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Review

Towards sustainable agroecosystems: A life cycle assessment review of soil-biodegradable and traditional plastic mulch films

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ABSTRACT

The increasing use of traditional agricultural plastic mulch films (PMs) has raised significant environmental concerns, prompting the search for sustainable alternatives. Soil-biodegradable mulch films (BDMs) are often proposed as eco-friendly replacements; however, their widespread adoption remains contentious. This review employs a comparative life cycle assessment perspective to evaluate the environmental impact of PMs and BDMs across their production, use, and end-of-life stages, providing strategies to mitigate their impact on agroecosystems. BDMs generally exhibit lower energy use and greenhouse gas emissions than PMs but contribute to greater land-use demands. Reported eutrophication and acidification potentials are less consistent, varying based on feedstock types and the scope of assessment of BDM, as well as the end-of-life management of PM. The environmental burden of both mulch types is influenced by the life cycle stage, polymer composition, farming practices, additives, film thickness, and local climatic conditions. The manufacturing stage is a major contributor to energy use and greenhouse gas emissions for both PMs and BDMs, despite their shared benefits of increasing crop yields. However, post-use impacts are more pronounced for PMs, driven by end-of-life strategy and adsorbed waste content. While starch-based BDMs offer a more sustainable alternative to PMs, uncertainties regarding the residence time of BDM residues in soil (albeit shorter than PM residues) and their effects on soil health, coupled with higher production costs, impede widespread adoption. For BDM end-of-life, soil biodegradation is recommended. Energy and material recovery options are crucial for PM end-of-life, with mechanical recycling preferred, although it requires addressing eutrophication and human toxicity. This review discusses these complexities within specific contexts and provides actionable insights to guide the sustainable integration of mulch films into agricultural practices.

1. Introduction

The use of mulch films in agriculture is increasing, with the global agricultural film market projected to reach USD 18.2 billion by 2028, growing at 6.9 % annually from 2023 to 2028 [1]. Plastic has, therefore, found its way into agricultural and adjacent natural ecosystems through soil-biodegradable mulch films (BDMs) and non-biodegradable traditional plastic mulch films (PMs). This is due to their remarkable benefits, including increased productivity, increased crop yield, improved crop quality, weed suppression,

Abbreviations: ACP, Acidification potential; ADP-fossil, Abiotic depletion (fossil); BDM, Soil-biodegradable mulch film; CH_4, Methane; CO_2, Carbon dioxide; CO, Carbon monoxide; dLUC, Direct land use change; ECP, Ecotoxicity potential; EUP, Eutrophication potential; GHG, Greenhouse gas; GWP, Global warming potential; HTP, Human toxicity potential; iLUC, Indirect land use change; LCA, Life cycle assessment; LDPE, Low-density polyethylene; LUP, Land use occupation potential; N_2O, Nitrous oxide; NH_3, Ammonia; NO_2, Nitrogen dioxide; NREU, Non-renewable energy use; PBAT, Polybutylene adipate-co-terephthalate; PBS, Polybutylene succinate; PHB, Polyhydroxybutyrate; PE, Polyethylene; PET, Polyethylene terephthalate; PHA, Polyhydroxyalkanoates; PLA, Polylactic acid; PM, Plastic mulch film; POF, Photochemical ozone formation; PP, Polypropylene; SO_2, Sulfur dioxide; TPS, Thermoplastic starch; VOC, Volatile organic compound; UV, Ultraviolet.

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optimized crop development, efficient water use, conservation of soil moisture, moderation of soil temperature, enhancement of microbial activities and growth, pest management, food safety compliance, and reduction of soil erosion [2–8]. Mulch films have, therefore, been widely used in high-value specialty crops such as strawberries, tomatoes, peppers, cucumbers, and pumpkins [7], as well as in other crops like corn and potatoes, where they can enhance agroecosystem services. We refer to the agroecosystem (or agricultural ecosystem) as a natural environment engineered for producing food, fibers, and resources through anthropogenic practices. In a similar definition, Swift et al. [9], as cited in Alhameid et al. [10], define agroecosystems as ecosystems in which humans have exerted deliberate selectivity on the composition of the biota (crops and livestock) and, to some extent, replaced the natural flora and fauna of the site. Agroecosystems provide various services, including food production, soil conservation, climate regulation, waste decomposition, carbon sequestration, and habitat maintenance [11,12].

