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GREENHOUSE GAS IMPACT OF BIOENERGY PATHWAYS

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Contents

Summary and key findings

In June 2014, the International Renewable Energy Agency (IRENA) released the first edition of 'REmap 2030: A Renewable Energy Roadmap'. This roadmap shows that modern bioenergy could represent 60% of the global renewable energy use in 2030, if the world is to achieve a doubling of its renewable energy share in total final energy consumption between 2010 and 2030. In REmap 2030, greenhouse gas emissions from renewable energy have not been considered, but obviously during the production process of renewable energy greenhouse gases are emitted. Bioenergy, especially, has a complex relationship with greenhouse gas emissions.

IRENA commissioned PBL Netherlands Environmental Assessment Agency to write a short technical background report on the greenhouse gas emission benefit and impacts of different bioenergy technology pathways, primarily but not only based on existing PBL material and references therein. This study is not aimed at providing a comprehensive overview of all pros and cons of bioenergy, but rather combining estimates of supply-chain emissions, direct and indirect land-use change emissions, and changes in carbon cycle dynamics, for various conventional and advanced bioenergy pathways. The result of this exercise is shown in Table S.1.

The study in hand finds that overall ranges of emission factors are wide, due to uncertainty about supply-chain emissions, ranges in land-use-change (LUC) emissions and ranges in carbon debts, and might even be wider if all uncertainties addressed in studies beyond what is covered here would have been included. Also, lower greenhouse gas emission factors than presented here are possible if strict policies would be implemented in the production of bioenergy. The results presented here in this report are largely based on the first edition of REmap 2030 and takes into account only minor revision of country results.

Supply-chain emissions

It is shown that supply-chain emissions of liquid biofuels and biomethane can be significant. Based on the life-cycle assessment (LCA) used in this study, greenhouse gas emissions could range from around 20 grams of carbon dioxide equivalent per megajoule (gCO_2eq/MJ) for liquid biofuels from woody crops (advanced biofuels) and biomethane from manure and organic waste up to almost 60 gCO_2eq/MJ for ethanol from wheat. There are four pathways that achieve significantly higher emission reductions per hectare than others: biomethane from woody crops, ethanol from sugar beets or sugar cane, and FAME or biodiesel from palm oil. The carbon impact for wood pellets is higher than for chips due to the additional energy consumption involved in drying, milling and pelletising, and ranges from 8 to 30 gCO_2eq/MJ. Large sources of uncertainty are the N_2O field emissions and the assumed yields of the woody crops.

Table S.1 *Emission factors based of different bioenergy pathways based on material presented in this study for power plants, liquid and gaseous transport biofuels and heat plants. Applied efficiencies are based on the TIMER model (Stehfest, Van Vuuren, Bouwman, and Kram, 2014) and (PBL, 2008).*

Bio-energy carrier	**From**		**Power**	**Transport**	**Heat**
			Units: grams CO_2 equivalent per MJ		
Crude vegetable oil	Palm with CH_4 capture		n/a	63 [45,74]	62 [40,81]
	Soya beans			85 [64,112]	88 [63,120]
	Sunflower seeds			71 [69,73]	71 [69,74]
	Rapeseed			89 [53,139]	93 [49,153]
Ethanol	Starch crops (gas CHP)		n/a	59 [44,90]	n/a
	Sugar cane			41 [27,70]	
	Sugar beet			38 [36,42]	
	Woody	Switchgrass		44 [33,66]	
		Miscanthus		27 [16,38]	
Bio-CH_4	Manure and Waste		46	20	25
Pellets	Fast growing	Agricultural land	40 to 110	25 to 65	20 to 60
		Marginal land	35 to 70	20 to 40	20 to 35
	Agro-residues	Crop harvest			
		Processing			
	Forest residues	Ref = Burning	20 to 45	10 to 25	10 to 25
		Ref = Decay	85 to 150	50 to 90	45 to 80
	Increase in	Thinning	180 to 300	110 to 180	95 to 160
		Felling	415 to 520	250 to 315	220 to 255
	Waste	Ref = Burn	20 to 45	10 to 35	10 to 30
		Ref = Landfill	-360 to 250	-210 to 150	-180 to 130
Fossil energy source		**Emission factor**			
Coal		93	195 (48%)	-	117 (79%)
Gas		56	98 (57%)	56 (100%)	65 (87%)
Oil		84	-	84 (100%)	102 (82%)

Land-use change emissions

Agricultural land expansion caused by the demand for biofuels can possibly lead to greenhouse gas emissions due to direct land-use change (DLUC) and indirect land us change (ILUC), where DLUC can be defined as 'the situations in which land use is changed from any previous use to bioenergy feedstock production itself' and ILUC as 'the change in land use outside a feedstock's production area that is induced by changing the use or production quantity of that feedstock'.

Calculations of DLUC emissions based on the RED methodology (EC, 2010) show that the conversion of forest land to bioenergy cropland emits large amounts of greenhouse gas (up

to 360 gCO_2eq/MJ). Conversion of grasslands shows a range from -74 gCO_2eq/MJ (palm oil in Indonesia, negative indicating a saving)) to +83 gCO_2eq/MJ (biodiesel from soya beans in Brazil, positive indicating additional emissions). Other feedstocks that sequester significant amounts of carbon when converted from grasslands are switchgrass, miscanthus, sugar cane, Jathropha and forest plantations. The level of uncertainty is high for DLUC emissions based on the presented method (EC, 2010) using default carbon stock values for soil and vegetation. Therefore, if available, real-world data should be used.
DLUC emissions are just part of the effect as the demand for biofuels products often leads to ILUC as well. The extent to which indirect effects occur depends on many economic factors (e.g. yield increase, consumption changes, availability of the feedstocks, and input prices).
According to a number of recent studies, uncertainty in overall LUC emissions is high. Based on the economic studies examined various types of conventional bioethanol have a LUC factor of approximately 20 gCO_2eq/MJ (computed as the mean of the averages in the studies considered), with a range of 3 to 61 gCO_2eq/MJ and conventional biodiesel around 35 gCO_2eq/MJ with a range of 7 to 94 gCO_2eq/MJ. For palm oil biodiesel and biodiesel in general the use of peatland in Malaysia and Indonesia play an important role in the greenhouse gas effects.
Direct or indirect conversion of forest should be avoided since these will lead to high emissions, using a 30 years allocation period for land-use emissions. This plays a relatively larger role for biodiesel.
Perennials have the potential to have relatively lower LUC factors since they have higher living biomass carbon and higher soil organic matter carbon. Harvest residues have the potential to have LUC factors close to 0. Using marginal land – land not used for any economic purpose, such as agriculture, forestry, or other uses, now or in the scenario period – result in low LUC emissions, but there is often a reason that it is not used, for example low fertility or limited accessibility.

Forestry
Wood taken from forests is a carbon-neutral energy source in the long term, but it takes time before net emission reductions are actually achieved. This is called the carbon debt which is defined as the carbon emitted due to harvesting the bioenergy (including e.g. residues) minus the carbon that would be emitted by the alternative system (mostly fossil energy).
Using harvesting residues for bioenergy production results in a relatively small carbon debt and carbon payback times. Likewise, using processing and post-consumer waste can have a carbon debt that can be very small in some cases, but this is strongly dependent on the reference situation (e.g. landfills with or without methane capture). Wood plantations on agricultural land have very low payback times because of the uptake of CO_2 in the years before the wood is harvested. However, it requires land and therefore LUC emissions have to be taken into account.
Using wood from thinning in boreal and temperate forests for bioenergy could produce a significant carbon debt and payback times between 40 and 135 years, when used for replacing coal in power generation. Thinning in forest plantations may have much shorter

payback times. Additional felling for bioenergy in boreal and temperate forests could result in large carbon debts, requiring payback times of decades, up to more than three centuries. For the short term, an efficient climate mitigation measure would be to refrain from additional final felling for the purpose of bioenergy. In that way more carbon would remain stored in forests and an effective carbon sink would remain intact.

Bioenergy demand and supply

The bioenergy demand and potential supply estimates underpinning the REmap 2030 study fall within the ranges published by the Intergovernmental Panel on Climate Change (IPCC), PBL and others at the level of global totals. Despite this, the REmap projections must be considered ambitious and attainable only under favourable conditions and strong policies mainly because the ranges published in other studies are compiled for a more distant future (2050 instead of 2030 in REmap). While global totals seem to be in reasonable agreement, the underlying details per supply category, at country level, are ambitious for some cases. For example, in REmap, some countries seem to be large exporters of biofuels, others are presented as large importers, while this is not always consistent with current trade patterns, historical agricultural land expansion since 1980 and current land-use policies.

Costs, strategies and policy directions

As indicated, large-scale bioenergy deployment is an important contributor to reaching ambitious climate change targets. Bioenergy options can deliver net cost benefits compared to fossil fuel alternatives, and more so if greenhouse gas emission reductions are valued in monetary terms. However, from the global perspective, net benefits are lower when compared to the avoided emissions of the replaced fossil fuels. Cost implications at smaller scales and from different stakeholder perspectives can vary enormously from the global perspective. Firstly because net emissions reductions differ under varying system boundaries. And secondly, because the unit price of emissions depend on specific rules and regulations for countries and sectors.

Negative impacts of ambitious bioenergy schemes on natural ecosystems may be reduced significantly by simultaneous introduction of measures to keep land conversion in check.

In particular schemes to protect forest areas can be instrumental to limit net land-use change and related greenhouse gas emissions, leading to beneficial effects for nature protection and biodiversity conservation of highly valued forest areas. Introducing land protection policies could bring about costs, for consumers in the form of higher agricultural commodity prices. Furthermore policies should be developed that are aimed at 1) increasing (biomass) productivity of land-use systems in general and in particular in the agriculture and forestry sectors, 2) increasing the carbon stock of land in both biomass and the soil, and 3) better use of waste products and improved efficiency in the supply chain.

Finally, sustainability criteria need to be developed carefully with wide consultation and good systems understanding about the natural and anthropogenic processes and their interactions. That will be essential to ensure that proposed measures have the desired consequences.

1 Introduction

Biomass stores carbon dioxide (CO_2) from the atmosphere that is released once the biomass decomposes. When biomass is combusted instead of decayed, carbon is released into the atmosphere instantaneously. If the total biogenic carbon released during biomass decay and/or combustion is sequestered again by regrowing the same amount of biomass, the system continues to be in balance. As a result, the amount of CO_2 in the atmosphere does not increase.

Biomass carbon can also be stored for long period in standing aboveground plants or in roots and as soil carbon. The carbon cycle could, however, change in different ways when large amounts of bioenergy are used as fuel. With increasing bioenergy use, the carbon stored in living plants and soil may also change, but the dynamics of soil carbon are not well understood.

Growing energy crops require land. This can be existing agricultural land. Non-agricultural land such as forest or pasture land could be converted to grow energy crops as well. It is common to distinguish between direct and indirect land-use changes. Direct land-use change (DLUC) involves changes in land use on the site used for bioenergy feedstock production. Indirect land-use change (ILUC) refers to the changes in land use that take place elsewhere as a consequence of the bioenergy project. Especially the latter is a major source of uncertainty since it involves both biophysical and socio-economic factors.

In June 2014, the International Renewable Energy Agency (IRENA) released the first edition of 'REmap 2030: A Renewable Energy Roadmap' (IRENA, 2014), hereafter referred to as 'REmap 2030'. This roadmap shows that modern bioenergy could represent 60% of the global renewable energy use in 2030, if the world is to achieve a doubling of its renewable energy share in total final energy consumption between 2010 and 2030. In REmap 2030, greenhouse gas (greenhouse gas) emissions from renewable energy have not been considered but, obviously, during the production process of renewable energy, greenhouse gases are emitted. Especially, bioenergy has a complex relationship with greenhouse gas emissions.

Numerous studies have looked into the assessment of the greenhouse gas impacts of bioenergy. However, the results show great divergence and uncertainty. These differences arise from the fact that there are a number of sources of greenhouse gas emissions in the bioenergy life cycle, and depending on how these are accounted for, total emissions significantly differ. Another reason for these ranges is that carbon footprints of bioenergy pathways are in most cases site or case specific.

IRENA asked PBL Netherlands Environmental Assessment Agency to write a short technical background report to IRENA's second edition of REmap 2030, on the greenhouse gas (greenhouse gas) benefit and impacts of different bioenergy technology pathways, primarily but not only based on existing PBL material and references therein, i.e. without performing new analysis or research. In other words, this request was *not* meant to write a comprehensive overview of all pros and cons of bioenergy, but just a first and quick attempt to combine estimates of supply-chain emissions, direct and indirect land-use change emissions, and changes in carbon cycle dynamics for different conventional and advanced bioenergy pathways. Where relevant limitations of the methodology are highlighted and briefly discussed.

This report shows that bioenergy-related greenhouse gas emissions can be significant and in some cases higher than the displaced fossil fuels. However, this does not lead to the conclusion that bioenergy would be last on the list of greenhouse gas mitigation options. In fact, many PBL studies show that a significant contribution from bioenergy is key to limit global warming to two degrees Celsius by 2100 (Vuuren, Bellevrat, Kitous, and Isaac, 2010), where the use of bioenergy in combination with Carbon Capture and Storage plays a crucial role, and biomass potential dominates the cost of reaching this target.

2 Greenhouse gas impacts of bioenergy pathways

Key messages:

- greenhouse gas-impacts of liquid biofuels and biomethane can be significant. Based on the life-cycle assessment used in this study, from around 20 gCO_2eq/MJ for liquid biofuels from woody crops (advanced biofuels) and biomethane from manure and organic waste up to almost 60 gCO_2eq/MJ for ethanol from wheat.
- There are four pathways that achieve significantly higher emission reductions per hectare than others: biomethane from woody crops, ethanol from sugar beets or sugar cane, and FAME or biodiesel from palm oil.
- The carbon impact for wood pellets is higher than for chips due to the additional energy consumption involved in drying, milling and pelletising, and ranges from 8 to 30 gCO_2eq/MJ depending on the type of wood used to produce the pellets, its country of origin, and method for drying the wood and pre-pelletisation.
- Large sources of uncertainty are the N_2O field emissions and the assumed yields of the woody crops.

2.1 Introduction

This chapter discusses what emissions, in terms of CO_2 equivalents, are involved in the feedstock-to-product or supply chains of different forms of bioenergy. Section 2.2 discusses emission factors that are related to the production of conventional and advanced liquid biofuels that are mainly used in the transport sector and biomethane. Section 2.2.5 includes a short discussion on uncertainty ranges in supply-chain emissions of liquid biofuels. Section 2.3 briefly describes the supply-chain emissions in woody source categories (e.g. residues, woody crops, and forests plantations) that can be used as solid fuel in power or heat production (chips and pellets) or as fuelwood. In this chapter, no emissions with respect to direct or indirect land-use change (LUC) are taken into account. This issue is covered in Chapter 3.

2.2 Liquid biofuels and biomethane

In general, the supply chain of liquid biofuels can be divided into three main components:

1. cultivation of the feedstock,
2. processing the feedstock into the biofuel, and
3. transport of the feedstock to the production site and transport from the production site to the end user.

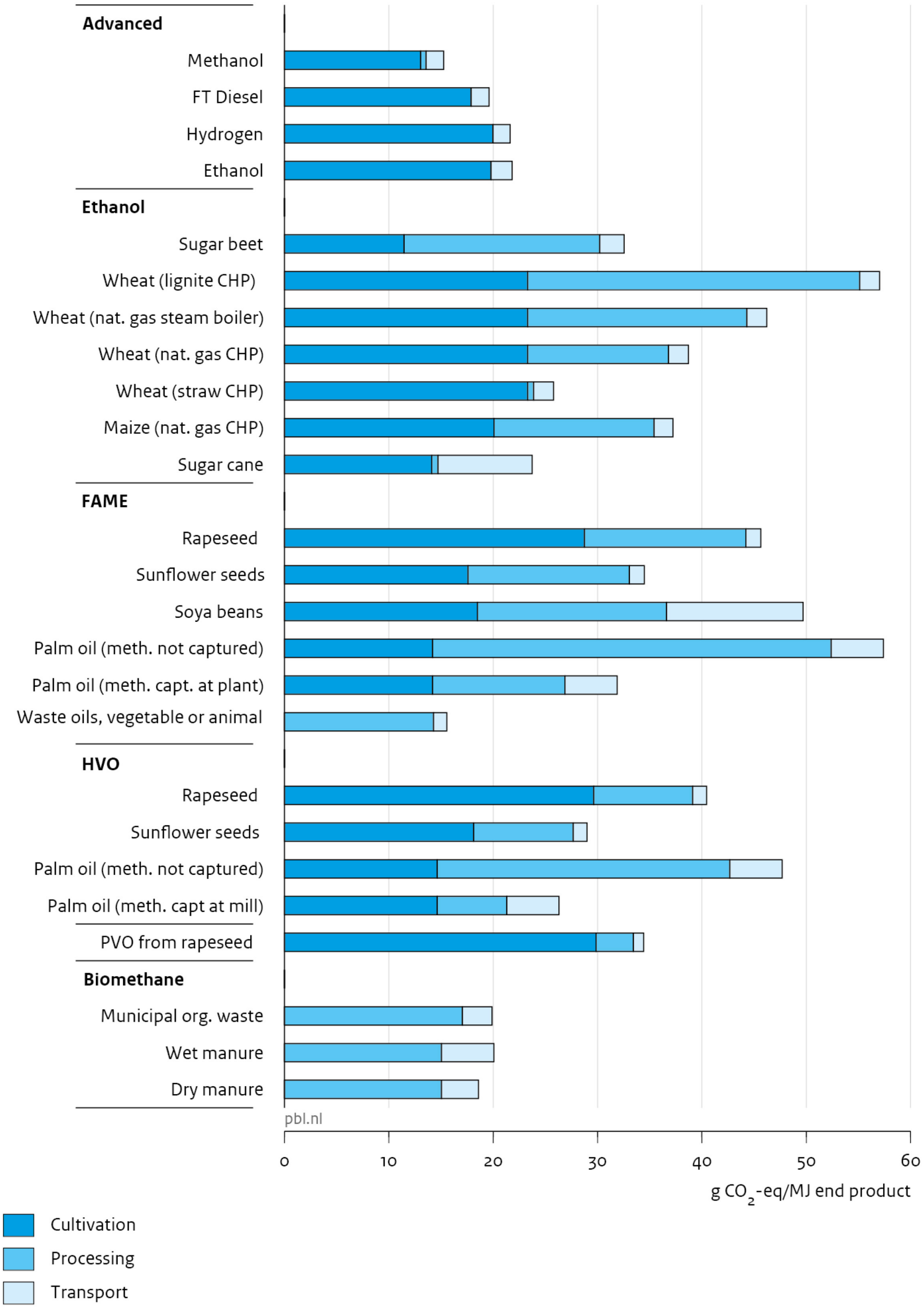
Figure 2.1 shows the emission factors – i.e. the emissions of greenhouse gases per MJ of biofuel – for these main components of the supply chain for 25 different liquid biofuel and biomethane pathways. In this figure, six categories are distinguished:

1. advanced liquid biofuels made from lignocellulosic biomass through gasification (methanol, FT Diesel and hydrogen) or through hydrolysis and fermentation (ethanol);
2. ethanol made from sugar and starch through fermentation;
3. Fatty Acid Methyl Esters (FAME) made from vegetable oils and vegetable and animal waste oils through esterification;
4. Hydro-treated Vegetable Oil (HVO) or hydro-biodiesel from vegetable oils through hydro-treatment;
5. Pure Vegetable Oil (PVO) from vegetable oils (only rapeseed in this study);
6. Compressed biomethane from manure and organic waste through fermentation;

The emission factors of categories 2 through 6 are based on the most recent version 4b of BioGrace (JRC, EUCAR, and CONCAWE, 2015). This is a tool that allows reproduction of the detailed calculation method of computing CO_2 equivalent emissions of the production chain of liquid biofuels and biomethane as described in Annex V of the European Renewable Energy Directive (EC, 2009). The emission factors of category 1, the advanced liquid biofuels, are based on a BioGrace-like spreadsheet that has been developed at PBL (PBL, 2008), based on IMAGE data (Stehfest et al., 2014) and following the methodology as described in (Hamelinck and Hoogwijk, 2007). Obviously, these sources do not cover the full range of supply-chain emission factors that can be found in the literature since the results are site and case specific due to, for example, technological differences. Or, to put in another way, you will hardly find two oil mills or biodiesel plants with the same energy or mass balances. However, in this short study, we chose BioGrace which it is a detailed data source that provides a consistent and recent overview of all steps of the supply chain of a large set of conventional liquid biofuels and that serves as a starting point to compute emissions in the context of an important policy directive: the RED.

FAME can be blended with fossil diesel fuel up to a certain percentage. Due to their reactive properties with metals (oxidation) and rubber, most engines do not allow for percentages of FAME above 20%. HVO or Hydro-treated Vegetable Oil or hydro-biodiesel is a high quality fuel that can be used instead of fossil diesel without beneficial effects or even damage to the diesel engine (Hartikka, Kuronen, and Kiiski, 2012). However, its production process is significantly more expensive than that of FAME because it involves treatment with explosive hydrogen (H_2).

Supply chain emission factors of liquid biofuels



Source: Harmelinck and Hoogwijk, 2007 & JRC et al, 2015

Figure 2.1

Supply-chain emission factors for liquid biofuels based on (Hamelinck and Hoogwijk, 2007) for the advanced biofuels and based on BioGrace (JRC et al., 2015) for the others, taking co-products into account. Text between parentheses in ethanol category refers to energy source for processing. FAME=Fatty Acid Methyl Esters or Biodiesel, HVO=Hydro-treated Vegetable Oil or Hydro-biodiesel, CHP=Combined Heat Power, PVO=Pure Vegetable Oil.

In computing the emission factors, it has been taken into account that in the production chain of conventional biofuels, co-products are produced that can be used as animal feed (e.g. soya or rapeseed cake) or in the chemical industry (refined glycerol for soap). Therefore, part of the emissions in the supply chain can be assigned to these co-products since they replace the production of animal feed or glycerol that would otherwise be produced elsewhere. Table 2.1 reflects the fractions in energy terms of the crops or intermediate products that end up in a co-product according to BioGrace, which go from 0% for sugar cane (no co-product) to 67% for soya beans.

Table 2.1 *Allocation factors and co-products (DDGS= Dried Distillers Grains with Solubles). Source: Biograce (JRC et al., 2015).*

Crop	**(Intermediate) Product**	**Fraction to co-product**	**Co-product**
Sugar beet	Ethanol	29%	Sugar beet Pulp
Wheat[1]	Ethanol	40%	DDGS
Maize	Ethanol	45%	DDGS
Sugar cane[2]	Ethanol	0%	None
Rapeseed	Crude vegetable oil	41%	Rapeseed cake
Sunflower seeds	Crude vegetable oil	37%	Sunflower seed cake
Soya beans	Crude vegetable oil	67%	Soya cake
Palm oil	Crude vegetable oil	5%	Kernel Meal
Crude vegetable oil	FAME or biodiesel	4%	Glycerol

[1] Wheat also produces straw, but no emissions are assigned to it.

[2] Electricity and heat used in the production of ethanol from sugar cane are produced by CHP from bagasse and other residues, implying no emissions.

Emission factors in Figure 2.1 show a wide range from advanced methanol (15 gCO_2eq/MJ) to FAME from palm oil without methane capture (57 gCO_2eq/MJ):

- The lowest emission factors are biomethane and FAME from waste products because these pathways do not include a cultivation phase. However, the supply of waste products is limited, also because especially in Western countries these waste products are often burnt to produce electricity.
- Ethanol from sugar cane, which has a large share in the total ethanol production, has a relatively low emission factor because in the processing step it is assumed that electricity and heat used for the production of ethanol are produced by CHP from bagasse implying no emissions (see Section 2.2.2). Transport emissions on other hand are relatively high because in the Renewable Energy Directive (RED) it is assumed that ethanol from sugar cane has to be shipped over long distances (see Section 2.2.3) to Europe. So for other regions, the overall emission factor would be even lower (see Section 2.2.5).

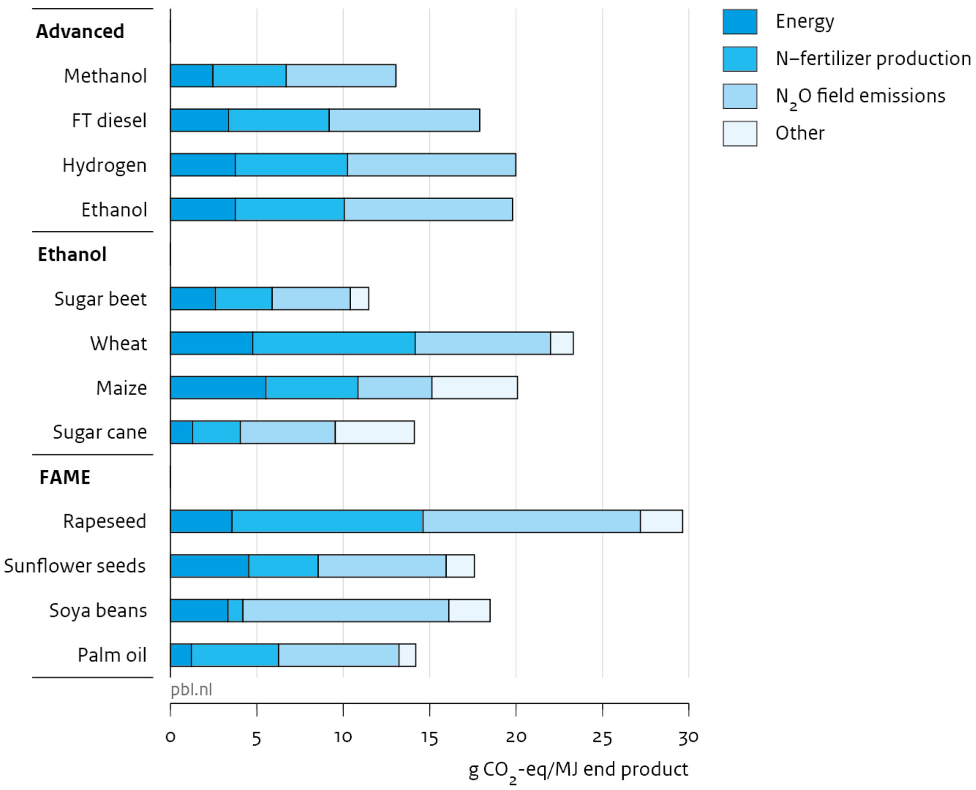
- HVO and FAME from palm oil have high emission factors of methane, which is produced in large amounts in the processing phase (see Section 2.2.2), is not captured. However, these emissions can be captured at the oil mill relatively easy resulting in lower emission factors.
- In the supply chains of ethanol from wheat, there are large differences in the processing phase (see Section 2.2.2) based on the fuel used in the ethanol plant. If lignite is used in the CHP boiler, it results in high processing emissions.
- Advanced liquid biofuels have relatively low emission factors, where 80% to 90% is assigned to cultivation emissions (mainly the use of N fertilisers, see Section 2.2.1). The fossil energy consumption in the conversion step (or the processing phase) is limited, since most of the conversion energy is derived from the biomass itself. However, in terms of emission reductions per hectare (see Section 2.2.4), their performance is comparable to – or even slightly worse than – biofuels from sugar crops and palm oil.

2.2.1 Cultivation emissions

The variables that determine cultivation emissions are the production of chemical N fertilisers, field emissions of N_2O, energy use (e.g. diesel for tractors), and a fraction allocated to manure (if applied), and the production of other chemical fertilisers, pesticides, and seeds and in case of sugar cane it also contains CH_4 emissions due to trash burning (see Figure 2.2). Note that the cultivation emissions for FAME in Figure 2.2 also apply to HVO. For advanced biofuels (based on woody crops or short rotation plantations), the emissions from cultivation cover 80% to 90% of the total emissions of the supply chain because, as indicated, the energy consumption in the conversion step is from the biomass itself. In the production of methanol, FT diesel and Hydrogen this energy is generated during the gasification process.

The use of fertilisers implies CO_2, N_2O and CH_4 emissions in the production phase and N_2O emissions when applied in the field. The values, as used in BioGrace and in (PBL, 2008) for wood, for the application of N and other fertilisers and pesticides are shown in Table 2.2a. For calculating the field emissions, the DNDC model (Gilhespy et al., 2014) was used for European crops and the IPCC Tier 1 for non-European crops.

Emissions from cultivation of liquid biofuels



Source: Harmelinck and Hoogwijk, 2007 & JRC et al, 2015

Figure 2.2

Emissions due to cultivation of major biofuel crops (allocated results, based on Table 2.1). The category 'other' refers to N_2O emissions from manure (if applied), emissions related to the production non-N chemical fertilisers (see Table 2.2a), pesticides, and seeds and in case of sugar cane it also contains CH_4 emissions due to trash burning. The cultivation emissions of FAME also apply to HVO.

