, litter, and wood C input to the soil.

(iv) C-loss or gain as an effect of changed transport of organic compounds with runoff water.

Among them it is only (i) that could be measured/estimated easily and the short-term direction as compared to stem-only harvest is clear with a soil C loss, whereas the long-term effect is less clear. The long-term direct effect of logging residue harvest on the soil C pool depends on the proportion of C in logging residues that, if they were left on site, will end up in a more stable C-pool due to recalcitrance or because it ends up in locations in the soil with poor conditions for decomposition.

A study on islands in boreal Sweden suggests that a large proportion of stored C in the soil is derived from roots and associated microorganisms rather than from aboveground biomass (Clemmensen et al. 2013). They particularly pointed out the importance of fungal residues in late successional forests. In a follow-up study, they particularly pointed out the importance of melanised hyphae of ericoid mycorrhiza associated with dwarf shrubs for the build-up of the forest soil C stock, while hyphae of ectomycorrhizal fungi associated with trees was linked to rapid turnover, efficient nitrogen mobilization and low C sequestration (Clemmensen et al. 2015).

Climate as the prime control of decomposition rates has been questioned (Bradford et al. 2016) and evidence for the importance of litter traits and decomposer biomass have been presented (Bradford et al. 2017). Recalcitrance of certain soil organic compounds has also been questioned and many issues on soil C dynamics is to a large extent still unknown (Schmidt, et al. 2011), making it difficult to model and predict the long-term fate of soil C (c.f. Dungait et al. 2012).

More process-based knowledge is needed to understand soil C dynamics and thus, to model the long-term development of the soil C pool.

For (ii) a common statement is that clear-cutting and soil disturbance, such as that following mechanical site preparation and stump-harvest, is likely to increase the decomposition rate of soil organic C, and thereby increase the carbon dioxide (CO_2) release to the atmosphere. However, studies do not give a general support for that statement (c.f. Yanai et al. 2003).

Results from *The Biomass Fuel Program* include a study where soil respiration following different mechanical site preparation methods and stump-harvested plots were compared during the first two years (Strömgren et al. 2012). The study showed that the effect of stump harvesting on the short-term soil CO_2 flux was negligible compared to mechanically site prepared plots. In another study litter decomposition in different soil layers and soil CO_2 flux following different soil disturbances and stump harvesting were compared with "undisturbed" clear-cut plots (Mjöfors et al. 2015). Despite the fact that buried litter decomposed faster than litter on the soil surface, soil CO_2 flux was highest on undisturbed plots.

Studies of soil CO_2 flux during 2 years in conjunction with the establishment of 14 stump-harvest experiments distributed over Sweden gave similar result, i.e. a lower or equal soil CO_2 flux on clearcuts following stump-harvest and different mechanical site preparation methods as compared to "undisturbed" clearcuts (Strömgren et al. 2017). Based on the fact that these 14 sites represent different climates and site properties they conclude that their results are reliable and generalizable for Nordic forest conditions. This adds to other studies suggesting that the default forest soil CO_2 flux response to mechanical soil disturbances not necessarily is an increased flux.

There is no support for the general assumption that mechanical soil disturbances increase net soil CO_2 emissions.

Two-year measurements of fluxes of other important GHGs (nitrous oxide (N_2O) and methane (CH_4)), on 3 mesic sites out of the 14 recently established stump-harvest experiments, showed small treatment effects of mechanical site preparation and stump-harvest, and the potential climate impact off these fluxes were small compared to size and changes in CO_2 fluxes on these clear-cuts (Strömgren et al. 2016). A shift from a net methane sink to a net source as a result of clearcutting and stump harvest has also been shown by Sundqvist et al. (2014). A raised water table was suggested as an explanation for this shift, thus, a transient effect should be expected as vegetation and trees re-establish.

Fluxes of other GHGs like N_2O and CH_4 are generally small from Nordic forest soils.

For (iii) there is strong evidence that increased growth results in increased soil C pools – at least for within species comparisons. Thus, silvicultural measures that increase forest growth, such as fertilization, will most likely increase the soil C stock (Jandl et al. 2007; Lal, 2005). Correspondingly, if additional nutrient export with harvested logging residues cause a reduction in forest growth a reduced soil C sequestration should be expected. A key question here is how forest owners respond to altering future market outlooks, i.e. with or without a market for biomass, in terms of willingness to invest in measures to increase forest production.

For (iv) there are no obvious reason to expect large effects of harvesting logging residues on C export with runoff water. However, an initially higher evaporation from slash on the ground will reduce the amount of water available for that transport and if harvest of logging residue results in increase soil damages this may change the runoff pattern on the site. A study in *The Biomass Fuel Program* measured water chemistry, including organic carbon, in catchments following mechanical site preparation and stump harvest. The study found no difference between the treatments, although the total organic C concentrations increased compared to unharvested catchments (Eklöf et al. 2012). The fate of C lost with runoff water adds to the CO_2 flux uncertainty, although dissolved organic carbon (DOC) reaching surface waters to a large extent releases its C to the atmosphere (Wallin et al. 2013).

These uncertainties and the complexity of soil C dynamics make it difficult to model soil C over time, and there is a need for empirical data from long-term field experiments to calibrate the models. A problem here is that soil sampling for accurate C-pool estimates and the interpretation of the results are difficult due to the large spatial and temporal variation of C-content in forest soils (Yanai et al. 2003). The spatial variation can be further amplified by harvest intensity (n.b. stump harvest) and mechanical site preparation.

One approach to find possible general patterns is to gather data from the published literature and use the full dataset in a meta-analysis. The meta-analysis on soil C pool changes following harvest by Johnson & Curtis (2001) is often referred to. Based on published results they compared the soil C pools after harvest with the pools in unharvested reference stands and found, over all studies included, that the soil C-pool increased following stemwood harvest (logging residues left on site), whereas it decreased following whole-tree harvest. Data included in the study were not necessary relevant for northern forests, with some originating from temperate and subtropical areas, and more important, treatments with forest floor removal were included in the whole-tree harvest category.

A more recent meta-analysis, based on 432 soil C response ratios from primarily temperate forests, showed an average soil C loss over all soil layers of 8%, with forest floors (30% loss on average) being more likely to lose C than mineral soils as compared to unharvested reference stand or pre-treatment values (Nave et al. 2010). In this study, the response was similar for both harvest intensities (stem only and whole-tree), suggesting that more intense harvest to supply a bioenergy market did not make a difference.

However, results from a new meta-analysis by Achat et al. (2015), suggested that soil C losses in the forest floor following stem only harvesting was fully compensated for by accumulation in deeper soil layers, whereas whole-tree harvest resulted in C losses in all soil layers. Based on their results they argued that the ability of forest soils to store C might be threatened by more intensive biomass harvesting. But again, also in this study treatments with forest floor removal were included in the whole-tree harvest category. The results are therefore not directly applicable to operational whole-tree harvest.

These meta-analyses have their advantages since they use large datasets that can be used to identify general treatment effects – but it is also important to look carefully into the data used before any final conclusions are drawn. This includes the experimental design that many times differs from practical forestry – not the least by being more intense. There are also important issues related to temporal and spatial scales.

The increased C pool in deeper soil layers following stem-only harvest reported by Achat et al. (2015) could be the result of mechanized harvest operations and site preparation that buries logging residues and litter into the mineral soil as suggested by Yanai et al. (2000). With logging residues harvested for energy purposes or with a full-tree harvesting system, where whole trees are transported to the landing where they are delimbed, this accumulation potential is reduced. A key question is then the fate of logging-residue C since logging residues, as a labile C source, will decompose and release its C to the atmosphere over time. Or in the case of a full-tree harvesting system, where the slash piles often are burnt on site, immediately emit its C (and other climate forcing agents) to the atmosphere (Aurell et al. 2017).

The supplementary material supplied by Achat et al. (2015) shows that a majority of the data was sampled just a few years up to 10 years after harvest, thus, left slash on stem-only harvested plots is still decomposing. This temporal aspect also has to be seen in the light of the spatial scale of a forest landscape managed for sustainable yield over time, where just a small area of the landscape is harvested each year, while other areas sequester carbon with a large proportion distributed to the soil through root exudates and litter. This is illustrated for spodosols (a soil type common in Sweden) in Nave et al. (2010), where forest floor C following harvest recovers slowly over a period of 60–80 years.

To fully capture the temporal or the spatial dimensions of soil C pool changes in forest soils and potential CO_2 emissions to the atmosphere, relevant temporal or spatial scales, i.e. a full rotation period and the whole production landscape, have to be considered.

Furthermore, in both the study by Johnson & Curtis (2001) and the study by Achat et al. (2015) the whole-tree harvest treatment also included studies with forest floor removal. This reduces the relevance of these studies for practical harvest operations where instead the forest floor and a substantial part of the logging residues is left on site (Thiffault et al. 2015).

Results from *The Biomass Fuel Program* include one study where the soil C pools after slash and/or stump harvest were compared with stem only reference plots 32–39 years after harvest on eight sites located in northern, central, and southern Sweden. Although there was a negative trend, none of the increased harvest intensities resulted in significant reductions of the soil C pool (Jurevics et al. 2016). Non-significant negative trends on the soil C pool were also reported by Hyvönen et al. (2016), from a survey-study comparing soil C pools following slash and stump-harvested sites with slash-harvested sites in Finland 8–13 years after clear-cutting. In another study, the soil C pool following stump harvest and intense deep soil cultivation (100% soil disturbance down to 60 cm, i.e. far beyond soil disturbance expected in practical forestry) was compared with manually patch scarified stem-only harvested reference plots more than 20 years after harvest. This intense treatment resulted in a significant decrease in total soil C (Egnell et al. 2015).

Long-term studies on mechanical site preparation effects can be used to evaluate whether soil disturbance may have an impact on the soil C-pool by stimulating mineralisation. Örlander et al. (1996) reported soil C-pool reductions following mechanical site preparation as compared to untreated control plots in five old experiments on poor Scots pine sites in Sweden. Soil C reductions, although not significant, after 25 years were also reported for the most intensive mechanical site preparation method (ploughing) as compared to control plots without any site preparation (Mjöfors et al. 2017). A conclusion based on these soil-disturbance-induced C losses reported could be that they will have an impact on the climate as more C has been emitted to the atmosphere.

An alternative interpretation is that the decay of soil organic matter shows that there is biological activity in the soil resulting in released nutrients that could benefit regrowth of plants and trees. This “soil carbon dilemma” is discussed in a paper by Janzen (2006), with the open question concerning soil C: *“Shall we hoard it or use it?”* The answer probably lies somewhere in between, where soil C losses due to decomposition is necessary to drive important biological processes in the soil and not the least to support plant growth and thereby C sequestration through mineralisation of plant nutrients otherwise locked up in organic matter.

A significant short- to medium-term change in soil C as a result of a silvicultural measure cannot be used as a basis for final conclusion on the long-term climate impact.

6.2 GHG-emissions in primary production – The above-ground carbon pool

With soil C changes following silvicultural measures being insufficient to support final conclusion on climate impacts, additional information on C sequestered in plant biomass provides a better basis. In two of the three studies above, showing reductions in the soil C pool following stump harvest and/or intense mechanical site preparation, the soil C loss was fully compensated for by increased tree growth (Jurevics et al. 2016), or even overcompensated, with a significantly larger total C pool (soil + trees) following the most intense site preparation methods (Mjöfors et al. 2017). Örlander et al. (1996) reported a higher top height following mechanical site preparation suggesting that the production potential was increased.

If climate impact data is limited to carbon pools in the biomass primary production system, a minimum requirement is that both soil and crop C-pools are included in the analysis.

These examples are all site-level studies, where the relevant time-period for final conclusions on C-stock changes preferably should be based on data covering one rotation period or more, and even better covering the whole forest production landscape. Unfortunately such data is lacking.

In the lack of empirical data, modelling approaches have been used. Within *The Biomass Fuel Program,* Cintas et al. (2016) modelled forest C stocks dynamics at the landscape level over 100 years with forest fuels integrated as an equally important assortment as pulpwood and saw timber. The C stocks for the three different real landscapes analysed showed different responses to the new assortment with increased C stocks in two of the landscapes and a more expected C stock decrease in the third as compared to a reference scenario where pulpwood and saw timber plus a limited amount of slash in final felling were harvested (Figure 11). This was explained by differences in initial forest conditions between the three forest landscapes together with management responses related to the "new" forest fuel assortment including extended rotation periods and more frequent thinning regimes. This also holds true for any forest landscape managed for feedstock. Managed forest landscapes with a lot of mature forests are likely to experience an initial C loss, whereas landscapes with a lot of plantations and young forests are likely to experience an initial C gain when modelled over time.

