

Strategies to Reduce the Global Carbon Footprint of Plastics

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Abstract

Over the last four decades, global plastics production has quadrupled¹. Continuing this trend, the greenhouse gas (GHG) emissions from plastics would reach 15% of the global carbon budget by 2050². Strategies to mitigate the life cycle GHG emissions of plastics, however, have not been evaluated on a global scale. Here, we compile a new dataset covering ten conventional and five bio-based plastics and their life cycle GHG emissions under various mitigation strategies. Our results show that the global life cycle GHG emissions of conventional plastics was 1.7 Gt CO₂e in 2015, which would grow to 6.5 Gt CO₂e by 2050 under the current trajectory. However, an aggressive application of renewable energy, recycling, and demand management strategies in concert has the potential to keep the 2050 emissions comparable to the 2015 level. In addition, replacing fossil feedstock by biomass can further reduce the emissions to achieve an absolute reduction from the current level. Our study demonstrates the need for integrating energy, materials, recycling, and demand management strategies to curb the growing life cycle GHG emissions from plastics.

Main

Global production of plastics grew from 2 Mt to 380 Mt between 1950 and 2015, at a compound annual growth rate (CAGR) of 8.4%¹. Globally, 58% of plastic waste was discarded or landfilled, and only 18% was recycled in 2015¹. It is estimated that 4.8-12.7 Mt plastic waste generated by coastal countries entered the ocean in 2010³. Growing along the volume of global production and consumption of plastics are the diverse concerns on their impacts to the ecosystem and human health⁴⁻⁷. However,

relatively little attention has been paid to their contributions to climate change. While the chemical industry as a whole is responsible for about 15% of global anthropogenic greenhouse gas (GHG) emissions⁸, the magnitude of global life cycle GHG emissions from plastics has yet to be quantified.

Various strategies to reduce GHG emissions from plastics have been discussed in the literature. Replacing fossil-based plastics by bio-based plastics, for example, is one of them⁹⁻¹¹. Bio-based plastics generally show lower life cycle GHG emissions compared to fossil-based counterparts¹². Substituting 65.8% of the world's conventional plastics with bio-based plastics is estimated to avoid 241 to 316 Mt CO₂e per year¹³. Both biodegradable and non-biodegradable forms of bio-based plastics are available in the market¹⁴. Bio-based non-biodegradable polymers such as Bio-Polyethylene (Bio-PE) and Bio-Polyethylene Terephthalate (Bio-PET), also referred to "drop-in" polymers, offer virtually identical properties with their fossil-based counterparts. While bio-based biodegradable polymers, such as Polylactic Acid (PLA), Polyhydroxyalkanoates (PHAs) and Thermoplastic Starch (TPS) display somewhat different mechanical and chemical properties¹². Strategies to promote bio-based plastics have been initiated by the European Commission and other countries including Japan, Korea and Thailand^{15,16}. In 2017, the total global production of bio-based plastics reached 2.05 Mt, and is projected to grow by 20% over the next five years¹⁷.

Low carbon energy is another strategy to reduce life cycle GHG emissions of plastics. Under a 100% renewable energy, the GHG emissions from the United States plastics production could be reduced by 50-75%¹⁸. Another strategy to reduce GHG emissions of plastics is recycling, which reduces, in part, carbon-

intensive virgin polymer production¹⁹ while preventing GHG emissions from some end-of-life (EoL) processes such as incineration²⁰.

However, the literature to date has focused on a subset of plastic types, mitigation options, or geographical locations in isolation^{18,21}. Here, we develop a new dataset that covers GHG emissions from resin production, conversion, and EoL of ten fossil-based and five bio-based plastics. We then integrate the dataset with global plastics demand projections and GHG mitigation strategies. We evaluate the following mitigation strategies and their combinations:

(1) Bio-based plastics: fossil-based plastics are gradually substituted by bio-based plastics until a complete phase-out of fossil-based plastics by 2050. While bio-based plastics can be derived from a variety of feedstock, modelled here are corn and sugarcane given their dominance in current market¹¹.

(2) Renewable energy: the energy mix of plastics supply chain is gradually decarbonised and reaches 100% renewables (i.e. wind power and biogas) by 2050. Emissions under the current energy mix are modelled as comparison.

(3) Recycling: recycling rates of EoL plastics gradually increase and reach 100% by 2050. In comparison, we also model the emissions under a projected EoL management mix scenario and a 100% incineration/composting scenario.

(4) Reducing demand growth: the current annual growth rate of global plastics demand, which is 4%, is reduced to 2%.

We examine these strategies as illustrative scenarios, rather than as realistic projections of future trajectories, with the purpose of envisioning their potentials for GHG mitigation. We acknowledge that achieving 100% recycling or renewable

energy may be neither practical nor economically feasible in reality. Details on these scenarios can be found in Supplementary Table 1.

Our analysis shows that conventional plastics (fossil-based) produced in 2015 generated 1.8 Gt CO₂e emissions over their life cycle, excluding any carbon credits from recycling (Fig. 1). The amount corresponds to 3.8% of the 47 Gt CO₂e global emissions that year²². Resin production stage generated the majority of the emissions (61%), followed by conversion stage (30%). Of all plastic types, PP&A fibers had the highest GHG emissions at both stages. Polyolefin family (PP, L/LLDPE, and HDPE), which accounts for nearly 50% of the world's plastics consumption, was also a significant contributor. GHG emissions from bio-based plastics are not considered for 2015 given their negligible market share (<1 percent).