Table 2.2a *Use of fertilisers and pesticides. Source: BioGrace (JRC et al., 2015) and (PBL, 2008).*

Crop	**N**	**CaO**	**K_2O**	**P_2O_5**	**Pesticides**
Sugar beet	120	400	135	60	1.3
Wheat	109	-	16	22	2.3
Maize	52	1600	26	35	2.4
Sugar Cane	63	367	74	28	2.0
Rapeseed	137	19	49	34	1.2
Sunflower seeds	39	-	22	30	2.0
Soya beans	8	-	62	66	2.7
Palm Oil	128		200	144	8.4
Woody	60	-	-	35	-

2.2.2 Processing emissions

Ethanol and methanol

Processing emissions in ethanol and methanol production are shown in Figure 2.3. The negative electricity emissions occur when a CHP ethanol plant produces more electricity than needed in the production process. The emissions that would otherwise be emitted by a power plant are reported as 'negative' emissions in BioGrace. In the case of sugar cane, it is assumed in BioGrace that electricity and heat used in the production of ethanol are produced by CHP on bagasse, implying no emissions. The chemicals used in sugar cane production mainly are quicklime (CaO) and sulphuric acid (H_2SO_4) used in the sugar making process.

For advanced biofuels, we assume that the fossil energy consumption in the conversion step is limited or zero, since the conversion energy, or most of it, is generated by the biomass itself (see (PBL, 2008) and (Hamelinck and Hoogwijk, 2007)).

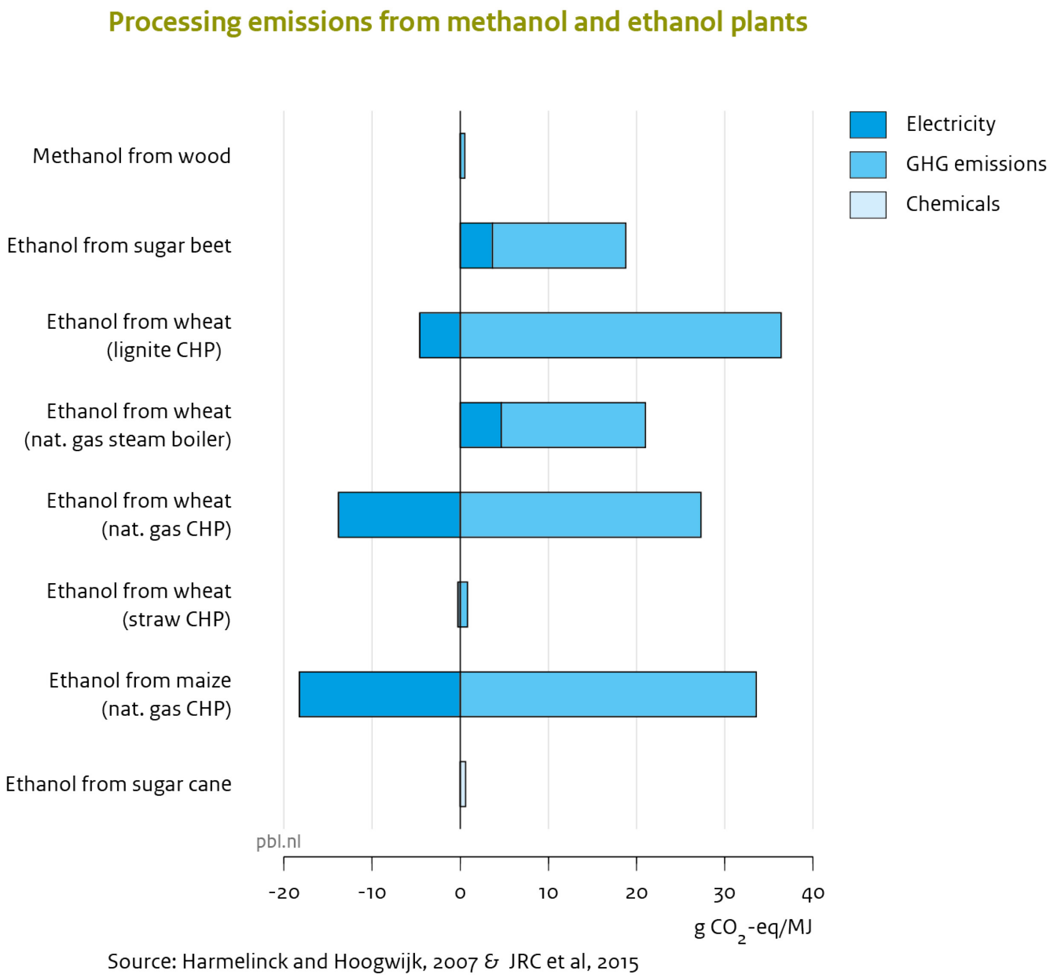


Figure 2.3 *Processing emissions (m)ethanol plant. In case of CHP, more electricity can be produced than needed in the processing phase, resulting in 'negative' emissions. greenhouse gas emissions refer to CH_4, N_2O and CO_2 emissions from the burning of natural gas or lignite in the processing phase. The sum of negative and positive emissions result in overall processing emissions as presented in Figure 2.1. Source: (JRC et al., 2015).*

FAME

Processing emissions in FAME production (biodiesel) can be divided in three steps: extraction, refining and esterification (see Table 2.2b).

Table 2.2b *Processing emissions in FAME production. Source: Biograce (JRC et al., 2015).*

	Extraction	**Refining**	**Esterification**
	gCO_2eq/MJ end product		
Rape seed	2.8	0.7	12
Sunflower seeds	2.8		
Soya beans	5.4		
Palm Oil (CH_4 capt)	0		
Palm Oil	21		17
Waste oil	0		14

In the extraction phase of palm oil it is assumed that the demand of heat and electricity is met by combusting biomass residues that are locally available implying no emissions. However, at the palm oil mill large quantities of palm oil mill effluent (POME) are generated, mainly from the sterilisation and clarification processes of the palm oil mill. If POME is treated in open ponds, where anaerobic digestion takes place automatically due to the high organic content, large quantities of the greenhouse gas methane are emitted; about 1 g of methane or 25 gCO_2eq/MJ FAME (Solomon et al., 2007). This methane can be captured relatively easily and more and more palm oil mills are equipped with gas tight tanks resulting in much lower greenhouse gas emissions.

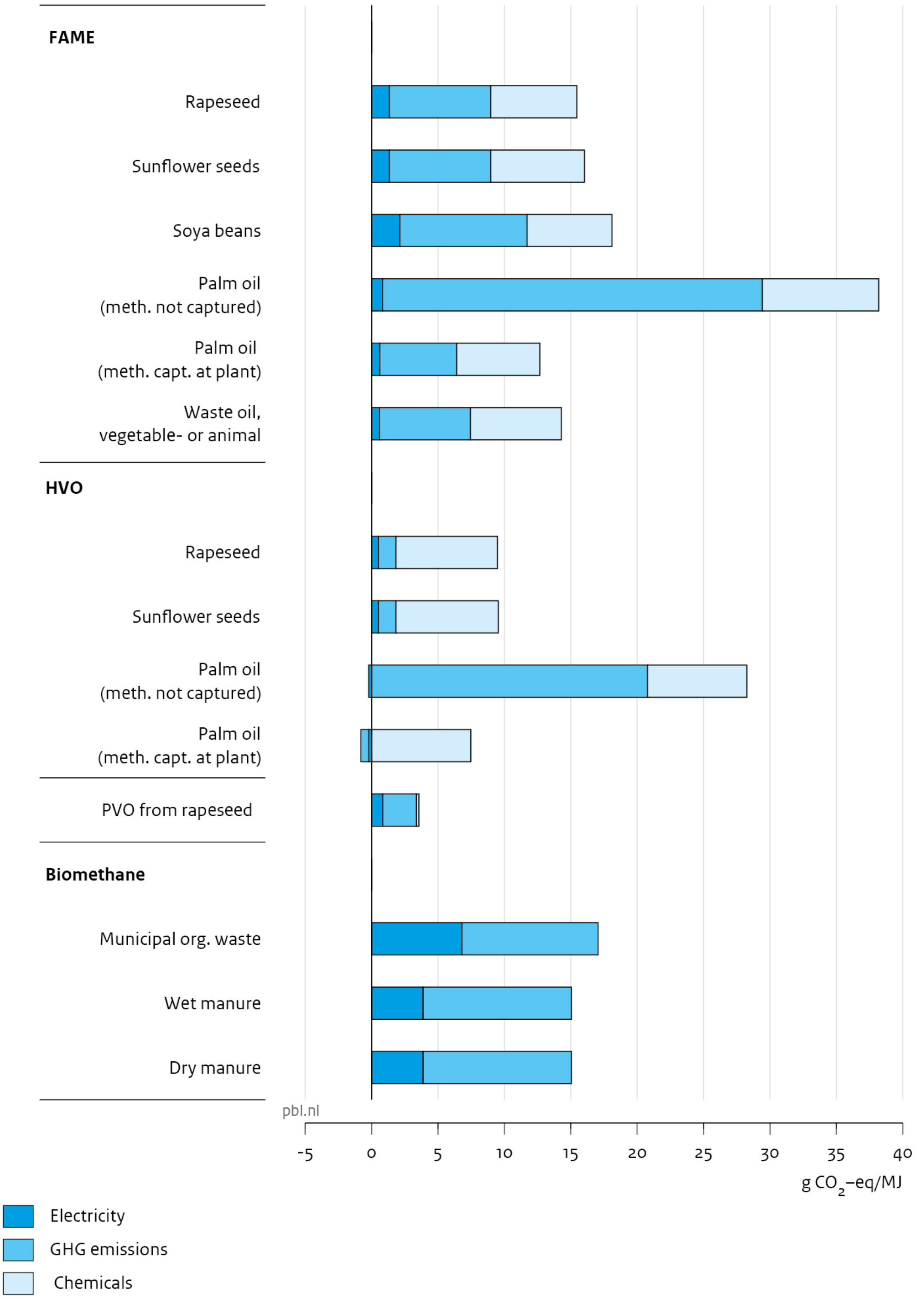
In Figure 2.4 emissions of the FAME processing phase are subdivided in emissions from electricity use, chemicals and CH_4, N_2O and CO_2 emissions from the burning of natural gas.

Chemicals in FAME production are mainly used during esterification, i.e. phosphoric acid (H_3PO_4), hydrochloric acid (HCl), sodium carbonate (Na_2CO_3), sodium hydroxide (NaOH) and methanol (MeOH). When producing PVO (Pure Vegetable Oil) from rapeseed, no esterification process is involved and therefore emissions are lower.

HVO

In case of HVO (hydro-biodiesel) the processing phase can be divided in the extraction and the hydrogenation of the vegetable oil. The extracting phase is the same as for normal biodiesel (FAME). The hydrogenation phase produces electricity. In some cases electricity and steam production is (slightly) higher than needed in the production process. Net electricity production results in (small) 'negative' energy emissions and net steam production in negative greenhouse gas emissions that reflect the avoidance of N_2O and CH_4 emissions. The emissions in hydrogenation are related to the use of hydrogen (H_2) that needs energy in its production process resulting in CO_2 emissions. In Figure 2.4, the chemicals-related emissions in HVO production mainly reflect the input of H_2 during hydrogenation.

Processing emissions from FAME, HVO, PVO and biomethane



Source: Harmelinck and Hoogwijk, 2007 & JRC et al, 2015

Figure 2.4 *Processing emissions PVO, HVO, FAME and compressed biomethane. For advanced biofuels, fossil energy consumption in the conversion step is close to zero and therefore not shown. Negative emissions refer to the net production of electricity or steam. greenhouse gas emissions refer to CH_4, N_2O and CO_2 emissions from the burning of natural gas in the processing phase. Details can be found in (JRC et al., 2015).*

Compressed biomethane

The processing of biomethane involves two steps: biogas generation from fermentation followed by methane extraction via pressurised water scrubbing. Almost all greenhouse gas emissions in both steps are related to leakage of methane.

2.2.3 Transport emissions

For most pathways , the contribution of transport to the total greenhouse gas balance is small. However, this is not the case for conventional biofuels from sugar cane, soya beans and, to a lesser extent palm oil, as is shown in Figure 2.1. This is largely because these goods have to be transported over more than 10,000 km by ship, from tropical regions to Europe, although the amount in emissions per kilometre in international transport is small, because sea ships have very large cargo capacities. Sometimes, greenhouse gas emissions from local transport of feedstock to ports or central conversion facilities contribute significantly, when inefficient trucks are used over long distances, or when the biomass is very wet or only partially useful.

Table 2.3 *Average transport distances (in km) for Europe for different biofuel pathways as used in BioGrace. Sources: (JRC et al., 2015) and (PBL, 2008).*

<table>
<tr><th>Pathway</th><th>Feedstock to plant</th><th>Fuel to port</th><th>Port to port</th><th>To depot</th><th>To filling station</th></tr>
<tr><td>Compressed biomethane from municipal organic waste</td><td>0</td><td rowspan="3">0</td><td rowspan="3">0</td><td rowspan="3">0</td><td rowspan="3">10</td></tr>
<tr><td>Compressed biomethane from dry manure</td><td rowspan="2">10</td></tr>
<tr><td>Compressed biomethane from wet manure</td></tr>
<tr><td>FAME or HVO from palm oil</td><td rowspan="2">20</td><td>150</td><td rowspan="3">10,000</td><td rowspan="8">150</td><td rowspan="8">150</td></tr>
<tr><td>Ethanol from sugar cane</td><td rowspan="2">700</td></tr>
<tr><td>FAME from soya beans</td><td rowspan="2">30</td></tr>
<tr><td>Ethanol from sugar beet</td><td rowspan="6">0</td><td rowspan="6">0</td></tr>
<tr><td>Ethanol from wheat</td><td rowspan="4">50</td></tr>
<tr><td>Ethanol from maize</td></tr>
<tr><td>FAME, HVO or PVO from rapeseed</td></tr>
<tr><td>FAME or HVO from sunflower seeds</td></tr>
<tr><td>Ethanol, Methanol, FT Diesel or Hydrogen from wood</td><td>260</td><td>0</td><td>450</td></tr>
</table>

In BioGrace, transport-related greenhouse gases are emitted during different phases of the production chain (see Table 2.3), i.e. when the feedstock is transported by truck from the field to the production plant, when ethanol and biodiesel are transported from the production plant to the storage depot, and from the depot to a filling station. In case of sugar cane and palm oil, the biofuels are transported by ship over long distances to Europe and other continents. Soya beans are shipped as raw biomass to the destination countries where they are converted into biodiesel (or used as animal feed). Emissions related to the power used at depots and filling stations are also counted as transport emissions. Compressed biomethane from manure and organic waste is assumed to be transported through pipelines, thus consuming very little energy. However, energy consumption at filling stations is relatively high.

In BioGrace, transport emissions in Europe are based on average distances along the production chain (see Table 2.3). In general, these distances also apply to other regions, except when sea transport is involved (columns 'Fuel to port and 'Port to port'). For example, if sugar cane is used in the producing region (Brazil) there would be no emissions from shipping. In the case of advanced biofuels, default transport distances are used of 260 km from the plantation to the processing site followed by 450km to the filling station. If these biofuels are shipped overseas, emissions related to sea transport should be taken into account as well.

2.2.4 Supply-chain greenhouse gas emission reduction per hectare

Another important indicator of the impact of biofuel production is their land use. Figure 2.5 reflects the supply-chain greenhouse gas emission reduction per hectare, computed as:

$$ER = Yield(Petrol - EF)(gCO2eq/ha) \quad [2.1]$$

Where *ER* is the Emission Reduction per hectare, *Petrol* is the emission factor of petrol (=84 gCO_2/MJ), *EF* is the emission factor as presented in Figure 2.1 and *Yield* refers to the Total Yield in Table 2.4, i.e. the sum of the biofuel and the co-products yields.

The figure clearly shows that establishing new forests on marginal agricultural land is effective under nearly all circumstances, if this is not causing any ILUC effects (see Chapter 3). For example, willow wood has a payback time of only a few years when grown on marginal lands and not causing any indirect land-use change. This is because of the high production level and short rotation cycles, e.g. (Elbersen et al., 2013; Tsarev, 2005).

The figure clearly shows that establishing new forests on marginal agricultural land is effective under nearly all circumstances, if this is not causing any ILUC effects (see Chapter 3). For example, willow wood has a payback time of only a few years when grown on marginal lands and not causing any indirect land-use change. This is because of the high production level and short rotation cycles, e.g. (Elbersen et al., 2013; Tsarev, 2005).

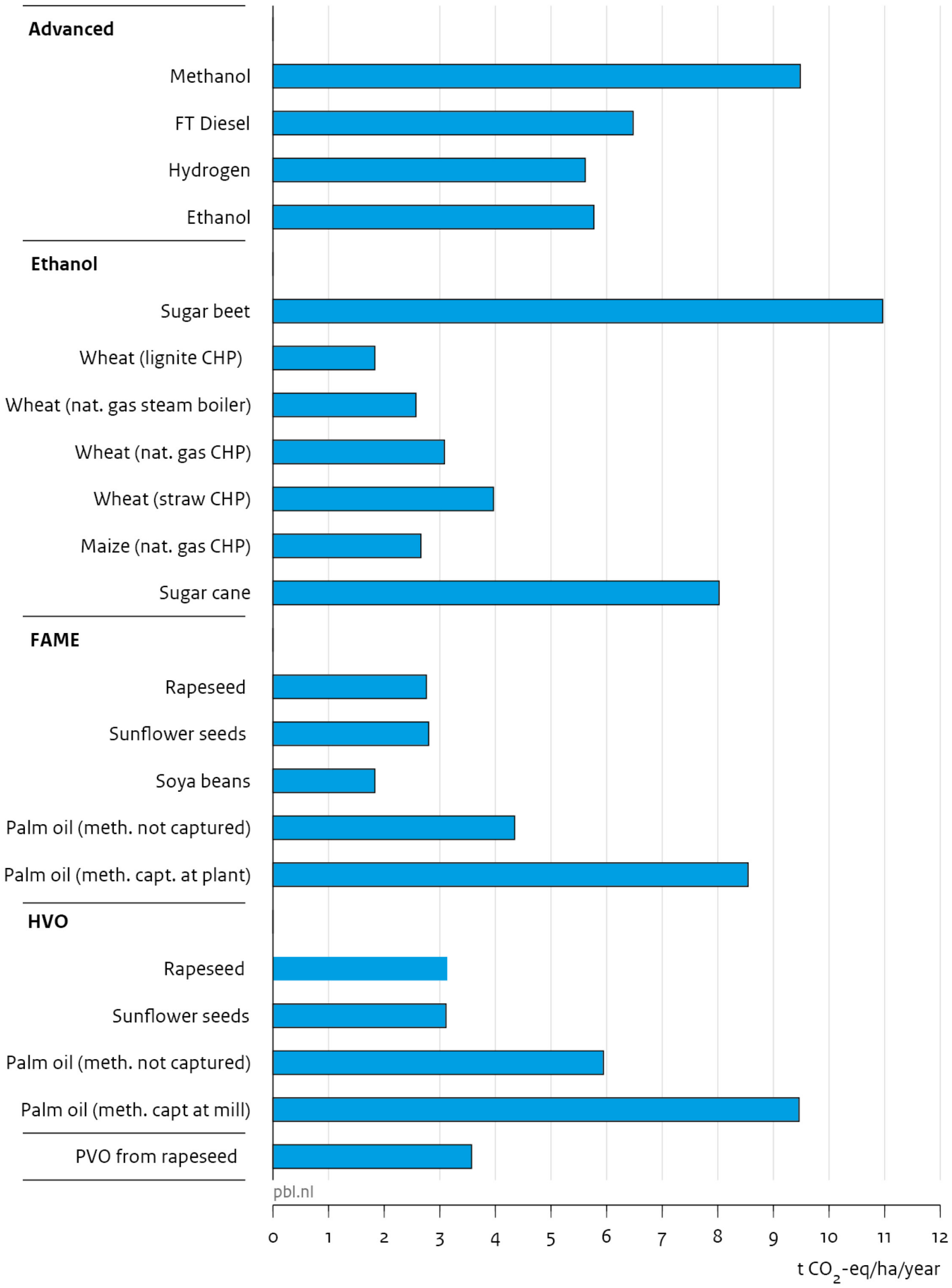
Table 2.4 *World average crop yields and Highest regional yields based on (FAO, 2013), where USA=United States of America, WEu=Western Europe, CAm=Central America, EAsia=Eastern Asia and SEAsia=Southeast Asia. Raw yields, Liquid biofuel yields and co-products yields are based on BioGrace and (PBL, 2008). Total yield is the sum of liquid biofuel yield and co-product yield. Efficiency is defined as the liquid biofuel yield divided by the raw yield.*

Crop	**World average raw yield** (t/ha)	**Highest regional average raw yield** (t/ha)	**Raw yield in this study** (t/ha)	**Raw yield in this study** (GJ/ha)	**Efficiency** (%)	**Liquid biofuel**	**Liquid biofuel yield**[a] (GJ/ha)	**Co-products yield** (GJ/ha)	**Total yield** (GJ/ha)
Sugar beet	56.2	87.0 (USA)	68.9	281	54	Ethanol	153	61	214
Wheat	3.3	7.5 (WEu)	5.2	77	53	Ethanol	41	28	68
Maize	5.5	10.0 (USA)	3.9	61	51	Ethanol	31	26	57
Sugar Cane	70.7	82.9 (CAm)	68.7	370	36	Ethanol	134	0	134
Rapeseed	2.0	3.5 (WEu)	3.1	74	58	FAME,HVO,PVO	43[a]	29	72
Sunflower seeds	1.7	2.6 (EAsia)	2.4	58	63	FAME,HVO	36[a]	21	57
Soya beans	2.5	2.9 (USA)	2.8	56	32	FAME	18	35	53
Palm oil	14.7	18.9(SEAsia)	19.0	301	50	FAME,HVO	150[a]	14	164
Woody crops	6 to 15		15.3	144	90	Methanol	138	0	138
					65	Ethanol	93	0	93
					70	FT Diesel	101	0	101
					63	Hydrogen	90	0	90

[a] Difference between FAME, HVO and PVO yields is less than 2%. Numbers refer to the average liquid biofuel yield.

There are five pathways that achieve emission reductions per hectare that are significantly higher than all the others; HVO and FAME made from palm oil, ethanol from sugar cane and sugar beet, and methanol - and to a lesser also ethanol, FT-Diesel and hydrogen - from woody crops. This is because these crops have, (much) higher liquid biofuel yields than the other crops (Table 2.4). For most crops, BioGrace yields are higher than the world average as shown in the second column of Table 2.4. This is because it is assumed that expansion of bioenergy production will use more modern techniques than used on average worldwide. The energetic crop yield (in GJ/ha) of sugar cane (third column Table 2.4) is by far the highest, but because of its low supply-chain efficiency of 36% – mainly due to the co-production of bagasse that is being used in the ethanol production phase – the liquid biofuel yield (ethanol) is lower than that of palm oil and sugar beet.

Emission Reduction per hectare



Source: Harmelinck and Hoogwijk, 2007 & JRC et al, 2015

Figure 2.5 *Emission reduction per hectare for different biofuels.*

2.2.5 Uncertainties

The previous sections have provided insight in the different elements of supply-chain emission factors and the relative importance of these elements based on BioGrace. However, we acknowledge there are studies indicating that the efficiencies (energy use, chemical use, oil yields) for first generation biofuels are different to those assumed in Biograce. This is even more relevant for second generation fuels where almost no actual data exists. More specific, important sources of uncertainty are:

- **Yields.** The emission factors are based on the yields presented in Table 2.4. In the real world yields differ on a spatial and temporal scale. If all other factors remain equal, higher yields imply lower emission factors in the cultivation phase of the supply chain, since more GJ will be produced. Likewise, lower yields result in higher emission factors. However, the simplest way to increase yields is to increase the quantity of nitrogen fertiliser, which will (partially) offset the decrease in the emission factor. For example, in (Stehfest, Ros, and Bouwman, 2010) it is shown that if higher yields are merely achieved by increasing the quantity of nitrogen fertiliser this could lead to additional emissions of up to 150 gCO_2eq/MJ of fuel. For the same reason, also in (Hamelinck and Hoogwijk, 2007), the results for 2050 in terms of emission factors for most supply chains based on food/feed crops, are very comparable with the results for 2005. However, in (Stehfest et al., 2010) it is computed that if higher yields would be achieved by simultaneous improvement of management, crop varieties and fertiliser input, additional emissions could stay below 5 gCO_2eq/MJ of fuel. For example, the application of fertilisers can decrease, using finer in-field techniques.
- **Production of N fertilisers.** There are large differences in the upstream emissions of the various N fertilisers available. In BioGrace, emissions from the production of N fertilisers are based on older fertiliser production technology. As can be seen in Figure 2.4, disregarding these emissions would significantly lower the cultivation emission factors for most pathways from just 1 gCO_2eq/MJ for FAME from soya beans up to 11 gCO_2eq/MJ for FAME from rapeseed. Although N_2O emissions associated with the production of fertiliser could be avoided at rather low costs (Hamelinck and Hoogwijk, 2007), up to now there was no incentive to do so.
- **Application of N fertilisers.** This factor gives rise to large uncertainty in the cultivation phase because N_2O field emissions from the application of N fertilisers are computed following the IPCC Tier 1 method and the DNDC model (Gilhespy et al., 2014). An indication of the uncertainty is the -70% to +300% uncertainty range given for the direct emission default factors provided by IPCC. And even this range does not capture all field measurements made. Also, DNDC and IPCC methods are not yet applicable worldwide. Furthermore, it can be expected that fertiliser application will decrease in the future, using finer in-field techniques.
- **Fossil fuel emissions** The fuels (mainly natural gas) used in boilers or CHP plants in the processing phase could be replaced by biomethane. Also instead of fossil diesel, biodiesel could be used in trucks and ships to transport the biofuels and also

in the equipment used to cultivate the crops (i.e. tractors). This would lower the energy and greenhouse gas emission factors in all phases of the supply chain, but it would also raise significantly the amount of land needed to produce the same amount of biofuels (i.e. the efficiency of the supply chain as shown in Table 2.4 would be reduced) and thus it would raise the direct and indirect emissions from land-use change as presented in Chapter 3. Although there are examples in the real world, it is unlikely that large scale fossil fuel replacement will take place in the production of biofuels in the short term. It would require strict policies and sustainability criteria.

2.3 Solid biofuels

Since there is no PBL material on supply-chain emissions of solid biofuels (chips and pellets), our starting point was a representative publication by AEA (Bates and Henry, 2009) on supply-chain emissions from chips and pellets in the United Kingdom. This study distinguishes the following sources of wood fuel:

- Forestry residues: unused timber (e.g. branches) from conventional forestry operations.
- Short rotation coppice: an energy crop (typically willow) which is grown and harvested every few years.
- 'Clean' wood waste: wood waste from sawmills, or wood waste (if untreated) from furniture production.

For all source categories AEA takes into account emissions of CH_4 and in particular N_2O during combustion, the sum being about 1,7 gCO_2eq/MJ. The supply-chain emission factor for chips, without combustion, from national (in this case, the United Kingdom) forest residues are about 3 gCO_2eq/MJ. When residues are imported from abroad additional transport emissions between 3 and 7 gCO_2eq/MJ can be expected, in this case from the Baltic States (Estonia, Latvia and Lithuania) and Canada, respectively.

The carbon factor for pellets is higher than for chips due to the additional energy consumption involved in drying, milling and pelletising, and ranges from 8 to 30 gCO_2eq/MJ depending on the type of wood used to produce the pellets, its country of origin, and method for drying the wood and pre-pelletisation. The highest emissions are associated with the processing of short rotation coppice. This is mainly due to the high moisture content of the wood and therefore the energy requirements in the drying process. The impact of transporting wood from abroad can be seen with higher emissions from the Baltic States and Canadian sources, especially for wood processing waste. The emissions for chips from woody crops are based on the cultivation emissions of woody crops for ethanol (see Figure 2.2) assuming an efficiency of 65% (PBL, 2008). These emissions mainly reflect the production and application of N fertilisers which are, as indicated in Section 2.2.5, subject to large uncertainty ranges.

