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1 | Introduction

There is hardly any material that better symbolizes modern lifestyle than plastic. In a variety of ways, it has invaded value chains and people's homes due to its superior properties and its usability for a wide range of applications. Plastic material is light, durable, tearproof, corrosion resistant and easy to modify. For a long time, these obvious advantages have tempted people to shut their eyes to the downsides of a plastic economy. Foremost, these concern the environmental consequences of plastic use. First, plastic production in its current form is emission-intensive with respect to greenhouse gases and pollutants across all process stages. Second, mainly due to their technical properties, plastics pose a considerable challenge for end-of-life treatment. In case of uncontrolled littering, plastic material can cause longlasting damage to ecosystems and food chains. The most striking example is the emergence of huge plastic accumulation zones in the oceans, such as the great pacific garbage patch. It is especially this unsettling phenomenon that has gained increasing media attention recently, spurring a public debate on improved waste collection and alternatives to plastic use.

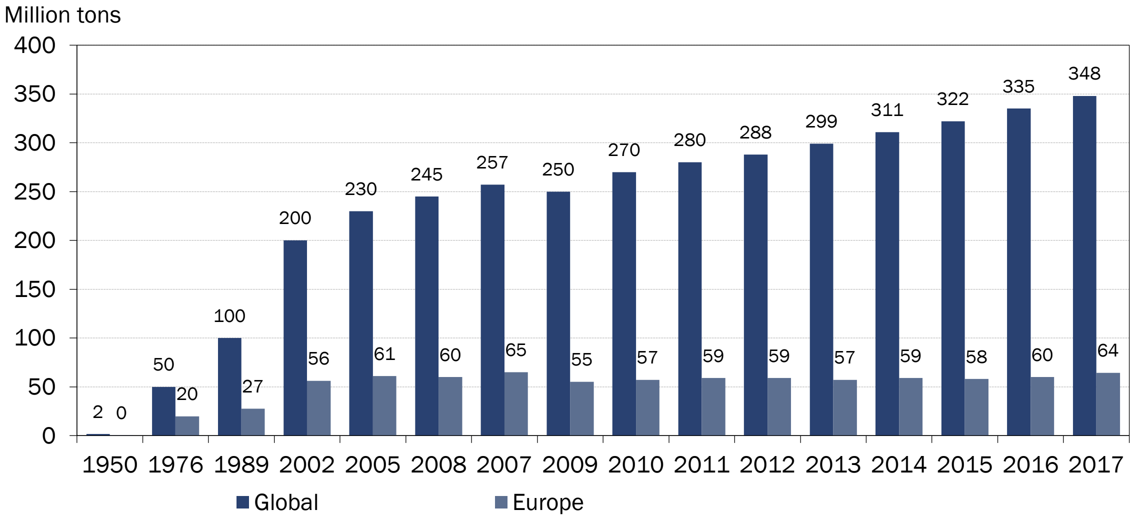
To come up with sound recommendations for economics and policy-making, it is however essential to be informed on the exact nature of the environmental impacts of specific materials and to what extent potential technology alternatives represent remedies in this regard. Over the last decades, Environmental Life Cycle Analysis (ELCA) has established as an interdisciplinary tool to investigate the various ways in which a product interacts with the environment, from the first stages of its production to its destiny after use ("cradle-to-grave" approach). By now, it has also widely been applied to assessments of plastic products and their alternatives. Given the societal dimension of the problem, the significant amount of information gathered by this literature should not exclusively remain in the realms of technicians, but be disseminated to a wider scientific and non-scientific audience as well.

The main purpose of this article is to provide an introduction to the methodological approach and current insights of the ELCA literature related to plastic from the point of view of a non-technician. Concerning potential remedies, we set our focus on the question to what extent the use of biomass as an alternative feestock in plastic production can contribute to a reduction of the environmental burden. The paper starts with a brief overview on current statistics related to plastics in section 2. Section 3 lays out the fundamentals of the ELCA method and discusses its challenges and limitations. Section 4 adresses life cycle issues of conventional plastics, first by discussing the types of impacts occuring at single process stages and then by summarizing recent results of the ELCA literature in this respect. Section 5 introduces the emerging industry of bioplastics and presents some existing results on the environmental performance of their products.

2 | Facts on plastic production and use

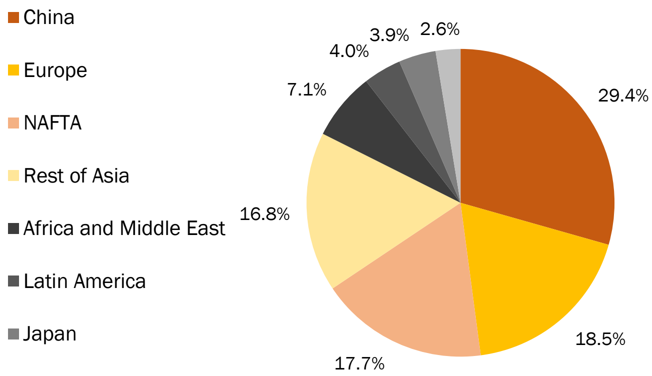
Originally, the word “plastic” describes the property of a substance not to break when being deformed. In practice, it is an umbrella term comprising any synthetic or semisynthetic organic polymer that fulfills this property. Plastic thus refers to a wide range of materials serving a multitude of purposes. Over the last decades, it has become an increasingly indispensable part of our everyday lifes. It is used for packaging, as construction material, in the production of electronics, textiles, vehicles, machinery, and as a component of cosmetics and many other consumer goods. The increased demand can be showcased by the development of the worldwide plastic production that is steadily increasing over the last decade (Fig. 1). From 2002 to 2017, the worldwide production increased by about 75 %. The only exception marks 2009 as the sole decrease during that time. While the production in Europe has seemingly hit a plateau in recent years, the world production kept rising. This is mainly due to the production of Asian countries (and especially that of China) that has increased sharply. This is apparent in the regional distribution of plastic production in comparison of 2009 and 2017 (Fig. 2 and Fig. 3). The similar amounts of Europe with 55 and 64 tons, respectively, represent vastly different shares of 25 % and 18.5 %. This demonstrates that there are growing as well as stable submarkets for plastic production. According to estimates by Geyer et al. (2017), packaging applications currently make up the by far biggest share of global plastic use (approx. 47 % in 2015), followed by textiles and transport.

Figure 1: Worldwide and European production volumes of plastics



Source: PlasticsEurope (2018).

