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Land use mediated GHG emissions and spillovers from increased consumption of bioplastics

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Supplementary material for this article is available online

Abstract

Bioplastic production is a small but fast growing sector in the global bioeconomy, which may benefit from public support measures in the future as governments seek to promote more sustainable consumption patterns. Here we assess the potential net economy-wide impacts of a 5% bioplastic target relative to current plastic consumption in the main producing regions. We compare two alternative policy strategies to achieve the target in a general equilibrium framework that allows for substitution between conventional and bio-based plastics: a subsidy on bioplastics versus a tax on fossil-based plastic consumption. Our study is the first to quantify global greenhouse gas (GHG) emissions from an increased demand for bioplastics on a global scale, produced from arable crops, considering both direct and indirect land use change (LUC). The tax provokes a contraction of all sectors that employ plastics, which leads to a drop of 0.07% in global real GDP, whereas the subsidy has no significant effect on the global economy. Both tax and subsidy reduce world demand for petroleum products, by 0.37% and 0.07%, respectively, boosting demand for sugar- and starch-based feedstock in the bioplastic industry. This leads to emissions from LUC globally, which correspond to a carbon payback time of 22 years on average, with the associated annual abatement costs of over US\$2000 per tonne of CO_2-eq. The tax has greater GHG reduction potential in bioplastic producing regions but generates greater economic and environmental spillover effects in countries that do not enforce the target. Results show that promoting bioplastic consumption is not a cost-effective strategy for climate change mitigation if based on conventional feedstock, due to market-mediated GHG emissions from LUC. Bioplastics are not necessarily more sustainable than conventional polymers just because they are bio-based, although further assessment of potential environmental gains associated with biodegradability and recyclability is desirable.

1. Introduction

Plastics are increasingly important worldwide and across economic sectors due to their versatility, durability, and low production costs. Roughly 90% of conventional plastics are produced from heavy crude oil and hence associated with fossil fuel depletion, also giving rise to approximately 400 million tonnes of greenhouse gas (GHG) emissions per year globally, including waste incineration (European Commission (EC) 2018). These emissions occur when the carbon embodied in petroleum resources is suddenly released to the atmosphere by degradation or burning (Hottle *et al* 2013). Global plastic production has been growing exponentially and could reach up to 1.2 billion tonnes annually in 2050; which would then represent 20% of the total oil consumption in the world and 15% of the annual CO_2 emissions (Ellen MacArthur Foundation 2017). Stability and durability of conventional plastics generate additional environmental problems at end-of-life when plastic debris pollutes the oceans or terrestrial and freshwater ecosystems. Since only a small share of fossil-based plastics is biodegradable (OECD 2013), it is found that between 60% and 90% of marine debris is derived from petroleum resins with a long degradation time (UNEP 2016).

Public concern about the environmental costs of plastic consumption and a trend towards bio-based economic transformation in more and more countries are boosting investments in biomass-based plastics or simply *bioplastics* (BÖR 2018). Global bioplastic production capacity has increased from 1.5 to 1.9 million tonnes between 2012 and 2015 and may reach 6.7 million tonnes in 2018 (Rivero *et al* 2016). Despite high growth rates, bioplastics still account for only 1% of the total global plastic market (van den Oever *et al* 2017) and consist mainly of non-biodegradable drop-in products that allow for direct substitution in the industry, such as bio-polyethylene (bio-PE) and bio-polyethylene terephthalate (bio-PET). The share of plastics that are both bio-based and biodegradable, such as polylactic acid (PLA), is predicted to grow substantially in the next few years, as production costs gradually decrease (Aeschelmann and Carus 2015, van den Oever *et al* 2017). China, South Korea, the United States (US), the European Union (EU) and Brazil are currently the leading bioplastic producers, with capacity increases expected in the Asia-Pacific region (Aeschelmann and Carus 2015). Future market developments will depend on international trade, new conversion technologies for feedstock diversification, recycling infrastructure and logistics, and accompanying policies. Indeed, plastics are identified as one of the five priority areas in the 'EU Action Plan for the Circular Economy' (European Commission (EC) 2015). The 'European Strategy for Plastics' (European Commission (EC) 2018) prioritizes recycling over biodegradation to simultaneously increase the sustainability of the plastic industry and curb plastic waste. Since sector-specific incentives for bioplastics have been limited to date, as compared to those for biofuel production, policy support is increasingly demanded by bioplastic producers worldwide (Aeschelmann and Carus 2015, Hermann *et al* 2011, OECD 2013).

Feedstock currently used in bioplastic production varies between regions, but essentially consists of food crops. Second-generation technologies, which supposedly reduce competition with food production, are not yet operated on a commercial scale (Lewandowski 2015, Brodin *et al* 2017). According to European Bioplastics (2017), biomass for material uses accounts for no more than 2% of the total harvested area. However, increased biomass demand for non-food uses can certainly put additional pressure on limited resources, such as land and water, with implications for food security, climate change, and biodiversity (Scarlat *et al* 2015). The extent of the impacts will depend on biomass productivity and conversion efficiency, product functionalities and technical substitution rates in the industry as well as on global and national policy responses.

Impacts of replacing conventional with bio-based plastic have mainly been studied by means of Life Cycle Assessment (LCA) on a case-by-case basis (Groot and Borén 2010, Tsiropoulos *et al* 2015, Dietrich *et al* 2017). Posen *et al* (2017) estimate that GHG reductions from transitioning to biomass feedstock in the US plastic industry would only be achieved if using second-generation technologies. To our knowledge, no comprehensive assessment exists of potential land use mediated impacts of bioplastics, including both direct and indirect land use change (dLUC and iLUC), which are global in scope. More specifically, GHG emissions from iLUC represent spillover effects, which may imply decades of carbon payback time, as found for instance for biofuels (Fargione *et al* 2008, Gibbs *et al* 2008, Searchinger *et al* 2008, Lapola *et al* 2010). Only Piemonte and Gironi (2011, 2012) show the potential of total LUC to negate possible GHG benefits from bioplastics, by taking default emission factors from Searchinger *et al* (2008). As a market-mediated effect, iLUC is difficult to trace (Henders and Ostwald 2014) and therefore frequently addressed by means of global computable general equilibrium (CGE) models, coupled with biophysical modules (Hertel *et al* 2010, Plevin *et al* 2010). These can capture production-consumption linkages across all economic sectors and regions under physical and economic accounting identities. In contrast to product-oriented LCA, CGE-based assessments reflect average input-output relationships across firms in a sector, but respond to changes in the market and policy environment. CGE models are particularly well suited for the assessment of policy- and technology-driven land spillovers because they cover bilateral international trade and region-specific land scarcity; as critical aspects influencing global land use dynamics (Hertel 2018).

Global CGE models, by and large, rely on the GTAP database (Timilsina *et al* 2011, Aguiar *et al* 2016), which does not explicitly account for bioplastics. Lee (2016) was the first to analyze the economic effects of bioplastic expansion in key Asian countries—ignoring further environmental implications—by extending the now outdated GTAP 8 database (Narayanan *et al* 2012). In order to fill this gap, we seek to quantify global GHG emissions from increasing the market penetration of bioplastics in major producing regions. To do so, we extend the latest GTAP 9 database (Aguiar *et al* 2016) in the CGE model framework CGEBox[4] (Britz 2017) by introducing conventional and bio-based plastics as additional sectors. Due to data limitations, this approach focuses on GHG and land use impacts for sustainability assessments of bioplastic-related policies. Additional work would be necessary to explore the implications of biodegradability and recyclability aspects at end-of-life.

