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A Review of the Environmental Impacts of Biobased Materials

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Summary

Concerns over climate change and the security of industrial feedstock supplies have been opening a growing market for biobased materials. This development, however, also presents a challenge to scientists, policy makers, and industry because the production of biobased materials requires land and is typically associated with adverse environmental effects. This article addresses the environmental impacts of biobased materials in a meta-analysis of 44 life cycle assessment (LCA) studies. The reviewed literature suggests that one metric ton (t) of biobased materials saves, relative to conventional materials, 55 ± 34 gigajoules of primary energy and 3 ± 1 t carbon dioxide equivalents of greenhouse gases. However, biobased materials may increase eutrophication by 5 ± 7 kilograms (kg) phosphate equivalents/t and stratospheric ozone depletion by 1.9 ± 1.8 kg nitrous oxide equivalents/t. Our findings are inconclusive with regard to acidification (savings of 2 ± 20 kg sulfur dioxide equivalents/t) and photochemical ozone formation (savings of 0.3 ± 2.4 kg ethene equivalents/t). The variability in the results of life cycle assessment studies highlights the difficulties in drawing general conclusions. Still, common to most biobased materials are impacts caused by the application of fertilizers and pesticides during industrial biomass cultivation. Additional land use impacts, such as the potential loss of biodiversity, soil carbon depletion, soil erosion, deforestation, as well as greenhouse gas emissions from indirect land use change are not quantified in this review. Clearly these impacts should be considered when evaluating the environmental performance of biobased materials.

Introduction

Biobased wood, paper, and textile materials have been produced for centuries. Together, these materials account for 14% of global bulk materials production, whereas synthetic materials, predominantly produced from fossil fuel-based feedstock, account for a 7% share (estimates based on the work of Deimling et al. 2007; IAI 2010; Lasserre 2008; OGJ 2007; Saygın and Patel 2010; UN 2008).[1] Concerns about greenhouse gas (GHG) emissions and the security of industrial feedstock supplies have triggered relatively recent interest in also substituting biomass for conventional fossil fuel-based feedstock in the production of synthetic materials (Deimling et al. 2007; Patel et al. 2006; Shen et al. 2009a, 2009b). As a consequence, the production of biobased synthetic materials such as polymers, lubricants, and fibers has grown continuously in the past decade. As of 2008, biomass already provided 10% of the feedstock of the European chemical industry (Rothermel 2008). Biobased polymers, such as alkyd resins or polylactic acid, accounted for 7% of the total polymer production (PlasticsEurope 2007), while biobased

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plastics (comprising polymers with a molecular mass greater than 20,000 unified atomic mass units) are still in their infancy: by the end of 2007 they accounted for only 0.3%, or 0.36 megatons (Mt) of the worldwide plastics production (Shen et al. 2009a, 2009b).[2] Technological innovation will likely continue to open a wide range of new applications for biobased materials (Hermann et al. 2010; Shen and Patel 2008, 2009a). Breakthroughs can be expected in the coming years for integrated biorefineries, which may optimize the use of biomass by providing a whole range of materials and energy products from biobased feedstock.

However, the prospects for novel biobased materials present scientists, policy makers, and industry with an environmental challenge: the benefits of replacing fossil fuel-based feedstock and reducing GHG emissions may come at the cost of additional land use and related environmental impacts. Strategic decision making thus requires a thorough analysis of all environmental impacts of biobased materials in comparison with their conventional fossil fuel-based or mineral-based counterparts. To this end, life cycle assessment (LCA) has been applied to a large range of biobased materials, including starch-based polymers (e.g., Dinkel et al. 1996; Patel et al. 2006; Würdinger et al. 2002), fiber composites (e.g., Müller-Sämann et al. 2002; Wötzel et al. 1999; Zah et al. 2007), and hydraulic oils and lubricants (e.g., Reinhardt et al. 2001). Detailed reviews of the LCA literature revealed an initial focus on nonrenewable energy use and GHG emissions only (e.g., Dornburg et al. 2003; Kaenzig et al. 2004) that grew to include additional environmental impact categories such as eutrophication and acidification (e.g., Deimling et al. 2007; Oertel 2007; Weiss et al. 2007). A comprehensive quantification of the environmental impacts associated with a large range of biobased materials, however, is still missing. Here we address this problem by presenting a meta-analysis that summarizes the results of the existing LCA literature on biobased materials. We do not claim absolute completeness with respect to all published LCA studies. Instead, we seek to identify general patterns in the environmental impacts of a wide range of biobased materials. This article excludes technical, economic, and social aspects (e.g., production costs or the impact of nonfood biomass farming on food prices and the livelihood of smallholders in the tropics). These aspects should be considered, however, in a comprehensive evaluation of biobased materials.

Background Information and Methodology

Biobased materials comprise materials that are produced partially or entirely from biomass, that is, from terrestrial and marine plants, parts thereof, as well as biogenic residues and waste. Biobased materials include traditional wood, paper, and textile materials, as well as novel biobased plastics, resins, lubricants, composites, pharmaceuticals, and cosmetics. The manufacturing processes of biobased materials range from extraction and simple mechanical processing of natural fibers to fermentation and advanced enzymatic or catalytic conversions.