Figure 2: Distribution of global plastic production by region in 2017

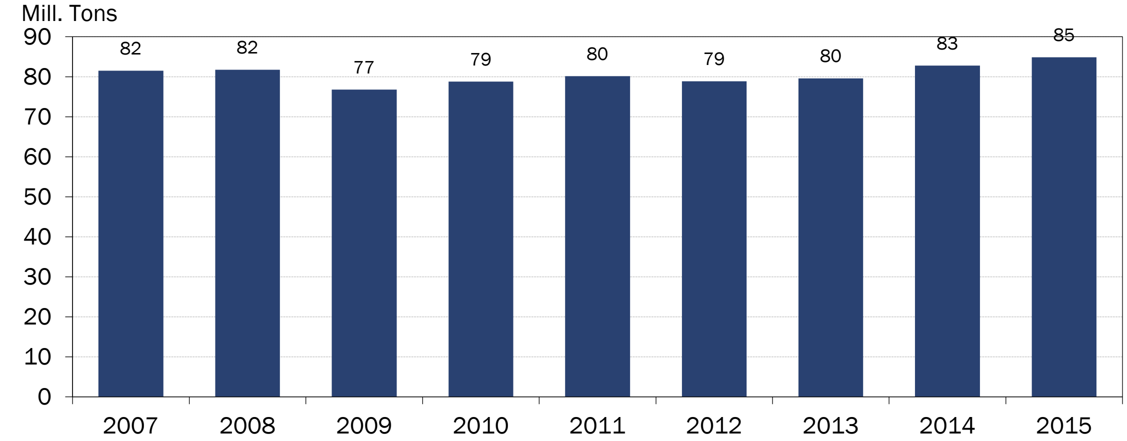


Source: PlasticsEurope (2018).

The vast quantities of plastics result in waste after the use stage, whether the product takes the form of an intermediate good or is used by consumers. The issue of pollution and plastic waste receives intense media coverage with hotly debated topics such as microplastic pollution and upstreaming in the food chain or marine pollution in general. Concerning the latter issue, current projections see the problem of marine pollution worserning increasingly (Jambeck et al., 2015). It results from land-based plastic litter entering the ocean from coastlines and through rivers, but also marine-based plastics used in fisheries. These plastics tends to accumulate in certain ocean hotspots, which are, due to transport by ocean currents, in parts also located in the less populated Southern hemisphere. The fact that plastics break down more easily in an aquatic environment makes this development even more worrysome. Smaller particles are more likely to be swallowed by marine animals and thereby enter the food chains.

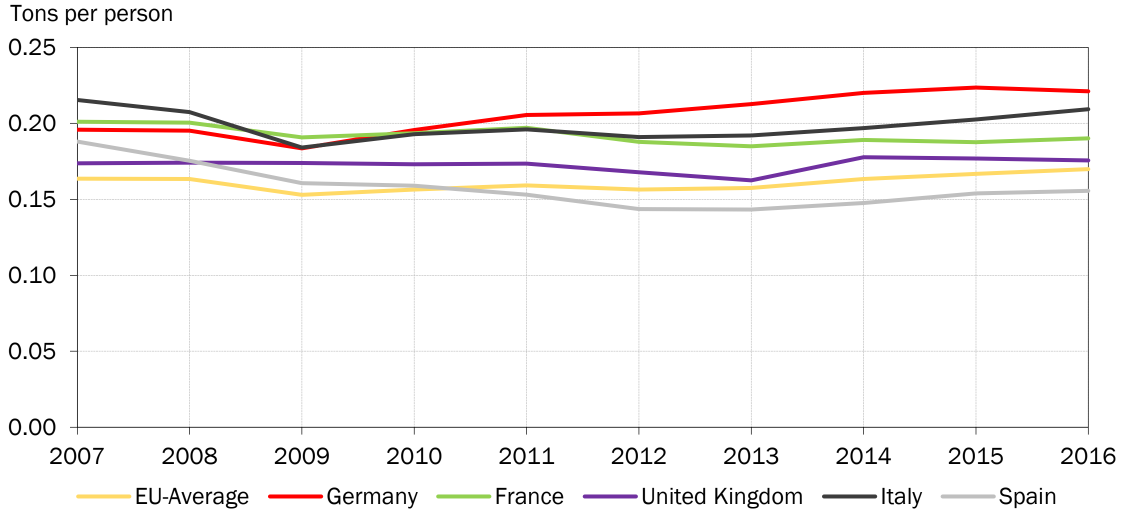
Over recenty years, the level of waste production of plastic products in Europe seems to be quite stable. Plastic waste from packaging origins lies in annual magnitude of around 80 Mill. Tons in the past years (Fig. 3). A dip of five million tons is recorded for 2009, which might be based on the financial crisis in that year. It took until 2014 to reach higher levels of waste than in 2008. The European average for plastic packaging waste is around 170 kg per person per year (Fig. 4). Developments at country level are heterogeneous, but generally, countries are relatively stable in their plastic waste production. The amount produced in Germany increased between 2007 and 2016 from under 200 kg pp per year to around 225 kg pp per year, while Spain saw a significant dip from 190 kg to around 150 kg over the same time period.

Figure 3: Generated plastic packaging waste in the EU



Source: Eurostat (2019).

Figure 4: Plastic packaging waste per person in EU countries

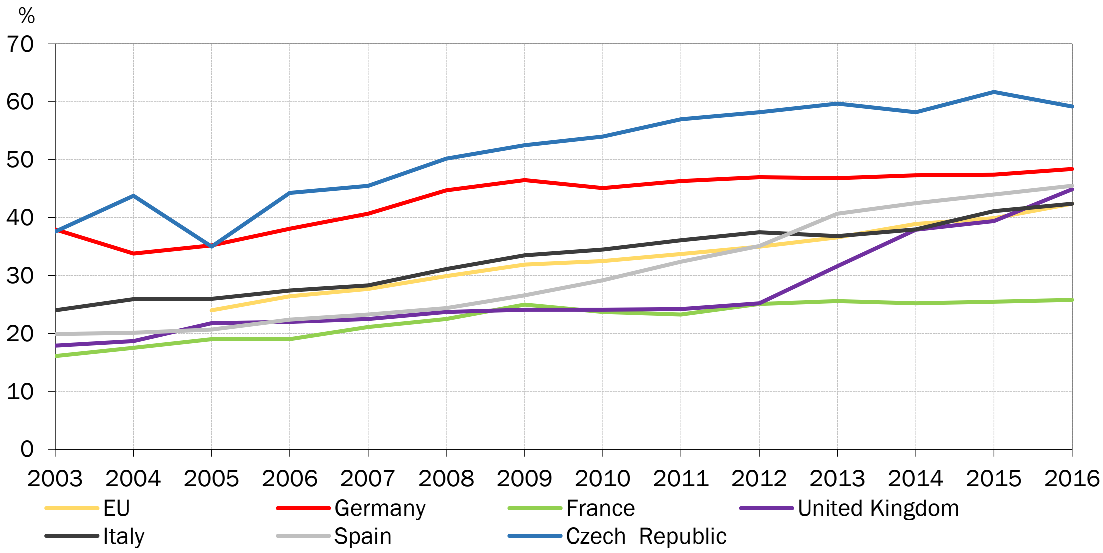


Source: Eurostat (2019).

