Sustainable biomass for energy and materials: A greenhouse gas emission perspective

Working paper[1]

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1 Introduction

Bioenergy is a major source of renewable energy. Its global contribution to fulfil energy needs is at present about 55 EJ per year and is expected to strongly increase in the coming decades. However, its sustainability is debated in both the scientific arena and the public domain. Two recent examples are related to the publications of a vision paper published by the Royal Netherlands Academy of Arts and Sciences (Katan et al. 2015) and a report by the World Resources Institute (Searchinger & Heimlich 2015). Both sparked a heated debate in the Netherlands and the United States, respectively. Katan et al. (2015) sketch a pessimistic picture of the benefits, impacts and potential of bioenergy. Also Searchinger and Heimlich (2015) provide a similar perspective: they claim that bioenergy can neither contribute significantly to global energy supply without increasing food insecurity, nor can it help reduce emissions compared to fossil fuels. While there is complete agreement that biomass for energy must be produced sustainably, the two reports' views on very limited contribution from bioenergy to energy supply and emission reductions are contested. For example, the Netherlands Environmental Assessment Agency (PBL) responded to the vision paper by Katan et al. (2015) by emphasizing the importance of bioenergy (particularly in combination with carbon capture and storage) in limiting climate change (PBL 2015). PBL further explains that the question is not about whether or not to use biomass but about how to produce biomass so that it indeed reduces emissions compared to fossil fuels. Also in the US, immediate responses were made to the Searchinger and Heimlich (2015) report, presenting a different perspective. For example, Wang and Dunn (2015) from the Argonne National Laboratory provide numerous arguments opposing the conclusions drawn by Searchinger and Heimlich. For example, Wang and Dunn (2015) point out that by-products of biofuel production are not included in the analysis although they supply food or feed that does not need to be produced elsewhere. Similar to PBL, also they conclude that a sound design of bioenergy policies is the key (Wang & Dunn 2015). In addition, the United Nations Food and Agriculture Organization recently suggested that it is necessary to move from a "food vs. fuel" to a "food and fuel" debate (FAO 2015). The inclusive debate indisputably considers food first and only afterwards searches for generating opportunities from bioenergy.

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One of the reasons for this heated debate is that biomass production systems (and associated land use changes), supply chains and end-uses differ widely, and so do their environmental and socio-economic impacts, e.g. on carbon stocks, water, soil, air, biodiversity, land tenure and food security. The direction (positive or negative) and the magnitude of these impacts primarily depend on the type of energy crop, the biophysical and socio-economic conditions of the location of production, production technologies, the design of the supply chain and the final use (including which fossil fuel or material use is replaced). The present paper focusses on greenhouse gas (GHG) emissions because a key reason for stimulating the use of bioenergy and for an increasing interest in biomaterials is the need to reduce GHG emission associated with using fossil fuels. It aims at i) providing an overview of current research on the GHG emissions performance of biomass for energy and material applications, ii) clarifying the nuances between different studies and underlying reasons for apparent disagreements, and iii) discussing the risks and possible opportunities of bioenergy. A better understanding of these aspects allows identifying key entry points for policy making in order to ensure that bioenergy and biomaterials contribute to the reduction of GHG emissions. Clearly, for bioenergy and biomaterials to be sustainable, also other impacts than just GHG emissions need to be considered. This requires an integrated analysis of all potential environmental and socio-economic impacts and the trade-offs between them, as is already discussed in, for example, Chum et al. (2011) and Dornburg et al. (2010).

Before addressing the GHG emissions from different components of the supply chain and the aims of this study mentioned above, the next section first provides an overview of biomass supply and demand. This is done in order to show the anticipated role of biomass in the future and placing the discussions around the GHG emissions from biomass for energy and materials in context.

2 Supply and demand of bioenergy and biomaterials

2.1 Biomass potentials for energy and novel materials[2]

Biomass resources for energy and novel biomaterials can be roughly separated into two categories: residues and dedicated crops. Residues encompass a wide range of agricultural and forestry residues from harvesting and from processing[3] as well as post-consumer residues[4]. Biomass from dedicated crop production can come from surplus agricultural land (assuming a food-first principle) and from marginal or degraded land. Another source of biomass feedstock could come from increasing the output of existing forests by management changes. Next to land-based biomass, aquatic biomass

[2] The technical potential of biomass supply in resource assessments accounts for key competing uses of land and biomass. For example, the future use of biomass for food, fodder, and traditional materials (e.g. paper, timber, fibres) is already included and given priority. However, the additional use of biomass for novel biomaterials (e.g. biochemicals and bioplastics) are not included. The potentials therefore must be considered for energy and novel material purposes.

[3] Agricultural residues include, for example, corn stover, wheat and rice straw (harvest residues) and sugarcane bagasse, and risk husks (process residues). Forestry residues encompass, for example, tree tops and branches (both harvest residues), sawdust, shavings, and bark (process residues).

[4] Examples of post-consumer residues are demolition wood, the organic fraction of municipal solid waste, sewage sludge.

(microalgae, seaweed, sea grass) is another category of biomass resources. This category is not addressed here because of the lack of estimates and high level of uncertainty (GEA 2012).

Slade et al. (2014) recently identified key factors and preconditions necessary to achieve different levels of biomass production worldwide (Figure 1). The preconditions are mostly related to how much residues are used, how much agricultural land expands, how food demand develops, and how agricultural productivity changes. A detailed literature assessment combined with model assessment by Dornburg et al. (2010) approximated the technical potential to range between 200 – 500 EJ/yr in 2050. Key parameters identified in their study defining the technical potential are agricultural efficiency and crop choice. However, other assessments have suggested lower technical potentials. According to GEA (2012), the total technical potential varies between 162 – 267 EJ in 2050, mainly due to lower expectations about dedicated crops. Haberl et al. (2010) have estimated the technical potential for 2050 to range between 64 – 161 EJ/yr (where the range depends on climate impacts, yields and human diet).

