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Life Cycle Environmental Impacts of Plastics: A Review

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Abstract

Global production and consumption of plastics and its deposition in the environment are growing rapidly, while their life-cycle impacts to the environment are yet to be fully understood. This study reviews the existing literature with the goal of evaluating the state of current data and knowledge on plastics' life-cycle environmental impacts. We identified 98 peer-reviewed journal papers, 25 reports, 8 databases as well as 21 existing reviews on the environmental impacts of plastics. Our review shows that life-cycle approaches have been instrumental in gaining new insights on the environmental implications of plastics. Global life-cycle greenhouse gas (GHG) emissions from plastics, for example, is estimated to be around 1.7 Gt CO_2e yr^{-1}, which is substantially larger than the GHG emissions from global aviation. LCAs have also been widely employed in various comparative studies such as biomass v.s. petroleum-based plastics, single-use plastics v.s. multi-use alternatives, which often result in counterintuitive outcomes. However, our analysis also indicates that there still is a paucity of reliable data and tools for conducting LCAs of plastics. In particular, we find that Asia, the major plastic-producing region, is scarcely covered by existing literature and databases. In addition, the life-cycle impacts of plastic additives, and the degradation pathways and associated environmental, human and ecological impacts of plastics are poorly understood, limiting our ability to gauge the life-cycle impacts of plastics. Furthermore, we find that a consensus on the allocation of environmental impacts for the recycling and reuse of plastics is yet to be reached. We identify four major areas of future research including: (1) developing the method for reliable estimate of the amount of plastics entering the environment, (2) understanding plastics' environmental degradation pathways, degradation byproducts, and their human and ecological impacts, (3) expanding the coverage of life-cycle inventory data across geographies and additives, and (4) building consensus on key methodological issues including those around allocation.

Key words

Plastics; life cycle assessment; human and ecological impacts; environmental degradation; additives.

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1. Introduction

The word ‘plastics’ is a colloquial term for a wide range of synthetic or semi-synthetic polymer materials that are used in a growing range of applications[1]. While the first man-made plastic was created in 1862, it was not until the 1940s when industrial production of plastics became possible through the use of fossil fuels as feedstock [1]. Soon after, the growth of the global plastics market took off in the 1950s [2].

Global plastics market is currently valued at around USD 570 billion and is expected to grow at an annual rate of 3.2 %. It is estimated that 6,300 Mt of plastics waste has been generated since the 1960s, around 9 % of which has accumulated in natural environments or landfilled [3]. If the current production trend continues, roughly 12,000 Mt of plastics waste will be landfilled by 2050 [3].

Plastics serve as a key material in our daily lives. Though plastics have brought many benefits to human wellbeing and materials welfare thanks to their durability, versatility, and low cost, the production, consumption, and end of life management of plastics have raised environmental concerns ranging from persistence in the environment, marine debris, human health risks, greenhouse gas emissions, among others [4-9].

One of the approaches employed by various studies to measure the environmental and human health impacts of plastics is Life Cycle Assessment (LCA) [10-13]. LCA is a science-based approach that quantifies the environmental impacts of products across their life cycle including extraction of raw materials, manufacturing, transportation, consumption, and disposal [14-16]. However, our ability to understand the life-cycle environmental impacts of plastics hinges upon the availability and quality life cycle inventory (LCI) data and impact assessment methods. This study aims to review existing databases, literature, and recent activities related to LCAs of plastics with the goal to evaluate whether the body of knowledge and data accumulated in the literature and existing databases provide a sufficient basis for understanding plastics' life-cycle environmental impacts. We will also identify the key challenges and gaps in current studies, standards, and tools. Finally, we will discuss the currently on-going initiatives that are aiming to fill known research gaps.

2. Review of Plastics LCA

2.1. Method

Relevant research for this review was identified through searching the literature in sources including Google Scholar, Web of Science, and Google. We used a variety of keyword combinations to develop a comprehensive list of relevant studies. We used "plastics" and "polymer" in combination with "LCA," "life cycle," "life-cycle" and "life cycle assessment" to capture the LCAs on plastics. In addition, we used specific application types, material characteristics, and life-cycle stages to identify the relevant literature that specifically addresses them, including: "recycling," "packaging," "biodegradable," "bio-based," "bioplastics," "agriculture," "plasticizers," "automotive," "building," "construction," "colorants," "resin," "fillers," "nanoplastics," "microplastics," "lightweight," "single-use," "fiber reinforced," "impacts," "database," "materials," "climate," and "reusable." In addition to keywords, searches also specified the regional interest areas including Europe, the United States, and Asia. The searched literature was considered relevant if the principles and research it contained discussed at least one environmental impact or aspect of plastics. The search focused only on the papers published in or after 1990, but we also included highly cited articles published prior to 1990, if relevant to this review. Databases and reports that were determined relevant but not covered by the keyword searches were manually added to the list. Resulting list of data sources includes 98 peer-reviewed journal papers, 25 reports, 8 databases as well as 21 reviews. See Supporting Information for a complete list of data sources used in this review.

After finalizing the list, we analyzed each item for its objectives, main findings, and key discussion points. The main topics covered by the list of sources collected include: environmental characteristics of plastics (46 references), life cycle assessment methods (5 references), databases (3 references), bio-based and biodegradable plastics (16 references), carbon dioxide-derived plastics (4 references), packaging (15 references), agriculture (11 references), building and construction (11 references), automobiles (13 references), consumer goods (8 references), electronics (2 references), functional additives (12 references), and fillers and reinforcements (6 references).

In the next section, we will provide an overview of plastics LCA, data sources, common material uses, and impacts to be considered. More details on terms and definitions discussed in the report are in the Supporting Information. We will first review existing LCA databases and their coverages of plastics. Following that, we will review the LCAs on plastics by their feedstock categories and then by application types. Given that the plastics that use various feedstock types are applied in various sectors, the feedstock-based and application-based reviews are two different angles to look at the same pool of LCA studies. We reviewed plastics by feedstock type, highlighted bio-based and captured CO_2-based plastics in section 3.2, while the following section, 3.3, covers all feedstock types across various applications.

2.2. LCA Databases

The literature often distinguishes two primary approaches to life cycle inventory (LCI): processed-based and economic input-output (IO) approaches [17-20]. Processed-based LCA focuses on the material flows and environmental impacts of every life-stage in the life cycle, which would generate results with high precision and granularity. However, process-based LCA requires detailed data for every life-stage, thus requiring significant efforts on data collection, the absence of which often leads to incompleteness or “truncation error” [18]. IO-LCA is an approach based on monetary transactions or economic input-output data to trace along the supply chain, which covers a wider spectrum of sectors but can only provide sectoral or product group averages at a lower resolution. Table 1 provides a summary of the most widely used databases for plastics-related LCA modeling from both process-based and IO-based LCA databases.

Altogether, these databases provide LCIs of several hundred plastics materials with varying temporal and spatial system boundaries and level of aggregation. These LCI databases can be applied to various life cycle impact assessment (LCIA) methods that quantify plastics impacts on climate change, ozone layer depletion, acidification, and various human and ecotoxicological impacts.

Overall, Europe is better represented by these databases than other continents: 4 of the 8 databases listed have data drawn primarily from European industry and the release year of these databases ranges from 2003 to 2017, while some of the underlying data may be older. These databases also differ in the feedstock types and application categories they cover. USEEIO, CEDA, and the GREET model cover plastics product types including bags, bottles, foams, pipes, laminated products, and others. On the other hand, Plastics Europe and the US LCI database cover specific resin and monomer types including PET, HDPE, PVC, LDPE, PP, PS, and others. However, these databases often have limited coverage in plastics additives and bio-based plastics.

Table 1. Available LCA databases sources, description, and references.