[Insert Figure 1]

Figure 1. Global life cycle GHG emissions of conventional plastics in 2015 by life cycle stage and plastic type.

Carbon credits generated by recycling are not included. Blue, orange, and green represent resin production, conversion, and end-of-life management stage, respectively. The emissions from each stage are broken down by plastic type or end-of-life treatment method, indicated with different shades of the corresponding color. Abbreviations: Polyethylene terephthalate (PET), High density polyethylene (HDPE), Polyvinyl Chloride (PVC), Low-density/linear low-density polyethylene (L/LLDPE), Polypropylene (PP), Polystyrene (PS), Polyurethane (PUR), Polyester, Polyamide and Acrylic fibers (PP&A), Additives, and Others.

The EoL stage accounted for 9% of total life cycle emissions, excluding the carbon credits from recycling. Incineration was the dominant source of GHG emissions among EoL processes. Landfill generated the least GHG emissions although the process handles the largest share of plastic wastes (58%). The recycling process itself generated 49 Mt CO₂e emissions. However, if the displacement of carbon-intensive virgin polymer production by recyclates is considered, the GHG emissions of recycling would go down to -67 Mt, and the total emissions from EoL stage would be reduced from 161 Mt to 45 Mt CO₂e. In this case, the total global life cycle GHG emissions of plastics become 1.7 Gt CO₂e, or 3.5% of the global annual GHG emissions in 2015.

Under the current trajectory, the global life cycle GHG emissions from plastics are poised to grow rapidly (Fig. 2a). The global economy produced 407 Mt plastics in 2015, with an average annual growth rate of 4% between 2010-2015¹. Following this trend, annual plastics production is expected to grow to 1,606 Mt by 2050, and the life cycle GHG emissions are expected to grow from 1.7 Gt CO₂e in 2015 to 6.5 Gt CO₂e in 2050, using the projected EoL management mix change¹, and maintaining current energy mix (baseline: red solid line in Fig. 2a). If all plastic waste are incinerated by 2050, total annual emissions will reach 8.0 Gt CO₂e (a 22% increase from the baseline). Recycling all plastic waste, however, would reduce the emissions to 4.9 Gt by 2050 (a 25% reduction from the baseline).

[Insert Figure 2]

Figure 2. Global life cycle GHG emissions of plastics under scenarios of different feedstock sources, energy mix, end-of-life management and plastics demand growth, 2015-2050.

Solid lines: projected end-of-life management mix (Supplementary Table 10); shaded areas: ranges due to EoL options; right-side bar of each panel: ranges due to different EoL options in 2050. **a**, plastics demand grows at 4% year⁻¹ under current energy mix. **b**, plastics demand grows at 4% year⁻¹, and energy mix decarbonises until 2050. **c**, plastics demand grows at 2% year⁻¹ under current energy mix. **d**, plastics demand grows at 2% year⁻¹, and energy mix decarbonises until 2050.

With a plastics demand growth rate of 4% year⁻¹, a complete replacement of fossil-based plastics by corn-based plastics is estimated to reduce global life cycle GHG emissions of plastics to 5.6 Gt CO₂e by 2050 under current energy mix and the projected EoL mix, which is 1.0 Gt or 15% less than the baseline (Fig. 2a). If all EoL “drop-ins” are incinerated and all EoL biodegradable plastics are composted, global life cycle GHG emissions of corn-based plastics would increase to 6.7 Gt CO₂e. Recycling all EoL bio-based plastics, however, would reduce the emissions to 4.4 Gt CO₂e. Sugarcane-based plastics can further reduce global life cycle GHG emissions of plastics to 4.9 Gt CO₂e, which is 1.7 Gt or 25% less than the baseline, with a range between 5.8 Gt (100% incineration/composting) and 4.0 Gt (100% recycling). Our model shows that fossil-based plastics under 100% recycling scenario achieves similar or even lower emissions compared to bio-based plastics with the projected EoL mix (Fig. 2a and 2b, sidebars). This implies that the recycling of conventional plastics may be as beneficial as using renewable feedstock.

Decarbonising energy shows a significant potential to reduce GHG emissions (Fig. 2b and Fig. 2d). On average, switching to 100% renewable energy reduces life cycle GHG emissions from plastics by 62% in 2050, assuming 4% year⁻¹ demand growth. Even if fossil sources (petroleum, natural gas and coal) serve as the sole feedstock for future plastics production, using 100% renewable energy can achieve 51% reduction (projected EoL mix) compared to the baseline, although the absolute total emissions would double the 2015 level by 2050. However, recycling all EoL plastics under 100% renewable energy allows 77%, 84% and 86% reductions in life cycle GHG emissions from fossil, corn and sugarcane-based plastics, respectively. This result shows that absolute reduction of emissions can only be achieved by combining aggressive deployment of renewable energy and extensive recycling of plastics.