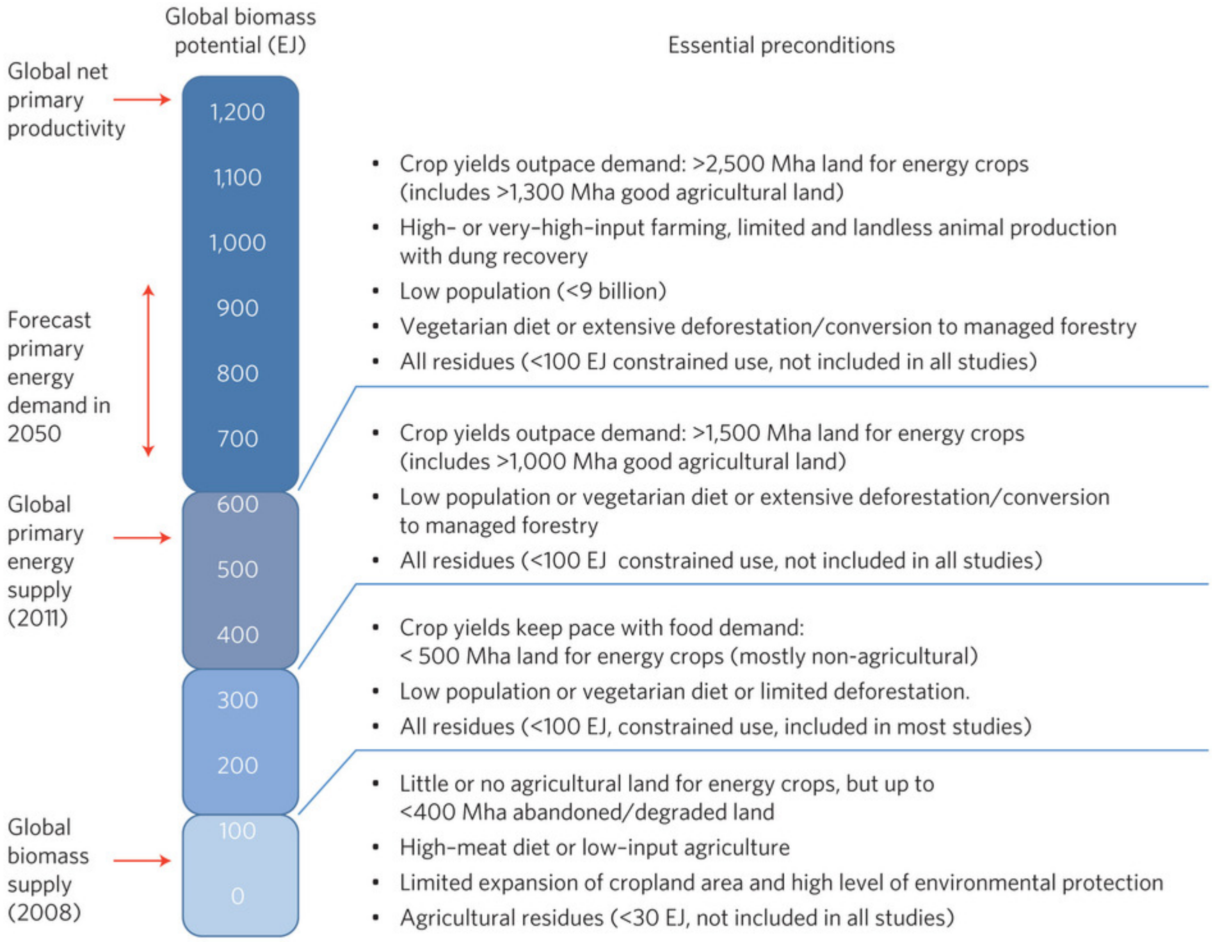

Figure 1: Essential preconditions for increasing levels of biomass production (Slade et al. 2014).

Although nearly all technical potential studies account for key environmental and socio-economic constraints (e.g. food first principle, exclusion of forests and protected areas), few studies evaluate the sustainable implementation potential. Especially the sustainable potential of dedicated energy crop production is not well understood (Slade et al. 2014). Accounting for additional environmental, social, economic and institutional constraints will undoubtedly result in a sustainable implementation potential that is lower than the technical potentials presented above (Searle & Malins 2014). For example, Schueler et al. (2013) conducted a detailed analysis of how the sustainability criteria of the European Renewable Energy Directive reduce the theoretical potential and found that only 10% (or about 100 EJ) of the theoretical potential would not be affected by the sustainability criteria. Another study by Searle and Malins (2014) reassessed existing potential studies by aligning their key assumptions on land area, energy crop yields, production costs, political stability and heating value of the biomass. They found that the maximum sustainable potential for 2050 would then be 60 – 120 EJ/yr.[5] .

2.2 The need for bioenergy and novel biomaterials

With increasing anthropogenic GHG emissions, largely caused by the use of fossil fuels and land-use change, many integrated assessments show that we need a range of (renewable) technology options to meet stringent global warming targets (e.g. keeping CO_2 concentration below 440 ppm by 2050). A review of 164 scenarios from 16 large-scale integrated models revealed that there is not one single dominant renewable technology (Fischedick et al. 2011), But rather, all available low-carbon renewable energy options have to be developed to meet the set targets. To meet a <440 ppm target, the use biomass for energy is projected to increase from currently about 55 EJ per year to between 80 - 180 EJ per year (depending on the model and scenario) by 2050 (GEA 2012; Fischedick et al. 2011). This large increase in biomass for energy is partly due to the fact that liquid biofuels have high energy densities and are therefore particularly well suited for heavy transport (e.g. trucks, marine shipping and aviation), which at least for the short and medium term has few or no renewable alternatives available.