2. Methodological framework

Economic analysis in a CGE framework assumes that production and consumption decisions are motivated by incentives that arise from, for example, policies and technological change. Shifts in supply and demand

[4] A regularly updated and extended model documentation can be found at: www.ilr.uni-bonn.de/em/rsrch/cgebox/cgebox_GUI.pdf.

translate into price adjustments across markets, which ultimately generate environmental impacts through altered material flows. Given the limited resources to satisfy demand, incentives for one sector translate into disincentives for another, requiring economy-wide adjustments to bring all the markets back to equilibrium. Simulating an exogenous shock in comparative-static analysis, such as an increased bioplastic demand, hence requires specifying the incentives that encourage market actors to adjust consumption and production. Coming back to the example of biofuels, governments have used a variety of policies to increase the share of biofuels in the transportation mix, such as tax incentives or blending mandates (Timilsina and Shrestha 2011). In CGE analysis, these policy instruments are often defined as ad-valorem taxes and subsidies, which must adjust endogenously to drive the model to the desired biofuel demands. Comparative-static analysis allows then determining how endogenous variables in the model react, without providing information on the transition path from the original equilibrium.

2.1. Policy experiment

We simulate a 5% target for bioplastic consumption, relative to total plastic consumption in the baseline—i.e. 2011—simultaneously in Brazil, China, the EU, and the US as the major bioplastic producers. The 5% assumption is realistic given the current market share and level of technology, but conservative as compared to projections of up to 85% in market shares (Storz and Vorlop 2013, Schipfer *et al* 2017). We consider two alternative policy instruments to foster bio-based plastic use: consumption subsidies on bio-based plastics versus consumption taxes on fossil-based ones, such as product-specific sales or value-added tax rates. We treat bioplastics and conventional plastics as imperfect substitutes in demand; hence, in order to reach the target, the price of bioplastics must drop relative to that of fossil-based plastics. The subsidy scenario implies that users pay a price below production costs for bio-based plastics, whereas in the tax scenario, fossil-based plastics are taxed beyond the baseline, making bioplastics more competitive. Technically, the aggregated bioplastic demand from firms, government, households and investments is fixed to the regional target, so that it can and will trigger different supply increases reflecting price responsiveness in each case, while subsidy and tax rates are endogenously determined.

2.2. Model extension and implementation

The study departs from the GTAP 9 database (Aguiar *et al* 2016), which depicts the world economy in 2011. The underlying data do not explicitly capture conventional fossil- or bio-based plastics; instead, both are part of the chemical industry aggregate. Following a top-down approach, we hence disaggregate the latter into three sub-sectors, namely 'fossil-based plastics', 'bio-based plastics' and 'rest of chemicals', by means of the split utility in CGEBox (Britz 2017). The split factors arise from calculated output values and feedstock cost shares for the aforementioned regions based on production capacities (Shen *et al* 2009). Only first generation biopolymers, i.e. derived from edible biomass, are considered. The data suggest that Brazil focuses on bio-PE, although polyhydroxybutyrate (PHB) is also produced in small amounts; in both cases, by using sugarcane. The EU employs wheat (83.3%) and other cereal grains (16.7%) for producing thermoplastic starch blends. China relies on maize (85.7%) and wheat (14.3%) for the production of both PLA and PHB, while the US mainly uses domestic maize (88.2%) for the same purposes but also some wheat (11.8%). Beyond Lee (2016), we allow for substitution between 'bioplastics' and 'fossil-based plastics' in intermediate demand of firms (see figure A2, available online at stacks.iop.org/ERL/13/125005/mmedia). To date, only Nowicki *et al* (2010) had considered the possibility of substitution, assuming an elasticity of 3 for substituting PLA for fossil-based plastics in the short-term (2005–2010). Since our study also covers drop-ins such as Bio-PE, which allow for direct substitution with no change in the final product, we assume a higher elasticity of 15. This also reflects enhanced technical characteristics of bioplastics in the long-run, in line with Posen *et al* (2017). Additionally, we conduct sensitivity analysis for values of 5, 10 and 20. Further methodological details regarding the database extension and model setup are documented in the annex (see figures A1–A2 and tables A1–A2).

In addition to the standard GTAP model, we employ various extensions available in CGEBox, namely: GTAP-Agr (Keeney and Hertel 2005) to better represent the characteristics of the agricultural sector; GTAP-E (Burniaux and Truong 2002) to incorporate substitution between energy sources in production and calculate CO_2 emissions from the combustion of fossil fuels; GTAP-AEZ (Lee 2005) to capture competition for land between uses at the level of agro-ecological zone (AEZ). Non-CO_2 emissions from consumption (e.g. fertilizers), endowment use (land and capital), and production are also quantified (Aguiar *et al* 2016). GHG emissions from dLUC and iLUC arise from land use transitions across 18 AEZs and subsequent changes in carbon pools. These include flows from forest regrowth and forgone carbon sequestration over a 30 year period; primary forest conversion is not considered due to the nature of the underlying economic accounts (Plevin *et al* 2014). Carbon stock data (Gibbs *et al* 2014) are included in the Annex (see tables A3–A5). In this way, we can quantify environmental spillover effects from both extensification and intensification of agriculture on a global scale. Below we summarize our findings in terms of percentage changes due to the 'shock' to bioplastic demand, relative to the benchmark. All

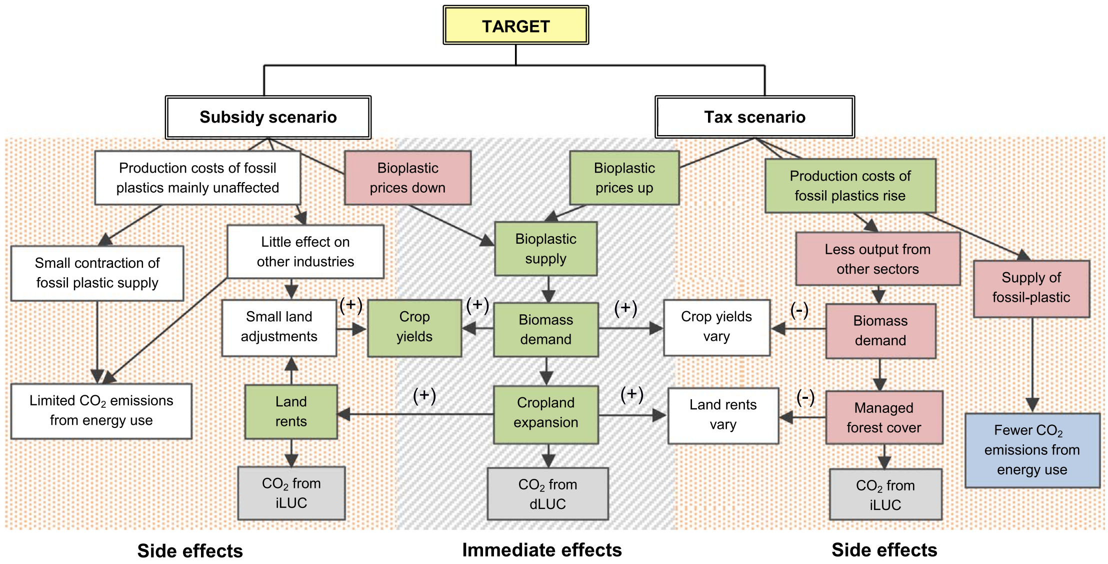

Figure 1. Successive market responses triggered by the target in the two scenarios. *Immediate effects* refer to responses across the bioplastic and related feedstock markets. *Side effects* propagate across the rest of the economy through the fossil-plastic market segment. Boxes in green indicate an increase in the variable; red boxes indicate a decrease; boxes are colourless when there is no significant change; grey shows an increase in CO_2-eq. emissions, while blue boxes refer to a decrease in CO_2-eq.; dLUC: direct land use change; iLUC: indirect land use change.

quantities are expressed in constant US$ 2011, while the environmental indicators are measured in physical units.