Table 2.5 *Supply-chain emissions of chips and pellets for different source categories in the United Kingdom. Based on (Bates and Henry, 2009) and on Section 2.2.1 for chips from woody crops. All numbers include 1,7 gCO_2eq/MJ due to N_2O (1,6g) and CH_4 (0,1g) emissions during combustion.*

<table>
<tr><th rowspan="2">Energy Carrier</th><th rowspan="2">From</th><th colspan="2">United Kingdom</th><th>Baltic States</th><th>Canada</th></tr>
<tr><th>Batch</th><th>Bulk</th><th colspan="2">Bulk</th></tr>
<tr><td rowspan="4">Chips</td><td>Forest residues (wet)</td><td colspan="2">5</td><td>8</td><td>11</td></tr>
<tr><td>Short rotation coppice (wet)</td><td colspan="2">6</td><td colspan="2"></td></tr>
<tr><td>Wood processing waste (wet)</td><td colspan="2">4</td><td>6</td><td>10</td></tr>
<tr><td>Woody crops (dry)</td><td colspan="2">15</td><td colspan="2"></td></tr>
<tr><td rowspan="3">Pellets</td><td>Forest residues</td><td>8</td><td>12</td><td>15</td><td>18</td></tr>
<tr><td>Short rotation coppice</td><td>16</td><td>30</td><td></td><td></td></tr>
<tr><td>Wood processing waste</td><td>9</td><td>16</td><td>20</td><td>25</td></tr>
</table>

Obviously, there are other studies that report on the supply-chain emissions of chips and pellets, but that could not be incorporated in this study due to the short timeframe. For example, a recent study on the life cycle assessment and uncertainty analysis of wood pellet-to-electricity supply chains from forest residues (Röder, Whittaker et al., 2015) showed that pellets can reduce greenhouse gas emissions by up to 83% compared to coal-fired electricity generation, but when parameters such as different drying fuels, storage emission, dry matter losses and feedstock market changes were included the bioenergy emission profiles showed strong variation with up to 73% higher greenhouse gas emissions compared to coal. Especially in the case of large scale storage and/or transport of wood chips anaerobic conditions and the formation of methane cannot be excluded. The impact of methane emissions during storage has shown to be particularly significant regarding uncertainty and increases in emissions. Investigation and management of losses and emissions during storage is therefore key to ensuring significant greenhouse gas reductions from biomass.

3 Emissions from land-use change

Key messages:

- Direct land-use change (DLUC) is 'the situation in which land use is changed from any previous use to bioenergy feedstock production'. Indirect land-use change (ILUC) is 'the change in land use outside a feedstock's production area that is induced by changing the use or production quantity of that feedstock'.
- DLUC calculations based on the RED methodology (EC, 2010) show that the conversion of forest land to bioenergy cropland emits large amounts of greenhouse gas (up to 360 gCO_2eq/MJ). Conversion of grasslands shows a range from -74 gCO_2eq/MJ (palm oil in Indonesia) to +83 gCO_2eq/MJ (biodiesel from soya beans in Brazil). Other feedstocks that sequester significant amounts of carbon when converted from grasslands are switchgrass, miscanthus, sugar cane, Jathropha and forest plantations.
- Uncertainty in DLUC emissions based on the presented method (EC, 2010) using default carbon stock values for soil and vegetation is high. Preferably real world data should be used.
- DLUC emissions are just part of the effect as the additional demand for biofuels products often leads to ILUC, as well. The extent to which indirect effects occur depends on many economic factors (e.g. yield increase, consumption changes, availability of the feedstocks, prices of inputs).
- According to a number of recent studies, uncertainty in overall LUC emissions is high. Based on the economic studies examined various types of conventional bioethanol have a LUC factor of approximately 20 gCO_2eq/MJ, with a range of 3 to 61 gCO_2eq/MJ and for conventional biodiesels this is around 35 gCO_2eq/MJ with a range of 7 to 94 gCO_2eq/MJ.
- For palm oil biodiesel and biodiesel in general the use of peatland in Malaysia and Indonesia play an important role in the greenhouse gas effects.
- Harvest residues have the potential to have LUC factors close to 0.
- Direct or indirect conversion of forest should be avoided since these will lead to high emissions, using a 30 years allocation period for land-use emissions. This plays a relatively larger role for biodiesel.
- Perennials have the potential to have relatively lower LUC factors since they have higher living biomass carbon and higher soil organic matter carbon.
- Using marginal land – land that is not used for any economic purpose, now or in the scenario period – results in low LUC emissions. However, this land is often not used for a reason; for example, because it has a low level of fertility or limited accessibility.

3.1 Introduction

Biomass is produced on land. Therefore, the cultivation of crops dedicated to energy production influences the land-use system. This chapter deals with the land-use emission caused by bioenergy from (woody) energy crops, harvesting residues, and fuelwood (if a conversion to a plantation is involved). The crops included are starch, sugar and oil crops, or 'food' crops, and lignocellulosic crops, which include short rotation coppice such as willow and poplar, and grasses such as miscanthus and switchgrass. The former are referred to as conventional biofuels and the latter are examples of advanced biofuels. Additionally, we present figures for forest plantations, which is also lignocellulosic but having a longer growth period than the lignocellulosic crops.

Land-use change emissions can be divided in direct land-use change (DLUC) and indirect land-use change (ILUC) emissions. In Section 3.3 DLUC emissions are quantified based on a methodology of the European Commission. For ILUC emissions a quantification is made in Section 3.4 based on the scientific literature . The calculations are accompanied by uncertainty ranges and the major sources of uncertainty are described in Section 3.5. In addition, spatial and temporal scales with respect to land-use change also are discussed. This includes global crop locations and the role of current vegetation and soil conditions and peatlands. On temporal scales, we discuss the effect of different amortisation periods (also called allocation periods).

3.2 Defining direct and indirect land-use change

(Searchinger et al., 2008) and (Fargione, Hill, Tilman, Polasky, and Hawthorne, 2008) show that agricultural land expansion caused by the demand for biofuels can possibly lead to greenhouse gas emissions from land-use change. Assuming the biofuels are taken from the commodity market their reasoning is as follows: By diverting crops from other uses to biofuel production prices will rise and farmers will respond by producing more. Part of the higher production will occur by expansion of cropland at the expense of natural land. This conversion may lead to greenhouse gas emissions because the vegetation biomass and soil organic matter of the cropland is often lower than that of the original land use.

These land-use change emissions originating from crop expansion can be divided in two categories: DLUC and ILUC. (Wicke, Verweij, Van Meijl, Van Vuuren, and Faaij, 2012) define DLUC as 'the situations in which land use is changed from any previous use to bioenergy feedstock production itself' and ILUC as 'the change in land use outside a feedstock's production area that is induced by changing the use or production quantity of that feedstock'.

The land-use effects of the demand for biofuels on the agricultural system are illustrated in Figure 3.1. To meet the demand for biofuels as a feedstock, one option is to use land that is currently in agricultural or forestry production, or other economical use, and another option is to convert land that currently is not in production. In the latter case, there is a clear one-to-one relation between the production of feedstock and land-use change emissions. This would be DLUC. In the case the feedstock is grown on land previously in production for food or feed crops, there are theoretically three options. One is that the former production is realised elsewhere through the conversion of unproductive or natural land into agricultural land. Second is through intensification of agriculture to increase yields (K. P. Overmars, Stehfest, Ros, and Prins, 2011; Stehfest et al., 2010). And third is to reduce the original consumption and to use this spared land for biofuel feedstocks.

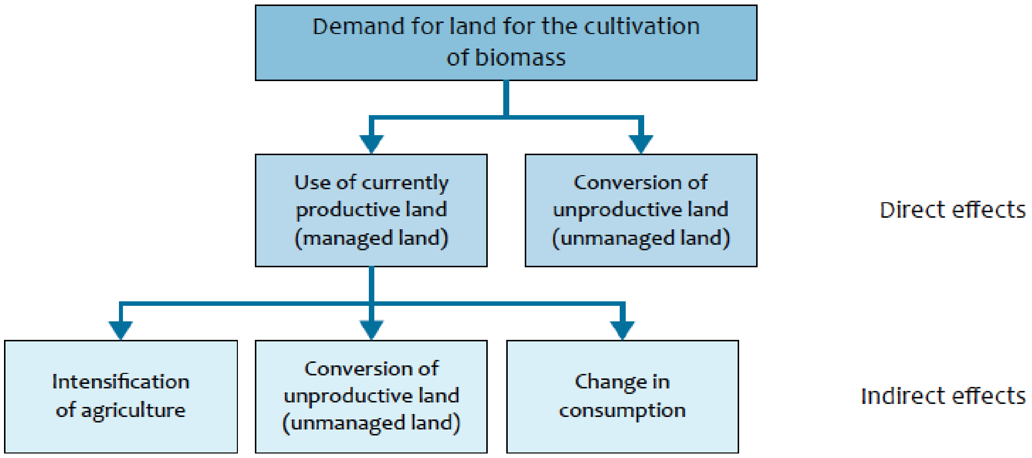


Figure 3.1 *Land-use effects of the additional demand for biofuel crops. Source: (J. P. M. Ros et al., 2010)*

One could imagine scenarios in which intensification is high and land is abandoned, while having an increasing productivity, or that scenarios with low (meat) consumption or reduced food losses and reduced food waste would save land. This abandoned land (saved land) then could be used for bioenergy production. Several remarks here: Firstly, scenarios with land abandonment are only projected for specific regions (e.g. the EU27 and the United States). Total global agricultural land use is projected to increase, despite intensification and despite land abandonment in some regions, resulting in net emission increase over time for the total system. Secondly, in assessing the effects of using abandoned land with scenario studies one always have to take the fate of the land in the baseline situation into account. This baseline might project regrowth of forest in these locations. Then it is not evident that using this land for bioenergy would lead to higher greenhouse gas reductions. Thirdly, some believe that the extra demand for biofuels might speed up yield increase in such a way that land is saved from areas currently in production for food, feed and fibre. Theoretically this is difficult to defend. The price of land would decrease drastically and economic forces would drive agricultural back to these locations and the pressure to invest in productivity would decrease. In general, there are no models that show a decrease in agricultural land use in any of the scenarios. Feedstock sources that have no to very little influence on land use, neither direct or indirect, are harvesting residues, forest residues and waste from the food chain (i.e. household waste or oil from food preparation).

3.3 Emissions from direct land-use change (DLUC)

In the context of the Renewable Energy Directive (RED) of the European Union[1] (EC, 2009), a methodology has been developed to estimate DLUC emissions (EC, 2010). It is a simple and transparent methodology that can be used for assessing carbon impacts due to land-use change for bioenergy in any region of the world. If data are available on carbon content of the soil and the vegetation for a certain plot from before and after this land was converted to be used for growing biofuel feedstocks, the annual land-use emissions (or their sequestration) can be calculated using this methodology. If no accurate data are available, the methodology can be applied for inorganic soils using standard values that can be taken from 18 tables in the 'Guidelines for the calculation of land carbon stocks' (EC, 2010). If carbon stocks in organic soils are affected by drainage, this can result in additional soil carbon losses that are not covered by this methodology.

Technical details of the methodology are summarised in Annex 1. In short, the direct emission factor of a specific conversion into land for the production of bioenergy is based on:

1. **Soil type.** To compute the Soil Organic Carbon (SOC) that is naturally present in the 0–30 centimetre topsoil layer, the methods distinguish high activity clay soils, low activity clay soils, sandy soils, spodic soils, volcanic soils and wetland soils.
2. **Land-use type.** To compute the loss of SOC in the new situation, the original land-use type must be known. Seven land-use types are distinguished: cultivated cropland, perennial cropland, grassland including savannah, undegraded native forest, managed forest, and shifting cultivation (either mature or shortened). In general the conversion of the natural situation to cultivated or perennial cropland leads to lower SOC values.
3. **Vegetation/crop type.** Different vegetation/crop types have different amounts of above- and below-ground living biomass. The methodology distinguishes sugar cane, Miscanthus, perennial crops, other crops, forest plantations, grassland, scrubland and forests having between 10% and 30% canopy cover or having more than 30% canopy cover.
4. **Climate region.** Climate affects the SOC in the 0–30 centimetre topsoil layer and the carbon in the different land-use types. Several climate regions are distinguished: tropical (dry, moist, wet and montane), subtropical or warm temperate (dry and moist), cool temperate (dry and moist) and boreal (dry, moist and wet).

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[1] The RED is an overall policy for the production and promotion of energy from renewable sources in the EU. It requires the EU to meet at least 20% of its final energy needs with renewable energy by 2020 – to be achieved through the attainment of individual national targets. All EU countries must also ensure that at least 10% of their transport fuels come from renewable sources by 2020. First generation biofuels, which are based on food crops, are capped on 7%.

5. **Ecological zone.** The ecological zone affects the carbon in the different vegetation types. Within climate regions, different ecological zones are distinguished. For example the tropical dry region is divided into tropical dry forests and tropical scrubland.
6. **Land-use management.** Refers to the amount of tillage in the case of (perennial) cropland; the more tillage is applied the quicker soil organic matter will oxidise. In the case of grassland and savannah, it refers to the level of management going from 'improved management' to 'severely degraded'.
7. **Fertilisation.** The application of chemical fertilisers and manure lead to more soil organic matter accumulation. The fertilisation levels included are: low, medium, high with manure, and high without manure.
8. **Yield of bioenergy crop/plantation.** The methodology results in an accumulated carbon loss (or gain) per hectare that is divided by an allocation period of 30 years to obtain an annual rate[2]. To convert this into an annual emission factor in terms of gCO_2eq/MJ end product, annual yields in terms of energy content are needed. In this study we apply the average energy yields from Table 2.4, which are based on BioGrace (see Chapter 2).

The methodology offers a wide range of combinations of values for the variables described above. However, the number of realistic combinations is limited. Table A1.1 in Annex 1 shows a number of combinations that apply to typical land conversions – from grassland/savannah and forests to a set of key biofuel crops – for a number of world regions, and possible combinations of climate, soil fertilisation and management, resulting in a range of possible DLUC values. Table 3.1 shows the energy yield and emissions per hectare due to changes in the carbon content of the soil – which is a function of climate, land use, input and management – and the resulting DLUC values in gCO_2eq/MJ. The latter is also shown in Figure 3.2. Conversion emissions from grassland to plantations are presented in Table A1.2.

[2] See Section 3.4, under 'Allocation or amortisation period', for more information on the allocation period.

Table 3.1 *Tonne C per ha (i.e. total C lost), energy yields, and maximum and minimum DLUC emission values, using an allocation period for the initial C loss of 30 years*[3] *and based on possible land-use conversions using methods and tables from (EC, 2010).*

	Yield (MJ/ha)	Minimum (tC/ha)	Maximum (tC/ha)	Minimum (gCO_2eq/MJ)	Maximum (gCO_2eq/MJ)
Conversion of grassland/savanna					
Maize ethanol, US	57	7	36	15	78
Sugar cane ethanol, Brazil	134	-34	-4	-31	-4
Sugar beet ethanol, EU	214	-9	29	-5	16
Wheat ethanol, EU	68	-9	29	-17	51
Miscanthus ethanol, EU/US	61	-18	-8	-35	-15
Rapeseed biodiesel, EU	72	-9	29	-16	48
Soya biodiesel, US/Brazil	54	10	36	22	83
Palm oil, Indonesia/Malaysia	165	-99	-61	-74	-45
Sunflower biodiesel, EU	57	-9	29	-20	62
Jathropha biodiesel, Africa	91	-24	-16	-33	-22
Forest plantations[4]	144	-81	-2	-68	-1
Conversion of forest land[5]					
Maize ethanol, US	57	99	122	211	262
Sugar cane ethanol, Brazil	134	92	121	84	111
Sugar beet ethanol, EU	214	76	109	43	62
Wheat ethanol, EU	68	76	109	135	195
Miscanthus ethanol, EU/US	61	73	77	146	154
Rapeseed biodiesel, EU	72	76	109	128	184
Soya biodiesel, US/Brazil	54	102	157	232	359
Palm oil, Indonesia/Malaysia	165	125	161	93	120
Sunflower biodiesel, EU	57	76	109	163	234
Jathropha biodiesel, Africa	91	128	178	172	238

[3] See Section 3.4, paragraph 'Allocation or amortization period' for more information on the allocation period.

[4] See also Table A1.2 in Annex 1.

[5] Forest land – excluding forest plantations – having more than 30% canopy cover (EC, 2010)

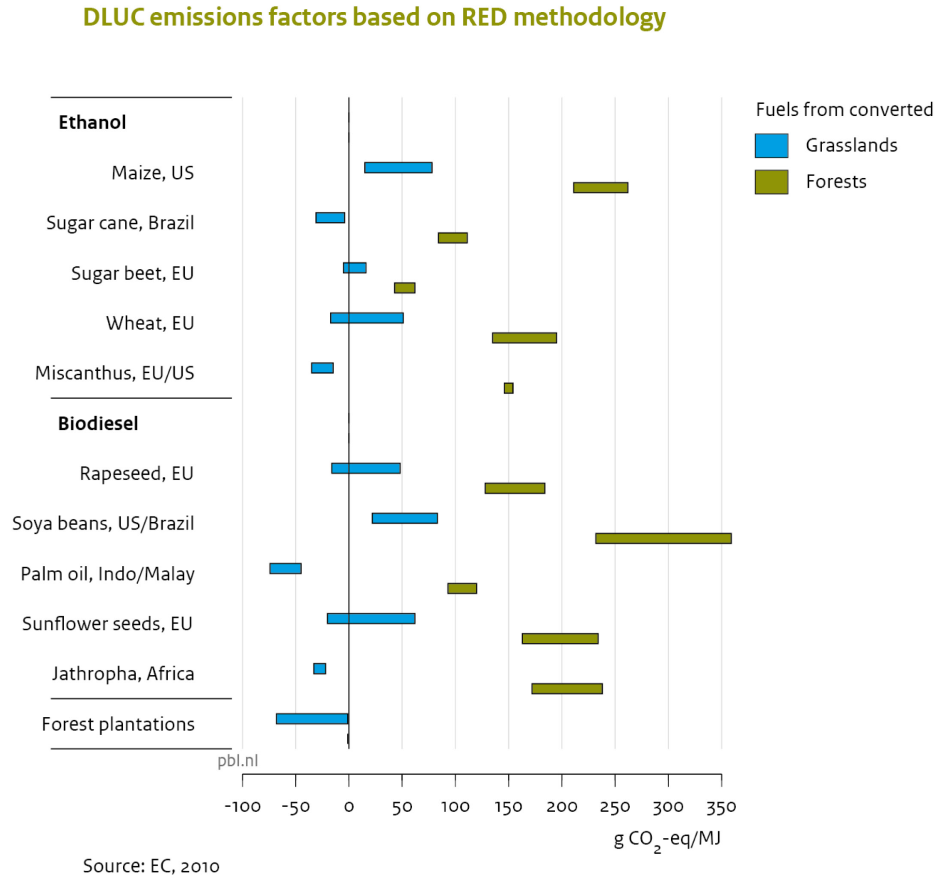


Figure 3.2 *DLUC emission or sequestration values using an allocation period for the initial C loss of 30 years*[6]*. Based on possible land-use conversions calculated using methods and tables from (EC, 2010).*

In calculating DLUC values using this methodology we can draw several general conclusions:

1. Conversion of grassland is emitting less than conversion of forest, since forest has a larger carbon pool.
2. Conversion of forest leads to larger emissions than the use of fossil fuels (fossil fuels = 84 gCO_2eq/MJ) using the amortisation period of 30 years.
3. Perennials hold more carbon in the soil and living biomass than arable crops. Conversion to perennials can even lead to CO_2 sequestration.
4. Manure and fertiliser increase soil organic matter but can also increase emissions of greenhouse gas, such as N_2O. In principle this is accounted for in the supply-chain emissions (Chapter 2).
5. Reduced tillage will help maintain soil organic matter and therefore to limit net carbon emissions.

[6] See Section 3.4, under 'Allocation or amortization period', for more information on the allocation period.

Discussion
The methodology presented is a straightforward and simple way to estimate DLUC emissions of biofuel production. However, it is difficult to assess the fate of the land in the future. Theoretically, the only case where DLUC is the only effect of land conversion is when there is no influence on other types of land use (e.g. agriculture) over a period of 30 years. Otherwise, there will also be an indirect land-use change effect, as is described in the next section. Another drawback of the method is that the drainage and oxidation of peatland is not included. Overmars, Edwards, Padella, Prins, and Marelli (2015) report a value of 27.3 tC/ha/yr for oil palm on peatland, considering 33% of the plantations are on peatland. Using the yield figures on palm oil from Table 3.1 this would lead to an additional emission of 20 gCO_2eq/MJ. (Marelli, Ramos, Hiederer, and Koeble, 2011) report that up to 55% of EU ILUC emissions (20 gCO_2eq/MJ) could be attributed to palm oil based on the EU mix of several biofuels. For palm oil alone this value would be higher in this study.

3.4 LUC factors based on literature

DLUC emissions as presented in the previous section are just part of the effect of growing energy crops. Potentially, the additional demand for bioenergy leads to indirect effects as well. This occurs in case an energy crop is grown on a location that already is in production, for example to produce food. A DLUC calculation to assess the effects of land-use change emissions is only sufficient in case the land is not in use for another purpose at the time of conversion nor in the future, in other words if there is no interference with food production or other uses. The magnitude of ILUC effects – i.e. land expansion elsewhere, yield increase and consumption changes, see Figure 3.1 – is determined by market forces; prices and availability of feedstocks, land and other production factors. In case an energy crop is grown on agricultural land, the price of the formerly cultivated crop will increase due to its diminished supply. This price increase is an incentive to farmers to grow more of this commodity on other land through intensification and/or land expansion. On the consumption side higher prices cause a decrease in consumption (K. P. Overmars et al., 2011). Together these forces lead to a new equilibrium in demand and supply and the prices of the commodities. In this section, we use a series of model studies on land-use change emissions from growing energy crops to estimate an emission factor, and a range for each biofuel feedstock. The calculations are based on a series of steps or principles, which are described below.

General calculation principles
Calculations of ILUC of energy crop production start with the gross area needed to produce a certain amount of biofuel. This gross area is literally the land on which the crop grows. This gross area depends on the crop and the yields that are achieved, which is subject to location and management. Additionally, the processing efficiency of the raw material into biofuel determines the final area needed to produce a certain amount of energy (see Chapter 2).

Secondly, the net land effect of the additional agricultural demand is calculated. The net effect is lower than the gross area needed due to different processes. The additional demand for biofuel crops has an effect on price, supply and demand of those crops in particular, but of agricultural crops in general as well. This leads to agricultural intensification (e.g. increased fertilisation, and improved management), changes in consumption and land expansion (see also Figure 3.1). How these three effects contribute to fulfilling the demand for biofuel crops is dependent on the global economy, trade and policies. The resulting land expansion that is actually necessary to produce the additional demand is the net area effect.
In this second step the so-called co-products should also be taken into account. Co-products are all other products that are produced from the harvested crop, for example animal feed or glycerine (see also Chapter 2). These co-products will diminish the production of these commodities elsewhere, and therefore reduces the net land effect of the biofuels production.
The third step is to determine where land-use conversions are actually taking place, and subsequently the carbon content of that land. By subtracting the carbon content per hectare, above ground and below ground, (i.e. soil organic matter and vegetation) of the original land use from the carbon content of the new land use a carbon effect can be determined. Often this is reported as gCO_2eq/MJ of biofuel (K. P. Overmars et al., 2011; Prins, Overmars, and Ros, 2014).

Allocation or amortisation period
Emissions from land-use change are typically high in the period right after conversion (e.g. deforestation). By averaging these emissions over the time span of the policy or scenario, biofuels produced in a different period of the scenario are treated equally over the time span of the scenario. This time span is called allocation period or amortisation period. The allocation period used in this study is 30 yrs. In many studies either 30 years or 20 years is used. Generally, the US uses 30 years and the EU 20 years. Converting the 30 years allocation period to a 20 years allocation period the numbers should be multiplied by 1.5 (i.e. the emission factors would be 50% higher). So, using a longer allocation period will lead to lower emissions factors.
It is important to realise that, after the allocation period, the LUC emission factor will be 0. In the timespan of the allocation period all land-use emissions are equally allocated to these years. There are several reasons to include an allocation period. One is that one wants to know the actual emissions/emission reductions at a certain point in time, for example, after 20 or 30 years or at the end of a specific scenario. Another reason is that it is uncertain how long a technology will be used. If the technology, in this case biofuels from agriculture, is abandoned after a certain period the land-use change emissions cannot be immediately reversed.

ILUC, DLUC and LUC

Modelling studies normally start with a fixed demand for feedstock for biofuels, based on policy targets, for example. This demand is used in the economic modelling to determine the total future demand and supply for all uses of agricultural products, including food, biofuel and other uses. This total demand is used in land-use modelling. The interplay between economic and land-use modelling determines how (by intensification or by land expansion) and where (i.e. by replacing other crops or by exploiting new land) this demand is met. The result of this (economic) process is unknown beforehand. Therefore, it is impossible to distinguish between DLUC and ILUC in such scenarios. Similar to what happens in reality when a feedstock is bought on the market, it is not exactly clear which land the biofuel feedstock came from. Therefore, most studies describe the total the carbon emissions from land-use change, i.e. the sum of both ILUC and DLUC emissions (e.g. (Wicke et al., 2012)). The numbers reported below are the total emissions due to land-use change effects, referred to as LUC emissions.

Calculation of LUC emission factors based on recent literature

Because of uncertainties and differences in modelling assumptions it is difficult to assign one LUC emission factor to a certain biofuel or biofuel pathway (see also Section 3.5). For each different feedstock and for each region, the emission factor is different and also variable in time. Nevertheless, in this report we present a set of emission factors, including uncertainty ranges, which can be regarded as the current state of knowledge as presented in the literature.

We base our LUC factors on two recent literature reviews of (Wicke et al., 2012) and (Ahlgren and Di Lucia, 2014). From these reviews we selected the most recent (2010 or later) publications because later studies are more complete and more advanced than the earlier ones. Science progressed and studies have converged over time on what processes to include and how to approach the problem. In case more studies from the same organisation are included we selected only the most recent one, e.g. (Laborde, 2011) and (Al-Riffai, Dimaranan, and Laborde, 2010), both from IFPRI. In case we were aware of an update not included in Wicke et al. or Ahlgren and Di Luca we included the update instead of the older version, e.g. (K. Overmars et al., 2015) substituting (K. P. Overmars et al., 2011). We collected one average for each biofuel feedstock in each study. From these averages we report the mean, minimum and maximum. Using this method the calculation can be regarded as an inter study comparison which does not reflect the full range of outcomes since each study has its own range for each biofuel covered in that study. In other words, within model uncertainty (see also Section 3.5) is not included. If this would be included, there would be a larger range than presented in Tables 3.2, 3.3 and 3.4, as is shown for example in Plevin, Beckman, Golub, Witcover, and O'Hare (2015) and Laborde (2011).

Results

Table 3.2 shows LUC emission factors based on economic and descriptive studies combined in gCO_2eq/MJ biofuel using a 30 years allocation or amortisation period. Descriptive studies refer to studies with a causal descriptive approach or simple descriptive effect relations and the economic studies refer to studies using economic models such as general or partial equilibrium models. The highest emission factors are found for biodiesels, which on their own cause average emissions that vary from 52% to 84% of the emissions from fossil oil (84 gCO_2eq/MJ) depending on the feedstock. LUC emissions from first generation petrol substitutes (i.e. bioethanol) are 7% to 24% and from advanced biofuels (ethanol) -1% to +21% compared to emissions from fossil oil.

An important effect in determining the emissions of biodiesels, is the substitution of vegetable oils with palm oil; i.e. an increasing demand for vegetable oils other than palm oil to produce biodiesel, result in a higher demand for palm oil if these vegetable oils are substituted with palm oil in other sectors. Consequently, all vegetable oils have a relatively high emission factor because palm oil is often grown on former forest land and/or peatland implying high greenhouse gas emissions. This effect even influences the ILUC factors of ethanol crops since ethanol crops may replace oil crops. Marelli et al. (2011) estimate that, for the EU mix of biofuels, about 55% of the ILUC factor is related to emissions from peatland in Indonesia and Malaysia.

Table 3.2 *LUC emission factors based on economic and descriptive studies combined using a 30 years allocation or amortisation period. The characters between brackets refer to studies listed in Table 3.5.*

Unit: gCO_2eq/MJ	*n*	Minimum	Maximum	Mean
Conventional biofuels (petrol substitutes)				
Maize ethanol	11	6.3 (h,m)	48.5 (e)	18.7
Sugar cane ethanol	10	3.5 (p)	60.3 (j)	20.0
Sugar beet ethanol	4	3.4 (k)	9.4 (d)	6.3
Wheat ethanol	8	-29.1 (a)	61.4 (e)	12.8
Advanced biofuels (petrol substitutes)				
Switchgrass ethanol	4	1.3 (m)	44.0 (n)	17.2
Willow or poplar ethanol	2	2.0 (m)	25.3 (f)	13.7
Wheat straw ethanol	1	0.8 (m)	0.8 (m)	0.8
Miscanthus ethanol	3	-6.1 (c)	16.6 (n)	4.0
Maize stover ethanol	2	-1.3 (n)	-1.2 (c)	-1.3
Conventional biofuels (diesel substitutes)				
Rapeseed biodiesel	8	1.3 (j)	136.2 (m)	45.9
Soya biodiesel	10	14.5 (d)	149.3 (m)	52.4
Palm oil biodiesel	8	12.5 (j)	138.5 (m)	43.2
Sunflower biodiesel	3	35.0 (i)	137.0 (m)	70.3
Jathropha biodiesel	1	62.0 (m)	62.0 (m)	62.0

Tables 3.3 and 3.4 and Figure 3.3 report the results for descriptive studies and economic studies separately, as did Ahlgren et al. and Wicke et al. Figure 3.3 shows that the average LUC factors in descriptive studies for biodiesels are higher than those in economic studies. The average EU and US values for ethanol are lower in the descriptive studies. The tables also show that the descriptive studies have higher variability than the economic studies.