Depending on the initial forest conditions a managed forest landscape may experience initial C loss (a lot of mature forests) or C gain (a lot of recent plantations and young forests) as a result of forest management.

Many studies have shown that more intense harvest, including logging residues like slash and stumps, result in a moderate increase in biomass removal, but a significant increase in nutrient removal from the site. This is particularly the case for slash (e.g. Kimmins 1977; Freedman et al. 1981) and less so in stumps (e.g. Uri et al. 2015). Furthermore, due to the different morphology in Scots pine and Norway spruce, the dominating tree species in the managed forests in Sweden, spruce holds more biomass and nutrients in slash than pine (Palviainen and Finér, 2012). One concern when nutrient-rich biomass is harvested is that it will have a negative impact on future site and stand productivity – an effect that would decrease the potential C sequestration.

A fair amount of studies on forest production following slash harvest, often referred to as whole-tree harvest, in clear cutting have been published. Although statistically significant negative effects on stand production have been revealed (e.g. Egnell and Leijon 1999), non-significant effects on stand productivity is a common result (e.g. Kaarakka et al. 2014). Suggested reasons for this are that stand productivity effects following slash harvest go beyond effects of the nutrient loss and also includes effects of microclimate and competing vegetation on the subsequent tree crop (Thiffault et al. 2011). In the case of slash harvest in thinnings there is a residual stand ready to respond on nutrients released following thinning, thus, nutrients harvested with logging residues is more likely to have a direct impact on growth. To some extent this has been proven correct, with negative growth effects reported following slash harvest in thinnings (Helmisaari et al. 2011; Egnell and Ulvcrona, 2015; Tveite and Hanssen, 2013), but no growth effect following slash harvest in thinnings have also been reported (Egnell and Leijon, 1997). Although growth losses have been reported in both spruce and pine, the tendency is that growth losses are more likely in spruce (Egnell 2017). The fact that compensatory fertilization with primarily nitrogen compensates for production losses following slash harvest (Helmisaari et al. 2011), suggests that it is the effect on nutrient availability that explains the growth loss. Furthermore, evidence suggests that growth effects following slash harvest are temporal rather than permanent (Egnell 2011; Egnell and Ulvcrona, 2015).

Compensation measures directly linked to slash harvest include shedding of needles before the slash is removed (Egnell and Leijon, 1999; Stupak et al. 2008) or fertilization (Helmisaari et al. 2011). Wood ash recycling has also been suggested – but it has a minor effect on forest growth on upland mineral soils unless also nitrogen is added (Jacobson et al. 2014). However, wood ash addition on peat soils has an immediate impact on forest growth (Rutting et al. 2014) and is highly recommended following slash harvest on peat soils. Since logging residues, slash in particular, constitute a physical impediment for site preparation and planting it is also reasonable to assume that harvest of logging residues will facilitate and improve site preparation and planting quality and thereby regeneration success (Saarinen 2006).

The limited amount of publications reporting on forest production following stump harvest suggest that stump harvest does not reduce growth of the subsequent stand in spruce and possibly even increase growth in pine (Egnell 2016, 2017; Hope 2007; Karlsson and Tamminen 2013). One particular case is when stump harvest is used as a method to reduce root rot infections in the subsequent stand. This has proved to be efficient with increased growth as a result (Cleary et al. 2013).

All together these results suggest that in order to reduce the impact on forest growth, pine sites should be targeted before spruce sites and stumps should be targeted before slash. In practice, spruce sites are targeted before pine sites since there is more slash biomass in spruce than in pine. Furthermore, spruce stumps often have a more superficial rot systems making them easier to harvest with current harvest technologies. Slash is also favoured before stumps since it is cheaper than stumps to procure (Lundmark et al. 2015).

Over a forest landscape, moderate negative effects of slash harvest on forest production should be expected unless the production loss is compensated for by more ambitious silviculture promoting forest growth, whereas stump-harvest appears to have no effect.

Expectations on future forest commodity markets are drivers for forest owner's silvicultural choices and their willingness to invest in measures to increase forest production. This includes fertilization, well-performed site preparation, and use of genetically improved seedling stock or fast-growing tree species (c.f. Nilsson et al. 2011). Although forest biomass for energy currently is a low-priced commodity, the potentially huge market may have such an impact. In more productive areas with alternative land use, as in south eastern US, market responses may also include expansion or reduction of forest land (Abt et al. 2012).

Forest owner response to a new market potential has to be addressed when analysing effects of using forest biomass for energy on the total forest carbon pool.

Biomass production for energy in long-rotation forestry is produced together with two other main assortments, namely pulpwood and saw timber. Part of the C that is sequestered into growing forests end up in forest products. Especially, C stored in long-lived products like construction wood in buildings needs to be considered in order to give a true and fair view of the C sequestration potential for managed forests. Within *The Biomass Fuel Program,* this has been demonstrated in a project where C stored in soil, trees, and products in Sweden have been modelled over time for a number of different management strategies.

With the Swedish forests as the modelling case, Gustavsson et al. (2017) found that compared to a "business as usual" scenario (BAU), where annual harvest levels equal annual growth, a "production" scenario, including a number of measures to increase forest growth, will increase the C-stock in the soil, in tree biomass and in forest products over 100 years (Figure 3). Although the C-stock decreased in forest products as compared to BAU, a positive effect on the total C-stock was achieved also with a "set-aside" scenario where the area of set-aside productive forest land in Sweden was doubled. With an evaluation based on the C-stock only the set-aside scenario appears to be the best option to mitigate climate change, followed by the production scenario. It should be noted that the BAU-scenario in this study is more intense than current practice in Sweden with harvest levels below annual growth (c.f. Figure 2).

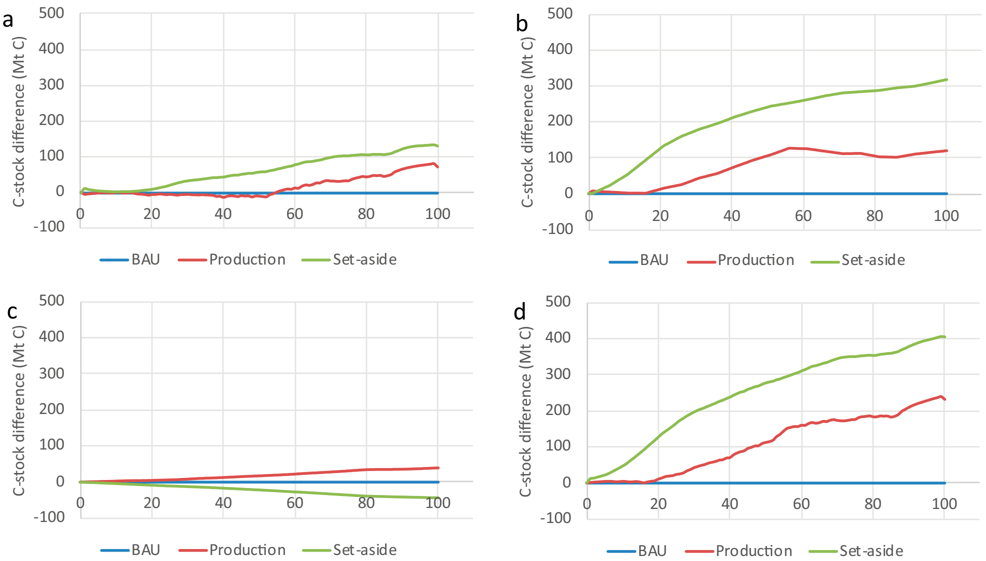

Figure 3. Difference in C-stocks (million t C) in a) forest soils, b) tree biomass, c) hard wood products, and d) total (a+b+c) over 100 years of forestry in Sweden. A business as usual scenario (BAU), here set to 0, is compared with a production scenario where a number of silvicultural measures to increase forest growth is introduced, and a set-aside scenario where the area of set-aside forest land is doubled as compared to BAU. In all three scenarios, annual harvest equals annual growth and slash corresponding to 8 TWh/y is harvested for generation of heat and power each year. Source: Based on data from Gustavsson et al. (2017).

The C-stock approach behind the results presented in Figure 3 does not include treatment/transport generated GHG-emissions and the substitution effect of using the harvested biomass and will therefore not capture the full climate impact of the alternate scenarios. See section 7.5 for further development of the results in the study.

Analyses including C stock changes in soil, trees and forest products do not capture the full C balance of managed forests – this also includes treatment/transport generated emissions and the substitution effect of using the harvested biomass.

7 Climate impact of biomass use for energy and biomaterials

7.1 Climate impact assessment methods and metrics

With forest biomass for energy integrated in the production of other forest products with their climate impacts, and an energy system under transition towards more renewables, complicates the picture further. Addressing all these issues is a challenging task. Figure 4 shows aspects that need to be considered in order to answer different questions. Already the most basic of the three questions in Figure 4 requires that one consider a cause-effect chain associated with GHG emissions, illustrating how effects along the chain implies increasing uncertainty as successively more complex aspects need to be considered.

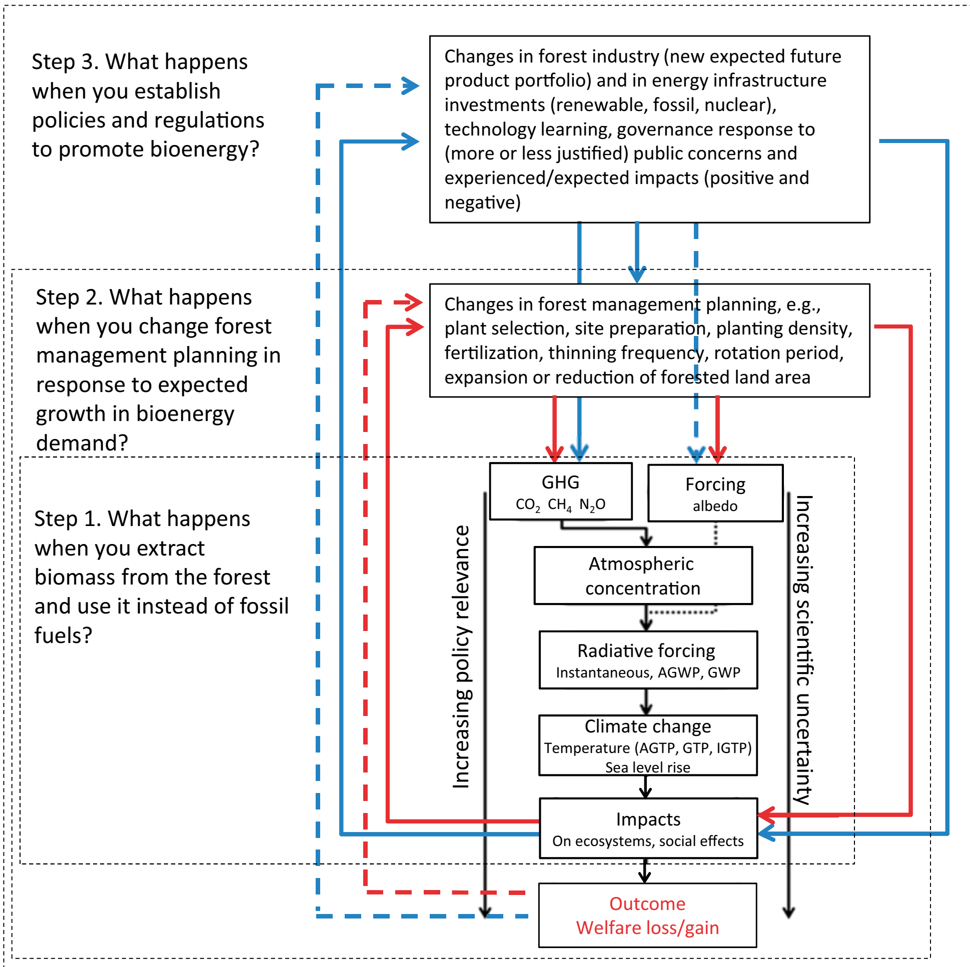

Figure 4. Cause-effect chain of greenhouse gas emissions. Based on Ericsson (2015) adopted from Fuglestvedt et al. (2003).