Source of Data	Description	Plastics Categories Included	Ref.	Additional Information
US LCI database	National Renewable Energy Laboratory (NREL) created the US Life Cycle Inventory Database to provide gate-to-gate, cradle-to-gate, and cradle-to-grave accounting for plastics environmental impacts	Process-based resins and polymers including PET, HDPE, LLDPE, LDPE, PLA, PP, EPS, PVC, biodegradable fill, open molding, and others	[21]	Main datasets were collected between 2003 and 2005. Fuel production data for the U.S. based on Department of Energy national statistics and data. The national average U.S. electricity grid (from the U.S. LCI Database) was used.
Ecoinvent	Life cycle inventory database encompassing around 18,000 processes including energy supply, agriculture, transport, chemicals, construction, and waste treatment.	Process-based resins and polymer markets including electronics, agriculture, waste treatment, extrusion, and industry	[22]	Underlying data were drawn mainly from European sources.
Plastics Europe	Association of plastics manufacturers in the European plastics industry.	Process-based resins and polymers including LDPE, LLDPE, HDPE, PP, PVC, VCM, PTA, PET, and others	[23]	Most of the datasets can be found as such in their aggregated form in Simapro industrial database, and Gabi database (which avoid conversion and format problems)
Gabi Plastics Extension Database	Gabi provides life-cycle inventory data on mass plastics (e.g., PE with various densities, PP, PS), vinyl polymers (e.g., PVC, PVAL), technical plastics (e.g., ABS, PMMA, PTFE), polyamide (e.g., PA 6, PA 6.6, PA 6.12), special plastics (e.g., PPS, PEEK, SMA)	Process-based polymers including vinyl polymers, technical plastics, polyamide, and special plastics	[24]	
US Environmentally-Extended Input-Output (USEEIO) database	The Environmental Protection Agency (EPA) provides an LCA database on US products and services. This database includes datasets for multiple plastics products (e.g., plastic bottles).	Plastics resins and polymers as well as plastics products including plastic bottles, foam products, bags, pipes, and others	[25]	USEEIO uses an environmental-extended input-output (EEIO) LCA model.
Comprehensive Environmental Data Archive (CEDA)	CEDA is an extensively peer-reviewed suite of environmentally extended input-output databases first launched in 2000. These are designed to assist various environmental systems analyses including life cycle assessments (LCA), carbon, energy, water, waste, and toxic impact assessment throughout the supply chain.	Plastics products including plastics and rubber industry machinery manufacturing, plastics and resin materials manufacturing, bottles, urethane and other foam products, rubber and plastics hoses, and others	[26]	CEDA uses an environmental-extended input-output (EEIO) LCA model.
CarbonMinds	CarbonMinds has developed a CM.CHEMICAL database from a regionalized model of the global chemical industry, built from plant-level. The database provides life cycle inventory datasets for national averages of consumption, production and per technology.	Process-based, supplier- and country-specific as well as market average LCIs of feedstock, resin, intermediate and final petrochemical materials and products	[27]	CM.Chemical uses sophisticated process simulations and market intelligence data.
Greenhouse gases, Regulated Emissions, and Energy use in Technologies (GREET) Model	GREET model is developed by Argonne National Laboratory to address life-cycle emissions of various fuel-vehicle combinations. Since plastic is widely used in automobile manufacturing, this model includes LCA data for plastic resins and products.	Process-based resins and polymers including ethylene propylene diene monomer (EPDM) resin, flexible polyurethane foam, compression molding, and average transformed plastic products	[28]	LCA data for the majority of the processes in this model is extracted from the aforementioned NREL and Plastics Europe databases.

2.3. LCAs on Plastics by Feedstock Types

Currently there are three types of feedstock categories used for producing plastics commercially: (1) petroleum, (2) biomass, and (3) captured CO_2. The majority of the plastics produced and consumed today are petroleum-based, with only slightly above 1 % of global plastics produced in 2019 being bio-based [29]. From the following section, we will review LCAs of plastics that use bio-based and captured CO_2 against petroleum-based plastics.

2.3.1. Petroleum-based Plastics

A majority of plastics are made of petroleum-based polymers. The Society of the Plastics Industry in the UK and the Plastics Industry Association in the US classify them into seven categories: PET, HDPE, PVC, LDPE, PP, PS, and others [30, 31]. Petroleum-based plastics, also known as conventional plastics, are the most common form of plastic with production on the order of 350 Mt annually and are responsible for 4 % to 8 % of the annual global oil consumption [32, 33]. Petroleum-based plastics have replaced other materials due to its lightweight, corrosion resistance, and ability to process in lower temperatures. As demand for plastics continues to increase, concerns for polymers to leak from waste streams and not degrade has increased [33]. The slow degradation of petroleum-based plastics characteristics is favorable for transportation, storage, and use, but not for disposal [5, 33].

Not surprisingly, given the volume of production, the life-cycle environmental impacts of petroleum-based plastics are reported to be substantial. A recent study on life-cycle GHG emissions from plastics, for example, showed that conventional plastics generate 1.7 Gt $CO_2e\ yr^{-1}$ over their life-cycle [9]. This level of emission rate is substantially higher than the direct annual GHG emissions from global aviation.

2.3.2. Bio-based Plastics

Bio-based polymers can be generally grouped into two types, (1) polymers synthesized by living organisms (natural or bio-engineered) and (2) polymers obtained from bio-based monomers [34]. An example of the first type is poly(hydroxyalkanoate) (PHAs), and an example for the second type is polylactic acid (PLA) [34]. These two polymers are the two most studied bio-based and biodegradable plastics materials [35]. Both materials are produced from renewable sources, such as corn (maize) and sugarcane [35]. Their mechanical properties are comparable to other commodity plastics, and both materials are found to have a simpler and faster degradation mechanism due to their renewable feedstock [35]. However, some bioplastics' properties are different, and as such, their applications and sector focus differs from conventional plastics. The most common replacement of conventional plastics occurs in packaging, consumer goods, and textiles (Figure 1).

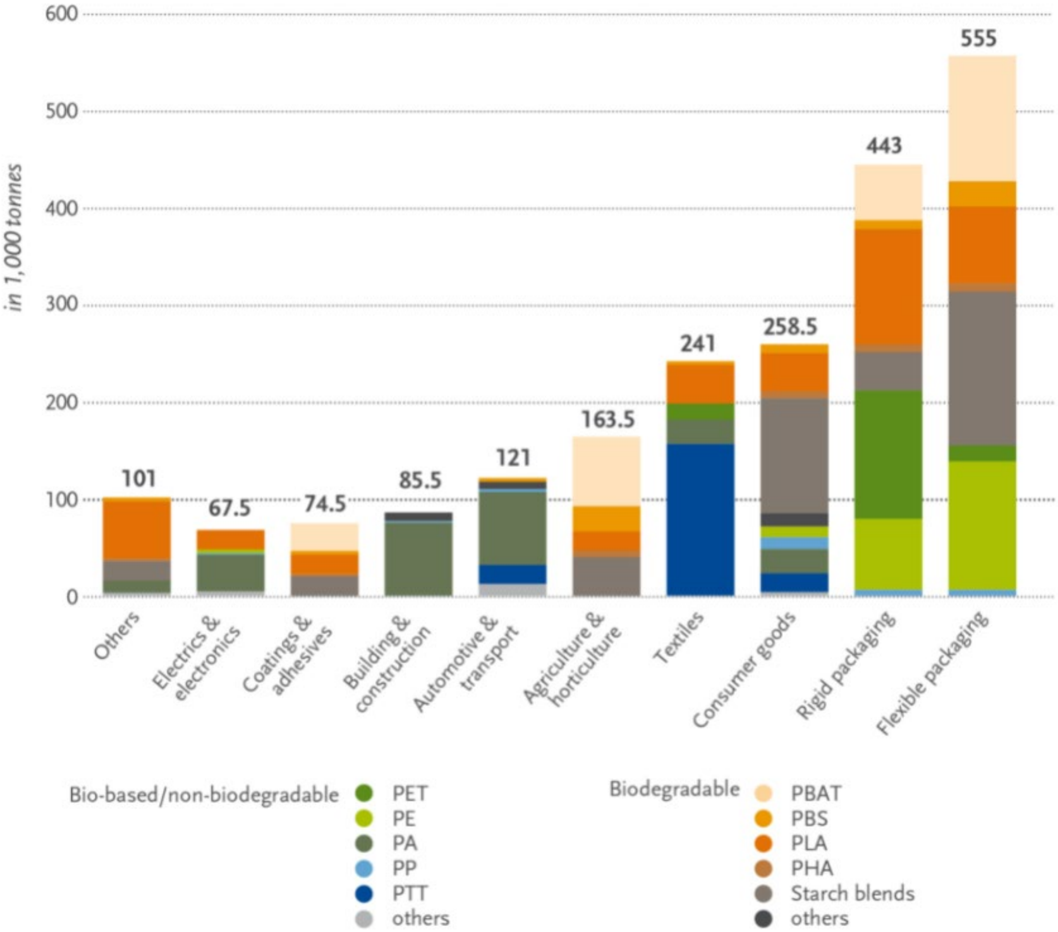

Fig. 1. Breakdown of bioplastics production capacities and applications [29].

However, it is worth noting that not all bio-based polymers are biodegradable [34]. For example, bio-based PE, PP, PET, and poly(trimethylene terephthalate) (PTT) can be fully or partially manufactured from bio-based feedstock, but are not biodegradable [34]. Furthermore, some petroleum based polymers, such as poly(ε-caprolactone) (PCL), polybutylene adipate terephthalate (PBAT), and poly(butylene succinate) (PBS), are biodegradable [36]. Sometimes, bio-based but non-biodegradable polymers and petroleum-based but biodegradable polymers are collectively (and confusingly) referred to as 'biopolymer' or 'bioplastic' (Figure 2) [36].

Bio-based plastics may, though not always, degrade faster, while degradation rates depend on the environmental conditions [37]. For example, one of the most common bio-based plastics, polylactic acid (PLA)-based plastics, are biodegradable in soils much faster than conventional plastics, but their half-lives are similar to that of petroleum-based polyethylene in the marine environment [5, 38]. Nonetheless, the concerns around the dependence on fossil fuels and persistence of plastics in the environment propelled the demand for bio-based and biodegradable plastics over the last few decades, reaching 3.8Mt, or about 1 % of the petrochemical polymers, in 2019 [39].