3. Results

The main economic and environmental response pathways triggered by both the subsidy and the tax scenarios are conceptually described in figure 1, whereas detailed results are presented in the annex. The exogenous target drives up bioplastic consumption in producing regions, so that production has to increase too. Subsidy and tax rates adjust endogenously (table A6), with the subsequent effects on consumer and producer prices of both conventional plastics and bioplastics (figure A3). As a result, the global tax income decreases with the subsidy (−0.17%), while it increases with the tax (+2.14%). Changes in tax income are greater in China where the target imposes a sharper increase in the market penetration of bioplastics (table A6). The subsidy lowers the average consumer price of plastics as a whole in the four producing regions (figure A3(a)), so that aggregate world total plastic demand increases by 0.32% (table A6). The tax on conventional plastics, on the other hand, affects the lion's share of the plastic market by pushing up average consumer prices; as a result, the world plastic market shrinks by 7.24%. As expected, both policy alternatives lead to a reduction in the global demand for petroleum and coal products, larger with the tax (−0.37%) as compared to the subsidy (−0.07%). Further responses can be explained by firstly analyzing the *immediate effects* of the target, i.e. through the expansion of the bioplastic market segment; and the *side effects* due to the contraction of the fossil-plastic market segment. In both scenarios, *immediate effects* across agricultural markets translate into spillover effects in terms of food prices as well as iLUC and associated emissions, as observed in the past under biofuel mandates. *Side effects* spread across all sectors that employ plastics directly or indirectly; with sizeable environmental impacts mediated by energy markets, since conventional plastic production is fossil fuel and energy intensive. The combination of *immediate* and *side effects* generates substantially different outcomes in the two scenarios, which underlines the need for careful policy instrument choice and design.

Following the immediate effect pathway (central column in figure 1), the subsidy increases intermediate demand for agricultural feedstock in the expanding bioplastic industry, though to a relatively smaller extent in those regions that already produce relevant amounts of bioplastics in the baseline, namely the US and Brazil (see table A6 in the Annex). The production of feedstock expands globally and especially in the bioplastic producing regions, i.e. cereal grains in China (+10.19%) and the US (+4.01%), sugarcane in Brazil (+0.95%), and wheat in the EU (+2.88%). This is due to both higher yields (intensification) and cropland expansion. Changes in land cover occur due to successive adjustments in agricultural markets on a global scale, with the subsequent GHG emissions. Since bioplastics remain a small economic sector, impacts of the subsidy on production costs of fossil-based plastics are minor, also on producer prices (see also figure A3(b)). Hence, *side effects* arising from factor reallocation across sectors, other than land, remain small as compared to those in the tax scenario.

The tax scenario triggers similar *immediate effects* as it pushes demand for bioplastics in the main producing regions to exactly the same quantity as the subsidy. As shown in table A6, however, the tax provokes a notably greater contraction of the fossil-plastic sector; also of the world plastic market as a whole, increasing intermediate input costs in all sectors that rely on plastics. Producer prices of conventional plastics increase as reported in figure A3(b), which reflects higher production costs. Note that taxing conventional plastics reduces the competitiveness of the sector in the focus regions, relative to the rest of the world (ROW). Other world plastic producers thus tend to compensate for the conventional plastic production shortfall in what can be considered a spillover effect of the regional bioplastic target. However, the tax discourages the use of plastics as a 'non-environmentally-friendly' input for firms in the US, China, the EU and Brazil, which together account for a large share of the plastic market (see figure A1). As a result, global production of fossil-based plastics shrinks by 9.72% under the tax, driving down global oil demand (−0.32%) to a larger extent than the subsidy scenario, in which it remains almost unchanged (−0.05%).

Immediate effects drive up land rents in the two scenarios due to increased land competition, which translates into higher crop prices, especially in the bioplastic producing regions (see figure A4). The only exception is China in the tax scenario, in which the *side effects* are especially large, generating a contraction of all sectors that consume plastics across the supply chain, including food production. Crop yields increase for bioplastic feedstock, but eventually decrease in primary sectors from which production factors are being shifted, depending on the region (see table A7). While these mainly involve livestock and forestry sectors in the subsidy scenario, the tax increases production costs in agriculture as well (through intermediate input use), which translates into lower yields, e.g. for oilseeds in Brazil (−0.40%) and the US (−0.25%). In general, the tax creates a strong distortion of the allocation of resources across the economy; as a result, global real GDP decreases by 0.08%, while it is slightly affected (−0.02%) by the subsidy (table A6). In both cases, China and the EU experience the greatest GDP losses, due to the relatively large share of the plastic sector in these countries.

3.1. Land use change and associated GHG emissions

The environmental implications of the target are analyzed in terms of total LUC, including dLUC and iLUC, and GHG emissions. The latter include CO_2 emissions from global land cover changes, non-CO_2 emissions, i.e. N_2O and CH_4 mostly from agriculture and livestock, and CO_2 emissions from overall energy consumption. The difference in CO_2-eq. before and after the policy shocks can thus be interpreted as the life cycle emissions associated with the specific increase in demand for bioplastics, reflecting related adjustments in material and factor use in the global economy. Land use transitions depend on differential land rents based on relative returns to land. Although the overall managed land extension is fixed in GTAP, the carbon stocks vary as land is shifted between uses in a region. In this way, the GTAP model captures global GHG spillovers at the extensive and intensive land use margin.

Cropland expands by 0.10% globally under the subsidy and by 0.17% under the tax, at the expense of managed forest and pasture. While the contraction in managed forest area is around 0.03% in the subsidy scenario, it reaches 0.17% in the tax scenario. Pastureland decreases by 0.04% under the subsidy but increases by 0.01% under the tax, partly due to the reduction in oilseeds production caused by the tax in major producing countries. Relative changes in cropland and forest area are shown in figures 2 and 3(a), (b), whereas shifts in pastureland extension are reported in the annex (figure A5).

The subsidy induces greater increases in cropland area (between 0.08% and 1.38%) in the regions that promote bioplastic use (figure 2(a)); especially China, where the target is largely met with domestic feedstock, and the US, which provides the EU with maize. Sharp cropland expansions are also detected in other parts of North and South America, Oceania, or Mediterranean Africa, which produce grains and other agricultural commodities for export; while cropland equally expands (up to 0.08%) in the ROW. This suggests that, in the subsidy scenario, LUC mostly arises from the *immediate effects*. On the contrary, the tax generates greater *side effects*, which deliver even opposite outcomes in terms of cropland area expansion in some regions, reflecting a shrinking global economy (figure 2(b)). Cropland expands (between 0.16% and 1.38%) across the EU, China, Former USSR, North America, Oceania and Northern Brazil. A contraction (up to 0.30%) is observed in vast extensions of Sub-Saharan Africa, Southeast Asia or South America, where overall agricultural output is decreasing.