The environmental impacts of biobased as well as conventional fossil fuel-based or mineral-based materials are typically quantified through the internationally standardized LCA methodology (ISO 2006a, 2006b). Here we review the LCA literature on biobased materials by applying (1) a standard Internet search for peer-reviewed articles contained in the online databases “Web of Science” and “Scopus,” as well as (2) a Google-based search for scientific and governmental reports, workshop documents, and working papers. We include LCA studies if they provide a minimum of methodological background information, quantify the environmental impacts of biobased materials in physical units for at least one impact category, and are published in the English or German languages before December 2011. LCA results presented in flyers or oral presentations are excluded from this review. Likewise, we exclude LCA studies that report environmental impacts in percentages or indices only (e.g., Gironi and Piemonte 2011; Guo et al. 2011). This approach does not allow complete coverage of all relevant LCA studies published to date; still, it is suitable for identifying the general pattern in the environmental impacts of biobased materials.

We include in our review a total of 44 LCA studies that cover about 60 individual biobased materials and 350 different life cycle scenarios (see tables S1 and S3 in the supporting information available on the Journal’s Web site). Preliminary scanning through the literature reveals that most studies focus on biobased materials of European origin being manufactured by both small pilot installations and large-scale industrial plants. The reviewed LCA studies generally differ from each other in many, if not all, assumptions and choices made regarding, e.g., system boundaries, functional units, life cycle scenarios, or allocation procedures. They also differ in the detail of explanation of the methodology and results. We refrain from correcting for differences in choices and assumptions. This approach is justified given the scope of this meta-analysis and the generally limited availability of necessary background information.

Individual LCA studies may also differ from each other in the method chosen for aggregating inventory data into individual environmental impact categories. The commonly used methods have been described by Heijungs and colleagues (1992), the German Federal Environment Agency (UBA 1995), and Guinée (2001). Our review covers the environmental impacts of biobased materials in six categories, which we characterize and quantify as follows (Guinée 2001):

- nonrenewable energy use (NREU), quantified in gigajoules (GJ);
- climate change, quantified in metric tons of carbon dioxide equivalents (t CO_2-eq) by considering the global warming potential of GHG emissions over a time horizon of 100 years;
- eutrophication, quantified in kilograms of phosphate equivalents (kg PO_4-eq);

- acidification, quantified in kilograms of sulfate equivalents (kg SO_2-eq);
- stratospheric ozone depletion, quantified in kilograms of nitrous oxide equivalents (kg N_2O-eq); and
- photochemical ozone formation, quantified in kilograms of ethene equivalents (kg ethene-eq).[3]

We express the relative environmental impacts of biobased materials in comparison to the environmental impacts of conventional materials as

$$D_{ij} = EI_{biobased,ij} - EI_{conventional,ij},$$

where

D_{ij} = the difference in the environmental impact of the biobased and conventional material;
$EI_{biobased,ij}$ = the environmental impact of the biobased material;
$EI_{conventional,ij}$ = the environmental impact of the conventional material;
i = the specific material; and
j = the specific environmental impact category.

This approach follows the methodology applied in most of the reviewed LCA studies and results in negative values if biobased materials exert lower impacts on the environment than their conventional counterparts. We quantify the relative environmental impacts of biobased materials per metric ton of product and per hectare of agricultural land and year (ha*a).[4] The latter metric allows us to obtain insight into the land use efficiency of biobased materials. Several LCA studies report impacts for other functional units (e.g., Hermann et al. 2010; Madival et al. 2009). We include these studies in our meta-analysis and recalculate the environmental impacts based on the background information provided in each respective study. If the information in the respective LCA studies is insufficient for estimating the relative environmental impacts of biobased materials, we use information from additional literature sources in the following manner (see also table S1 in the supporting information on the Web):

- A few LCA studies on plastics present the environmental impacts of the biobased materials only (e.g., Vink et al. 2007). In these cases, we calculate the difference in the impacts of biobased and fossil conventional materials based on data provided by Boustead (2005a, 2005b, 2005c, 2005d; see table S1 in the supporting information on the Web).
- The reviewed LCA studies typically express the environmental impacts of biobased and conventional materials by using product-based functional units (e.g., 1 t or 1 square meter [m^2] of material). To express the findings of part of these studies also in terms of per hectare and year, we include in our meta-analysis the review results of Haufe (2010) and Weiss and Patel (2007).

This approach ensures a relatively complete coverage of published LCA studies.

We estimate stratospheric ozone depletion based on N_2O emissions only and exclude chlorofluorocarbons that are of minor importance since their banning by the Montreal Protocol in 1994 (Müller-Sämann et al. 2002; Würdinger et al. 2002). We present the results in each impact category disaggregated for nine individual groups of biobased materials and as a total over all materials. We report the arithmetic mean and the standard deviation of the environmental impacts for each group of biobased materials. The standard deviation can be regarded as indicative of the uncertainty interval in our results and reflects, to some extent, the diversity of life cycle scenarios and methodological choices made in the reviewed LCA studies. We calculate the overall average impact of biobased materials in each impact category based on the mean environmental impacts of the individual groups of biobased materials (see table S1 in the supporting information on the Web). This approach ensures an equal representation of all nine groups of materials in the overall result.