PMs are fossil-based and non-biodegradable. Polyethylene (PE) mulch, specifically low-density polyethylene (LDPE)—is the most common commercially used PM today due to its relatively low cost, high durability, and flexibility [7,13]. In contrast, BDMs are derived from biodegradable polymers such as starch, polylactic acid (PLA), polyhydroxyalkanoates (PHA), etc. Degradation can occur through biotic or abiotic processes. Abiotic factors include bulk erosion, ultraviolet (UV) radiation, heat, pollutants, and weather elements like wind and precipitation, while biotic factors involve the activity of microorganisms such as bacteria, fungi, and algae. While PMs are fragmented primarily through abiotic means, BDM degradation involves a complex interplay of abiotic and biotic factors. Complete field biodegradation varies depending on climate and soil conditions and may take 21–58 months for 90 % degradation [14]. During biodegradation, BDMs are assimilated into microbial biomass, with carbon dioxide (or methane) and water as primary byproducts of microbial respiration [13]. Conversely, PMs used on the field degrade via abiotic factors and are fragmented into tiny pieces during harvest and collection. These fragments can take hundreds of years to fully degrade under natural environmental conditions [15], necessitating complete removal after use. Also, residual plastic fragments in the soil and watershed areas contaminate the environment and require high labor and retrieval costs [16].

Despite their benefits, different PMs have noticeable adverse environmental impacts, varying across scenarios depending on climatic conditions, farming practices, resources used, and end-of-life (EOL) choices. Life cycle assessment (LCA) is a valuable tool for evaluating these impacts and assessing the environmental footprint of mulch films throughout their life cycle—from production to field application and final disposal or EOL. LCA studies have attributed PMs to higher carbon footprints, non-renewable energy use (NREU), and global warming potential (GWP) [17–22], as well as particulate matter air pollution [20] and human toxicity and health concerns [23,24]. The production stage of PMs exerts substantial material use, having the highest impact on abiotic depletion potential (ADP) and human toxicity; however, ADP can be mitigated through energy recovery EOL methods such as incineration [24]. For instance, incineration and landfilling with energy recovery reduced the human health impact of cabbage production by 21 % and 16 %, respectively [20]. NREU and greenhouse gas (GHG) emissions were also reduced when incinerated with energy recovery [25].

The field application stage of PMs was found to have the highest GWP, acidification potential (ACP), and eutrophication potential (EUP) compared to no-mulch scenarios [24]. GWP during field application resulted from GHG emissions, such as carbon dioxide (CO_2), and changes in the net ecosystem CO_2 balance. EUP is attributed to the leaching of excess nitrogen and phosphorus from fertilizers, while ACP is linked to ammonia emissions from ammonium-based fertilizers. The post-use stage of PMs poses significant health and environmental threats, such as the release of plastic residue into the soil [26], ecotoxicity and emissions from additives and plasticizers in plastic fragments [27,28], and the emission of toxic pollutants like dioxins and particulate matter during plastic waste combustion [29], etc. Human toxicity, freshwater aquatic ecotoxicity, and marine aquatic ecotoxicity can account for up to 90 % of the total post-use impact due to mulch film waste treatment in incineration and landfilling [24]. In addition, mulch fragments left after removal can lead to micro- and nanoplastic accumulation, contaminating agricultural soils and potentially harming plant growth [30–32]. This raises health concerns about plastic ingestion from bioaccumulated plastics in products [7,32]. Furthermore, plastic films can adsorb pesticide residues 20 times more than the soil [33], causing ecotoxicity and eutrophication when migrated into the environment.

As PMs and BDMs offer comparable agronomic benefits—such as improving crop yield [34–36], preserving soil moisture, regulating soil temperature, and suppressing weeds [37], depending on specific mulch colors and types—BDMs are increasingly being considered a more sustainable alternative to PMs. Does this mean BDMs are all set to replace PMs without environmental concerns? This review employed a comparative LCA methodology to answer this, so a balanced assessment is done. Few studies reviewed comparative LCA studies, with all presenting case studies and not entirely on mulch films [38–41]. This study instead presents a detailed review of comparative LCA studies of mulch films to (a) explain the environmental burdens and economic considerations associated with the production, use, and EOL of PM and BDM on land, soil, and air, (b) examine the causes of these environmental impacts, (c) postulate impact mitigation strategies, and (d) provide insights for sustainable BDM adoption. This review provides a handy decision-making tool for agronomists, farmers, crop consultants/advisors, mulch film manufacturers, recyclers, and policymakers.

2. Polymeric categorization of agricultural mulch films

The market offers various agricultural mulch films for various applications with distinct or similar characteristics intricately linked to the polymers and additives used in their formation. In general, agricultural mulch films are categorized based on the source of polymers into two categories: traditional plastic films and biodegradable films.