Waste management options of bio-based plastics differ from those of petrochemical plastics and often cannot be recycled alongside each other. However, bioplastics can be composted alongside conventional waste streams, such as landfills, incineration, recycling, and anaerobic digestion [37]. Some studies suggest that anaerobic digestion is the preferred degradation method, while others favor incineration [40]. Studies have pointed out that clear standardization of labeling and sorting systems is needed to ensure optimal end of life treatment of biodegrade plastics, which affect LCA results [41].

While bio-based plastics can be recycled along with conventional wastes, separating bio-based plastics from conventional plastic waste is a better option; it can improve the composting rate of biodegradable waste and the energy recovery rate for non-recyclable bioplastics [42]. Although fossil fuel-based plastics currently have a price advantage over renewable ones, accounting for the external costs and future optimization is expected to bring the price of fossil fuel-based polymers 44 % higher than those of bio-based polymers [34]. However, a direct comparison of environmental impacts between fossil fuel-based and biomass-based polymers based on the literature is still challenging due to the underlying differences among LCA studies. End of life scenarios are, for example, modeled differently in almost every LCA study, making the environmental impacts from waste streams difficult to compare [37]. A study has shown that bioplastic production for replacement of petrochemical plastics can be an effective way of reducing end of life emissions, with mechanical recycling being the optimal method for energy savings [43]. Another study also showed that the use of biomass as a feedstock for plastics can reduce life-cycle GHG emissions from plastics [9].

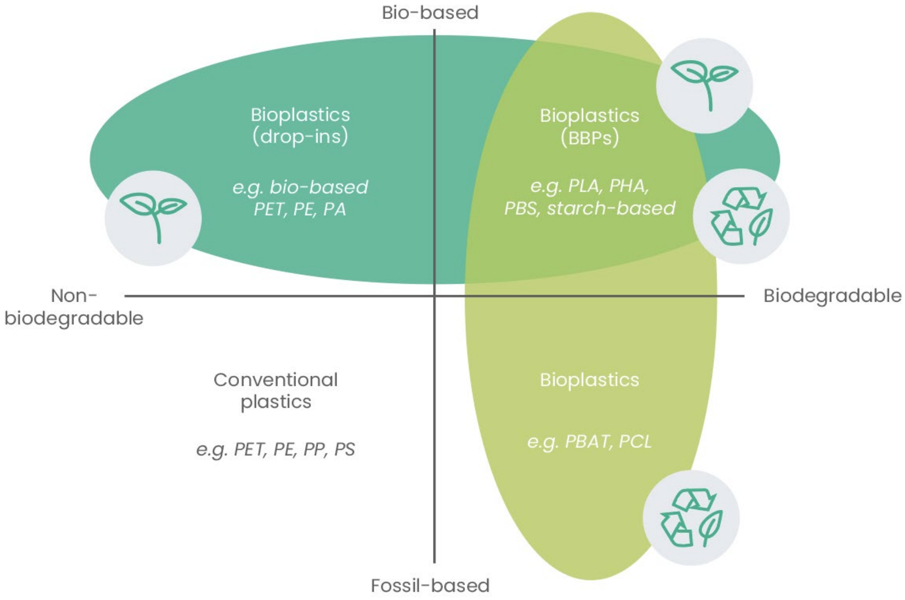

Fig. 2. Common types of bioplastics and their position in biodegradability and feedstock type spectra [44].

The majority of LCAs on biopolymers (e.g., PLA, PHA, and starch-based polymers) focused primarily on their GHG emissions. The results demonstrated several outcomes examining the use and potential reductions of plastics' environmental impacts. Ref. [45]

and Ref. [44] show that using these materials may reduce climate impacts, but may result in other environmental tradeoffs. According to Ref. [35], the process efficiency needs to be further improved to optimize the climate benefits, though such improvement might have taken place since the publication of study. A more recent review by Ref. [46] shows that the shift to biopolymers can potentially reduce 241 to 316 million tonne CO_2e GHG emissions. A similar review demonstrates that many LCA studies on PHA show that energy usage comparing bio-based and fossil-fuel based plastics are comparable, and while climate change impact is typically lower for bioplastics, they often perform worse in other impact categories [47]. In addition to GHG emissions, Ref. [33] and Ref. [48] show that bio-based feedstocks, especially large-scale agricultural products, are usually dependent on fossil-derived pesticides and fertilizers, which create higher impacts in eutrophication and acidification.

In addition to improving process efficiencies, Ref. [35] points out that using energy- and chemical-intensive crops is associated with considerable environmental impacts. Ref. [48] demonstrate that for certain types of feedstocks, there will be a significant amount of energy consumption during the process of converting agricultural products into plastics materials, potentially creating a higher global warming impact. Ref. [48] and Ref. [49] show that land-use change is also an important factor when determining the environmental impacts of biobased plastics, and it depends on the geographical location. According to Ref. [50], land use change and excessive harvesting of biomass cause the release of soil organic carbon to the atmosphere and decreases soil's ability to sequester organic carbon. Extensive land use for bio-based plastics production will make it a competitor with agricultural production, threatening the food value chain. From a study by Ref. [51], it is estimated that a complete replacement of conventional plastics with bio-based one would consume between 30 and 219 million hectares of additional lands, and around 307 to 1652 billion m^3 of water per year.

Overall, the literature generally warns that large-scale production of bio-based plastics may potentially create higher environmental impacts through the emissions from land use and land management. Therefore, it would be crucial to take into account land-based emissions when assessing the life-cycle impacts of biomass feedstock [33, 52].

2.3.3. Captured CO2-based Plastics

As compared to petroleum-based plastics that took off in the mid-20th century, CO_2 capture and utilization for plastics is still an emerging subject, as a majority of the research has occurred in the 21st century [32]. The overall process of separating, capturing, and using CO_2 for the creation of polymers may be done in several ways, and companies are driving innovative processes for each method. Among the most active companies specifically making captured CO_2-based plastics are Covestro, Econic Technologies, and Novomer [32]. While there are these companies making CO_2-based plastics, the processes are relatively new and CO_2-based plastics are still developing on a commercial scale [53].

Among the active companies making CO_2-based plastics, most of the information is contained outside of peer-reviewed literature. Each of these companies use different proprietary catalysts to convert CO_2 into polymers. As most of the information is coming from news outlets or from the companies themselves, it is unclear the full environmental impacts from CO_2-based plastics. Most emphasis on company websites are placed on CO_2 reduction rather than other impact indicators [54, 55]. Given the nascency of the technologies behind CO_2-based plastics, peer reviewed literature that compares the life-cycle environmental impacts of CO_2-based plastics against petroleum-based or biomass-based plastics has yet to be developed.

2.4. LCAs on Plastic Materials and Products

In this section, we reviewed 53 studies from 6 different market segments, including packaging, agriculture, building and construction, automotive, consumer goods, and electronics as well as the plastics industry. These industries have products of different lifetimes, and therefore will have different scopes of impacts associated with them. Product categories include but are not limited to bags, straws, cutlery, mulch films, printers, and nappies. Different types of plastics and alternative materials considered include PET, PP, and PVC. Also, some studies that specifically focus on different plastic types are included in the review. The major takeaways are summarized by the market sector as follows.

2.4.1. Packaging

The packaging sector utilizes the biggest portion of primary plastics produced. In 2015, 42 % of primary non-fiber plastics produced were used by the packaging industry [2]. Also, the product life-times of packaging products are very short, mostly less than a year, making the contribution of packaging in the near-term plastics waste generation even greater; globally, packaging accounted for 54 % of non-fiber plastic waste in 2015 [2].

Plastics are used for many applications in the packaging industry such as food containers, bottles, bags, and cutlery. LCA studies on packaging often compare various plastics and non-plastics materials such as HDPE, PET, PP, PS, PLA, PE, cotton, paper [49, 56-58]. These studies show that the weight of the material, use pattern, and waste management are critical factors when determining environmental impacts [49, 57, 58].

One of the most common uses of plastics is shopping bags. Ref. [58] assess the life-cycle impacts of 7 shopping bags and reported that use pattern, material type and weight were the key determinants of their environmental impacts and being more preferable depends on the environmental impact indicator that is being considered. The research here and in other studies shows that a single-use paper bag has higher environmental impact in all categories except eutrophication when compared to single-use plastic bags due to the weight and paper production processes [58, 59]. In the study, paper bags are assumed to weigh 47 g whereas the alternative single-use products made of HDPE weigh 6-8 g [56, 58]. UNEP's report compares the two studies from Finland and Hong Kong, China and India [60, 61]. The report states that paper bags seem to perform better in the Finnish study mostly due to the Finnish paper production using integrated pulp and paper mills whereas Muthu et al.'s study states a high dependence on fossil fuels in the paper production process [60, 61]. The studies tend to show better life cycle emissions results depending on the efficiency of the production processes.