The *immediate effects* of the subsidy lead to a decrease in forestland by up to 0.20% mainly in the bioplastic producing regions; while greater decreases (between 0.20% and 0.98%) are only reported in Northeast China (figure 3(a)), where the main maize producing provinces are located. In the tax scenario, the decline in forest cover (between 0.03% and 0.98%) spreads across North America, most of the EU and former USSR; while it becomes particularly pronounced in the US, Brazil, China and Central Asia, and also Sub-Saharan Africa (figure 3(b)). This is due to *side effects*: the contraction in all sectors that rely on fossil-based plastics also reduces the demand for forest products, which is substantial in sectors such as construction or manufacturing in the countries involved. The

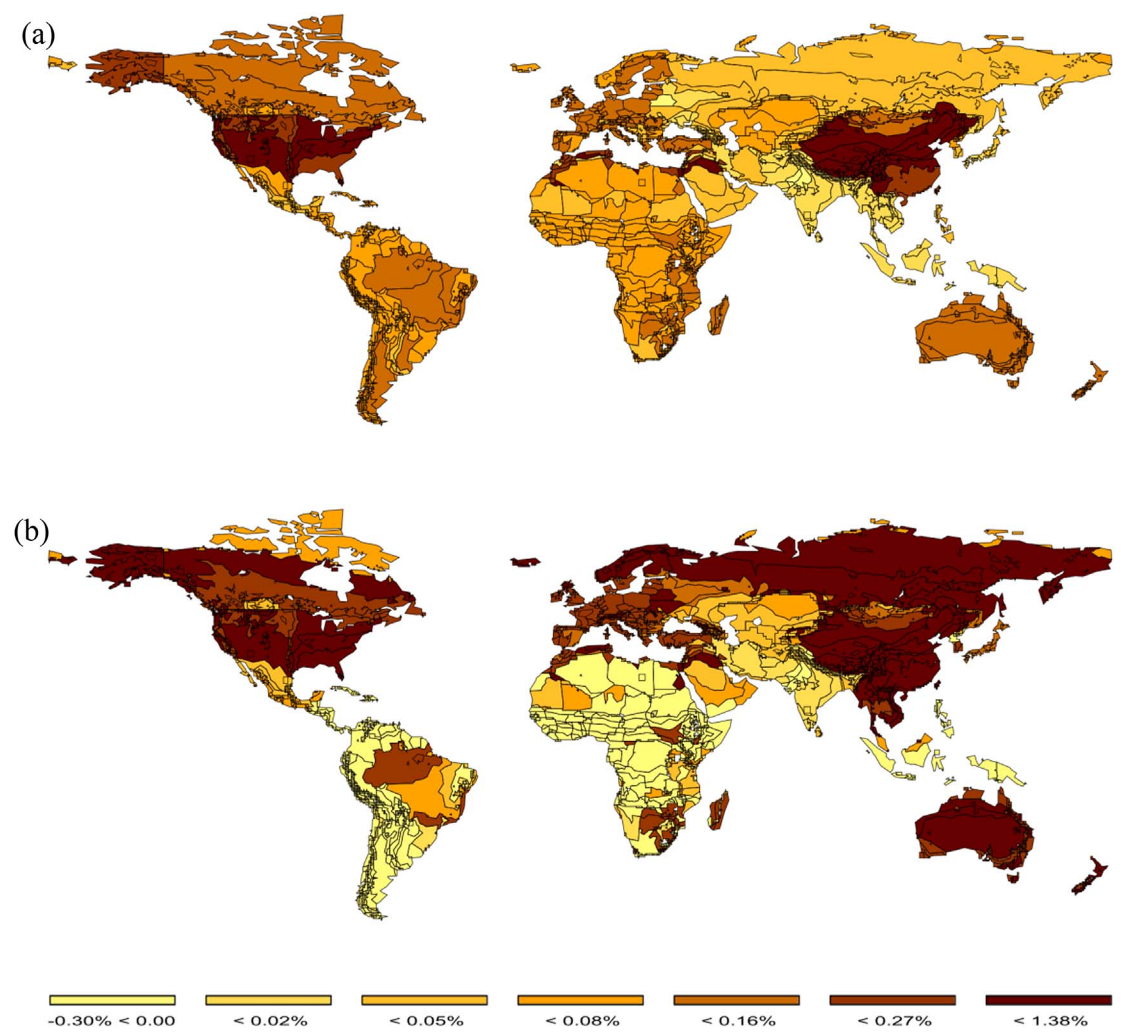

Figure 2. Relative changes (%) in cropland area across AEZs in the subsidy scenario (a) and the tax scenario (b), as compared to the baseline.

decrease in firms' demand for forest-based bioenergy is especially marked in the US, EU and China. Hence, intermediate demand for forest products decreases globally by 0.76% under the tax and remains almost unchanged (−0.01%) under the subsidy. Forest area expands up to 0.29% in the rest of South America and Southeast Asia, where forestry becomes more profitable than other sectors such as oilseeds (figure 3(b)).

GHG outcomes from the policy experiments are documented in table 1. Non-CO_2 and CO_2 emissions from production reflect average annual emission flows across the economy, whereas CO_2 emissions from LUC are considered as a one-time effect from carbon stock changes (Plevin *et al* 2014). These two kinds of estimates allow us to quantify the global carbon payback time of the bioplastic target, defined as the time that it takes for the GHG savings from fossil-based plastic substitution in aggregated demand to compensate for the 'carbon debt' (Fargione *et al* 2008), i.e. CO_2 emissions incurred by the subsequent LUC effects, which are global in scope. Payback times are calculated following equation (1) (Gibbs *et al* 2008) for both subsidy and tax:

$$\text{Carbon payback time (years)} = \frac{\Delta\text{Carbon Stock[Mt]}\,\frac{44}{12}}{\Delta CO_2\,[\text{Mt}] + \Delta\text{non}CO_2[\text{Mt}]}, \quad (1)$$

where the numerator is the difference in total carbon stock before and after the shock, expressed as CO_2-eq. by considering the conversion factor of carbon into CO_2; the denominator captures the change in annual CO_2-eq. emissions from all economic sectors as the sum of CO_2 emissions from energy use and non-CO_2 emissions from agricultural and industrial production, excluding LUC.