The occurring plastic waste can be handled by different kinds of treatment. Common end-of-life scenarios are landfills, incineration with energy recovery and recycling (a detailed discussion of these technologies can be found in section 4). The rate of recycling varies among countries of the EU, but the trend over the last two decades seems to be converging and increasing (Fig. 5). In 2016, recycling overtook landfills in plastic waste treatment for the first time, with 31.3 % and 27.1 %, respectively, while energy recovery was at 41.6 % (PlasticsEurope, 2018). Among the biggest economies of the EU, Germany has started with the highest rate of around 37 %. At the same time, however, progress over recent years is not very impressive, with about 10 percentage points, reaching a rate of 47 % in 2016. By contrast, the UK started at a much lower level and experienced a significant increase, reaching a recycling share of about 44 % in 2016. France, on the other hand, started at a low rate and could so far not achieve a significant improvement, exhibiting a share of only 26 % in 2016.

The sheer size of these numbers showcases the social relevance exerted by a mutual influence of the plastic industry, prevailing behavorial patterns of consumers and the existing regulatory framework. The issue adresses several dimensions of justice, e.g. related to the longevity of the bequeathed waste, the use of resources for the production of plastics or to the global distribution of this production. However, in order to seize appropriate measures, reliable information on the overall environmental impact of the various stages of plastic production and use is required. This information can only be obtained based on a sound methodological framework that ensures transparency and allows for comparisons among technology alternatives. A widely applied method that is supposed to achieve such a comprehensive assessment is the Environmental Life Cycle approach. Its potentials and limitations are discussed in the subsequent sections, first in general terms and then applied to plastics.

Figure 5: Recycling rates for plastic packaging in EU countries



Source: Eurostat (2019).

3 | The Environmental Life Cycle Assessment (ELCA) method

The general concept of an *Environmental Life Cycle Assessment (ELCA)* is a holistic approach. The environmental effects of a specific product or production technology are captured, listed and evaluated along the whole cycle of use ("from cradle to grave"). This life cycle thinking considers not only the environmental burden of resource extraction, production and transport, but also the ecological impact of use and (potential) reuse as well as the long-term effects of the final disposal as waste. As a consequence, the execution of an ELCA requires a thorough understanding of the relevant process stages and their linkages. Impacts of the single stages are measured by heterogeneous indicators collected from different sources. Missing information often has to be replaced by general guidelines from the literature. Apart from the technical complexity, this raises further issues related to validity and comparability of the figures utilized. Over time, these challenges have fostered the establishment of generally accepted principles for the analytical framework, manifested in own ISO standards (ISO 14040).

At the top level, an ELCA can be decomposed in two major steps: an inventory analysis and the actual assessment. The *Life Cycle Inventory (LCI)* comprises the process of collecting and listing the various flows associated with the single stages of the life cycle. Depending on the nature of the task, these flows can take two different forms. First, they could constitute the total impact of the considered product/production technology, regardless of the impact of alternative products or means of production. Such an LCI is termed an *attributional LCI*, as it measures all the flows that can be attributed to the investigated object. Alternatively, the flows could measure changes (instead of absolute levels) in the environmental impacts caused by a certain action (e.g. a change in product design, the substitution of materials). It involves the definition of a benchmark case as a starting point. This type of analysis is termed a *consequential LCI* (Rebitzer et al., 2004). It can be used as a tool in the development process, in contrast to the more ex-post oriented attributional LCI. Provided that the benchmark case is reasonably specified, it offers more insightful information than a pure look at a single technology.

Given extent and heterogeneity of the information required, the researcher has to explicitly define *system boundaries* for the LCI to ensure interpretability and comparability. This both concerns the spatial and the time dimension. In the spatial dimension, a challenge might lie in the fact that production of the investigated product is split into various stages scattered over different regions (or countries). It has to be clarified whether the analysis captures the impact at a global or at a local level and, in the latter case, how the spatial boundaries are exactly defined. This gets further complicated in the likely instant when effects differ in their spatial scope. For instance, impacts of the emissions of greenhouse gases are of a global nature, while emissions of air pollutants (aerosols, trace gases) might primarily affect only the local environment. In principle, this can be dealt with by transforming indicators into spatially consistent measures in the latter analytical stages of *characterization* and *normalization* of impacts. Concerning the time dimension, one challenge is typically the treatment of the end-of-use scenario. It involves decisions to what extent recycling (or other conversion) processes and their outcomes (second life scenarios) are incorporated. Concerning the final disposal of products as waste, one also needs to clarify to what extent impacts of long-term emissions related to the disposal (e.g. methane emissions from landfills) are covered (Hauschild et al., 2008).

Other methodological challenges associated with an LCI involve the identification of processes, the definition of precise functional units and the product-specific assignment in presence of multifunctionality. For process identification, it is necessary to consider and disentangle all the process steps within the defined system boundaries as well as the commodity flows between them. Common tools in process analysis like flow diagrams or matrix representations can be assistant in this (Suh & Huppes, 2005). More recent approaches propagate the integration of Input-Output techniques into the analysis, allowing processes to be modelled based on readily available Input-Output-Tables (Durairaj et al., 2002). In principle, this allows to widen the analysis to an economy-wide dimension, including indirect environmental effects in up- and downstream sectors. A crucial requirement is however that Input-Output-Tables exist in the sectoral and spatial resolutions needed.

The functional unit is a quantitative performance measure (expressed in a specific unit of measurement) for which the various types of environmental impacts are identified, i.e. it is the reference point of the analysis. For instance, when applied to the life cycle of a certain product, this performance measure can take the form of a specific production quantity. An appropriate choice of functional units is essential for a meaningful comparison of the results to related studies. This holds especially in presence of nonlinearities in the extent of damages, i.e. cases where damages are not proportionate to the functional unit chosen. In such cases, functional units should be specified by means of assumptions on actual performance levels (e.g. expected production quantities).

Multi-functionality is the issue that arises when an investigated process is characterized by not one, but several outcomes. For instance, a certain production process could generate not one, but several products. Undertaking an LCI for a single product then requires researchers to specify which share of the overhead environmental costs of production (e.g. electricity, resources used for machinery and maintenance) to assign to the specific product. This is especially challenging in case of fixed costs, where production quantities might not represent appropriate distribution keys. The literature has proposed several guidelines how to distribute impacts among products under circumstances like these (Rebitzer et al., 2004).