Next to bioenergy, novel biomaterials have received increased attention due to concerns about finite fossil fuel resources, GHG emissions, and reducing the generation of non-degradable plastic waste. The worldwide annual capacity of biobased plastics has increased from 0.36 million metric tonnes (Mt) in 2007 (Shen et al. 2010) to approximately 1.6 Mt in 2013. Still, the biobased polymer industry is small compared to the synthetic polymer industry (ca. 315 Mt p.a. in 2007). The total (biobased) feedstock energy required for today's novel biomaterials is estimated at 0.04 EJ[6], compared to 22 EJ final fuel use as feedstock for total synthetic chemicals and polymers (Saygin et al. 2014). However, the global production of bio-based plastics is expected to continue growing strongly in the near future to about 6.7 Mt by 2018 (0.16 EJ) (IfBB 2015). Long term projections show that the economic potential to use biomass as a feedstock for synthetic chemicals and polymers varies between 15-17 EJ in 2050 (Saygin et al. 2014).

[5] Searle and Malins (2014) warn that the actual mobilization of this sustainable potential requires such large commitment that actual implementation is likely to be lower. At the same time, they make conservative assumptions, for example, with regard to the use of forest residues from natural forest, which they consider would not be possible at all. However, this is an assumption which is challenged by e.g. Lamers et al. (2013).

[6] Based on the market volume for 2013 (1.6 Mt production capacity) and the volume projection for 2018 (EuBP (2014): http://en.european-bioplastics.org/market/). Only biobased feedstock energy are taken into account based on HHV. These numbers represent the theoretical minimal biomass feedstock requirements. They are not the same as non-energy use as defined by the International Energy Agency (IEA); that is, non-energy use is fuel that is not consumed for energy, but used for raw materials.

3 GHG emission performance of supply chains (excluding carbon stock changes)

Processes related to the cultivation, collection, pre-treatment, transport, conversion and final use of bioenergy and bio-based materials comprises greenhouse gas (GHG) emissions from land use (change), the consumption of (fossil) energy, process utilities and waste production. Treatment of bioenergy as carbon neutral therefore implies in most cases an overestimation of GHG savings from real systems. This section deals with the GHG emissions of supply chains excluding land use-related net changes in carbon stocks. These are addressed the next section.

Many life cycle assessments (LCAs)[7] of biomass supply chains have been conducted, especially for first generation bioenergy crops, and large ranges in energy balances and GHG emission reductions have been found. Figure 2 shows the ranges found in LCA studies for transportation fuels, electricity and heat compared to fossil reference systems per unit of energy output counterparts if land use-related net changes in carbon stocks and land management impacts are excluded (Chum et al. 2011). The large ranges are mainly explained by the variety in biomass supply chains, agricultural management applied, the biophysical conditions of the location of production, the design and management of the supply chain, the conversion technologies, the choice of process energy (fossil fuels vs. renewables) and the reference system the biomass supply chain is compared with. In addition, variations in outcomes are caused by differences in research methods applied including differences in system boundaries, allocation procedure and uncertainties in data sets (e.g. Cherubini et al., 2009; Chum et al., 2011). Including land use-related emissions makes this variation even larger (see Section 4).