If emissions from LUC are not considered, the two scenarios deliver GHG reductions on a global scale and in the bioplastic producing regions individually. The tax has a greater GHG mitigation potential (−0.25%), due to the contraction of all sectors that intensively use conventional plastics. Energy demand decreases as well, which constitutes the largest source of GHG emissions at both global and national levels, except for hydropower dependent countries like Brazil. The greatest reductions are reported for China (−1.18%) and the EU (−1.01%), where the target

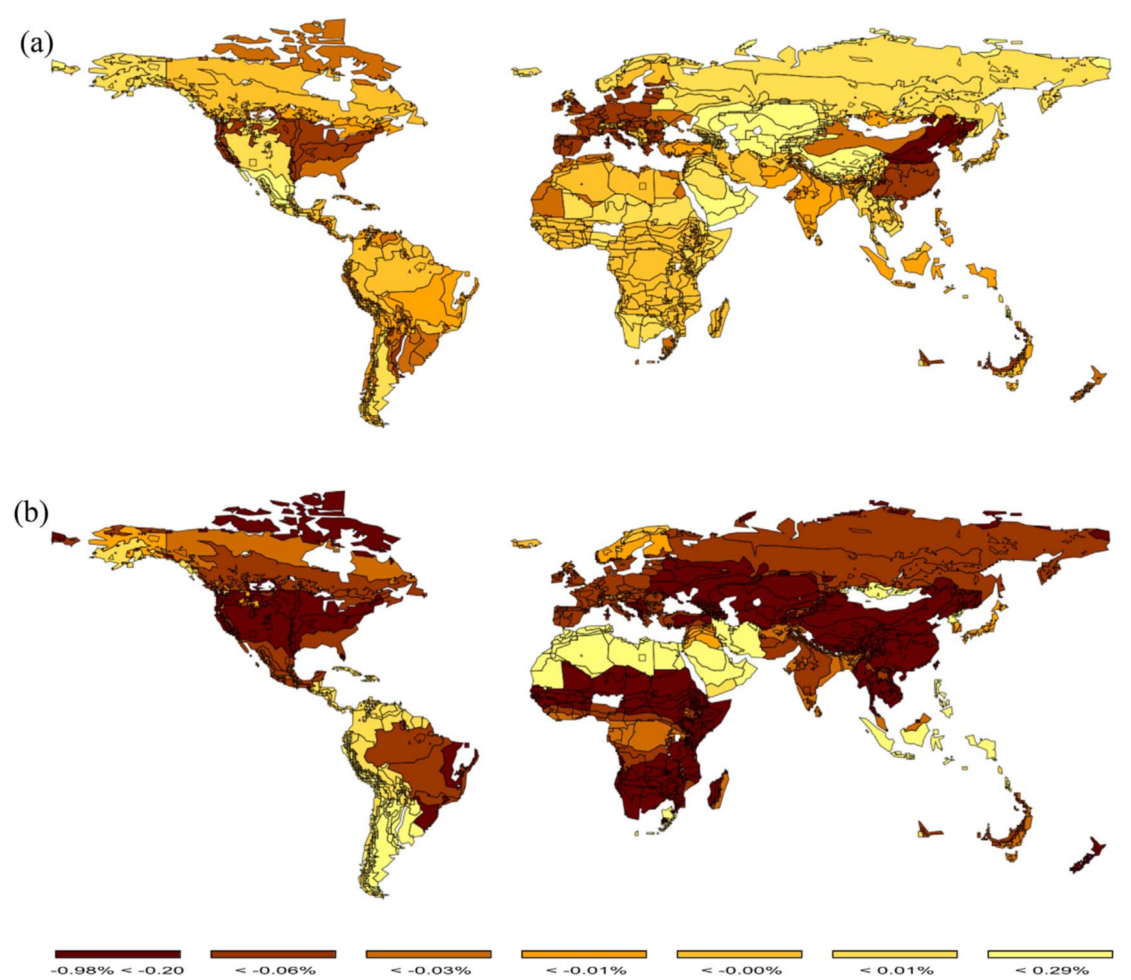

Figure 3. Relative changes (%) in forest area across AEZs in the subsidy scenario (a) and the tax scenario (b), as compared to the baseline.

triggers greater *side effects*. The subsidy scenario entails minor adjustments in material and factor use across sectors, which only generate a 0.05% reduction in GHG emissions globally. Note that primary factor endowments are fixed in GTAP, i.e. the availability of managed land and natural resources such as oil and gas remains unchanged regardless the policy scheme. Differences in GHG emissions thus only arise from factor reallocation and demand shifts across sectors and industries.

When LUC emissions are considered, the bioplastic target is associated with carbon payback times of 21.6 years in the subsidy scenario and 22.5 years in the tax scenario, on a global scale. This is because the tax entails greater emission reductions from fossil substitution across economic sectors but also greater emissions from LUC globally, with differences between countries (table 1). In the subsidy scenario, the greatest LUC emissions come from land cover changes in the US (see figure 3(a)), which partially responds to the demand for maize from the EU, as combined with limited GHG savings in the US industry. The target translates into an especially marked increase in bioplastic demand in the EU and China (table A6). Hence, emissions from LUC are also substantial in the latter, since China mainly relies on domestic feedstock for bioplastic production. The tax generates significant spillover effects in terms of forest cover loss across the world (figure 3(b)), with the subsequent emissions from LUC; especially in China, where there is a notable decrease in demand for forest biomass for energy purposes. GHG emissions greatly increase in Brazil, where forest loss involves carbon-rich ecosystems and LUC is historically the largest contributor to climate change (Lapola *et al* 2014). The stronger economic contraction observed under the tax also triggers greater CO_2-eq. reductions from energy savings, which ultimately translates into a payback time that is similar to that estimated for the subsidy scenario. The tax leads to an overall expansion of pasture across the globe, except for regions where grain production is dramatically increasing (see figure A5(b)). The estimated payback times serve as an indication of the potential climate change impact of an increased bioplastic consumption in major producing regions at the same time, hence also responding to *side effects*.

Table 1. Total (GHG) emissions as CO_2-eq. (million tonnes) from the entire economy, broken down by source, and absolute changes in annual CO_2-eq. emissions (million tonnes) relative to the baseline, under the two policy scenarios.

		Emissions from LUC as CO_2-eq. (Mt)	Non-CO_2 emissions as CO_2-eq. (Mt)	CO_2 emissions from energy consumption as CO_2-eq. (Mt)	Annual change in GHG emissions, without LUC, as CO_2-eq. (Mt)
Baseline	World		12962.8	27941.5	
	US		975.4	4996.8	
	Brazil		614.3	356.8	
	China		2743.7	6974.8	
	EU28		1183.6	3596.8	
Subsidy scenario	World	458.9	12952.3	27930.9	−21.21
	US	151.4	973.8	4996.0	−2.34
	Brazil	26.9	613.4	356.4	−1.31
	China	135.4	2737.2	6968.1	−13.19
	EU28	78.2	1182.3	3594.6	−3.42
Tax scenario	World	2340.8	12907.5	27893.0	−103.85
	US	253.7	968.2	4984.5	−19.48
	Brazil	209.4	613.5	354.9	−2.74
	China	698.2	2688.8	6915.4	−114.37
	EU28	153.7	1172.3	3560.0	−48.14

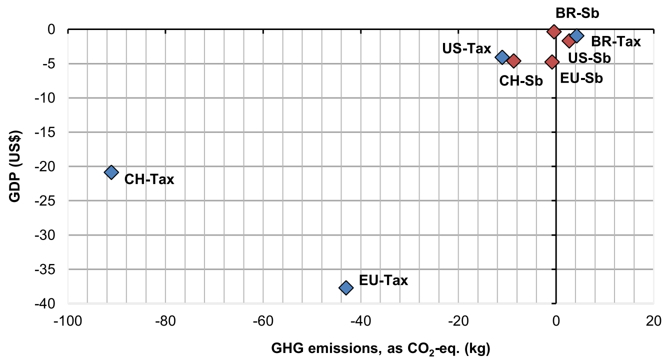

Figure 4. Absolute annual changes in GDP (US$) and GHG emissions as CO_2-eq. (kg) in each region and policy scenario, relative to the baseline. BR: Brazil; CH: China; EU: European Union 28; US: United States; sb: subsidy scenario (red dots); tax: tax scenario (blue dots).