Reducing plastics demand growth rate from 4% to 2% year⁻¹ achieves 56% (under the current energy mix) to 81% (under low carbon energy) reduction from the baseline in 2050 (Fig. 2c and 2d). Using 100% renewable energy keeps the emissions flat at 2015 level for fossil-based plastics with projected EoL mix, and replacing them with bio-based ones brings the emission levels down further. Among all the scenarios tested, the global life cycle GHG emissions of plastics were the lowest under the 100% sugarcane-based plastics with 100% renewable energy combined with 100% recycling and reduced demand growth, which achieved 0.5 Gt CO₂e/year, or 93% reduction from the baseline. This demonstrates that a drastic reduction in global life cycle GHG emissions of plastics would be possible in a technical sense, but it would require implementing all of the four strategies examined at an unprecedented scale and pace.

[Insert Figure 3]

Figure 3. GHG emissions breakdown by life cycle stage of plastics derived from different feedstock types under two energy mix scenarios in 2050.

a, GHG emissions per kilogram of plastics under the current energy mix scenario in 2050. **b**, GHG emissions per kilogram of plastics under a 100% renewable energy scenario in 2050. Emissions results are based on the scenario of 4% annual plastic demand growth rate and the projected end-of-life management mix (Supplementary Table 10). Carbon credits of recycling are considered.

Figure 3 shows the breakdown of GHG emissions by life cycle stage, normalised to per kilogram of plastics derived from different feedstock types. The total life cycle GHG emissions for fossil-based, corn-based and sugarcane-based plastics are on average 4.1, 3.5 and 3.0 kg CO₂e/kg plastic in 2050, respectively, under current energy mix (Fig. 3a). Under a 100% renewable energy scenario, however, the average life cycle emissions will be reduced to 2.0, 1.4 and 1.3 kg CO₂e/kg plastic, respectively (Fig. 3b). Plastics derived from renewable feedstock (assuming projected EoL mix) generate less GHG emissions over the whole life cycle compared to their fossil-based counterparts regardless of the energy system used.

Resin production and conversion stages are major contributors to the life cycle GHG emissions of all feedstock types under current energy mix (Fig. 3a). However, under the 100% renewable energy scenario, incineration becomes the largest contributor to the total emissions for bio-based plastics (Fig. 3b). Under the 100% renewable energy scenario, recycling generates fewer carbon credits, as the low GHG

emissions of renewable energy undercuts the carbon benefits of avoided virgin polymer production.

In sum, our results show that none of the four strategies, namely bio-based plastics, renewable energy, recycling, and demand management, can achieve sufficient GHG mitigation for absolute reduction below the current level on its own; only when implemented in concert, these strategies can achieve the much-needed absolute reduction. Among them, decarbonisation of the energy system, which is an economically more favorable option for GHG mitigation as compared to the use of bio-based plastics¹⁸, shows the largest potential. Even if fossil feedstock is used as the sole source for plastics production, 100% renewable energy will reduce the average life cycle GHG emissions by half from the baseline emissions. If combined with extensive recycling or demand management, decarbonisation of energy can virtually keep the current level of GHG emissions until 2050. Reducing GHG emissions even further to achieve absolute reduction from the current level requires large-scale adoption of bio-based plastics in addition to implementing all the other three strategies examined.

Going forward, we see both opportunities and challenges in reducing the life cycle GHG emissions of plastics. The current global average plastics recycling rate of 18%¹ certainly presents a significant room for further improvement. The low price of fossil-based plastics, however, is a key barrier to dramatically increasing recycling rates. Together with technological innovations in plastics recycling, fiscal policies, such as carbon pricing and incentivising recycling infrastructure expansion, should be considered to overcome such barriers^{23,24}.

Replacing fossil-based plastics with bio-based plastics plays an important role in GHG mitigation. Nevertheless, our results show that the emissions of bio-based plastics are highly dependent on the EoL management method chosen. Composting or incinerating bio-based plastic waste, for example, showed similar or even higher GHG emissions than the case of using 100% fossil-based plastics under projected EoL mix in 2050. Moreover, EoL management of bio-based—especially biodegradable—plastics requires systematic changes such as separate collection and recycling infrastructure, since inclusion of biodegradable plastics in the mix of conventional plastic waste can affect the quality of the recyclates²⁵. Furthermore, composting of biodegradable plastics in home composting conditions or natural environments is much less effective than in industrial composting facilities¹⁴. Lastly, the land use implications of a large-scale shift to bio-based plastics need further research. In 2017, land use for bioplastics was reported to be 0.82 million hectares, or 0.016% of global land area, which would increase to 0.021% in 2022 under the projected market growth¹⁷. A complete shift of the plastics production of approximately 250 million tones to bio-based plastics would require as much as 5 percent of all arable land²⁶, which, depending on where they take place, may undermine the carbon benefits of bio-based plastics. The use of lignocellulosic or waste biomass as feedstock and growing material crops in fallow lands would alleviate the pressure of cropland expansion and associated GHG emissions from land use change.

Our study shows that an aggressive implementation of multi-layered strategies would be needed in order to curb the GHG emissions from plastics. GHG mitigation strategies are often implemented within energy, materials, waste reduction and management policies in isolation. Our results indicate that absolute reduction in life

cycle GHG emissions of plastics requires a concerted action among decarbonisation of energy infrastructure, improvement of recycling capability, adoption of bio-based plastics, and demand management.

