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Biomass Provision and Use, Sustainability Aspects

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Glossary

Biobased economy The biobased economy encompasses all activities of the bioeconomy with a focus on non-food applications and includes bioenergy and traditional and modern biomaterials [1].

Biobased material Biobased materials are materials that are completely or partially produced from biomass. It covers traditional material such as wood timber, pulp and paper, and textiles as well as novel biobased plastics, chemicals, natural fibers, pharmaceuticals, and cosmetics [2].

Bioeconomy The bioeconomy encompasses the production of renewable biological resources and their conversion in agriculture, forestry, fisheries, food, bioenergy, pulp and paper, and part of chemicals and biotechnological sectors [3].

Bioenergy Bioenergy covers the use of biomass for energy purposes. Traditional bioenergy is biomass consumed in the residential and small industries in developing countries for inefficient heating and cooking. The majority of biomass (around 80%) is used for traditional purposes [4]. Modern biomass covers efficient heating, electricity generation, and transport biofuels [5].

Carbon debt The loss of sequestered biogenic carbon per land area due to the initial harvest for bioenergy.

DLUC (abbreviation for direct land use change) Land use is changed from a previous use to the cultivation of feedstock for energy and materials.

ILUC (abbreviation for indirect land use change) Change of land use that is induced by bioenergy or biomaterial production but

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occurs geographically disconnected from the biomass feedstock production [6].

LCA (abbreviation for life cycle assessment) Life cycle assessment is a methodological approach to analyze and assess environmental impacts associated with all stages of a product's life from the raw material extraction through materials processing, manufacture, distribution, use, repair and maintenance, and disposal or recycling.

LUC (abbreviation for land use change) Land is converted from one use to another. This can refer to both direct and indirect land use change resulting from crop cultivation for energy and materials.

Definition of the Subject

The use of biomass for energy and material purposes (i.e., bioenergy and biobased materials) has been promoted for a number of reasons; the most important one relates to substituting fossil fuels to mitigate climate change, reduce dependence on finite fossil fuels, and increase diversity of energy sources. The key reasons have varied over time, across countries, and for different applications. European energy policy has long focused on increasing the share of renewable energy and reducing its energy impacts on the climate. Bioenergy, in the beginning particularly biofuels, has been seen to play a key role in achieving these aims. Biomass is renewable and as a starting point carbon neutral, as carbon is sequestered during growth of biomass and released when combusted. However, also the production of bioenergy and biobased materials causes greenhouse gas (GHG) emissions throughout the whole life cycle (cultivation, harvesting, transport, processing, distribution, use, and disposal). In addition, the possible temporal imbalance between the release and sequestration of forest carbon has raised concerns about the short-term climate mitigation potential and carbon neutrality of forest biomass for energy.

As a result, bioenergy and biobased materials are not fully carbon neutral by default. Large variation exists in the magnitude of GHG emissions from bioenergy and biobased material and therefore also in emission reductions compared to fossil fuels. This is a result of the multitude of biomass production systems and the complex integration of biomass supply chains in the much larger (bio)economy. The biophysical and socio-economic conditions of the location of production, the type of land use change resulting from feedstock cultivation, energy and material production technologies, the design of the supply chain, and the final use (including which fossil fuel or material use is replaced) are key determining factors for how bioenergy and biobased materials perform compared to their fossil counterparts.

This paper will analyze the GHG emissions performance of biomass for energy and material applications, provide an overview of the current progress in understanding the impacts from direct and indirect land use change as well as carbon debt, and discuss the current and possible future policies for addressing these challenges so that bioenergy and biobased material applications can contribute to climate change mitigation.

Introduction

In the last decades, deployment of modern bioenergy systems, in particular electricity generation, liquid biofuels, and efficient heating, has grown in many cases rapidly. Furthermore, next to traditional material sectors such as timber and pulp and paper, biomass is increasingly used as a feedstock for biobased materials that replace fossil materials such as bitumen, chemicals, and plastics. This development has been stimulated by policies to mitigate greenhouse gas (GHG) emissions and improve energy supply security by diverting from fossil fuels to renewable energy. Other motivations to stimulate modern bioenergy and novel materials include stimulation of local development by income and job generation or securement in forestry, agriculture, and other bioeconomy sectors [4]. Traditional applications of bioenergy, e.g., simple and inefficient wood stoves, generally have poor environmental

performances and create serious negative impacts on health and living conditions [5]. In contrast, modern bioenergy generally yields positive impacts on energy independence and reduces GHG emissions by direct substitution of fossil fuels [7]. However, when impacts over the whole life cycle (cultivation, harvesting, transport, processing, distribution, use, and disposal) are considered, the impact of various biobased supply chains for energy and materials differs greatly [5, 7].

The large variation in the direction (positive or negative) and the magnitude of the GHG impacts of biobased supply chains are the result of the multitude of biomass production systems and the complex integration of biomass supply chains in the much larger (bio)economy. Each of the system components, including feedstock production, logistics, conversion, and end use, is embedded in different social, economic, and environmental contexts. Biophysical and socioeconomic conditions of the location of production, previous land use, energy and material production technologies, the design of the supply chain, and the final use (including which fossil fuel or material use is replaced) are key determining factors [4].

Life cycle assessment (LCA) is a widely employed method to determine the environmental performance of bioenergy over the whole supply chain. It has been standardized by many LCA-based norms or technical specifications, of which the most widely applied are the ISO standards (ISO 14040: 2006, ISO 14044:2006 and ISO 14067:2013), PAS 2050:2011 (Publicly Available Specification), and GHG Protocols (for organizations). However, none of these methods provide guidance on more complex issues associated with GHG emissions of the production of bioenergy and biomaterials, such as indirect land use change and carbon debt, due to a lack of methodology and data. Clear conclusions on the GHG impact of bioenergy and biomaterials are therefore often lacking [7]. Nevertheless, given the heavy reliance of climate mitigation scenarios on the use of biomass for energy and materials, it is of utmost importance to understand the GHG impact of bioenergy and biomaterials in relation to its context conditions.

The aim of this paper is to (i) provide a consistent overview of the GHG emissions performance of biomass for energy and material applications based on existing LCAs, (ii) provide an overview of the current progress in understanding the impacts from indirect land use change and carbon debt, and (iii) discuss the current and possible future policies dealing with full-chain GHG emissions, ILUC, and carbon debt, in the context of the role of biomass in a low-carbon economy.

Besides the GHG performance of biobased supply chain, an increased use of biomass for energy and materials could have several other environmental and socioeconomic impacts such as deforestation, loss of biodiversity, impact on food security, etc. Clearly, for bioenergy and biomaterials to be sustainable, also these other impacts need to be considered. This requires an integrated analysis of all potential environmental and socioeconomic impacts and especially also the trade-offs between them [5]. However, as one of the key objectives of using biomass for energy and materials is to mitigate climate change, this paper focusses only on the GHG emissions.

Life Cycle Assessment of Bioenergy and Biomaterials: A Review

Life cycle assessment (LCA) is a tool to assess the environmental impacts of a given product or a service through its life cycle. The methodology has been standardized by the International Standardization Organization in ISO 14040:2006 [8] and ISO 14044:2006 [9]. LCA is the backbone methodology of many environmental footprint tools, such as life cycle GHG emissions (i.e., carbon footprint). According to the ISO standards, an LCA shall be conducted in four steps: (1) goal and scope definition, (2) life cycle inventory analysis (LCI), (3) life cycle impact assessment, and (4) interpretation. Widely applied LCA-based methods for GHG emissions accounting include PAS 2050:2011 [10], ISO 14067:2013 [11], and GHG Protocols [12].

Depending on the goal of an LCA, the inventory models can be distinguished into either attributional or consequential models.

- An *attributional* model, sometimes called "accounting LCA," uses historical, fact-based, and/or average data with known uncertainties and includes all the processes through the life cycle stages [13]. Typically, the results can be used for microlevel decision context, for instance, to help the producer to identify environmental hotspots for the potential supply chain or technology improvements.
- A *consequential* LCA is sometimes called "change-oriented" or "perspective" LCA. It is used to assess the impact by introducing changes in the product system. Consequential LCA is used in meso-macro level decision support [14]. It does not reflect the actual or average supply chain but gives insights into a hypothetical supply chain that is modeled based on known market mechanisms.

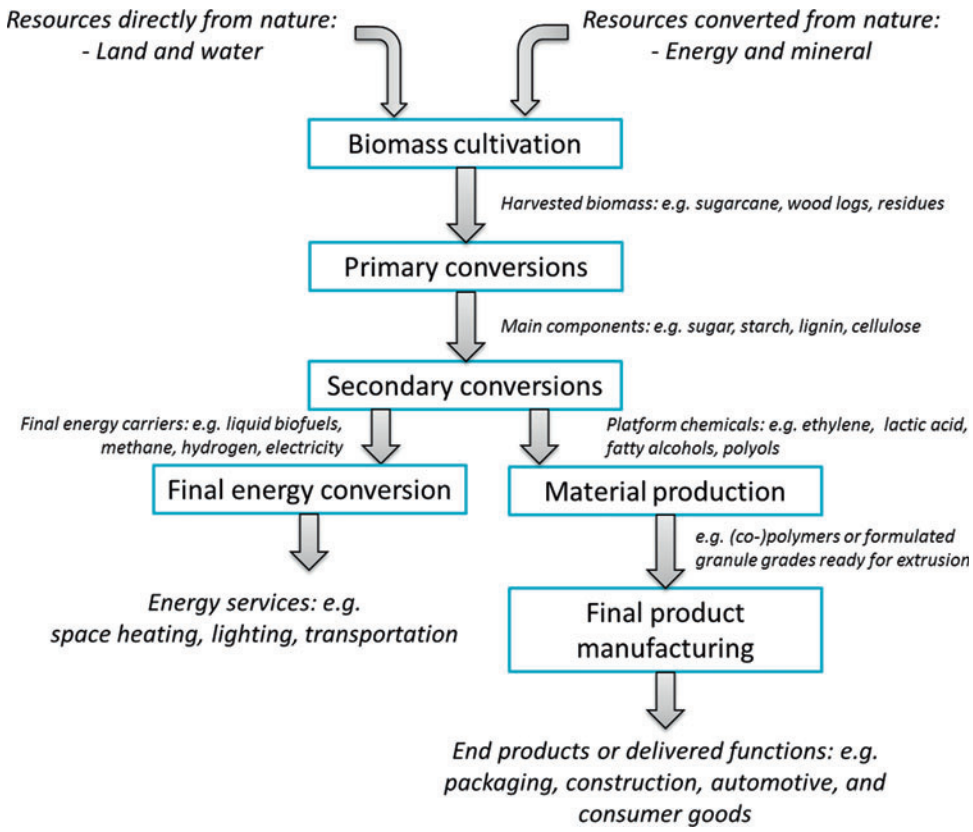
An emerging third type of LCA is the so-called *advanced attributional* LCA [13]. In practice, the clear line between attributional and consequential LCA is often found blurred. In many cases, LCA can be used to understand the environmental impacts of an existing technology or a supply chain, where fact-based, average data are used. However, when a constraint resource is introduced in the system, it is important to understand the impact "beyond the immediate system boundaries" [13]. For example, in a biobased economy, available land is considered a constraint resource. In such a case, additional GHG emissions associated with land use changes are also taken into account.