3.2. Environmental versus economic tradeoffs

The payback times discussed above can be interpreted as the minimum period for the bioplastic target to remain in force in order to deliver long-term GHG savings on a global scale. A shorter period would generate increased GHG emissions together with real GDP loss under the two policy regimes. In order to estimate the associated cost-effectiveness of the policy, a period has to be assumed during which the target remains binding. Considering that the policy, i.e. subsidy or tax, entails constant average annual costs under our comparative-static approach, we take 30 years as the reference period, which is consistent with the analytical horizon for the carbon stock calculation in the AEZ-EF model (Plevin *et al* 2014) and is often applied in biofuel-related studies (Searchinger *et al* 2008, Hertel *et al* 2010). LUC emissions are thus annualized accordingly, considering a linear amortization over 30 years. We then jointly evaluate economic versus environmental tradeoffs by quantifying the overall welfare loss associated with a decrease of one unit of CO_2-eq. globally due to the combined target in bioplastic producing regions. Annual abatement costs are hence calculated as the ratio of the change in real GDP to the change in annual GHG emissions as CO_2-eq., including LUC as annual carbon stock loss, according to equation (2):

the tax in Brazil due to large region-specific LUC effects combined with small domestic GHG savings. Note that these LUC effects are partly driven by increases in feedstock demand in other bioplastic producing regions. For the EU and China, the tax generates both substantially greater GHG reductions and GDP losses than the subsidy. The effects of the subsidy on the non-bioplastic producing regions are minor, whereas the tax generates an increase of overall GHG emissions in the ROW; especially in those countries that take over part of the fossil-based plastic market share, such as Japan, South Korea and India, where GDP increases significantly.

4. Discussion and policy implications

CGE modeling in combination with the GTAP 9 database and its so-called satellite accounts provides an adequate basis for the quantification of policy-induced spillover effects in terms of land use and GHG emissions. Spillovers occur when impacts spread beyond the geographical boundaries of the intervention, i.e. the bioplastic consumption target, and should thus be evaluated at the global level (Kim *et al* 2014). In our simulation experiment, GHG emissions do not only arise from LUC, referring to both dLUC and iLUC, but also from adjustments in material and factor

$$\text{Annual abatement cost per unit of } CO_2\text{-eq.}(\$\text{kg}^{-1}) = \frac{\Delta\text{GDP}(\$)}{\Delta CO_2\ (\text{kg}) + \Delta\text{non } CO_2(\text{kg}) - \Delta\text{Carbon Stock(kg)}\frac{44}{12}\frac{1}{30}}. \quad (2)$$

The target is associated with average global abatement costs per kg of CO_2-eq. of \$2.04 when enforced through a subsidy and \$2.19 in the case of the tax. Figure 4 shows absolute changes in both GDP and GHG in each region due to the combined target. GHG emissions increase with the subsidy in the US and with

use in the global economy. In this way, the bioplastic target delivers changes in GHG emissions in the countries that enforce it but also foreign emissions or 'emission spillovers' through price and trade-mediated adjustments. Outcomes are however subject to (1) uncertainty in default model parameters, such as supply and demand elasticities, emission factors, and land productivity (Henders and Ostwald 2014); and also (2) model constraints on (fixed) primary factor supply and (no) technological change (Gerlagh and Kuik 2014). Particularly the substitutability of bioplastics for their fossil counterparts, here captured by a substitution elasticity, can play a critical role in bio-based transitions. A mandate for bioplastic consumption is very likely to trigger technological innovation allowing for a larger degree of substitutability in the future, mainly in the industry, while adding new functionalities in final demand. The real substitution potential ultimately depends on the specific bioplastic family, being 1:1 for drop-in products such as bio-PE (Posen *et al* 2017). However, this distinction is not yet possible given the level of disaggregation of the available data.

We explore the sensitivity of land and GHG spillover effects to changes in the substitution elasticity between conventional plastics and bioplastics in intermediate demand, which is the only parameter introduced and not based on empirical data. Hence, values of 5, 10 and 20 are assessed, which reflect the range from quite limited to almost full substitutability. As mentioned above, Nowicki *et al* (2010) chose a substitution elasticity of 3 for short-term replacement with PLA, which motivates the lower limit of 5 as drop-ins are already in the market. Technically, the substitution elasticity determines the ability to substitute for fossil-based plastics in order to reach the target in total aggregated demand. The lower the substitution elasticity, the more inputs (feedstocks, other intermediates, capital, labor) are necessary to produce the amount of bioplastic needed to replace a physical unit of the fossil counterpart. Results from the sensitivity analysis are discussed in the Annex and shown in figure A6 in terms of cropland area and GHG percentage changes relative to the baseline, in the bioplastic producing regions versus the ROW; together with associated carbon payback times for both subsidy and tax. The global carbon payback time decreases with increasing elasticities of substitution in both policy scenarios, although the variability is greater under the subsidy due to the greater influence of *immediate effects* on the annual emission changes. To assist policy design further, both the payback times and abatement costs must be analyzed for each region individually. This entails additional experiments that consider increasing country-specific bioplastic targets, since market-mediated GHG emissions and spillovers vary in a nonlinear fashion.

Bioplastics may receive growing attention by policy makers as a mean to achieve sustainability goals, including energy security and climate change mitigation. Our results however show that increased bioplastic consumption leads to deforestation and GHG emissions from LUC on a global scale. A conservative 5% target for bioplastic consumption in the main producing regions as a whole is associated with payback times of around 22 years globally, when met with current technologies. The policy should thus be in force for decades in order for GHG savings from fossil raw material substitution to compensate for emissions from LUC (including the so-called iLUC). The payback times estimated by us are close to that obtained for instance for the 2015 US ethanol mandate, which is around 28 years if entirely based on maize (Hertel *et al* 2010); despite the fact that we simulate four regional targets simultaneously, including multiple raw materials such as sugarcane. We find high global average abatement costs for the bioplastic target of over \$2000 per tonne of CO_2-eq., both for the subsidy and tax scenarios. This is in the upper range of estimated abatement costs for biofuel policies, e.g. between \$960 and \$1700 in the US, EU and Canada in the period 2013–2017 (Timilsina and Shrestha 2011). Scientific evidence for biofuels encouraged the consideration of iLUC emissions factors for inclusion in biofuel policies as 'sustainability requirements' (Gawel and Ludwig 2011, Finkbeiner 2014). Khanna *et al* (2017) estimate the cost-effectiveness of a US biofuel mandate exclusively met with low-iLUC biofuels, i.e. supply chains in which annualized land carbon stock losses are especially low compared to CO_2 savings, at between \$61 and \$187 per tonne of CO_2-eq. According to the authors, this is still substantially higher than the social cost of carbon of \$50, even if the study neglects further potential market feedback effects.