2.1. Traditional plastic mulch films

Traditional PMs are produced from synthetic polymers, i.e., petroleum/fossil-based polymers. These plastics impose a substantial environmental footprint and require significant energy inputs during manufacturing [42]. Common plastic polymers for PM production include polyethylene (PE) and polypropylene (PP), with LDPE being the most prevalent. While polybutylene adipate terephthalate (PBAT) and polybutylene succinate (PBS) are fossil-derived, they are designed to biodegrade in soil or compost [41]. In this study, we categorized mulch films based on the biodegradability of the polymers for better comparative assessment. Therefore, only non-biodegradable, fossil-based polymers are categorized as traditional PMs. Both PBAT and PBS are biodegradable polyesters (see Table 1).

2.2. Biodegradable mulch films

BDMs are composed of biopolymers or biodegradable polymers. ASTM D883-23 [56] defines biodegradable plastics as those that undergo significant changes in chemical structure, physical properties, or appearance through naturally occurring microorganisms, such as bacteria, fungi, and algae [56], transforming into microbial biomass, water, and CO_2 aerobically or methane (CH_4) anaerobically [57]. The biodegradation of BDMs is a multi-step process involving (a) biodeterioration: fragmentation into small particles via abiotic factors and microorganisms; (b) depolymerization: conversion of polymeric molecules into oligomers, dimers, and monomers by microbial action; (c) assimilation: integration of these molecules into microbial cells; and (d) mineralization: release of oxidized simple molecules like CO_2, nitrogen, CH_4, and water into the soil [58]. The biodegradation period is a strong metric for biodegradability but varies with biopolymer types and environmental conditions. Some biodegradable polymers require higher temperatures for biodegradation, and their intended EOL is a composting environment, while others can be biodegraded in the soil after tillage.

Shen [41] categorizes biopolymers into biodegradable and bio-based biopolymers, noting that bio-based polymers are not necessarily biodegradable and vice versa, while biodegradable polymers are not necessarily made from biological resources. The following categorizations were established: (a) fully bio-based and biodegradable polymers, (b) fully bio-based but non-biodegradable polymers, (c) partially bio-based biodegradable or non-biodegradable polymers, and (d) fossil-based but biodegradable polymers. Please see Supplementary Material Block S1 for details.

This study categorizes BDMs into starch-derived and fossil-derived mulch films based on polymer sources. The starch-derived category encompasses starch-based films, e.g., TPS, films derived from starch fermentation, e.g., PLA and PHA films, and starch plastics, i.e., blends of TPS and biodegradable polyesters. The fossil-derived category includes biodegradable polyester, e.g., PBAT and PBS. Bio-PET, bio-PE, and bio-PP are not considered in this category because they do not biodegrade and are more suitable for water bottles, food packaging, and other consumer products. The characteristics, production pathways, and commercial examples of these biodegradable mulch polymers are presented in Table 1, while Fig. 1 represents categories of agricultural mulch film polymers.

3. Comparative life cycle assessment review of mulch films

3.1. Overview of life cycle assessment

The term “life cycle assessment (LCA),” also referred to as “life cycle analysis,” refers to a structured, systematic, comprehensive, and iterative method for evaluating the environmental impact associated with the life cycle stages of a product system, from raw material acquisition/extraction to disposal/EOL [59]. LCA methodology involves four stages: goal and scope definition, life cycle inventory (LCI), life cycle impact assessment (LCIA), and result

Table 1
Categorization of biodegradable mulch film polymers: Characteristics, production pathway, and some commercial examples.