An LCI as such is not a useful tool for evaluation and decision-making, as its final result merely consists of a set of environmental indicators measured in different units. These indicators stem from different categories, harming the environment to a different extent and in different ways. An actual impact evaluation is therefore required as a subsequential analysis, officially termed *Life Cycle Impact Assessment (LCIA)*. According to the guidelines of the official standards, it can consist of the following steps: characterization, normalization and weighting. *Characterization* is the process of assigning damage measures to the single indicators and summing up damages by category. It involves the determination of a characterization factor, which transforms the indicator to an environmental damage measure. Those indicators to which the same damage measure can be applied are assigned to a common category. For instance, emissions of CO_2 and CH_4 (methane) could be summarized by means of their Global Warming Potentials (GWP) and thereby allocated to the common category "greenhouse gases". System boundaries are also in the specification of characterization factors a difficult issue. Different damage categories are often associated with characterization factors of different time or spatial horizon (for instance, the immediate damages caused by the emissions of pollutants vs. the long-term nature of the impact of greenhouse gases). This limits comparability across categories.

The next step, *normalization*, consists of putting the characterization factors in relation to damages occurring in external reference situations. A practical reason is to consolidate the different factors to one and the same dimensionless unit, as a preparatory step before applying a weighting scheme. If reference situations are carefully selected, the normalized values can also have sensible interpretations. For instance, if reference values represent some sort of typical damage level observed for standard processes, the normalized characterization factors can be interpreted as the relative extent to which the impact of the investigated process differs from the standard case (Pizzol et al., 2017). In this way, researchers can assess whether the introduction of a new technology is associated with progress or regress in specific impact categories. However, while shedding light on potential environmental trade-offs in the choice between different processes, normalization itself does not establish a clear ranking of alternatives.

To arrive at a conclusion, which process should be regarded as the environmentally friendliest, the different impact categories need to be summarized to an aggregate measure. This involves assigning weights to the single categories. Understandably, this is the by far most controversial step in an LCIA. The relevant ISO standards take a very critical view on weighting, questioning its scientific nature (Finkbeiner et al., 2006). Many studies therefore refrain from this last step and simply list the outcomes of the different impact categories as a final result. In any case, weighting has to be value-based and therefore involves some degree of subjective judgement outside the realms of technical and natural sciences. Fortunately, this doesn't mean that it has to be arbitrary. Researchers can draw upon methods from Economics and Social Sciences to define appropriate weights, e.g. by ascertaining people's willingness-to-pay for the prevention of certain damages. Over time, standard approaches for this have also evolved. For instance, the Virtual Eco-Cost approach assigns monetary weights to different forms of pollutions based on the respective prevention costs, which allows to quantify the aggregate ecological costs of a specific technology (Vogtländer et al., 2001).

4 | Life cycle assessment of conventional plastic products

4.1 | Assessment by process stages

4.1.1 | Resource extraction

Crude oil and natural gas are the major raw materials in plastic production. A discussion of the lifecycle impact of plastic products therefore has to address environmental issues related to oil and gas drilling. Oil drilling is the challenging process of drilling and pumping of oil from underground wells. One immediate impact consists of the contamination of the local surroundings by oil spills, which are to some degree unavoidable. Their toxicity and the slow rate of natural degradation cause them to be a considerable threat especially to aquatic ecosystems, where the oil forms an emulsion with water. Oil can thereby enter marine food chains and contribute to a depletion of dissolved oxygen in the water. On land, oil spills can impair the growth of crops (Onwurah et al., 2007). Furthermore, the infrastructure required for the drilling activities can transform local landscapes in an irrevocable manner, eliminating vegetation and wildlife habitats (Allred et al., 2015). It can also boost soil erosion, increasing the exposure of areas towards floods. Both oil and gas drilling are associated with significant emissions of the greenhouse gases carbon dioxide and methane. This takes the form of accidental leaks, but also deliberate releases. On oil sites, natural gas gained as a by-product of the extraction process is sometimes combusted in absence of appropriate transport infrastructure ("gas flaring"). Moreover, also unburned gas is on some occasions discharged intentionally during the extraction process ("gas venting"), e.g. for safety reasons. Environmental consequences of the latter activity are more drastic than those of the former, because the latter is associated with massive methane emissions, while the form oxidizes gases to carbon dioxide, the gas with the comparatively smaller global warming potential (Ismail & Umukoro, 2012). Therefore, drilling activities also immediately contribute to global warming.

4.1.2 | Refining

Both crude oil and natural gas are not chemically homogeneous substances. They consist of mixtures of different chemical compounds, mainly different forms of hydrocarbons. These compounds exhibit individual chemical and physical properties, making them eligible for specific products. The major task of the refining industry is to isolate these single substances from the mixture, a process referred to as fractioning. In case of crude oil, this is achieved by means of fractional distillation. Crude oil is heated to cause evaporation, with the resulting gas rising into a fractioning tower. Due to the fact that boiling points of the single substances differ, they will condense at different heights and thus end up in different chambers of the fractioning tower, achieving a spatial separation. One of the substances that can be distilled in such a way is *Naptha*, itself a mixture of liquid hydrocarbons. It serves as the main building block for many plastic products. Before that, however, it has to be divided by a subsequent thermal process into its components ethylene and propylene. These rather simply structured molecules are termed *monomer molecules*. They are linked through chemical bonding to form more complex polymers as part of the actual plastic production process.

The refinery process also intervenes into the environment in several ways. Foremost, this concerns air pollution hazards. Refineries emit hazardous air pollutants such as BTEX compounds (benzene, toluene, ethylbenzene, and xylene), both as a consequence of heating processes and due to leakages (Baltrėnas et al., 2011). Exposure to critical amounts can cause significant health risks, especially regarding benzene, which is a known carcinogen (Maltoni et al., 1983). A range of other air pollutants with detrimental health impacts like particulate matter (PM), nitrogen oxides (NO_x), carbon monoxide (CO), hydrogensulfide (H_2S) and sulfur dioxide (SO_2) are likewise emitted, potentially contributing to an increase in secondary pollutants like ozone (O_3) as well. Impacts also concern the aquatic environment: the affluents emitted by the process contain a wide range of chemicals, which are at least in parts toxic, imposing danger on aquatic species (Wake, 2005). Soil contamination with its potential adverse effects on land productivity is another issue (Iturbe et al., 2004).