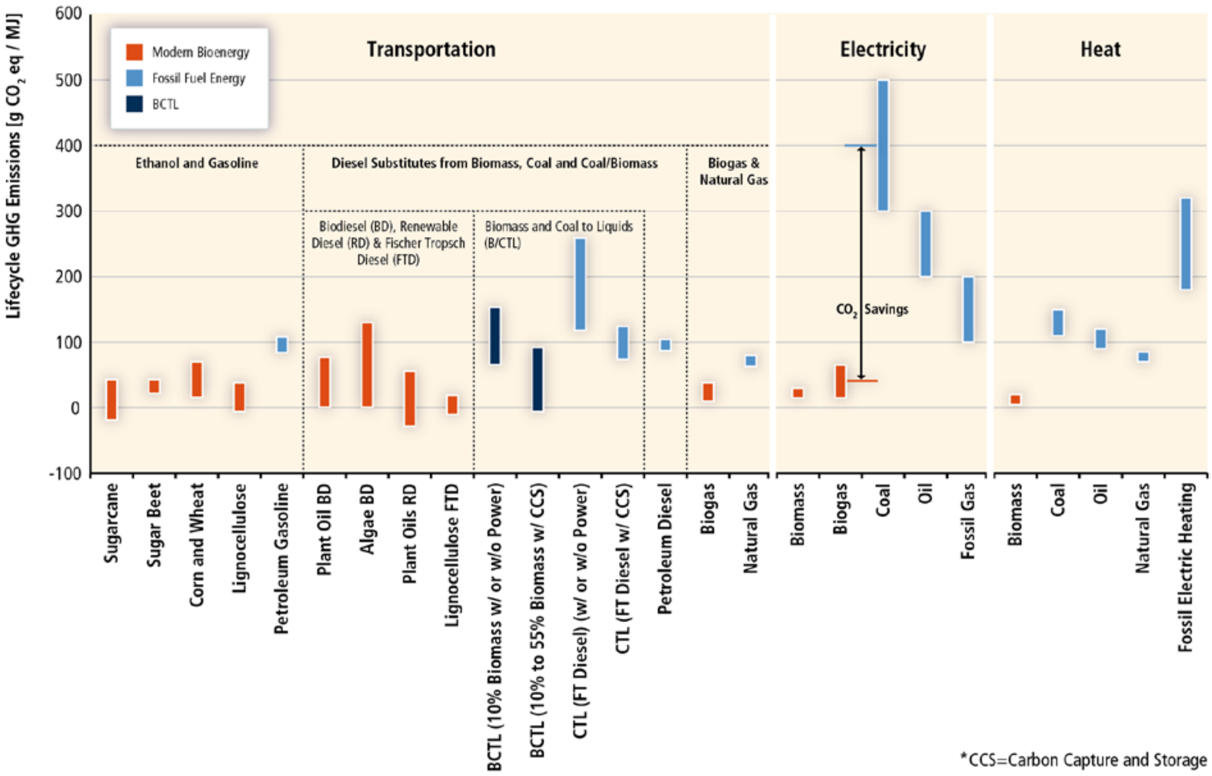

Figure 2 Ranges of GHG emissions per unit of energy output compared to selected fossil reference systems (land use-related net changes in carbon stocks and land management impacts are excluded) (Chum et al. 2011).

[7] LCA has been a widely accepted tool to assess the GHG emissions and (fossil) energy use in bioenergy production systems and potential GHG emission reductions compared to fossil fuels (Plevin et al. 2014).

Novel biomaterials cannot be compared directly to bioenergy systems because of the differences in functional units (energy versus materials). However, like for energy, large ranges are found for biomaterials (Chen & Patel 2011; Hottle et al. 2013; Weiss et al. 2012). Table 1 shows an overview cradle-to-factory gate GHG emissions of various biobased polymers compared to their petrochemical counterparts. From the table it can be seen that the GHG emissions of biobased polymers are strongly influenced by the type of biomass feedstock (e.g. PHA with and without the use of agricultural residues used to fuel the process and biobased PE made from sugar cane or from corn), technological progress (e.g. today and tomorrow's PLA and biobased PTT) and the allocation methods used by an LCA, especially for avoided emissions from by- or co-products when agricultural residues are used to fuel the process (Kim & Dale 2008; Hottle et al. 2013).

Table 1: Cradle-to-factory gate GHG emissions of biobased polymers as compared to petrochemical PE (polyethylene), PP (polypropylene), PET (polyethylene terephthalate) and PTT (polytrimethylene terephthalate), adapted from Chen & Patel (2011).

Product	Feedstock	Technology	Cradle-to- factory gate GHG emissions ($t\ CO_2$ eq./t plastic)
PHA	Corn	Today	0.7 to 4.2[a]
	Corn	Near future	-1.2 to 1.7[b]
PLA	Corn	Today	0.4 to 1.3
PE	Corn	Today	-0.34
	Sugar cane	Today	-2.05
	Petrochemical	Today	1.8 to 2.1
PP	Corn	Medium term	> 0.2 to 0.3
	Petrochemical	2005	2.0
PET	Corn	Today	1.4
	Sugar cane	Today	1.0
	Petrochemical	Today	2.15
PTT	Corn	Today	1.6 to 2.51
	Petrochemical	Today	1.9 to 3.55
PBS	Bio-SA[c]	Future	2.3 to 3.9

[a] Without use of agricultural residues. [b] With (-1.2 t CO_2 eq./t) and without (1.7 t CO_2 eq./t) use of agricultural residues. [c] Biobased succinic acid.

The biophysical conditions of the location of crop cultivation, management and related yields, fertilizer application and soil N_2O emissions are key factors that determine the GHG performance of feedstock production for biofuels. For that reason, systems that use organic waste or unused residues as a feedstock are in most cases found to have the lowest GHG footprint (highest savings) followed by systems that use lignocellulosic energy crops. In general , these crops perform better than system that use annual plant food crops because these crops have higher yields[8], require less

[8] Lignocellulosic crops in Europe (miscanthus, switchgrass, short rotation): 90 – 225 GJ/ha/yr compared to 54-58 GJ/ha/yr for wheat and 60 – 70 GJ/ha/yr for rapeseed. Sugar beet crops have higher yields (116-158 GJ/ha.yr) (Chum et al. 2011), but only under good agro-ecological conditions (van der Hilst et al. 2012).

fertilizers, and require less pesticides as they are less susceptible for plagues and diseases (Chum et al. 2011).

Today however, most biofuels are produced from food type crops (1st generation) as progress of 2nd generation biofuels has been slower than expected (Janssen et al. 2013). Nevertheless, also 1st generation biofuel systems can result in GHG emission reductions well over 60%. For example, ethanol produced from Brazilian sugar cane crops using unconvertible biomass (bagasse) for heat and power results in GHG savings between 65-80% (Cherubini et al. 2009). In contrast, if coal is used for process energy, biobased production could lead to an increase in GHG emissions compared to the petrochemical reference of biofuel (Turkenburg et al. 2012) and novel biomaterials (Gerssen-Gondelach et al. 2014; Hottle et al. 2013).

Lignocellulosic biomass from forestry and agriculture residues or dedicated crops, are in their raw form often difficult to mobilize and need to be processed (pre-treated) in order to improve handling for transport to end-users; especially if long distance transport is required. These operations results in additional energy requirements and GHG emissions. However, even long-distance transportation does not need to result in prohibitive GHG emissions. JRC (Giuntoli et al. 2014) estimated GHG emissions from wood pellet supply chains between the US and Europe to contribute between 4 CO_2-eq /MJ for wood pellets from industrial residues to 30 g CO_2-eq /MJ for stemwood dried with a natural gas fired boiler.

Wood pellet supply chains between North America and Europe that are used today mainly for large scale electricity generation, achieve GHG savings above 60%, and above 70% if efficient conversion systems are used (Giuntoli et al. 2014) or 80% if substitution of coal is assumed (Sikkema et al. 2010). Similar or higher savings could be achieved when 2nd generation biofuels would become commercially available. Well-designed lignocellulosic biofuel production systems use unconvertible biomass to generate process heat and electricity avoiding the use of fossil fuels for process energy and providing excess electricity to the grid (Turkenburg et al. 2012). Overall GHG system performance can be further improved by using CO_2 capture and storage (CCS, see also Figure 2). However, the technology is not yet applied on a large scale and the future deployment is uncertain.