The life cycle of bioenergy and biomaterials can be briefly described in Fig. 1. The product systems include natural resources extraction, processes related to biomass cultivation, collection, pretreatment, transport, and conversion, and the final use of bioenergy and biomaterials comprises GHG emissions from land use (changes), the consumption of (fossil) energy, process utilities, and waste production. In this section, literature reviews of published LCAs on bioenergy (section "GHG Emissions of Bioenergy") and biomaterials (section "GHG Emissions of Biobased Materials") are provided.

GHG Emissions of Bioenergy

The life cycle GHG performance of bioenergy production systems has been studied extensively. Figure 2 shows ranges of major electricity, heat, and road transportation bioenergy systems found in mostly peer-reviewed literature published after 2010 and collected by Creutzig et al. [4]. Emissions from land use change are excluded and would likely increase overall life cycle emissions (see Fig. 5 in [4] and section "Land Use Change-Related GHG Emissions"). Renewable jet fuels (aviation) have been added from [15]. The studies that came to these results followed an attributional approach. The large ranges in bioenergy systems are therefore mainly explained by real system variations. Variability of the LCA method applied is mainly limited to allocation procedures, for example, energy versus economic allocation and the scope of the study [17]. Real system variations include, for example, agricultural management applied, the biophysical conditions of the location of production, the design and management of the supply chain, the conversion technologies, and the choice of process energy (fossil fuels vs. renewables).

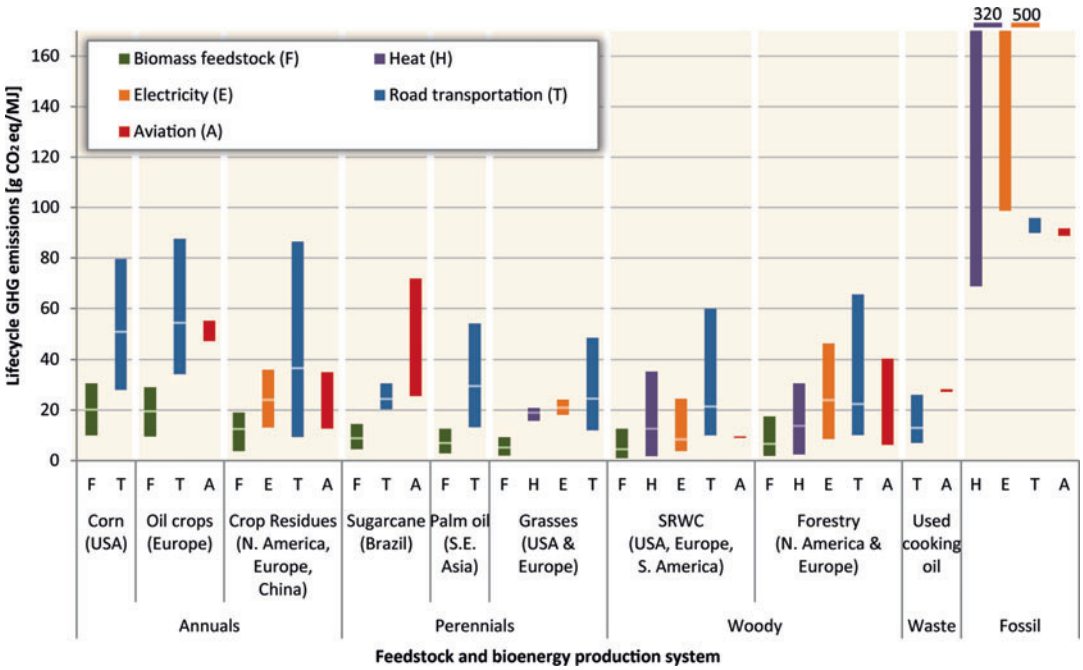
In addition, variations in outcomes are induced by differences in research methods applied including differences in system boundaries, allocation procedure, and uncertainties in data sets [17, 18]. Figure 2 excludes credits to account for coproducts and indirect or market-mediated effects such as land use change. These factors increase the uncertainty of LCAs significantly. Furthermore, production systems that are not yet commercially available use the so-called prospective analysis that is inherently more uncertain than existing systems based on retrospective analysis [17]. These and other uncertainties are not properly reflected in Fig. 2 in which narrow ranges are also the result of the limited number of cases included. For example, the literature base on the lifecycle GHG performance of renewable jet fuels is still small.



Biomass Provision and Use, Sustainability Aspects, Fig. 1 Generic flow scheme of cradle-to-factory gate bioenergy and biomaterial production

The biophysical conditions of the location of crop cultivation, management and related yields, fertilizer application, and soil nitrous oxide (N_2O) emissions are key factors that determine the GHG performance of feedstock production. Soil N_2O emissions, induced by fertilizer and manure use, are with 10–80% the largest contributor to direct GHG emissions of annual food crop-based biofuels [4, 19]. For that reason, systems that use organic waste or residues as a feedstock are found to have the lowest GHG footprint (highest savings). Forestry biomass varies by biogenic carbon fluxes and how they are accounted for over time. Short rotation wood crops (SRWC) and herbaceous (perennial) crops generally require less fertilizers and require less pesticides as they are less susceptible for plagues and diseases and have higher yields compared to annual food-based crops. Realistic yields of lignocellulosic energy crops are probably between 3 and 15 t/(ha a) (dry matter) depending on the climatic zone and if grown on land that is marginal for agriculture (land with a poor suitability for food production) [20]. Some annual crops, such as sugar beet crops, can have higher yields (11.6–15.8 t/(ha a) (dry matter)) [5] but only under good agroecological conditions [21].

Sugarcane ethanol, produced in Brazil, contributes to 20% to the world biofuel production by volume (27 Billion L in 2016) [22] and shows to have relatively low direct (attributional) GHG emissions compared to other food-based biofuels. If used for renewable jet fuels in aviation, the range becomes substantially larger because of the processing required to produce a drop-in biofuel from ethanol (alcohol-to-jet) or from fermentation to farnesene (direct sugars to hydrocarbon). The lower conversion efficiency and hydrogen



Biomass Provision and Use, Sustainability Aspects, Fig. 2 Greenhouse gas emissions of major bioenergy pathways (sum of CO_2eq. emissions, global warming potential over 100 years). Land use-related net changes in carbon stocks are excluded. The ranges describe the minimum and maximum values found in literature; markers indicate median values. Emissions from feedstock are in g CO_2eq/MJ LHV feedstock, and emissions from heat (H), electricity (E), road transportation (T), and aviation (A) are in g CO_2eq/MJ LHV of final product. Based on [4] with additional renewable jet fuels [15], fossil electricity and heat [5], and fossil transportation fuels [16]

consumption of renewable jet fuels are in most cases the result of the higher lifecycle GHG emissions compared to biofuels used in road transportation [15]. Note that emissions of aviation are considered to have a greater climate impact 2–2.7 times larger in terms of radiative forcing compared to ground-based emissions, among others by cloud formation, and ozone and methane production form NO_x emissions [23]. Those impacts are not considered in Fig. 2.

To the right, Fig. 2 also shows ranges of lifecycle GHG emissions of conventional fossil energy systems. The relative GHG savings of bioenergy largely depend on the type of fossil reference (fuel type, efficiency), especially in case of electricity and heat. GHG emissions from fossil electricity can be below 100 g CO_2eq/MJ from an efficient natural gas-fired combined-cycle plant (NGCC) and up to 500 g CO_2eq/MJ when supplied from an inefficient coal power plant [5]. The highest emissions from heat are from fossil high-carbon-intensive electric heating and lowest if heat is directly generated from natural gas [5]. Fossil road transportation fuels (gasoline and diesel) and aviation (kerosene) can cover conventional fuels and variations in allocating upstream emissions [16]. Nonconventional fossil oil sources such as oil from tar sands and coal-to-liquids (both not included in Fig. 2) have substantially higher emissions compared to conventional oil sources [24].

Model projections of climate mitigation scenarios show that biomass will be needed in all sectors of the biobased economy (electricity, heat, transport, and chemicals). However, because the highest GHG savings can be achieved when biomass is used to replace carbon-intensive electricity generation and heat, the GHG mitigation effectiveness of biomass will be the largest if amounts of biomass are allocated to these sectors [25]. However, the marginal effectiveness of biomass in heat and electricity sectors decreases

when the most carbon-intensive systems are replaced. Furthermore, increasing amounts of biomass will be needed in sectors including aviation, shipping, and non-energy uses that have little or no renewable alternatives to biomass. The optimal effectiveness of biomass in climate mitigation is therefore most likely a mix of end-use sectors [26]. The mix of mitigation options not discussed here includes biomass with CCS (abbreviation for carbon capture and storage) or BECCS (abbreviation for bioenergy with carbon capture and storage); marine biofuels and biobased materials are discussed in section "GHG Emissions of Biobased Materials."