Category	Polymer	Characteristics	Production pathway	Commercial film blends
Starch-derived mulch film	TPS	• Made from renewable source—starch. • Thermoplastic, biocompatible with other polymers, and biodegradable • Fragile, retrograde, poor water resistance, unsatisfactory mechanical properties [43–45].	Starch granule disruption + plasticizer/ additives + film processing [43,44].	Biomax TPS, Mater-Bi, EcoWorks, Bionolle [46,47].
	PLA	• Made from renewable sources—starch and sugar. • Eco-friendly, biodegradable, biocompatible, has low production cost and is abundant. • Inherent hardness, brittleness, and limited thermostability [42,48].	Starch/sugar fermentation to LA + LA ring-opening polymerization to lactide monomers + lactide polymerization to PLA + film processing [42,44].	Ingeo, Bio-Flex, Ecovio, [46,49].
	PHA	• Water-insoluble, non-swelling, and hydrolytically degradation-resistant but biodegradable, especially in sediments. • Biocompatible and non-toxic but reduces water sensitivity of other biopolymers. • High production costs, energy-intensive, limited mechanical properties. • Biodegrades in aerobic and anaerobic environments [13,48].	Sugars/lipids fermentation to PHA + polymerization/processing + film processing [48,50,51].	Biomer L, GreenBio, ReNew [49].
Fossil-derived mulch films	PBAT	• Fossil-based aromatic–aliphatic polymer. • Favorable mechanical properties: stretchability, impact resistance, extensibility, heat resistance. • Biodegradable based on soil structure and microorganisms. • Expensive, highly UV sensitive. • Degradation leaves plastic fragments in the soil and releases toxins (adipic acid and terephthalic acid) [46].	Polycondensation of butanediol, adipic acid, and terephthalic acid + film processing [52].	Eastar Bio, Eco-Flex, Ecovio, EcoWorks, Enviro [47,49].
	PBS	• Favorable mechanical and thermal properties, e.g., meltability. • Similar properties to PET and PP. • Low toxicity, low degradation rate, relatively high rigidity, and high production cost. • Breaks down naturally in liquid cultures, compost, and soil. • Some PBS-based blends are immiscible, e.g., PBS/PHB, PLA/PBS [13,44,46,53–55].	SA from petro-based monomers or glucose fermentation to SA + SA and 1,4-butanediol esterification to oligomers + polycondensation to PBS + film processing [13,44,46,53].	Bionolle, Biosafe [47,49].

Abbreviations: LA, Lactic acid; PBAT, Polybutylene adipate-co-terephthalate; PBS, Polybutylene succinate; PET, Polyethylene terephthalate; PHA, Polyhydroxyalkanoates; PHB, Polyhydroxybutyrate; PLA, Polylactic acid; PP, Polypropylene; SA, Succinic acid; TPS, Thermoplastic starch; UV: Ultraviolet.

evaluation/sensitivity analysis.

An LCA study begins with the goal and scope definition. The goal establishes the purpose and intended audience, while the scope outlines the system and its functions, defines the functional unit, sets the system boundary, and identifies the impact categories, methods, tools, and data quality control measures. The functional unit serves as a quantifiable reference metric for normalizing input and output data [59,60] and ensuring equivalent comparison between two products based on equivalent levels of function [61]. System boundaries specify the processes to be included, often excluding minor inputs or outputs based on well-defined cut-off criteria. Depending on the system boundary, LCA can take different forms: cradle-to-grave, which encompasses the entire life cycle from manufacturing to mulch use and post-use disposal or EOL; cradle-to-factory gate, which focuses only on the production phase at the factory; or gate-to-grave, which includes either the use stage and post-use stage or only the post-use stage. The choice of system boundary depends on its relevance to the study's goal. The LCI phase follows, collecting quantitative data on materials, resources, and energy inputs, as well as product outputs and emissions. All LCA studies adhere to ISO 14044 [59] and ISO 14040 standards [60]. Detailed guidelines for LCA methodology can be found in the ILCD Handbook [62], Jensen et al. [63], and US EPA [61].

LCIA involves evaluating the environmental impact of unit processes within a product system using impact categories—classes representing environmental concerns chosen based on the scope and to achieve the goal of an LCA study. Impact categories relevant to mulch films include global warming potential (GWP), non-renewable energy use (NREU), abiotic depletion potential (ADP), eutrophication potential (EUP), acidification potential (ACP), ecotoxicity potential (ECP), human toxicity potential (HTP), land-use occupation potential (LUP), particulate matter potential (PMP), ozone layer depletion (OLD), and photochemical oxidation or photo-oxidant smog formation (POF). Table 2 summarizes the definitions of these impact indicators. Fig. 2 shows the relevant impact categories used for the environmental impact assessment of PMs and BDMs with their broad damage categories.