Each step down the chain is a consequence of the previous step, and a direct cause of the following step. In every step, multiple responses can appear. For example, a change in radiative forcing, which describes the energy balance of the earth, causes a climate change. This can in turn give responses such as temperature change, sea level change and changed weather patterns, etc. These will eventually propagate down the cause-effect chain and impact on human society. An important aspect of the cause-effect chain is that for each step down the chain additional assumptions and uncertainties have to be introduced in the modelling. However, at the same time the relevance to human society increases, as the impacts are evaluated closer to actual damages to human society.

As the complexity of the GHG cause and effects of emerging bioenergy markets increases additional assumptions and uncertainties have to be introduced in the modelling, but at the same time the relevance of the results increases due to increased completeness.

Bioenergy systems are rarely fully C neutral and the size and timing of C flows and C stock changes have been studied from various perspectives for decades. However, during recent years the C balance associated with bioenergy systems has become a topic for debate also outside the scientific world, especially in the context where governments and authorities want to ensure that bioenergy initiatives contribute to reducing net GHG emissions in the short term. The increased attention has resulted in a large number of studies, which have presented diverging conclusions. One reason to this disagreement is that C emissions and sequestration associated with bioenergy systems are not necessarily in temporal balance with each other and different methodological approaches capture the C dynamics in contrasting ways.

Especially, the dynamics of C uptake and release in long-rotation forestry vary greatly and can differ substantially for the bioenergy use on the one hand and decomposition/regrowth processes in the forest ecosystem on the other hand. This leads to GHG mitigation trade-offs between biomass extraction for bioenergy and the alternative to leave the biomass in the forest to decay. This C dynamic also causes challenges for the quantification of climate effects in relation to how different metrics capture the dynamics.

Various metrics have been proposed for quantifying climate change effects. Depending on the purpose of the assessment, different metrics may be preferred. Global warming potential (GWP) is commonly used as a climate impact indicator in life cycle assessments (LCA). GWP is the integrated radiative forcing over a chosen time horizon, expressing all GHGs in carbon dioxide equivalents (CO_2-eq), and would in Figure 4 be placed in step 3. The GWP metric has been accepted by the IPCC and is implemented in the Kyoto Protocol with the purpose to compare the potential climate impacts of different gases (Shine et al. 2005). However, attention has been drawn to several disadvantages of using this metric in LCA, e.g. the use of fixed and often arbitrary chosen time horizons and neglect of the temporal variations of emissions over time (Fuglestvedt et al. 2003).

One commonly used alternative metric, global temperature potential (GTP) expressed as the global mean temperature for a selected year in the future (Shine et al. 2005), reflects effects at the last step in the cause-effect chain (Figure 4). This increases the uncertainty in the results, but also approaches the actual damage that emissions of GHGs may cause. Another important difference between GWP and GTP is that the latter only evaluates the climate impact at a specific point in time after an emission has taken place. The GWP, on the other hand, integrates the impacts between the time of emission and the chosen time horizon. This means that the GTP says nothing about impacts between the time of emission and the chosen point in time, while the GWP "remembers" all impacts from the time of emission up to the end of the chosen point in time. Metrics formulated like the GTP, in this sense, are commonly referred to as instantaneous, while metrics formulated like the GWP are commonly referred to as cumulative (Ericsson, 2015).

Different methodological approaches capture the C dynamics in contrasting ways.

Other time dynamic climate metrics include dynamic characterisation factors (Levasseur et al. 2010), time-adjusted warming potentials (Kendall 2012), the Lashof-method (Fearnside et al. 2000), the fuel warming potential (O'Hare et al. 2009) and the GWPbio characterisation factors (Cherubini et al. 2011). An alternative way to approach the issue of timing is to not calculate the climate impact as a single score, but instead graphically present the change in radiative forcing or climate impact over time.

As another alternative, Berndes (2012) proposed to use the concept of 'greenhouse gas emissions space', which focuses on accumulated emissions up to a given year. This concept is relevant in relation to temperature targets when CO_2 is the dominating GHG, since the peak warming appears to be insensitive to the CO_2 emissions pathway, i.e., timing of emissions or peak emission rate. Depending on the atmospheric lifetime of specific GHGs the trade-off between emitting more now and less in the future is not one-to-one in general. But the relationship for CO_2 is practically one-to-one, so that one additional ton CO_2 emitted today requires that future CO_2 emissions be reduced by one ton. Likewise, one fewer ton CO_2 emitted within the emission space today allows for a future increase in CO_2 emissions by one ton. The reason for this is the close to irreversible climate effect of CO_2 emissions.

Application of more than one metric is informative for policy development since they complement each other.

7.2 Case studies of bioenergy supply chains

In this section, we summarize some of the recent research on climate effects of Swedish bioenergy, looking beyond the simple C-pool thinking and into assortments such as forest residues for heat and power, pellets and biofuels for transportation. We present results and conclusions from studies that have a relatively narrow scope as well as studies that involve more comprehensive approaches. Presented studies should be considered complementary in that they together provide insights beyond what can be gained from one single study. It should be noted that the climate is influenced not only by GHGs and all climate forcers should ideally be included, however, the focus below is on GHGs.

7.2.1 Pellets for heat and electricity

In a literature review by Höglund (2011), Swedish pellets were found to have low GHG emissions compared to other producing countries, mainly due to use of renewable fuel for drying. However, transportation of pellets to the Swedish market can have substantial influence on the GHG performance, especially transportation to small-scale end users.

Similar results were found in a study within *The Biomass Fuel Program* by Hansson et al. (2015), were the GHG emissions from production of wood pellets in different countries, integrated with use of these pellets in the energy sector in Sweden in different scales of heat and/or electricity production facilities, were studied. The calculations followed the methodology described in the Renewable Energy Directive (RED, 2009/28/EC), i.e., biospheric C stock dynamics (C in soils and above-ground biomass) were not considered. The total factory to gate GHG emissions were found to range between 2 and 25 g CO_2-eq/MJ pellets with Swedish pellets at the lower end, and Russian pellets using natural gas for drying the raw material at the higher end. The potential GHG reduction as compared to the RED fossil fuel default energy comparator is 64–98% for the electricity produced in the pellet value chains studied (Figure 5) and 77–99% for the heat produced. Thus, the study concludes that many wood pellet value chains on the Swedish market will be able to meet relatively high requirements on net GHG savings, if quantified as currently described in the RED.

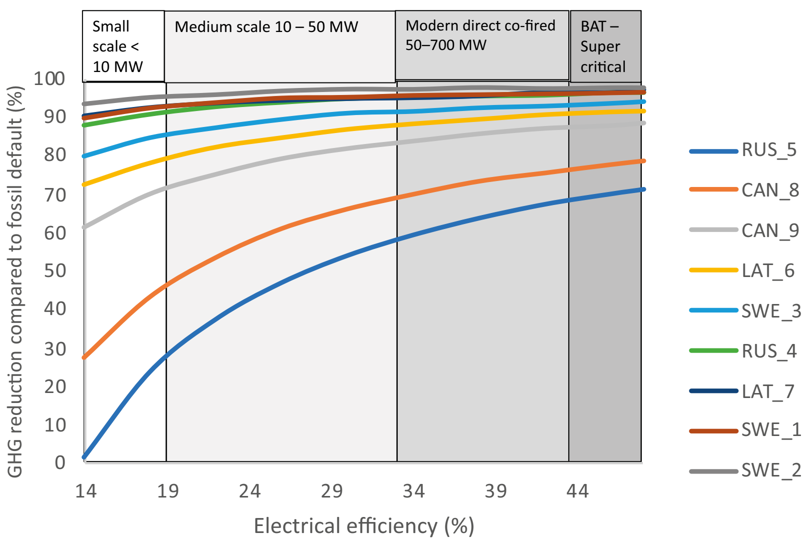

Figure 5. GHG emission reductions for different wood pellet supply chains for electricity production in Sweden as a function of different conversion efficiency and size of the dedicated power plant up to the best available technology, supercritical power plants (BAT). The wood pellet supply chains studied were located in four different countries, Sweden (SWE), Lativia (LAT), Canada (CAN) and Russia (RUS). The denotations 1–7 are for different production chains, with different raw material (saw dust, shavings, wood residues etc), different fuels for drying, and different modes of transportation. Based on a figure in Hansson et al. (2015).

Many wood pellet value chains on the Swedish market will be able to meet the requirements on net GHG savings, if quantified as currently described in the RED (2009/28/EC, will be replaced by RED II for the period 2021-2030), excluding changes in soil and crop C-pools.

Complementing Hansson et al. (2015), the GHG balance in willow and poplar-based pellets production chains were considered in another study within *The Biomass Fuel Program* by Porsö and Hansson (2014). They found that soil C changes were very influential on the GHG balance in this specific case with willow and poplar grown on previous fallow land (Figure 6).

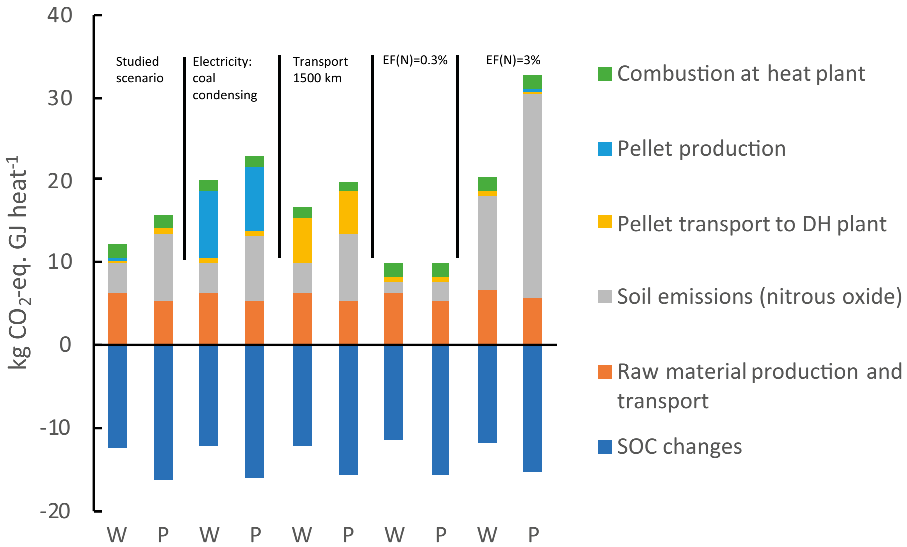

Figure 6. GHG emissions for pellets based on willow (W) or poplar (P) for the "Studied scenario", assuming Swedish electricity mix (which mainly consists of hydro and nuclear power with relatively low GHG emissions), a pellet transport distance to the district heating plant of 150 km, and a N_2O direct soil emission factor of 1%, compared with scenarios with changed electricity production (coal condensing), changed transport distance to the district heating plant or changed N_2O direct soil emission factors. Based on a figure in Porsö and Hansson (2014).

Porsö and Hansson (2014) also calculated the temperature change over time for willow and poplar pellets (Figure 7), in which the fluxes of CO_2 to and from biomass, soil and combustion are taken into account. The sequestration of C in the live biomass and the increased soil C pool resulted in an overall cooling effect on the temperature during the study period for both willow and poplar. Striving towards a new steady state, the soil C will contribute to a continuously increasing cooling effect. This highlights the importance of also including C pool changes in biomass and soil for accurate estimates of the GHG balance.

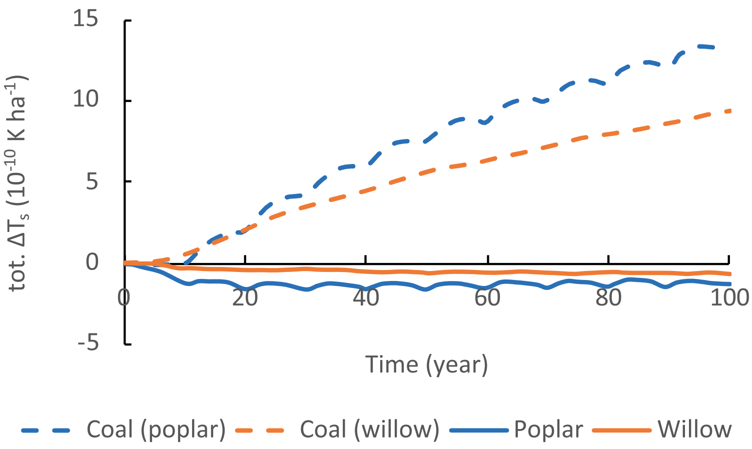

Figure 7. Changes in the global mean surface temperature, for one ha of willow or poplar used for district heating compared with the temperature change when coal is used to produce an equivalent amount of heat. Note: The willow and poplar produce different amount of energy per unit area. Based on a figure in Porsö and Hansson (2014).