The impact is doubled when the weight of the material used for a bag is doubled, and if a bag is used twice, the impact is the half of a single-use bag [56]. This makes the assumptions around the use pattern very relevant in the analysis. For example, it was found that if a reusable polypropylene bag is used 52 times instead of 104 as assumed, then its impact is higher than all the other single-use bags except the paper bag [57, 58].

UNEP reviewed the comparative LCAs on shopping bags covering a wide global range including Europe, USA, and Asia [56]. The report concluded that the waste management practices are important in determining the environmental impact and different waste management practices are being applied across geographies [56, 60]. This becomes important when comparing paper bags to plastic bags. Landfilled paper bags result in methane emissions, which have higher global warming potential, whereas plastic bags are relatively inert when landfilled [56, 60]. However, when incinerated, CO_2 emissions from biogenic carbon in paper bags are carbon-neutral since biomass captures CO_2 by photosynthesis while growing, whereas incinerating petroleum-based plastic bags generate net emissions of CO_2 [56, 62]. Composting biodegradable bags decreases the environmental impacts compared to conventional plastics, however, the availability of composting facilities is highly dependent on the region [63].

Another common use of plastics in packaging is bottles, for which PET is the main material in use. UNEP reviewed 7 LCA studies on single-use plastic bottles and alternatives [64]. The alternatives suggested include single-use and reusable glass bottles, single-use aluminum cans, single-use carton laminated packaging systems, reusable steel, and aluminum bottles [64]. Production technology and end-of-life management practices, which vary widely across geographies, were found as the key factors that determine the LCA results [64]. Similar to plastic bags, bottle alternatives behave differently in different impact categories. For example, compared to PET bottles, PLA bottles have lower global warming potential, fossil energy use, and human toxicity, but PET bottles have lower impact in acidification and eutrophication categories [64, 65]. Glass bottles would have to be 3 times lighter than they are now to have a lower footprint than PET ones [66]. End of life treatment of plastic bottles has also been studied: while even though the majority of PET bottles are discarded, a study in China examined the production of blankets made entirely from recycled plastic bottles where impacts stemmed mainly from additional energy inputs resulting in global warming and fossil fuel depletion [67]. Ref. [2] analyze the potential impact of China's ban on the import of plastic wastes, of which the dominant category was PET. The study shows that in the absence of about 2 million metric tons of recyclable PET imported prior to the ban, China is likely to use virgin coal as a substitute to produce PET fiber, increasing the life-cycle environmental impacts for all categories considered [2].

Another use of plastics in packaging is take-away food containers. UNEP reviewed 6 LCA studies on take-away food packaging and alternatives including materials such as fossil and biobased plastics, aluminum, paper, cardboard, wood and glass [68]. Chemical contamination becomes an important factor for the LCA of food containers because it deteriorates the recyclability of the material [68]. Ref. [69] suggest that compostable tableware can be a better alternative since they can be sent to a composting facility together with the food waste and the analysis show that compostable tableware perform better in 7 out of 15 categories including climate change, when compared to traditional fossil based plastic tableware [69]. Cutlery in take-away food containers is disposed of and incinerated often without energy recovery. Shifting to different compostable materials could result in a tenfold reduction in energy consumption [70]. Given the preference for single-use containers for take-away food, the UNEP report recommended that the consumer behavior and functional differences between take-away food packaging should be taken into consideration when assessing different alternatives [68]. Also, Some studies point out that the ability for food packaging to prevent food waste, which is often ignored in food packaging LCA studies, is an important consideration in LCA, because food wastes often embody higher environmental impacts than packaging itself [44]. Most of the time, food waste has greater environmental impact than the packaging itself [68]. The weight of the material, production technology, resource use, geographical context, and end-of-life management are crucial factors for deciding the environmental impacts of food containers [40, 68].

The literature on plastics LCA applied to packaging generally agrees that plastics should be lighter and use more reusable alternatives [57]. However, there are some disagreements and inconsistencies on the impacts of single-use packaging products when compared to plastics [11, 12, 49, 57, 58]. Overall, packaging is often not the major contributor to a product's life cycle impacts, and the weight of the material, resource use, geography, process efficiency, and waste management determine the significance of its contribution [68]. Functional use of each packaging should be defined clearly, and a systems approach should be taken to find the best solution for each use.

2.4.2. Agriculture

Plastics are used in greenhouses, mulching, irrigation systems, crop transportation, and many other applications in the agriculture industry [1]. We have reviewed 6 LCA studies on agriculture which focus primarily on fossil fuel consumption, acidification, and eutrophication. According to Plastics Europe, agriculture made up 3.4 % of the total plastics demand in Europe in 2018 [71]. While the percentage of agriculture seems to be low, agricultural plastics are difficult to collect from open fields and plastics can be transported to the open waters along with other materials [71]. Chemicals like pesticides adhere to the plastic surfaces and microplastics can carry pesticides into nearby water bodies [72]. Biodegradation of these microplastics accelerates the release of associated organic contaminants, many of whose effects we do not yet understand (Figure 3). The release of plastics will in turn affect plant health and transfer to human toxicity effects.

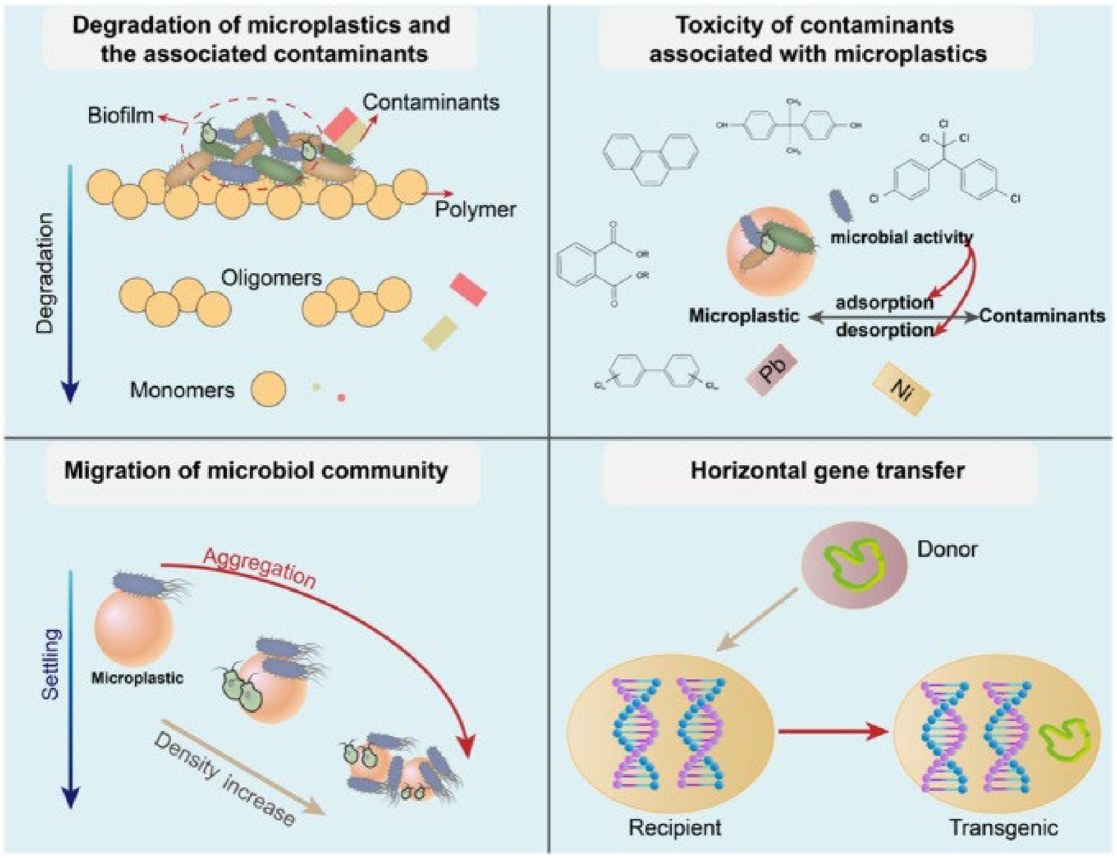

Fig. 3. Implications of microbial habitation and biofilm formation on microplastics [73].

Mulch film is responsible for the largest portion of agricultural plastics use (41 %) followed by greenhouses and tunnels [74-76]. Mechanical recycling is challenging for mulch films because they are highly contaminated with soil, stones, and biological waste. The alternative to mechanical recycling of mulch film is the use of biodegradable materials. These alternatives are compared to non-biodegradable plastics for mulch film to cover 1 hectare of mulched agricultural land [76]. The potential impacts are reduced by 25 % to 80 % depending on the impact categories and the end-of-life scenario [76]. Granule production was the biggest contributor to global warming and nonrenewable energy consumption for both biodegradable film and Polyethylene (PE) film [76]. Biodegradable plastics can be a good alternative since the waste at the end of the crop cycle is reduced to zero [76].