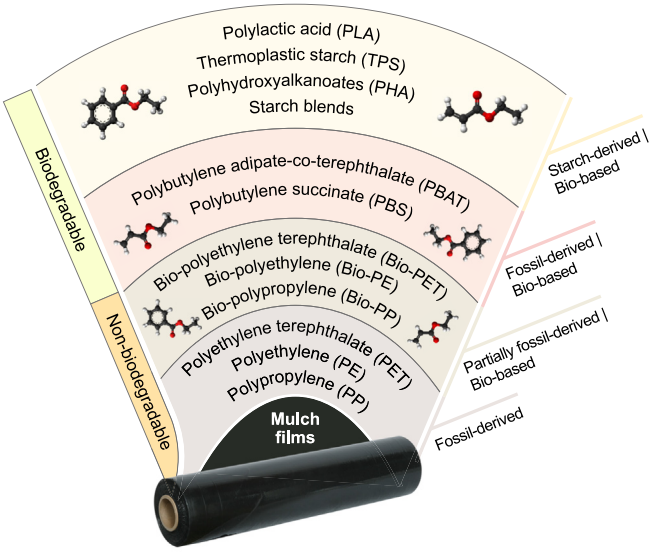

Fig. 1. Categories of biodegradable and non-biodegradable polymers used for agricultural plastic mulch film production. Not all bio-based mulch films are biodegradable, and not all biodegradable mulch films are bio-based. Biodegradable mulch includes starch-based mulch (e.g., TPS and starch blends), mulch derived from starch (e.g., PLA and PHA), and fossil-derived mulch (e.g., PBAT and PBS mulch films). Fossil-derived PE, PP, and PET, as well as their partially bio-based derivatives, including bio-PE, bio-PP, and bio-PET mulch films, are not biodegradable.

3.2. Comparative life cycle assessment of plastic and biodegradable mulch films

A comparative LCA evaluates two or more product systems, in this case, PMs and BDMs. Effective comparison is achieved when there is the same functional unit, methodological considerations, and impact assessment indicators for both PM and BDM [63]. However, inherent limitations exist both within and between comparative LCA studies. Within a single study, limitations can arise from varying assumptions and the lack of suitable data, which can restrict the scope of the LCA. Between different comparative studies, limitations may arise from differences in assumptions, scope, modeling approaches, and climatic or geographic conditions. These discrepancies can lead to variations in impact assessment results and should be interpreted within their respective contexts [66]. To minimize these limitations, this section only reviews LCA studies that performed comparative LCA of both PM and BDM within the same study, examining the entire life cycle of mulch films, from material manufacturing to field use and EOL stages. A systematic literature search was conducted on the Web of Science, Science Direct, and Google Scholar databases using a combination of specific keywords in the form "(life cycle assessment OR LCA) AND (plastic mulch film OR polyethylene mulch film) AND (biodegradable mulch film OR BDM mulch film)." Fig. 3 shows some of the reviewed articles' quantifiable metrics, indicating the percentage of environmental benefits between PM and BDM across ten impact categories under various scenarios.

3.2.1. Manufacturing stage: resin, granule, and film production

The material manufacturing stage of mulch films contributes dominantly to the overall environmental impact. The manufacturing of PMs consumes a substantial amount of energy and fossil resources, leading to increased fossil fuel consumption and GHG emissions [37]. Starch blends are widely available BDMs on the market today [67]. Although the manufacturing of starch-derived mulch films offers advantages in terms of energy consumption and GHG emissions over traditional PMs, their contribution to EUP is often higher due to bioplastic feedstock manufacturing, depending on the specific type of starch-derived film. In a cradle-to-grave study with a functional unit of 1 ha of black-mulched agricultural land, Razza et al. [68] found that starch-based BDM (Mater-Bi®) granule production had higher impacts in EUP and ACP than in PE mulch. However, when the overall life cycle impact was considered, including the post-use stage, starch-based BDMs demonstrated lower impacts across all categories, including EUP and ACP (Fig. 3). This was attributed to the additional environmental burden associated with the mandatory removal and disposal of PE films. This highlights the significance of the post-use stage to the overall environmental impact of PE films. The study revealed that the granule production phase dominates environmental impacts for both mulches due to material use, although impacts are higher for PE mulch, particularly in NREU and GWP.

Similarly, Broeren et al. [69] conducted a cradle-to-factory gate LCA on six types of starch plastics, using a functional unit of 1 kg of starch plastic granules in the Netherlands. The study found that starch plastic granule production reduces net GHG emissions (up to 80 %) and NREU (up to 60 %) compared to petrochemical-based LDPE and PP on a per-weight basis. However, starch plastics showed significantly higher EUP (up to 400 %) and agricultural LUP requirements (about 0.3–1.3 $m^2yr\ kg^{-1}$). Meanwhile, the study highlighted substantial uncertainties in the EUP results, citing challenges in accurately modeling eutrophication-relevant emissions and advising caution in their interpretation. In addition,

Table 2
Definitions of relevant life cycle assessment impact category indicators (major information extracted from Acero et al. [64]).