Studies highlights the importance of also including C pool changes in biomass and soil for accurate estimates of the GHG balance when shifting crops on farmland.

The notion that establishment of biomass plantations can cause C sequestration in soils and above-ground biomass, enhancing the climate benefit, is well established since long (e.g., Schlamadinger and Marland 1996). It was proposed in association with COP6 in 2000 that C sinks could enter the UNFCCC agreement via bioenergy projects that utilize an accumulating sink. In other words, that bioenergy plantation projects would be allowed to expand the system boundary to include additional C accumulation on the land from which the biomass was drawn (Schlamadinger et al. 2001). Crediting such C sinks could significantly increase the incentives for dedicated bioenergy crop plantations (e.g., Berndes and Börjesson 2002).

Crediting C sinks in soils and crops could significantly increase the incentives for new dedicated biomass plantations with perennial crops.

But policymakers contemplating such measures need to be aware of challenges associated with permanence and leakage: the sequestered C would be emitted back to the atmosphere if the plantations at some stage are removed and previous land uses become re-established. GHG emissions may also arise if the plantation establishment induce land conversion elsewhere to make place for the displaced land use activities (iLUC).

Policy challenges for C-sink credits include permanence and leakage of C sequestered in soils and crops together with indirect land-use change (iLUC).

Further, in regions with seasonal snow cover or a seasonal dry period (e.g. savannahs), reduction in albedo due to the introduction of perennial green vegetative cover can counteract the climate change mitigation benefit of bioenergy plantations (Betts 2000). Conversely, albedo increases associated with clearcutting or the conversion of forests to energy crops (e.g. annual crops and grasses) may counter the global warming effect of CO_2 emissions from the clearcutting or deforestation (Sjølie et al. 2013).

7.2.2 Forest biomass for heat and electricity production

Within *The Biomass Fuel Program* the global temperature change associated with a slash harvest from a single stand and used in a district heating system, was studied by Hammar et al. (2015) for three different Swedish climate zones. The reference situation was that the residues were left in the forest and fossil fuels were used in the district heating system. No additional effects on the forest or energy sectors were considered. Replacing coal with logging residues gave a direct climate benefit, while the climate benefit was delayed some 8–12 years if instead natural gas was replaced (Figure 8). Although the difference between the three climate zones were relatively small, harvesting logging residues for bioenergy in the south of Sweden yielded the highest temperature change mitigation potential per unit energy over the study period since residues left in the forest decompose faster in in southern Sweden than in northern Sweden. An important note is that decomposition in northern Sweden eventually will catch up.

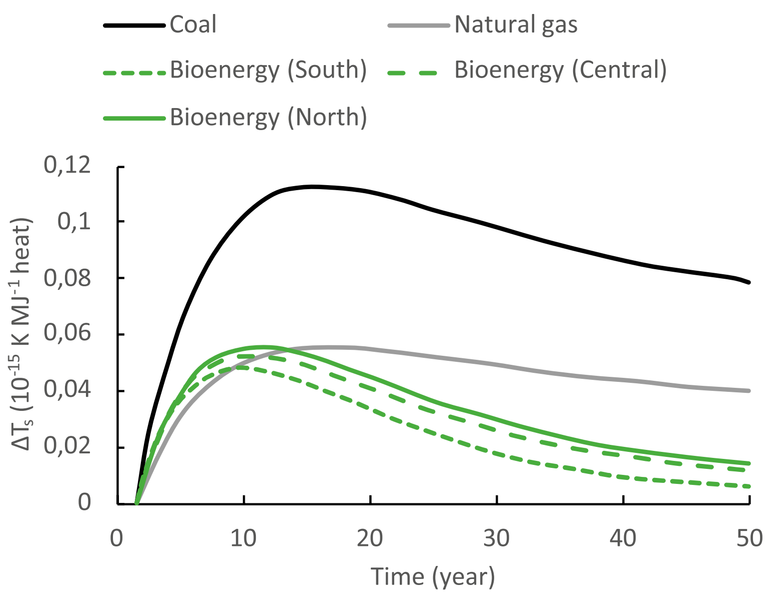

Figure 8. Net temperature effect (ΔT_s) during the first 50 years following combustion of a single harvest of slash after final felling of a single stand in southern, central and northern parts of Sweden compared with two fossil alternatives. The bioenergy curves include biogenic CO_2 emissions from biomass combustion, changes in soil organic carbon and decomposition of forest litter, as well as non-biogenic greenhouse gas emissions from the supply chain. Based on a figure in Hammar et al. (2015).

Gustavsson et al. (2015) studied the use of forest residues such as slash, stumps and small diameter trees from thinnings for heat and electricity production over 300 years and reached similar conclusions. Three different decay models and different transport distances were applied in their modelling and their conclusion was that the results were insensitive to the choice of soil carbon model and transport distances. The difference between slash, stumps and small diameter trees were relatively small and they conclude that all bioenergy systems, independent of feedstock, contribute positively to climate change mitigation in the long-term perspective.

Another Swedish study on forest residue use for energy by Zetterberg and Chen (2015) reached the same conclusions and noted that the use of different metrics resulted in some variation concerning the time required before the use of logging residues instead of natural gas yielded a positive effect.

LCA-studies suggest that the choice of soil C model and feedstock transport distance have limited impact on the climate change mitigation potential of forest biomass substituting fossil fuels in combined heat and power production.

Stumps were given special attention in *The Biomass Fuel Program* in the sub-program *Tree-stump harvesting and its environmental consequences*, managed by the Swedish University of Agricultural Sciences. A study derived from that sub-program used time-dependent LCA methodology and ecosystem forest carbon modelling to evaluate climate effects of using stumps as a fuel in district heating compared with coal and natural gas (Ortiz et al. 2016). From their results, presented with two different metrics, for a single harvest and for continuous supply in a landscape, and for three different climate zones, they concluded that stumps in district heating in Sweden gives a climate benefit even in a rather short time perspective (2 decades) as compared with the alternative of using coal or natural gas and leave the stumps to decompose in the forest. Climate benefits were immediate when replacing coal, whereas the parity time was 12–16 years for a single harvest and 22–28 years for a continuous supply of stumps in a landscape when replacing natural gas. Climate benefits were only slightly delayed (4 years) in northern Sweden due to a slower decomposition of left stumps in the reference case.

Independent of type of forest feedstock (slash, small-diameter trees, stumps) combined heat and electricity production contributes positively to climate change mitigation in the long-term perspective.

7.2.3 Forest and agricultural biomass for transport biofuel production

Over the years, the number of LCA studies on biofuels for transportation has rapidly increased. Several review studies have also appeared (Cherubini & Strømman, 2011; van der Voet et al. 2010; Von Blottnitz & Curran, 2007), as well as a review of Swedish biofuel LCA studies (Martin et al. 2015). Most of the studies have a strong focus on climate impacts.

Naturally, results from LCA-studies vary due to different assumptions and choices of methodologies and data. In general, it is common to separate between two types of LCAs; attributional LCA (ALCA) - and consequential LCA (CLCA). While ALCA typically accounts for the environmental impact of an existing product, CLCA calculates the environmental impact of a change. Although the border between the two types of LCA may be difficult to draw, they are often used to answer different types of questions. A typical issue for ALCA could be: "Where are the environmental hotspots in production of product A?". A typical research question for CLCA might be: "What is the impact of increased demand for product A?". This implies that CLCA to a higher degree must take surrounding systems into account, and the impact on them.

In the EU renewable energy directive, an ALCA approach has been chosen for calculation of GHG emissions from biofuels. Since the framework is developed for regulatory purposes with the aim of an easy and fair comparison of different type of biofuels, this can be justified. However, since it is very simplified it does not show the full picture of the biofuel impacts.

It is important that the choice of LCA-methodology and choice of data are in line with the research question and the intended use of the results.

For Swedish biofuel production, a study within *The Biomass Fuel Program* compared LCA-studies that use the EU RED (2009/28/EC) calculation framework (will be replaced by RED II for the period 2021–2030) and calculations done with an expanded system boundary according to the international ISO-standard on LCA (ISO 14040-series) and in line with CLCA-thinking (Börjesson et al. 2016).

Several conclusions can be drawn from the results (Figure 9). First, independent of LCA approach, all biofuels have lower emissions than the fossil references. Further, biogas from manure and waste offers the greatest reduction of greenhouse gases when using system expansion calculations. Emissions are here often negative, thanks to avoided methane emissions from storage of undigested manure. Biofuels based on gasification, HVO from tall oil, biogas from forage crops and mixed ethanol and biogas from lignocellulose also provides large reductions in greenhouse gas emissions (80–95%).

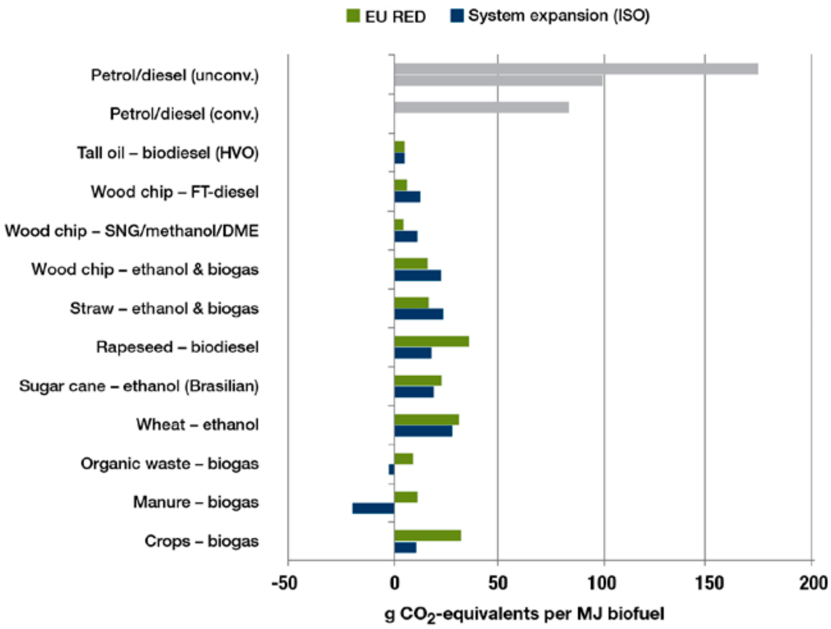

Figure 9. Greenhouse gas emissions from biofuel production systems, calculated according to the methodology of the EU's Renewable Energy Directive (RED) and the ISO standard (ISO 14040-series) for life cycle assessment (system expansion). For comparison, the emissions from conventional fossil fuel according to EU RED's default value and high and low values for unconventional fossil fuels (JRC, 2014) are shown Source: Börjesson et al. 2016.

LCA-studies suggest that Swedish biofuel production from agricultural and forest raw material have a large GHG reduction compared to fossil fuels

Even though the numbers of LCAs have been increasing, several challenges remain, which have been identified and categorized in various ways (e.g. Ahlgren et al. 2015; Cherubini & Strømman, 2011; McManus et al. 2015; McKone et al. 2011). In this literature, several of the identified challenges are related to the modelling and methodological framework, e.g. development of relevant scenarios, uncertainties in predicting future development, applying relevant system boundaries and accounting for time in impact assessments. Soil carbon modelling is also mentioned as a major challenge. Several challenges also relate to variability in data; for agricultural feedstock especially emissions of nitrous oxides are pointed out.

Some of these challenges have been addressed in Swedish studies. For example, most biofuel LCA-studies nowadays include soil carbon changes directly related to the cultivation/extraction of raw material. There are several soil carbon models developed which can be integrated in the LCA-framework. There are also a few studies on Swedish biofuel production that take into account the timing of greenhouse gas emissions (e.g. Gustavsson et al. 2015).