We put our findings in perspective by normalizing the overall average environmental impacts of biobased materials based on the worldwide average inhabitant-equivalent values for the year 2000 with regards to primary energy consumption (EIA 2011) and the five environmental impact categories (Sleeswijk et al. 2008) covered here. We discuss additional environmental impacts and secondary effects of biobased materials alongside the uncertainties of our review in semiquantitative terms after presenting the results.

Results

Nonrenewable Energy Use and Climate Change

We find that biobased materials save, on average, 55 ± 34 GJ/t and 127 ± 79 GJ/(ha*a) of nonrenewable energy. These savings exceed the worldwide average per capita primary energy consumption in the year 2000 by a factor 8 ± 5 and 18 ± 11, respectively. Furthermore, biobased materials save, on average, 3 ± 1 t CO_2-eq/t and 8 ± 5 t CO_2-eq/(ha*a) of GHG emissions relative to conventional materials (figure 1 and table S2 in the supporting information on the Web). This is equivalent to, respectively, 37 ± 21% and 111 ± 79% of the worldwide average per capita GHG emissions in the year 2000.

The results vary across large ranges. This makes it infeasible to identify individual groups of biobased materials that are environmentally superior with respect to nonrenewable energy use and climate change.

Special attention has been paid in recent years to the production of biobased chemicals. Patel and colleagues (2006) conducted an extensive cradle-to-factory gate (C-FG) analysis on the nonrenewable primary energy use and the GHG emissions of a wide range of biobased chemicals (figure 2 and table S3 in the supporting information on the Web). Their findings are in line with our results for a larger range of biobased materials, indicating that the production of biobased chemicals saves,

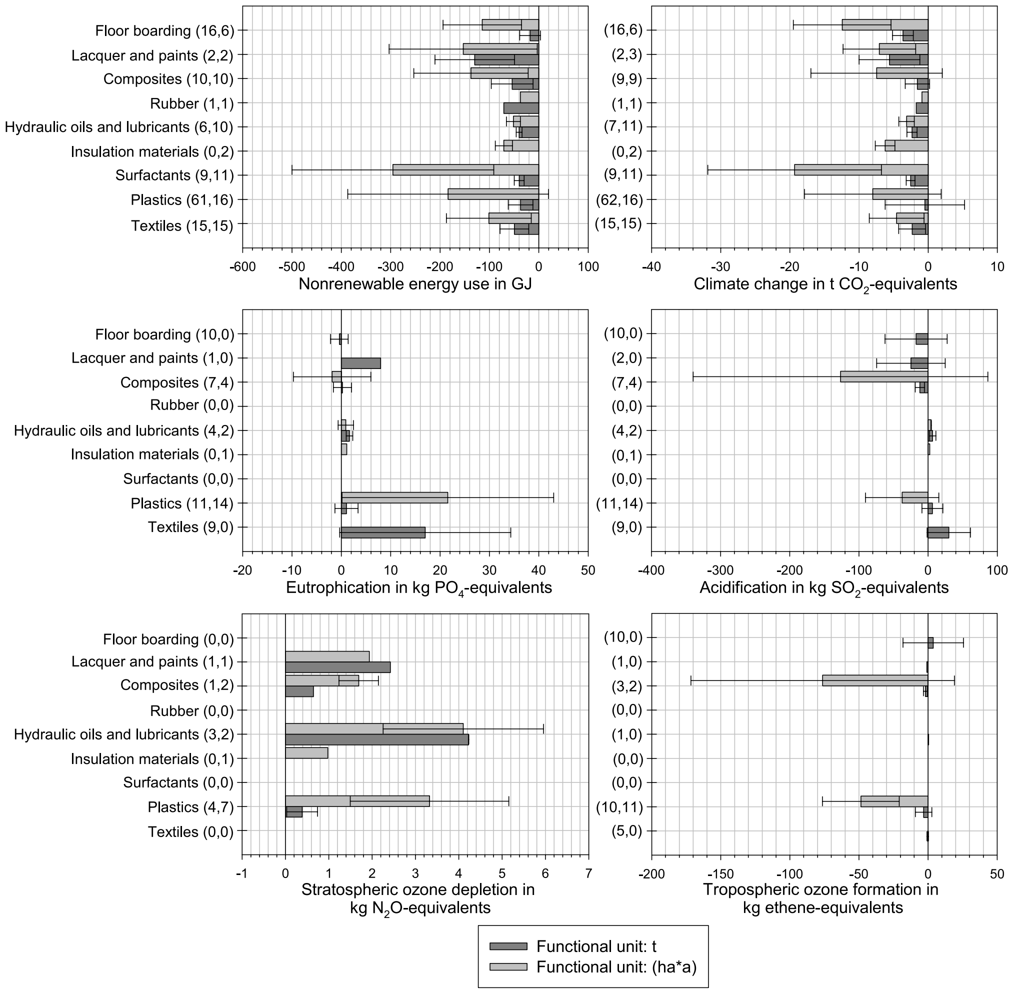

Figure 1 Average product-specific environmental impacts of biobased materials in comparison to conventional materials (D_{ij}). Uncertainty intervals represent the standard deviation of data. Numbers in parentheses indicate the sample size for the functional units of per metric ton and per hectare and year, respectively.