Methods

Life cycle GHG emissions of plastics were compiled for three feedstock types, considering effects of energy mix transformation, different end-of-life management options, and different growth rates of plastics demand. 2015 was selected as the base year, with GHG emissions modelled until 2050 under different scenarios. GHG emission data were collected for three life cycle stages: (a) resin production stage, which includes all activities from cradle to polymer-production factory gate; (b) conversion stage, covering the manufacturing processes that turn polymers into final plastic products; and (c) EoL stage, which refers to the treatment and disposal processes of plastic waste. The use stage was excluded. To calculate the total GHG emissions of a certain year, the annual plastics production and waste generation volumes are multiplied with the life cycle GHG emissions of each plastic type as shown in equation (1):

$$GHG_{s,t} = \sum Q_{s,j,t} \times E_{s,i,j,k,t} \quad (1)$$

Where $Q_{s,j,t}$ represents the annual global production or waste generation amount of type j plastic in year t under scenario s , and $E_{s,i,j,k,t}$ represents the per-unit weight emissions of GHG i by plastics type j at its life cycle stage k in year t under scenario s . Index i indicates different GHG types including carbon dioxide, methane and nitrous oxide; j indicates different type of plastics including L/LLDPE, HDPE, PET, PVC, PS, PUR, PP&A for fossil-based plastics, and Bio-PE, Bio-PET, PLA, PHAs, TPS for corn- or sugarcane-based plastics; k indicates the life cycle stage of plastics from

resin production, conversion, to end-of-life management; t indicates a year between 2015 and 2050, and s indicates scenarios of different combinations of feedstock, end-of-life options, energy mix and plastics demand growth.

Life cycle GHG emissions of fossil-based plastics

For resin production stage of fossil-based plastics, GHG emissions data from ecoinvent 3.4 database²⁷, European Life Cycle Database (ELCD)²⁸ and various literature sources were used. Detailed unit processes of resin production are listed by polymer type; emission data of some polymer types with subtypes were calculated as weighted sums according to their market share information (Supplementary Table 2). There is a large gap in life cycle inventory data of plastics additives²⁹. Hence, we chose Di-isononyl phthalate (DINP) as a proxy for plastics additives, which is frequently used as a general, all-purpose plasticizer³⁰. For the last group, “Others”, average GHG emission values of all plastic types were used.

After resin production, the polymers are transformed into various final products for specific applications. Injection molding, blow molding and extrusion are commonly used conversion technologies³¹. There are limited data on plastic products conversion processes in ecoinvent 3.4 and additional data was compiled from the literature. Ref. 31 and ref. 32 served as main data sources for this stage. Ref. 5 shows average GHG emissions from converting various plastic parts for a typical vehicle, and we used the data for GHG emissions from general conversion processes. For PS conversion process, data was drawn from ref. 33. For PUR conversion process, due to the scarcity of data, the average emissions from PP and PE conversion processes was used. For PP&A fibers, data from ref. 34 was used and the emission values were weighted based on the market share of polyester,

polyamide and acrylic. Due to the complex supply chain of textile industry, we cut off at yarn production and exclude the following conversion processes including fabric production and garment production. Since the amount of additives added into different types of polymers varies, average emissions data were used for the conversion of all the other types for “Additives” and “Others”.

For landfill and incineration processes, we used the life cycle GHG emissions data of mixed plastics from ecoinvent 3.4. For landfilling process, given that fossil-based plastics hardly degrade, only a small amount of GHG emissions is produced during collection and transportation. Incineration of per kilogram of plastic waste generates 3.92 MJ electricity and 7.66 MJ heat plastic according to ecoinvent 3.4, and these credits were used to calculate GHG emissions for incineration process.

The recycling process includes collection, transportation, sorting, separation and material recovery of the waste. The average emission value from PET and HDPE recycling processes (906 kg CO₂e/ton polymer) was calculated and used³⁵. To account for the GHG emissions credits from recycling EoL plastics, a substitution ratio of 80% is applied, meaning that 1 kg of recycled plastics avoid producing 0.8 kg of average market-mix plastics²⁰. As recycled content of average market-mix plastics changes over time under some scenarios, GHG credits from displacing them are calculated each year and subtracted from the GHG emissions generated from recycling.

The resulting GHG emissions data of fossil-based plastics at different life cycle stages can be found in Supplementary Table 3.

Life cycle GHG emissions of bio-based plastics

The most readily available feedstock for a specific region can be different. For example, Thailand and Brazil have excellent conditions for growing sugarcane, the USA is predominantly growing corn, while Europe has good farmland for growing sugar beet³⁶. In our study, corn and sugarcane are chosen. The emission data of Bio-PE, Bio-PET, PLA, PHAs and TPS production derived from corn and sugarcane were collected separately, with the direct and indirect land use change (LUC) emissions already included or calculated as elaborated later in this section. System expansion method was used to handle co-products such as electricity, heat, and digestate. The biological carbon sequestration credits were subtracted from corresponding life cycle GHG emission values for bio-based plastics (e.g. 3.14 kg CO₂/kg Bio-PE, 1.83 kg CO₂/kg PLA, 2.05 kg CO₂/kg PHB³⁷ and 1.94 kg CO₂e/kg TPS³⁸).