Biopolymers do not necessarily outperform fossil-based polymers in terms of sustainability outcomes, due to potential negative impacts from land and other agricultural input use (Hottle *et al* 2013). This is in contrast to previous studies reporting potential annual CO_2-eq. savings between 241 and 316 Mt (Spierling *et al* 2018), without considering LUC. Our approach quantifies upstream GHG emissions across the bioplastic supply chain by capturing feedback effects from agricultural input intensity and LUC, among other factors. This methodologically complements existing studies on carbon payback times of biofuels (e.g. Fargione *et al* 2008, Gibbs *et al* 2008). One limitation of GTAP-based CGE experiments relative to those is however that neither degraded grasslands nor primary forests are included as available land uses. As Bentsen (2017) points out, further methodological consensus is needed for the calculation of carbon payback times of alternative policies for bio-based transitions. Similarly, agreement on the amortization time of LUC emissions is advisable for policy coherence and cross-country comparison, since it proves critical in the quantification of emissions from fossil raw material substitution (Hertel *et al* 2010).

The assessment of environmental versus economic tradeoffs of the target in section 3.2 does not capture benefits from increased recyclability and biodegradability of some bioplastics at end-of-life. These characteristics, also found in specific fossil-based plastics, could translate into reduced waste streams and lower impacts from waste treatment; the cost from which are often borne by governments. For instance, treatment options for non-biodegradable plastics frequently entail burning, especially in developing countries; releasing CO_2 and carcinogen compounds (Hottle *et al* 2013, Nkwachukwu *et al* 2013). In other countries, incineration with energy recovery offers sustainability gains due to electricity generation (Philp *et al* 2013). Recycling is usually the preferred option for plastic disposal but is often hampered by technical and logistic barriers (Kaiser *et al* 2017); these are however lower for drop-in products, which can enter existing plastic recycling infrastructure (Philp *et al* 2013). Although highly context-dependent, end-of-life options are energy intensive, delivering increased emissions throughout life cycles of plastic products. In spite of technical challenges, mechanical recycling can become the most cost-effective alternative by implementing closed-loop approaches in the industry to reduce demand for raw material (Hopewell *et al* 2009). This is deemed to deliver energy and emission savings relative to virgin plastic production. Thus, alignment of processes between the chemical industry and the waste collection sector is required for innovative and fully recyclable plastics. However, adequately implementing circular economy aspects in a CGE framework requires additional data and model adjustments beyond the scope of this study.

5. Conclusions

We have assessed the economic and environmental implications of the promotion of bioplastic use on a global scale, by extending the GTAP 9 database (Aguiar *et al* 2016) to include both 'fossil-based plastics' and 'bioplastics'. Our study is the first to quantify global land use mediated GHG emissions due to an increase in demand for bioplastics, considering dLUC and also iLUC, which arises from economy-wide interactions. We show that a hypothetical 5% target for bioplastic consumption relative to total plastic demand in major producing regions is not an effective strategy for climate change mitigation if based on food crops. As in the case of first generation biofuels, an early but currently stagnating sector in the bioeconomy, global LUC emissions can offset the GHG abatement potential associated with the substitution for fossil raw materials in plastic production. Our findings encourage research in second-generation technologies, which do not compete with food and feed uses.

From a policy perspective, our results emphasize that bioplastic promotion with tax instruments affects relative prices, leading to factor reallocation and GDP loss, which could only be justified by equivalent environmental benefits. Payback times are calculated at around 22 years at the global level irrespective of the policy instrument used to achieve the target. This means that the policies would have to remain in force for many years in order to deliver cost-effective GHG reductions. Both policy scenarios are associated with a global annual abatement cost of around US$2000 per tonne of CO_2-eq., but generate different global distributions of GDP loss, LUC, and GHG savings. Revenues from the tax could however be used to finance GHG mitigation strategies in countries that implement the policy, such as improvement of conversion efficiencies or promotion of renewable energy to bring further environmental gains.

The main implication of this study for policy initiatives that seek to promote transformation towards sustainable bioeconomies is that 'bio-based' may not be enough. Future biomaterial strategies should focus on enhancing biodegradability and recyclability as much as on the origin of the feedstock. Unless new and less land intensive types of feedstock are available, circular economy principles, such as reuse and recycling, may represent more sustainable means of securing material supply to growing bioeconomies than fossil resource substitution. This ultimately points to the need for governance frameworks that promote cascading uses and feedstock diversification (see, for example, the EU Action Plan for Circular Economy) in order to minimize tradeoffs among sustainability dimensions, such as among the Sustainable Development Goals 12–15.

The improved CGEBox (Britz 2017), based on GTAP 9, constitutes a valuable tool to simulate bioplastic policies in a CGE framework, capturing global economy-wide impacts under different levels of technological development in the bioplastic industry. Future research should aim at relaxing model assumptions, such as fixed total primary factor (e.g. land) use, which substantially influence GHG outcomes.

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References

Aeschelmann F and Carus M 2015 Biobased Building Blocks and Polymers in the World: Capacities, Production, and Applications–Status Quo and Trends Towards 2020 *Ind. Biotechnol.* **11** 154–9

Aguiar A, Narayanan B and McDougall R 2016 An overview of the GTAP 9 data base *J. Glob. Econ. Anal.* **1** 181–208