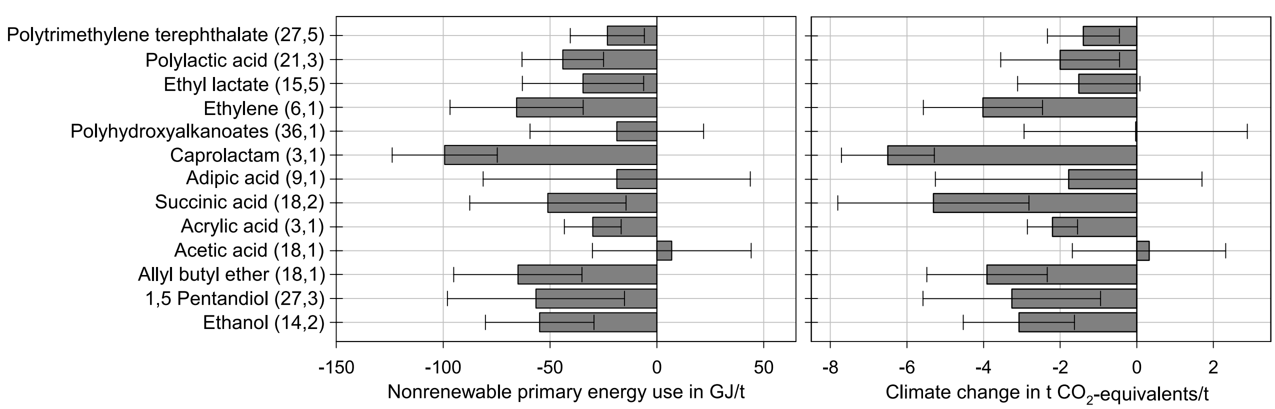

Figure 2 Average nonrenewable primary energy use and greenhouse gas (GHG) emissions of biobased chemicals in comparison to conventional chemicals (D_{ij}). Uncertainty intervals represent the standard deviation of data. Numbers in parentheses indicate the sample size for the biobased and conventional chemicals, respectively. Source of data: Patel and colleagues 2006.

on average, 43 ± 27 GJ/t and reduces the GHG emissions by 3 ± 2 t CO_2-eq/t in comparison to conventional fossil fuel-based chemicals. The variability in the results for individual biobased chemicals partially stems from differing assumptions regarding the type of biomass feedstock and the applied production technology. The GHG emission savings identified so far, however, disregard the potentially substantial effects of indirect land use change (see the Discussion section).

Eutrophication and Acidification

Biobased materials may induce, on average, 5 ± 7 kg PO_4-eq/t and 6 ± 11 kg PO_4-eq/(ha*a) higher eutrophication potentials on the environment than conventional materials (figure 1). These additional impacts account, respectively, for 66 ± 98% and 79 ± 157% of the worldwide average per capita freshwater eutrophication caused in the year 2000. The high eutrophication potentials mainly result from biomass production with industrial farming practices that causes nitrate and phosphate leaching from the applied nitrogen fertilizers, as well as ammonia emissions from manure applications (e.g., Cherubini and Jungmeier 2010; Deimling et al. 2007; Würdinger et al. 2002).

Our findings are inconclusive with respect to acidification, indicating that biobased materials relative to conventional materials may save 2 ± 20 kg SO_2-eq/t and 39 ± 61 kg SO_2-eq/(ha*a). This result translates, respectively, into a factor of 0.3 ± 2.9 and 5.7 ± 8.9 of the worldwide average per capita acidification potential in the year 2000. The relatively large uncertainty intervals indicate that acidification is case specific. Biobased plastics and composites, for example, seem to decrease acidification, whereas biobased lubricants are likely to increase acidification relative to their conventional counterparts (e.g., Müller-Sämann et al. 2002; Reinhardt et al. 2001). Acidification is mainly caused by emissions from the application of manure and mineral fertilizers in agriculture, as well as from combustion processes. The first source is relevant for biomass production in general, the second is predominantly for lubricants. Even if, e.g. biobased composites are incinerated after use, the resulting emissions may constitute only a minor portion of the total acidifying emissions along their relatively long product life cycle.

Stratospheric Ozone Depletion and Photochemical Ozone Formation

A limited number of seven LCA studies indicates that biobased materials may increase stratospheric ozone depletion by, on average, 1.9 ± 1.8 kg N_2O-eq/t and 2.4 ± 1.3 kg N_2O-eq/(ha*a) relative to their conventional counterparts (figure 1). The additional impacts thereby account, respectively, for 28 ± 26% and 35 ± 18% of the worldwide average per capita ozone depletion potential in the year 2000. The impacts in this category largely result from N_2O emissions that originate from fertilizer application in agriculture (Müller-Sämann et al. 2002; Würdinger et al. 2002). Because fertilizer application is characteristic for industrial farming used for growing biomass, high stratospheric ozone depletion potentials may be found for a wide range of biobased materials.