Bio-PE and Bio-PET are two major bio-based non-biodegradable plastics used today¹⁷. The production processes of Bio-PE including corn or sugarcane cultivation and harvest, ethanol fermentation and distillation, bio-ethylene production through dehydration, and polymerisation of bio-ethylene to polyethylene^{18,39}. To produce Bio-PET, instead of directly going through polymerisation, bio-ethylene is oxidised to ethylene oxide and hydrolysed to ethylene glycol, which then is polymerised with purified terephthalic acid (PTA) to obtain Bio-PET polymers⁴⁰. For corn-based PE and PET, we averaged Bio-HDPE and Bio-LDPE emission data¹⁸. For sugarcane-based PE, after adding LUC emissions, the net emissions in 2015 under the baseline scenario ranged from -0.7 to 1.8 kg CO₂e/kg Bio-PE⁴⁰ and average value was taken. Average value of emissions data from three geographical locations for Bio-PET resin production was used⁴⁰.

Polyhydroxybutyrate (PHB), the most common PHAs polymer, was selected as a representative PHAs type. A typical corn-based PLA/PHB polymer production process covers corn cultivation, corn wet milling, fermentation and polymerization/recovery, successively. The sugarcane-based production follows similar process with only the difference of sugarcane milling instead of corn milling. The production process of TPS involves corn cultivation, starch production and compounding. The emissions data of resin production for corn-based PLA/PHB and TPS are from ref. 37 and ref. 41, respectively. And the ones for sugarcane-based PLA and PHB are from ref. 42 and ref. 43, respectively.

For corn-based plastics, LUC emissions data of 89 kg CO₂e/ton corn was used³⁷. We used ref. 18 for the amounts of corn required for Bio-PE, Bio-PET, PLA, PHB and starch production. For sugarcane-based plastics, LUC emissions range between 0.16-2.38 kg CO₂e/kg for Bio-PE and 0.03-0.4 kg CO₂e/kg for Bio-PET⁴⁰; we used an average value for each plastic type. For sugarcane-based PLA, 63.6 kg CO₂e/ton PLA was used for LUC emissions⁴⁴.

Regardless whether the feedstock is fossil fuel or plants, further conversion of ethylene to Bio-PE or Bio-PET polymers remains the same⁴⁰. Therefore, the emission values of Bio-PE/PET conversion process are the same with fossil-based ones. The manufacturing technologies for plastics conversion into final products do not differ much between biodegradable plastics and conventional plastics^{37,45}. For example, PLA is usually processed by existing methods such as extrusion, thermoforming, injection molding, blow molding or cast film and sheet¹⁶. One slight difference is that prior to melting processing of PLA, the polymer must be dried sufficiently to prevent excessive hydrolysis which can compromise the physical properties of the polymer⁴⁶. However, no particular life cycle inventory data could be found for

biodegradable plastics processing so far. Therefore, we assumed that the emission values for biodegradable polymers conversion is also the same with conventional polymers.

The EoL treatments of Bio-PE and Bio-PET are no different from their fossil-based counterparts, given that they have identical properties and appearances. Therefore, they follow the same EoL mix of fossil-based plastics including recycling, incineration and landfill. In comparison, EoL management methods for biodegradable plastics can be recycling, incineration, landfill, composting or anaerobic digestion. Credits were given to generation of electricity, heat and digestate during incineration and composting processes. The efficiency of waste plastics to substitute virgin polymers is assumed as 80% for all recycling processes, except 74% for TPS, which will undergo higher quality loss during recycling³⁸. Recycled contents are assumed to replace an average market-mix of plastics for that year with 80% substitution rate as explained earlier.

The resulting emission values for bio-based plastics at different life cycle stages can be found in Supplementary Table 4.

Life cycle GHG emissions under low carbon energy scenario

Building upon the methodology in ref. 18, we explored the emissions under low carbon energy scenario (i.e. electricity from 100% wind power and biogas).

Contribution analysis of the life cycle emissions data was performed wherever possible. By closely examining the references, the amount of electricity and heat used in the production, conversion and EoL treatment processes were parsed out for different plastic types. Then we recalculated the emissions from the electricity and heat from low carbon energy sources in 2050, and assumed a linear increase of

low carbon energy in the energy mix from 2015 to 2050 to model a gradual energy decarbonisation process. The GHG emissions of each plastic type in 2050 under low carbon energy scenario was calculated by Equation (2):

$$E_{lc2050,j,k} = E_{j,k} - (E_{elec} + E_{heat}) + (E_{elec_{lc}} + E_{heat_{lc}}) \quad (2)$$

Where $E_{lc2050,j,k}$ is the GHG emissions of plastic type j in its life cycle stage k under low carbon energy scenario in 2050; $E_{j,k}$ is the GHG emissions of plastic type j in its life cycle stage k under current energy mix; E_{elec} and E_{heat} are the emissions produced from the generation of electricity and heat under the current energy mix, respectively; $E_{elec_{lc}}$ and $E_{heat_{lc}}$ are the emissions from the generation of electricity and heat under a low carbon energy scenario, respectively. All the emissions values are based on one unit of weight (i.e. one kilogram).

For fossil-based plastics resin production stage, the ratios between the emissions under low carbon energy scenario and that under conventional energy scenario in ref. 37 were applied. For conversion stage, blow molding, injection molding and extrusion processes fromecoinvent 3.4 were selected as representative conversion processes to calculate the average contributions of electricity and heat to GHG emissions (81.3% and 9.5%, respectively). For EoL stage, the electricity and heat generation credits from incineration were calculated using low carbon energy emission values. In addition, we calculated the process emissions from recycling by using the energy profile of recycling depicted in ref. 35. It is assumed that the diesel used for vehicles in waste collection in recycling process is replaced by electricity from wind. Supplementary Table 5 lists the GHG emission data of energy sources used to calculate our results under low carbon energy scenario.