The effect of higher variability in descriptive studies can be explained as follows. The descriptive studies often assume a quite specific case. For example, a crop on degraded land in the EU leading to negative emissions (i.e. sequestrations) or crops on forest land or peatland leads to high emissions. Of course, these studies do include substitution effects, but mostly a first order effect based on the assumptions.

Table 3.3 *LUC emission factors based on descriptive studies using a 30 years allocation or amortisation period. The characters between brackets refer to studies listed in Table 3.5.*

Unit: gCO_2eq/MJ	*n*	Minimum	Maximum	Mean
Conventional biofuels (petrol substitutes)				
Maize ethanol	2	6.3 (h, m)	6.3 (h,m)	6.3
Sugar cane ethanol	5	7.0 (l)	60.3 (j)	23.0
Sugar beet ethanol	1	8.2 (m)	8.2 (m)	8.2
Wheat ethanol	4	-29.1 (a)	31.3 (f)	4.3
Advanced biofuels (petrol substitutes)				
Switchgrass ethanol	1	1.3 (m)	1.3 (m)	1.3
Willow or poplar ethanol	2	2.0 (m)	25.3 (f)	13.7
Wheat straw ethanol	1	0.8 (m)	0.8 (m)	0.8
Miscanthus ethanol	1	1.3 (m)	1.3 (m)	1.3
Maize stover ethanol	0	-	-	-
Conventional biofuels (petrol substitutes)				
Rapeseed biodiesel	4	1.3 (j)	136.2 (m)	48.4
Soya biodiesel	4	37.4 (a)	149.3 (m)	77.8
Palm oil biodiesel	4	12.5 (j)	138.5 (m)	54.7
Sunflower biodiesel	1	137.0 (m)	137.0 (m)	137.0
Jathropha biodiesel	1	62.0 (m)	62.0 (m)	62.0

The economic models simulate substitution and trade in more detail and, therefore, there are ILUC effects for many crops in many regions. The economic studies are much more detailed in the economic effects they incorporate. They have a more detailed, or higher order, cascade of effects of commodities substituting each other. Therefore, differences in ILUC between crops tend to fade out, partly; they converge more since they are built of components of all crops (due to crop diversion and crop substitution). The consequence of this is that in descriptive studies, EU and US ethanol has a rather low LUC emission factor. The regional land-use effect is small and differences between the crops fade somewhat due to the above-mentioned reasons. Sugar cane, soya and palm oil have relatively higher emission factors in the descriptive approach, because their first-order effects in the regions include forest conversion and peat oxidation.

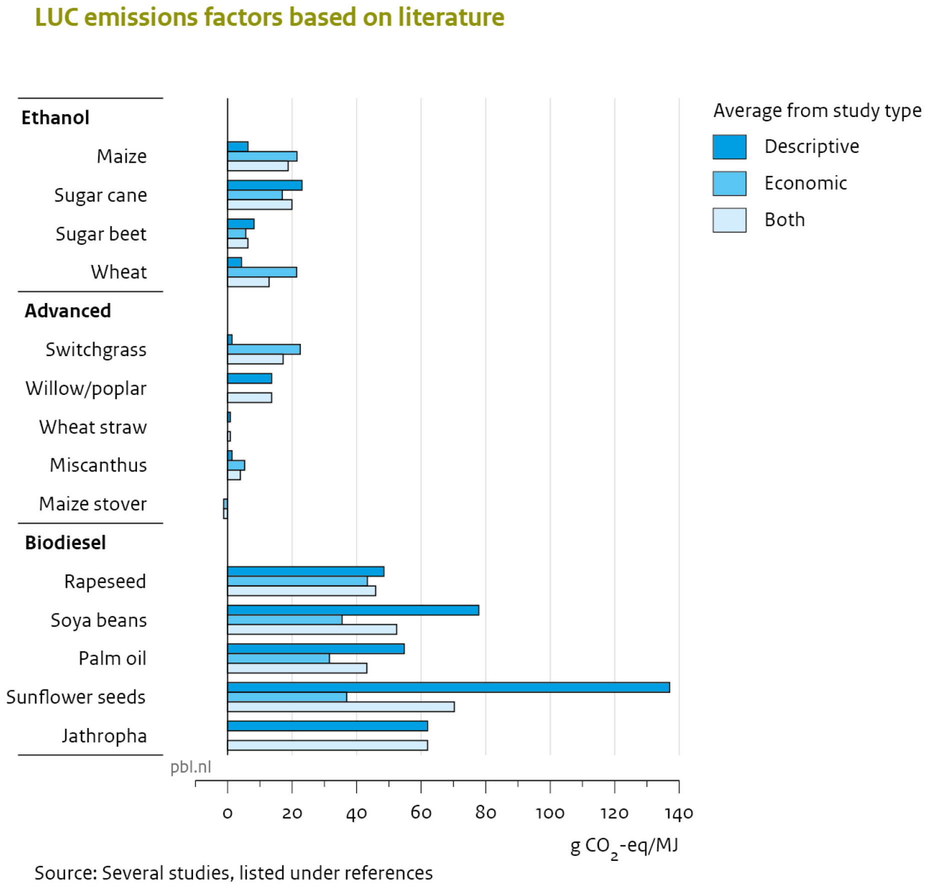


Figure 3.3 *LUC emission factors based on literature using a 30 years allocation period.*

The average LUC factor of conventional bioethanol in the economic studies is approximately 20 gCO_2eq/MJ, with an range of 3 to 61 gCO_2eq/MJ, and for conventional biodiesel this is around 35 gCO_2eq/MJ, with a range of 7 to 94 gCO_2eq/MJ. Wheat straw and maize stover ethanol has values close to 0. Although there are only few studies on these feedstocks, the results can be realistic, given that straw and stover are waste products. However, attention has to be paid to the influence of removing wheat and straw from the land on soil organic matter. Additionally, these harvest residues do in fact represent economical value in many cases (K. Overmars et al., 2015). Additional demand for these products may therefore result in indirect land-use change. The results for the advanced biofuels based on lignocellulosic feedstocks are less straightforward to interpret. The number of studies incorporated here is low. Some of them are of the descriptive type, possibly leading to low estimates, for example in (K. Overmars et al., 2015), where for EU miscanthus and switchgrass no trade effects were foreseen outside Europe. In general one would expect that the LUC factor for these crops would deviate from the conventional crops based on the differences in their energy yields, where the conventional biofuels are corrected for co-products. (K. Overmars et al., 2015) assume for both miscanthus and switchgrass dry-matter yields of 1.57 times the EU wheat yield at traded water content, while (Dunn, Mueller, Kwon, and Wang, 2013) report a 46% higher energy yield for miscanthus and a 25% lower energy yield for switchgrass, compared to corn. A positive aspect of these crops, leading to lower LUC emissions, is the perennial character (higher soil organic matter) and higher standing biomass.

Table 3.4 *LUC emission factors based on economic studies using a 30 years allocation or amortisation period. The characters between brackets refer to studies listed in Table 3.5.*

Unit: gCO_2eq/MJ	*n*	Minimum	Maximum	Mean
Conventional biofuels (petrol substitutes)				
Maize ethanol	9	7.0 (i)	48.5 (e)	21.5
Sugar cane ethanol	5	3.5 (p)	46.0 (b)	16.9
Sugar beet ethanol	3	3.4 (k)	9.4 (d)	5.6
Wheat ethanol	4	3.6 (d)	61.4 (e)	21.4
Advanced biofuels (petrol substitutes)				
Switchgrass ethanol	3	10.9 (c)	44.0 (n)	22.5
Willow or poplar ethanol	0	-	-	-
Wheat straw ethanol	0	-	-	-
Miscanthus ethanol	2	-6.1 (c)	16.6 (n)	5.3
Maize stover ethanol	2	-1.3 (n)	-1.2 (c)	-1.3
Conventional biofuels (diesel substitutes)				
Rapeseed biodiesel	4	7.1 (d)	93.7 (e)	43.3
Soya biodiesel	6	14.5 (d)	62.0 (b)	35.5
Palm oil biodiesel	4	13.0 (d)	47.9 (e)	31.6
Sunflower biodiesel	2	35.0 (i)	38.9 (k)	36.9
Jathropha biodiesel	0	-	-	-

Table 3.5 *Descriptive and economic studies used in the calculation of the LUC emission factors presented in Tables 3.2 to 3.4.*

a	Bauen, Chudziak, Vad, and Watson (2010) present a causal descriptive approach to model the greenhouse gas emissions associated with the indirect land-use impacts of biofuels. It is a study for the UK Department for Transport and it uses a set of scenarios with changing assumption on a variety of subjects. Palm oil scenarios differ in deforestation rate, continuous or single plantation, yield increase, and peatland expansion. The rapeseed oil scenario differs in the quantity of rape produced, effects on production in the Ukraine, deforestation rate in Indonesia and Malaysia, share of co-products used, and has varying co-product substitution rates. The soya oil scenario describes a case with oil substitution in China with different rates of rape and palm substituting soya. In the palm oil substitution they use the high and low ILUC palm scenario mentioned before. The wheat scenario includes changes in wheat trade balance, yield assumptions, different rates of deforestation in Indonesia and Malaysia, and varying shares of co-products. The sugar cane scenarios vary in demand, sugar cane production in other countries, yield increase, pasture displacement assumptions, pasture intensification rates, crop displacement location assumptions, and deforestation assumptions. This mix of assumption leads to a spread in LUC factors between 20 (sugar cane) to 80 gCO_2eq/MJ (palm oil).	Descriptive
b	CARB (2009) on regulations to achieve greenhouse gas emission reductions contains two lookup tables with direct emissions and LUC emissions for many different types of bioethanol and biodiesel production pathways in different areas of California.	Economic
c	Dunn et al. (2013) on LUC and greenhouse gas emissions from corn and cellulosic ethanol, have varying scenario and model settings regarding soil cultivation effects, crop yields and erosion. This leads to LUC factors of 24 for switchgrass, 12 for miscanthus, 0 for corn stover and 9 gCO_2eq/MJ for corn.	Economic
d	Darlington, Kahlbaum, O'Connor, and Mueller (2013) on LUC emissions of European biofuel policies utilising the Global Trade Analysis Project (GTAP) Model.	Economic
e	Edwards, Mulligan, and Marelli (2010) on ILUC from increased biofuels demand present a series of modelling approaches using the same scenario of marginal increase in various biofuel feedstocks. Differences in outcome are the effect of the different models and their different sets of model assumptions. Results differ between 89 for US maize, 67 for EU wheat, 174 for rapeseed, and 62 gCO_2eq/MJ for palm oil.	Economic
f	Fritsche, Hennenberg, and Hünecke (2010) on sustainability standards for internationally traded biomass assume 25% and 50% ILUC area relative to the cropping area. The types of land-use changes considered are from arable land, grassland, degraded land, savannah, and forests. As expected, converting degraded land leads to the lowest ILUC factors or even carbon sequestration and converting forest generally leads to the highest ILUC figures. This last assumption leads to differences in ILUC estimates of about 30 gCO_2eq/MJ for wheat, rapeseed and short rotation coppice. For palm oil, soya and sugar cane the differences between scenarios are about 180, 100 and 120 gCO_2eq/MJ.	Economic
g	Hertel et al. (2010) on the effects of US maize ethanol on global land-use and greenhouse gas emissions, present uncertainty using a sensitivity analysis for land-use change locations, land-use change emissions and yield (increase) factors in the model. This leads to a range of 15 to 90 gCO_2eq/MJ maize ethanol.	Economic
h	Kim, Dale, and Ong (2012) allocate greenhouse gas effects among different uses of land. They have two different sets of assumptions on co-products for their maize case. In one maize is replaced with crops for a vegetable-based human diet and the other with an animal-based diet. Although having low ILUC estimates, it leads to a difference greater than a factor 2 in ILUC emissions (3.9 vs 8.6 gCO_2eq/MJ).	Descriptive
i	Laborde (2011) assesses the land-use change consequences of European biofuel policies. It contains two scenarios: one assuming current trade policies and one assuming trade liberalisation. The outcomes between these scenarios differ by 2 gCO_2eq/MJ.	Economic
j	Lahl (2010) presents a regional quantification of climate relevant land-use change and options for combating it. For wheat and rapeseed assumptions are made on the amount of grassland and forest converted. The palm oil scenarios differ in the levels of deforestation. The soya scenarios vary in the level of deforestation and livestock replacement. This leads to LUC emission factors between 8 (rapeseed) and 91 gCO_2eq/MJ (wheat).	Descriptive
k	Marelli et al. (2011) estimate greenhouse gas emissions from global LUC scenarios. They present two sets of assumptions on sugar cane and palm oil. One including burning of residues from sugar cane and one considering palm and sugar cane as long-term crops with less loss of soil organic matter. The low and high estimates differ by 10 gCO_2eq/MJ at most.	Economic
l	Nassar, Antioniazzi, MR, Chiodi, and Harfuch (2010) describe an allocation methodology to assess greenhouse gas emissions associated with LUC. They present two numbers varying in land-use change assumption. One considers only native vegetation to convert and another that includes all land-use changes, including change from one agricultural use to another. The resulting numbers differ by a factor of 1.2.	Descriptive

m	Overmars et al. (2015) on ILUC estimates from biofuels based on historical data. Two models are used having different land-use allocation rules. This leads to values that differ typically a factor 1.1–1.2, but in more extreme cases a factor 2, 3, or 4 (respectively wheat, lignocellulosic crops and sugar cane). They have also two ways of including co-products in the calculation. One with allocation of land to co-products based on energy content and one based on economic value. These differ typically a factor 1.25 and in the most extreme case (Jathropha) a factor of 2.	Descriptive
n	Taheripour and Tyner (2013) on LUC emissions due to conventional and advanced biofuels and uncertainty in land-use emissions factors, use three different land-use emissions sources: Woods Hole, CARB and the Terrestrial Ecosystem Model. Additionally they assume one scenario with emissions from cropland pasture, which is a subcategory in GTAP modelling, to cropland and one without. The latter assumption has hardly any effect. The total variation is in the range of 1 to 54 gCO_2eq/MJ.	Economic
o	Tyner, Taheripour, Zhuang, Birur, and Baldos (2010) present a comprehensive analysis on land-use changes and consequent CO_2 emissions due to US maize ethanol production. It includes three scenarios: one using the 2001 GTAP database isolating the US effects, a second including the 2006 GTAP database and including world economy and a third with adding yield and population growth compared to the second scenario. Results for US corn differ from 15–21 gCO_2eq/MJ maize ethanol.	Economic
p	US EPA (2010) is a regulatory impact analysis for the US. The estimates in are the high end and low end of a 95% confidence interval (Monte Carlo simulation) of uncertainty in satellite data and emission factors of land-use conversions. The range widths are 26 gCO_2eq/MJ for maize ethanol, 17 gCO_2eq/MJ for switchgrass ethanol and 21 gCO_2eq/MJ for sugar cane ethanol and 69 gCO_2eq/MJ for soya biodiesel.	Economic

3.5 On the uncertainties and variability in LUC modelling

As discussed in the previous section, (integrated) modelling techniques are often used to determine the overall LUC effects. Bioenergy policies have impacts across. and even beyond, the whole chain of the agricultural economy. Models need to make use of assumptions, which inevitably lead to uncertainty. Additionally, models use different data sets, all having their own uncertainties.
Here we describe important sources of uncertainty and variability in LUC studies. Table 3.5 presented sources of uncertainty and variability in the descriptive and economic studies used in the assessment of Section 3.4. These within study factors causing differences in outcome are similar factors that cause differences between studies.
The assumptions made in descriptive modelling are quite different from those in the economic models. In many cases the descriptive studies use prescribed assumptions on land-use changes, where this is endogenous in the economic modelling studies. However, also in the economic modelling studies, many assumptions must be made. It should be realised that to describe *all* differences in assumptions and model settings between the studies goes beyond the scope of this report. Important sources of uncertainty and variability are:

1. Scenarios
Explicitly or implicitly all calculations use a reference scenario and one or more alternative scenarios. In the scenarios many choices are made leading to different model settings and input variables. For example, scenarios with a higher biofuels target lead to more greenhouse gas emissions per MJ. 'As expected, the direct emission saving coefficient is reduced as the level of the mandate increases. Greater pressure for biofuel production from a higher target results in increasing use of less efficient feedstock.' (Al-Riffai et al., 2010).
Other examples are policies such as trade policies and land-use policies (i.e. protected areas) that have to be included. These policies co-determine where land expansion will take place. Examples of scenario differences in Table 3.5 are *a*, *c*, *i* and *j*.

2. Co-products
Most models include the effect of co-products by attributing part of the land conversion to this product and not to the biofuel. However, the level of substitutability of different products, the level of uptake of the co-products in the economy and therefore the actual use of co-products depend on assumptions and different model set-ups of the economic models. Examples of scenario differences in descriptive approaches can be found in Table 3.5 (*h* and *m*).

3. Climate change feedback
Some models include feedback of climate change on agricultural production, others do not. Increased CO_2 concentrations may lead to changes in the climate. This can lead to either higher or lower agricultural outputs, dependent on whether the new climate is more or less

favourable by agricultural crops. Furthermore, increased CO_2 concentrations lead to higher CO_2 uptake by plants, which can lead to higher agricultural output. All these effects will have an impact on the agricultural system and thus on the LUC effect of biofuel production (J. P. M. Ros et al., 2010).

4. Impacts on oil price and production

There is an indirect effect on fossil fuels that may cause additional emissions. More biofuels mean a greater supply of fuels and therefore the price may drop and the consumption of transport fuels may increase; the so-called rebound effect (e.g. Smeets et al., 2014). In general economic studies account for this effect and descriptive studies do not.

5. Emissions from intensification

greenhouse gas emissions may increase due to intensification. As depicted in Figure 3.1, agriculture will possibly intensify due to the extra demand for agricultural products for biofuel use, so on top of yield increase under the reference scenario. One way to increase agricultural yields is to use more fertilisers. The increased use of fertilisers can increase the emission of N_2O, which is a strong greenhouse gas (Stehfest et al., 2010). Some models incorporate this effect while others are purely based on the greenhouse gas effect of land expansion. This is especially an important issue if marginal lands are used to increase bioenergy production, since growth on these lands depend on high (fertiliser) inputs.

6. Future yield levels, Marginal yield assumptions and cropping intensity

Another assumption that has to be made is on future yield levels. For example, technological change over time largely determines future yield levels. The technological change is a key variable that can be an input variable, but can also be endogenous. In some cases (part of) the increase in yield per hectare is made dependent on the demand.

Besides the yield improvement over time another aspect of yield is included in the model assumptions. Models use different algorithms with respect to the yield of unmanaged land that is taken into production. Some assume that the best yielding land is used first. This implicates that newly converted land is of lower quality than the existing land and therefore the average yield will decrease. Others take into account that agricultural land use not only depends on yield, but also on other factors such as accessibility and labour availability. In this case yields on newly developed land can be quite similar as other yields in the region.

Other assumptions associated to yield have to be made such as rotation patterns and cropping intensities. Increasing the cropping intensity, by reducing rotational fallows or by harvesting multiple crops per year, can reduce the amount of land expansion need for extra demand for agricultural products as compared to a constant cropping intensity.

7. Peatlands

As mentioned before, if land expansion – mainly for palm oil – in Indonesia and Malaysia occurs on peatlands, this can result in huge amounts of CO_2 emissions from peatland oxidation. The exact expansion of palm oil plantations on peatland is uncertain as well as the

carbon emissions of the peatland itself. This depends on total carbon stored (peat depth) and drainage levels. Related are assumptions on substitutability of one vegetable oil with another. Due to this substitutability and trade in vegetable oils, the feedstocks for biodiesel based on vegetable oil, other than palm oil, and even feedstocks for ethanol, can indirectly result in peatland conversion. 'Prices of rapeseed oil, soya oil and palm oil are well correlated, suggesting that the markets for these oils are well connected' (ICCT, 2013).
Studies that do not link the use of other vegetable oils to palm oil substitution or studies that have lower estimates of the use of peatland have much lower LUC emission factors for biodiesels.

8. Models and model settings

Model settings are an important determinant of the outcomes. Many of the settings are determined by the modellers and cannot always be determined precisely by theory or (empirical) study. An example are model settings in economic models. Outcomes of economic models are highly dependent on so-called elasticities. Elasticity settings determine, for example, the change in supply or demand in relation to changes in price. Often it is not possible to determine these elasticities empirically and therefore assumptions for the value of the elasticity are made. (Plevin et al., 2015) examined the effect of uncertainty in model parameterisation on the outcomes. They found that 95% of the outcomes are within the range of ±20 g CO_2eq/MJ of the mean. An example of the effect of different models (while using the same scenario) is study *e* in Table 3.5.

9. Land-use modelling

Besides the question of how much land expansion is necessary to accommodate the biofuels also the question where this occurs is of great importance. Allocation models or modules are used for this part. Again the data and parameter settings are crucial here. For example, each land-use type has its level of above- and below-ground carbon per hectare, influencing the choice of where and how much land will be converted. Studies *a*, *f*, *g*, *j*, *l*, and *p* in Table 3.5 are good examples of studies with different land-use modelling approaches.

10. Spatial scale

The spatial scale, both extent and resolution are important to the model outcomes. Extent is often as large as the complete world. However also in regional or local studies the context of the wider world should be taken into account. In the economic part of a model often countries or blocks of countries (having similar conditions) are used as the unit of analysis (i.e. resolution). This is a reasonable approach since most trade data is available at country level and policies on trade are functioning at this level as well. For the biophysical part of the calculation, the location of land-use changes and the physical conditions at these locations are modelled at different resolution in the various models. This can range from averages for countries or regions to detailed grids at the 10km or 1 km level or grid sizes between 0.5 degrees and 5 minutes; approximately 50 to 10 km (Stehfest et al., 2010; Stehfest et al., 2014). Others combine land properties (e.g. soil, land use) to construct a mosaic of units with unique properties. As with all data used, the resolution and quality does influence the outcome of the ILUC calculations. However, it is not said that more detail is always better.

4 Carbon impact of using energy feedstocks from the forestry sector

Key messages:

- Wood taken from forests is a carbon-neutral energy source in the long term, but it takes time before net emission reductions are actually achieved. This is called the carbon debt which is defined as the carbon emitted due to harvesting the bioenergy (e.g. including residues) minus the carbon that would be emitted by the alternative system (mostly fossil energy).
- Based on a modelling exercise with EFISCEN (European Forest Information Scenario Model), harvesting residues for bioenergy produce a relatively small carbon debt and carbon payback times between 2 to 15 years when the wood replaces coal, between 20 and 50 years when it replaces gas, and 5 to 25 years when oil-based transport fuels are replaced.
- Using processing and post-consumer waste wood for bioenergy produces a carbon debt that can be very small in some cases, but this is strongly dependent on the reference situation (e.g. landfills with or without methane capture).
- Using wood from thinning in boreal and temperate forests for bioenergy could produce a significant carbon debt and payback times between 40 and 135 years, when used for replacing coal in power generation. Thinning in forest plantations may have much shorter payback times.
- Additional felling for bioenergy in boreal and temperate forests could result in large carbon debts, requiring payback times of decades, up to more than three centuries.
- For the short term, an efficient climate mitigation measure would be to refrain from additional final felling (for the purpose of bioenergy). In that way more carbon would remain stored in forests and an effective carbon sink would remain intact.
- Wood plantations on agricultural land have very low payback times because of the uptake of CO_2 in the years before the wood is harvested. However, it requires land and therefore LUC emissions have to be taken into account (see Chapter 6).

4.1 Introduction

Forests can act either as a carbon source or sink, depending on the balance between uptake of carbon through photosynthesis and the release of carbon through respiration, decomposition, fires, or removal through harvesting activities. On aggregate, forests are estimated to have acted as sinks over the last decades, on both a European and global scale (Le Quéré et al., 2013). Different types of forest management can influence its carbon balance (Eggers et al., 2007). Forest management activities can influence carbon pools, fluxes and productivity, either directly, for example, by transferring carbon from 'growing stock' to 'product' pools (e.g. through thinning or harvesting), or indirectly, by altering tree growth conditions (e.g. through liming or fertilising). Effects can be immediate (e.g. from thinning) or evolve slowly (e.g. due to fertilisation). Activities may affect current stands (e.g. thinning regime) or future stands (e.g. regeneration), or may be transient (e.g. minimising site preparation). This chapter discusses the possible carbon impacts of harvesting and using wood as a source of bioenergy and is mainly based on a recent report written by PBL (J. Ros, Minnen, and Arets, 2013), which is based on the literature available in June 2013, including reviews of scientific information, and in combination with additional model calculations and analyses. New data or studies published since that date could impact the reported ranges and uncertainties.

4.2 On the carbon dynamics of trees and forests

4.2.1 Tree growth

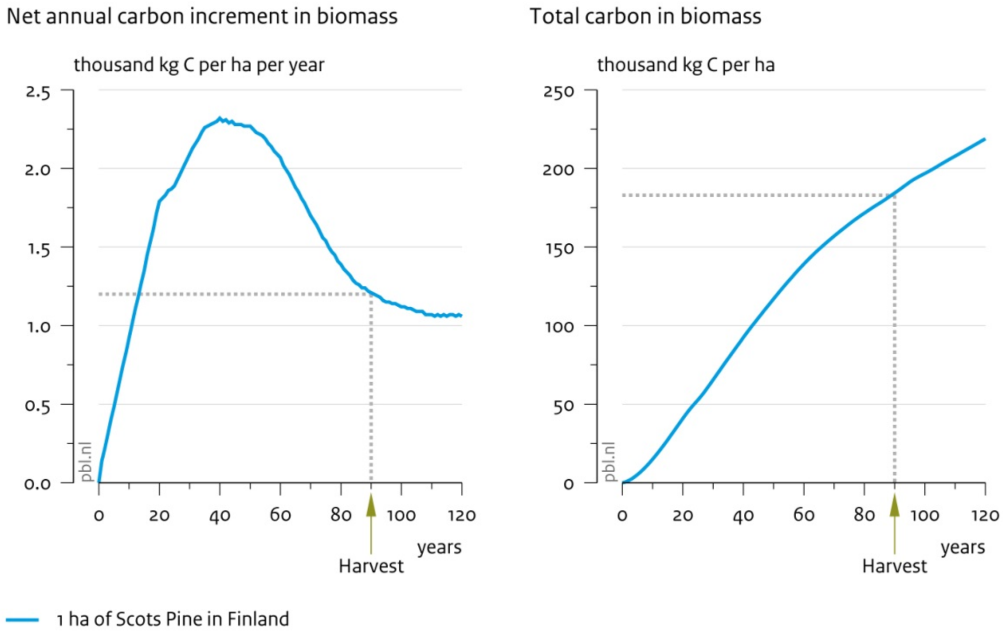
Tree growth is one of the main processes that determine a forest's net carbon sequestration potential. As shown in Figure 4.1, this growth is not constant over time. Small trees in young forest stands sequester relatively little carbon. The rate of net biomass increment in these young forests increases up to a maximum, which is species and site specific. After the peak in growth at intermediary ages, growth rates gradually decrease again. In very old forests, net increment (balance between losses, disturbances and tree mortality, and the growth of individual trees) will further decrease and could, assuming constant atmospheric and climatic conditions, eventually become zero. In Europe this seldom occurs, as forests are usually harvested in rotations of a certain time span, the length of which depends on species, growth rate and the tree size required for the intended purpose (see Table 4.1 for some characteristic values of rotation periods).

Table 4.1 *Examples of European tree species with different carbon dynamics. Source: (J. Ros et al., 2013).*

Species	**Region/ country**	**Total area covered (1000 ha)**	**Growth**	**Rotation (years)**
Sitka spruce	Scotland	800	Fast	40–60
Beech	Germany	1560	Relatively slow	120–140
Norway spruce	Germany	2980	Relatively fast	80–120
Scots pine	Finland	10560	Slow	76–90
Scots pine	Poland	4320	Moderate	80–120
Oak coppice	Bulgaria	540	Slow	60–90
Maritime pine	France	1360	Fast	45–55
Poplar	France	140	Fast	20–25

Timber yields per rotation period are highest when trees are harvested after the peak in growth, when net annual increments would start to stabilise. Harvesting removes the tree stems and most of the branches. Small branches and bark are often left behind because removing them is often (economically) inefficient, and they are also needed to keep enough nutrients in the forest soils to sustain future growth.