Our findings are largely inconclusive with respect to photochemical ozone formation, indicating that biobased materials may save, on average, 0.3 ± 2.4 kg ethene-eq/t and 62 ± 20 kg ethene-eq/(ha*a) as compared with conventional materials (figure 1). These savings account, respectively, for 5 ± 35% and a factor of 9 ± 3 of the worldwide average ozone formation potential in the year 2000. The large uncertainty intervals warrant caution and suggest that impacts in this category are case specific. Substantial parts of the uncertainty stem from LCA studies on wood floor boarding, which typically show a high ozone formation potential in comparison with both conventional floor boarding and other biobased materials. The impacts stem from volatile organic compounds that are emitted from solvents contained in the glues and surface finishing of parquet floors (Nebel et al. 2006). Parquet floors are included in the analysis of the product-specific photochemical ozone formation potentials, but not in the analysis of the land use-specific potentials. This causes relatively large uncertainty intervals in the first analysis but not in the second one.

Additional Environmental Impacts

In addition to these results, biobased materials exert a large variety of environmental impacts that are not quantified by most LCA studies, and thus by this review. Several LCA studies suggest that biobased materials may

- exert lower human and terrestrial ecotoxicity as well as carcinogenic potentials than conventional materials (e.g., Corbière-Nicollier et al. 2001; Harding et al. 2007; Shen and Patel 2010; Wötzel et al. 1999; Würdinger et al. 2002), and
- exert higher aquatic ecotoxicity than conventional materials (Shen and Patel 2010).

Additional land use-related impacts, such as water consumption for biomass cultivation, soil erosion, soil carbon losses, and changes in biodiversity, have received recent attention (e.g., Geyer et al. 2010a, 2010b). These impacts are predominantly relevant at the local and regional scales, but are difficult to quantify and are therefore excluded from both the majority of LCA studies and this review.

Discussion

Our findings confirm the results of previous LCA reviews that cover a smaller group of biobased materials and fewer environmental impact categories (Dornburg et al. 2003; Kaenzig et al. 2004; Patel et al. 2003; Weiss et al. 2007). Furthermore, biobased materials show a similar tendency to bioenergy and biofuels in their relative environmental impacts (e.g., Larson 2006; Quirin et al. 2004; Reinhardt et al. 2000; Schmitz et al. 2009; von Blottnitz and Curran 2007; WBGU 2008).

However, our findings scatter over a wide range, spanning both negative and positive values for several impact categories (figure 1 and table S1 in the supporting information on the Web). Decision making should account for this variability by considering individual cases, potentially weighing global environmental concerns (e.g., climate change, stratospheric ozone depletion) against local and regional concerns (e.g., eutrophication, photochemical ozone formation, land use change).

Discussion of Uncertainties

The results of our review are subject to uncertainties and limitations that arise from (1) the method used for analyzing the LCA studies and (2) the uncertainty in the inventory data, as well as the diversity of methodological choices applied in the individual LCA studies.

Addressing the first source of uncertainty, our review only quantifies the environmental impact of biobased materials in six categories. These categories are typically the most prominent ones addressed in the LCA literature. Still, they enable only a partial evaluation of biobased materials because of insufficient accounting for (1) other relevant land use-related impacts as well as (2) the potential risks resulting from the use of genetically modified crops and microorganisms.

Our approach to refrain from harmonizing the reviewed LCA studies with respect to differences in methodological choices and inventory data may limit the reliability and accuracy of our results. Methodological differences are likely to result in a random error for impact categories of sufficiently large data samples (e.g., nonrenewable energy use or GHG emissions). However, caution is required when interpreting the results for impact categories in which only a few data points are available (e.g., acidification, photochemical ozone formation) because methodological inconsistencies may lead to systematic errors. In these cases, the data samples are often highly skewed, making the mean a less reliable estimator of the general tendency of the sample. To analyze whether the skewness of data samples affects the interpretation of our results, we calculate the median environmental impacts (see table S2 in the supporting information on the Web). The deviations between the arithmetic mean and median are typically negligible for the totals of all biobased materials as well as for impact categories and groups of materials for which large data samples are analyzed (e.g., as is the case of nonrenewable energy use and GHG emissions). The deviations, however, become larger for cases in which small data samples span large value ranges. Such a case appears, for example, in the acidification potential of floor boarding, where the mean and median values may lead to different conclusions (see table S2 in the supporting information on the Web). Caution is therefore necessary before drawing conclusions solely based on one indicator for the central tendency of data samples.

Addressing the second source of uncertainty in all rigidity is beyond the scope of this article.

The data used for the inventory analysis in the respective LCA studies are typically subject to substantial uncertainties. Dinkel and colleagues (1996) quantify error ranges of inventory data of 40% and deviations resulting from differences in allocation methods of up to 90% of the final result. Miller and colleagues (2007) emphasize the considerable variability and uncertainty in the emission profiles of agricultural systems due to differences in geography, climate, and farming practices. The aggregate volatile organic compound and N_2O emissions, for example, vary in their study of soybean-based lubricants by more than 300% due to variability in agricultural processes, cropland characteristics, and soybean oil extraction. Furthermore, the reviewed LCA studies often vary from each other in the number of pollutants included in their inventory analysis as well as in the method used for aggregating inventory data into individual environmental impact categories (compare, e.g., Corbière-Nicollier et al. 2001; Dreyer et al. 2003; Müller-Sämann et al. 2002; Turunen and van der Werf 2006; Wötzel et al. 1999). The latter inconsistency, however, led to a small and random error in our analysis and can thus be regarded as negligible.