4 GHG emissions from direct and indirect land use change effects

The cultivation of dedicated energy crops results in direct land use change (dLUC): land use is changed from a previous use to the cultivation of energy feedstock. This, in turn, can lead to indirect land use change (iLUC): a change of land use that is induced by a biofuel project or mandate but occurs geographically disconnected from the biofuel feedstock production (Ahlgren & Di Lucia 2014).
The conversion of land from its original use to the cultivation of biomass for bioenergy could result in carbon sequestration (removing CO_2 from the atmosphere) or in carbon emissions due a change in above and below ground biomass and/or soil organic carbon. The loss of carbon due to the land use or the land management change for bioenergy is called the carbon debt of bioenergy (Fargione et al. 2008). *Over time,* bioenergy from biomass from converted land *can repay this carbon debt if their lifecycle GHG emissions are less than the emissions of the fossil fuels they displace* (Fargione et al. 2008). However, when this debt is very large the carbon payback time will be long and it becomes

increasingly uncertain if the carbon benefits will ultimately occur. Also, the time-lag in reducing atmospheric concentrations of greenhouse gasses resulting from the pulse emissions from the use of bioenergy and the low sequestration in soil and plant biomass, should be taken into account when considering biomass for a short term climate change mitigation option (Berndes et al. 2011). The magnitude of the change in above and below ground biomass and soil organic carbon as a results of converting on type of land use/cover to bioenergy crop cultivation depends on the original land use, the type of biomass cultivated after conversion, the management of land before and after conversion, and the local biophysical conditions such as soil and climate conditions (Gibbs et al. 2008). Even if the land cover remains the same (e.g. forest) but the management or the use changes, it could result in a change in carbon stocks. Large carbon losses occur when e.g. forests are cleared or when organic soils are managed. This results in carbon payback times of decades or even centuries. However, when low carbon stock land e.g. (abandoned or marginal) agricultural land or degraded pastures are used for the cultivation of high yielding crops or woody / herbaceous biomass, soil and phytomass carbon is likely to increase.

4.1 Carbon debt for wood-based energy

The topic of carbon debt has been particularly prominent for woody biomass. This is because a tree can theoretically be harvested and used for energy in a matter of weeks or months, releasing CO2 to the atmosphere. But it can take over 100 years (in boreal forests) for a new tree to re-sequester this carbon, whereas in a hypothetical reference scenario this tree would still have sequestered carbon till the tree has grown up and dies (releasing CO2). The increasing use of wood pellets in e.g. coal power plants in North-Western Europe combined with the uncertain climate benefits of using wood for electricity production has been highlighted by NGOs (Birdlife 2010; RSPB 2012) and has led to a substantial debate amongst scientists (see e.g. (Agostini et al. 2013; Lamers & Junginger 2013; Miner et al. 2014; Ter-Mikaelian et al. 2015; IEA Task 38 2014)). The time it takes before the use of wood for bioenergy achieves higher GHG emission savings than a reference scenario can range from several years to more than several 100 years (Agostini et al. 2013; Lamers & Junginger 2013; Buchholz et al. 2015). The 'pay-back' time highly depends on a number of factors, such as which fossil fuel is replaced (coal leads to a shorter time than natural gas), the efficiency of the supply chain, the forest regrowth rate, whether the effect of wildfire is included, and the counterfactual scenario, i.e. what would have happened if the feedstock had not been used for bioenergy (Lamers & Junginger 2013; Buchholz et al. 2015). The exact methodology how to calculate a carbon debt and pay-back time is still debated amongst scientists (Ter-Mikaelian et al. 2015; IEA Task 38 2014).

To illustrate the complexity of the carbon debt debate, we take the South-eastern United States (SE US) as an example given it is currently a major source of wood pellets to replace coal in the Netherlands, the UK, Belgium and Denmark (Abt et al. 2014; Goetzle 2015). The use of especially pellets from hardwood from bottomland swamp forests to produce electricity in EU power plants has been heavily criticized by NGOs, because of long carbon payback times issues related to sustainable forest management practices and biodiversity (NRDC 2013). However, the current predominant source for pellet production is from existing softwood pine plantations, also used to produce pulp and paper products and timber for construction (Abt et al. 2014). Different counterfactual scenarios could e.g. be i) the forest is allowed to grow further, or ii) forest biomass is harvested for material purposes only (e.g. pulp and timber) and replanted, or iii) harvested for material purposes and afterwards converted to agriculture or urban development. Studies show that depending on the specific counterfactual scenario, using wood from pine plantations in the SE US results in payback times between 12-50 years (Colnes et al. 2012; Jonker et al. 2014), or (in an even wider set of scenarios) emissions from electricity production can range from achieving negative emissions (net carbons sequestration) to being worse than burning coal (Stephenson & MacKay 2014). But especially if the productivity of existing plantations for bioenergy through changes in management can be increased, this may result in short payback times (12-23 years, based on Jonker et al. 2013). In fact, Abt et al. (2014) show that due to higher demand for bioenergy, conversion from natural pine stands to (more productive) pine plantation may occur. In such a scenario, effectively the same amount of carbon would be sequestered as in a no-pellet scenario – thus avoiding a carbon debt (but at the same time causing a major loss of biodiversity). Which mix of counterfactual scenarios is going to occur is difficult to predict, but has a major influence on the time period until any carbon debt is repaid. A key aspect to consider is that more than 85% of the forests in the SE US are owned by private landowners (Kittler 2015). The vast majority of these land owners uses their forests for multiple purposes like recreation, and hunting) but ultimately as a form of investment for wood production. However, different segments of the housing market and also of the pulp and paper markets (the two major markets of wood from the SE US) have either remained stable or declined in the past decade (McKeever & Howard 2011). Without markets for wood, land owners are more likely to convert their land to agriculture or sell it to a real estate developer. In fact, until 2060, up to one quarter of the current forested area in the SE US is projected to be deforested because of urban expansion (Wear & Greis 2013). The sustained, long-term demand for wood for pellet production can help slow this trend down (Miner et al. 2014).