To conclude this section: Studies show that Swedish biofuel production from agricultural and forest raw material have a large GHG reduction compared to fossil fuels. However, it is important that the choice of LCA-methodology and choice of data is in line with the research question and the intended use of the results. In some cases (such as regulation) ALCA is sufficient; in other cases CLCA is needed to provide the full picture. Several other challenges exist for future development of biofuel LCA-studies, e.g. inclusion of timing of GHG emissions in the impact assessment, how to deal with variability, a better understanding of nitrous oxide emissions from soils. What is also clear is that most CLCA-studies of Swedish biofuel production are limited in their scope and do not include all types of market mediated effects including indirect land use changes (see further discussion in section 7.4).

> There are still many uncertainties and thus challenges involved in LCA studies, not the least the inclusion of indirect market and land use effects

7.3 Influence of methodological choices and assumptions

Figure 10 below illustrates how contrasting conclusions and positions can be associated with methodology choice, in this case the time at which accounting is commenced, relative to the first harvest for bioenergy. The upper diagram represents a case where accounting starts at the time of final felling of a single forest stand. If logging residues are extracted and used for energy instead of being left to degrade on site, this accounting would detect an initial C loss from the stand and if the avoided fossil C emissions are lower than the C emissions associated with the biomass use for energy, a C debt would be recorded that needs to be paid before the bioenergy systems reduces net GHG emissions. Based on this, a conclusion might be that it is better to continue using fossil fuels and leave the forest residues in the forest to decay over time. This is especially the case if the prime objective is to reduce net GHG emissions on the near term rather than to have an impact on the climate in the long term.

> Too much focus on near term GHG emission reduction targets can be counterproductive in combating climate change in a long-term perspective.

The lower diagram represents a case where the accounting instead starts at some time before the final felling. It reflects the view that a tree has to grow before it can be harvested and used. As an example, the World Bioenergy Association has stated that *"... the majority of assumptions in the theories on carbon debt and payback time of biomass are wrong, because they assume that first you burn the tree and then you grow it"* (WBA 2012).

This logic reasoning however does not reflect that studies commonly quantify net effects rather than absolute effects, i.e., the bioenergy case is compared with a reference case without bioenergy and it is the difference in C dynamics that is quantified so as to obtain information about the climate effects of bioenergy systems. Thus, also in a situation where the accounting starts at some time before the final felling, the conclusion can be that the extraction of harvest residues is associated with a net C emission pulse. This situation is only avoided if the C sequestration that takes place in the forest prior to the final felling is sufficiently much higher in the bioenergy case than in the reference case. Stand level assessments however rarely consider how bioenergy implementation affects forest management and growth (and they do not consider other dynamic effects).

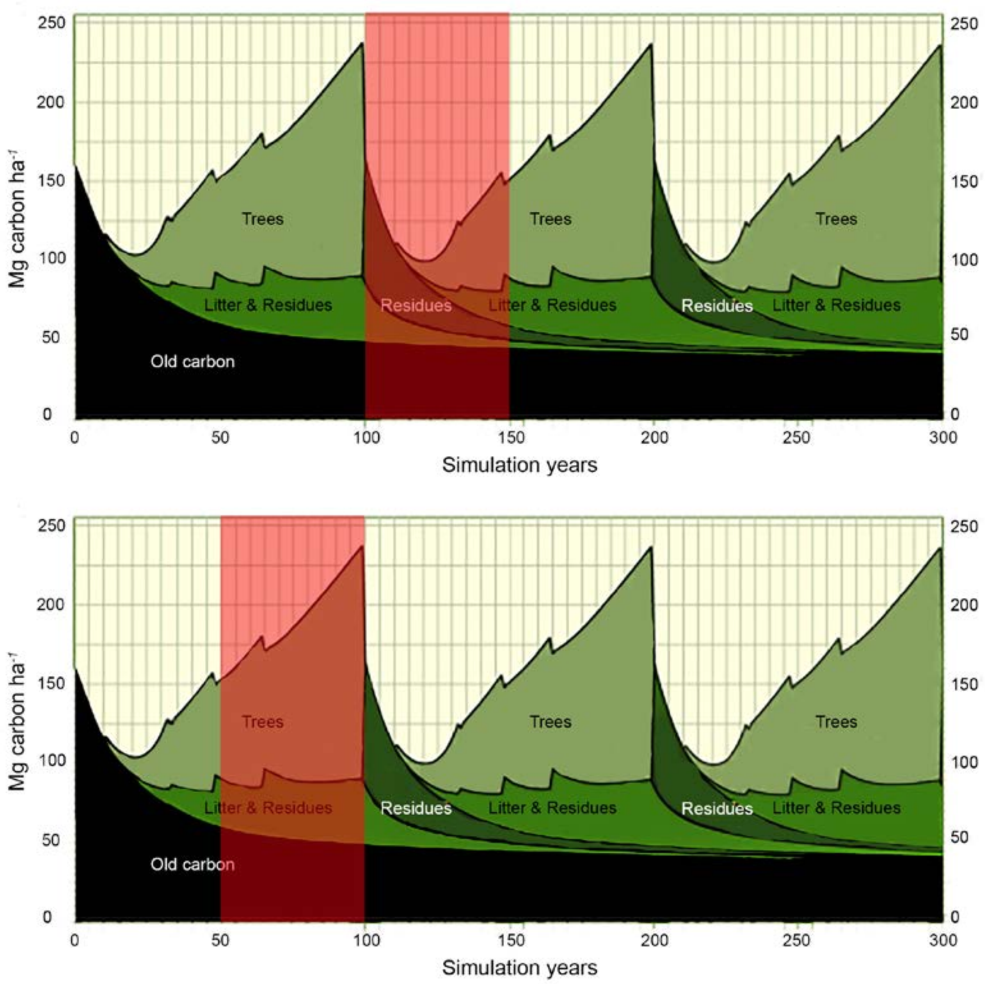

Figure 10. Accounting windows placed over the C dynamics in a managed forest stand over three rotations with three thinnings before final felling. Modified from Eliasson, et al. 2013.

> Studies that use a stand-level perspective have been questioned as potentially misleading due to that the outcome is strongly influenced by how the C accounting is made.

Studies that use a landscape scale – and consider location specific aspects – are more appropriate for addressing the question whether changes in forest C stocks, resulting from bioenergy incentives, affect the GHG mitigation benefits of bioenergy, and the timing of such benefits. The answers vary between different locations, due to variation in environmental and socio-economic factors. The change in forest management and harvesting regimes due to bioenergy demand depends among others on forest type, climate, forest ownership and the character and product portfolio of the associated forest industry. The forest carbon stock response to changes in forest management and harvesting in turn depends on the characteristics of the forests.

This is illustrated in Figure 11 showing modelled carbon pool changes in forests and cumulative additional biomass harvested over 100 years for three real forest landscapes in Sweden in response to an increased demand and therefore adaption to and harvest of forest biomass for energy as compared to a reference scenario with moderate demand for biomass for energy (Cintas et al. 2016). Due to differences in the forest conditions at the beginning of the modelling period and management responses (longer rotation periods and more frequent thinnings) the C-pool increases in two of the landscapes and decreases in one (Gbg) as compared to the reference case. Note that the C-pool decrease is moderate compared to the cumulative biomass harvest.

A larger landscape perspective, rather than the stand/field level, would in general be the appropriate scope for studies that intend to inform policy development.

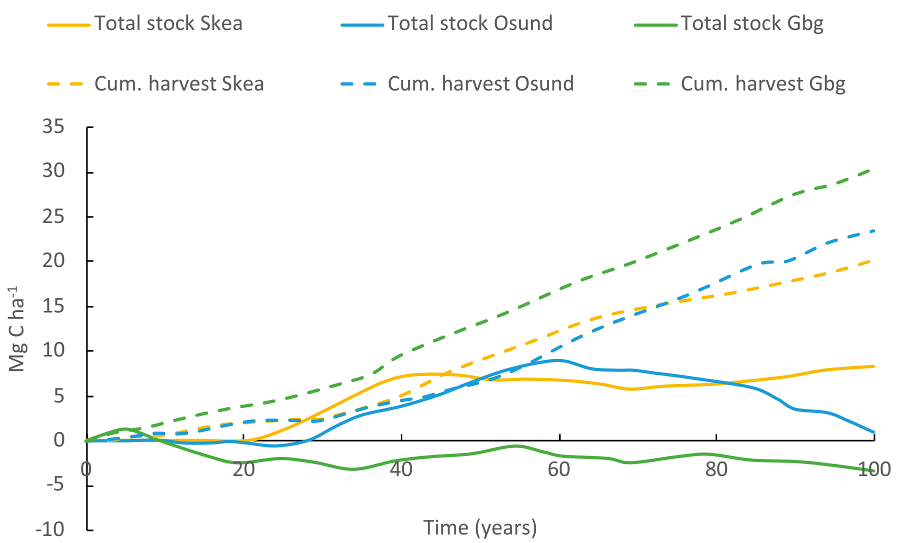

Figure 11. Modelled net C stock difference (BIO-REF) in forest pools (i.e., C in trees, soil, and litter) and in cumulative biomass harvested between a reference scenario with stemwood harvest (saw timber and pulpwood) plus slash harvest (tops and branches) in 40% of the final cuts (REF) and a bioenergy-adapted scenario as in REF, but with intensified extraction of slash i.e. slash extracted in approximately 45% of the thinnings and 60% of final cuts (BIO). Results are based on data from three real forest landscapes around the cities of Skellefteå (Skea, 9 170 ha), Östersund (Osund, 1 712 ha), and Göteborg (Gbg, 4 216 ha). Based on a figure in Cintas et al. 2016.

The cumulative additional biomass harvest (slash) in the bioenergy-adapted scenario (Figure 11) can then be used to displace fossil fuels in the energy sector. However, the character of existing energy systems determines the fossil fuel displacement – and thus the GHG savings – achieved from bioenergy use. This is illustrated in Figure 12, where GHG savings as a result of displacing coal or natural gas in the energy system have been added to the net C stock in the forests. As in many other studies displacing coal falls out as more beneficial than displacing natural gas.

Policy makers have to consider regional and local conditions for biomass production and energy system conversion. Common policies over large areas are unlikely to be optimal for climate change mitigation.

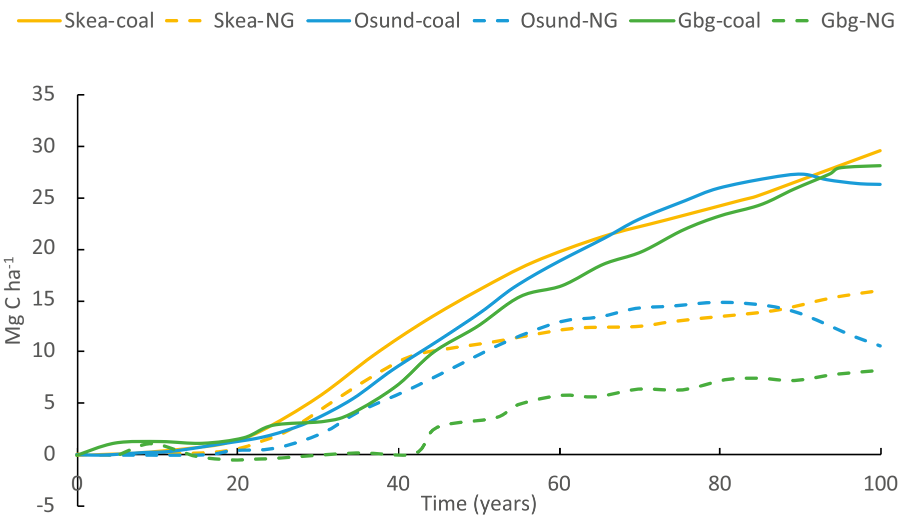

Figure 12. Net C stock (BIO-REF) in Skellefteå (Skea), Östersund (Osund), and Göteborg (Gbg) with bioenergy in bioenergy-adapted scenario (BIO) used to displace natural gas (NG) or coal used in the reference scenario (REF). Scenarios described in Figure 11. Based on a figure in Cintas et al. 2016.

Forestry in the Nordic countries is by far dominated by even-aged forestry and national forest inventory data have provided a lot of empirical data to fine tune growth models and thereby C sequestration estimates based on that silvicultural system. Empirical data from uneven-aged forestry systems and old growth forests is however scarce. This adds uncertainty into modelling studies comparing different silvicultural systems and studies with a counterfactual scenario where the forests are left to grow old. In Nordic forestry biomass is an integrated assortment together with saw timber and pulpwood and decisions to cut is driven by more valuable assortments than biomass for energy. Leaving the trees to grow in the absence of a bioenergy market is therefore not a likely counterfactual.