Discussion of Critical Aspects in the Life Cycle Assessment of Biobased Materials

The large range of results for biobased materials stems from the diversity of product systems, methodological choices, and plausible assumptions made in the reviewed LCAs (e.g., Patel et al. 2006). Usually it is a combination of choices that leads to substantial differences in the outcome of individual LCA studies, even if similar product scenarios are analyzed. We now discuss five choices and aspects, which may be particularly critical in the life cycle assessment of biobased materials.

Secondary Effects—The Case of Indirect Land Use Change

Indirect land use change, that is, the unintended expansion of farmland elsewhere due to the rededication of existing farmland, may add substantially to the overall environmental impacts of biobased materials. The effects of indirect land use change are excluded from the reviewed LCA studies but have been studied in the context of biofuels. Plevin and colleagues (2010) suggest that the GHG emissions from indirect land use change of corn ethanol production in the United States span between 10 and 340 kg CO_2-eq/GJ ethanol, thereby ranging from small to several times greater than the life cycle emissions of gasoline. Substantial indirect land use change effects of biofuels production in the United States and in Brazil have been identified by Arima and colleagues (2011), Lapola and colleagues (2010), and Searchinger and colleagues (2010).

Direct land use change due to the production of biobased plastics has been analyzed by Piemonte and Gironi (2011), who find that land use emissions have a substantial and largely negative impact on the GHG emissions savings unless waste biomass or biomass grown on degraded or abandoned land is used as feedstock. Wicke and colleagues (2008) found that direct land use change is the most decisive factor in the GHG emissions of palm oil energy chains. Hoefnagels and colleagues

(2010) show that the GHG emissions directly associated with the production of biodiesel from palm oil quadruple if plantations are located on former peat lands and rainforests instead of on degraded land or logged-over forests. Based on these considerations it is reasonable to assume that indirect land use change is likely to increase the GHG emissions of biobased materials; the extent to which this is happening remains, however, uncertain. In general, the impacts of land use change can be reduced by

- producing nonfood biomass on degraded lands that need to be restored;
- producing biobased materials from crops that provide high yields in feedstock and useful coproducts (Patel et al. 2006);
- achieving yield and productivity increases in regions with lagging yield developments, such as sub-Saharan Africa (Bruinsma 2009; Hubert et al. 2010; Nellemann et al. 2009);
- more intensive use of farmland by planting so-called *agricultural intercrops* between main cropping periods, which serve energy or material purposes (Karpenstein-Machan 2001);
- introducing comprehensive land use management guidelines; and
- establishing programs and policies of sustainable resource management that also consider and limit the consumption of global resources, including global land use (Bringezu and Bleischwitz 2009).

Treatment of Agricultural Residues

LCA studies generally assume that residues remain on the field as a substitute for mineral fertilizers (e.g., Würdinger et al. 2002) or they simply exclude residues from the product system (e.g., Corbiére-Nicollier et al. 2001). The environmental impacts of biobased materials may, however, substantially decline if the part of residues that is nonessential for maintaining soil organic matter is used for producing materials or energy. Dornburg and colleagues (2003) identified reductions in the nonrenewable energy use and GHG emissions of biobased polymers of up to 190 GJ and 15 t CO_2-eq/(ha*a), respectively, if agricultural residues are used for energy. Similar effects can be expected if agricultural and forestry residues or biogenic wastes are utilized for the production of biobased materials and second-generation biofuels in integrated biorefineries (Cherubini and Jungmeier 2010; Williams et al. 2009).

Farming Practices

The high environmental impacts of biobased materials in the categories of eutrophication and stratospheric ozone depletion (along with a substantial share of their nonrenewable energy use and GHG emissions) arise from biomass production with industrial farming practices. Kim and Dale (2008) found that no-tillage farming and the cultivation of winter crops can decrease the environmental impacts of corn production by up to 70%. Würdinger and colleagues (2002) have demonstrated that extensive farming practices, which differ from conventional farming in that they apply no synthetic pesticides and nitrogen fertilizers, substantially reduce the eutrophication, acidification, and stratospheric ozone depletion potential of starch-based loose-fill packaging materials (figure 3).

Since the restriction of ozone-depleting substances under the Montreal Protocol (UNEP 2000), fertilizer induced N_2O emissions constitute the single most important driver of stratospheric ozone depletion on a global scale (Ravishankara et al. 2009). N_2O is also a potent GHG that might not always be appropriately accounted for in the assessment of GHG emissions from farming (Bringezu et al. 2009). Smeets and colleagues (2009) suggest that N_2O emissions might contribute 10% to 80% to the total GHG emissions of biofuels, depending on crop type, climate conditions, and reference land use scenario. Results for biofuels indicate that altogether 3% to 5% of the provided nitrogen might be converted to N_2O (Crutzen et al. 2007). Although contested (see, e.g., Ogle et al. 2008; Smeets et al. 2009), this finding suggests that the commonly used average emission factor of 1% (IPCC 2006) may substantially underestimate the actual N_2O emissions. Optimized farming practices and crop management, which includes avoiding stagnant anaerobic soil conditions, may substantially reduce N_2O emissions (Komatsuzaki and Ohta 2007; Scheer et al. 2008; Sehy et al. 2003; Würdinger et al. 2002). However, as changes in farming practices are generally constrained by prevailing economic and social factors, it remains doubtful as to whether substantial reductions in the environmental impacts of biomass production are achievable on a large scale. Furthermore, the expansion of extensive farming may come at the cost of decreasing yields, and thus higher land requirements for biomass production (Würdinger et al. 2002). This complexity requires a thorough evaluation of biobased materials on regional, national, and global scales.