S/N	Impact category	Definition	Unit
1	Global warming potential (GWP)	Measures the potential of global warming caused by GHG emissions.	kg CO_2-eq
2	Non-renewable energy use (NREU)	Quantifies depletion of energy resources such as fossil fuels (e.g., coal, oil, natural gas).	MJ
3	Abiotic depletion potential (ADP)	Assesses depletion of non-biological resources, including minerals, metals, fossil fuels, and water. Unlike NREU, ADP also includes non-energy non-biological resources.	kg Sb-eq., kg minerals, MJ fossil fuels, or m^3 water.
4	Eutrophication potential (EUP)	Categorized into freshwater, marine, and terrestrial EUP. EUP assesses the potential impact of nutrient buildup in freshwater, marine, or terrestrial ecosystems, specifically nitrogen or phosphorus-containing chemical nutrients like NH_3, NO_3, NO_x, and P.	kg PO_4^{3-}-eq or kg N-eq.
5	Acidification potential (ACP)	Measures soil and water acidification potential due to emissions of acidic substances, including NO_x *and* SO_x.	kg SO_2-eq.
6	Ecotoxicity potential (ECP)	Categorized into freshwater, marine, and terrestrial ECP. ECP assesses the potential impact of emitted toxic substances such as heavy metals to air, water, and/or soil on freshwater, marine, and land organisms.	kg 1,4-DB-eq.
7	Human toxicity potential (HTP)	Evaluates potential harm to humans (including cancer, respiratory diseases, and other non-carcinogenic effects) from exposure to toxic substances like arsenic, sodium dichromate, and hydrogen fluoride.	kg 1,4-DB-eq.
8	Land-use occupation potential (LUP)	Assesses the potential impact of land use on terrestrial environments, including species loss, soil degradation, and changes in soil quality due to agricultural activities, anthropogenic settlements, and resource extractions.	m^2 a
9	Particulate matter (PMP)	Measures the potential health impacts from extremely small, suspended particles ($PM_{2.5}$ and PM_{10}) emanating from combustion, resource extraction, etc.	kg PM
10	Ozone layer depletion (OLD)	Assesses the potential destruction of the stratospheric ozone layer by emitted ozone-depleting substances such as CFCs, halons, and HCFCs.	kg CFC-11-eq.
11	Photochemical oxidation or photo-oxidant smog formation (POF)	Measures the potential for emitted substances, such as CO, SO_2, NO, ammonium, and non-methane VOC, to contribute to photochemical ozone formation.	kg C_2H_4-eq.

Abbreviations: 1,4-DB-eq, 1,4-dichlorobenzene-eq; PMP, particulate matter; CFCs, chlorofluorocarbons; HCFCs, hydrochlorofluorocarbons; NH_3, ammonia; NO_3, nitrate; NO_x, nitrogen oxides; m^2a, square meters annum; P, phosphorus; SO_x, sulfur oxides; GHG, greenhouse gas emissions; CO, carbon monoxide; Sb, antimony; SO_2, sulfur dioxide; NO, nitrogen oxide; VOC, volatile organic compounds.

replacing LDPE or PP mulch films with starch plastic films can save between 10 and 200 MJ energy per m^2 per year. Furthermore, using reclaimed starch instead of virgin starch significantly reduced environmental impacts, lowering agricultural LUP by up to 60 % and EUP by up to 40 %.

Similar findings have been reported for resin production. Using a functional unit of 1 kg of polymer, Hottle et al. [70] examined the production, transportation, and EOL of starch-derived polymers