Plastic greenhouses are mainly located in eastern Asia and the Mediterranean Basin, accounting for about 10 % of agricultural plastics use [74]. The environmental impact of using plastics materials in greenhouses is dependent on factors such as its carbon emissions during the production phase and its generation of waste during its use in greenhouses [77]. It has been found that plastics used in greenhouses have a smaller environmental impact than glasses used in greenhouses during the production phases of each material [78]. Recycling of greenhouse plastics must also be considered. Mechanical recycling of plastics is only beneficial if they are sorted properly. Therefore, the utility of using plastic waste to substitute other materials, such as concrete, in greenhouses is nonvalue adding [78]. Additionally, PVC greenhouse materials can release heavy metal stabilizers into the environment during mechanical recycling [78].

Another LCA study has been conducted on agricultural plastic nets. Plastic nets are oftentimes used in agricultural operations to enhance agriculture yields and nutritional properties as well as limit exposure to insect infestation and solar radiation [79]. However, it is often impossible to measure the global volume of agricultural net production due to a lack of data availability from agricultural and manufacturers associations [74]. Like mulch film, biodegradable plastic nets made of poly-lactic acid (PLA) require a lower energy demand for production and have a lower GHG emission equivalent than non-biodegradable plastic nets made of polymer (PE) [79]. Recycling of agricultural plastic nets seems like a sustainable disposal method as it diverts landfill volume and removes excess plastics from the environment. However, recycling plastic nets can trigger water and air pollution and is confined by technological limitations that inhibit the process from being fully effective [79].

2.4.3. Building and Construction

Plastics are an important component in the building and construction sector due to their unique properties, such as being lightweight and corrosion resistant [80]. A number of LCA studies have been conducted to evaluate the environmental impacts of building materials and construction processes. We have examined 8 LCA studies in the sector from the U.S., Europe, and Australia.

One of the most common uses of plastics in construction materials is fiber reinforced plastics (FRP). FRP is a composite material with a polymer matrix blended with fibers, such as carbon and glass [81]. FRP has the advantages of high durability and high resistance under extreme environmental conditions, making it superior compared to construction materials [80]. Using FRP instead of traditional steel significantly reduces the amount of reinforcement by 67 % due to FRP's comparatively low weight [81]. It is also found that FRP reinforced pavement's environmental impact is significantly lower than that of steel-reinforced one, since FRP reinforced material requires less maintenance [82]. Regarding buildings and construction, a recent study indicates that the FRP reinforced bridge produces 50 % less carbon dioxide and requires 59 % less embodied energy than conventional steel reinforcement [83].

Polypropylene (PP) fiber is another successful application of plastics in the building sector [84]. PP is widely used as a popular alternative to steel reinforcing mesh (SRM), which is traditionally used to reinforce concrete to prevent cracks [85]. Detailed LCA shows that, compared to SRM, recycled PP fiber consumes 99 % and 91 % less water and fossil fuel, respectively [85]. Moreover, it produces 93 % less CO_2 equivalent and 97 % less PO_4 equivalent [85].

In addition to using plastics as replacement materials for concrete, they are also used to replace other building materials, such as those for plumbing, flooring, and insulation. Cross-linked polyethylene (PEX) may be used for plumbing as a replacement for copper, and consumes 42 % less CO_2 equivalent, contains 47 % less embodied energy, and has lower environmental impacts across impact categories than the metal [86]. PVC may be used for flooring instead of wood or linoleum (cork), but creates 2.6 times the carbon emissions of linoleum and 9.8 times the emissions of wood flooring, and has higher impacts on water quality and creates more waste than the other materials [87]. Extruded polystyrene (XPS) is a rigid cellular plastic with a closed cell structure that, along with fiberglass and corkboard, is used as an insulation material. XPS has 57 % CO_2 equivalent more than fiberglass, and 2.8 times the CO_2 equivalent of corkboard [88]. Moreover, it produces 50 % and 29 % more SO_2 equivalent than fiberglass and corkboard, respectively [88].

PVC is commonly used in water and sewer pipes in buildings. Compared to other materials used for pipes, PVC has 61 % less embodied energy than HDPE and 56 % less embodied energy than ductile iron [89]. However, the manufacture, use, and disposal of PVC pose significant environmental and human health hazards. It creates long-lasting pollutants including chlorinated dioxins, chlorinated furans (polychlorinated dibenzofurans), polychlorinated biphenyls (PCBs), hexachlorobenzene (HCB), and octachlorostyrene (OCS). Aside from pipes, PVC is also used for wall and floor coverings, which is linked to higher rates of exposure to health hazards such as toxic mold growth, heavy metals, and PVC phthalate [90]. Energy consumption and GHG emissions from PVC are lower than similar materials, but toxicity exposure is higher. It is important to look at various environmental and human health indicators while accessing different plastics materials. As in the case of PVC, they might be performing well in one of the indicators but might be very harmful in another category.

Research also found that the end-of-life treatment method is an important factor when determining life-cycle environmental impacts. For example, the production of recycled PP fiber emits 50 %, 28 %, and 78 % less CO_2 equivalent, water, and fossil fuel compared to virgin PP fiber, respectively [85]. In the building and construction sector, using recycled plastics could be a feasible and promising solution without compromising the structural properties.

2.4.4. Automotive

Polymers and composite materials are often used in the automotive sector as a replacement for metals [91]. One main driver is the more stringent fuel efficiency standard, such as the Corporate Average Fuel Economy (CAFE) standard. These fuel efficiency standards require vehicles to be more fuel efficient; weight reduction is one of the most cost-effective approaches. Reducing automobile weight is a key method in improving fuel economy and energy efficiency of automobiles. Compared to using metals such as steel, using fiber reinforced plastics offers weight reduction potential in the range of 50 % to 60 % [92]. In a review of 33 LCA studies, all studies indicate that using aluminum, glass-fiber reinforced plastic, and high strength steel to replace conventional steel reduces the vehicle's GHG emissions [93]. Due to the high flexibility of options given by the variety of materials present, these material changes will be able to reduce life cycle emissions. Furthermore, studies have examined lightweighting specific automotive parts through the addition of glass-fiber reinforced polymers (GFRPs). GFRP products consistently perform better than steel alternatives in the global warming potential, energy demand, and acidification potential categories [94].

In addition to making cars weigh less, plastics have great design flexibility [95]. They can be molded to tight tolerances and complex shapes, with lower tooling costs than most metal components [95]. We found 7 LCA studies mostly from Europe focusing on plastics in the automotive sector, of which 2 focus on carbon-reinforced plastics, while others examine plastics car parts, residue, and alternative feedstocks. Environmental impact categories varied, but the majority included energy consumption, global warming potential, acidification, and human toxicity.

Plastics have been widely applied in automobiles and one example is carbon fiber reinforced polymer (CFRP). CFRP are composite materials that use carbon fiber to strengthen the materials, while polymer enhances the cohesiveness [96]. It is estimated that, for a passenger vehicle, applying CFRP indicates a vehicle life cycle energy savings potential of 17 % to 25 % with assumed weight savings potential of 65 % to 70 % [97]. A similar study compares lightweight solutions including recycled magnesium and conventional steel to other options in a cradle-to-exit gate stage modality. Out of those solutions, CFRP has the best performance [98]. However, concerns about high energy intensity and cost may become major barriers to large-scale CFRP production and application. It is estimated that the energy intensity of carbon fiber production is 14 times higher than conventional steel production [92]. CFRP and steel energy consumption is found to be comparable when considered from the perspective of input data variability due to numerous sources used [92]. The high energy content of carbon fibers results in higher energy recovery in recycling, and a lighter weight in lower use phase energy consumption. Other common reinforcement materials include glass fiber, clay, and natural fiber, which is further discussed in section 2.5.3 [7, 99].

Similar to other sectors, end-of-life treatment is also an important consideration for plastics usage in the automotive industry [100]. In 2015, the European Union's End-of-Life Vehicle (ELV) Directive set specific recycling targets for vehicles and components [100]. It requires that since January 2015, for all end-of-life vehicles, a minimum of 95 % of vehicle weight needs to be recycled [100]. In Europe, the recycling rate of automobiles is high because the metallic fraction (usually 75 % of the vehicle weight) has been recycled [101, 102]. The remaining 25 % of material consists of plastics, fibers, and glass, which is also called automotive shredder residue (ASR). Therefore, the use of plastics in ASR is a hurdle in achieving a higher recycling rate of the vehicles.

Several studies evaluate the environmental impacts of ASR treatment. Ref. [102] evaluated different end-of-life treatment methods, including landfill, incineration, mechanical recycling, and chemical recycling. The results indicate that small-scale recycling plants, modeling with mechanical recycling and chemical recycling options, achieved the lowest impacts due to energy and material recovery [102].

2.4.5. Consumer Goods

Plastics are widely used in many consumer products such as toothbrushes, wipes, nappies, feminine products, masks and other personal care products. We found eight studies examining plastics in consumer goods, two of which were analyses of already-published studies. Most of these studies analyzed plastics materials in Europe, however, other studies were held in the United States and Brazil. Environmental impact categories varied from study to study, but common ones include global warming potential, ozone depletion, and human toxicity.