4.2 ILUC

In addition to the direct effects of LUC on the carbon balance, a key point of discussion relates to indirect land use change (ILUC) effects from bioenergy production. This happens when food crops are displaced or diverted to biofuels while demand for food remains constant, and/or when a biofuel mandate results in high crop prices that incentivize crop production elsewhere (Wicke et al. 2012). While direct LUC for biofuel feedstock production can be observed, measured and easily allocated to biofuel production, an actually occurring LUC elsewhere cannot easily be allocated as an indirect effect of biofuels. Therefore, ILUC is modelled. GHG emission impacts of ILUC, generally expressed as an emission factor for direct and indirect land use change[9] per unit biofuel (in g CO_2-eq/MJ biofuel), have been assessed with different models. But economic models (especially general equilibrium models, e.g. (Tyner 2010; Laborde 2011; Laborde et al. 2014)) are most often used as they can account for interlinkages between economic sectors and regions. Various studies have come to (largely) varying LUC factors (see Figure 3 for LUC emissions from corn ethanol by different studies). Although varying, all studies find positive but significantly lower results than the values originally suggested by Searchinger et al. (2008). Differences exist between biofuel types (bioethanol and biodiesel), and assumptions in the key modelling parameters. Regarding the biofuel type, sugar and starch-based bioethanol performs significantly better than vegetable oil-based biodiesel (Malins 2011). Key reasons are that i) demand for vegetable oil is less elastic than for cereals so that prices increases have less effect on vegetable oil demand than on cereals, ii) different vegetable oils are largely substitutable and, because palm oil is the cheapest vegetable oil, use of any vegetable oil for

[9] Economic models cannot distinguish between direct and indirect LUC effects. Therefore, results refer to total LUC-related GHG emissions induced by biofuels, also referred to as LUC factor.

biodiesel results in palm oil production expansion, and iii) oil palm cultivation is associated with high GHG emissions from deforestation and peatland conversion.

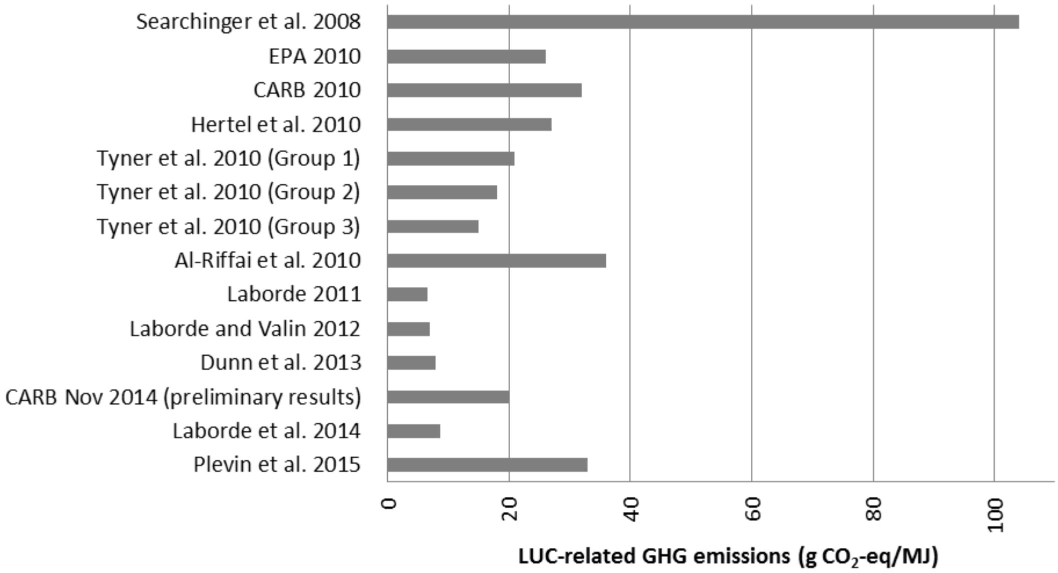

Figure 3: LUC-related GHG emissions from corn ethanol (30 year allocation period) estimated by different studies, sorted by date of publication , updated from Wicke et al. 2011

The differences between studies can be explained by the different assumptions made for key parameters. Prins et al. (2014), Malins et al. (2014), and Plevin et al. (2015; 2010a) identified the key modelling parameters as i) level of agricultural intensification (including the price-yield elasticity), ii) amount of agricultural expansion (including the elasticity for land expansion) and yield on newly converted land, iii) use of co-products, iv) consumption level (effect of price changes on food demand), v) trade, vi) land use/cover change and resulting emissions. In a detailed uncertainty analysis of LUC emissions with an economic model (GTAP-BIO-ADV) linked to an emission factor model (AEZ-EF), Plevin et al. (2015) identified the crop price yield elasticity (through which the development in yields on existing cropland is modelled) as the factor that contributed by far the most to variation in LUC emission factors.