Typical carbon increment curves for an even-aged stand of trees



Source: Alterra CO2fix model, 2013; Koivisto, 1959

Figure 4.1 *On the left, typical carbon or biomass increment curve for a single even-aged stand, in this case 1 ha of Scots pine in Finland. In year 0, the stand is established after which trees start to grow. Initially, tree growth is slow, peaking after 30 to 50 years, after which the increment in carbon decreases again. Harvesting time, in the graph, points to the age at which this forest type is usually harvested (at around 90 years). The right-hand graph shows the resulting development of carbon in the same stand, over time. Source: (J. Ros et al., 2013).*

4.2.2 Carbon debt, payback time and the carbon impact indicator

Wood taken from forests is a carbon-neutral energy source in the long term, but it takes time before net emission reductions are actually achieved. In this respect, the term 'carbon debt' or 'greenhouse gas investment' has been introduced. It indicates that if there is a decline in average carbon stock in the bioenergy system, this needs to be overcome before the bioenergy system delivers mitigation benefit. In a way, for any infrastructure (e.g. building a railway line) intended to reduce emissions in the longer term, a greenhouse gas investment is required. The carbon debt depends on two factors:

1. When timber is harvested or forest residues are collected, biomass in the forest decreases. The amount of regrowth that would be needed to recover this decrease

takes time, for example if the growth rates of more mature forests are higher than those in the early stages of a regrowing forest, this temporarily can reduce carbon sequestration capacity, and

2. the amount of fossil fuel emissions displaced by bioenergy. That is, the greenhouse gas displacement factor of the bioenergy system, which reflects the relative greenhouse gas emissions per unit energy, and the relative efficiency of bioenergy versus fossil-fuel systems.

The term carbon debt is frequently used very loosely in the literature. Here we define it as the carbon emitted due to harvesting the bioenergy (e.g. including residues) minus the carbon that would be emitted by the alternative system (mostly fossil energy). A related term is the 'carbon payback time' which is defined as the time it takes for the carbon debt to become zero. This is when the greenhouse gas emissions related to bioenergy minus the CO_2 uptake in the forest due to biomass regrowth equals the greenhouse gas emissions in the alternative system. From that moment on, real emission reductions occur.
Another metric used in this chapter is the 'carbon impact indicator' (CI) to assess the carbon impact of using feedstocks from the forestry sector. CI is the quotient of total carbon losses from a forest and carbon removed through harvesting (adapted from (J. Ros et al., 2013)):

$$CI(t) = \frac{C_{r(t)} - C_{s(t)}}{\sum_{i=startyear}^{t} C_{harvest(i)}} \qquad (\text{-})\ [4.1]$$

Where

$C_{r(t)}$= The total accumulated carbon in all forest carbon pools – i.e. living biomass, standing dead trees, soil carbon – at year *t* of a 'reference' situation, i.e. when no harvesting would take place.

$C_{s(t)}$= The total accumulated carbon in all forest carbon pools at year *t* in a situation *with* harvesting.

$C_{harvest(i)}$= The carbon in the harvested wood in year *i* where different harvesting methods can be considered: final felling, thinning, harvesting residues, salvage logging of dead wood, (see Sections 4.3.2 and 4.3.3).

$\sum_{t=start\ year}^{end\ year} C_{harvest(t)}$ = Total cumulative carbon in the harvested wood for bioenergy between a given start year (for example 2015) and an end year (for example 2030 as in Remap 2030).

The value of CI gives useful information on the carbon dynamics of the forest considered:

CI > 1	Losses from the forest system are larger than the carbon in the harvested woody biomass. Carbon losses occur not only due to wood harvesting, but regrowth after harvesting is also slower than under circumstances without harvesting.
CI = 1	Carbon losses from the forest system are exactly counterbalanced by the carbon in the harvested wood.
CI between 0 and 1	The use of woody biomass results in a net carbon benefit, as the harvested carbon pool is larger than the net carbon losses from forests.
CI = 0	The losses due to wood harvesting is *completely* compensated by forest regrowth. In this case harvesting of carbon has no impact on the sum of all carbon pools in the forest.
CI < 0	Carbon losses due to wood harvesting are even *more* than compensated for by forest regrowth, which in some cases may be the end result of intensified forest management.

4.2.3 Landscape level versus stand level

When assessing carbon balances in forests, a distinction is often made between stand level and landscape level. Stand levels are especially useful for analysing well-defined specific (model) situations and for studying time-dependent processes. In this case, the focus can be from single trees to a small well-defined area of (even-aged) trees. Landscape levels are larger in scale and concern a complete forest or even a whole region. These landscapes may include many different stands with different properties, i.e. different species, age classes and management regimes.

On stand level, the impact on carbon storage and carbon sinks of harvesting and regrowth can be calculated. Because the same operation happens across the entire stand at the same time, the impact is relatively big. On landscape level, the impact of harvesting and regrowth of a tree is the same, but the relative impact on carbon storage is 'diluted' by the growth of all the other trees within this landscape. However, because the carbon sink of all the other trees in the landscape is not changed, the absolute change in carbon sink on the landscape level is the same as on the stand level. Therefore, the impact per unit of bioenergy is not dependent on the scale.

Increased harvesting may still result in increasing carbon stocks, as long as the harvested volumes are lower than the net annual increment. Such increases, however, will result in changes in the equilibrium between harvest and increment and in a decrease in carbon stocks compared to the situation without additional harvesting.

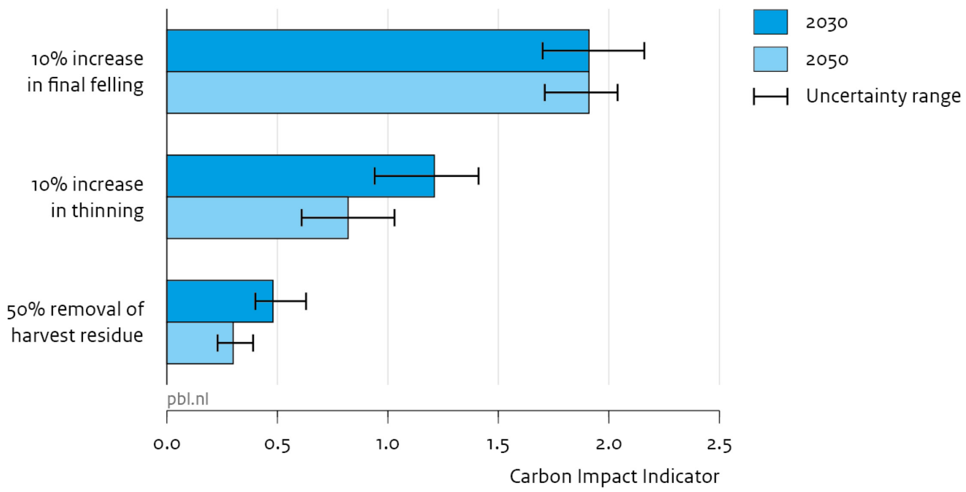
4.3 Carbon impact of different harvesting strategies

4.3.1 Final felling and selective cutting

In Ros et al. (2013), the European Forest Information Scenario Model (EFISCEN) was applied to representative forest types and forest management systems across Europe, with contrasting growing conditions as shown in Table 4.1. The model calculations assumed that, in 2015, wood harvesting would have structurally increased over each defined area. Although calculations were done for European forests only (all with still increasing carbon stocks), results also apply to many other forests at mid and high latitudes around the world, as these are comparable in composition and structure.

In EFISCEN, two different harvest systems are distinguished: 1) final felling, where all trees are harvested in a stand or 2) thinning, where only part of the trees are harvested on a stand. Thinning is a common practice in forestry where a small part of (young) trees are removed to create space for the bigger trees. It is usually performed several times before a forest or plantation is harvested. Wood from thinning is often used in the paper and pulp industry.

Impact of different harvesting practices on the carbon balance



Source: Ros et al, 2013/Alterra Efiscen model, 2013

Figure 4.2 *Impact on the carbon balance – expressed in terms of the Carbon Impact Indicator, see Section 4.2.2 – of different harvesting practices in Europe, starting in 2015 according to the EFISCEN model based on the average of the tree species shown in Table 4.1. The spread in the results is also shown. Results also apply to many other forests at mid and high latitudes around the world, as these are comparable in composition and structure.*

A 10% increase in felling in existing forests – in addition to current practice in a baseline situation – involves the risk of negative impacts on the carbon balance for decades to come. Simulations with the EFISCEN model, assuming one new young tree for every tree harvested[7], show that carbon losses from forests in 2030 and 2050 would still be more than two times higher than the amount of carbon in the additionally harvested wood (Figure 4.2). It may take more than a century to realise a situation with the carbon losses in the forest equal to the carbon harvested. As can be expected, thinning performs better than felling. Over a period of 15 years a 10% increase in thinning as compared to a baseline situation, carbon losses would, on average, still be higher than the carbon in the harvested wood.

4.3.2 Harvesting residues

Harvest residues consist of remnants and portions of trees, such as tree tops, stumps, branches, foliage and pieces of bark, resulting from silvicultural activities (thinning and final felling). Results from the EFISCEN modelling experiment showed that extracting 50% of harvest residues could lead to a positive carbon balance within 5 years following the initial increase in extraction, compared with the baseline situation. By 2030 and 2050, carbon losses from the forest were shown to be considerably lower than the amount of carbon in the harvested wood residue (Figure 4.2).

So it can be concluded that harvesting residues is, in terms of forest carbon dynamics, a preferable strategy. However, their use can conflict with other sustainability criteria (Lamers and Junginger, 2013; Zanchi, Pena, and Bird, 2012). For example, the fraction of dead wood in a forest is one of the indicators for biodiversity (Schuck, Meyer, Menke, Lier, and Lindner, 2004).

As shown in (Lamers and Junginger, 2013) current global wood pellet production is predominantly residue-based. Currently, between 20% and 35% of total felling consists of residues (Mantau et al., 2010). Up to now, these residues often are left in the forest or burned along roadsides, because of their relatively low economic value. As such, forest residue potentially represents a substantial biomass resource that could be used to replace fossil fuel (Repo et al., 2012), even though only a part of it is easily accessible and could be harvested, from an ecological and economic perspective (Lippke et al., 2011).

Despite the considerable overall potential, residues should only be partially removed, to ensure soil fertility can be maintained although an option could be to return the ashes from wood combustion (Agostini, Giuntoli, and Boulamanti, 2013). The percentage that could be removed depends on soil type and fertility, local conditions and climate. EEA (2007) assumes removal rates for various soil types, varying from 15% to 75%.

4.3.3 Salvage logging

Salvage logging is a potential source of biomass for energy. Salvage logging refers to the removal of damaged and dead stems, due to for example storms, forest pathogens, insects and diseases. Dead wood includes wood lying on the forest floor (which otherwise would not be extracted), roots, and large stumps. Dead wood that remains in the forest has clear biodiversity benefits, but large amounts of dead wood may increase the risk of forest fires. According to the Food and Agriculture Organization (FAO) of the United Nations, the current global amount of dead wood is estimated at 67 Gt, although this Figure is only a rough

[7] Carbon uptake might be enhanced by planting more than one tree of rapidly growing species in short rotations of 10 to 15 years.

estimate and will vary in time. It equals about 11% of the total global biomass demand (FAO, 2010). Regions with large amounts of dead wood are located in Russia and parts of Africa.

On a global level, close to 40 million hectares of forest are adversely affected by insect infestations and diseases, annually, but not all of these areas are equally accessible. The Mountain Pine Beetle in western North America is of special concern, because of the unprecedented magnitude of the infestations. Since the late 1990s, the beetle has devastated more than 11 million hectares of forest in Canada and the western United States, and it is still spreading today. In British Columbia, by 2012, the infestations had killed an estimated 710 million m^3 of commercially valuable pine timber (IINAS, 2012). Some of this dead wood could be used in energy production (Lamers, Junginger, Dymond, and Faaij, 2013), resulting in a positive climate effect. Removing the dead trees would enable regrowth and/or replanting, thus increasing the average growth rate of the forest. If this wood would otherwise be burned at the roadsides or be left in the forest without valorisation of its energy content, then any bioenergy alternative would be beneficial to the climate. (Lamers and Junginger, 2013) showed this for beetle-impacted pine forests in British Columbia. An important limitation to the use of salvaged wood from beetle-infested mountain pine forests is the high costs associated with future harvests, as accessibility decreases and transport strongly increases (Niquidet, Stennes, and Van Kooten, 2012).

4.3.4 Waste wood

There are two types of waste wood: waste originating from industrial processing of wood into various products and wood coming from the end of life of its various uses. Most of the waste from the first category is already used to produce energy, to a large extent in the industry itself. Wood can be used as a building material, for all kinds of products (e.g. furniture), as well as for paper and cardboard. The carbon contained in these products remains effectively stored during their lifetimes, which vary from 1 to 10 years for most paper products, and between 20 and more than 100 years for some building materials. Even if the wood is burned in the end, the delay of the emission of the carbon that was temporarily stored in these products and materials can be quite relevant. If the carbon is stored in products that last for about 10 years, the impact of the related emissions on global warming 100 years from now will be reduced by almost 10%. If stored for 40 years, the impact will be reduced by about 30% (Cherubini, Guest, and Strømman, 2012), compared to the impact of an immediate CO_2 emission at the time of harvesting.

In practice, the use of wood can be optimised by the 'cascading principle', whereby the same wood is used in several successive applications. This is not only the case in paper recycling; wooden materials also can be recycled. Finally, waste wood and other woody residues from industry and households can be used for energy or, possibly, in the chemical industry. Burning the woody materials is the easiest way to use them. However, producing green polymer (e.g. polyethylene) from monomers in the chemical industry, or liquid and gaseous biofuels in the transport sector, or 'green' gas in various applications, may be more advantageous, because of a likely lack of low-carbon alternatives, in the coming decades, in these sectors. For those types of applications, more advanced technologies than incineration (e.g. gasification or fermentation) need to be implemented. The carbon conversion efficiency (carbon from the biomass that ends up in the product) is about 50% to 60%, whereas carbon capture and storage or reuse would be an option that eliminates most of the emissions from industrial processes.

Although, theoretically, the cascading principle is an attractive one, an optimal application of this principle requires that the demand for bioenergy becomes attuned to the use of wood as a resource material. Furthermore, our current society also is a carbon sink. More wooden materials enter the societal system than leave it as waste. An increase in the share of waste wood in the energy system, therefore, requires patience.

The emission reduction achieved by using waste wood for energy is determined by the emission levels of the various alternatives, such as using incineration, landfill or composting. In case of waste incineration used for generating energy, the replacement of fossil fuels already leads to emission reductions. They are being realised in many European waste incineration plants, today, but a higher level of reduction could be reached by developing more efficient installations for processing the waste wood.

In the landfill option, some parts of the wood (cellulose and hemicellulose) can be degraded under the anaerobic conditions found in landfills. In practice, landfills serve as an effective carbon stock, because even after long periods of time most of the woody materials are still present in the landfill. Overall, between 25% and 35% of the carbon in woody forest products in landfills (consisting of large amounts of paper) is emitted (Mann and Spath, 2001). For solid pieces of wood within the waste, only a few per cent of the carbon would be released, even after many decades (Wang, Padgett, Cruz, F.B., and Barlaz, 2011; Ximenes, Gardner, and Cowie, 2008). Part of the carbon in the decaying wood will be released as methane, a strong greenhouse gas. The amount of methane emissions is strongly determined by local circumstances, such as moisture content, temperature and anaerobic conditions. In practice, in many cases, 50% to 60% of the carbon is released in the form of methane (Mann and Spath, 2001). In terms of CO_2 equivalent emissions this is far *more* than would be emitted when burnt. In some cases, methane is (partly) recovered, especially in the first 5 to 20 years, significantly reducing these emissions.

4.3.5 Forest plantations

When forests are planted, CO_2 uptake starts immediately but it also requires land, and the impact of direct or indirect land-use change has to be included in calculations of the carbon balance, similar to that related to biofuel production based on agricultural crops. Although, for the latter, CO_2 emissions from indirect land-use change may be somewhat lower, as the carbon stock in forests is generally greater than in agricultural crops. Indirect deforestation elsewhere also cannot be excluded. No (model) analysis is currently available that quantifies the overall and especially the indirect effect for forest plantations.

It will take a while before new forests are able to provide wood as a resource for bioenergy. This period largely depends on the type of tree species and its rotation period. If, for example, relatively fast growing or short-rotation species (SRC) are selected, such as willow or eucalyptus, biomass becomes available relatively soon and on a regular basis, see also (EEA, 2007). Multiple studies have shown that, on average, wood production in willow plantations is in the range of 6 to 15 tonnes dry matter per hectare per year (in energy terms, between 110 and 275 GJ/ha per year), harvested over 2 to 5-year cycles, e.g.(Elbersen et al., 2013). The production range depends on location (production levels are lower in high-latitude countries) and, especially, on management intensity. High production levels are only possible if plantations are grown on fertile (agricultural) land and with a high level of management (Elbersen et al., 2013; Werner et al., 2012).

Such short-rotation cultivation is similar to an agricultural activity and its level of sustainability should be judged in the same way, including the effects of indirect land-use change (ILUC, see Chapter 3), and considering the specific carbon stocks on such a plantation. As for other energy crops, willow plantations on marginal or degraded land (with production levels of around 6 to 9 tonnes dry matter per hectare per year) may be an interesting sustainable option to produce woody biomass, but as business cases these are generally not very attractive, and it is difficult to formulate effective and enforceable criteria.

The picture is different when natural forests are converted into fast-growing plantations (Mitchell, Harmon and O'Connell, 2012), because the carbon stored in the original vegetation will be lost. Wood production levels for bioenergy may be still high (although less than plantations on agricultural land, as forested lands are often less fertile). However, the compensation of carbon losses due to the conversion could require a considerable period of time. For example, typical above-ground biomass pools in natural boreal and temperate forests contain, on average, about 60 and 150 tonnes dry matter per hectare, respectively (FORM, 2013; IPCC, 2003), which would equal a period of more than 10 years to compensate for the related carbon losses. A situation where more carbon is stored than is lost will seldom be reached in the short term (Agostini et al., 2013; Zanchi et al., 2012). Furthermore, these conversions often have considerable negative effects on other ecosystem goods and services, such as biodiversity (Brockerhoff, Jactel, Parrotta, ·, and Sayer, 2008).

4.4 Payback times

Figure 4.3 gives an overview of the carbon payback times (for a definition, see Section 4.2) for power generation as reported in the literature and by the EFISCEN modelling experiment discussed in the previous section. The literature shows a considerable range in payback times. The reasons for this wide range are the following:

- The replacement of fossil fuel. Replacing coal by wooded biomass has a significantly shorter payback time than when replacing natural gas.
- Wood characteristics, such as moisture content.
- The forest species and residue type considered.
- Current and future forest growth rates. Using wood from relatively young, still fast growing forests is less attractive. Given the fact that European forests are often in this phase, it would be more efficient, from the perspective of emission reduction, to leave the trees in the forest than to harvest them for energy.

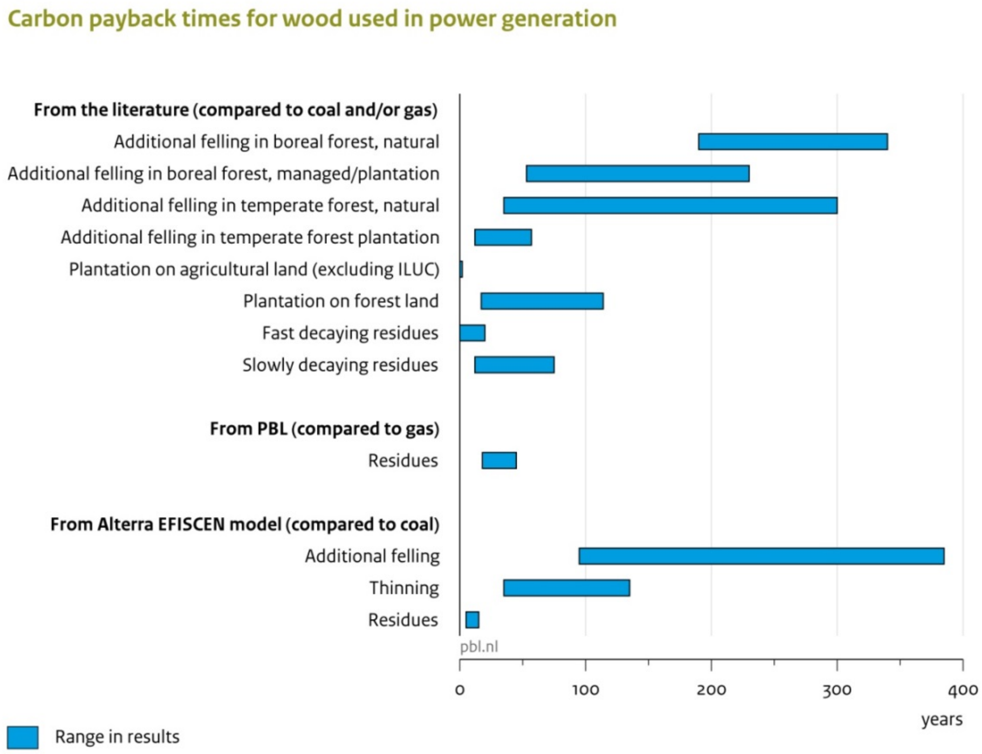
- Management may increase forest growth rates and, thus, may shorten payback times, compared to those of unmanaged forests, see (Agostini et al., 2013).
- For forest residues, particular and additional factors determine the payback time, due to differences in pool sizes, types of residues (fast decaying bark, twigs and leaves or slowly decaying dead stem wood or stumps), and alternative residue use (natural decay leads to longer payback times than when residues are burned on site without the energy being used).

These results can be compared with those from the EFICSCEN modelling experiment for European forests (Figure 4.3) and for the use of residues and woody waste (Figure 4.4), in general. They are of the same order of magnitude. Payback times are long for felling or even thinning of European forests that take place over the coming decades. If thinning is done to harvest more wood for bioenergy, payback time calculation also include the impact on forest growth. If thinning is carried out as an essential part of forest management, in order to produce the required wood quality, the wood harvested thus can be considered a residue, but in actual practice it is often applied for other uses.

Figure 4.3 shows that for the coming decades, the risk of negative impacts is high if additional felling in existing forests is used for the production of bioenergy assuming no substantial change in management, e.g. (Agostini et al., 2013; Zanchi et al., 2012). Payback times are longer in boreal regions than in temperate latitudes/regions. (Holtsmark, 2012), for example, mentions a payback time of 190 to 340 years for boreal forests. For natural forests in temperate regions, the range is between 35 and 300 years (Colnes et al., 2012; McKechnie, Colombo, Chen, Mabee, and MacLean, 2011; Zanchi et al., 2012). Payback times are substantially shorter if wood is used from forest plantations instead of from natural forests (Agostini et al., 2013; Jonker, Junginger, and Faaij, 2014).

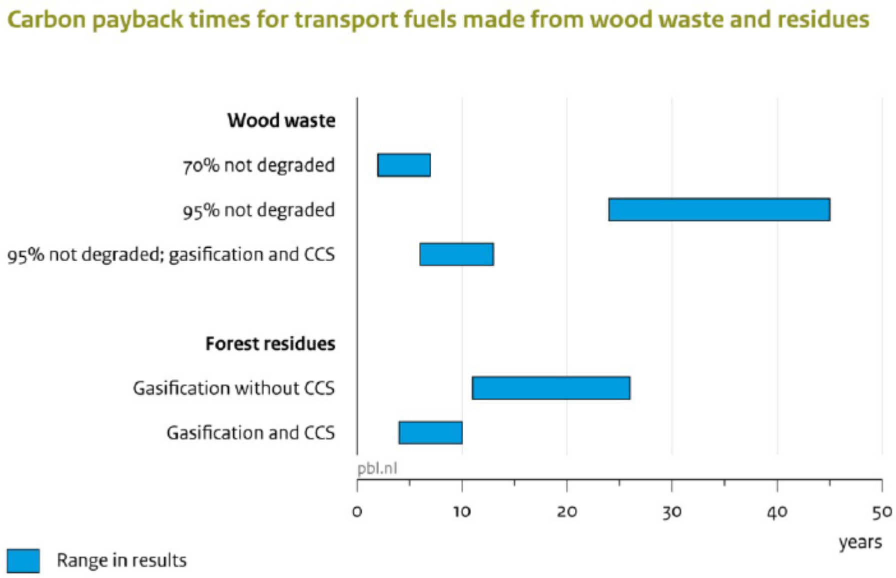
The figure clearly shows that establishing plantations on (marginal) agricultural land is effective under nearly all circumstances, if this is not causing any ILUC effects (see Chapter 3). For example, willow wood has a payback time of only a few years when grown on marginal lands and not causing any indirect land-use change. This is because of the high production level and short rotation cycles, e.g. (Elbersen et al., 2013; Tsarev, 2005).

The payback time for waste wood (Figure 4.4) is strongly dependent on the assumptions concerning the reference situation, especially if it would be a landfill. The fate of the waste, especially the methane emissions due to degradation in the landfill determines the payback time. In case of 5% degradation and 50% of it resulting in methane emissions, the payback time would be 30–50 years. In case of 30% degradation it would be about five years. Even if only a few per cent of methane would be captured, this would reduce the payback time considerably. If the reference situation is incineration without any use of the energy, the payback time is zero (not shown).



Source: Walker, 2010; Holsmark, 2011; McKechnie et al., 2011; Colnes et al., 2012; Mitchel et al., 2012; Repo et al., 2012; Zanchi et al., 2012; Jonker et al., 2013; JRC, 2013; Lamers et al., 2013; Lamers and Junginger, 2013; Alterra EFISCEN model, 2013; PBL, 2013

Figure 4.3 *Ranges of carbon payback times for wood used in power generation.*



Source: PBL, 2013

Figure 4.4 *Payback times calculated for wood waste (compared to storage in landfill) and residues used in the production of transport fuels (based on gasification)*

5 REmap 2030 in perspective

Key messages:

- The bioenergy demand and potential supply estimates underpinning the REmap 2030 study fall more or less within the ranges published by IPCC, PBL and others at the level of global totals.
- Despite this, the REmap projections must be considered ambitious and attainable only under favourable conditions and strong policies mainly because the ranges published in other studies are compiled for a more distant future (2050 instead of 2030 in REmap).
- While global totals seem to be in reasonable accordance, the underlying details by supply category and at country level is ambitious for some cases.

5.1 Technical potential and deployment levels according to IPCC

The inherent complexity of biomass resources makes the assessment of their combined technical potential controversial and difficult to characterise.

Table 5.1 shows that the global technical potential for a number of categories of land-based biomass supply for energy production based on an extensive literature review goes from less than 50 EJ all the way up to more than 1000 EJ (Chum et al., 2011). The technical potential considers the limitations of the biomass production practices assumed to be employed and also takes into account concurrent demand for food, fodder, fibre, forest products and area requirements for human infrastructure.

Narrowing down the technical potential of the biomass resource to precise numbers is not possible. In summary, Chum et al. (2011) conclude that the potential depends on a number of factors that are inherently uncertain and will continue to make the long-term technical potential unclear. Important factors are population and economic/technology development and how these translate into fibre, fodder and food demand (especially share and type of animal food products in diets) and development in agriculture and forestry. Additional important factors include:

1. climate change impacts on future land use including its adaptation capability,
2. considerations set by biodiversity and nature conservation requirements, and
3. consequences of land degradation and water scarcity.

Studies point to residue flows in agriculture and forestry and unused (or extensively used) agricultural land as an important basis for expansion of biomass production for energy, both in the short term and in the longer term. Consideration of biodiversity and the need to ensure maintenance of healthy ecosystems and avoid soil degradation set bounds on residue extraction in agriculture and forestry. Grasslands and marginal/degraded lands are considered to have potential for supporting substantial bioenergy production, but biodiversity considerations and water shortages may limit this potential. The possibility that conversion of such lands to biomass plantations reduces downstream water availability needs to be considered.