Decisions on harvest of trees are driven by other, more valuable, assortments than biomass for energy. Leaving the trees to grow in the absence of a bioenergy market is therefore not a likely counterfactual for the managed Nordic forests where biomass is an integrated assortment together with saw timber and pulpwood.

As forest biomass for energy in Nordic forests is an integrated assortment harvested together with saw timber and pulpwood, changes in the use of forest biomass for energy can have a direct impact on the output and use of other forest products with their potential climate impacts i.e. through material substitution. In a case where the indigenous output decreases and the use is maintained, wood has to be sourced from elsewhere causing an indirect effect with potential impacts on the climate. The same reasoning is valid also for impacts on the energy market with bioenergy included among other energy options. These market mediated effects complicate analyses further.

7.4 Market mediated changes caused by increased use of bioenergy

An increased demand for bioenergy will influence the bioenergy market and the bioenergy prices. However, increased demand for bioenergy can also affect other markets. For example, using grain for biofuel production can affect food prices and using wood for pellets can affect pulp and paper prices, causing effects on the production, distribution and use that in turn can change GHG emissions. The effect on other product´s markets is in this report referred to as the indirect effects of an increase in bioenergy demand. Note however that there is no coherence in use of the terminology in literature; indirect effects can also be referred to as leakage, spillover or rebound effects.

In the bioenergy debate, many different indirect effects have been discussed. The most commonly considered indirect effect is connected to land use, namely indirect land use change (iLUC). Before we go further into the issue, let us first distinguish between direct and indirect land use change. Converting land from one state to another (e.g. from forest cover to a crop field) to grow biofuel crops is referred to as direct land-use change (dLUC). Direct land-use change, however uncertain to project when contemplating effects of bioenergy incentives, can be observed and carbon stock changes can be measured.

If, on the other hand, biofuel crops are cultivated on previously existing farmland this might displace other crop production which may lead to land conversion elsewhere, referred to as iLUC. These effects are closely coupled with demand and supply of agricultural commodities, which ultimately can lead to a change in market behaviour leading to changes in land use and related GHG emissions. In other words, iLUC are the changes in land use that take place as a consequence of a bioenergy project, but are geographically disconnected to it. In contrast to direct land-use change, iLUC cannot easily be observed or measured, as it is the result of a series of consequences.

iLUC estimates tend to be more uncertain than dLUC estimates.

LUC emissions and inefficiencies due to leakage has been a concern for decades and LUC emissions were discussed in relation to the 2nd assessment report of the IPCC where bioenergy-intensive stabilization scenarios were included. The connection between bioenergy and LUC received wide recognition also outside the scientific community in 2008, when Fargione et al. (2008) and Searchinger et al. (2008) published studies claiming that LUC emissions associated with biofuel expansion could negate the GHG savings from displacing fossil fuels with biofuels. Since then the debate on iLUC has been intense. Many efforts have been made, primarily using global economic equilibrium models, to quantify the magnitude of iLUC (c.f. Ahlgren and Di Lucia, (2014). When compared to the early estimates of iLUC emissions caused by bioenergy projects (Fargione et al. 2008; Searchinger et al. 2008), subsequent estimates are lower as models have been updated to consider improved efficiencies in feedstock production, decreasing deforestation rates, and increasingly stringent regulation of agricultural practices, although large uncertainties remain (Macedo et al. 2014).

While the number of LCA-studies of biofuel has exploded, including iLUC in LCA studies is still rare, se e.g. review of Swedish biofuel LCAs in Martin et al. (2015). The reason is mainly because of the lack of consensus on the methodologies and the uncertainties of the results. The debate associated with the year-2008 publications (see above) also still frames the consideration of iLUC in the sense that many takes for granted that iLUC and other indirect effects are per definition associated with increases in net emissions, which is not true.

Many takes for granted that iLUC and other indirect effects are per definition associated with increases in net emissions, which is not true.

The discussions of indirect effects have mainly concerned the use of agricultural crops for biofuel production. However, the same theories are applicable to forestry products. A diversion of wood to the energy sector implies that other sectors will have to source wood from elsewhere, or they will use other, potentially more GHG intensive materials like concrete and metals. Similar effects can also be achieved by decisions or policies suggesting using forests to sequester carbon. These indirect effects on the wood market are sometimes called indirect wood use change, iWUC (Agostini et al. 2013).

Evaluations of such effects need to consider regional and global markets, e.g., whether a shift from pulpwood to biofuel production occurs due to increasing competition from an expanding bioenergy sector or due to declining pulpwood demand in the region, a situation in which forest owners (in the absence of a bioenergy market) may leave forests unattended, change their forestry practices, or shift to other land uses.

Also, the GHG and other consequences of ramping up pulp and paper production elsewhere (to compensate for lower production where biofuels are prioritized) can vary, e.g., new pulpwood production may stem from conversion of mature forests into tree plantations (forest carbon loss), tree planting on degraded pastures (carbon gain), or improvement of previously neglected forests that might otherwise have been converted into pastures (avoided carbon loss).

An increase in demand for wood in one sector can also have effects on fuel use in other sectors. For example, if wood for production of transportation fuel is increasing, the price and use of energy for cooking, heat and electricity might be affected. This is referred to as indirect fuel use change, iFUC (Agostini et al. 2013).

To complicate matters more, bioenergy also affects the fossil energy systems (which is desirable since the ultimate goal of promoting bioenergy and other non-fossil options is to support the dismantling of fossil energy infrastructure). In many LCA-studies, it is assumed that one energy unit of bioenergy replace one energy unit of fossil fuel. However, an increased use of biofuels can lower oil prices and therefore result in a rebound effect, i.e., increased crude oil consumption (Rajagopal et al. 2011; Rajagopal, 2015). It can also lead to reduced investments into maintaining or extending oil supply capacity. As with iLUC, these indirect effects are difficult to estimate.

A review study by Smeets et al. (2014) of first generation biofuel rebound effects report a wide range of values, from −20% to 119%, meaning that one unit bioenergy substitute between -0.2 and 1.2 units of fossil energy. Results depend among others on the biofuel policy, the applied method and the model parameter assumptions.

Focus has been put on direct land use change (dLUC) and on indirect land use change (iLUC), but the reasoning is relevant also for indirect effects on markets for forest products (iWUC) and energy (iFUC).

To summarize, an increase in demand of bioenergy can lead to different indirect effects/leakages of climate impact in other geographic areas and other sectors. To quantify these leakages, some type of modelling is required.

A combination of biophysical, climate and socio-economic models is required to understand the full climate effects of bioenergy, including effects on parallel industries (wood products, agriculture and energy).

The modelling exercises have so far given large varieties in results, revealing major uncertainties and inconsistencies in used methodologies. Therefore, the inclusion of leakage effects in LCA studies has been limited. For Swedish bioenergy systems, there is a large lack of such studies.

7.5 How is biomass best utilized?

Biomass can be used for a number of different applications, e.g. production of biofuel, chemicals, heat, electricity, or as feedstock in the forest product industry. Is there an optimal allocation of different biomass assortments to different applications, from a cost-effective, resource or environmental point of view? And can implementation of policies, such as CO_2-taxes, steer the development?

These are of course very complex and dynamic questions to answer, requiring a system modelling approach. One such model is the MARKAL Sweden model, which is a bottom-up based cost-optimization model developed for analysing the Swedish energy sector, allocating biomass to different sectors. Results from the model vary, mainly depending on the studied time horizon (assumptions of technological development) and policies being applied.

For a policy scenario with 80% CO_2-reduction (compared to 1990 emission level) in the national energy sector, the MARKAL model results show that bioenergy makes up 36% of the total energy supply in Sweden in 2050. The largest increase in bioenergy use is in the transport sector; 43% of the total primary biomass goes to biofuel production in 2050. This is explained by increasingly stringent CO_2 restrictions and biomass availability constraints, which gives higher biomass prices. As a consequence, sectors more easily able to switch to other energy sources will do so (within the constraints of the model and scenarios), e.g. wind power becomes more cost-efficient in the stationary energy system (Börjesson et al. 2015).

This higher competition for biomass feedstock is modelled to lead to high biomass prices, which gives a high degree of utilization of high-cost biomass sources such as stumps, dedicated energy crops, and pulpwood. Figure 13 shows bioenergy sources that would be used in a scenario with stringent CO_2-reduction policies implemented across all energy sectors.

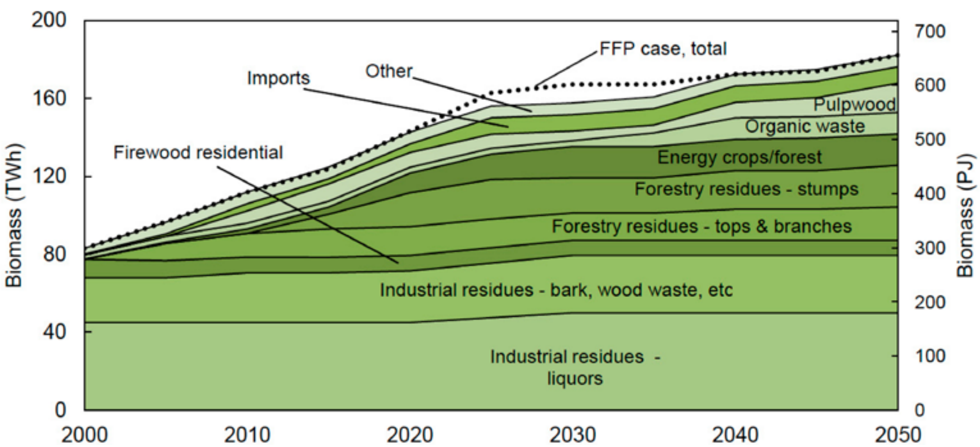

Figure 13. Biomass deployment from MARKAL-model in a policy scenario (Fossil Fuel Phase-out, FFP) where 80% CO_2-reduction (compared to 1990 emission level) in the national energy sector is mandated (Börjesson et al. 2015).

The fact that the substitution effect often is higher when material like concrete and metals are substituted rather than fossil fuels (Sathre and O'Connor 2010) has been used as an argument for cascading use of wood. The concept of cascading has been raised frequently in discussions concerning renewable energy, the "bio-based" economy and the "circular" economy within EU without a clear definition (Olsson et al. 2016). Although there is a lack of consensus on what cascading means, biomass for energy tends to be given the lowest value in cascading hierarchies. However, practical experiences do not support cascading as an efficient tool to reach set goals (Olsson et al. 2016). Rather than putting a ban for some markets to use certain biomass, stimulations for preferred use of biomass could be an alternative. One example could be to stimulate the use of wood in buildings and other constructions at the expense of i.e. concrete and metals. Since construction wood is a higher priced commodity this could stimulate investments in forest management and growth among forest owners at the same time as the C-pool in buildings increases, substitution effects increase, and the stream of lower value residues from harvested sites, sawmills and end-of-life construction wood increases. This increased residue stream could then feed into the energy industry or other industry that can make use of low quality wood.

Based on a number of different scenarios and a multi-model approach Braun et al. (2016) assessed the GHG-dynamics of Austrian forests and wood use. They did confirm that material use of wood had a much higher climate mitigation efficiency than energy use. But, since the energy demand in the scenarios was much higher than the material demand, the climate mitigation potential was eight times higher for energy use than for material use. This is important to remember in the cascading discussions if the prime aim is to mitigate climate change.

> Material substitution has a higher climate mitigation efficiency per unit of wood used than fossil fuel substitution – but due to the extensive use of fossil fuels the climate mitigation potential is much higher for fuel substitution.

We have also seen studies showing that leaving the trees to grow rather than harvest and use them for energy gives the best climate mitigation outcome (e.g. Holtsmark 2013; Soimakallio, et al. 2016). Although both studies are rather optimistic concerning the growth rates of forests as they grow older, the results are, within their system boundaries, probably correct given the models used and assumptions made.

Holtsmark (2015) did not consider the fact that within the Nordic countries biomass for energy is an integrated assortment together with saw timber and pulpwood and that saw timber is the prime driver for harvest – not biomass for energy. Thus, leaving the trees to grow is not the most probable counterfactual in managed forests.