Uncertainty analyses of LUC factors (e.g. (Plevin et al. 2010b; Plevin et al. 2015; Laborde 2011; Verstegen et al. 2015)) indicate large ranges in outcomes. For example, as a result of varying the values of the key parameters listed above, Plevin et al. (2015) estimate the 95% confidence interval to range from 18 to 55 g CO_2-eq/MJ (for scenario "food consumption fixed") with a mean of 33 g CO_2-eq/MJ. The uncertainties in these parameters thus have great influence on the results, but are not likely to decrease in the short term due to the nature of ILUC that it needs to be modelled (Prins et al. 2014). But models include complex interactions of multiple processes on local, regional, and global level and apply various inherently uncertain assumptions. In addition to parameter uncertainties that have are examined in the uncertainty analyses listed above, "changes to the model structure have the potential to shift the mean by tens of grams of CO_2e per megajoule and further broaden distributions for ILUC emission intensities" (Plevin et al. 2015). Therefore, more research is needed to better understand model structure uncertainties, to improve approximation of the key assumptions and to further develop modelling approaches for assessing ILUC. In addition, given ILUC is a key factor in the GHG balance of bioenergy, also a better understanding of how ILUC can be mitigated is needed.

5 Conclusions and policy implications

Many studies and models have indicated that by 2050 between 80-180 EJ of biomass for energy will be needed to achieve stringent climate change targets. Additionally, approximately 15 EJ of biomass may be demanded for biomaterials. It is thus not a matter of *whether or not* modern biomass applications should be part of the energy and material mix, but of *how* it can be produced and used sustainably. While technical biomass potentials are in principle sufficient to meet this demand, sustainability constraints may limit biomass availability to 100 EJ or less, depending largely on future land-use efficiency for food production. Competition between different applications is therefore possible in the future.

At the same time, large differences exist in direct and indirect GHG emissions between different feedstocks and supply chains. These differences are largely defined by the type of crop, design of the supply chain, end-use, counterfactual feedstock use, and previous and counterfactual land use (as explained in the previous sections). Given potential competition between various applications for sustainable biomass feedstocks and differences in their GHG emission performance, policy must aim to simultaneously achieve large reduction of GHG emissions in the supply chain, address emissions from land use change (particularly indirect effects) and meet increasing demand for biomass. Therefore, policy makers in collaboration with other stakeholders need to clearly define boundary conditions in which biomass feedstock is preferred under different conditions, where and how to produce it, how it is converted and how it is used. The implementation of criteria related to GHG emissions can help answer these questions and ensure that the use of biomass indeed leads to robust savings of energy and GHG emissions.[10] Existing criteria differ for different applications of biomass: GHG criteria for *biofuels* in the Renewable Energy Directive (2009/28/EC) require minimum savings (including direct land use change and supply chain emissions) of 35% today increasing to 50% in 2017 and 60% for new installations in 2018. Savings of 35% can be met easily by most systems when net carbon emissions from land use change are not considered. But increasing the minimum saving criteria or making other policy changes[11] will stimulate better GHG performance of biofuels or the use of better performing systems. Solid biomass used for *power, heating and cooling* are currently exempted from EU wide criteria. However, some member states have implemented sustainability criteria, including GHG saving criteria, in national schemes. For example, in the UK support for heat under the Renewable Heat Incentive, and electricity (>1 MW) require a minimum

[10] Important to note is that results of LCA studies should be dealt with caution when defining system emissions for policy targets. This is because there is a large variation in results induced by methods and real system variations (Plevin et al. 2014). Legislation can account for this by allowing producers to prove that they can perform better than the default system emission defined for policy targets.

[11] For example, Germany recently changed its system for biofuels in Germany from an energy based quota system to a GHG based quota system to increase GHG reduction in transport from 3.5% in 2015 to 6% in 2020. This stimulates the use of better performing supply chains, which in turn incentivizes improvements in the GHG emission performance. But whether this change will also ensure that Germany reaches its 10% renewable transport energy target in 2020 is uncertain. http://www.energypost.eu/new-german-legislation-will-shake-eu-biofuels-market/

GHG savings of 60% and are scheduled to increase to 75% for new installations by 2025 (Pelkmans et al. 2014). In the Netherlands, it was recently agreed on minimum savings of 70% for heat and electricity compared to EU reference values (Energie Nederland, 2015). Setting such increasing targets over time both incentivizes the development of cleaner technologies, e.g. 2nd generation lignocellulosic biofuels, and provides industry with certainty which targets (both quantitative volumes of bioenergy and GHG emission reduction) they have to achieve. This limits the regulatory risk and makes long-term investments in better performing technologies possible. GHG emission criteria or broader sustainability criteria for the use of biomass for *biomaterials* are not yet defined. However, only applying GHG emission reduction criteria for energy end-uses could lead to unwanted leakage effects, e.g. the use of inefficient food crops for biomaterials. If biomaterials are supposed to help reduce GHG emissions compared to oleochemicals, the same issues as for bioenergy need to be addressed in future policy making.

Carbon debt and ILUC are the key challenges for the GHG emission performance of bioenergy[12] (see Section 4). Therefore, regulation and governance is needed in order to ensure that bioenergy contributes to GHG emission reductions. As discussed in section 4.1, the overall GHG reduction potential of woody biomass also depends on whether its use for bioenergy is linked to a carbon debt and associated payback time. The question what payback times are acceptable for bioenergy options to make meaningful contributions to mitigate climate change is still debated amongst policy makers and other stakeholders. The size of any carbon debt and time that is required to repay it depends to a large extent to which fossil fuel is assumed to be replaced, what changes in forest management would occur and what the relevant counterfactual scenario would have been. While in the UK attempts have been made to quantify these to support policy design, so far no policy measures to address carbon debt have been taken. This is due to the large number of possible scenarios and substantial variation in results. In the absence of a broadly agreed on framework to quantify carbon debts of woody biomass, it may still be possible to address such risk by relatively simple measures. For example, the recent Dutch covenant by industry and NGOs aims to mitigate the risk of high carbon debts and limit the competition with material applications from wood pellets used in Dutch power plants. To this purpose, the covenant amongst others i) defines specific feedstock categories, ii) limits the share of biomass used for energy purposes to less than 50% of the total woody biomass extracted in any given year and area, disallows conversion of (semi-natural) forests after 2008 and requires written evidence that C-stocks in forest must be maintained or increased (Energie-Nederland 2015). While such criteria are not invulnerable to misconduct, they significantly limit the risk of stimulating the use of feedstocks with high carbon debt risks, and instead allow the development of sustainable woody biomass production and supply chains.