Johnson & Johnson analyzes different handle materials for its toothbrushes to reduce environmental waste and integrate environmental issues in its product design and development [103]. The results show that incorporating pre-consumption plastics waste into toothbrushes reduces both production costs and environmental impacts by 44 % [103].

A study from the UK comparing single-use nappies (diapers) to reusable ones concludes that there is ultimately no better product [104]. The drivers of environmental impacts are different in single-use and reusable nappies [104]. Single-use nappy manufacturers can reduce nappy' weight and improve the manufacturing process, whereas reusable nappy users can reduce the energy consumption for washing and drying to lower their environmental impacts [104]. Use patterns and end-of-life disposal are also important factors while determining environmental impact of reusable nappies [104]. Biobased nappies demonstrate potential reductions in environmental impact if they are composted at the end-of-life [104].

When comparing different feminine products, reusable menstrual cups perform better than single-use pads and tampons [105]. Single-use tampons have a lower environmental impact than single-use pads, especially if there is no applicator [105]. Additionally, reusable menstrual cups produce less than 10 % of the costs of disposable products [105]. Again, the life-cycle stage that has the biggest contribution changes depending on the product. The use-phase is an important stage for the menstrual cups, that depends on water use and washing frequency [105].

Today, personal protective equipment such as masks are widely used, primarily due to the global pandemic. There are various single-use and reusable masks on the market. According to a study, reusable masks have a lower environmental impact than disposable masks [106]. The most important contributors to the impact change depends on the reusability of the product [106]. Material production, packaging, and end-of-life management are significantly impactful stages for the disposable masks, while the use-phase (washing and sterilization) are the most important stages for the reusable masks [106].

UNEP's report on single-use plastics products emphasizes the importance of geographical and cultural context in the analysis [105]. Geographical location affects resource availability and use, energy mix, and production technology [105]. Cultural context affects the adoption of alternative materials and user behavior [105]. Both cultural and geographical factors affect recycling rates and end-of-life management practices [105]. While a comparison of many LCA studies indicated that recycling is on average the preferred option if it could replace at least 70 % to 80 % of virgin plastics, this depends on the performance of recycled plastics as well as the efficiency of mechanical recycling and energy recovery that results [107]. The majority of LCA studies conclude that single polymer plastics recycling with minimal contamination replace virgin plastics at a ratio close to 1:1, and recycling is generally the preferred treatment option [108]. However, these conditions do not always occur. As demonstrated in several studies, recycling performance is optimized when pretreatment is adapted to the recycling technology, which may reduce emissions from the most demanded plastics by up to 73 % [109]. However, post-treatment of plastic solid waste from goods is also important with reviews showing that pyrolysis is often the preferred advanced waste treatment technique with advantages from bio-oil and biochar production [40].

Each study had varying results, however, it was found that recycled and reusable materials found in consumer goods generally have a lower environmental impact than virgin plastics materials.

2.4.6. Electronics

The market for consumer electronics is rapidly growing, and historically environmental impacts have not been considered in the product design and development processes. Fossil-based plastics are widely used in electronics. Additives in the composition make it difficult to recycle and reuse; therefore, the recycling efficiency is low [110].

Two studies that are reviewed in this report focus on laptops and printer panels as case studies and evaluate the impacts of the use of fossil-based plastics and alternative materials, including bio-based materials, aluminum, and bamboo [110, 111]. Many electronics products are still lacking in life cycle assessment studies and do not have many publicly facing studies available. The laptop case study found that in order for metals to be a better option than plastics, the recycled content of the product should be above a certain percentage depending on the metal, thus replacing the primary metal production [110]. To be able to understand the impacts of bio-based materials, there needs to be more research on end-of-life of bio-based materials. In general, increasing the post-consumer recycled content in consumer electronics will result in tangible reductions in environmental impacts [110].

Another study shows that additives in printer panels are the biggest contributors to cradle-to-gate GHG emissions of a printer's GHG emissions (up to 40 %) [111]. For both flame retardant and non-flame retardant panels, two alternative biobased plastics are suggested [111]. For flame retardant panels, biobased material provides similar environmental and economic performance, whereas, for non-flame retardant panels, the environmental performance is improved with the biobased alternative [111].

2.5. LCAs on Plastics Additives

Additives in the plastics industry are chemical compounds that are added to basic polymers to enhance the performance, functionality, and aging properties of plastics products [7]. Generally, there are four main categories of additives: (1) functional additives, (2) colorants, (3) fillers, and (4) reinforcements [7]. Specifically, functional additives include plasticisers, flame retardants, stabilisers, slip agents, lubricants, curing agents, blowing agents, and biocides [7]. Under each of these categories, there are many potential available chemical substances, ranging from a simple substance such as zinc oxide (an inorganic pigment) to complicated organic substances such as bis (2-ethylhexyl)phthalate (DEHP, a common plasticiser) [7]. Due to the heterogeneity of these substances, the amount and quality of the studies vary significantly across the categories. Overall, additives are often omitted from plastics LCA studies [37].

There are two main types of LCAs for these additives. The first one is the cradle-to-gate LCA for the substances themselves. There are more studies of this type, considering there are existing LCA tools and databases for chemical substances. Some examples include the Chemical Life Cycle Collaborative (CLiCC) and the Environmental Assessment Tool for Organic Synthesis (EATOS) [112, 113]. If the additives do not affect the life-cycle characteristics of the products significantly (e.g., the colorants which only change the color of the products), this type of study can be simply integrated into the LCA of the plastics products. However, for additives such as stabilisers, which would significantly increase the life-time of the product, comprehensive LCAs for the product with and without such additives will be necessary to understand their environmental impacts. The second type of the LCA, which is the cradle-to-grave LCA for the plastics products with the consideration of the additives, would be a better fit to understand the impact. There are fewer studies of this type in the existing literature, and most of them focus on specific industries, probably due to the initiatives in such industries for sustainability. To develop this type of comprehensive analysis, it is necessary to predict how various additives would affect the service life-span and degradation rate of a plastics product.

In the following sections, 12 LCA-related articles are reviewed regarding additives in plastics production. If an additive category is not reviewed, it means we cannot identify a significant amount of LCA studies focusing on its applications. However, it is still possible to find LCAs of the substances in the category in databases like CLiCC or EATOS.

2.5.1. Functional Additives

2.5.1.1. Plasticizers

Plasticisers are small organic molecules, which are used to improve the flexibility and processability of polymers [114]. The impacts of plasticisers on human health and the environment have been widely studied, and there are existing tools to help evaluate the risks of certain plasticiser substances [114, 115]. However, there are only a few studies that use LCA to analyze the environmental impacts of plasticisers, and these studies are usually cradle-to-gate LCAs for specific plasticiser substances. The European Council for Plasticisers and Intermediates (ECPI) applies cradle-to-gate LCA on the production of di-isononyl phthalate (DINP) [116]. Since this study only focuses on DINP instead of evaluating a range of plasticiser substances, the result cannot provide much insight. Another study in 2013 applies cradle-to-gate LCA to DEHP, and compared its impacts with acetyl tributyl citrate (ATBC) and diisononylcyclohexane dicarboxylate (DINCH) in a semi-quantitative fashion (full LCAs for ATBC and DINCH are not applied) [117]. The study concluded that DINCH has a higher cradle-to-gate impact than DEHP, but has a lower impact when considering the use phase [117].

2.5.1.2. Flame Retardants

Flame retardants (FR) are used to inhibit, suppress, or delay the ignition in plastics [118]. Similar to other additives, FRs are usually ignored in plastics LCA studies, yet there are still several cradle-to-grave studies focusing on the use of FRs in various products [118]. Deng and co-workers apply cradle-to-grave LCAs to two printed circuit board (PCB) substrates--a biobased one using melamine polyphosphate (MPP) as FR and a conventional one using tetrabromobisphenol A (TBBPA) as FR [119]. They show that the biobased PCB has better overall environmental performance, but has a worse land-use impact [119]. Jonkers and co-workers apply cradle-to-grave LCAs to two types of laptops, one using brominated flame retardants (BFR) and another one using halogen-free flame retardants (HFFR) [120]. They conclude that BFR has worse environmental impacts compared to HFFR, which is mainly caused by the high human toxicity impact of BFR in the end-of-life stage of the product [120].

The main function of FR is to prevent the risk of fire, which would affect the life-cycle characteristics of the plastics products. Since conventional LCA assumes a normal operation during the life-cycle of the product, it cannot necessarily capture the potential environmental benefits brought by the prevention of the fire events due to the FRs. To tackle the problem, a research group in Sweden developed a Fire-LCA model [121]. This model is a cradle-to-grave LCA model including fire event as a potential end-of-life scenario for the product and includes all the corresponding emissions of the fire event in the LCA analysis [121]. In another study from the same group, they applied the Fire-LCA model to two types of TV enclosures--one with FR and one without FR and concluded that the TV enclosure without FR has significantly higher life-cycle emissions due the emissions from the potential fire event [122].