In the study by Soimakallio, et al. (2016) all harvest operations in Finland are terminated in one scenario. For such a scenario to be relevant all use of wood sourced from Finnish forests including the significant export of forest products will have to be terminated too. Again, such a scenario is not particularly likely considering the importance of the forest industry in Finland and the market demand for forest products. A maintained market demand for forest products in such a scenario is likely to result in sourcing of wood from elsewhere (iWUC, iLUC) with its climate impacts.

Giving up forestry in a country also leads to a rapid loss of forestry infrastructure including knowledge, manpower and machinery. This means the readiness to salvage-log trees after major disturbances will be highly limited leading to potential future GHG-losses to the atmosphere (c.f. Barrette, et al. 2015). This risk is likely to increase with forest age and possibly also as an effect of climate change (c.f. Reyer, et al. 2017; Seidl, et al. 2014). It is also reasonable to assume that in the absence of incomes from forestry, some forests will be converted to other land use – a conversion that in most cases coincides with C-loss to the atmosphere (cf. Guo and Gifford 2002).

Long-term carbon storage in forests is a risky business due to natural disturbances from fire, wind, snow, pests and diseases – a risk that increases with stand age and a changing climate.

Based on C-stock changes in soil, trees and durable forest products only, the study by Gustavsson et al. (2017) suggested that a set-aside scenario would gain the best climate mitigation benefits from the managed in Sweden during the next 100 years (cf. Figure 3). However, this C-pool approach does not consider GHG-effects of input energy in logging operations and transport, GHG-effects of fertilization, or effects of substituting fossil fuels and materials (concrete and steel). Comparing the same scenarios as in Figure 3, but accounting also for GHG-emissions and substitution effects gives a different result. The set-aside scenario shows a moderate reduction in cumulative GHG-emissions that declines over time as compared to the BAU scenario, whereas the production scenario continues to deliver reductions in GHG-emissions over the whole assessment period (Figure 14).

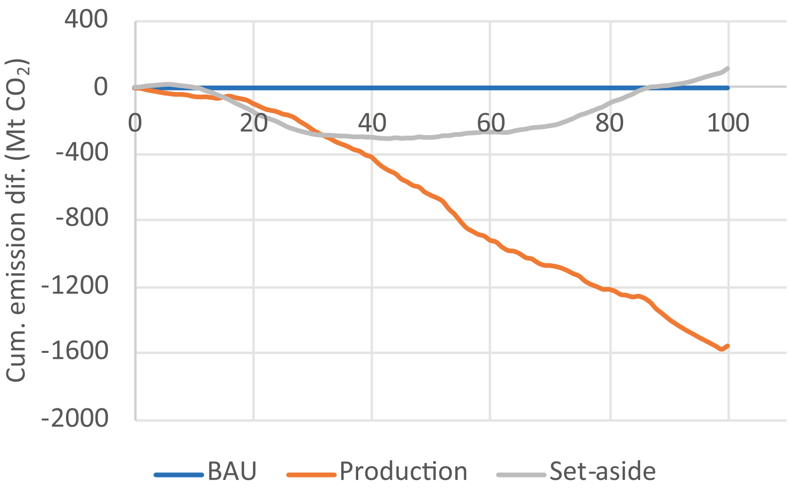

Figure 14. Difference in cumulative CO_2 emission (million t CO_2) as a result of different management strategies in Swedish forestry over 100 years. A business as usual scenario (BAU), here set to 0, is compared with a production scenario where a number of silvicultural measures to increase forest growth is introduced, and a set-aside scenario where the area of set-aside is doubled as compared to BAU. In all three scenarios, annual harvest equals annual growth and slash corresponding to 8 TWh is harvested for generation of heat and power each year. Based on data from Gustavsson et al. 2017.

When interpreting these results and results from other studies, assumptions behind the studies are critical. Here the harvest level in the BAU-scenario equals annual growth, whereas current levels in Sweden are below annual growth (c.f. Figure 2). Volumes of construction wood withhold from the market in the Set-aside scenario is here assumed to be substituted with concrete and steel. However, with a maintained market for forest products these quantities may as well be sourced from forests harvested elsewhere with its impact on the global carbon stock. In the Production scenario, additional construction wood on the market is assumed to substitute concrete and steel. This presupposes that the market is willing to accept more wood at the expense of steel and concrete which is not necessarily the case. At least not without (political) actions facilitating and promoting increased use of wood in buildings and other constructions.

Assumptions made are often critical for the outcome of modelling studies on climate impacts of biomass use.

Although reductions in GHG-emissions are often targeted by policy the ultimate target is to reduce the energy input to the atmosphere. One metric for this is cumulative radiative forcing that quantifies the damage to the planet from GHG-emissions over a determined length of time. Applied on the CO_2-emissions in figure 14 this metric gives a slightly different result – but still with the production scenario delivering the best climate mitigation impact (Figure 15a). In all three scenarios logging residues corresponding to 8 TWh bioenergy is harvested each year (approximately current deliveries in Sweden). If more of available slash and stumps are harvested for bioenergy purposes the climate mitigation impact is further enhanced (Figure 15b.).

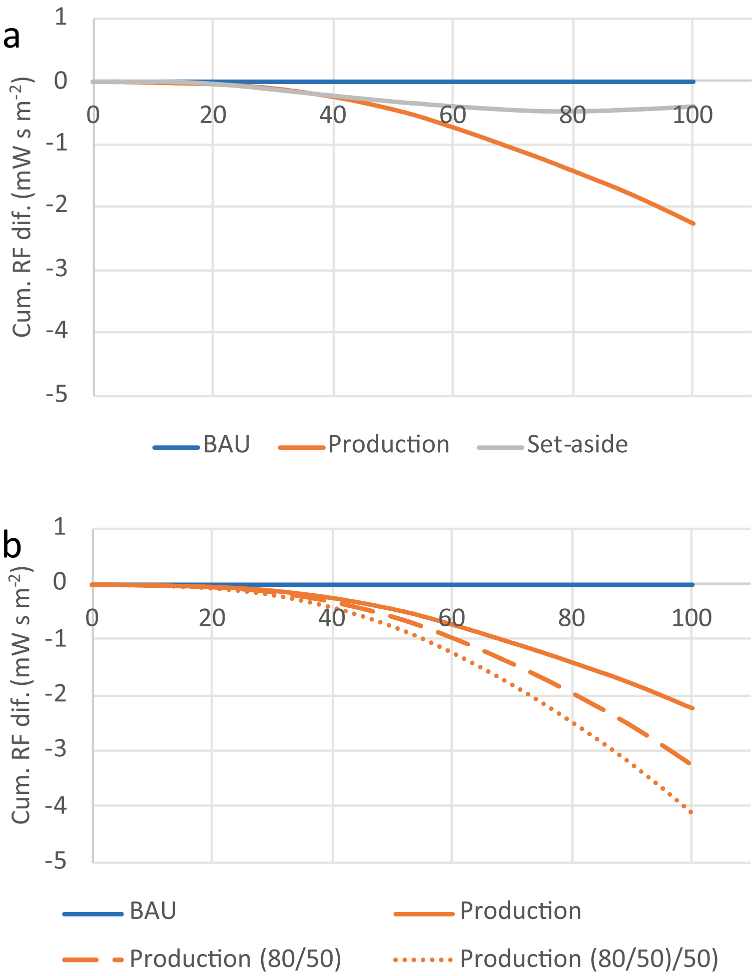

Figure 15. Difference in cumulative radiative forcing (mW s m^{-2}) as a result of Swedish forestry over 100 years. A business as usual scenario (BAU), here set to 0, is compared with a production scenario where a number of silvicultural measures to increase forest growth is introduced, and a set-aside scenario where the area of set-aside forestland is doubled as compared to BAU. In all three scenarios, annual harvest equals annual growth and slash corresponding to 8 TWh is harvested for generation of heat and power each year (a). If more logging residues are harvested in the production scenario, 80/50 = 80% of slash in final thinning and 50% in thinnings, 80/50/50 = as 80/50 plus 50% of the stumps in final felling, the climate mitigation impact is further enhanced (b). Based on data from Gustavsson et al. 2017.

An important message from the study by Gustavsson et al. (2017) is that depending on the system boundaries chosen, different management strategies falls out as the most beneficial to mitigate climate change. Although all assumptions in model studies, e.g. the magnitude of the substitution effect and the C-sequestration potential of set-aside over time can be questioned, we argue that these analyses have to include all C-stock changes, GHG emissions, and the substitution effect of the full suite of products produced to provide useful results for policy.

> To be useful for policy, analyses of the GHG-balance of biomass production systems have to include all C-stock changes, GHG emissions, and the substitution effect of the full suite of products produced.

Another important message from the study is that the increased use of wood to substitute materials and fossil fuels in the production scenario continues to deliver climate mitigation over time, whereas the positive effect of setting aside diminishes over time.

> The effect of using wood to substitute materials and fossil fuels continues to deliver climate mitigation over time, whereas the positive effect of setting aside forest land diminishes over time.

While focus here has been on GHGs, climate is also influenced by changes in the atmospheric concentration of aerosols, solar irradiation (cloudiness), and land surface albedo. Ideally these effects should also be included in the analyses making them more complete and complex.

The study by Gustavsson et al. (2017) also suggests that with annual harvest levels not exceeding annual growth levels, climate benefits from forestry in Sweden are gained both through means to increase forest production and by harvesting more of the available tree biomass (stumps and slash) to substitute fossil fuels.

> A management strategy that maintains or increases the forest C stocks while supplying the marked with a large annual yield will generate a positive and sustained GHG mitigation.

However, this implies a more intense forestry with potential impacts on other ecosystem services that forests supply and on biodiversity. These effects also need to be balanced to meet the requirements of sustainable forestry.

8 Sustainability – not just a matter of climate impacts

8.1 Sustainability criteria in the EU

In 2009, EU adopted the Renewable Energy Directive (RED) (Directive 2009/28/EC, will be replaced by RED II for the period 2021–2030). The Directive mandates that all Member States shall have 10% (on energy basis) biofuels in the transport sector by 2020. In order for a biofuel to be accounted within the national reporting, it must meet a number of sustainability criteria as described in the Directive. Biofuels must also meet the sustainability criteria to receive financial support, such as tax exemptions.

To be counted as a sustainable according to RED, raw material for the biofuel production cannot be sourced from primary forests, nature protection areas or highly biodiverse grasslands. Land with high carbon stocks such as wetland or peatland can only be used under certain circumstances (i.e. if the land use has not been changed since January 2008). Social and economic sustainability criteria are included in the directive but are not mandatory for a biofuel producer to meet.

The sustainability criteria in the RED only apply for biofuel for transportation, although the process to also include solid biomass for heat, electricity and cooling is on-going. In 2010 the European Commission issued a first report which contained a calculation methodology as well as typical and default values for solid and gaseous bioenergy pathways. These proposed values have been updated on a couple of occasions (Giuntoli et al. 2015).

Another step was taken in 2013, in a draft proposal from the European Commission for solid and gaseous bioenergy. The document points out three types of no-go areas; (1) land with high biodiversity value i.e. primary forests, nature protection areas and highly biodiverse grassland; (2) converted land with high carbon stocks i.e. conversion of wetlands and continuously forested areas to other land use categories; and (3) drained peatland. Further, any forest biomass used in energy installation must come from sustainably managed forest in line with international principles and criteria.

Several of the definitions in the draft sustainability criteria for solid biomass, such as primary forest, highly biodiverse grassland and wetlands, have already been identified as problematic (Fritsche & Iriarte, 2014; Thiffault et al. 2015). For example, the term "primary forest" is defined as forestland without clear human activity. However most of European forests have been affected by human activity for centuries or millennia. If the policy makers want to protect highly biodiverse boreal forests, other definitions are needed. Thiffault et al. (2015) suggest that "old-growth" forest might be a better definition. Further, Thiffault et al. (2015) point out that common sustainability criteria will be difficult to realize in the forestry sector due to a tradition of national and regional differences in implementation, reporting, monitoring and auditing in already existing governance systems.

In early 2014 the EC announced that binding sustainability criteria for solid and gaseous bioenergy would not be proposed in the short term (EU SWD, 2014). In the meantime, a few Member States (e.g. Netherlands, Belgium, UK) have adopted their own binding GHG saving criteria for biomass used in electricity/heating. The Netherlands has also implemented a volume cap on biomass co-firing in coal-fired power plants (Fern, 2016).