For corn-based Bio-PE and Bio-PET resin production, emissions data in low carbon scenario were from ref. 18. For corn-based PLA and PHA resin production, the low carbon emissions data from ref. 22 was used, and the LUC emission data from ref. 12 was applied. For TPS production, the maize starch production process inecoinvent 3.4 was used as a proxy process, and the contribution of electricity and heat to the GHG emission are 24% and 17%, respectively. For sugarcane-based Bio-PE, 3.09 kWh electricity and 10.5 MJ diesel are used for per kilogram of Bio-LDPE produced³⁹, and they served as representative data for Bio-PE and Bio-PET due to unavailability of detailed energy use data on Bio-HDPE/PET production. For sugarcane-based PLA, the emissions from electricity and steam are 600 kg and 675 kg CO₂e/ton polymer, respectively⁴². For sugarcane-based PHB, the electricity and steam production are 1.1 kWh and 14.8 MJ per kilogram of polymer, respectively⁴³. Bio-PE and Bio-PET are assumed to produce the same amount of GHG emissions as their fossil-based counterparts during EoL management stage. As for biodegradable plastics, GHG emissions are assumed to stay unchanged for landfilling process; for incineration, composting and digestion, electricity and heat generation data from ref. 47 were used.

The GHG emissions values for fossil-based plastics and bio-based plastics under low carbon energy scenario can be found in Supplementary Table 6 and 7, respectively.

Plastics demands

Beginning with the amount of plastics produced in 2015¹, two scenarios are evaluated until 2050, assuming an annual resin production growth rate of 4% (average annual growth rate of 2010-2015) and 2% (a slower growth trend of plastics production). For 100% fossil-based plastics scenario, the market share of

each plastic type is assumed to remain unchanged. For corn- and sugarcane-based plastics scenario, the market share of bio-based plastics is assumed to linearly grow from zero in 2015 to 100% in 2050, given that the global market share of bio-based plastics in 2017 was less than one percent¹⁷. Additionally, it is assumed that bio-based plastics substitute for conventional plastics on a 1:1 scale by weight.

Substitution assumptions

Today, there is a bio-based plastic alternative for almost every conventional plastic and the corresponding application¹⁷. A report regarding the technical substitution potential of bio-based polymers concludes that 90% of the conventional polymers can be technically replaced worldwide⁴⁸. Considering biopolymer technology advancement, it is assumed that all fossil-based plastics can be replaced by bio-based plastics scenarios by 2050.

In 2017, bio-based non-biodegradable plastics accounted for 56% of the global bioplastics market. These so-called “drop-in” solutions have the same properties, conversion processes and disposal methods with their fossil-based counterparts and therefore serve as perfect substitutes. Bio-PE are assumed to replace the majority of fossil-based PE, PVC and PUR, while Bio-PET to replace PET and PP&A fibers. Other types of bio-based non-biodegradable plastics are not considered in this study since they are not yet available at a commercial scale or there is a lack of data in the literature.

Bio-based biodegradable plastics make up the rest 44% of the bioplastics market, with PLA and PHAs driving the growth¹⁷. PLA is the most versatile biodegradable plastic type and has wide applications across food packaging, medical devices, agriculture films, among others⁴⁹⁻⁵¹. It has comparable mechanical and thermal

properties with PS and PET, and can also replace PE, PP, and PVC in some applications³⁷. The use of PLA to replace nylon and PET in the textile industry is also increasing⁴⁵. PHAs have been used in fibers, non-woven materials, disposable products⁵², cosmetic and food containers⁵¹. Commercialized PHAs can frequently replace PE, PP and PS, and may also substitute for PET and PVC³⁷. The high price of PHAs is a major barrier to its large scale commercialization⁵². TPS is used in specialized agricultural applications, as filler in plastic composites, or in single-use items like bags, containers, diapers and tampons^{51,53}. Pure TPS has poor mechanical properties and is susceptible to water, which limits its potential product applications⁵⁴. However, it is a common practice to blend starch with other polymers such as PLA, PCL, and PHAs to obtain composites to improve its properties⁵⁰.

Based on the technical substitution potential, comparable properties, common application areas and the market growth reviewed above, a substitution plan was developed for bio-based plastics to replace conventional plastics (see Supplementary Table 9).

End-of-life management

The projected EoL management mix change of all plastic types (fossil-based and bio-based) between 2015 and 2050 is shown in Supplementary Table 10. The mix in 2050 is determined based on the projections of future EoL change¹, as well as the historical changes of the plastic waste EoL management in Europe and the United States. Linear change of the rate of each EoL method is assumed between 2015 and 2050.

Limitations

There are uncertainties and limitations associated with the data and the model employed in this study. We made various assumptions to simplify the processes involved in plastics life cycle. For example, we assumed that the indirect land use change and the GHG emissions from agricultural expansion for bio-based plastics would remain at the current level. We also extracted and combined emissions data from multiple sources. Conventional plastics data are from ecoinvent 3.4, which are originally Eco-profiles of the European plastics industry (PlasticsEurope). The data contains outdated numbers and uses extrapolation for the regions other than Europe. Therefore, the temporal and geographical representation of the data was identified as a weakness, while no better data sources were identified. The methods to calculate LUC emissions associated with bio-based plastics production vary in the literature and warrant further research.