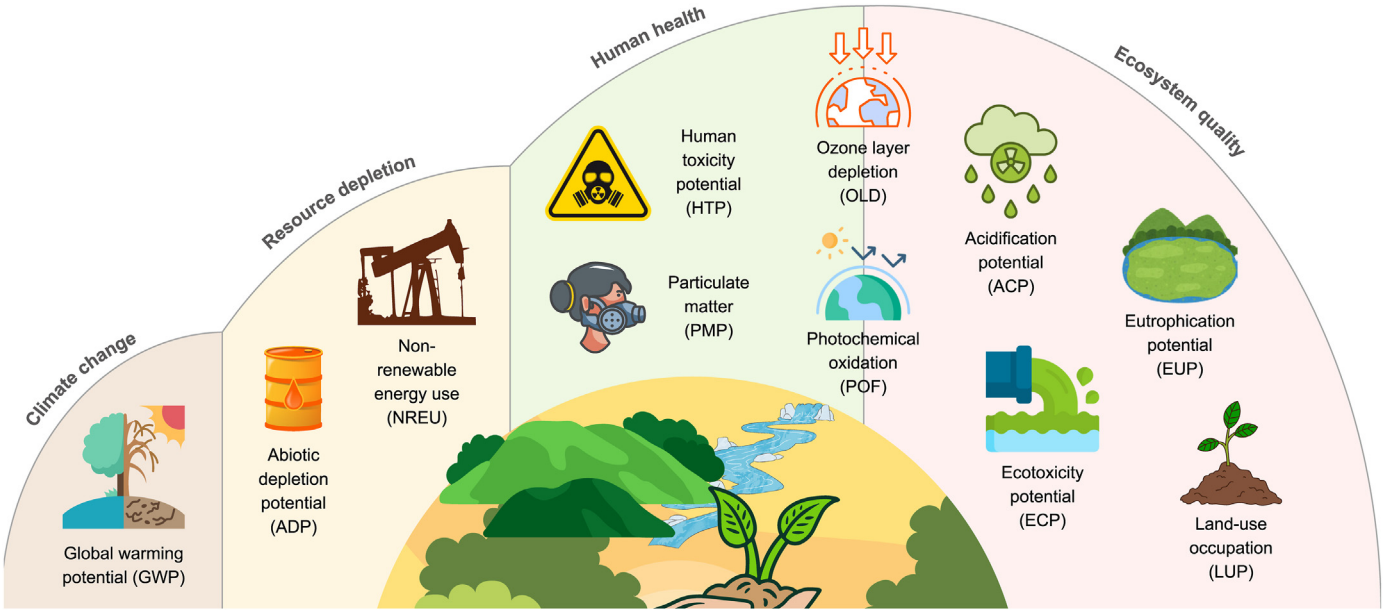

Fig. 2. Environmental impact categories with corresponding damage categories. The impact categories are grouped into damage categories—climate change, resource depletion, human health, and ecosystem quality—based on IMPACT 2002^+ methodology [65]. Damage categories represent areas of protection from potential damage of the environmental impact assessed. GWP contributes to climate change, while ADP and NREU lead to resource depletion. HTP and PMP contribute to adverse human health impacts. ACP, EUP, ECP, and LUP collectively affect ecosystem quality. Overlapping impacts from OLD and POF primarily affect human health and ecosystem quality.