Treatment of Temporary Carbon Storage

When assessing the GHG emissions of biobased materials, there are three ways of addressing biobased carbon:

- ignoring it by considering biobased carbon as neutral because it has been withdrawn from the atmosphere and will be returned to the atmosphere within a limited time period; this approach implies that allocation issues are implicitly treated through the relative carbon content of coproducts (Guinée et al. 2009);
- regarding it as neutral but, as opposed to the previous option, allocating biobased carbon consistently with the allocation of other environmental burdens,
- crediting it by considering biobased carbon to be sequestered in biobased materials; this approach is more relevant for consequential LCAs, where the consideration of reference land uses is critical (Brandão and Levasseur 2011).

Crediting additional carbon storage implies that both the uptake of carbon as well as its release during use or waste

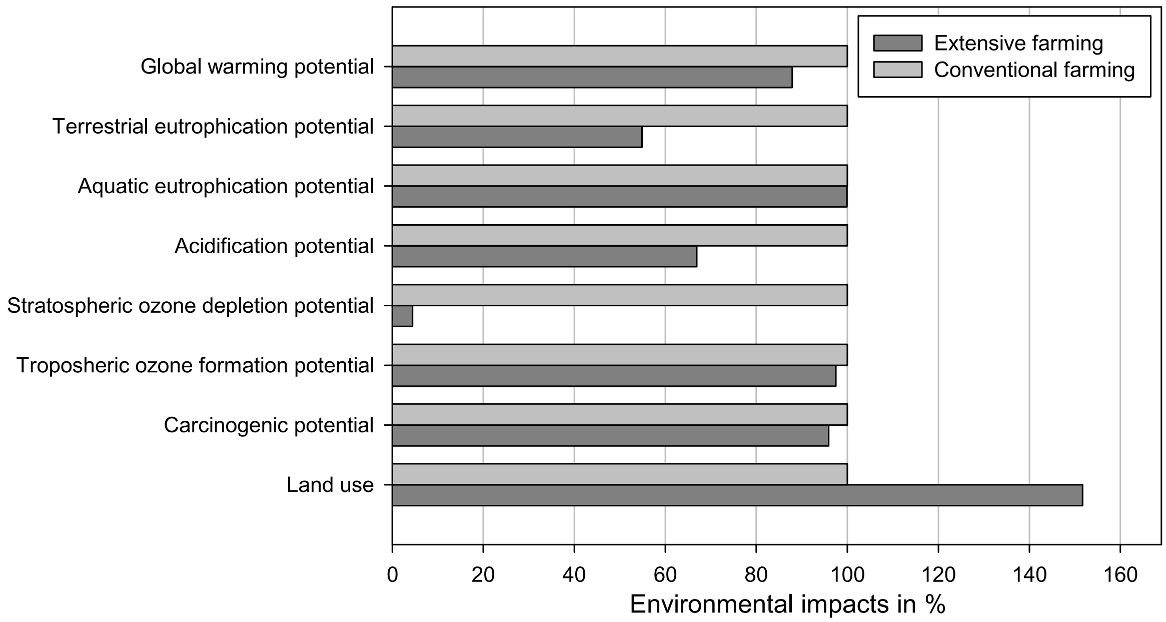

Figure 3 The relative environmental impacts of a defined volume of loose-fill packaging materials produced from conventional and extensively grown wheat starch. Source of data: Würdinger and colleagues 2002.

treatment must be accounted for as positive or negative emissions. As a consequence, there may be systematically lower GHG emissions for a cradle-to-factory gate analysis than for a cradle-to-factory gate analysis.

End-of-Life Waste Treatment

Covering end-of-life waste treatment allows investigating a wide range of waste treatment scenarios and specifically addresses the question of whether biodegradability is an environmentally favorable property of biobased materials. Hermann and colleagues (2011) show for polylactic acid that the product-specific GHG emissions might vary by approximately 20% depending on whether or not energy is recovered during waste incineration. In the LCA study of loose fills, Würdinger and colleagues (2002) suggest that the differences in GHG emissions between various waste treatment scenarios are similar to the differences between biobased and fossil fuel-based loose-fills. These findings call for a detailed assessment of all major waste management options, including landfilling, composting, waste-to-energy conversions, municipal waste incineration, digestion, and recycling (Amlinger et al. 2008; Edelmann and Schleiss 2001). Carbon cascading by using biomass first for material purposes and then recovering energy through incineration at the end of the product life cycle can maximize the GHG emissions savings of biobased materials (Bringezu et al. 2009; Dornburg et al. 2003; Oertel 2007). However, recent LCA studies have shown that composting can be more attractive than incineration (thus carbon cascading) if compost is used to replenish carbon stocks in agricultural soils (Hermann et al. 2011; Khoo et al. 2010).