2.5.2. Colorants

During our review, we could not find LCA studies focusing on the application of colorants in the plastics industry. However, unlike other additives which are usually ignored in existing LCAs for plastics products, colorants are sometimes included. For example, both USEEIO and CEDA include synthetic dyes and pigments in their analyses for plastics products [25, 26].

2.5.3. Fillers and Reinforcements

The term “functional fillers” is sometimes used to describe both fillers and reinforcement in plastics production. The purposes of adding fillers and reinforcements into plastics products are usually cost reduction and property (especially mechanical property) enhancement [99]. Typical filler/reinforcement materials include talc, clay, natural fiber, carbon fiber, and glass fiber [7, 99]. Natural fiber, such as wood, cellulose, cotton, and flax, are gaining more attention in current studies due to their “green” characteristics [123]. Compared to other additive categories, there are relatively abundant studies on this topic.

The first type of study compares plastics with fillers to virgin plastics using LCA. Vidal and co-workers applied LCAs to virgin plastics and recycled thermoplastic with rice husks and cotton linters; they concluded reduced environmental impacts for the latter [124]. Wotzel and co-workers apply LCAs to car side panels made of acrylonitrile butadiene styrene (ABS) and hemp fiber composite and discover that the latter has the better environmental performance [125].

The second type of study focuses on comparing plastics with different types of fillers using LCA. Corbiere-Nicollier and co-workers compare the environmental impacts of plastic transport pallets made of PP reinforced by glass fiber and PP reinforced by China reed fiber [126]. They show that China reed fiber reinforced PP has lower environmental impacts [126]. Korol and co-workers studied the environmental impacts of plastic pallets made of PP reinforced by glass fiber and three different natural fibers (cotton, jute, and kenaf) [127]. They found better environmental performances for jute and kenaf reinforced PP compared to glass fiber reinforced PP. However, cotton reinforced PP has the worst environmental performance due to its cultivation process with significant environmental impacts [127]. Luz and co-workers researched the environmental impacts of sugarcane bagasse reinforced PP and talc reinforced PP and concluded a better environmental performance for the former [128]. It is worth mentioning that most of these studies focus on products in the automobile industry, probably due its initiative in improving product sustainability.

2.6. Overall Trend Summary

In general, literature agrees that use pattern assumptions are critical when comparing single-use and multi-use products [11, 12, 64, 105, 129]. Multi-use products might not perform better if they are not used sufficiently many times.[98] The production phase of multi-use products might have more environmental impacts than the single-use products, since they might require more inputs to make them more durable.

Various papers showed that waste generated from plastics can cause a variety of environmental impacts, and end-of-treatment methods greatly affect LCA results. Compared to landfilling, recycling has lower environmental impacts, including GHG emissions, water consumption, and fossil fuel consumption [85, 102, 130]. But the cost of pre-sorting and collecting waste makes it hard to improve the recycling rates globally [131]. In addition, regional waste management policy is an important determining factor. For example, the EU requires that plastics packaging is either reusable or recycled in a cost-effective way by 2030, which incentivizes plastics recycling in the EU market.

Regarding the LCA for recycling itself, papers showed that allocation of the environmental benefits of recycling lacks a consistent approach across the industry, and various industrial guidelines provide different suggestions [132]. This obscurity would potentially lead to double counting of credits across the whole economy [132].

Many papers show that factors that are being considered in the assessments depend on geographical and/or cultural context [56, 68]. These factors include resource availability and use, feedstock mix (virgin materials, fillers, and recycled materials), energy mix, adoption of new materials and manufacturing technologies, user behavior, recycling rates, and end-of-life management practices [56, 68].

3. Challenges and Gaps in Plastics LCA

There are several challenges in understanding the life-cycle environmental impacts of plastics that have been identified through our review.

First, there is a lack of reliable, up-to-date, high-quality, and regionalized data on plastics. The reviewed LCA databases are highly concentrated on the EU and North America markets. In the seven reviewed LCA databases, USEEIO, NREL, and CEDA are based in the United States. PlasticsEurope only covered European countries. Gabi covers both US and EU countries and CM.Chemicals has broader regional distribution. Ecoinvent has limited data for the Asian market, while Asia has emerged as the major contributor to global plastics production with approximately 51 % of the world's plastics produced in Asia in 2019 [1]. Region-specific emission factors are needed to accurately represent the background energy mix scenarios and end-of-life treatment options. Some of the recent datasets such as CM.Chemicals, however, started to address region- and supplier-specific life-cycle environmental impacts.

Second, existing LCA studies do not address the environmental, ecological and human health impacts of microplastics and the degradation byproducts from plastics. Microplastics are abundant in marine and freshwater environments [133]. However, the health impacts of microplastics are still not well understood [38, 133]. Research shows that microplastics, due to their composition and relatively large surface area, might transport contaminants up the food chain which can accumulate in the tissues of the organisms [134]. The research in this space of understanding plastic degradation is only emerging, and there are several pathways of degradation that must be considered (Figure 5). Weathering proceeds along multiple synergistic pathways that would be difficult to model in an LCA study due to the variety of pollution-caused damages that occur, including those to the carbon and nutrient cycles, habitat changes, changes to ecotoxicity, and societal impacts [38].

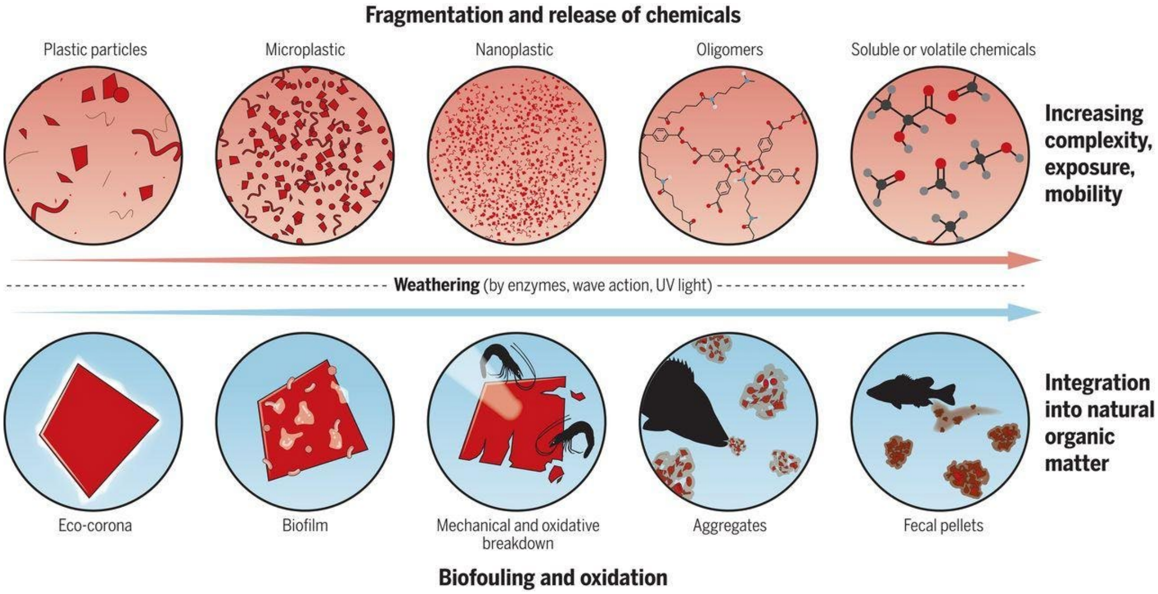

Fig. 4. Weathering processes of poorly reversible plastic pollution [38].

Third, additives are essential ingredients for plastics products, but additives are poorly covered in LCA studies. We reviewed twelve articles utilizing LCA to analyze the environmental impacts of additives in the plastics industry--one for plasticizers, five for flame retardants, and six for fillers and reinforcements (with special focus on organic reinforcement materials in the automobile industry). Other functional additives such as stabilizers, slip agents, and lubricants are usually missing in the existing literature [37]. Integrating additives into LCA for plastics products is challenging for several reasons. There are many potential substances, and no generalized LCA can cover them all. Also, some additives will alter the characteristics of the product (e.g., prolonged lifetime, reduced fire risk), and such characteristics need to be embedded in the cradle-to-grave LCA for accurate results. The impact of plastics additives also depends on the end of life (EOL) treatment for the product, which would decide the final destination of the additives [135]. On the other hand, additives, especially fillers, may also limit the EOL options for plastics products [123]. For example, the industry tends to incinerate plastics with high content of filler, since it is hard to separate the materials and recycle them. With EOL remaining a major gap in plastics LCA studies, it is difficult to accurately capture the impact of additives.

Fourth, currently, there is no publicly available tool with which manufacturers and researchers can assemble reliable LCIs of plastics. This problem becomes particularly challenging when developing an LCI for a new plastics material or product with no background data available in the public domain. When using existing LCI data, interoperability and compatibility among them with regard to technological, geographical, and temporal boundaries can pose additional challenges, since most of the existing studies only provide results with specific parameters. In reality, the ability to adjust such parameters would be far more helpful in the decision-making process.