Bentsen N S 2017 Carbon debt and payback time—lost in the forest? *Renew. Sustain. Energy Rev.* **73** 1211–7
BÖR 2018 *Bioeconomy Policy: Update Report of National Strategies around the World* Part III German Bioeconomy Council
Britz W 2017 CGEBox: A flexible, modular and extendable framework for CGE analysis in GAMS *20th Annual Conf. Glob. Econ. Analysis*
Brodin M, Vallejos M, Opedal M T, Area M C and Chinga-Carrasco G 2017 Lignocellulosics as sustainable resources for production of bioplastics—a review *J. Clean. Prod.* **162** 646–64
Burniaux J M and Truong T 2002 GTAP-E: an energy-environmental version of the GTAP model *GTAP Technical Paper* No. 16 Center for Global Trade Analysis (Purdue University)
Dietrich K, Dumont M J, Del Río L and Orsat V 2017 Producing PHAs in the bioeconomy—towards a sustainable bioplastic *Sustain. Prod. Consum.* **9** 58–70
Ellen MacArthur Foundation 2017 The new plastics economy: rethinking the future of plastics and catalysing action
European Bioplastics 2017 *Bioplastics market data 2017-Global production capacities of bioplastics 2017-2022 Berlin* (https://docs.european-bioplastics.org/publications/market_data/2017/Report_Bioplastics_Market_Data_2017.pdf) European Bioplastics
European Commission (EC) 2015 Closing the loop—an EU action plan for the Circular Economy COM (2015) 614 *European Commission* (www.eea.europa.eu/policy-documents/com-2015-0614-final)
European Commission (EC) 2018 A European Strategy for Plastics in a Circular Economy COM (2018) 28 final *European Commission*
Fargione J, Hill J, Tilman D, Polasky S and Hawthorne P 2008 Land clearing and the biofuel carbon debt *Science* **319** 1235–8
Finkbeiner M 2014 Indirect land use change—help beyond the hype? *Biomass Bioenergy* **62** 218–21
Gawel E and Ludwig G 2011 The iLUC dilemma. How to deal with indirect land use changes when governing energy crops? *Land Use Policy* **28** 846–56
Gerlagh R and Kuik O 2014 Spill or leak? Carbon leakage with international technology spillovers: a CGE analysis *Energy Econ.* **45** 381–8
Gibbs H K, Johnston M, Foley J A, Holloway T, Monfreda C, Ramankutty N and Zaks D 2008 Carbon payback times for crop-based biofuel expansion in the tropics: the effects of changing yield and technology *Environ. Res. Lett.* **3** 034001
Gibbs H K, Yui S and Plevin R 2014 New estimates of soil and biomass carbon stocks for global economic models *GTAP Technical paper* No. 33 Center for Global Trade Analysis (Purdue University)
Groot W J and Borén T 2010 Life cycle assessment of the manufacture of lactide and PLA biopolymers from sugarcane in Thailand *Int. J. Life Cycle Assess.* **15** 970–84
Henders S and Ostwald M 2014 Accounting methods for international land-related leakage and distant deforestation drivers *Ecol. Econ.* **99** 21–8
Hermann B, Carus M, Patel M and Blok K 2011 Current policies affecting the market penetration of biomaterials *Biofuels Bioprod. Biorefining* **5** 708–19
Hertel T W 2018 Economic perspectives on land use change and leakage *Environ. Res. Lett.* **13** 075012
Hertel T W, Golub A A, Jones A D, O'Hare M, Plevin R J and Kammen D M 2010 Effects of US maize ethanol on global land use and greenhouse gas emissions: estimating market-mediated responses *Bioscience* **60** 223–31
Hopewell J, Dvorak R and Kosior E 2009 Plastics recycling. Challenges and opportunities *Phil. Trans. R. Soc.* B **364** 2115–26
Hottle T A, Bilec M M and Landis A E 2013 Sustainability assessments of bio-based polymers *Polym. Degrad. Stab.* **98** 1898–907
Kaiser K, Schmid M and Schlummer M 2017 Recycling of polymer-based multilayer packaging: a review *Recycling* **3** 1–26
Khanna M, Wang W, Hudiburg T W and DeLucia E H 2017 The social inefficiency of regulating indirect land use change due to biofuels *Nat. Commun.* **8** 15513
Keeney R and Hertel T W 2005 GTAP-AGR: a framework for assessing the implications of multilateral changes in agricultural policies *GTAP Technical Paper* No. 25 Center for Global Trade Analysis (Purdue University)
Kim M K, Peralta D and McCarl B A 2014 Land-based greenhouse gas emission offset and leakage discounting *Ecol. Econ.* **105** 265–73
Lapola D M *et al* 2014 Pervasive transition of the Brazilian land-use system *Nat. Clim. Change* **4** 27–35
Lapola D M, Schaldach R, Alcamo J, Bondeau A, Koch J, Koelking C and Priess J A 2010 Indirect land-use changes can overcome carbon savings from biofuels in Brazil *Proc. Natl Acad. Sci. USA* **107** 3388–93
Lee D H 2016 Bio-based economies in Asia. Economic analysis of development of bio-based industry in China, India, Japan, Korea, Malaysia and Taiwan *Int. J. Hydrog. Energy* **41** 4333–46
Lee H L 2005 The GTAP land use data base and the GTAPE-AEZ model: incorporating agro-ecologically zoned land use data and land-based greenhouse gases emissions into the GTAP framework *8th Ann. Conf. Glob. Econ. Analysis (Lübeck, Germany)*
Lewandowski I 2015 Securing a sustainable biomass supply in a growing bioeconomy *Glob. Food Secur.* **6** 34–42
Narayanan B, Aguiar A and McDougall R 2012 *Global Trade, assistance, and production: the GTAP 8 data base Center for Global Trade Analysis* Purdue University
Nkwachukwu O, Chima C, Ikenna A and Albert L 2013 Focus on potential environmental issues on plastic world towards a sustainable plastic recycling in developing countries *Int. J. Ind. Chem.* **4** 34
Nowicki P, van Leeuwen M, Bos H, Chant L, Molenveld K and Tabeau A 2010 Influence of the biobased economy on agricultural markets *Preparation of a Modelling Approach* (The Haghe: LEI Wageningen UR)
OECD 2013 Policies for Bioplastics in the Context of a Bioeconomy *OECD Science, Technology and Industry Policy Papers* (https://doi.org/10.1787/5k3xpf9rrw6d-en)
Philp J C, Ritchie R J and Guy K 2013 Biobased plastics in a bioeconomy *Trends Biotechnol.* **31** 65–7
Piemonte V and Gironi F 2011 Land-use change emissions: how green are the bioplastics? *Environ. Prog. Sustain. Energy* **30** 685–91
Piemonte V and Gironi F 2012 Bioplastics and GHGs saving: the land use change (LUC) emissions issue *Energy Sources A* **34** 1995–2003
Plevin R J, Gibbs H K, Duffy J, Sahoko Y and Yeh S 2014 Agro-ecological Zone Emission Factor (AEZ-EF) Model (v47). A model of greenhouse gas emissions from land-use change for use with AEZ-based economic models *GTAP Technical Paper* No. 34 Center for Global Trade Analysis (Purdue University)
Plevin R J, O'Hare M, Jones A D, Torn M S and Gibbs H K 2010 Greenhouse gas emissions from biofuels' indirect land use change are uncertain but may be much greater than previously estimated *Environ. Sci. Technol.* **44** 8015–21
Posen I D, Jaramillo P, Landis A E and Griffin W M 2017 Greenhouse gas mitigation for US plastics production: energy first, feedstocks later *Environ. Res. Lett.* **12** 34024
Rivero C P, Hu Y, Kwan T H, Webb C, Theodoropoulos C, Daoud W and Lin C S K 2016 Bioplastics from solid waste *Current Developments in Biotechnology and Bioengineering: Solid Waste Management* ed W C Jonathan *et al* (Amsterdam: Elsevier) pp 1–26
Scarlat N, Dallemand J F, Monforti-Ferrario F and Nita V 2015 The role of biomass and bioenergy in a future bioeconomy: policies and facts *Environ. Dev.* **15** 3–34
Schipfer F, Kranzl L, Leclère D, Sylvain L, Forsell N and Valin H 2017 Advanced biomaterials scenarios for the EU28 up to 2050 and their respective biomass demand *Biomass Bioenergy* **96** 19–27

Searchinger T *et al* 2008 Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change *Science* **319** 1238–40

Shen L, Haufe J and Patel M K 2009 *Product overview and market projection of emerging bio-based plastics* PRO-BIP Final Report University of Utrech

Spierling S *et al* 2018 Bio-based plastics–a review of environmental, social and economic impact assessments *J. Clean. Prod.* **185** 476–91

Storz H and Vorlop K D 2013 Bio-based plastics: status, challenges and trends *Appl. Agric. Forestry Res.* **4** 321–32

Timilsina G R, Csordás S and Mevel S 2011 When does a carbon tax on fossil fuels stimulate biofuels? *Ecol. Econ.* **70** 2400–15

Timilsina G R and Shrestha A 2011 How much hope should we have for biofuels? *Energy* **36** 2055–69

Tsiropoulos I, Faaij A P C, Lundquist L, Schenker U, Briois J F and Patel M K 2015 Life cycle impact assessment of bio-based plastics from sugarcane ethanol *J. Clean. Prod.* **90** 114–27

UNEP 2016 *Marine Plastic Debris and Microplastics—Global Lessons and Research to Inspire Action and Guide Policy Change DEP/2010/NA* (Nairobi: United Nations Environment Programme)

van den Oever M, Molenveld K, van der Zee M and Bos H 2017 Bio-based and biodegradable plastics—Facts and figures: focus on food packaging in the Netherlands *Research Report* No. 1722 Wageningen: Food and Biobased Research