Table 5.1 *Global technical potential overview for a number of categories of land-based biomass supply for energy production. Source: (Chum et al., 2011).*

Biomass category	Comment	2050 Technical potential (EJ/yr)
Category 1. **Residues from agriculture**	By-products associated with food/fodder production and processing, both primary (e.g., cereal straw from harvesting) and secondary (e.g., rice husks from rice milling) residues.	15 – 70
Category 2. **Dedicated biomass production on surplus agricultural land**	Includes both conventional agriculture crops and dedicated bioenergy plants including oil crops, lignocellulosic grasses, short-rotation coppice and tree plantations. Only land not required for food, fodder or other agricultural commodities production is assumed to be available for bioenergy. However, surplus agriculture land (or abandoned land) need not imply that its development is such that less total land is needed for agriculture: the lands may become excluded from agriculture use in modelling runs due to land degradation processes or climate change (see also 'marginal lands' below). Large technical potential requires global development towards high-yielding agricultural production and low demand for grazing land. Zero technical potential reflects that studies report that food sector development can be such that no surplus agricultural land will be available.	0 – 700
Category 3. **Dedicated biomass production on marginal lands**	Refers to biomass production on deforested or otherwise degraded or marginal land that is judged unsuitable for conventional agriculture but suitable for some bioenergy schemes (e.g., via reforestation). There is no globally established definition of degraded/marginal land and not all studies make a distinction between such land and other land judged as suitable for bioenergy. Adding categories 2 and 3 can therefore lead to double counting if numbers come from different studies. High technical potential numbers for categories 2 and 3 assume biomass production on an area exceeding the present global cropland area (ca. 1.5 billion ha or 15 million km²). Zero technical potential reflects low potential for this category due to land requirements for, for example, extensive grazing management and/or subsistence agriculture or poor economic performance if using the marginal lands for bioenergy.	0 – 110
Category 4. **Forest biomass**	Forest sector by-products including both primary residues from silvicultural thinning and logging, and secondary residues such as sawdust and bark from wood processing. Dead wood from natural disturbances, such as fires and insect outbreaks, represents a second category. Biomass growth in natural/semi-natural forests that is not required for industrial roundwood production to meet projected biomaterials demand (e.g., sawn wood, paper and board) represents a third category. By-products provide up to about 20 EJ/yr implying that high forest biomass technical potentials correspond to a much larger forest biomass extraction for energy than what is presently achieved in industrial wood production. Zero technical potential indicates that studies report that demand from sectors other than the energy sector can become larger than the estimated forest supply capacity.	0 – 110
Category 5. **Dung**	Animal manure. Population development, diets and character of animal production systems are critical determinants.	5 – 50
Category 6. **Organic wastes**	Biomass associated with materials use, for example, organic waste from households and restaurants and discarded wood products including paper, construction and demolition wood; availability depends on competing uses and implementation of collection systems.	5 – >50
Total		**<50 – >1000**

Notes: Based on Fischer and Schrattenholzer (2001); Hoogwijk et al. (2003, 2005, 2009); Smeets and Faaij (2007); Dornburg et al. (2008, 2010); Field et al. (2008); Hakala et al. (2009); IEA Bioenergy (2009); Metzger and Huttermann (2009); van Vuuren et al. (2009); Haberl et al. (2010); Wirsenius et al. (2010); Beringer et al. (2011).

Based on this considerations and an expert review of available scientific literature, (Fischedick et al., 2011) estimate that potential deployment levels of biomass for energy by 2050 are in the range of 100 to 300 EJ (see Figure 5.2). This coincides with a scenario review conducted in Chapter 10 of the same report indicating that by 2050, in the median case bioenergy contributes 120 to 155 EJ to global primary energy supply, or 150 to 190 EJ for the 75th percentile case, and up to 265 to 300 EJ in the highest deployment scenarios.

5.2 Availability of biomass according to PBL

5.2.1 Expert judgement

Recently PBL developed an infographics on biomass (PBL, 2014) containing PBL's expert judgement on the availability of sources of biomass on a global scale in 2050. Three projections are distinguished: low, middle and high. In the 'low' projection potential land-based biomass supply is 50 EJ, 145 EJ in the 'middle' case and in the high projection it more

than doubles to 310 EJ (see Figure 5.1).There is also an aquatic potential of 5 EJ in the 'middle' projection and even 90 EJ in the 'high' case. Aquatic biomass includes all biomass growing in the aquatic environment (fresh water and saline), such as fish, seaweed and algae. This type of biomass can be from oceans, seas, lakes or rivers, but increasingly more often from specific aquaculture. These compounds have a high market value and their production does offer opportunities, but the technology is still at an experimental stage, and the possibilities for large-scale production for energy are very uncertain. Therefore, in this study, we exclude the potential from aquatic sources.
In the PBL estimate, wood production refers to both natural forests and forest plantations, supplemented by fast-growing types of wood, such as willows, grown on land no longer used in agriculture. Because the types of energy crops that will be chosen in the future are as yet uncertain, the distribution of agricultural land for fast-growing grass and fast-growing wood, in percentages, is kept at a 50:50 ratio.
A strong downward trend is expected in the use of woody biomass as a traditional energy source. For the future, it is furthermore expected that certain sustainability criteria also will be applied to woody biomass. It is, however, also expected that a limited amount of wood can always be harvested in an acceptable, sustainable way.

The 'low' estimate of PBL is based on pessimistic assumptions:

- policies are aimed at no further stimulation of energy from conventional food crops such as rapeseed, oil palm, sugar cane, maize and wheat (see Chapter 2) and therefore the production level remain at the current level of around 5 EJ,
- using additional land for growing wood (forest plantations) is regarded unsustainable,
- residues (branches, tree tops, dead trees) are left behind in the forest or are burned on location because taking them out is not considered economically viable,
- wood construction, demolition and furniture waste is assumed to remain at the 2010 level,
- crop yields hardly increase and sustainability criteria (ILUC) prohibit the expansion of agricultural land for bioenergy crops.
- the harvesting of residues (stalks, straw) will be utilised to a limited extent only, because it is not attractive enough from an economic perspective and has significant competition with other uses,
- regarding agricultural waste (i.e. losses during the transportation, storage, processing and consumption of food), the emphasis is on avoiding losses and utilisation as animal feed which result in a 'low' estimate of 20 EJ,
- using additional land for growing wood is regarded unsustainable,
- the use of wood from various types of forests is assumed at 10 EJ. The maximum – under the 'high' scenario – is assumed at 35 exajoules.

The 'high' estimate of PBL, which coincides with the upper range of IPCC's deployment level of 300 EJ, is based on optimistic assumptions:

- advanced biofuels will become important, i.e. biofuels made from fast-growing grasses, such as miscanthus, or fast-growing trees, such as willows and poplars. This will provide a potential of 75 EJ in energy (and 20 EJ in the medium case).
- the use of wood from various types of forests is 35 EJ,
- the demand for wood in the form of wooden products and as materials is expected to increase up to 25 EJ (OECD, 2012). Combined with an optimal utilisation of forest residues this result in a potential of 25 EJ,
- the demand for paper, wooden products and construction materials is expected to increase by 35% up to 2030, and 58% up to 2050 (OECD, 2012) and, therefore, more wood waste will become available. The 'high' scenario for 2050 assumes an increasing waste flow, equalling the amount of input in 2030, all of which is expected to be used in energy generation (20 EJ),
- an agricultural land area of 6 million km^2 (0.6 billion ha) which is in line with studies that assume an increase in agricultural productivity that is more or less equal to that of the past decade.
- the yield in energy crops per square kilometre is assumed to be 1.5 times higher than of current forest plantations. This would deliver around 150 EJ in potential energy, distributed over agriculture (80 EJ) and forest plantations (70 EJ),
- nearly all residues will be gathered and utilised for energy (around 30 EJ) under the only restriction that a certain amount of residues is assumed to be left on the land to maintain soil quality,
- a growth in production and improved utilisation of the largest part of all waste flows result in a maximum potential of around 45 EJ.

Studies that result in an even higher potential often assume production levels of agricultural crops will more than triple, in the long term. This would reduce the amount of land required for food crops and leave a large amount of land available for biomass crops. In addition, they also assume the yield of the biomass crops to increase substantially. We do not consider this a realistic scenario.

5.2.2 Marginal lands

As shown in Table 5.1, according to IPCC, potential biomass production on marginal land (category 3) is between 0 and 110 EJ. Zero technical potential reflects that marginal land is required for extensive grazing management and/or subsistence farming or it reflects poor economic performance if using the marginal lands for bioenergy. High potential assume biomass production on an area exceeding the present global cropland area of 1.5 billion hectares. However, they point out that there is no globally established definition of degraded/marginal land and that not all studies make a distinction between such land and other land judged as suitable for bioenergy. Therefore the high estimate from category 2 (biomass on surplus agricultural land) cannot be combined with the high estimate of category 3. This would result in double counting.

In 2011, PBL performed a study with the integrated assessment model IMAGE to estimate the global potential of energy crop production on degraded lands using detailed, spatially explicit data about the area, type and extent of degradation derived from the Global Assessment of Land Degradation data set, and by combining this data set with various spatially explicit data sets (Nijsen, Smeets, Stehfest and Van Vuuren, 2012). Next, an estimate was made of the possible yield of perennial energy crops on the degraded areas as a function of the type and degree of degradation. Lightly degraded areas were not included, as these areas might be suitable for conventional food production. The total global potential energy production on degraded lands was assessed to be slightly above 150 and 190 EJ, for grassy and woody energy crops, respectively. Most of this potential is on areas currently classified as forest, cropland or pastoral land, leaving a potential of around 25 and 32 EJ on other land cover categories, mostly grassland and savanna . Most of the potential energy crop production on degraded land is located in developing regions. China has a total potential of 30 EJ, of which 4 EJ from areas classified as other land. Also the United States, Brazil, western Africa, eastern Africa, Russia and India have a substantial potential of between 12 and 18 EJ, with up to 30% of the potential from areas classified as 'other land'. However, this global potential of 25–32 EJ cannot simply be added to the expert judgements presented in the previous section because it is not clear to what extent this would imply double counting. Also, (Chum et al., 2011) emphasise that main challenges in relation to the use of marginal and degraded land for bioenergy include (1) the large efforts and long time periods required for the reclamation and maintenance of more degraded land; (2) the low productivity levels of these soils; and (3) ensuring that the needs of local populations that use degraded lands for their subsistence are carefully addressed.

5.3 REmap 2030 demand and supply in perspective

In REmap 2030, demand and supply are presented for 26 countries separately (representing three quarters of the total final global energy demand) and for the 'Rest of the world'. Figure 5.1 summarises the demand in REmap 2030 subdivided in five broad demand sectors. The reference case reflects the global energy use in 2030 if current and planned government policies and targets are achieved. The REmap 2030 case reflects the implementation of all REmap options, where modern bioenergy would represent 60% of the global renewable energy use in 2030, resulting in a demand of 93 EJ. Note that we used the REmap data of 30 June 2015.

In primary energy terms this demand translates into a supply of around 110 EJ, assuming optimistic conversion efficiencies of 50% for biofuels (to large extent used in transport), 55% for biomethane (mainly used in power generation) and 100% for heat. In Figure 5.2 we compare this primary energy demand in terms of the supply sources with the global 'low' and 'high' potential supply estimates from REmap 2030, PBL ('low', 'middle' and 'high', see Section 5.2) and the low and high deployment levels from IPCC (see Section 5.1). To put these numbers in perspective, global biomass used for energy in 2010 amounts to approximately 50 EJ per year (see Figure 5.1), and all harvested biomass used for food, fodder, fibre and forest products, when expressed in equivalent heat content, equals 219 EJ. In other words, the entire current global biomass harvest would be required to achieve a 200 EJ deployment level of bioenergy by 2050.

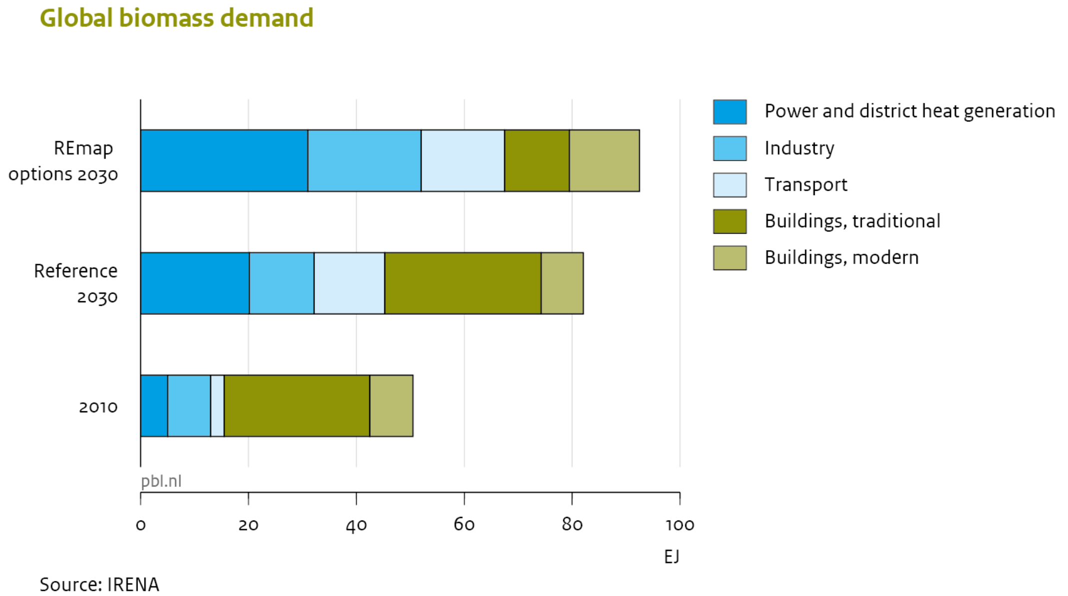


Figure 5.1. *Sectoral global bioenergy demand in REmap 2030.*

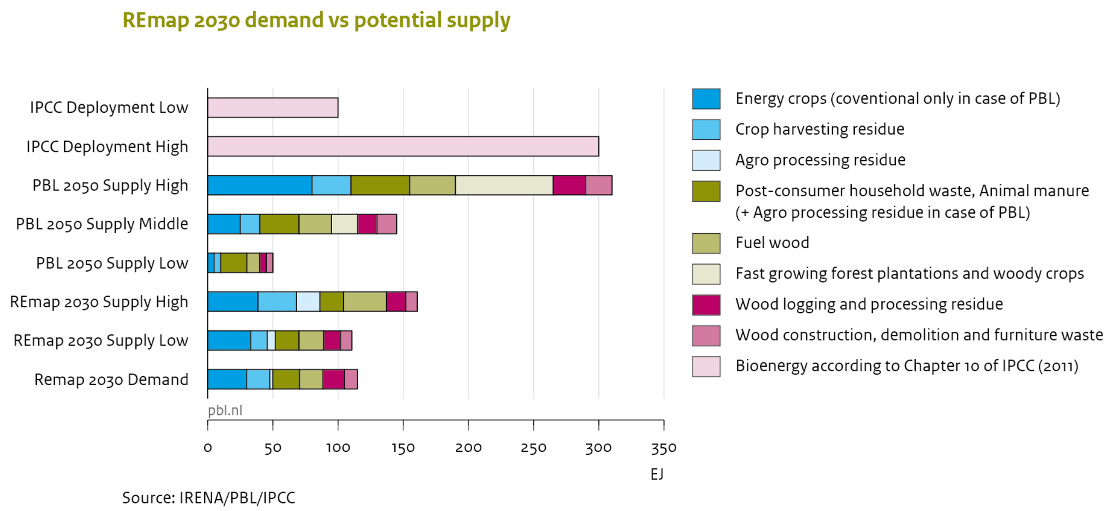


Figure 5.2 *Global REmap 2030 demand in terms of source categories versus global potential primary supply estimates from REmap 2030, PBL (see Section 5.2) and from the Special Report on Renewable energy (IPCC, 2011).*

When comparing the REmap demand with the PBL potential supply estimates, it shows that the 'low' estimate of PBL is by far too low to cover the REmap demand, but it compares well with PBL's 'middle' estimate and it is only 17% higher than the low deployment level of IPCC. As indicated, the 'low' estimate of PBL is pessimistic and thus, at the global level REmap's demand seems to be reasonable. The global potential primary biomass supply in REmap

2030 is between 110 EJ (low) and 160 EJ (high), see Figure 5.2. Again, it fits well within the range as depicted by PBL and IPCC.

However, a key aspect is that the potential supply estimates of PBL and IPCC are related to the year 2050. REmap has a much shorter time horizon of 2030, only 15 years from now and therefore for several source categories it is questionable or even impossible that the 2050 potential can become available twenty years earlier. In the following paragraphs we will discuss the limits to the availability of the various source categories of REmap 2030 in more detail.

5.3.1 Energy crops

The category 'biomass from surplus non-forest land' in REmap 2030 refers to the supply from first generation and advanced woody energy crops and is estimated between 33 and 38 EJ. The designation 'surplus non-forest land' suggests these crops are grown on areas that do not imply LUC-effects. In reality, bioenergy production will often compete with (growing) food and feed production for the same productive areas. In other words, there could be LUC-effects unless strict policies are implemented to avoid this. As also stated in various literature (e.g., Slade, Bauen, & Gross, 2014), terms like 'abandoned agricultural land' and 'surplus land' are prone to misinterpretation and should be used with care. In the PBL estimate, the source category 'agricultural production' is estimated at between 5 and 80 EJ (Figure 5.2), but this covers conventional energy crops only. Advanced biocrops and fast-growing trees, such as willows and poplars, are covered by a separate source category – 'fast growing forest plantations and woody crops' in Figure 5.2 – which is estimated between 0 and 75 EJ, and a middle estimate of 20 EJ. So, the PBL estimate results in a very broad range of 5 to 155 EJ, with a middle estimate of 45 EJ (25 EJ from conventional crops and 20 from forest plantations and advanced crops). This large range is partly explained by differing views on sustainability. To what extent are we ready to allow competition with food production? Should energy crop production take place on vacant agricultural land and should it exclude areas with degraded soil or water scarcity, or areas with large amounts of carbon stored in the vegetation and soil (peat)?

The middle estimate is higher than both the demand (30 EJ) and the supply (33–38 EJ) of REmap 2030, but, as indicated, it is assumed to be achieved in 2050, not 2030, and the low estimate of PBL is far below the demand in REmap 2030. To get a better insight in the likelihood or bottlenecks in achieving a global production from biocrops of 30 EJ within the next 15 years, we performed an analysis of the REmap demand and supply figures.

Table 5.2 contains an optimistic estimate of the agricultural land expansion in 2030 for the REmap countries and the rest of the world, based on the assumption that the demand for liquid biofuels in transport as shown in the columns under 'REmap demand' is covered by the 'biomass from surplus non-forest land' and that 50% is conventional liquid biofuels produced from the main biofuel crop today ('Main crop' in Table 5.2) and the remainder advanced biofuels (i.e. from woody crops). The REmap low and high supply estimates (columns three to six) refer to the agricultural production in PJ/yr *before* conversion to liquid biofuels, i.e. the raw yields.

Table 5.2 *Agricultural expansion in REmap 2030 in historical perspective. Colours indicate difference with historical trend, from large (red) to small (green). It is assumed that 50% of potential supply and demand is cultivated as 'main crop' and 50% as woody biomass (Details see text).*

	Main Crop	**Supply low**		**Supply High**		**REmap demand**		**Agr 2012**	**Expansion comp. to 2012**			**Historical comp. to 2012**		
		[PJ/yr]	kHa	[PJ/yr]	kHa	[PJ/yr]	kHa	kHa	Supply low	Supply high	Demand	1982	1992	2002
Australia	Wheat	1025	10244	1149	11482	754	7539	405474	2.5%	2.8%	1.9%	18%	15%	10%
Brazil	Sugar cane	6888	33178	8235	39664	5307	25562	275605	12%	14%	9.3%	-17%	-10%	-3.5%
Canada	Maize	0		0		398	4637	65346			7.1%	1.4%	3.8%	3.3%
China	Wheat	0		0		1716	17153	515361			3.3%	-14%	0%	0.8%
Denmark	Wheat	7	74	16	156	45	446	2624	2.8%	5.9%	17.0%	10%	5.0%	1.6%
Ecuador	Palm oil	4	21	24	123	116	595	7507	0.3%	1.6%	7.9%	-7.3%	5.9%	-0.2%
France	Rapeseed	988	6677	1270	12982	940	9614	28839	23%	45%	33%	9.7%	5.4%	3.0%
Germany	Rapeseed	527	5393	879	8987	852	8716	16664	32%	54%	52%	10%	1.7%	1.8%
India	Wheat	0		0		350	3499	179300			2.0%	0.8%	1.1%	0.7%
Indonesia	Palm oil	0		0		2693	13811	56500			24%	-33%	-27%	-15%
Italy	Wheat	212	2115	245	2450	518	5175	13729	15%	18%	38%	28%	16%	11.3%
Japan	Wheat	237	2365	261	2611	506	5056	4549	52%	57%	111%	32%	23%	4.7%
Malaysia	Palm oil	0		0		49	249	7750			3.2%	-34%	-11%	-9.8%
Mexico	Wheat	0		0		222	2224	106705			2.1%	-7.9%	-0.5%	-0.1%
Morocco	Wheat	0		0		0	0	30403			0%	-4.0%	0.7%	-0.4%
Nigeria	Wheat	0		51		179	1789	72000			2.5%	-33%	-10%	-1.9%
Russian Fed.	Wheat	846	8456	1047	10470	295	2952	214350	3.9%	4.9%	1.4%		3.4%	1.1%
Saudi Arabia	Wheat	0		0		0	0	173390			0%	-100%	-29%	0.2%
South Africa	Wheat	0		179		220	2199	96341			2.3%	-2.4%	0.1%	1.8%
South Korea	Wheat	56	562	59	587	260	2599	1788	31%	33%	145%	25%	19%	7.3%
Turkey	Wheat	41	406	115	1149	221	2208	38407	1.1%	3.0%	5.7%	-3.2%	3.9%	7.3%
UAE	Wheat	0		0		0	0	397			0.0%	-43%	-15%	44%
Ukraine	Wheat	343	3433	349	3487	115	1151	41297	8.3%	8.4%	2.8%		1.5%	0.2%
UK	Wheat	511	5110	624	6239	84	840	17182	30%	36%	4.9%	6.4%	5.2%	-1.2%
US	Maize	6569	76556	7475	87120	3604	42005	408707	19%	21%	10%	5.6%	4.1%	1.1%
ROW	Wheat	14823	148156	16575	165669	10471	104654	2141993	6.9%	7.7%	4.9%		-2.0%	-1.8%
Total world		**33077**	**302746**	**38553**	**353176**	**29916**	**264674**	**4922207**	**6.2%**	**7.2%**	**5.4%**	**-5.3%**	**-1.0%**	**0.1%**

We estimated the hectares (in kHa) that would be needed to produce those amounts using the raw yields from the fourth column of Table 2.4. The column 'Agr 2012' in Table 5.2 refers to the total agricultural area in the country or region considered. The column 'Expansion comp. 2012' indicates the relative agricultural area expansion compared to 2012 needed to cover the REmap low and high supply and the demand in 2030 as shown in the preceding columns. In the columns under 'Historic comp. to 2012' the historic agricultural area is compared to the agricultural area in 2012. For example, '10%' indicates that the agricultural area in that year was 10% larger than in 2012. A negative number indicates it was smaller.

So, for example, in Brazil it is assumed that 50% of the potential supply and the primary demand is covered by ethanol from sugar cane (the main energy crop in Brazil), with an (optimistic) average raw yield of 370 GJ/ha (taking co-products into account), and 50% by advanced liquid biofuels from woody biomass with a raw average yield of 144 GJ/ha. For Brazil, the potential biofuel supply in REmap 2030 is between 6.9 and 8.2 EJ, which would require between 33 and almost 40 Mha or an increase in agricultural land of between 12% and14%, compared to 2012. To meet Brazil's demand in 2030 through Brazilian agriculture would require almost 26 Mha or an land increase of 9.3%, compared to 2012. In 1982, 1992 and 2002, Brazilian agricultural land surface was 17%, 10% and 3.5% smaller than in 2012, respectively. So, to achieve an increase of 9.3% over a period of 18 years (i.e. 2012 to 2030), the high increase in the forgoing 20 years (i.e. since 1992) must be repeated, where the full increase should be assigned to growing (woody) biofuel crops. However, food demand will increase due to economic growth requiring additional land and/or a significant increase in agricultural yields. Also the increase in agricultural land surface has decreased since 2002. So, the question is where additional land can be found without causing deforestation. Brazil could import liquid biofuels from other regions, but at the global level demand in 2030 will be close to the lower supply levels, implying that Brazil, which is the largest supplier of liquid biofuels in REmap 2030, should actually produce more than the domestic demand, exporting ethanol to other regions just as it does today. Some argue that if a 2 °C target is set and strong policies are implemented, this growth may very well happen, provided that simultaneously also agricultural practices and efficiencies are improved.

Another large supplier is the United States. To cover its own demand (growing maize and woody biomass), agricultural land should increase by 10% in 2030. However, to cover the global demand they should expand even more, i.e. up to 21% in the high supply case. However, agricultural land has been abandoned in the past 30 years and this trend should be reversed into a growth of up to 20% in 2030. One could argue that this abandoned land could be put back into production. However, as also indicated in Section 3.2, it is not evident that this will lead to greenhouse gas reduction.

In France and Germany, a huge expansion of agricultural land would be needed to cover the primary demand, or even to cover the low supply levels. This would imply that even more agricultural land than has been taken out of production in the past 30 years, should be taken into production in the next two decades.

An interesting observation is that Indonesia has no potential supply, but at the same time a high demand in 2030. The reason of this discrepancy is that, in the REmap 2030 scenario, Indonesia does not have any 'surplus non-forest land'. This would imply huge amounts of biomass import, which is rather unlikely. However, there might be space for increased bioenergy production which is not covered by the REmap scenario, such as energy crop production on degraded forest land or further intensification of agriculture. For example, the Senior Advisor to the Minister on Renewable Energy of Indonesia reported in October 2015 that Indonesia has identified some 70 million hectares of degraded rainforest to be replanted with a mix of high-yielding crops in close consultation with local stakeholders.
For China the situation might be different because they already are a large importer of agricultural products, which is expected to grow further in the decades to come. Furthermore, the 'Rest of the World' is assumed to be a large net exporter of liquid biofuels implying a growth in agricultural land of roughly 7% for biomass crops alone. Again this would be a major achievement given a growth of 2% since 1992.
At the global level, the agricultural land should grow by at least 5.4% to cover the biofuel demand in 2030. Given the fact that in the past 30 years, the growth has been less than 6% and that no growth was obtained since 2002 a growth of 5.4% in the next 15 years implies a huge change in current trends. Theoretically, this land is available. In REmap 2030, the theoretically available land is calculated as potential suitable land minus current agricultural land, forest, protected area and built-up area. However, in many cases, and especially in European countries it is unclear whether it is realistic that this land can be used in the current situation and at such a short notice. Most often this land is privately owned and has a purpose. Converting these lands into productive biofuel land would be a great challenge. Moreover, surplus agricultural land that is not used for food production will probably less attractive economically, possibly resulting even lower yields than assumed in Table 5.2.

5.3.2 Crop harvesting residues

The demand in REmap 2030 of crop harvesting residues is almost 18 EJ, almost 3 EJ higher than the middle estimate of PBL. The range of potential supply in REmap is between 12.6 and almost 30 EJ. The high estimate equals the high estimate of PBL, which is partly based on an in-depth integrated analysis of global biomass flows (Born, Van Minnen, Olivier, and Ros, 2014), summarised in Figure 5.3. It shows that in 2010, 11 EJ of a total of 77 EJ were used in the biofuel sector to produce 4 EJ of secondary bioenergy.
If unused and burned crop residues, which are classified as the sustainable potential, were used for energy and materials, extraction could increase by 1,180 Mt, equivalent to almost 24 EJ (see Table 5.3). This could increase the proportion from 5% to 17% of the energy content of crops and residues only, and to 11% of the energy content of the total primary produced agricultural production, including grassland and rangeland.
The large production of rice and the relatively low residue flow to the soil makes rice residues the residue with the highest potential for bioenergy, followed by residues from oil crops, cereals, maize and sugar cane. Although rice residues have an high potential (Lim, Abdul Manan, Wan Alwi, and Hashim, 2012) concluded that further research is required on optimal

allocation of rice straw and rice husk resources in rice mills and on industrial commercialisation of these technologies. The sugar cane agro-industrial system already incorporates many residues in the food-fuel-energy chain (see Chapter 2), but improvements and innovations in land management and in sugar and ethanol processing are needed in order to use the full potential.