Conclusions and Outlook

The findings of this meta-analysis allow us to draw the following conclusions:

- Biobased materials save nonrenewable energy; they enable the manufacturing industry to substitute renewable feedstock for part of its fossil fuel-based or mineral-based feedstock.
- Biobased materials generally exert lower environmental impacts than conventional materials in the category of climate change (if GHG emissions from indirect land use change are neglected).
- Biobased materials may exert higher environmental impacts than their conventional counterparts in the categories of eutrophication and stratospheric ozone depletion; our results are inconclusive with regard to acidification and photochemical ozone formation.
- Normalizing our results with worldwide average inhabitant-equivalent values suggests that biobased materials can contribute more to nonrenewable energy savings than to a decrease or increase of impacts in the five analyzed environmental impact categories.
- The environmental impacts of biobased materials span a wide range, partly due to the diversity of plausible methodological choices and assumptions made in the reviewed LCA studies. Thus caution must be taken when interpreting the outcome of this meta-analysis.
- Our analysis only quantifies part of the environmental impacts of biobased materials. More comprehensive quantitative analyses should address, in particular, land

use-related impacts (such as effects on biodiversity and soil organic matter, soil erosion) as well as the risks related to the use of genetically modified crops and microorganisms.
- Biomass cultivation with conventional farming practices is the key contributor to the high eutrophication and stratospheric ozone depletion potentials of biobased materials. These impacts can be reduced by improving fertilizer management and employing extensive farming practices. However, it should be considered that agricultural extensification by, for example, decreasing the application of agrochemicals, may result in lower crop yields, thus increasing land requirements for biomass production.
- The GHG emissions savings identified here are uncertain because the reviewed LCA studies (1) may only insufficiently account for N_2O emissions from biomass cultivation and (2) exclude the effects of indirect land use change. Depending on product scenarios and time horizons, especially the latter factor may substantially lower the established GHG emissions savings. Further research is needed.

The entire life cycle of biobased materials offers the potential for decreasing environmental impacts. However, the reduction of land use and its impacts on GHG emissions, eutrophication, and stratospheric ozone depletion might be most critical. Three strategies could be pursued: (1) expanding the feedstock base by utilizing organic wastes as well as forest and agricultural residues; (2) deploying integrated biorefineries that allow a more complete use of the biomass for producing biobased materials, energy, fuels, and heat; and (3) carbon cascading by using biomass first for material purposes and second for energy at the end of product life cycles. In developing countries, increasing yields and optimizing agricultural production are of paramount importance. A comprehensive accounting of global land use for both food and nonfood biomass production would allow assessing the overall effect on direct and indirect land use change as a basis for policy adjustments (Bringezu et al. 2012).

To date, biobased materials are produced largely by small-scale pilot plants. Progress in biotechnology, technological learning, up-scaling of production facilities, and process integration will likely reduce both the environmental impacts and the costs of biobased materials (Hermann et al. 2010; Vink et al. 2010). To quantify existing possibilities, LCA results can be combined with assessments of the technical and economic potential of biobased materials (e.g., Saygın and Patel 2010). In spite of current growth rates, biobased materials will require at least a decade to reach substantial market shares, even at high fossil fuel prices (Saygın and Patel 2010; Shen et al. 2009b). The economy-wide impact of biobased materials on nonrenewable energy use and GHG emissions will remain limited in the long-term because feedstocks for synthetic materials production accounts for only 6% in the global fossil fuel supply. The vast majority of fossil fuel use and related GHG emissions can be attributed to energy conversions elsewhere in the economy (IEA 2009).

This meta-analysis has highlighted the environmental potentials and challenges of biomaterials. Addressing persisting challenges may enable biobased materials to substantially decrease the environmental impacts from the production, use, and disposal of industrially manufactured materials.

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Notes

1. The total global production of bulk materials reached approximately 6,500 megatonnes (Mt) in 2009. This figure comprises the production of cement, iron and steel, bricks, glass, polymers and other petrochemicals, lubricants, bitumen, aluminum, textiles, wood, and paper (estimate based on Deimling et al. 2007; Lasserre 2008; OGJ 2007; IAI 2010; Saygın and Patel 2010; UN 2008).
2. One megatonne (Mt) = 10^6 tonnes (t) = one teragram (Tg, SI) $\approx 1.102 \times 10^6$ short tons.
3. One gigajoule (GJ) = 10^9 joules (J, SI) $\approx 2.39 \times 10^5$ kilocalories (kcal) $\approx 9.48 \times 10^5$ British thermal units (BTU). One metric ton (t) = 10^3 kilograms (kg, SI) $\approx$ 1.102 short tons. One kilogram (kg, SI) $\approx$ 2.204 pounds (lb). Carbon dioxide equivalent (CO_2-eq) is a measure for describing the climate-forcing strength of a quantity of greenhouse gases using the functionally equivalent amount of carbon dioxide as the reference. Ethene is C_2H_4, commonly known as ethylene.
4. One hectare (ha) = 0.01 square kilometers (km^2, SI) $\approx$ 0.00386 square miles $\approx$ 2.47 acres.

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Supporting Information

Additional supporting information may be found in the online version of this article.

Supporting Information S1: This supporting information lists the reviewed LCA studies and provides an overview of the environmental impact categories covered in each individual study. It also presents a detailed overview of the average impacts, expressed as differences between biobased and conventional materials, for six environmental impact categories and nine groups of materials.

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