Data availability

The authors declare that the main data supporting the findings of this study are available within the article and its Supplementary Information file. Extra data are available from the corresponding author upon reasonable request.

References

1. Geyer, R., Jambeck, J. R. & Law, K. L. Production, use, and fate of all plastics ever made. *Sci. Adv.* **3**, e1700782 (2017).
2. World Economic Forum, Ellen MacArthur Foundation & McKinsey & Company. *The New Plastics Economy - Rethinking the future of plastics.* (2016).
3. Jambeck, J. R. *et al.* Plastic waste inputs from land into the ocean. *Science* **347**, 768–771 (2015).

4. Law, K. L. Plastics in the Marine Environment. *Annu. Rev. Mar. Sci.* **9**, 205–229 (2017).
5. Law, K. L. & Thompson, R. C. Microplastics in the seas. *Science* **345**, 144–145 (2014).
6. Rochman, C. M. *et al.* Policy: Classify plastic waste as hazardous. *Nature* (2013). doi:10.1038/494169a
7. Lithner, D., Larsson, Å. & Dave, G. Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. *Sci. Total Environ.* **409**, 3309–3324 (2011).
8. Fishedick, M. *et al.* *Climate Change 2014: Mitigation of Climate Change Ch. 10 - Industry.* (2014).
9. Hillmyer, M. A. The promise of plastics from plants. *Science* **358**, 868–870 (2017).
10. Weiss, M. *et al.* A Review of the Environmental Impacts of Biobased Materials. *J. Ind. Ecol.* **16**, S169–S181 (2012).
11. Yates, M. R. & Barlow, C. Y. Life cycle assessments of biodegradable, commercial biopolymers—A critical review. *Resour. Conserv. Recycl.* **78**, 54–66 (2013).
12. Chen, G.-Q. & Patel, M. K. Plastics Derived from Biological Sources: Present and Future: A Technical and Environmental Review. *Chem. Rev.* **112**, 2082–2099 (2012).
13. Spierling, S. *et al.* Bio-based plastics - A review of environmental, social and economic impact assessments. *J. Clean. Prod.* **185**, 476–491 (2018).
14. Albertsson, A.-C. & Hakkarainen, M. Designed to degrade. *Science* **358**, 872–873 (2017).

15. OECD. *Policies for Bioplastics in the Context of a Bioeconomy*. (2013).
doi:10.1787/5k3xpf9rrw6d-en
16. European Commission. *A European strategy for plastics in a circular economy*. (2018).
17. European Bioplastics. *Bioplastics market data 2017*. (2017).
18. Posen, I. D., Jaramillo, P., Landis, A. E. & Griffin, W. M. Greenhouse gas mitigation for U.S. plastics production: energy first, feedstocks later. *Environ. Res. Lett.* **12**, 034024 (2017).
19. Hopewell, J., Dvorak, R. & Kosior, E. Plastics recycling: challenges and opportunities. *Philos. Trans. R. Soc. B Biol. Sci.* **364**, 2115–2126 (2009).
20. Lazarevic, D., Aoustin, E., Buclet, N. & Brandt, N. Plastic waste management in the context of a European recycling society: Comparing results and uncertainties in a life cycle perspective. *Resour. Conserv. Recycl.* **55**, 246–259 (2010).
21. Hottle, T. A., Bilec, M. M. & Landis, A. E. Sustainability assessments of bio-based polymers. *Polym. Degrad. Stab.* **98**, 1898–1907 (2013).
22. Olivier, J. G. J., Schure, K. M. & Peters, J. A. H. W. *Trends in global CO₂ and total greenhouse gas emissions - 2017 Report*. 69 (2017).
23. Zhu, J.-B., Watson, E. M., Tang, J. & Chen, E. Y.-X. A synthetic polymer system with repeatable chemical recyclability. *Science* **360**, 398–403 (2018).
24. Bing, X., Bloemhof-Ruwaard, J., Chaabane, A. & van der Vorst, J. Global reverse supply chain redesign for household plastic waste under the emission trading scheme. *J. Clean. Prod.* **103**, 28–39 (2015).
25. Soroudi, A. & Jakubowicz, I. Recycling of bioplastics, their blends and biocomposites: A review. *Eur. Polym. J.* **49**, 2839–2858 (2013).