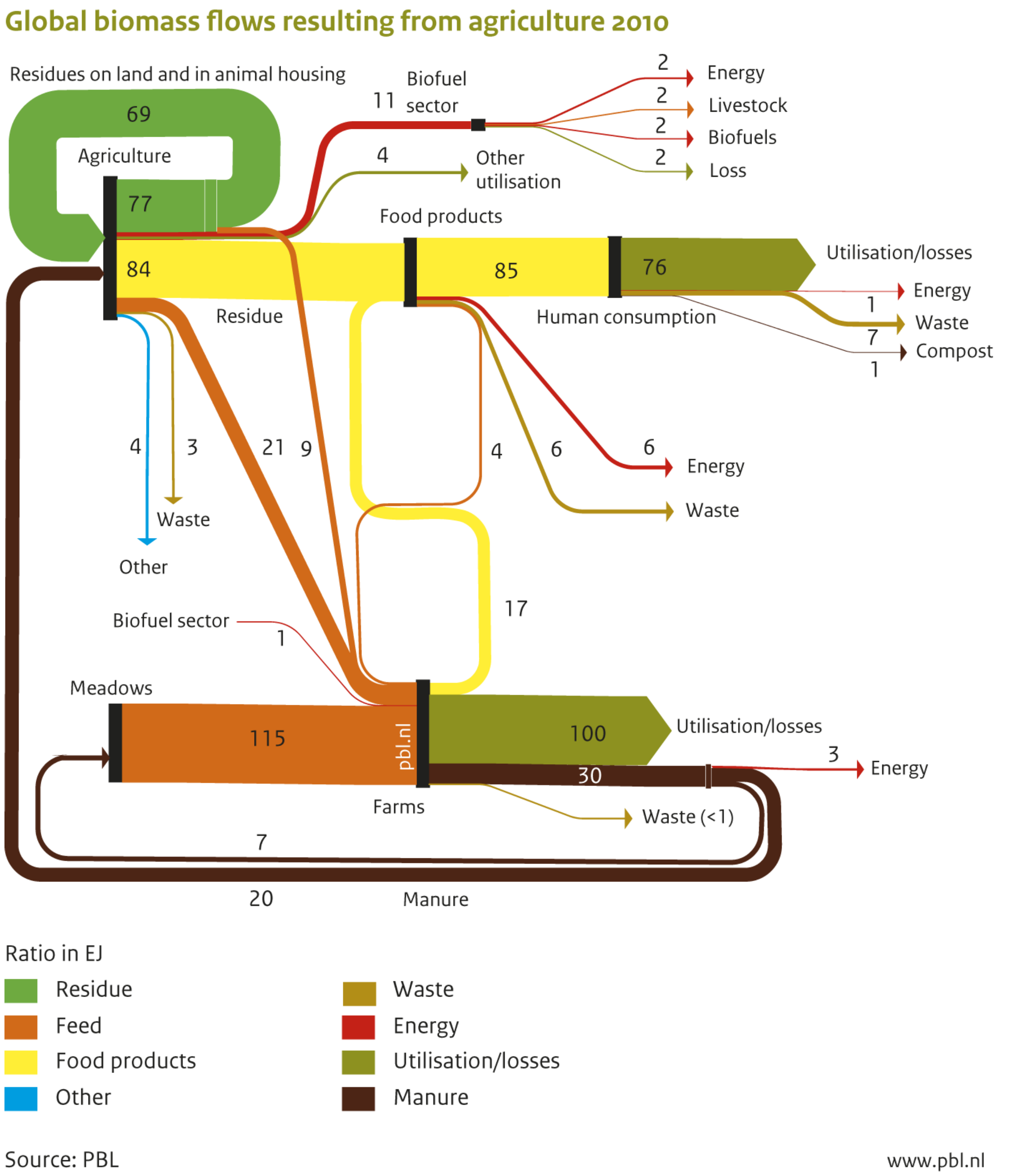


Figure 5.3 *Sankey diagram of the global biomass flows in agriculture in 2010. Source: (Born et al., 2014).*

Also, this estimate is in line with a recent study on the availability and cost of residues from agriculture and forestry (Daioglou, Stehfest, Wicke, Faaij, and Van Vuuren, 2015) applying a methodology which projects residue availability within the integrated assessment model IMAGE (Stehfest et al., 2014). Depending on the scenario, theoretical potential in this study was projected to increase from approximately 120 EJ today to 140–170 EJ by 2100, coming mostly from agricultural production. In order to maintain ecological functions approximately 40% is required to remain in the field, and a further 20% to 30% is diverted towards alternative uses. Of the remaining potential (approximately 40 EJ/yr in 2030 and 45 EJ/yr in 2050), more than 90% is available at less than 10 USD_{2005}/GJ.

Table 5.3 *Potential for energy and materials from crop residues including energy co-production. Source: (Born et al., 2014).*

Commodities:	(Million tonnes)	share	(EJ)	share
sugarcane	120.4	9%	2.2	8%
oilcrops	190.4	15%	4.1	15%
cereals	189.3	14%	4.7	18%
corn	131.8	10%	3.2	12%
paddy	483.9	37%	8.2	31%
other crops	64.5	5%	1.3	5%
	1,180.2		23.6	
sugarcane bagasse	126.2		3.0	
	1,306.4	100%	26.6	100%

units: Mtonnes (left column) EJ yr-1 (right column), all countries, 2010

5.3.3 Agricultural processing residues

With respect to agricultural processing residues, Figure 5.3 shows that in 2010, 6 EJ was converted into energy and 6 EJ is wasted, equal to the low supply level in REmap 2030. To satisfy the relatively low REmap 2030 demand of 2.3 EJ, less than 40% of the currently wasted processing residues have to be used as bioenergy.

5.3.4 Post-consumer household waste and animal manure

According to REmap 2030, the global primary demand for post-consumer household waste and manure is almost 21 EJ, and the potential supply, both low and high, is 18 EJ. In Figure 5.2, this demand and supply is compared with the waste flow from agriculture according to PBL, which is 20 to 45 EJ in 2050, but this also includes agricultural processing residues with a maximum potential of 6 EJ in 2010 and maybe more by 2030 (see Section 5.3.2).

Based on Figure 5.3, it can be concluded that 7 to 8 EJ of household waste in 2010 is potentially available for bioenergy. Assuming a growth of 30% between 2010 and 2030 (OECD, 2012), the maximum available waste flow would be ~10 EJ in 2030.

With respect to manure, Figure 5.3 shows that 3 EJ out of 30 EJ was used in 2010 to produce bioenergy. The remaining 27 EJ is returned to agricultural and pasture land. REmap assumes

that up to 50% of this flow can be recovered sustainably, dependent on the agricultural system (i.e. rangelands vs stables). According to Table 15 in (Nakada, Saygin, and Gielen, 2014), the recoverable fraction in REmap 2030 of manure ranges between 0% for buffalo, goat and sheep in South America and 56% for India. Assuming an average recoverable fraction of 25%, this translates into a potential of 7.5 EJ in 2010 (25% of 30 EJ) and almost 10 EJ in 2030 assuming the earlier mentioned growth of 30%. However, REmap 2030 assumes a conversion efficiency (into biomethane) of 100%. According to BioGrace this efficiency is between 50% and 55%, reducing the actual potential to 5 or 6 EJ.
So, the total potential in 2030 based on the global biomass flow from agriculture is 10 EJ from household waste and 6 EJ from manure, resulting in 16 EJ. This is close to the global demand of 18 EJ according to REmap.

5.3.5 Fuelwood

In REmap, the category 'biomass from surplus forest land' refers to fuelwood where it is assumed that traditional biomass is replaced by 'modern biomass'. Again the designation 'from surplus forest land' suggest no LUC effects should take place, which is difficult to achieve in a real world situation. The demand in REmap 2030 is 18 EJ and the potential supply between 19 and 33 EJ (see Figure 5.2). The PBL estimate in 2050 is between 10 and 35 EJ, depending, for example, on the sustainability criteria that might be applied to woody biomass in the future. This range is partly based on the earlier mentioned PBL study on global biomass flows (Born et al., 2014) from which Figure 5.4 is copied. The FAO data set was the primary data source for quantifying wood product/timber production flows (FAO, 2013), supplemented with data from other institutes on fuelwood (IEA, 2012), illegal logging (Nellemann C and Interpol, 2012) and primary residues (Mantau et al., 2010). The diagram shows that about 66 EJ biomass was harvested in 2010, of which about 49 EJ yr-1 was used for energy production. The largest part of this, 39 EJ, was used as a traditional energy source. Some trees were felled for this purpose, while another part was in gathered residues; but the ratio between the two is unknown, nor is it known whether this was done in a sustainable manner. As indicated, a strong downward trend is expected in the highly inefficient use of woody biomass as a traditional energy source. It should be replaced by 'modern biomass' for either power generation or heating. Furthermore, an additional 15 EJ in wood is assumed to be needed by 2050 (and 7.5 EJ in 2030) for more wooden products and paper, which must be subtracted from the projected amount of wood available for energy.

5.3.6 Wood logging and processing residues and wood waste

The REmap 2030 supply of wood logging and processing residues (or wood residues) is between 13 and 15 EJ. The supply of wood construction, demolition and furniture waste (or wood waste) is a small range between 8.4 and 8.8 EJ. The REmap demand for both source categories combined is 26.5 EJ. Assuming that the ratio between the categories wood residues and wood waste in the demand equals the ratio in the supply (i.e. 60:40), the combined demand of 26.5 EJ consists of 16.3 EJ of wood residues (60%) and 10.2 EJ (40%) of wood waste (see Figure 5.2). So, the demand is slightly higher than in the high supply estimate. The range for wood logging and processing residues (including dead wood) in the PBL estimate is between 5 and 25 EJ. The middle estimate is 15 EJ, close to the demand and high supply estimates of REmap. The PBL range for wood waste is between 5 and 20 EJ. The middle estimate is substantially higher than the REmap demand and supply estimates.

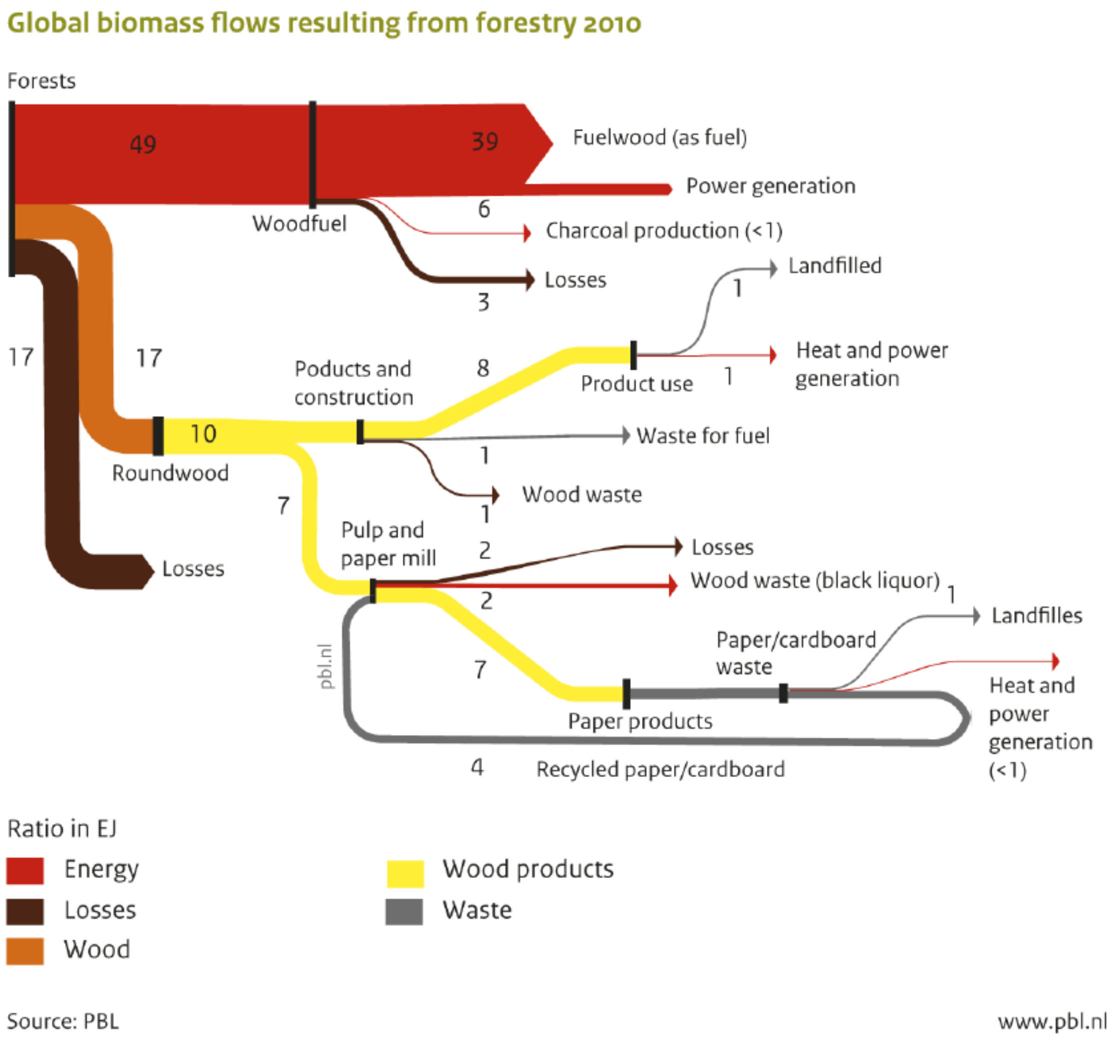


Figure 5.4 *Sankey diagram of the global biomass flows in the forestry sector in 2010 in EJ. Source: (Born et al., 2014).*

Starting point of PBL's wood residues estimate is the Sankey diagram of Figure 5.4. Wood logging removes tree trunks and some branches from a forest for specific uses. But damaged trunks and branches and bark are often left or burned along roadsides (primary residues) because removal is often uneconomical. Between 20% and 35% of total felling consists of primary forest residue (Mantau et al., 2010), implying a loss of 11–19 EJ per year (average 17 EJ; see Figure 5.4 and Table 5.4). A large proportion could be removed from a forest without affecting the nutrient balance, because most nutrients are in the small biomass, mainly the leaves. The exception is forests on poor soils where harvest residue potentially represents a substantial biomass resource (14 EJ) for energy production (Repo et al., 2012).
The global stock of dead wood is estimated at about 1200 EJ of biomass (FAO, 2010). This large pool has built up over a long period of time and in the entire forest area. Assuming an

average rotation of 50 to 100 years, this implies a biomass pool of 10 to 20 EJ per year. When primary forests are excluded, about 7 to 14 EJ of dead biomass remains (FAO, 2010). Forests with large quantities of dead wood are located in Russia and in parts of Africa. As indicated in Chapter 4, a limitation to the use of salvaged wood is the high costs of access and transport (Niquidet et al., 2012). A conservative estimate of accessible planted forests reduces the pool of available dead wood to about 2 EJ per year (Table 4.2). When an additional assumption is made that half of the dead wood needs to remain in forests to maintain biodiversity (Verkerk, Lindner, Zanchi, and Zudin, 2011), the estimate is about 1 EJ biomass is available annually for energy production.

Table 5.4 *Wood production for energy production in 2010 and its potential if the wood is used more efficiently. Source: (Born et al., 2014).*

Source	Current total production	Waste/Unused residues	Currently used for energy & heating	Sustainable extra potential
Fuelwood and biomass use	49	0	49	0
Construction/saw logs	10	0.3	1.9	0.3
Paper/pulp production	7	0	3.6	0
Subtotal	66	0.3	54.5	0.3
Harvest losses / Primary residues	17	17	0	14
Wood product waste (cascading from construction/saw logs and paper/pulp)	3	1.9	1.1	1.9
Dead wood in forests	10-20	0	0	1
Subtotal	30-40	19	1.1	17-18
TOTAL	96-106	19	56	17-18

Unit: EJ yr-1

Figure 5.4 shows that 17 EJ was harvested in 2010 for paper and pulp (7 EJ) and for construction and saw logs (10 EJ). Wood waste occurs in processing, such as in sawmills and paper production, and is estimated at 6 EJ. In developed countries, much of this biomass is either lost or reused in the system, for example for energy, and is not an additional fuel source (IEA, 2012). Sawmills in developing countries produce about 0.3 EJ of unused residues from construction and saw logs (Table 5.3).

The annual global production of industrial roundwood is estimated at about ~17 EJ, which includes 3.6 EJ of illegal logging. Materials, such as timber, board and paper, are wasted (about 3 EJ) and end in landfill (1.9 EJ) or are used in energy production and co-firing (1.1 EJ). Assuming equal quantities from timber products and paper and pulp waste, this cascading provides a potential 1.9 EJ for energy production (Table 5.3). There is considerable stock build up in the system with more wood products (15 EJ) produced annually than disappear from the system (3 EJ). The current waste flow for energy production could increase considerably in the coming decades (1–8 EJ per year) as the system gains more equilibrium.

6 Overall emission factors

Key messages

- Based on the data sources and studies referred to in this report, overall greenhouse gas emissions factors of bioenergy pathways are potentially lower than fossil fuels, especially when compared to coal and fossil oil.
- However, the ranges are very wide due the uncertainty in supply-chain emission, ranges in LUC emissions and ranges in carbon debts.
- Lower emission factors than presented here are possible if strict policies would be implemented to avoid greenhouse gas emissions in the production of bioenergy.

6.1 Introduction

In this chapter an overview of the emission factors related to bioenergy is presented. The final impact of the introduction of more bioenergy on greenhouse gases in the atmosphere can be assessed in three steps: emissions related to the production of the bioenergy carrier (Section 6.2), emissions related to the use of this bioenergy carrier to produce a specific form of energy for final use (Section 6.3) and the comparison of the emission factors related to bioenergy with the emission factors of fossil energy sources.

6.2 Emission factors of bioenergy carriers

Table 6.1 is an overview of the supply-chain emissions, LUC emissions and the 'carbon impact' of a set of key bioenergy carriers[8] as discussed in this document. The 'C content' is the amount of CO_2 per MJ that has been captured from the atmosphere during growth.
Supply-chain emissions of crude vegetable oil are computed from supply-chain emissions of biodiesel (Figure 2.2) minus the emissions related to the esterification process (Table 2.2b) and assuming an efficiency from crude oil to FAME of 95% (JRC et al., 2015). Supply-chain emissions of ethanol and biomethane are copied from Figure 2.1; supply-chain emissions of pellets are based on Table 2.5. As indicated in the respective chapters, note that uncertainty ranges have not been quantified in this study.
Land-use change (LUC) emissions in Table 6.1 are based on Table 3.4 – i.e. the economic studies – for crude oil and ethanol, assuming an efficiency from crude oil to biodiesel of 95% (BioGrace) and from wood to ethanol of 65%, computed from (PBL, 2008). The LUC effect of wood pellets and chips from former agricultural land is based on descriptive studies of willow and poplar, because economic studies do not exist for this category. We report no LUC emissions if pellets are extracted from marginal (or degraded) lands. However, this is only true if these lands are unsuitable for agricultural use, which is, in our view, hardly ever the case (see Chapter 3 and Section 5.2.2).

[8] A bioenergy carrier is defined here as the form in which energy is transported and distributed.

Table 6.1 *Carbon content and greenhouse gas impacts of various bioenergy carriers in gCO_2eq/MJ end product as derived in this study. Note that uncertainty ranges for bio liquids and biomethane have not been quantified in this study and that longer amortisation periods result in lower emissions.*

Bio-energy carrier	**From**		**Carbon content**	**Supply chain**	**LUC 30 years**	**Carbon Impact 30 years**
			Units: grams CO_2 equivalent per MJ bioenergy carrier			
Crude vegetable oil	Palm oil	With CH_4 capture	71	19	30 [12,46]	n/a
		Without capture		39		
	Soya beans			36	34 [14,59]	
	Sunflower seeds			21	35 [33,37]	
	Rapeseed			32	41 [7,89]	
Ethanol	Starch crops (gas CHP)		71 – 82	38	21 [6,52]	
	Sugar cane			24	17 [3,46]	
	Sugar beet			33	6 [3,9]	
	Woody	Switchgrass		22	22 [11,44]	
		Miscanthus			5 [-6,17]	
Bio-CH_4	Manure and Waste		50	20	0	
Pellets	Fast growing[a]	Agricultural land	100	15 to 29	9 [2,16]	<0
		Marginal land			0	
	Agric. residues	Crop harvest				>0
		Processing				0
	Forest residues	Ref = Burning		8 to 18		
		Ref = Decay				35 [25,45]
	Increase in	Thinning				90 [70,110]
		Felling				185 [165,200]
	Waste	Ref = Burning		9 to 25		0
		Ref = Landfill[b]				-150 to 80

[(a)]Refers to source categories woody crops and short rotation coppice, see Table 2.5.

[(b)]Half-life paper and wood set to 100 years (IMAGE model). 0% to 50% emitted as methane.

The 'carbon impact after 30 years' is shown in the last column. This is defined as the amount of the carbon content that would remain after an amortisation period of 30 years in the reference case and is computed as the carbon content at the moment of harvesting times the value of the CI indicator (Carbon Impact indicator, see Section 4.2.2) after 30 years.
In case feedstocks are grown on former agricultural or marginal land, carbon impact can be negative since growing biomass on these areas (especially marginal land) often increase the carbon content of the soil. However, ranges are wide and no data are available that can be used to give a global estimate. For residues, the carbon impact highly depends on the reference situation. Obviously, if the reference case is that forestry residues are burnt on site

after harvest, the carbon impact is equal to zero. If residues are harvested that would otherwise decay 'slowly' in the forest, the remaining carbon in the reference case could still be substantial after 30 years as also shown by payback times that can be up to 80 years (see Figure 4.3). The range shown here is based on the results of the EFISCEN model (Figure 4.2). Thinning and especially felling result in long payback times and thus high carbon impacts.

In case of paper and wood waste, payback times and thus carbon impacts are zero if the reference situation is burning. It is more complicated when the reference situation is a landfill (see Section 4.3.4). The numbers presented in Table 6.1 are the extremes, i.e. from waste burning to landfill with and without methane capture. In the case of landfill, the half-life for wood and paper waste is set to 100 years (based on the IMAGE model) and it is assumed that 0 (=methane capture) to 50% (=no methane capture) is emitted as methane. So if it is assumed that 20% of the paper and wood waste decays in 30 years – which is equal to a half-life of 100 years – and 50% of the carbon is emitted as methane, then it would imply a CO_2 equivalent emission of 230 grams per MJ.

6.3 Bioenergy emission factors for power, transport fuels and heat

Three types of final energy are distinguished in Table 6.2: power, liquid and gaseous transport biofuels and heat. The emission factors are equal to the sum of supply-chain emissions from Figure 2.1, LUC emissions from Table 3.4 and carbon impacts from Table 6.1, assuming an efficiency of biomass plants of 43% and an efficiency of 80% in the production of heat from biomass. These efficiencies are equal to the global average of new large power plants in the TIMER model[9] in 2030. The efficiency from chips and pellets to transport fuels is 70%, based on the production of FT diesel and computed from (PBL, 2008). It is assumed that high quality biofuels such as crude oils and ethanol will not be used in electricity production at a large scale and therefore these numbers have not been computed.

The bioenergy emissions presented in Table 6.2 can be compared with fossil alternatives for the production of energy in final use as shown in the last three rows. These Emission factors are based on world average efficiencies in 2030 of the TIMER model (shown between brackets in Table 6.2). Supply-chain emissions are excluded and account for 10% to 20% of combustion emissions.

In most cases first and second generation bioenergy perform better than fossil fuels, especially coal and fossil oil, except for pellets from thinning and felling. However, as indicated before, uncertainty ranges are large and the carbon impact of pellets is time dependent, i.e. longer amortisation periods would result in lower emission factors. It is important to note that the emission factors shown are based on data sources and studies

[9] More details on the TIMER model can be found in Section 4.1 of Stehfest et al. (2014) and the TIMER pages of the website on the IMAGE model.

referred to in this study and could be lower if, for example, strict land-use policies would be implemented minimising or even avoiding direct and indirect LUC emissions (see Chapter 7).

Table 6.2 *Emission factors for power plants, liquid and gaseous transport biofuels and heat plants. Applied efficiencies are based on the TIMER model (Stehfest et al., 2014) and (PBL, 2008).*

Bio-energy carrier	**From**		**Power**	**Transport**	**Heat**
			Units: grams CO_2 equivalent per MJ		
Crude vegetable oil	Palm oil with CH_4 capture		n/a	63 [45,74]	62 [40,81]
	Soya beans			85 [64,112]	88 [63,120]
	Sunflower seeds			71 [69,73]	71 [69,74]
	Rapeseed			89 [53,139]	93 [49,153]
Ethanol	Starch crops (gas CHP)		n/a	59 [44,90]	n/a
	Sugar cane			41 [27,70]	
	Sugar beet			38 [36,42]	
	Woody	Switchgrass		44 [33,66]	
		Miscanthus		27 [16,38]	
Bio-CH_4	Manure and Waste		46	20	25
Pellets	Fast growing	Agricultural land	40 to 110	25 to 65	20 to 60
		Marginal land	35 to 70	20 to 40	20 to 35
	Agro-residues	Crop harvest			
		Processing			
	Forest residues	Ref = Burning	20 to 45	10 to 25	10 to 25
		Ref = Decay	85 to 150	50 to 90	45 to 80
	Increase in	Thinning	180 to 300	110 to 180	95 to 160
		Felling	415 to 520	250 to 315	220 to 255
	Waste	Ref = Burn	20 to 45	10 to 35	10 to 30
		Ref = Landfill	-360 to 250	-210 to 150	-180 to 130
Fossil energy source		**Emiss. factor**			
Coal		93	195 (48%)	-	117 (79%)
Gas		56	98 (57%)	56 (100%)	65 (87%)
Oil		84	-	84 (100%)	102 (82%)

7 Costs, strategies, and policy directions

Key messages

- Large-scale bioenergy deployment is an important contributor to reaching ambitious climate change targets, significantly reduces the greenhouse gas mitigation costs, and may be indispensable to meet the same objective if response actions are seriously delayed.
- Bioenergy options can deliver net cost benefits compared to fossil fuel alternatives, and more so if greenhouse gas emission reductions are valued in monetary terms. However, from the global perspective, net benefits are lower, and sometimes much lower, than the reduced emissions of the replaced fossil fuels suggest.
- Cost implications at smaller scales and from differing stakeholder perspectives can vary enormously from the global perspective. Firstly because net emissions reductions differ under varying system boundaries. And secondly, because the unit price of emissions depend on specific rules and regulations for the country or sector
- Simultaneous introduction of measures to keep land conversion in check, e.g. to protect forests are beneficial to reduce negative impacts of ambitious bioenergy schemes on natural ecosystems. Schemes to protect forest areas can limit net land-use change and related emissions of greenhouse gases, leading to beneficial effects for nature protection and biodiversity conservation of highly valued forest areas.
- Introducing land protection policies will bring about costs for consumers in the form of higher agricultural commodity prices.
- Four policy directions to limit the impact of large scale bioenergy production can be distinguished: 1) increase (biomass) productivity of land-use systems in general and in particular in agriculture and forestry, 2) protection of carbon rich land, 3) increase the carbon stock of land in both biomass and soil, and 4) to make better use of waste products and improve efficiency in the chain.
- Sustainability criteria need to be developed carefully with wide consultation and good systems understanding about the natural and anthropogenic processes and their interactions. That is the only way to ensure that proposed measures have the desired consequences.

7.1 Introduction

This chapter aims to provide a wider context by discussing costs, strategies and policies of biofuels in relation to climate policy. Section 7.2 describes if and to what extent the greenhouse gas reduction saves costs by reducing damage by climate change or reducing costs of climate change mitigation, and the economic viability of the biofuels including these benefits. Section 7.3 discusses the implication of various policies and targets of bioenergy use in climate policies for the abatement costs. In Section 7.4 the effects of additional forest protection are discussed. On the one hand forest protection is a climate policy that is likely to be implemented and influencing biomass production, on the other hand is can reduce the negative land-use change effects of biomass production. Section 7.5 provides policy directions to limit the impact of large scale bioenergy production.

7.2 Cost implications of greenhouse gas emissions

The supply chain of biomass incurs direct costs for production of the feedstock, transport to the markets, conversion to end products and distribution to end users. The cost for the end user, which is often further adjusted by taxes or subsidies, varies widely and is estimated in an earlier IRENA study between 2 and 80 USD/GJ (Nakada et al., 2014). Their value depends on the price of competing fossil fuels, estimated at 34 to 59 USD/GJ for liquid fuels and 7 to18 USD/GJ for other biomass in the same IRENA study. The question arises if and to what extent the greenhouse gas emission estimates presented in this report may have implications for the economic viability of the bioenergy alternatives. In the context of climate policies, greenhouse gas emissions represent an implicit value, either because of (future) damages they induce, or because of costs involved with mitigating them. So, mitigating emissions represents a financial benefit, as emissions have a price and thereby imply a cost for the emitter.

The level to which the use of biomass is emitting CO_2 ranges from close to neutral to very significant compared to the fossil fuels they replace. The magnitude depends on a range of factors, which is documented extensively in the previous chapters of this report.

Unfortunately there is no single number –or even a universally valid approach – to answer the question of costs of emission reductions. Several countries with differing stakes in greenhouse gas mitigation are often involved in the bioenergy supply chain. It is also very important to take into account the perspective of the cost-benefit analysis (global, national, sector, end user), the system boundaries, as well as relevant rules and regulations in countries or for sectors. It is important to know whether countries are committed to reduce their greenhouse gas emissions, to what degree, and what climate policies they put in place to bring about emission reductions.

For example, primary biomass can be produced in a country with no explicit policies to reduce greenhouse gas emissions, then shipped to a country with mitigation targets where the feedstock is processed to liquid biofuel and then used. As presented in Chapter 6, the overall greenhouse gas balance may be relatively poor for such a supply chain. Hence, in terms of the international, societal cost of the entire chain, the greenhouse gas benefits can be much smaller than the reduction arising from using less fossil energy.

In the absence of a global market, the implicit value of the greenhouse gas reduction from the global perspective can be approached by the concept of the social cost of carbon.