Fifth, modeling of plastics' EOL is still a significant challenge. When developing a new plastics material or product, developing a reasonable estimate for its recycling potential is often difficult given that collection rate, type and severity of contamination, and accessible recycling technologies vary widely across geographies and times. Furthermore, currently there is no consensus on how to allocate environmental impacts for recycling or reusing of EOL plastics, understanding of which is crucial when developing a consequential LCA model for recycling or reusing plastics. Inconsistencies in allocation may result in ambiguous claims about recycled content, or other plastic use cases. For instance, some studies or declarations claim products with 100 % recycled content while allocating recycling credit to only parts of the production process, while the remainder does not contain recycled content [136]. Certification programs are inconsistent with their methodologies, and further amplify the obscurity of such claims. It is also notable that the inability to access sensitive information, such as the technical details of commercial technologies, is a perennial challenge in LCA, accessing such information related to plastics including emerging recycling technologies is not an exception.

Finally, some authors warn that plastic LCAs potentially have blind spots and have the potential to be misused [137]. For example, it has been pointed out that many LCAs of plastics do not take the end of life phase into consideration, making plastics often the preferable material, even though plastics may continue degrading and leaching into the environment [138]. Furthermore, not all potential environmental and human and ecological health impacts of plastics and their degradation byproducts are fully understood or incorporated into life cycle impact assessment (LCIA) methods, limiting our ability to evaluate plastics using LCA [138]. Some studies point out that LCA studies often focus on recycling options rather than studying alternative materials [137]. Additionally, much of the impact of plastic products depends largely on the use phase and post-consumption phase scenarios--how long and how many times a product is used by a consumer and what EOL treatment options are chosen [137].

4. Discussion and Recommendations

Given the growing demand for plastics and their increasing presence in the environment, the capacity to understand life-cycle environmental impacts of plastics is urgently needed. Having gone through the literature and databases covered in this review, we ask ourselves the question that prompted us to embark on this study in the first place: have we accumulated enough knowledge and data to understand the life-cycle environmental impacts of plastics? Despite that there has been enormous progress in the literature, our best answer to this question is ‘not yet’. That’s primarily because of the scarcity of reliable data and tools for conducting LCAs of plastics. In addition, science needs to be further advanced to capture the full spectrum of impacts caused by plastics and their degradation byproducts, and such new knowledge should be actively incorporated into the development of LCIA methods. What can be done to close these gaps in the literature so that LCAs can be utilized to reliably and credibly evaluate the life-cycle environmental impacts of plastics? Based on our review, we summarize our recommendations:

First, we believe that developing a set of reference LCIs for major plastics materials, additives, and prevailing processes is needed to enable rapid and reliable LCA of plastics. In particular, LCI data in general is scarce in North America and Asia. When developing such reference LCIs, it would be crucial to ensure interoperability and compatibility with existing, widely used LCI databases. Not only LCIs of plastics but also the data on plastics emissions to the environment as a part of other non-plastic products’ LCI need to be built. Currently little is known how much plastics are entering the environment throughout the life cycle of plastics-using products, making it difficult to quantify their environmental implications. Such an effort can best be accompanied by standardization of nomenclature, system boundary definitions, materiality criteria, default scenarios use, and EOL phases, among others. Existing platforms such as Federal LCA Commons can be leveraged to maximize the value of existing resources. Such LCI data is highly dependent on regional resource use and geographical context. Building an open-source data exchange platform could help facilitate the data collection process and enhance collaboration in the plastics industry globally. Forming federal level working groups would help standardize and coordinate the efforts on plastics LCA across executive branches.

Second, the on-going research on the prediction of the service life of plastics with the consideration of additives can be tapped into LCA research. Understanding the service life of a product is crucial in developing use phase and EOL phase scenarios. Inventorying the on-going research that is potentially relevant to LCA research and communicating the needs from LCA practitioners with relevant researchers at the early stage of such projects would facilitate synergies.

Third, one major opportunity is to develop a life-cycle screening tool focusing on plastics life cycle, which can also provide the flexibility to adjust several key considerations such as energy mix, feedstock mix, functional additives, end-of-life treatment. With the tool, plastics manufacturers could make more informed decisions in product design. This comprehensive plastics screening tool would enable users to input relevant information such as polymer type, country of production, energy mix, feedstock mix, EOL scenarios, and the type and quantity of additives. Based on the users' inputs, a publicly accessible tool can help understand the greenhouse gas emissions, human health impacts, and ecological toxicity of plastics. Such a tool will enable users to consider the environmental impacts in the design and use of plastics products.

Finally, the research on the fate, transport, degradation pathways and exposure of plastics as well as their human and ecological impacts need to be incorporated into mainstream LCIA methods. The science of plastics' environmental degradation pathways, bioaccumulation potential, and their impacts is rapidly evolving [73, 139-144]. However, currently none of these findings have been incorporated into mainstream LCIA methods, making the human and ecological impacts of plastics' degradation byproducts including those of micro- and nano-plastics a complete blind spot. Without the protocol to incorporate the recent research findings into LCIA, LCAs of plastics can provide only a partial view on their environmental impacts. We believe that existing characterization methods, such as the USEtox, may be applied to plastics once the degradation pathways of plastics in various environmental conditions are well understood. We recommend incorporating such findings into relevant characterization methods, so that the full ecological and human health impacts of plastics can be assessed through LCA.

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Supporting Information

4.1. Terms and Definitions

Plastics are commonly grouped into polyethylene terephthalate (PET), high-density polyethylene (HDPE), poly(vinyl chloride) (PVC), low-density polyethylene (LDPE), polypropylene (PP), polystyrene (PS), and the rest [17]. Table 2 shows the polymer types and the demand of most common polymers in the plastics market by mass.

It is notable that biomass-derived plastics are not always biodegradable and compostable, and biodegradable and compostable plastics are not always derived from biomass [39]. A plastic's ability to biodegrade rather depends on its design and conditions it is exposed to after use. See section 2.2.2. For details. Table 3 outlines some common terms and definitions used in plastics.

Table 2. Distribution of plastics demands by mass of resin types in 2018 [31].

Plastic Type	Demand by mass	Common uses
PP	19.3 %	Food packaging, sweet and snack wrappers, hinged caps, microwave containers, pipes, automotive parts, bank notes
PE-LD/PE-LLD	17.5 %	Reusable bags, trays and containers, agricultural film, food packaging film
PE-HD/PE-MD	12.2 %	Toys, milk bottles, shampoo bottles, pipes, houseware
PVC	10 %	Window frames, profiles, wall and floor covering, pipes, cable insulation, garden hoses
PUR	7.9 %	Building insulation, pillows and mattresses, insulating foams, refrigerators
PET	7.7 %	Bottles for water, soft drinks, juices, cleansers
PS/EPS	6.4 %	Food packaging, building insulation, electronic equipment, refrigerator inner lining, eyeglass frames
Others	19 %	Hub caps (ABS), optical fibers (PBT), eyeglass lenses and roofing sheets (PC), touch screens (PMMA), cable coating and communications (PTFE), and many others in medical and surgical devices, membranes, valves and seals, and protective coats

Table 3. Overview of definitions on plastics.

Terms	Description
Polymer	A large molecule composed of monomers typically connected by covalent bonds.
Plastic	A colloquial term for a wide range of synthetic or semi-synthetic polymers [1]
Bioplastic	While exact definitions vary, it generally refers to both biodegradable plastics and bio-based plastics [29].
Compostable plastic	A plastic that is capable of biodegrading at elevated temperatures in soil under specified conditions and time scales, usually only encountered in an industrial composter. A composting facility is one that transfers or stores composting materials resulting in biological decomposition [63]. In Europe, compostable plastics are defined to be materials that disintegrate within 12 weeks and completely biodegrade after 6 months [30].
Degradable plastic	A plastic that is capable of a partial or complete breakdown as a result of e.g., UV radiation, oxygen attack, biological attack, which implies an alteration of the properties, such as discoloration, surface cracking, and fragmentation [35].
Microplastic	The generic term for small pieces of plastics the longest dimension of which is under 5 mm [133].
Bio-based plastic	A type of plastic derived from biomass such as organic waste material or crops grown specifically for the purpose, which may or may not be biodegradable [35]. It applies to both naturally occurring polymers and natural substances that have been polymerized into high molecular weight materials.
Biodegradable plastic	A plastic that is capable of degrading under biological processes of organic matter, which is completely or partially converted to water, CO_2/methane, energy, and new biomass by microorganisms (bacteria and fungi) [35].
Fiber reinforced plastic	Composite material made of a resin matrix reinforced with fibers to enhance the mechanical properties [31].

4.2. Supplemental Documents

Two supplemental documents are available at https://doi.org/10.6028/NIST.GCR.22-032s

(1) a literature review spreadsheet (XLSX format) with categorization and brief summaries of documents found related to the state of LCA in plastics

(2) a set of presentation slides (PPTX format) that summarizes the findings in this report