GHG Emissions of Biobased Materials

Novel biobased materials have received increased attention due to concerns for finite fossil fuel resources, GHG emissions, and post-consumer plastic waste which persists in the environment. Together, the chemical and petrochemical sector and the oil and gas refinery sector account for at least 8% of total global GHG emissions in 2012 [27]. The lion's share of the emissions originates from the combustion of fossil fuels for the synthesis and conversion processes.

Worldwide, biobased plastics have been experiencing a rapid growth in the past decades. The global annual capacity of biobased plastics has increased from 0.36 Million t/a (metric tonnes) in 2007 [28] to approximately 3.4 Million t/a in 2016 [29]. The biobased plastics industry is still very small, accounting for only about 1% of the petrochemical-based, synthetic polymer industry (ca. more than 300 Million t/a).

A generic flow scheme of a cradle-to-factory gate LCA of biobased products is shown in Fig. 1. Similar to bioenergy production, the production of biobased material starts with biomass cultivation, logistics, and primary conversion steps such as purification and pretreatments where the main biobased components are extracted (e.g., sugar, starch, or cellulose). To date, sugarcane and corn are the two most important biomass feedstock for the established industrial scale production of biobased materials. The biobased material product systems usually involve more complex conversion steps compared to the bioenergy production systems. Via biochemical or thermal-chemical/catalytic processes, the main biobased components are converted into platform chemicals, which can be drop-in chemicals, for which the biobased chemical has the identical chemical structure as the petrochemical counterpart, such as ethylene and butylene, or innovative ones such as lactic acid or carboxylic acids. The platform chemicals are often the precursors of many polymers (or co-polymers) which provide the "backbone" structures of many end applications. For instance, lactic acid can be directly polymerized to form polylactic acid (PLA). By adding plasticiers, compatibilizers, and other additives, desired functions can be achieved for many polylactic acid (PLA)-based plastic applications, e.g., film or foam packaging, injection moldings for consumer goods, and extrusion for construction materials, as well as textile fibers.

Various LCA studies reported the GHG emissions of novel biobased materials. Table 1 shows an overview of cradle-to-factory gate GHG emissions of various biobased polymers compared to their petrochemical counterparts based on published LCAs. The comparisons in Table 1 are based on a kg material. For drop-in chemicals, since the chemical structure is the same as their petrochemical counterparts, such a comparison can be justified. However, for those with novel chemical structures such as polylactic acid (PLA), starch blends, and polyhydroxybutyrate (PHB), a justified comparison should be made on a functional unit basis. In the table, the typical material properties are provided next to the comparison of the GHG emissions. It can be seen that all three novel biobased materials, namely, starch blends, polylactic acid (PLA), and polyhydroxybutyrate (PHB), could achieve comparable material strength properties but also have relatively higher density compared to their petrochemical counterparts. This could lead to, but not strictly necessary, a heavier end product compared to those made from the petrochemical polymers.

Fossil polyethylene (PE), polypropylene (PP), and polyethylene terephthalate (PET) together account for more than 55% of European plastics production in 2016, excluding the fiber applications [47], and thus are important plastics to find

Biomass Provision and Use, Sustainability Aspects, Table 1 Comparing the technical performance and cradle-to-factory gate net GHG emissions of novel biobased materials

Type of material		Typical technical properties			Net GHG emissions reported in literature[a]					
	Composition[b]	Density (g/cm^3)	Young's modulus (GPa)	Tensile Strength (MPa)	LCA published by	Cradle-to-gate ***net*** GHG emissions[c] (kg CO_2 eq./kg)	Biogenic carbon embedded in polymer[d] (kg CO_2 eq./kg)	Key biomass feedstock	Sources of process energy	Level of technology assessed
Starch blends	TPS/PLA (68% biobased)	1.3	1.2	20–23	[30]	1.22	1.27	Potato	Fossil fuels	Commercial
	TPS/PHB (85% biobased)	1.3	n/a	20		1.49	1.42	Potato	Fossil fuels	Commercial
	TPS/PLA/PBAT (56% biobased)	1.3	1.7–2.0	24		2.47	1.09	Potato	Fossil fuels	Commercial
	TPS/PBS (18% biobased)	1.5	0.8–1.1	21		2.84	0.31	Potato	Fossil fuels	Commercial
PLA	100% PLA	1.2–1.4	2.7–16	53–100	[31]	0.4–1.3	1.83	Sugarcane or corn	Fossil fuels	Lab + process design
					[32]	1.24		Corn	n/a	Commercial
					[33]	0.62		Corn	n/a	Commercial
					[34]	0.5		Sugarcane	Fossil fuels + bagasse	Commercial
PHB	100% PHB	1.17–1.25	1.2–3.0	18–40	[35]	−6.06 to −1.94	2.04	Biogas from waste	Biogas (for the low value), fossil fuels (for the high value)	Lab + process design
					[36]	−2.3 to −1.4		Corn	Biomass residues	Lab + process design
						1.7–1.9		Corn	Fossil fuels	Lab + process design
					[31]	−1.1 to 3.4		Sugarcane	Fossil fuels + bagasse	Lab + process design
						1.9–6.4		Corn	Fossil fuels	Lab + process design
					[37]	1.96		Sugarcane	Fossil fuels + bagasse	Lab + process design
					[38]	−0.24		Soybean	Fossil fuels	Lab + process design
						1.39		Corn		Lab + process design

PE	100% biobased PE	0.88–0.96	0.15–0.2 (film grade)	10–18 (film grade)	[39]	−2.05	3.14	Sugarcane	Fossil fuels + bagasse	Conceptual design
						−0.34		Corn	Fossil fuels	Conceptual design
					[40]	−0.75		Sugarcane	Fossil fuels + bagasse	Commercial
					[41]	−2.1		Sugarcane	Fossil fuels + bagasse	Commercial
	Petrochemical PE				[42]	2.1	n/a	Naphtha	Fossil fuels	Commercial
PET	Biobased PET: 30% biobased content	1.3–1.4	2.7	52	[43]	1.03	0.69	Sugarcane	Fossil fuels + bagasse	Conceptual design
						1.36		Corn	Fossil fuels	Conceptual design
					[40]	2.2		Sugarcane	Fossil fuels + bagasse	Commercial
						1.9		Sugarcane and molasses	Fossil fuels + bagasse	Commercial
	Petrochemical PET				[44]	2.15	n/a	Naphtha	Fossil fuels	Commercial
PP	Biobased PP	0.91	n/a	33–35	[39]	−0.2 to −0.3	3.14	Corn	Fossil fuels	Conceptual design
	Petrochemical PP				[45]	2.0	n/a	Naphtha	Fossil fuels	Commercial
PTT	Biobased PTT (ca 30% biobased content)	1.35	n/a	66	[46]	2.5	0.64	Corn	Fossil fuels	Commercial
					[31]	1.6		Corn	Fossil fuels	Lab + process design
	Petrochemical PTT				[46]	3.2–3.6	n/a	Naphtha	Fossil fuels	Commercial

[a]Reported GHG emissions in this table do not include the impact from direct or indirect land use changes

[b]Abbreviations: *TPS* thermoplastic starch, *PLA* polylactic acid, *PHB* polyhydroxybutyrate, *PBAT* polybutyrate adipate terephthalate, *PE* polyethylene, *PP* polypropylene, *PET* polyethylene terephthalate, *PTT* polytrimethylene terephthalate

[c]Cradle-to-factory gate net GHG emissions are defined as the total GHG emissions from the combustion of fossil fuels, deducted by the biogenic CO_2 embedded in the material. In a cradle-to-grave LCA, this part of carbon will be released again in the atmosphere when the used materials is fully oxidized

[d]Biogenic carbon embedded in the material refers to the carbon with biogenic origin (sequestered by the biomass), expressed in CO_2 equivalence

biobased substitutes for that also deliver GHG benefits. Drop-in polymers, biobased polyolefins, and two major (partially) biobased polyesters, i.e., biobased polyethylene terephthalate (PET) and biobased polytrimethylene terephthalate (PTT), all have lower or similar GHG emissions than their petrochemical counterparts. Three novel biobased polymers are extensively reported, namely, starch blends, polylactic acid (PLA), and polyhydroxybutyrate (PHB). Both starch blends and polylactic acid (PLA) are currently produced on an industrial scale. Polyhydroxybutyrate (PHB) is in its early commercialization stage, and most published LCAs are still based on lab designs. The GHG emission ranges appear to be much bigger compared to those of the drop-in polymers based on the studies reviewed.

Some general observations can be made from Table 1:

- The type of biomass feedstock plays an important role. The most favorable biomass feedstock is waste/residues, and the least favorable one is corn. Sugarcane is somewhere in between and is so far the most common feedstock for commercial scale productions.
- The GHG emissions from the process energy used by the conversion steps, including primary, secondary and material conversions (Fig. 1), are significant. The use of biomass residues for process heat and electricity could substantially lower the GHG emissions [34, 35, 37, 40, 41, 48, 49].
- The assessment uncertainties are very often related to the technology development level. When the technology is in its early development stage, the assessments show large ranges, partly caused by higher uncertainties related to the conversion processes (e.g., yields of the reaction), whether or not process integration is implemented and to the efficiencies of the downstream processes [33, 35, 38, 40, 49].
- Like for bioenergy production systems, variations are also caused by the chosen LCA method. This is especially true for large-quantity agricultural by-products. For example, system expansion leads to lower GHG emissions allocated to by-products than physical partitioning (e.g., mass allocation) [35, 36, 48, 49].