Though very uncertain, a recent estimate suggest a present value of 37 USD/tCO_2eq (US, 2013). So, by multiplying the overall net reduction in a certain bioenergy supply chain in tonnes by USD 37 we get a first-order estimate of the global benefit. If in the example given the direct cost of the biofuel was 20 USD/GJ and the net greenhouse gas saving 33% of 75 g/MJ product, this would account for (0.33x75x0.037=) 0.9 USD/GJ or only a 4.5% lower cost. However, from the perspective of the producing countries, greenhouse gas emissions occurring on their territory do not represent a cost if they have no emission reduction targets. Whereas from the perspective of the country where the biofuel is consumed, the greenhouse gas savings are larger than the net impact over the entire chain, as the emissions of the feedstock production abroad are not accounted for. The national price can be equal to a carbon tax, or the implicit value or price of emissions as the result of other policy instruments such as emission trading schemes, sector emission caps or product standards. This CO_2 price level can be much lower, compare the current price in the European ETS, but also much higher as in many domestic non-ETS policies, than the earlier mentioned social cost of USD 37. In the same example as before, national greenhouse gas savings could be 70%, and if the local (implicit) price was 100 USD/tCO_2eq, they amount to (0.7x75x0.100=) USD 5.25 or 26% of the market price of USD 20. So, leaving aside price distortions from taxes or subsidies, emissions that originate from the country of use, e.g. from domestic processing of the feedstock, are faced with an additional cost, which raises the product price. If the processing industry would fall under the ETS system, under current market conditions the cost increase from the emission is very small. End users face the cost of the biofuel, but save petrol or diesel, which price is bound to reflect the cost of greenhouse gas emission subject to the rules and regulations of the country, adding to other levies such as excise and value-added taxes.

7.3 The role of bioenergy in low-carbon climate strategies

A wide range of technological options exist that can play a larger or smaller role in future low-carbon strategies, such as those aiming to not exceed the 2 °C global warming target. Taking into account differences in accessible resources in regions and appreciation of the viability and affordability of options, varying political or societal preferences, and also methodological differences, all work together to produce widely varying outcomes for future low-carbon energy pathways. The challenge to stay below 2 °C is so steep that many options will need to be deployed on a large scale, and commonly bioenergy features strongly in such long-term projections. This underlines that bioenergy comes out as a relatively attractive, affordable option to choose from the menu available to the models in use. An important consideration, however, is to what extent the possible negative impacts, such as greenhouse gas emissions and land-use implications reported in this study, are recognised in the tools. But also other considerations such as food security and food prices, water availability, ecosystem services and biodiversity should also be taken into consideration, which are not elaborated on in this study.

To identify how important substantial use of bioenergy is in low-carbon energy projections, several studies have been conducted that explore the implications of higher or lower potential for key technological options. One recent example is the EU-AMPERE study (Riahi et al., 2015), in which limited availability of bioenergy is one of the cases analysed, which assumes an upper limit for global use of bioenergy of 100 EJ. As the models typically select available options in increasing order of cost, constraining one relatively low-cost option will tend to drive up total cost of the strategy by substituting it by another option. Many other cases were tested, but the AMPERE conclusions point at the importance of bioenergy as follows: 'Biomass is . . . of central importance for keeping mitigation costs relatively low, which has also been emphasised by other studies'. An important factor is that the option of combining bioenergy with CO_2 capture and storage (CCS) effectively removes CO_2 from the atmosphere. These 'negative emissions' allow for peak-and-decline emission and concentration pathways, attractive from an overall cost point of view, and due to slow response of the earth system consistent with the 2 °C mark. Across a range of eight global models, abatement cost of the 2 °C or 450 ppm scenario with limited availability of bioenergy increased by 60% to 100% compared to a scenario with a more abundant bioenergy supply. So, meeting the climate target is still feasible with less bioenergy, but at higher cost. The estimates given here assume a concerted, global climate action with full cooperation and efficient implementation of policies. If the climate response strategies were delayed, limiting bioenergy made the 2 °C target infeasible in more than half of the models. This indicates that under such conditions large amounts of bioenergy are crucial to meet the ambitious climate target.

7.4 Forest protection

The production of feedstock for biofuels may induce conversion of natural lands in the producing regions (see Chapter 3), with impacts on greenhouse gas emissions, ecosystems services and biodiversity. The main side effect for this study concerns the loss of carbon stored in above- and below-ground biomass and soils, when natural areas are converted to cultivated land. As shown, these carbon emissions vary between biomes and locations, but can offset the greenhouse gas emission reductions to a large extent, or at least initially, with impacts that will last for decades (see Chapter 4). The importance of limiting the conversion of natural lands is recognised from both a climate mitigation perspective and a nature conservation perspective. This has led to the introduction of policies to reduce deforestation, either for climate change alone (REDD) or to address mixed climate/biodiversity concerns (REDD++). As payments for conserved carbon stocks are part of the REDD mechanism, it implies transfer of funds from industrialised to developing countries in return for emission credits. In a study for the EU (7), PBL explored the potential gains of combining a very ambitious bioenergy expansion target with protection of remaining natural lands with relatively high carbon content, mostly forests.

The IMAGE model framework, combining economic and land-use modelling, was used to explore which currently unused lands would emit the biggest amounts of carbon after conversion, and ordered all land areas in accordance with their implicit carbon stock. The potential land supply for future expansion was adjusted using this information, such that all grid cells with high carbon stocks were assumed to be unavailable due to REDD protection arrangements. This alters the prospects for viable bioenergy feedstock production in those regions with predominantly high-carbon, forested lands, such as Indonesia and Brazil.

In order to still supply the bioenergy demands on top of the food and fibre demands, the model suite applies the following responses:

- A shift of production, and associated land, from 'ecosystem carbon-rich' regions to other parts of the world;
- An associated shift in the mix of oil crop feedstocks to starchy feedstocks;
- A price increase for agricultural commodities, inducing a (slight) increase in overall productivity and thus less land, and some reduction in consumption.

As a result, the same amount of bioenergy is delivered as in the absence of REDD, but with lower net carbon emissions, and with less impacts on nature quality, biodiversity and other ecosystem services. The latter in particular in high-valued ecosystems in the tropics and semi-tropical zones.

Results

The global, total production of food crops is hardly affected by the REDD regime, but oil crops are produced less in 2020 and hence less oil products are available for consumptive and industrial uses, and also for biofuel production. The loss of oil products for biofuels is offset by increases in the production of temperate cereals and maize. These relatively small changes at the aggregated, global scale work out very differently in the various world regions. For example, production of oil crops in Indonesia and other SE Asian countries is significantly reduced if REDD is assumed in combination with the high biofuel case, and is mostly offset by increased production in the temperate zones.

On the global level the net changes in production and prices imply relatively small differences in consumption for human food and livestock feed; see Table 7.1. Non OECD regions together are affected slightly more, but the decrease in consumption remains well below 0.2%. In some cases, however, the REDD measures induce more significant effects on the level of consumption, up to 4.8% in the Rest of Central America.

Table 7.1 *Change in consumption for food and feed in 2020 due to REDD, global total and selected regions. (measured in value terms)*

Consumption Food and Feed	Oil Seeds (Oil crops)	Coarse Grain (Maize)	Wheat (Temp.cereals)
World	0.02%	-0.05%	-0.12%
EU27+	0.01%	-0.03%	-0.04%
Non-OECD countries	-0.01%	-0.08%	-0.16%
Rest of Central America	-4.81%	-2.81%	-2.64%
Southeast Asia	-0.79%	-3.23%	-2.13%

By and large, trends in agricultural area will follow the trends observed for production, so a small decrease at the global level, composed of reductions in tropical and subtropical regions, partly offset by expansion in temperate regions, see Figure 7.1.

Effect of REDD on crop area

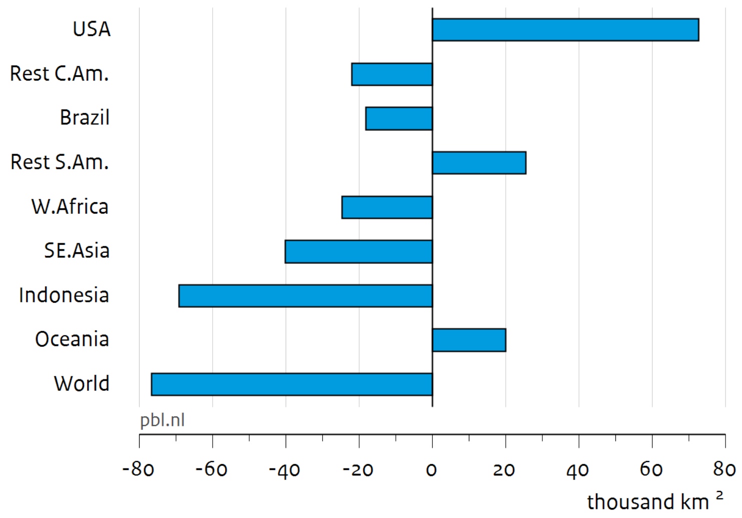


Figure 7.1 *Change in crop area in 2020 due to REDD*

Higher prices lead to slightly lower consumption, and together with higher yields caused by constraining land availability this means that some 80,000 km^2 less agricultural land is needed. Production and land use in tropical and subtropical countries is affected the most, and is partly offset by more production on more land in temperate regions.

As expected from the change in agricultural areas, the impact on natural lands from producing large amounts of biofuels shifts from tropical regions to temperate regions, and from forests to other ecosystems such as grasslands and scrublands. As a consequence, biodiversity loss in relatively species-rich forests in regions such as Indonesia, Brazil and Africa is reduced. The outcome for other ecosystems such as savannas, grass and scrublands is more mixed; in some regions these also benefit from REDD, in other regions they decline as the result of agricultural expansion.

For almost the same level of production of crops for human consumption, feed and biofuels, the land-use emissions of CO_2 are significantly reduced due to the REDD measures; see Figure 7.2. The major contribution to the reduction in CO_2 emissions from land use between the HiBF (purple line in Figure 7.2) and HiBF+REDD (orange line) cases is concentrated in tropical regions, where carbon-rich forests remain in place.

Production of biofuels will induce higher demand for the feedstocks from which they are produced, and this in turn tends to drive up the prices of agricultural commodities. If additional restrictions are imposed on where to allocate agricultural land to meet the total demand, e.g. through REDD, prices will rise further.

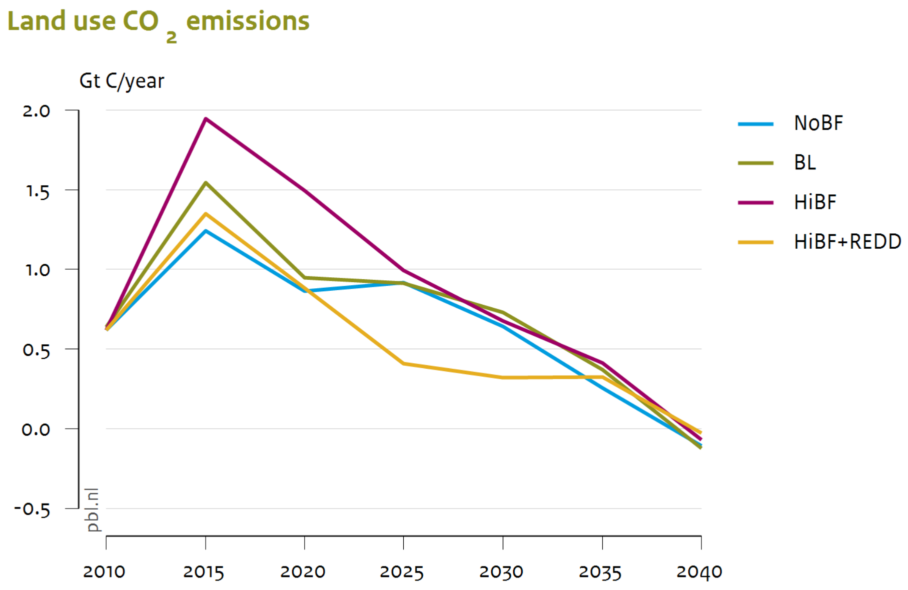


Figure 7.2 *Land-use CO_2 emissions. BL = Baseline: only 10% EU biofuel and US gasohol implemented, plus historic Brazil use continued. NoBF = No biofuels at all, to estimate the effect of BL. HiBF = High biofuel case: all policies and intentions globally implemented by 2020; estimated at 211 Mtoe or around 9 EJ of biofuels. HiBF+REDD = same as HiBF with restricted land supply due to carbon stock preservation.*

The REDD measures add another 1.5 percentage points to the price increase for the agricultural product groups concerned, compared with the HiBF scenario without REDD. As reported earlier; see Table 7.1, the price increases will reduce consumption of the commodities, but to a limited extent as demands are relatively insensitive to prices.
Higher prices and the resulting reduction in consumption have a downward effect on GDP. In 2020, global GDP is reduced by around USD_{2001}10.1 billion. Per tonne of carbon reduced in that same year (total 610 million tC), this indicates average costs of the REDD measures of around USD_{2001} 16.5 per tonne C, or USD_{2001} 4.5 per tonne CO_2. These costs represent global average opportunity costs, not including costs for implementation, monitoring and enforcement. The regional distribution of these costs is unequal, and they are mainly situated in the developing world. This aspect of forest protection should be addressed, for example some form of compensation before policies could be implemented successfully, this may include.

7.5 Policy directions

As discussed in this report the use of biomass from agriculture or forests has effects on land use with related greenhouse gas emissions. In order to actually reduce greenhouse gas emissions these emissions caused by land-use conversion should be as low as possible. This can be achieved by four policy directions that are characteristic for the land-biomass system in relation to climate change[10]:

1. Increase (biomass) productivity of land-use systems in general and in particular in agriculture and forestry.
2. Protection of carbon rich land (e.g. forest protection and protection of peatlands).
3. Increase the carbon stock of land in both biomass and soil.
4. A fourth element is to make better use of waste products and improve efficiency in the chain.

*Ad 1.*By increasing the productivity there is less land needed for the production of a GJ of bioenergy. Therefore, the most productive feedstocks (in GJ/ha) and production systems should be used and their productivity should be increased including in other sectors than bioenergy. Therefore, investing in research and technology in agricultural and forestry bioenergy systems and in general in all forestry and agricultural systems is essential.

Ad 2. Protection of the most valuable land in terms of carbon content and biodiversity creates boundaries to land expansion, but can also be an incentive to increase productivity. Forest land often is valuable in terms carbon and protecting them would affect the potential land for palm oil plantations and to a lesser extent sugar cane. As shown in Tables 6.1 and 6.2 forest-based biomass can have high carbon debts or payback times, using an allocation period of 30 years. In particular additional felling and thinning for bioenergy can have high carbon impacts. Longer allocation periods would result in lower carbon impacts, but if the policy target is to limit global warming to 2 °C this century, longer allocation periods will reduce the likelihood this can be achieved. In Section 7.4 it is shown that by constraining the land available for agricultural land expansion starting with the most carbon dense land there is a shift from oil-based biofuels to starch-based biofuels, as a result of economic forces. The constraint on land makes palm oil more expensive. This would work also the other way around; using more starch-based biofuels instead of oil-based biofuels would reduce the amount of forest land needed. Moreover, Section 7.4 shows that restricting agricultural area by forest protection is an incentive for higher production, so, contributing to point 1,, increase productivity for both oil crops as starch and sugar crops.

Ad 3. Increase the carbon content of land while maintaining its production is a no regret policy. For the crop and forest systems for bioenergy this translates in the stimulation of perennials and forest plantations on currently non-forested land (i.e. grassland, degraded

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[10] http://infographics.pbl.nl/biomassa/

land and possibly arable land). This increases the carbon in the vegetation and in the soil leading to sequestration and higher bioenergy production.

Ad 4. Waste has many forms. Most waste products do not affect the land carbon system, but crop harvesting residue do since they are important to maintain soil organic matter and thus soil fertility. So, it is important to determine what percentage of harvesting residues can be used sustainably. This varies per country and per system. Using processing residues are generally without land impact if they cannot be used as animal feed. So, best would be to only use processing residues that have no food or feed function or in other way needs to be replaced by agricultural products. Using wood logging residues have an impact on land since otherwise they would be part of the carbon pool of the forest. Consumer waste (food waste) could also have an animal feed use. Wood construction, demolition and furniture waste would be a waste stream with little to no influence on the land carbon system.

Overall efficiency of biomass use on a global level is rather low. For example, most of the solid biomass for heating purposes is used in traditional systems. It is important task to work on further improvement of the overall efficiencies of bioenergy technologies.
In general, there is potential to increase global bioenergy production without negative (e.g. LUC) effects, but it needs sustainability criteria to be developed carefully with wide consultation and good systems understanding about the natural and anthropogenic processes and their interactions. That is the only way to ensure that proposed measures have the desired consequences. To work effectively, such criteria should be adopted by all countries and biofuel producers. This would force the producers of bioenergy to make the right decisions and to create stability for investors. Sustainability criteria can be an important incentive to increase efficiency and greenhouse gas mitigation potential. Implementing worldwide sustainability criteria might be a difficult. However, country level implementation or implementation for specific pathways would already be a step forward. Obviously, many governments already have adopted bioenergy-related policies, but they not always fully implemented.

Annex 1 Emissions from direct land-use change

Using the European Commission's guidelines for the calculation of land carbon stocks, changes in Soil Organic Carbon (SOC) and carbon changes in above- and below-ground vegetation can be assessed (EC, 2010) following the methodology as laid out in the Renewable Energy Directive (EC, 2009). If enough data is available, the actual carbon changes in SOC and vegetation can be precisely calculated using these guidelines. If no actual or accurate data are available, the guidelines provide a general methodology, summarised below, using standard values from a set of 18 Tables.

The carbon stock of the actual land use is referred to as CS_A. The carbon stock of the land before conversion is referred to as the reference carbon stock or CS_R. The difference between CS_R and CS_A reflects the loss or gain of carbon due to the change in land use.

$$DLUC = CS_A - CS_R \qquad \text{tC/ha} \quad [A.1]$$

A gain of carbon can occur, for example, if existing agricultural land is converted into a palm oil plantation. For the calculation of CS_R and CS_A the following simple formula apply:

$$CS_i = SOC + C_{VEG} \qquad \text{tC/ha} \quad [A.2]$$

where

i = R(eference) or A(ctual)

SOC = soil organic carbon (tC/ha), see [A.3]

C_{VEG} = above- and below-ground vegetation carbon stock (tC/ha), which can be selected from Tables 9 to 18 in EC (2010), based on a combination of domain (tropical or subtropical), climate region, ecological zone, continent, and whether it currently is cropland (with annual or perennial crops), grassland, forest land or forest plantations.

In case of mineral soils (i.e. all soil types excluding organic soils), SOC is computed as:

$$SOC = SOC_{ST} \times F_{LU} \times F_{MG} \times F_I \quad \text{tC/ha} \quad [A.3]$$

where

SOC_{ST} = standard soil organic carbon in the 0–30 centimetre topsoil layer (tC/ha). A value can be selected from Table 1 in (EC, 2010), based on the appropriate climate region (see Figure A1.1) and soil type (see Figure A1.2) of the area concerned. For example, a tropical moist sandy soil has a standard organic carbon content of 39 tC/ha.

F_{LU} = land-use factor reflecting the difference in soil organic carbon associated with the type of land use compared to the standard soil organic carbon.

F_{MG} = management factor reflecting the difference in soil organic carbon associated with the principle management practice compared to the standard soil organic carbon;
F_I = input factor reflecting the difference in soil organic carbon associated with different levels of carbon input to soil compared to the standard soil organic carbon.
Values for F_{LU} , F_{MG} and F_I can selected from Tables 2 to 8 in (EC, 2010), based on a combination of land use (cultivated land, perennial crop, grassland, savannah, native forest, managed forest, shifting cultivation), climate region (Figure A1.1), management type (full tillage, reduced tillage and no till) and input level (low, medium, high with manure and high without manure).

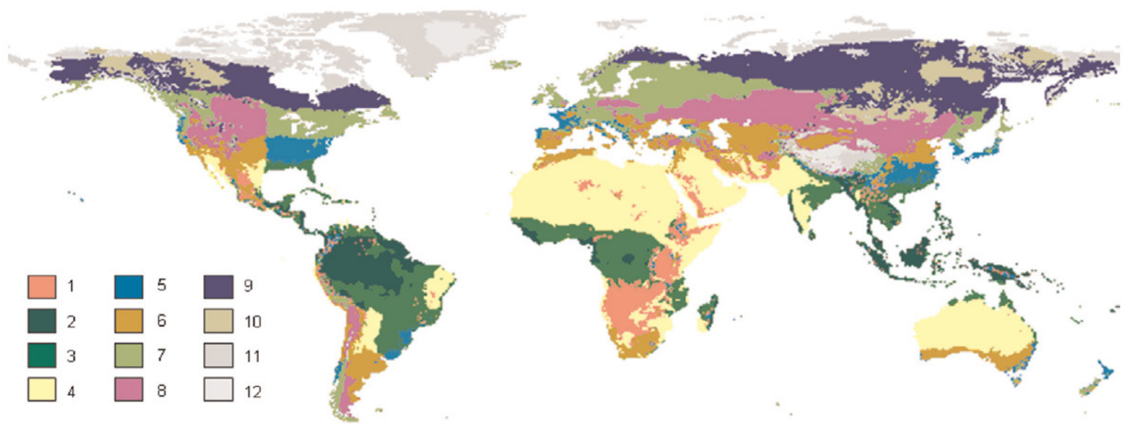

Figure A1.1 *Climate regions. 1 = Tropical, montane; 2 = Tropical, wet; 3 = Tropical, moist, 4 = Tropical, dry; 5 = Warm temperate, moist; 6 = Warm temperate, dry; 7 = Cool temperate, moist; 8 = Cool temperate, dry; 9 = Boreal, moist; 10 = Boreal, dry; 11 = Polar, moist; 12 = Polar, dry. Source: (EC, 2010)*

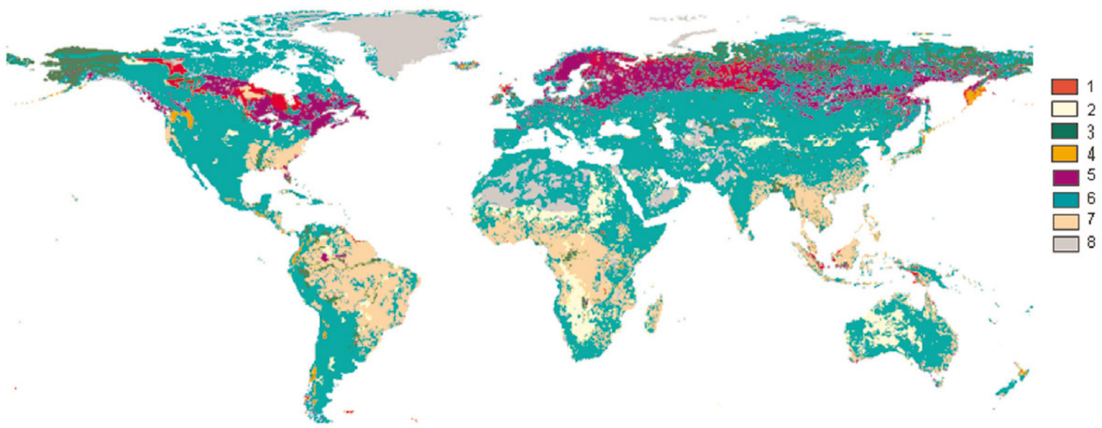

Figure A1.2 *Soil types. 1 = Organic; 2 = Sandy Soils; 3 = Wetland Soils; 4 = Volcanic Soils; 5 = Spodic Soils; 6 = High Activity Clay Soils; 7 = Low Activity Clay Soils; 8 = Other Areas. Source: (EC, 2010)*

In case of organic soils (which cover a relatively small fraction of the earth, see Figure A1.2) other methods have to be used to compute SOC that take into account the entire depth of the organic soil layer as well as climate, land cover and land management and input. Such methods may include measurements. Also, where carbon stocks are affected by soil drainage, this can result in additional soil carbon losses that should be taken into account.

Here, we illustrate the method with an example. For example we want to calculate the annual DLUC emissions of land-use conversion from grassland (the reference land use) to cropland with rapeseed (actual land use). We assume the climate to be 'cool temperate moist' and the soil to be 'high activity clay soils'. The change in land use causes a change in soil organic carbon (SOC) and a change in vegetation. The standard organic soil carbon (SOC_{st}) in high activity clay soils is 95 tC/ha (Table 1 in (EC, 2010)). This SOC_{st} is translated into the SOC of the original grassland and the SOC of the new cropland, using three factors for both grassland and cropland: the land-use factor, the management factor and the input factor (Table 2 in (EC, 2010)). Here we assume the grassland to be moderately degraded grassland and the cropland to be high tillage cropland without manure application. The SOC of the grassland is then calculated as 90.3 tC/ha and the cropland as 72.8 tC/ha. So, this is a loss of 17.5 tC/ha in SOC. The carbon content of the vegetation is 0 for cropland and 6.8 for the grassland (Tables 9 and 13 in (EC, 2010)). Thus, the conversion leads to a total carbon loss of 24.3 tC/ha or 89.0 tCO_2/ha. To convert this to an DLUC factor in gCO_2/MJ we use and energy yield from rapeseed of 72.3 GJ/ha/year (see Table 2.4). In 30 years (the allocation period) this is 2269 GJ. By dividing 89 tCO2/ha by 2269 GJ we calculate an DLUC value of 41 g CO_2/MJ. Thus, the DLUC emission of this conversion is almost 50% of the default CO_2 emission (i.e. 84 gCO_2 MJ) from fossil fuels.

Table A1.1 Combinations of crops, region, climate and soils and choices in the variables fertilisation and tillage.

Crop	Country/Region	Climate region	Soil type	Fertilisation	Tillage
Sugar cane ethanol (perenn)	Brazil	Tropical moist	Low Activity Clay Soils	Medium/high without manure	No tillage/reduced tillage
Maize ethanol	US	Cold and warm temperate, dry and moist.	High Activity Clay Soils		Full tillage/Reduced tillage
Sugar beet ethanol	EU			High with/without manure	Full tillage/Reduced tillage
Wheat ethanol					
Switchgrass ethanol (perennial)	US/EU			Medium	No tillage/reduced tillage
Wheat straw ethanol or Rapeseed biodiesel	EU			High with/without manure	Full tillage/Reduced tillage
Soya biodiesel	US			Medium	
	Brazil	Tropical moist	Low Activity Clay Soils		
Palm oil biodiesel (perennial)	Indonesia/Malaysia	Tropical wet	Organic Soils	Medium/high with manure	No tillage/reduced tillage
			Spodic Soils		
			Low Activity Clay Soils		
Sunflower biodiesel	EU	Cold temperate, moist	High Activity Clay Soils	High with/without manure	Full tillage/Reduced tillage
		Warm temperate, dry and moist			
Jathropha biodiesel (perennial)	Africa	Tropical, wet and moist	Low Activity Clay Soils	Medium	No tillage/reduced tillage
Miscanthus (perennial)	US/EU	Warm temperate, dry	High Activity Clay Soils		

Table A1.2 *Average, maximum and minimum DLUC emission values based on grassland to forest plantation conversion in gCO_2/MJ, for an allocation period of 30 years[11]. Assumed yield is 144 GJ/ha/yr. Ecological zone determines potential carbon stock of the plantation.*

Domain	**Ecological zone**	**Region**	**Min**	**Max**	**Avg**
Tropical	Rain forest	Africa	-42	-68	-55
		Americas	-30	-68	-49
		Asia	-25	-49	-37
	Moist deciduous forest	Africa	-23	-32	-27
		Americas	-15	-61	-38
		Asia	-18	-38	-28
	Dry forest	Africa	-12	-15	-13
		Americas	-12	-25	-18
		Asia	-12	-20	-16
	Scrubland	Africa	-1	-2	-2
		Americas	-4	-12	-8
		Asia	-4	-7	-6
	Mountain system	Africa	-12	-25	-19
		Americas	-10	-23	-16
		Asia	-9	-22	-16
Subtropical	Humid forest	Americas	-20	-65	-42
		Asia	-20	-42	-31
	Dry forest	Africa	-13	-16	-15
		Americas	-13	-27	-20
		Asia	-13	-22	-18
	Steppe	Africa	-2	-3	-3
		Americas	-5	-14	-10
		Asia	-2	-19	-11
	Mountain system	Africa	-13	-26	-20
		Americas	-11	-29	-20
		Asia	-10	-23	-17
Temperate	Oceanic forest	Asia/Europe	-45	-49	-47
		North America	-38	-42	-40
		New Zealand	-58	-62	-60
		South America	-21	-25	-23
	Continental forest and mountain systems	Asia/Europe	-41	-50	-46
		North America	-41	-43	-42
		South America	-24	-26	-25
Boreal	Coniferous forest and mountain systems	Asia/Europe	-7	-12	-9
		North America	-7	-12	-10
	Tundra woodland	Asia/Europe	-2	-5	-4
		North America	-2	-5	-4

[11] See Section 3.4 for more information on the amortisation period.

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