26. Reddy, R. L., Reddy, V. S. & Gupta, G. A. *Study of Bio-plastics As Green & Sustainable Alternative to Plastics*. (2013).
27. Wernet, G. *et al.* The ecoinvent database version 3 (part I): overview and methodology. *Int. J. Life Cycle Assess.* **21**, 1218–1230 (2016).
28. European Life Cycle Database (ELCD). Available at: <http://eplca.jrc.ec.europa.eu/ELCD3/index.xhtml?stock=default>. (Accessed: 23rd April 2018)
29. Voet, E. van der, Oers, L. van, Rydberg, T., Westerdahl, J. & Larsen, H. F. Life Cycle Assessment of Additives: Methodology and Data. in *Global Risk-Based Management of Chemical Additives II* 7–23 (Springer, Berlin, Heidelberg, 2012). doi:10.1007/698_2012_185
30. European Council for Plasticisers and Intermediates. *Eco-profiles and Environmental Product Declarations of the European Plastics Manufacturers: Diisononyl phthalate (DINP)*. (2015).
31. Keoleian, G., Miller, S., De Kleine, R., Fang, A. & Mosley, J. *Life cycle material data update for GREET model*. (2012).
32. Franklin Associates. *Life Cycle Inventory of Plastic Fabrication Processes: Injection Molding and Thermoforming*. 63 (2011).
33. Madival, S., Auras, R., Singh, S. P. & Narayan, R. Assessment of the environmental profile of PLA, PET and PS clamshell containers using LCA methodology. *J. Clean. Prod.* **17**, 1183–1194 (2009).
34. Thomas, B. *A Carbon Footprint for UK Clothing and Opportunities for Savings*. (2012).
35. Franklin Associates. *Life Cycle Inventory of 100% Postconsumer HDPE and PET Recycled Resin from Postconsumer containers and packaging*. (2010).
36. Lovett, J. Sustainable Sourcing of Feedstocks for Bioplastics. 17 (2016).

37. Posen, I. D., Jaramillo, P. & Griffin, W. M. Uncertainty in the Life Cycle Greenhouse Gas Emissions from U.S. Production of Three Biobased Polymer Families. *Environ. Sci. Technol.* **50**, 2846–2858 (2016).
38. Rossi, V. *et al.* Life cycle assessment of end-of-life options for two biodegradable packaging materials: sound application of the European waste hierarchy. *J. Clean. Prod.* **86**, 132–145 (2015).
39. Liptow, C. & Tillman, A.-M. A Comparative Life Cycle Assessment Study of Polyethylene Based on Sugarcane and Crude Oil. *J. Ind. Ecol.* **16**, 420–435 (2012).
40. Tsiropoulos, I. *et al.* Life cycle impact assessment of bio-based plastics from sugarcane ethanol. *J. Clean. Prod.* **90**, 114–127 (2015).
41. Shen, L. & Patel, M. K. Life Cycle Assessment of Polysaccharide Materials: A Review. *J. Polym. Environ.* **16**, 154 (2008).
42. Groot, W. J. & Borén, T. Life cycle assessment of the manufacture of lactide and PLA biopolymers from sugarcane in Thailand. *Int. J. Life Cycle Assess.* **15**, 970–984 (2010).
43. Harding, K. G., Dennis, J. S., von Blottnitz, H. & Harrison, S. T. L. Environmental analysis of plastic production processes: Comparing petroleum-based polypropylene and polyethylene with biologically-based poly- β -hydroxybutyric acid using life cycle analysis. *J. Biotechnol.* **130**, 57–66 (2007).
44. Tsiropoulos, I. *et al.* Life cycle assessment of sugarcane ethanol production in India in comparison to Brazil. *Int. J. Life Cycle Assess.* **19**, 1049–1067 (2014).
45. Castro-Aguirre, E., Iñiguez-Franco, F., Samsudin, H., Fang, X. & Auras, R. Poly(lactic acid)—Mass production, processing, industrial applications, and end of life. *Adv. Drug Deliv. Rev.* **107**, 333–366 (2016).

46. Lim, L.-T., Auras, R. & Rubino, M. Processing technologies for poly(lactic acid). *Prog. Polym. Sci.* **33**, 820–852 (2008).
47. Hermann, B. G., Debeer, L., De Wilde, B., Blok, K. & Patel, M. K. To compost or not to compost: Carbon and energy footprints of biodegradable materials' waste treatment. *Polym. Degrad. Stab.* **96**, 1159–1171 (2011).
48. Shen, L., Haufe, J. & Patel, M. K. *Product Overview and Market Projection of Emerging Bio-based Plastics*. 243 (Utrecht University, 2009).
49. Ashter, S. A. Commercial Applications of Bioplastics. in *Introduction to Bioplastics Engineering* 227–249 (Elsevier, 2016). doi:10.1016/B978-0-323-39396-6.00009-9
50. Luckachan, G. E. & Pillai, C. K. S. Biodegradable Polymers- A Review on Recent Trends and Emerging Perspectives. *J. Polym. Environ.* **19**, 637–676 (2011).
51. Babu, R. P., O'Connor, K. & Seeram, R. Current progress on bio-based polymers and their future trends. *Prog. Biomater.* **2**, (2013).
52. Chanprateep, S. Current trends in biodegradable polyhydroxyalkanoates. *J. Biosci. Bioeng.* **110**, 621–632 (2010).
53. Wang, X.-L., Yang, K.-K. & Wang, Y.-Z. Properties of Starch Blends with Biodegradable Polymers. *J. Macromol. Sci. Part C* **43**, 385–409 (2003).
54. Broeren, M. L. M., Kuling, L., Worrell, E. & Shen, L. Environmental impact assessment of six starch plastics focusing on wastewater-derived starch and additives. *Resour. Conserv. Recycl.* **127**, 246–255 (2017).

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Contributions

J.Z. performed the research and analysed the data. S.S. conceived the idea and designed the study. Both authors wrote the manuscript.

Competing interests

The authors declare no competing financial interests.

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