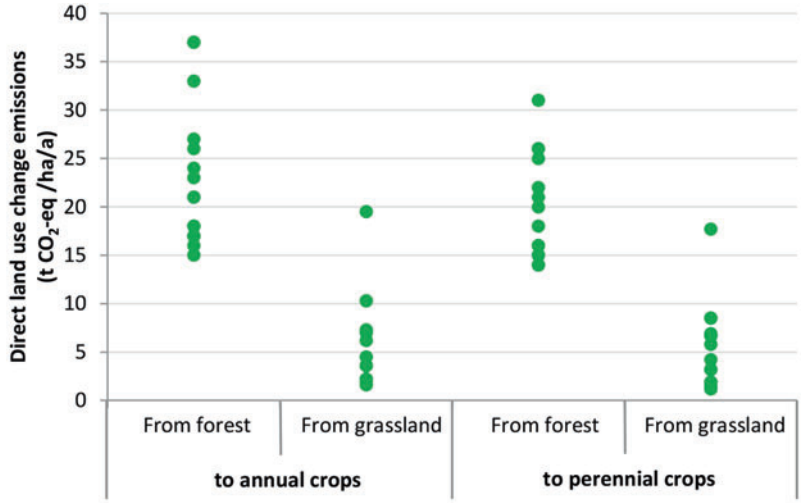
Based on the reviewed studies, biobased materials can offer great potential of GHG emissions reduction. However, it should be recognized that most published LCAs of biobased materials only reported the environmental impact from cradle-to-factory gate, although a large group of biobased plastics are designed for their biodegradability such as polylactic acid (PLA), polyhydroxybutyrate (PHB), and starch blends. The impacts of biodegradability are not captured by a cradle-to-factory gate analysis.

A full picture of the environmental impacts, not limited to GHG emissions and also including the impact of end-of-life stage, could be obtained by cradle-to-grave LCAs. However, the published LCAs covering a full life cycle scope are rarely reported for this group of products due to lack of empirical data and the complexity of the current waste management infrastructure across different countries.

On the feedstock side, the GHG emissions caused by land use changes (both direct and indirect) are often not included in a LCA due to lack of databases as well as methodology. Some recent development of inventory databases of direct land use changes, such as the *Agri-footprint* database [50], has made it possible to gain more insight into the consequences of land use changes in the transition of the material sector. A detailed overview can be found in section "LUC-Related GHG Emissions in LCA Studies."

Land Use Change-Related GHG Emissions

The cultivation of dedicated crops for bioenergy and materials results in direct land use change (DLUC): land use is changed from a previous use to the cultivation of energy and material feedstock. If another agricultural activity is displaced, this in turn can lead to indirect land use change (ILUC): a change of land use that is induced by bioenergy or biomaterial production but occurs



Biomass Provision and Use, Sustainability Aspects, Fig. 3 GHG emissions from direct land use change from natural land (forest or grassland) to crops for energy and materials (annual and perennial crops) in CO_2-eq per hectare per year for a selected set of countries (Argentina, Australia, Brazil, Canada, Finland, France, Germany, Indonesia, Malaysia, Mozambique, Pakistan, Poland, South Africa, Ukraine, UK, USA) assuming an amortization period of 25 years. The X-axis indicates the type of land use conversion, and the y-axis indicates the GHG emissions of LUC in tonne CO_2-eq per hectare per year (Based on the data of the Renewable Fuels Agency [57])

geographically disconnected from the biomass feedstock production [6]. In the following, the key challenges for quantifying land use change-related GHG emissions of biobased supply chains are discussed and explained.

GHG Emissions of LUC

The conversion of land from its original use to the cultivation of biomass for energy and materials could result in carbon sequestration (removing CO_2 from the atmosphere) or in carbon emissions, due a change in carbon stocks (above- and below-ground biomass, soil organic carbon (SOC), and dead organic matter) [51]. The loss of carbon due to the land use or the land management change is called the carbon debt [52]. Over time, energy or materials from biomass from converted land can repay this carbon debt if their lifecycle GHG emissions are less than the emissions of the fossil fuel-based alternatives they displace [52]. However, when this debt is very large, the carbon payback time will be long, and it becomes increasingly uncertain whether the carbon benefits will ultimately occur. The magnitude of the change in carbon stocks as a result of converting a certain land use to biomass cultivation for energy or materials depends on the former land use, the type of biomass cultivated after conversion, the management of land before and after conversion, and the local biophysical conditions such as soil and climate conditions [53]. Therefore, the LUC-related GHG emissions of biomass production for energy and materials are crop, management, and location specific [21].

Large carbon stock losses occur when natural land (e.g., forest and grass/shrubland) are converted to managed lands (e.g., cropland). The conversion of forest mainly results in the loss of biomass carbon, whereas the conversion of grass/shrubland results mainly in losses of soil organic carbon (SOC). Especially when natural lands are converted to intensively managed annual crops, high-carbon stock losses occur [54, 55]. In contrast, low-carbon stock losses or even carbon stocks gains can be achieved when marginal lands (low soil organic carbon (SOC), sparsely vegetated) or intensively managed croplands are converted to perennial crops or (short rotation) forest (plantations) [55]. The effect of the former land use on the LUC emissions is illustrated by

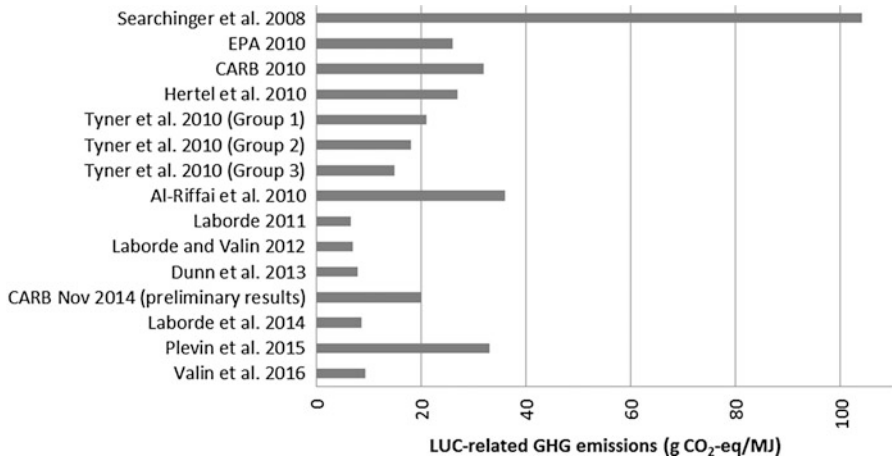
[24, 52, 56] showing that large GHG emissions (581–3452 Mg CO_2/ha) occur when oil palm replaces tropical rainforest but that no carbon losses or even carbon sequestration occur when cropland or (degraded) pastures are replaced by palm oil production. Figure 3 provides an example of the variation in GHG emissions from land use change converting natural land to crops for energy and materials. Higher GHG emissions occur when forest is converted, especially when converted to annual cropland. Lower GHG emissions occur when grassland is converted to perennial cropland; however, these can still be significant.

The local soil and climate conditions determine the impacts of land use change on the carbon balance. Natural-occurring soil organic carbon levels (without disturbance) vary from 10 to 146 t C/ha (for the 30 cm topsoil) depending on the soil type [51]. Also, the soil organic carbon (SOC) levels of soils are higher in cold temperate and moist climate conditions compared to the same soil types in dry climate regions. Furthermore, the soil and climate conditions affect the yield levels and therefore biomass carbon stock of the land use before and after conversion; e.g., in tropical rainforests the standing biomass carbon stock is much higher than forest areas in arid regions. The impact of local conditions on the GHG emissions from land use change is illustrated by Fig. 3, where a large variation in GHG emissions is shown for each specific conversion from natural land to cropland depending on the country the land use change takes place. This variation is the result of the heterogeneity in climate and soil conditions of the countries included in this figure.

In addition, the management applied in the previous land use and in the cultivation of biomass crops highly affects the changes in carbon stocks. For example, intensively managed annual cropland (e.g., high till, high residue removal rate) or degraded pastures (e.g., overgrazed, low inputs) generally correspond with low soil organic carbon levels, while extensively managed (e.g., no till) perennial crops or (short rotation) forest could foster high soil organic carbon (SOC) levels. There are also trade-offs when it comes to the impact of management decisions on GHG emissions; e.g., high residue removal results in lower soil organic carbon levels, while leaving large amounts of residues on the field could result in GHG emissions related to decomposition. Also, the application of nitrogen fertilizer contributes to sustaining soil organic carbon levels, but it also leads to N_2O emissions. The optimal residue removal and fertilizer application rates highly depend on local characteristics. Agricultural management decisions should therefore account for the spatial variation in biophysical factors.

Given the heterogeneity in land use that is converted, management applied, and climate and soil conditions, there is a large variation in LUC-related GHG emissions of biomass crop production for energy and materials. For that reason, it is not possible to provide a general GHG emission factor per type of biomass crop that is cultivated. Studies assessing the GHG impact of LUC using a statistical approach, and thus not accounting for the spatial variation in previous land use, soil and climate conditions, and the effect of management decisions, neglect the complexity and underestimate the large variation in GHG emissions resulting from land use change induced by biomass production for energy and materials. Comparing different studies on carbon losses due to land use change for biomass cultivation for energy and materials is not straightforward as the metric applied varies across studies (e.g., carbon loss per hectare, GHG emission per unit of bioenergy, payback period in years), emissions resulting from LUC and ILUC are sometimes combined, different allocation approaches are applied, various amortization periods are assumed, and it is not clear to what extent local conditions are accounted for.

However, in general it can be concluded that a high risk of large carbon losses exists when natural vegetation is converted to intensively managed cropland, especially when this is on organic or wetland soils. Low risks of carbon losses or even carbon gains can be achieved when intensively managed cropland or marginal land is converted to extensively managed perennial crops or (short rotation) forest (plantations). The effects of land use and land management changes (positive or



Biomass Provision and Use, Sustainability Aspects, Fig. 4 Direct and indirect LUC-related GHG emissions from corn ethanol (30-year allocation period) estimated by different economic modeling studies, sorted by date of publication (Updated from [60])

negative) are generally larger in moist climate regions and organic or other high-carbon soils.