4.1.3 | Manufacturing of plastic and plastic products

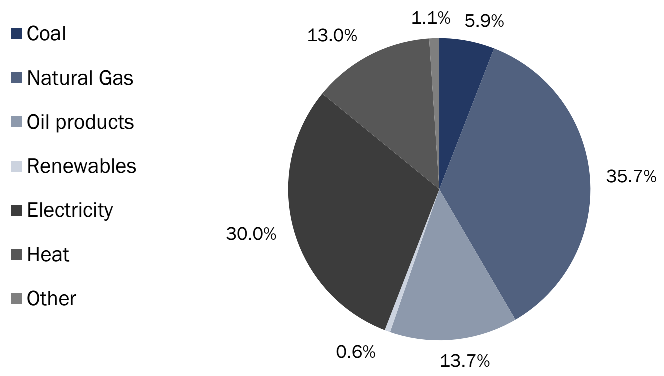
Over the years, a wide range of artificial polymers have been developed, differing substantially in their physical properties and typical uses. A basic distinction can be made between thermoplastics, which can be melted and then hardened again several times, and thermosets, which remain hard after the first cooling process. An obvious advantage of the former is its potential for recycling. In any case, production of plastic products involves two major steps, production of the plastic raw material (plastic resin) and its transformation into a specific plastic product. In material production, two types of chemical processes are common: polymerization and polycondensation. Polymerization consists of sequences of chemical reactions that form polymer chains from monomer molecules. By contrast, in polycondensation polymers are formed based on condensation reactions, potentially between monomers of different composition. A practical distinction is that polycondensation involves the production of small molecules as by-products, particularly water. In transformation, depending on the appearance of the final product, several molding techniques can be applied, such as injection molding, extrusion molding or thermoforming.

Regarding plastic packaging, production methods also differ between specific products. For instance, in the production of plastic bottles, Polyethylene Terephthalate (PET) commonly serves as a raw material, due to its robustness and comparatively low weight. It is generated based on a polymerization process and afterwards formed into a bottle by means of stretch blow molding. By contrast, basic materials for plastic wrap are typically Polyethylene, Polyvinyl chloride (PVC) or Polyvinylidene chloride (PVDC). All three are generated from simple hydrocarbons as part of polymeriziation processes. The actual wrap is then produced by extrusion molding, with a particular focus on achieving the desired thinness of the product (Gait & Hancock, 1970).

The manufacturing stage is also associated with significant emissions of greenhouse gases. For the most part, this is due to the energy use in the sector. In plastics production, considerable amounts of energy are required both in the forms of electricity and heat. For instance, Elduque et al. (2015) have demonstrated that electricity consumption is the most important factor in the environmental impact of injection molding. At the aggregate level, national energy balances unfortunately do not feature a sectoral resolution high enough to capture specific energy consumption (and therefore also GHG-emissions) of the plastics industry. For the chemical industry as a whole, Eurostat reports final energy consumption for energy use to amount to about 2.2 Mill. Terajoule (TJ) for the EU28 in 2017. In comparison, the same indicator for industries as a whole amounted to 10.9 Mill. TJ, implying a share of about 20 percent. Both this share and the absolute level of final energy consumption in the chemical industry have roughly stayed constant over the last years. Given the industry's dynamic growth (see Section 2), this implies considerable improvements in energy efficiency. In parts, this reflects technological process in injection molding. However, in terms of energy composition, the current outlook indicates a lot of room for improvement. Figure 6 splits total consumption up into distinct types of energy. It shows that externally provided electricity and heat together only made up about 43 % of total energy consumption. The remainder consisted of sector-internal energy conversion, with natural gas representing the dominant energy carrier. By contrast, direct use of renewables (i.e. not including the renewables used in the production of the externally provided electricity and heat) merely represented a share of 0.6

% (Eurostat, 2019). This demonstrates the sector's current reliance on GHG-intensive energy conversion.

Figure 6: Final energy consumption for energy use in the chemical industry by type of energy; EU 28 in 2017



Source: Eurostat (2019).

4.1.4 | End-of-life option: landfills

At the global level, storing plastic waste in landfills is the by far most common organized treatment option, disregarding pure littering (PlasticsEurope, 2018). Characteristics and management quality of these landfills differ significantly between countries, ranging from strictly regulated high-technology facilities to completely uncontrolled dumps. Of course, this also shapes the environmental impact of waste storage. An immediate restraint is the space used for landfilling. Atmospheric risks primarily take the form of greenhouse gase emissions and leachate. GHG emissions result from the (aerobic or anaerobic) decomposition of stored material by microorganisms. However, the fact that plastic takes a very long time to degrade limits the exposure, at least when compared to the storage of biomass (Ishigaki et al., 2004). Leachate comprises any liquid that drains from landfills, often as a consequence of rainwater entering the landfill. Depending on the type of waste, it can contain a wide range of dissolved substances that potentially threaten local ecosystems, especially when it inferes with the groundwater. Potential ways to improve the environmental balance of landfilling are the implementation of advanced cleaning techniques or the integration of facilities for energy recovery (see next subsection) (Zhou et al., 2014). Nowadays, technological development allows for the operation of sustainable landfills that combine systems of leachate recirculation with methan extraction for the purposes of power generation or biogas production (Surampalli et al., 2016).

4.1.5 | End-of-life option: waste-to-energy

Burning plastic waste represents another end-of-life option. From an energetic point of view, this can only be an efficient solution if the heat generated during combustion is captured for effective use, i.e. directly used for heating purposes or transformed into electricity. This is the case with many modern incineration plants, which recover both heat and electricity from the process (CHP). Then, a positive environmental effect can result from the replacement of fossil fuels for these purposes. However, the burning of plastic is also coupled with emissions of potentially hazardous substances, especially of dioxin and heavy metals (Shibamoto et al., 2007). The extent to which the emitted materials contaminate the environment is highly sensitive to the scrubber systems implemented. Thus, the extent of governmental regulation is again a key factor here. Incineration also involves the release of significant amounts of CO_2, which can not be captured by current technologies (Chen & Lin, 2010). The energy and material needed for the operation of the incineration plants must be taken into account as well. In the end, the balance strongly depends on the individual conversion efficiency reached in generating usable energy from plastic waste (Astrup et al., 2009a).