With regard to mitigating and preventing ILUC, Section 4.2 provided an overview of the challenges and uncertainties in determining emissions from ILUC. Given ILUC is found to be above zero, the question is how to deal with ILUC in the face of these uncertainties. Some studies (Finkbeiner 2014; Prins et al. 2014) have concluded that economic modelling results are not appropriate for determining LUC emission factors to be used in policy making because of these uncertainties.

[12] These challenges are also relevant for biomaterials, although implications of the use of biomass for biomaterials in terms of carbon debt and ILUC have not been studied in detail. Thus, in the following we focus our description on bioenergy. However, the implications are similar for biomaterials.

Although other researchers agree that results should not be used for single point estimates, they consider these LUC emission factors (and particularly their distribution curves from uncertainty analyses) still useful for policy making (Muñoz et al. 2014; Witcover et al. 2013; Plevin et al. 2015). This is because these factors and distribution curves can illustrate effects of possible pathways, which can be used for identifying risks and opportunities of these pathways. These differences in opinions and interpretations can also be found in policy making. For example, in the US, national and California-state regulations have already implemented LUC factors. In Europe, the potential use of LUC factors is still debated (e.g. the European Commission proposed to include estimated ILUC emissions in reporting only, while the European Parliament proposed accounting for ILUC-related emissions for the Fuel Quality Directive, Marques (2014)).

In addition to introducing a LUC emission factor that accounts for ILUC, several alternative and/or complementary approaches have been proposed to reduce the risk of ILUC. A key option is a combination of promoting feedstocks that rely less on land and reducing the risk of unwanted LUC (Witcover et al. 2013). In Europe, this approach is suggested to be implemented by capping the contribution of food-based biofuels to the renewable energy mandate (e.g. in Europe such a cap is discussed for 5 -7% of final energy use in road transport, Marques (2014)). Capping food-based (or so-called 1st generation) biofuels is considered because they are generally thought to have higher LUC emissions than lignocellulosic crops. This is a result of lower yields and lower carbon stocks (Chum et al. 2011). However, this approach (as well as the LUC emission factor approach) fails to account for ILUC being the direct LUC of another activity. For example, in Brazil sugarcane production has expanded at the cost of other crop and livestock production, which in turn expanded by clearing rainforest in the Amazon (Andrade de Sá et al. 2013; Barona et al. 2010; Sparovek et al. 2007). Only if this other activity is also incentivized to minimize its LUC, can ILUC be tackled. Therefore, another strategy to reduce ILUC is to take a sustainable approach to all land use, whether for food or non-food production. This entails sustainably increasing productivity and resource efficiency of all crop and forestry production, and appropriate zoning of land for all purposes (including protection of some areas and identification of other areas for production). Implementing such an approach in practice would mean only allowing those biofuels that do not displace other uses and that do not convert high carbon stock land (van de Staaij et al. 2012; Wicke et al. 2015). Key strategies to making additional biofuels available with low risk of causing ILUC are i) increasing agricultural productivity by modernising and sustainably intensifying agriculture especially in the currently low yielding areas, ii) applying under-utilized land for additional food and bioenergy feedstock production, and iii) zoning of land that should not be converted for any application, whether food or non-food, and its strict enforcement (Wicke et al. 2015). Although this is not an easy task, only an integrated perspective that covers all applications, functions and services of land can actually tackle ILUC. In the European Council Conclusions on 2030 Climate and Energy Policy (European Council 2014), such an approach is already suggested: Two goals are the promotion of sustainable intensification of food production and optimization of the sector's GHG emissions. If achieved, the environmental performance of the agricultural sector as a whole can be significantly improved (including reducing its currently large carbon footprint) and total production for food and non-food purposes can be increased.

Developing abovementioned policy measures and actually implementing them will take time, and requires substantial efforts and large investments in not only the bioenergy and biomaterial sectors but also the agricultural and forestry sector as a whole. It also will require increased cooperation and strong commitment of different stakeholders from the government, industry, civil society and academia at different scales (from local to global). Only if these stakeholders work together can sustainable solutions to increasing demand for bioenergy and biomaterial be found and implemented, and large GHG emission reductions be achieved. While this can take a decade, in the meantime, it is important to continue to develop sustainable production of the most promising biomass feedstocks under stringent monitoring of GHG emissions and other sustainability criteria – with a chance of corrective action needed in the future as part of the ongoing learning process. This is because simply excluding biomass from the energy and material mix (and ensuing failure to sufficiently rapidly mobilize sustainable biomass resources for a biobased economy) may impose risks in itself; namely, the risk of inaction for climate change and all related impacts. But it also diverts the attention away from other, often more important causes of e.g. deforestation, food insecurity, loss of carbons sinks and biodiversity, which are now often attributed to bioenergy. However, a sustainable approach to all land use that covers and integrates the different uses, functions and services (proposed above for tackling ILUC) can also help address a number of these issues. This overview of the state-of-the-art research shows that a lot of progress has been made on identifying promising biomass supply chains and strategies to optimise the performances. Steps must now be taken to not only minimize potential negative impacts but also develop the opportunities of biomass production for modern energy and material purposes.

6 References

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