High-resolution spatial analysis with high-quality data is required to properly estimate the changes in biomass and soil organic carbon. Although some local and national data on soil organic and biomass carbon stocks exist, so far there is no complete and up-to-data spatially explicit dataset available. The FAO Global Soil Partnership aims to develop a global soil organic carbon map at 1 km^2 resolution by the end of 2017 [58]. Liu et al. [59] developed a global map of aboveground biomass based on satellite images. These efforts help to identify where land use conversion to biomass production is likely to result in high-carbon losses or potential carbon gains. However, as both land use and soil organic carbon levels are highly dynamic, continuous efforts are required to improve these types of data sets and keep them up to date. This type of information is important for sound land use planning in order to avoid large LUC-related GHG emissions and to seek carbon sequestration possibilities.

GHG Emissions of ILUC

In addition to the effects of direct LUC on the carbon balance, a key point of discussion relates to indirect land use change (ILUC) effects. ILUC occurs when existing food or nonfood production is displaced or diverted to bioenergy or biomaterial feedstock production, while demand for the displaced products remains constant and/or when the additional demand results in higher crop prices that incentivize crop production elsewhere [60].

While direct LUC for biofuel feedstock production can be observed, measured, and relatively easily allocated to biofuel production, ILUC cannot easily be measured and allocated as an effect of biofuels. This is because it is not known whether and how land use change would have occurred in the absence of biofuel production. Therefore, quantification of ILUC and its effects needs to be modeled. GHG emission impacts of ILUC have been assessed with different models. Economic models (especially computable general equilibrium (CGE) models; e.g., [61–63]) are most often used as they can account for interlinkages between economic sectors and regions. But also partial equilibrium (PE) models (e.g., GLOBIOM [64]) and ILUC allocation approaches (e.g., [65]) have been applied. Partial equilibrium (PE) models do not account for all economic sectors and therefore do not consider all market-mediated effects. ILUC allocation approaches entail the risk of losing important details with respect to the complex market feedbacks and endogenous intensification [66].

Given the advantage of economic models to account for interlinkages between sectors and regions, most ILUC estimates are based on economic equilibrium models. Therefore, the following description of results and uncertainties focusses on these models. As economic models cannot distinguish between direct and indirect LUC effects, the results refer to total LUC-related GHG emissions induced by biofuels, also referred to as LUC factor, and are generally expressed as an emission factor per unit biofuel (in g CO_2-eq/MJ biofuel). Various studies have come to (largely) different LUC factors (Fig. 4 for LUC emissions from corn ethanol). But, while they differ in the exact results, all economic modeling studies agree that LUC factors are above zero and significantly lower than the values originally suggested by Searchinger et al. [67]. Differences between studies can be explained by the different assumptions made for key parameters.

Based on [68–70], the key modeling parameters can be identified as the (i) level of agricultural intensification (including the price-yield elasticity), (ii) amount of agricultural expansion (including the elasticity for land expansion) and yield on newly converted land, (iii) use of coproducts, (iv) consumption level (effect of price changes on food demand – the food price elasticity), (v) trade levels and policies, and (vi) type of land use/cover change and resulting emissions. Among these parameters, crop price-yield elasticity (through which the development in yields on existing cropland is modeled) has been identified as the factor that contributed by far the most to variation in LUC emission factors [70].

In addition, there are also large differences between bioethanol and biodiesel [60]. The comparison of biofuel types from different feedstocks shows that sugar and starch-based bioethanol performs significantly better than vegetable oil-based biodiesel [71]. Key reasons are that (i) demand for vegetable oil is less elastic than for cereals so that price increases have less effect on vegetable oil demand than on cereals; (ii) different vegetable oils are largely substitutable, and, because palm oil is the cheapest vegetable oil, the use of any vegetable oil for biodiesel results in palm oil production expansion; and (iii) oil palm cultivation is associated with high GHG emissions from deforestation and peatland conversion [72–78].

Uncertainty analyses of direct and indirect LUC factors (e.g., [62, 70, 79, 80]) indicate large ranges in outcomes. For example, as a result of varying the values of the key parameters listed above, the 95% confidence interval of the LUC-related GHG emissions from corn ethanol has been estimated to range from 18 to 55 g CO_2-eq/MJ with a mean of 33 g CO_2-eq/MJ [70]. The uncertainties in these parameters thus have great influence on the results but are difficult to decrease in the short term due to ILUC having to be modeled [68]. These models include complex interactions of multiple processes on local, regional, and global level and apply various inherently uncertain assumptions. In addition to parameter uncertainties, changes in the modeling structure may also largely change the size and distribution of the LUC factor [70]. Therefore, more research is needed to better understand model structure uncertainties (i.e., how the models are set up and structured, what parameters are used, and how they are defined) to improve approximation of the key parameters and to further develop modeling approaches for assessing ILUC. An additional point of uncertainty relates to projecting where in a country direct and indirect LUC will occur. Verstegen et al. [80] spatially explicitly quantified such uncertainties for Brazilian sugarcane production and found that it is difficult to project exactly where LUC will occur, emphasizing that there are more difficulties for ILUC than for direct LUC.

While uncertainties are large, ILUC remains a key factor in the GHG balance of first-generation biofuels, and it is similarly relevant also for second-generation biofuels, other energetic applications, and material purposes. Economic modeling studies have made first attempts at modeling second-generation biofuels. For example, LUC factors of 16 and 17 g CO_2-eq/MJ for Fischer-Tropsch-based biodiesel from agricultural or forestry residues, respectively, have been shown [64]. While no land is converted for residue removal, this factor is the result of reduced soil organic carbon buildup. Another study using a general equilibrium model (as opposed to the

partial equilibrium model used by [64]) estimated LUC factors of using corn stover for biofuels as 1.5 g CO_2-eq/MJ [81]. For bioethanol from short rotation plantations and perennial grasses, Valin et al. [64] find LUC factors of −29 and −12 g CO_2-eq/MJ as a result of increased carbon stocks on the land, indicating that energy crop-based second-generation biofuels may actually have a positive LUC effect. However, more research is needed on ILUC of second-generation feedstocks and their use for biofuels, electricity, and heat, as well as feedstock use for biomaterials.

In addition, given the significant effect on the overall GHG balance, also a better understanding of how ILUC can be mitigated is needed. Bottom-up assessments of strategies for ILUC mitigation have also been conducted. For example, [82–84] assessed how far ILUC mitigation is possible and what the role of different strategies for mitigating ILUC can be, i.e., improving agricultural crop and livestock yields, restricting land use change to only degraded and marginal lands with low-carbon stocks, using residues, and reducing losses in the supply chain. The results from the different case studies indicate that ILUC can be largely, and in some cases completely, mitigated with these strategies. Yield increases and the use of degraded and marginal land are the most promising options in terms of increasing the biomass resource potential without displacing other uses of land. However, increasing yields come with the risk of increased emissions, e.g., additional nitrous oxide emissions from fertilizer applications, indicating the importance of how agriculture is intensified. Gerssen-Gondelach et al. [85] showed that only if intensification is done sustainably, bioenergy can reduce enough emissions compared to fossil fuels to be in line with the proposed EC goal of 70% emission reduction [86]. The performance of the agricultural sector is thus key for a strong GHG emission performance of bioenergy. Therefore, an integrated perspective on land use for agriculture and bioenergy is needed [87]; see also section "Policies Addressing GHG Emissions of Bioenergy and Biobased Materials."

LUC-Related GHG Emissions in LCA Studies

Previously, the GHG emissions related to LUC were often ignored in life cycle assessment (LCA) of bioenergy and biomaterials due to lack of databases and methodologies. In recent years, LCA studies have tried to account for these LUC-related GHG emissions wherever data is available. For example, Tsiropoulos et al. [40] reported cradle-to-factory gate GHG emissions of high-density polyethylene (HDPE) using bioethanol from Brazil and polyethylene terephthalate (PET) produced from bioethanol from Brazil and India (see the results also in Table 1). In addition, the study reported the influence of including DLUC and ILUC by using existing DLUC and ILUC factors found in literature [60].

It is found that the influence from DLUC is rather limited for biobased high-density polyethylene (bio-HDPE) and biobased polyethylene terephthalate (bio-PET). The additional GHG emissions caused by DLUC only account for less than 3% (bio-HDPE) and 0.2% (bio-PET) of the emissions compared to the results without DLUC. However, when including ILUC by taking into account the ILUC factors found in literature [60], the GHG emissions of both bio-HDPE and bio-PET would increase [40]. For bio-HDPE, the additional GHG emissions (approx. 0.16–0.38 kg CO_2eq./kg bio-HDPE) do not change the conclusion that bio-HDPE still offers GHG emissions savings, largely due to the large biogenic carbon storage credit (−3.2 kg CO_2 eq./kg bio-HDPE). In contrast, the GHG emissions of bio-PET are dominated by the chemical conversion processes [40]. Although the additional emissions caused by ILUC are relatively low (0.03–0.4 kg CO2 eq./kg bio-PET), by including ILUC the benefit of the biogenic carbon storage credit of bio-PET (−0.45 kg CO_2 eq./kg bio-PET) is canceled out. This implies that bio-PET may even lead to an increase in GHG emissions compared to the petrochemical counterpart.

Many recently published life cycle inventory databases such as Agri-footprint version 3.0 [50] and Ecoinvent version 3.3 [88] also include DLUC factors using standard values per type of crop, making it possible to see more future LCAs taking into account emissions from DLUC.

However, to include ILUC into LCA remains a challenge due to available spatially explicit data and interpretation of the ILUC model uncertainty. Furthermore, ILUC is more complicated to be considered in LCA because of the different perspectives: consequential vs. attributional.