In addition to combustion, more sophisticated alternatives of recovering the energy content from used plastic have been devised. Pyrolysis is a technique to chemically de-compose organic material by exposing it to high temperature in the absence of oxygen in a special pyrolysis reactor. Applied to plastics, it can be used to convert plastic waste into the components pyrolysis oil, carbon black and hydrocarbon gas. Pyrolysis oil does not feature the same chemical properties as petroleum and is therefore not suitable for all of its applications. But quality can be increased by means of processes that reduce the oxygen content of pyrolysis oil, which is termed upgrading (Scheirs & Kaminsky, 2006). Of course, also the pyrolysis process causes emissions of greenhouse gases and several pollutants. Existing research suggests that the overall environmental performance compared to combustion is highly sensitive to the specific materials and techniques investigated (Conesa et al., 2009). Another alternative is gasification. It is an umbrella term for several chemical processes that transform plastic waste into gaseous products. The typical aim is to produce a synthetic gas that is ready to substitute natu-ral gas in its role as an energy carrier (Lopez et al., 2018). Again, an environmental as-sessment necessarily relies on the specific setup and filtering technology. In principle, however, researchers argue that the gaseous nature of the output facilitates emission control compared to processes with solid outputs (Kamińska-Pietrzak, 2013).

4.1.6 | End-of-life option: recycling

Plastic recycling is the process of recovering plastic waste and reprocessing the materials to usable products that can enter again the markets. Due to the enormeous diversi-ty of plastic materials and applications, recycling processes are technically complex and highly product-specific. With current technologies, some plastic materials can not be recycled at all. More generally, recycling involves a series of (resource-consuming) preparatory steps such as collection, sorting, washing, downsizing and separating (by their physical and chemical properties). The actual recycling consists of compounding the small particles to plastic pellets, which in turn are used to produce new plastic products (Subramanian, 2000). In comparison to alternative end-of-life options, recy-cling suggests an immediate savings potential with respect to space and air pollution. However, a consequential reduction in the environmental footprint has to be counter-weighed against the resource use and additional emissions associated with the recy-cling process. The related literature indicates that recycling can under some circum-stances be more resource-intensive in terms of energy, labor and machinery than land-filling or incineration (Kinnaman et al., 2014). In parts, this is due to the large hetero-geneity of plastic types, turning sorting and processing into very complicated tasks. It is also associated with the emission of Volatile Organic Compounds (VOC) (He et al., 2015). However, for a real lifecycle evaluation, it is not sufficient to compare recycling simply to alternative end-of-life treatments. The potential benefits caused by replacing plastic production from scratch need to be taken into account as well.

Therefore, a crucial factor is the capability of substitution. The idea to substitute plastics for other sorts of materials like wood is typically not assessed to have beneficial environmental consequences (Astrup et al., 2009b). The view on plastic recycling is usually very positive when it is compared to the production of virgin plastic. Of course, such a comparison makes only sense if recycled plastic can actually replace virgin plastics without substantial loss of usability in a wide range of applications. In this context, degradation within the recycling process ("downcycling") is a serious technical issue. Recycled products might turn out to be non-recyclable themselves, or of lower quality compared to the original product (e.g. synthetic clothes recycled to bottles). In addition, there are still practical obstacles to the implementation of large-scale recycling systems, as demonstrated by the currently low recycling rates in Europe (see section 2). This concerns a lack of demand for the recycled products, especially when they are an outcome of downcycling. It also concerns a partially underde-veloped recycling infrastructure, impeding improvements in quality and capacity of recycling facilities (Miller et al., 2014). As a consequence, current benefits of recycling have more to be seen in a delay of alternative end-use options (and their environmen-tal consequences) than in a persistent avoidance of such activities.

4.2 | Overall assessment

The ELCA literature has produced a vast bulk of comparative research on the properties of different kinds of plastic materials. For reasons of space and readability, we will not dive into the particularities of certain petrochemical polymers, but present some selected results highlighting the fundamental implications certain life cycle options.

Some studies allow for a comparison of the environmental performance of different process stages. Dormer et al. (2013) undertake a life cycle analysis for recyclable PET trays. They come to the conclusion that the stages resource extraction and manufacturing offer the biggest potential for a reduction in the overall carbon footprint, mostly due to the fact that they are the major contributors to GHG emissions. By contrast, end-of-life treatment, packaging and transport contribute only to a minor degree. Kang et al. (2017) specifically point to manufacturing as exerting the biggest global warming potential. Siracusa et al. (2014) assess the environmental impact of plastic bags. They identify the production of the granule not only as the most damaging process stage in terms of greenhouse gas emissions, but also in terms of non-renewable energy consumption and the emission of particulate matter. They also investigate the effect of a reduction in the thickness of the film layer, finding that it can contribute to a significant decline in the CO_2-emissions related to the production of the plastic resin. In parts, investigations concern the use of alternative plastic materials for similar purposes. Lewis et al. (2010), in an examination of the literature on LCA's for reusable plastic bags, arrive at the result that reusable plastic bags are only less emission-intensive in terms of GHG than non-reusable bags if the frequency of reusage reaches a significant level.

Another often investigated question is the environmental implication of different end-of-life treatments for plastic waste. In an analysis for non-recyclable plastic in municipal solid waste, Eriksson & Finnveden (2009) find that landfills often represent a better treatment option compared to incineration with energy recovery with respect to GHG emissions. This is only different if incineration plants are very efficient in terms of energy conversion and electricity-to-heat ratios are high. In studies comparing recycling with landfills, evidence is likewise mixed. In a simulation model aiming to optimize waste management in South Korea, Song et al. (1999) determine a CO_2-minimizing recycling rate of about 80 % for PET bottles, with the remaining bottles being sent to landfills. For plastic containers, Perugini et al. (2005) find that recycling scenarios are associated with a significant decrease of GHG emissions up to 80 % compared to nonrecycling scenarios. In a comparison of all three options for PET bottles, Simon et al. (2016) also identify recycling as the best option in terms of GHG emissions, while no significant differences between incineration and landfilling were detected. In their analysis of plastic waste management policy for Sweden, Milios et al. (2018) conclude that a combination of high target rates for recycling and a ban on plastic incineration represents the most sustainable policy strategy in terms of global warming.

There are also studies comparing the impact of different recycling technologies, especially a comparison between mechanical and feedstock recycling through pyrolysis (see last section). Based on a literature survey, Lazarevic et al. (2010) conclude that mechanical recycling is under most circumstances the environmentally preferable option, also with respect to Global Warming Potential. Perugini et al. (2005), in comparing mechanical recycling with both feedstock recycling and depolymerization (also termed "chemical recycling"), also find that mechanical recycling represents the smallest threat to climate when measured in CO_2-equivalents. However, as already mentioned above, several authors stress that the evaluation strongly hinges on the question to what degree recycled material is actually able to replace virgin plastic. By explicitly integrating the performance of recycled materials into an LCA, Rajendan et al. (2013) try to gain more insights into this condition. They find that mechanical recycling is only preferable over feedstock recycling if the recycled material is able to substitute at least 70 – 80 % of virgin plastics. Rigamonti et al. (2014) compare five different plastic waste management strategies mostly differing in terms of the sorting techniques. It turns out that a scenario with source separation of all plastic not only maximizes collection efficiency, but also minimizes GHG emissions compared to limited source separation or no separation at all.