During the preparation of this report new ambitious targets for the period 2021–2030 have been agreed among EU institutions in an update of RED (RED II, June 2018) including 32% renewables in the total energy use by 2030, a minimum share of 14% biofuels in the transport sector by 2030, a maximum share (7%) for first generation biofuels based on food crops, and a phasing out of biofuels based on palm oil. While RED only covered transport fuels, RED II also defines sustainability criteria for biomass fuels used for power, heating and cooling production. The new directive, will succeed the existing regulation and enter into effect on January 1, 2021.

An update of RED (RED II, 2021-2030), covering also sustainability criteria for biomass fuels used for power, heating and cooling production has been agreed upon and will enter into effect on January 1, 2021.

As a response to e.g. RED and a global call for sustainable bioenergy ISO (International Organization for Standardization) started its work with "Sustainability criteria for bioenergy" resulting in ISO 13065:2015. The standard specifies principles, criteria and indicators for the whole bioenergy supply chain, or a single process in it, including all three pillars of sustainability, environmental, social and economic.

A few countries have also introduced regulations aimed at potential competition with existing biomass uses. In Belgium, woody feedstocks suitable for the wood-processing industry are not eligible for the Flemish Green Power Certificates. In Poland a policy has been implemented excluding the use of stemwood with a diameter above a certain size from being eligible for national financial incentives for renewables (EC, 2014). The potential competition with other biomass uses and the prioritized order of use is also mentioned in several EU strategy documents (e.g. the bioeconomy strategy, the forest strategy and the circular economy package). This is often referred to as resource cascading and has parallels to the thinking on waste hierarchy (Olsson et al. 2016), i.e. high value products are prioritized over low value. This implies that e.g. chemical and material use of wood should be prioritized over energy use of wood.

There are several risks connected to implementing cascading principles in regulation. Previous experiences (e.g. from the Swedish Wood Fibre Law introduced in the 1970s and terminated in 1991) indicate that policy implementation of cascading principles can result in complicated legislative processes and difficulties in reaching agreement on what wood assortments should be used for material purposes and therefore should be excluded from energy use. Further complications are likely to arise if the cascading principle is enforced only in the EU. Without internationally harmonized rules, the efficiency of cascading policies could be compromised as market actors focus more on exploiting regulatory loopholes than on improving their performance (Olsson et al. 2016).

Cascading works as a guiding principle for raw material use – but several risks are connected to implementing cascading principles in regulation.

8.2 Sustainability criteria – effect on bioenergy production and trade

8.2.1 Biofuels for transportation

In Sweden, the sustainability criteria for biofuels and bioliquids have been implemented in law (Act 2010:598), regulations and guidelines. The Swedish Energy Agency is the authority to which the economic operators have to report annually.

The production and use of biofuels in Sweden have increased sharply since 2005. The share of renewable fuels in the domestic road transport was 12% in 2014 and when calculated according to the methodology of the RED, the share was 19%. Policy limits on biofuels for transportation according to RED II puts a cap on crop-based biofuels (max 7% from food and feed crops). Of the biofuels used in Sweden in 2014, 58% was produced from crops. This adds up to crop-based fuels being around 7.2% of the total use in the transportation sector in Sweden (Swedish Energy Agency, 2015a, b), meaning there is limited room for further producers of first-generation biofuels on the Swedish market.

Policy limits for food- and feed-based biofuels according to RED II currently leave limited room for further expansion on the Swedish market based on current conditions.

In the biofuel industry it was feared that the implementation of sustainability criteria would lead to increased administration and costs, lowering the competitiveness against fossil fuels that are not burdened with the same demands on sustainability criteria. However, Stupak et al. (2015) conclude from a survey study in the biofuel industry that most companies report that introducing new sustainability governance had resulted in positive or no changes to their production and trade. Another survey study however concluded that many actors avoid sourcing raw material from countries where sustainability issues may be a problem (Harnesk et al. 2015).

Several bioenergy assortments, such as bioethanol and wood pellets, are internationally traded with high dynamics in the flows due to policy and market factors and that new producers and users quickly emerge (Lamers et al. 2011; Pacini, 2015). In the presence of regulatory heterogeneity, there is a risk that producers seek markets with lower regulatory compliance costs, e.g. in countries such as USA, China, India, Japan and South Korea (Pacini, 2015). This could limit the competitiveness and marketability of biofuels with low GHG emissions. Further, investors could be discouraged to enter the emerging second-generation biofuel market if sustainability criteria require a high number of administrative tasks (Johnson, 2011).

With regulatory differences between bioenergy markets, producers may seek markets with low compliance costs rather than markets for bioenergy with low GHG emissions.

On the other hand, producers delivering biofuels with low GHG emissions can find markets for their products in countries with high GHG reduction demands. During the last year we have for example seen an increase of sales of ethanol from Sweden to Germany, since Germany in 2015 changed the biofuel quota from calorific value, to the reduction of GHG emissions. So, for Sweden, low GHG profile of biofuels gives good opportunities for trade, which can be expected to increase in the future.

8.2.2 Solid biomass

As previously mentioned the sustainability criteria for biofuel are already in effect, while regulations for biomass used for power, heating and cooling production will come with RED II that will enter into effect in 2021. A few studies have tried to predict what effects an implementation of solid biomass sustainability criteria in the EU would have on trade of biomass.

In a publication by Lamers et al. (2014) future global trade of solid biomass is modelled under different sustainability constraints. The study uses an optimization model coupling supply and demand nodes to reach minimum total biomass supply costs. The study explores different constraints and different bioenergy deployment rate scenarios. The findings show that the projected EU solid biomass demand by 2020 can be met almost exclusively via domestic biomass, given that use of domestic agricultural residues and energy crop potentials are increased sharply. Excluding pulpwood pellets may drive the supply costs of import dependent countries, foremost the Netherlands and the UK, whereas excluding forest biomass altogether (except for processing by-products, e.g. black liquor, bark, shavings, sawdust etc.) may entail higher costs for countries like Sweden that rely on regional biomass.

In a study by Galik and Abt (2015) supply of wood pellets from SE United States to the EU under different sustainability criteria is modelled. The study finds that, if the demand for pellets increase, the demand would primarily be met by pulpwood thus the price of pulpwood would be affected. This price effect is modelled to lead to a higher degree of harvesting in existing forest stands, especially of those forests which are most responsive to market conditions i.e. not sensitive low-productivity land. The increased demand of pellets is also projected to lead to new plantations of pine. In total, the increase in demand modelled (5,9 million tonnes of pellets) leads to little change in forest inventory and a net forest C gain rather than loss over time.

A model study suggests that increased EU demand for wood pellets from South Eastern US has little impact on the growing forest stock with a C gain rather than a C loss over time.

In a Swedish context, a couple of studies have analysed potential impacts of an implementation of sustainability criteria for solid biomass. Hansson et al. (2014) conclude that Swedish pellet producers are well-suited to meet demands of pellets with low associated GHG emissions and based on feedstock with low social and environmental impacts. Similarly, Gustavsson et al. (2014) find that the introduction of sustainability criteria for solid biofuels should not affect the market for pellets in Sweden to any great extent as Swedish pellets have good GHG performance. However, traceability of origin would be a requirement if sustainability criteria are implemented, something which is lacking at the moment in the pellets distribution chain. This could potentially lead to increased costs for biofuel suppliers.

To be effective, bioenergy policy needs to be more complete in targeting a wider scope of agricultural and forestry sectors and more comprehensive in its membership of countries (Frank et al. 2013). Another solution often brought up in the debate is to instead use broad and all-inclusive political instruments, such as a CO_2-tax. However, such broad legislation seems difficult to agree upon in the international arena.

The biofuel industry act on a market with many different stakeholders, governmental regulations and sustainability demands, and also have to deal with public opinion on how natural resources should be allocated. Although the intention of the sustainability criteria is to increase the sustainability of bioenergy, the effects are not always predictable. Detailed regulations have a way of leading to unexpected and unwanted development. Further, the regulation covers only the bioenergy sectors; the absolute effect of sustainability criteria is difficult to estimate as environmental impacts can be reallocated to other sectors. Further, there is a very narrow interpretation of sustainability in bioenergy regulation, in principle only GHG emissions are properly covered.

8.3 Other sustainability impacts of Swedish bioenergy systems

Climate impacts of bioenergy have been given a lot of attention among policy makers, in science, in the public debate, and in this report. There are also other environmental sustainability issues to consider when deploying large-scale bioenergy systems. A major issue in Sweden when biomass for energy from our forests, as a result of the two oil crises in the 1970s, were considered, was long term site productivity and thereby its future potential to produce feedstock to primarily the forest industry. Other important issues that have been studied over the years, also in *The Biomass Fuel Program,* are biodiversity, acidification, eutrophication and toxic substances (e.g. methyl mercury).

In a synthesis report de Jong et al. (2017) used published research results on these issues and the environmental objectives with defined milestones decided by the Swedish parliament to identify sustainable harvest intensities with focus on slash and stumps from conventional forestry. Their conclusions were that an increased use of wood for bioenergy would have impacts on environmental objectives, but that the impact could be both positive and negative. An increase from current harvest levels was suggested possible, particularly if specified mitigation and compensation measures were implemented. The latter includes wood-ash recycling, nitrogen fertilization, and for biodiversity, to compensate the harvest of low-value dead wood (e.g. slash of pine and spruce) with high-value dead wood (e.g. high stumps).

Although measures to increase primary production together with increased harvest intensities often results in increased pressure on the environment, there are cases when it doesn't or when it has the potential to strengthen environmental values. Examples are biodiverse semi-open forested landscapes that needs to be kept open to maintain their biodiversity values (Götmark, 2013) and biodiverse, extensively managed grasslands where ingrowth of woody species may reduce its biodiversity value (Ebenhard et al. 2017). Wherever invasive plants are combated bioenergy may offer a market that can contribute to the fight (Van Meerbeek et al. 2015). Other examples are purification of soils with short rotation woody crops (*Salix*) and trees (Pulford and Watson 2003), buffer strips with perennials to protect surface waters and as wildlife refuges in otherwise open agricultural landscapes (Börjesson 1999).

Keeping also options where the environment benefits from emerging biomass markets in mind in operational and policy planning processes, preferably at the landscape scale, will benefit the biomass market development and the environment.

9 Suggestions for future research

From reading the literature it is obvious that more process-based knowledge is needed to better understand soil C dynamics in agricultural and forests soils and to feed that knowledge into future soil C models. The modelling of tree growth and, hence, C sequestration in trees has been fine-tuned by means of a lot of empirical data from national forest inventories. But this data originates to a large extent from forests managed with an even-aged forestry system. More data emanating from uneven-aged forestry systems and from set aside forests as it grows older are needed in order to get better prediction about the C-sequestration potential following such forest management decisions. The same reasoning is valid also for farmland where we need to know more about the GHG-balance for e.g. dedicated energy crops and less common primary production systems. There is also need for more research on changes in the atmospheric concentration of aerosols and land surface albedo, with their potential impact on cloud formation and energy balance.

Since the climate impacts go beyond the GHG-balance of a bioenergy production system there is a need for more complete and complex modelling including a combination of biophysical, climate and socio-economic models. This is a challenge since the complexity adds uncertainty into the modelling. It is therefore desirable to strive towards a common protocol for these modelling approaches in order to make it possible to compare results from different studies.

Since sustainability goes beyond climate impacts there is also a need for studies balancing climate regulation against other ecosystem services provided by forestland and farmland at the landscape level (cf. de Jong et al. 2017).

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A sustainable energy system benefits society

The Swedish Energy Agency has an overall picture of the supply and use of energy in society. We work for a sustainable energy system, combining ecological sustainability, competitiveness and security of supply. The Agency:

- Develops and disseminates knowledge about a more efficient energy use to households, industry, and the public sector.
- Finances research for new and renewable energy technologies, smart grids, and vehicles and transport fuels of the future.
- Supports commercialisation and growth of energy related cleantech.
- Participates in international collaboration with the aim of attaining Swedish energy and climate objectives.
- Manages instruments such as the Electricity Certificate System and the EU Emission Trading System.
- Provides energy system.

Swedish Energy Agency

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