Currently, guidelines on LCA of biofuels and bioenergy differ in how they deal with DLUC and ILUC. The recent guidelines and input data on calculating the default GHG emissions from biofuels in the EU Renewable Energy Directive include DLUC, but not ILUC. The ISO standard ISO/FDIS 13065 on the sustainability criteria for bioenergy [89] refers to the ISO standard ISO/TS 14067:2013 [11] for the quantification of the carbon footprint. This standard requires the calculation of the DLUC-related GHG emissions in accordance with the tier 1 approach of the IPCC [51]. In this approach, the change in default carbon storage in the land use prior to the bioenergy project to that after the bioenergy project is quantified and amortized over 20 years. Given the complexity, uncertainty, and the lack of consensus on the calculation of the GHG emissions related to ILUC, it is not required to calculate and report on ILUC-related emissions according to the ISO/TS 14067:2013 standard.

Carbon Accounting for Wood-Based Energy

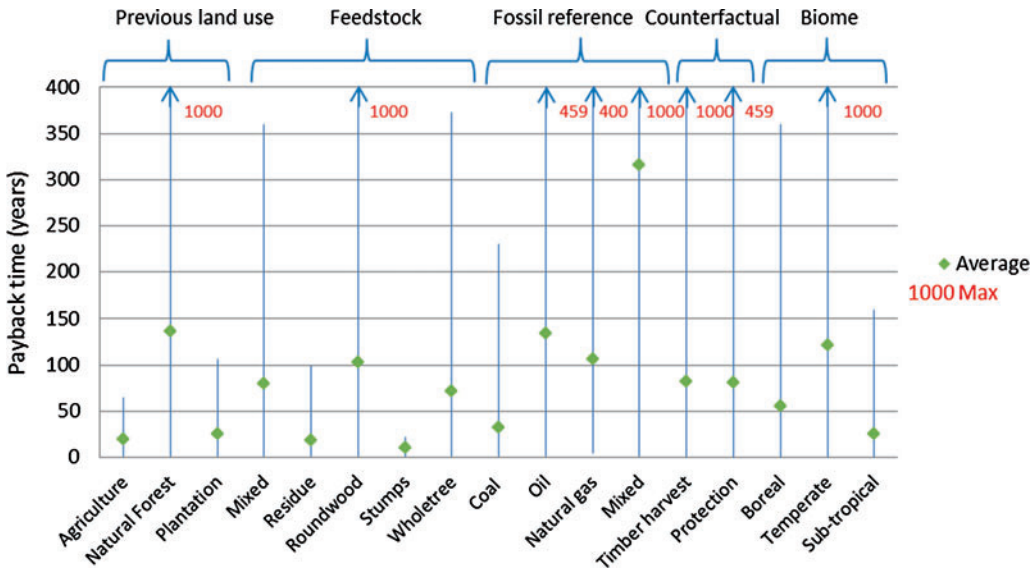
The concept of carbon debt addresses two issues typically not covered by LCA's. First, on LCA, all emissions are assumed to take place in year 0, i.e., emission profiles over time are simplified, and temporal aspects are usually omitted. Second, in most LCAs, comparisons are often made with a reference energy or material system, e.g., comparing bioethanol to gasoline or bioplastics to fossil oil-based plastics. However, many comparative LCA studies do not include what the alternative land use reference is. The ILUC approach (discussed in the previous section) investigates what indirect land use effects of expanding land use for bioenergy elsewhere are. However, when forests (including plantations) are used for bioenergy (often in addition to already existing use for materials), a management change occurs, often starting with a removal of carbon stocks. As in forestry, harvesting cycles can take several decades (opposed to much shorter rotation times for most agricultural crops), and most forests are important carbon sinks; many recent scientific studies have analyzed carbon flows over time for woody bioenergy systems and have compared them to the same scenarios with no bioenergy or other management changes.

While studies that take into account both the temporal and the (alternative) land use aspects have already been published in the mid-1990 (e.g., [90, 91]), two studies published in 2010 [92, 93] triggered a wider debate among scientists and between other stakeholders regarding the use of woody biomass for energy. The fundamental reason for this debate is the fact that a tree can theoretically be harvested and used for energy in a matter of weeks, rapidly releasing CO_2 to the atmosphere. But it can take over 100 years (in boreal forests) for a new tree to re-sequester this carbon, whereas in a hypothetical reference scenario, this tree might still have sequestered carbon till the tree has reached maturity and dies (after which it would have decomposed, releasing CO_2). This potential temporal imbalance and the uncertain climate benefits of using wood for electricity production combined with the increasing use of wood pellets in, e.g., coal power plants in Northwestern Europe, have been highlighted by NGOs [94, 95] and have led to a substantial debate among scientists (e.g., [96–100]).

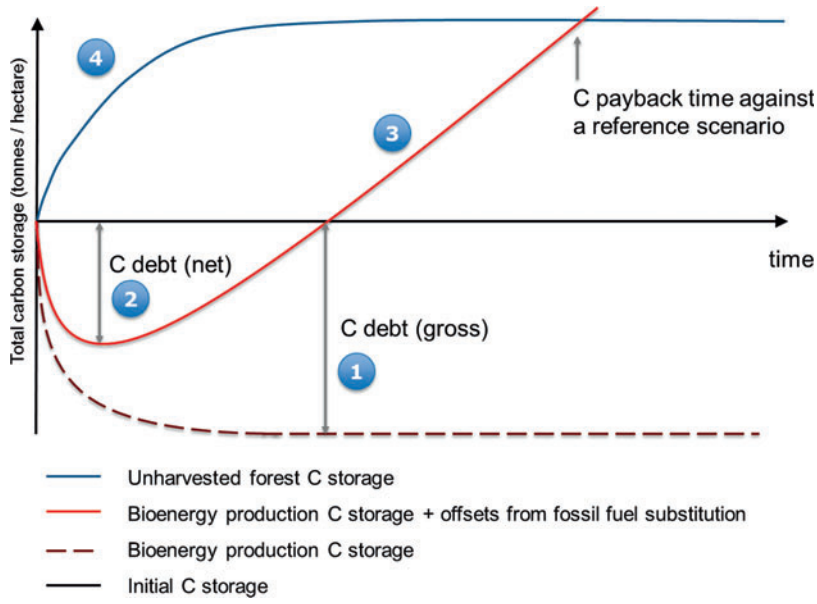
The time it takes before the use of wood for bioenergy achieves higher GHG emission savings than a reference scenario is called payback time and can range from 0 to 1000 years [96, 97, 101, 102]. Figure 5 gives an overview of these payback times found in literature.

However, this "payback" time highly depends on a number of factors:

- First, it depends on the previous land use and the amount of carbon removed in a bioenergy scenario, creating a gross carbon debt (see point 1 in Fig. 6). If the previous land use contained little carbon, such as agricultural land, payback times are typically much shorter



Biomass Provision and Use, Sustainability Aspects, Fig. 5 Average and range of carbon payback times in years across a number of independent variables (Data source: [101]). *Red* numbers indicate maximum payback times found in literature; *green* diamonds the average payback times



Biomass Provision and Use, Sustainability Aspects, Fig. 6 Conceptual representation of forest carbon accounting over time including a fossil and forest reference system, based on Mitchell et al. [103]. 1 Carbon stock change (due to harvest). 2 Reference energy system (counterfactual fossil fuel use). 3 Speed of carbon accumulation (after harvest). 4 Reference forest system (counterfactual wood/land use)

than if a natural forest is replaced by a bioenergy plantation (Fig. 5).

- Second, the fossil fuel replaced determines the net carbon debt (see point 2 in Fig. 6). Replacing dirtier fuels (such as coal) yields shorter payback times than, e.g., natural gas (Fig. 5).
- Third, the speed with which the carbon uptake of a new bioenergy plantation takes place (point 3 in Fig. 6) depends among others on the biomass; plantations in subtropical

climates grow much faster than in boreal climates (Fig. 5).

- Fourth, the choice and definition of the counterfactual scenario (what would have happened with no bioenergy production, point 4 in Fig. 6) have a major impact. It could be that the previous land use is maintained (e.g., protection in a natural forest), which typically causes longer payback times. But if the counterfactual is, e.g., conversion of forests to agriculture, then payback times are short.

Other impacts not shown in Figs. 5 and 6 are whether the effect of wildfire and other natural disturbances are included and whether leakage effects are included (also known as indirect wood use change). Lamers [97], Bentsen [101], and Buchholz [102] also point out that the model used also determines to a large extent the outcomes.

On top of this complex debate, while the metric of carbon payback times is used by most scientific studies (using cumulative or average CO_2/GHG emissions), several other metrics have been proposed, such as cumulative radiative forcing [104] and various forms of a GWP_{bio} factor (e.g., [105–107]), which (opposed to carbon payback times) also consider the impact of timing of emissions on the warming potential [108].

To illustrate the complexity of the carbon debt debate, the Southeastern USA is taken as an example. This region is currently a major source of wood pellets to replace coal in the Netherlands, the UK, Belgium, and Denmark [109, 110]. In 2014, the predominant source for pellet production was softwood from pine (both from natural stands and plantations, about 45%), followed by mill residues (about 35%) and hardwood (almost exclusively natural stands, about 20%) [109]. The use of especially pellets from softwood from natural pine stands and hardwood from bottomland swamp forests to produce electricity in EU power plants has been heavily criticized by NGOs, because of uncertain (and possible) long carbon payback times and related limited GHG benefits and possible impacts on biodiversity [111]. Counterfactual scenarios could for example include that (i) the forest is allowed to grow further; (ii) forest biomass is harvested for material purposes only (e.g., pulp and timber) and replanted, and material suitable for wood pellet production is left on site to decay or burnt in the field; or (iii) forest biomass is harvested for material purposes and afterwards converted to agriculture or urban development. Studies show that depending on the specific counterfactual scenario, using wood from pine plantations results in typical payback times between 0 and 50 years [112–114]. In an even wider set of scenarios evaluated for the UK government, emissions from electricity production from wood pellets can range from achieving negative emissions (net carbons sequestration) to being worse than burning coal [115]. But in a recent evaluation, it was found to be extremely difficult to determine the likelihood of these scenarios. If anything, low-carbon scenarios in the Biomass Emissions and Counterfactual (BEAC) model where deemed the more likely sources of fiber for pellets [116].