As mentioned above, one factor worsening the environmental balance of recycling is that recycled products themselves are in parts non-recyclable. Against this background, Toniolo et al. (2013) go one step further and compare the life cycle properties of two trays made of different forms of recycled plastic. One of them is made of nonrecyclable multilayer film and the other of an innovative recyclable mono-material PET film. Even when accounting for the additional process steps, it turns out that the mono-material is performing better in terms of climate change impact under all scenarios investigated. This matches other results in the literature (Nessi et al., 2012), indicating that treating recycled materials with specific additives that ensure their recyclability is an environmentally sound approach.

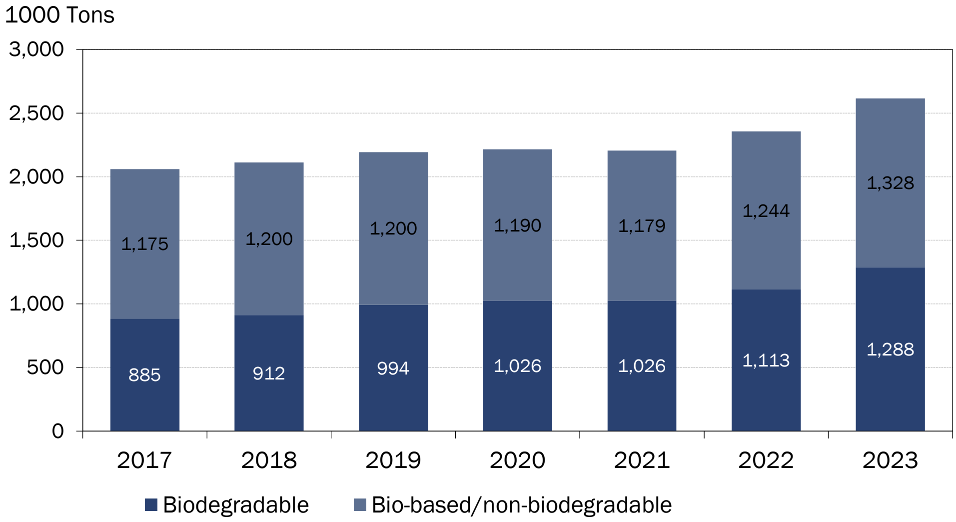
5 | Plastics from biological sources: a sustainable alternative?

Given the severe environmental impacts of conventional plastics, attempts have been made long ago to replace fossil resources as feedstock by less damaging sources. Those artificial polymers which are generated from biological resources and serve to replace are summarized under the term bioplastics. The first invention of a bioplastic product was celluloid, dating back to the 19th century. Initially, most bioplastic material was designed for use in packaging purposes. By now, the term refers to a wide range of different chemical compounds, covering a multitude of potential applications. Among those, according to estimates of the industry association European Bioplastics, flexible packaging is the by far most important one, with a share of about 44 % in 2017. Further noteworthy areas are rigid packaging, consumer goods and textiles (European Bioplastics, 2017).

Candidates for primary resources, which enter the production of bioplastic granules, are carbohydrates (e.g. starch, cellulose), proteins (e.g. gelatin, casein) and lipids (plant oils, animal fats) (Song et al., 2009). According to Chanprateep (2010), the granules can be classified into three groups based on their origin: polymers existing in nature, polymers produced through polymeriziation of natural monomers, and those gained from a combination of monomers from renewable resources with petrochemical-derived monomers (hybrid solutions). Hence, the meaning of the term bioplastic is somewhat blurred by the fact that the origin material can be of more or less biological nature. Concerning industry output, recently popular forms of granules are thermoplastic starchs, Polylactides (PLA) and Polyhydroxyalkanoates (PHA). PHAs, polyesters produced by microorganisms through fermentation processes, are viewed as particularly promising: they are purely bio-based, tend to biodegrade rapidly and can be structured for a multitude of uses (Koller et al., 2017).

Recent estimates predict significant growth for global capacities of bioplastic production in upcoming years, mostly driven by capacity increases for PLA and PHA (see Fig. 7). A general change in consumer attitudes towards a stronger focus on sustainability issues is seen as an important driving force for this development (European Bioplastics, 2018). Nevertheless, bioplastics currently make up merely 1 % of the global plastic markets. A central problem especially of the most sustainable bioplastic materials is still the high production costs (Chanprateep, 2010). This is not merely due to the different technologies used, but also due to the economies of scale exploited by the incumbent petrochemical firms. This renders it for bioplastic firms even harder to cover their consizeable sunk costs related to R&D and capital investment (Iles & Martin, 2013).

Figure 2: Global production capacities in bioplastic production



Source: European Bioplastics (2018). From 2019 onwards: forecasts

An obvious environmental advantage of biomass use is the savings of scarce fossil resources, including the greenhouse gas emissions involved with their extraction and processing. Moreoever, they tend to biodegrade much faster and do not release toxic substances during this process, reducing the dangers associated with plastic littering significantly. They also create alternative end-of-life options: used bioplastic products can enter the production of biogas and/or be utilized for composting.

On the downside, the production of the biomass itself involves the use of limited resources like land and freshwater. The fertilizers used in agricultural production can also contribute to the phenomena of acidification and eutrophication. Acidification refers to the damages done to ecosystems by the deposition of acidic substances in soil and water. These substances can i.a. be created through chemical reactions of the nitrogen contained in fertilizers (Alewell et al., 2000). Eutrophication describes an excessive enrichment of lakes with nutrients, leading to uncontrolled growth of algae and a deterioriation of water quality. This can result from situations in which agricultural soil is not able to assimilate the complete amount of nutrients included in fertilizers. In this case, the nutrients can be transported by rainfalls into groundwater or rivers flowing into lakes (Harper, 1992). And finally, not all bioplastic materials are biodegradable within a sensible amount of time. As for other materials, the degree of biodegradability depends on the specific environmental conditions. In a review of the related literature, Emadian et al. (2017) find that common types of bioplastics are analyzed to be highly degradable in soil and compost environments. However, in a study for carrier bags made from bioplastic, Accinelli et al. (2012) observe that decomposition takes place considerably slower when bags are submerged to water. Given the large amount of plastic that finds its way into oceans by direct disposal or through wastewater treatment, this is a worrying outcome.

The existing variety of origin materials implies a similar variety in production technologies, making a general assessement of their environmental qualities impossible. Therefore, just as in case of conventional plastics, a Life Cycle Assessment has to be technology-specific. As pointed out by Heimersson et al. (2014), undertaking an LCA for bioplastic products can be particularly challenging due a lack of sufficient data on technologies and the necessity to incorporate areas that are usually neglected (e.g. water use). Moreover, Philp et al. (2013) argue that a comparison of innovative bioplastic products with fossil-based plastics in an LCA can be unfair in a time perspective, because production of fossil-based plastics had a long time to optimize, while existing optimization potential for recent bioplastics is still partly unexploited.

Against this background, Spierling et al. (2018) restrict their attention to a comparison in terms of global warming potential. Based on assumptions for technical substitution potentials of existing plastic types, they estimate that a replacement of two-thirds of global conventional plastic production by bioplastics would yield an annual saving of 241 to 316 Mio. t of CO_2-eq.. However, given the omission of the more critical areas from the analysis, it is left unclear whether this implies an overall benefit or simply a shift of the environmental burden to other influences. Studies considering multiple impact categories tend to come to more mixed results. Belboom & Leonard (2016) compare fossil-based and bio-based high-density polyethylene (HDPE), where the bio-based version is produced with wheat and sugar beet. They show that the bio-based solution surely performs better in terms of GHG emissions and fossil resource depletion, but worse in the remaining categories acidification, eutrophication and also particulate matter formation. In an impact assessment of different types of starch plastics, Broeren et al. (2017) basically arrive at the same results, but additionally identify a reduction in non-renewable energy use compared to conventional plastics. Hottle et al. (2016) find in their review article that the existence of such trade-offs is a common characteristic of the LCA literature on bioplastics.

Furthermore, the literature points at the importance of reflecting on the origin of the biological feedstock. In a Life Cycle Analysis for bio-based PET bottles, Chen et al. (2016) consider ecotoxicity and smog formation as additional impact categories. For both categories, they document that the question whether the biomass used as feedstock is gained through additional agricultural production or as a forest residue is of crucial relevance. In the first case, the use of fertilizers and pesticitides raises ecotoxicity, while at the same time the smog impact is worsened by emissions resulting from the fuel combustion of agricultural machinery.

In addition, there are studies that focus on comparisons of end-of-life options for bioplastic products. One challenge with regard to end-of-life treatment of bioplastic is that existing technologies for mechanical recycling of conventional plastics tend not to be suitable (Davis & Song, 2006). Recyclability can be facilited by means of additives like natural fibre, but this can worsen the economic performance. Chemical recycling techniques are also only applicable to some bioplastics and endanger the overall environmental benefit with their high energy demands (Soroudi & Jakubowicz, 2013). For incineration, the environmental evaluation is found to be strongly sensitive to the efficiency of energy recovery (Hermann et al., 2011).

Potentials for composting is another area that has raised attention in the literature. Davis & Song (2006) argue that in the field of packaging bioplastic is most appropriate for single use packages, because it can be combined with local composting as a convenient way to recycle the product. An essential advantage of composting compared to storage on landfills is that biodegradation takes place under aerobic conditions. It therefore involves the release of CO_2 instead of (in terms of its contribution to global warming more dangerous) methane. From a life cycle perspective, this CO_2-release can not be considered a net emission, as it is anyway part of the organic carbon cycle (i.e. a re-release of the CO_2 absorbed during the production of the biomass). Further benefits of such a local solution include reductions in transport costs and related emissions (Davis & Song, 2006).

However, not all biodegradable plastics are compostable. Compostability requires that the substances biodegrade under the conditions of a composite site with rates similar to known material. A further restriction is the existence of local industrial composite sites, as research shows that the alternative of home composting at the consumer level delivers significantly less satisfying results (Song et al., 2009). Moreover, for a real comparison with recycling, the fact that composting implies the necessity to replace the composted product by a newly produced one should also be taken into account in an LCA. In a comparison of end-of-life options for PLA-type bioplastic, Cosate de Andrade et al. (2016) show that under these circumstances composting performs worse than both mechanical and chemical recycling in terms of impacts on climate change, human toxicity as well as fossil resource depletion. However, as pointed out by Yates & Barlow (2009), LCA studies do not consider potential practical problems in conjunction with a recycling of bio-based polymers: routines of sorting and cleaning have to be adapted to the different material properties, and the occurrence of rapid biodegradation might render the recycling process altogether impossible.

Hence, the LCA literature highlights the tremendous opportunities associated with an expansion of the bioplastic industry, but also adds a word of caution regarding potential undesired environmental side effects. Against this backdrop, data reliability and transparency of the underlying assumptions are even more crucial requirements for an appropriate LCA than usual. This not only concerns the evaluation of bio-based versus petrochemical plastics, but also a relative comparison of the different types of bioplastics with their heterogeneous properties and production methods. The fact that the industry is still in its infant stage contributes to the difficulty of the challenge. For many forms of environmental impacts, no widely accepted rule-of-thumbs on their quantitative dimensions yet exist. Furthermore, available plant-level data from pilot phases etc. is of only limited value for the assessment of future environmental consequences, as technologies have not exploited their optimization potential yet (Narodoslawsky et al., 2015).

6 | Conclusion

This paper has presented insights from the Environmental Life Cycle Assessment method on ecological side effects of plastic products. It became apparent that conventional petrochemical plastics exert adverse environmental effects in all stages of their complex production chains as well as related to end-of-life treatment. Improvements with respect to energy efficiency and end-use options can help to reduce emission intensity, but do not address the fundamental long-term isues related to by the slow degradation of petrochemical polymers. Therefore, searching for alternative feestocks that improve the biodegradability of plastic products seems a natural approach.

By now, a still small, but dynamically growing industry has emerged that produces plastics from biomass. Even though the specific properties of these bioplastic products differ to some extent, depending on the feedstock used, they share the advantage that no scarce fossil resources are used up in production. Many (but not all) bioplastics degrade rapidly in natural environments. Moreover, in life cycle perspective, emissions of greenhouse gases tend to be of a lower magnitude. At the same time, however, the literature points at a trade-off with respect to other impact categories: the use of biomass causes higher potentials of acidification, eutrophication and soil degradation in the context of agricultural production. Therefore, the use of by-products should be favored compared to the cultivation for pure bioplastic purposes. Recyclability is also a challenge that has to be addressed to further improve the environmental balance. From an economic perspective, that fact that bioplastic granulates are still substantially more costly than their petrochemical counterparts, while often not reaching the same quality, remains a serious barrier to market penetration. Nevertheless, continued upscaling efforts can, by fostering technological progress and exploiting economies of scale, principally overcome these obstacles. This is not an unlikely development. Given both the undiminished growth of global plastic consumption and the increasing consciuousness for the dimension of the plastic debris, it can be expected that conditions will become more and more favorable for bioplastics to prosper.

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