An effect that could be triggered by additional feedstock demand for wood pellets is that forest owners change their management strategies with the aim to produce additional woody biomass while minimizing carbons stock losses. If the productivity of existing plantations for bioenergy feedstocks can be increased through changes in management, this may result in short payback times (0–23 years [113]). By changing the planting density and management regime, and by also using tops and branches for energy, up to 25% additional woody biomass can be harvested for bioenergy without incurring any major carbon debts [117], while also not compromising the existing output of timber products, and at economically competitive cost levels.

Which mix of counterfactual scenarios is going to occur is difficult to predict, but it has a major influence on the time period until any carbon debt is considered to be repaid. Finally, a key aspect to consider is that more than 85% of the forests in the Southeastern USA are owned by private landowners [118]. The vast majority of these land owners use their forests for multiple purposes like recreation and hunting but ultimately as a form of investment through wood production.

However, different segments of the housing market and also of the pulp and paper markets (the two major markets of wood from the Southeastern USA) have either remained stable or declined in the past decade [119]. Without markets for wood, conversion to agriculture or use for urban development becomes more likely. In fact, until 2060, up to one quarter of the current forested area in the Southeastern USA is projected to be deforested because of urban expansion [120]. The sustained, long-term demand for wood for pellet production can help slow this trend down [98]. Galik and Abt [121] capture market responses to increased pellet demand using a subregional timber supply equilibrium model and show that "the market is responding positively to the imposition of pellet demand, but not so much that inventory is substantially reduced (thus driving net carbon storage down). The increased demand for biomass to produce pellets leads to an increase in forest rent, reducing the pressure on conversion of existing forests to some other, lower carbon use (e.g., agriculture). As the forests that are most likely to be lost to these other uses tend to be mature (and therefore high carbon) stands, any reduction in the rate of loss will yield a sizable increase in carbon storage" [121]. In a scenario with increased pellet demand, total forest area is expanded, but in some cases also management changes occur, potentially causing loss of biodiversity, as e.g., conversion of natural longleaf pine stands to loblolly plantation is seen as a major threat to a number of endangered species [122]. However, these land use change and management changes are likely going to differ from state to state [123].

Future Directions

On the basis of the review in sections "Life Cycle Assessment of Bioenergy and Biomaterials: A Review," "Land Use Change-Related GHG Emissions," and "Carbon Accounting for Wood-Based Energy," a short review of currently related policies and some suggestions for future policy directions are provided (section "Policies Addressing GHG Emissions of Bioenergy and Biobased Materials"), and some recommendations for future research are given (section "Future Directions for Research").

Policies Addressing GHG Emissions of Bioenergy and Biobased Materials

Given potential competition between various applications for sustainable biomass feedstocks (REF to book chapter on ▶ Bioenergy within the Global Energy Systems – Current and Future Contribution) and differences in their GHG emission performance (sections "Life Cycle Assessment of Bioenergy and Biomaterials: A Review," "Land Use Change-Related GHG Emissions," and "Carbon Accounting for Wood-Based Energy" of this chapter), policy must aim to simultaneously achieve large reduction of GHG emissions in the supply chain, mitigate emissions from land use change (particularly indirect effects), and meet increasing demand for biomass. Therefore, policymakers in collaboration with other stakeholders need to clearly define boundary conditions for GHG emission performance targets for bioenergy and biobased materials. Based on this, it can be decided which biomass feedstock is preferred, where and how to produce it, how it is converted, and how it is used. At the moment these boundary conditions are typically addressed by means of sustainability criteria that generally have a strong focus on GHG emission reduction. In this context:

- Current policies focus on full-chain GHG performance of bioenergy, including direct land use change (DLUC; section "Policies on Direct GHG Emissions").
- Indirect land use change (ILUC) is dealt with in a more qualitative and indirect manner (section "Policy Implications of ILUC").
- Carbon debt issues are still not (explicitly) part of biomass policy making (section "Implications of Temporal Accounting of Carbon for Forest Biomass").

In these three sections, it is also indicated in which directions policy improvements could be found and how policies can provide incentives to improve the performances of specific biobased

chains on the three elements (Although implications of the use of biomass for novel biomaterials in terms of temporal carbon accounting and ILUC have not been studied in detail, these challenges are just as relevant for biomaterials. Thus, while we focus on the description on bioenergy, the implications are similar for biomaterials).

Policies on Direct GHG Emissions

Existing GHG emission reduction criteria differ for different applications of biomass and across countries and regions. In Europe, criteria for biofuels are defined in the European Renewable Energy Directive (RED; 2009/28/EC), which requires minimum GHG emission savings (including direct land use change (DLUC) and supply chain emissions) of 35% today increasing to 50% in 2017 and 60% for new installations in 2018. In the newly proposed renewable energy directive (RED II [86]), the 60% threshold is proposed to be increased to 70%. However, this 70% emission reduction is based on a higher reference GHG emission of fossil fuels, which comes down to a GHG emission for biofuels comparable to the currently required 60%. Savings of 35% can be met easily by most systems when net carbon emissions from land use change are not considered, but the future 60% and 70% thresholds will stimulate better performing biofuels and improving the GHG performance of existing chains.

Also in other countries, biofuel targets set in energy terms (e.g., an amount of energy or a share of total energy demand in transport) are accompanied by specific greenhouse gas criteria that biofuels should meet in order to count toward the target. For example, the US Renewable Fuel Standard (RFS) comes with specific GHG thresholds (compared to a fossil reference) for four types of biofuels: 20% for ethanol from corn starch, 50% for biomass-based diesel and "advanced biofuels," and 60% for lignocellulosic biofuels. In LCA, calculations need to take into account direct and significant indirect emissions, including land use change [124]. Within the USA, the State of California has implemented different approach (i.e., a low-carbon fuel standard (LCFS)). This sets a GHG intensity target to all transport fuels, not a target in energy terms combined with GHG criteria, making the GHG performance of biofuels a more integrated element of the policy instrument [73]. This is roughly comparable with the German policy in this respect [9]. Also in Canada, national policy has a biofuels target with GHG thresholds, while some states apply an LCFS instrument [125, 126].

Solid biomass used for power, heating, and cooling are currently exempted from GHG criteria in the EU RED. However, some member states have already implemented sustainability criteria, including for GHG emission savings, in national schemes [127]. For example, UK support for heat under the Renewable Heat Incentive and for electricity (>1 MW) requires a minimum GHG saving of 60%. This is scheduled to increase to 75% for new installations by 2025 [128]. In the Netherlands, it was recently agreed on minimum GHG emission savings of 70% for heat and electricity compared to EU reference values [129]. Setting increasing targets over time both incentivizes the development of cleaner technologies, e.g., second-generation lignocellulosic biofuels, and provides industry with certainty which targets (both quantitative volumes of bioenergy and GHG emission reduction) they have to achieve in the longer term. This limits the regulatory risk and makes long-term investments in better performing technologies possible.

Also in other regions of the world, GHG standards for solid biomass are less popular than for liquid biofuels. One other example is the US State of Massachusetts in which solid biomass applications must lead to at least 50% GHG emission reduction, also taking into account a carbon debt factor [130].

The use of biomass for biomaterials is not yet addressed in GHG emission criteria or broader sustainability criteria in Europe or elsewhere. However, only applying GHG emission reduction criteria for energy end uses could lead to unwanted leakage effects, e.g., the use of inefficient food crops for biomaterials, and disregard the complexity of end-of-life waste management of biobased materials. If biomaterials are to help reduce GHG emissions compared to petrochemicals, boundary conditions must also be defined for

biomaterials, and the same challenges that bioenergy is facing now need to be addressed also for biomaterials in future policy making.

The following recommendations for future policy directions can be formulated.

- In general, chain-only GHG standards do not cover the full picture as they do not take into account ILUC and carbon debt effects. But generally increasing the minimum saving criteria or making other policy changes can stimulate better GHG performance of biofuels or the use of better performing systems. For example, Germany recently changed its system for biofuels from an energy-based quota system to a GHG-based quota system to increase GHG reduction in transport from 3.5% in 2015 to 6% in 2020. This stimulates the use of better performing supply chains, which in turn incentivizes improvements in the GHG emission performance. But this change does not ensure that Germany reaches its 10% renewable transport energy target by 2020 [131] and could push biofuels with a weaker GHG profile to other member states.
- Next to full-chain GHG criteria, specific support measures to develop and implement innovations that led to better GHG performing chains can be useful accompanying policy. In this respect, key parts of the chain are feedstock mobilization and conversion into energy carriers, which may require specific incentives for stimulating such innovations.

Policy Implications of ILUC

Given ILUC effects generally lead to extra GHG emissions, even if exact values remain uncertain, the question is how to deal with ILUC. Some studies [68, 80, 132] have concluded that economic modeling results are not appropriate for determining (I)LUC emission factors to be used in policy making because of these uncertainties. Although other researchers agree that results should not be used for single point estimates, they consider these (I)LUC emission factors (and particularly their distribution curves from uncertainty analyses) still useful for policy making [70, 133, 134]. This is because these factors and distribution curves can illustrate effects of possible pathways, which can be used for identifying risks and opportunities of these pathways. These differences in opinions and interpretations can also be found in policy making. For example, in the USA, national and California-state regulations have already implemented LUC factors; for example, in California crop-specific DLUC and ILUC emission factors are based on the macroeconomic general equilibrium model GTAP [135]. In Europe, at present the potential use of LUC factors in legislation is still debated. For now, the European Parliament decided that a LUC factor that includes ILUC will not be introduced (i.e., ILUC will not be counted toward the minimum GHG emission savings threshold of 60% for new installations in 2018) [136]. Instead, ILUC emissions must be included in reporting on GHG emissions from biofuels under the EU RED only [136].

In addition to introducing a LUC emission factor that accounts for ILUC, several alternative and/or complementary approaches have been proposed to reduce the risk of ILUC. A key option is a combination of promoting feedstocks that rely less on land and reducing the risk of unwanted LUC [134]. In Europe, this approach was recently agreed on by capping the contribution of food-based biofuels 7% of final energy use in road transport [136]. And in the RED II proposal [86], food crop-based biofuels are not allowed to count toward a specific transport target any more, only toward the general renewables target, and that with a cap that is gradually reduced from 7% to 3.8% [86]. This combination will strongly reduce the incentive to produce food crop-based biofuels. Capping food crop-based biofuels is considered adequate because they are generally thought to have higher LUC emissions than lignocellulosic crops. This is a result of lower yields and lower carbon stocks [5]. However, this approach (as well as the LUC emission factor approach) fails to account for ILUC being the direct LUC of another activity. For example, in Brazil sugarcane production has expanded at the cost of other crop and livestock production, which in turn expanded by clearing rainforest in the Amazon [47, 135, 137]. Only if this other activity

is also incentivized to minimize its LUC, can ILUC be tackled. Furthermore, generic capping of biofuels from food crops does not differentiate between those biofuels with very high LUC factors (think of palm oil) and those with LUC factors that only modestly reduce their climate benefits (e.g., cereals).

Therefore, other strategies to reduce ILUC focus on taking a sustainable approach to all land use, whether for food or nonfood production. This entails sustainably increasing productivity and resource efficiency of all crop and forestry production, and appropriate zoning of land for all purposes (including protection of some areas and identification of other areas for production). Implementing such an approach in practice would mean only allowing those biofuels that do not displace other uses and that do not convert high-carbon stock land [87, 138]. Key strategies to making additional biofuels available with low risk of causing ILUC are (i) increasing agricultural productivity by modernizing and sustainably intensifying agriculture especially in the currently low-yielding areas through good fertilizer management, integrated pest protection, reduced or no tillage, and the prevention of monocultures (i.e., intercropping), (ii) applying underutilized land for additional food and bioenergy feedstock production, and (iii) zoning of land that should not be converted for any application, whether food or nonfood, and its strict enforcement [87]. Although this is not an easy task, only an integrated perspective in policy making that covers all applications, functions, and services of land can actually tackle ILUC. This approach has the advantage that the environmental performance of the agricultural sector as a whole can be significantly improved (including reducing its currently large carbon footprint), and total production for food and nonfood purposes can be increased.

Our conclusion is that generic capping of broad clusters of biofuels is an adequate way to prevent major ILUC impacts only in the short term, but this approach is not adequate in the longer term. The innovative approaches that prevent ILUC discussed above should be exposed to adequate policy incentives to be further developed and implemented. For this, concrete options would be:

- The opening of exceptions in a generic capping system for those types of feedstocks of which scientific evidence is sufficiently convincing that LUC factors are small
- Specific land use programs aiming at the development and implementation of sustainable intensification, including intercropping, and the mobilization of underutilized land
- Development of more integrated regulations for land use, land rights, and protection of conservation areas that accounts for all land uses

These innovative approaches do not only deserve support because of their impact on LUC factors of biofuels from food crops but also for keeping LUC factors of woody and other lignocellulosic crops under control if these are to play the role that most long-term scenarios foresee.

Implications of Temporal Accounting of Carbon for Forest Biomass

The overall GHG reduction potential of woody biomass depends on carbon debt and the associated payback time (section "Carbon Accounting for Wood-Based Energy"). The question what payback times are acceptable for bioenergy options to make meaningful contributions to mitigating climate change is still debated among policymakers and other stakeholders. The size of any carbon debt and the time that is required to repay it depend to a large extent on the type of fossil fuel that is assumed to be replaced, the changes in forest management that would occur, and the relevant counterfactual scenario(s). While in the UK, attempts have been made to quantify these to support policy design [115], so far no policy measures to address carbon debt have been taken. This is largely due to the large number of possible scenarios (and the uncertainty how likely they are) and consequent substantial variation in results. In the absence of a framework to quantify carbon debts of woody biomass that is broadly agreed on, it may still be possible to address carbon debt risks by relatively simple measures. For example, the recent Dutch covenant by industry and NGOs aims to mitigate the

risk of high-carbon debts and limit the competition with material applications from wood pellets used in Dutch power plants [139]. To this purpose, the covenant, for example, (i) defines specific feedstock categories, (ii) limits the share of biomass used for energy purposes to less than 50% of the total woody biomass extracted in any given year and area, (iii) disallows conversion of (seminatural) forests into plantations after 2008, and (iv) requires written evidence that C-stocks in forest must be maintained or increased [129]. While such criteria are still vulnerable to misconduct, they significantly limit the risk of stimulating the use of feedstocks with high-carbon debt risks and instead allow the development of sustainable woody biomass production and supply chains.

As with ILUC, a mix seems necessary between simple, suboptimal policy mechanisms that prevent the worst at the short term and further development of knowledge and innovations that allow for a more refined treatment of the issue at the longer term. And for LUC effects as well as carbon debt considerations, bioenergy should be part of the development of a wider framework for land use, land use change, and land management, which encompasses products and services that land can provide.

Future Directions for Research

Based on the review in sections "Life Cycle Assessment of Bioenergy and Biomaterials: A Review," "Land Use Change-Related GHG Emissions," and "Carbon Accounting for Wood-Based Energy," some general lines for future research are shown as well as the way policy makes use of scientific insights. Our three key messages are:

1. Biomass and sustainability is a complex matter, but it should not paralyze progress.
2. Better use should be made of existing knowledge, and "new" sectors should learn from previous experiences.
3. Biomass sustainability efforts should integrate knowledge and policies on forestry and agriculture for all purposes, including food, feed, fiber, energy, and materials.

Ultimately, the overview of the state-of-the-art research provided in this chapter shows that a lot of progress has been made on identifying promising biomass supply chains and strategies to optimize the performances. Steps must now be taken to not only avoid potential impacts but also develop the opportunities of biomass production for modern energy and material purposes.

Moving Forward in a Complex Field

Developing a sustainable biobased economy will take time and requires substantial efforts and large investments from the public and private actors in not only the bioenergy and biomaterials sectors but also in the agricultural and forestry sector as a whole. It will also require increased cooperation and strong commitment of different stakeholders from the government, industry, civil society, and academia at different scales (from local to global). Only if these stakeholders work together can sustainable solutions to increasing demand for bioenergy and biomaterials be found and implemented and large GHG emission reductions be achieved. While this will take time, it is important to continue developing sustainable production of the most promising biomass supply chains in the meantime. This should be done under stringent monitoring of GHG emissions and other sustainability criteria – with a chance of corrective action needed in the future as part of the ongoing learning process and new scientific insights. At this point, it is important to stress that to a large extent, future impacts of biomass production for energy and other purposes are inherently uncertain (e.g., due to arbitrary methodological choices, limited data availability, and the inherent uncertainties of future global development). However, simply excluding biomass from the energy and material mix (and ensuing failure to sufficiently rapidly mobilize sustainable biomass resources for a biobased economy) until all scientific uncertainties are resolved is both unrealistic and may impose risks in itself; namely, the risk of inaction for climate change and all related impacts. Also, more research should be directed toward optimal land use for the overall production of food, materials, and energy rather than looking separately at these end uses. In parallel, uniform sustainability

criteria should increasingly be applied to all land use and land use change, irrespective of the end use, to avoid negative impacts of, e.g., deforestation, food insecurity, loss of carbons sinks, and biodiversity which are typically driven by a variety of factors (of which demand for biomass for, e.g., biofuels can be one).

Make Better Use of Existing Knowledge and Lessons Learned

Developing a sustainable biobased economy will need to make better use of existing knowledge. Many studies show that it is possible to assess the overall GHG emissions of potential biomass supply chains taking into account the type of feedstock, the design of the supply chains, the fossil reference it replaces, and the biophysical and socioeconomic conditions in the region of production. Therefore, it is possible to differentiate between biomass supply chains that do contribute to GHG emission reductions and supply chains that do not, without lumping them all together and label biobased as a good or a bad option.

Also, concerning policy making and setting sustainability criteria for biomass supply chains lessons learned should be taken into account, as the sustainability issues raised for bioenergy are equally important for biomaterials. Only applying GHG emission reduction criteria for energy end uses could lead to unwanted leakage effects due to ILUC and carbon debt impacts. If biomaterials are to help reduce GHG emissions compared to petrochemicals, boundary conditions must also be defined for biomaterials, and the same challenges that bioenergy is facing now need to be addressed also for biomaterials in future policy making. It is recommended to further develop methodologies that can be used to assess the environmental performance of specific biobased products (and of the simultaneous production of biobased materials and bioenergy, e.g., in biorefineries). Even though this is laborious and often challenging from both a scientific point of view and from the view of actual implementing such schemes, it is deemed preferable over no assessment at all or a "one size fits all approach."

Biobased Policy as an Element in a Broader Movement Toward Sustainability

Biomass production can only be sustainable if it is part of an integrated approach on sustainable land use which includes different uses, functions, and services. This approach has the advantage that the environmental performance of the agricultural and forestry sector as a whole can be significantly improved (including reducing its currently large carbon footprint), and total production for food and nonfood purposes can be increased. Attributing occurring negative impacts, e.g., deforestation, food insecurity, loss of carbons sinks, and biodiversity, solely to bioenergy diverts the attention away from other, often more fundamental causes of this, such as poor protection of forest areas (causing illegal logging), poorly developed protection of land rights (causing, e.g., land grab), and imperfectly functioning markets and trading schemes (allowing land and commodity speculation and suboptimal agricultural policies). The biobased economy should be developed in such a way that it is part of a wider movement toward a solution to these problems but should not be considered the sole responsible for these problems.

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