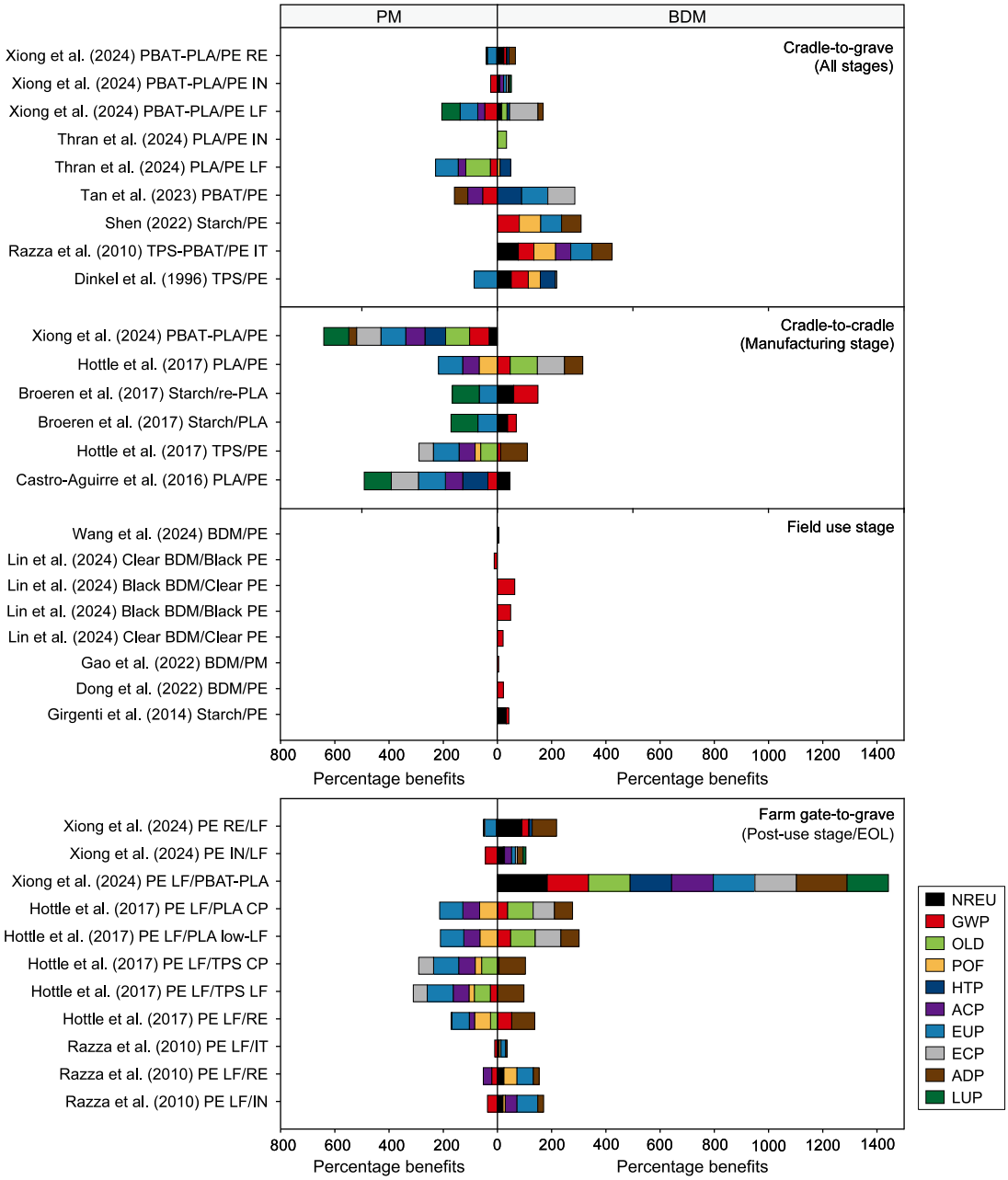

Fig. 3. Comparative life cycle assessment studies indicating the percentage of environmental benefits between plastic mulch (PM) and soil-biodegradable mulch (BDM) based on ten impact categories of different scenarios. For the Farm gate-to-grave (Post-use stage/EOL) block, the left and right bars represent categorizations before and after the "/" of the author label on the y-axis, respectively. ACP: acidification potential; ADP: abiotic depletion potential; CP: composting; ECP: ecotoxicity potential; EUP: eutrophication potential; GWP: global warming potential; HTP: human toxicity potential; IN: incineration; IT: Italian scenario of 84 % landfill and 16 % incineration with energy recovery; LF: landfilling; LUP: land-use occupation potential; NREU: non-renewable energy use; OLD: ozone layer depletion; PBAT: polybutylene adipate terephthalate; PE: polyethylene; PLA: polylactic acid; POF: photochemical oxidation or photo-oxidant smog formation; RE: recycling; re-PLA: post-industrial recyclate PLA; TPS: thermoplastic starch.

(PLA, TPS), petrochemical-based plastics (PET, HDPE, LDPE), and bio-based petrochemical equivalents (bio-PET, bio-HDPE, bio-LDPE). The study considered recycling, composting, and landfilling EOL scenarios. Biogenic carbon credits were given to raw material production processes, and the benefits of recycling were allocated to recycling EOL. Starch-derived plastic resin production had lower net GWP and NREU but higher ACP and EUP than petrochemical-based plastics. EUP and ACP were attributed to fertilizer manufacturing, effluent waste generation in starch production, and additional nitrogen from fossil-based plasticizers. On the other hand, sugarcane-produced bio-based petrochemical equivalents displayed high negative impacts in all categories except GWP and fossil fuel depletion. This benefit to GWP and NREU was also reported by Tsiropoulos et al. [71]. Using agricultural feedstocks such as sugarcane to produce ethylene and subsequent bio-ethylene introduces processes like farming, distillation, and dehydration. Notably, sugarcane farming and ethanol distillation contributed significantly to OLD, ACP, HTP, EUP, ECP, and POF. Ethylene dehydration, phosphorus-based fertilizers, and ocean freighter transportation were notable contributors to the negative environmental impact of bio-ethylene plastics.

In Castro-Aguirre et al. [72], the Ecoinvent resin production data

of 1 kg PLA and traditional polymers LDPE, PP, and polystyrene (PS) were presented. While PLA resin production offers benefits in NREU, it exhibits negative impacts in toxicity, EUP, ACP, OLD, agricultural LUP, and water consumption. Specifically, PLA resin production shows NREU benefits of 47 % more than LDPE. Notably, the data shows a higher climate change contribution for PLA than other traditional polymers, which, according to the author, is inaccurate as carbon sequestration was not accounted for. Also, PLA resin production significantly stresses water use compared to traditional polymers. Overall, PLA resin production has a considerable negative environmental impact compared to petrochemical resins, except for NREU and GWP. Meanwhile, using Ecoinvent 3, Thrän et al. [73] reported a higher carbon footprint for BDMs (PLA films) during the raw material stage than for LDPE films. This was attributed to emissions from planting, growing, harvesting, and initial processing, although it was unclear whether the data accounted for carbon credits from carbon sequestration. A functional unit of 1 ha of tomato greenhouse area in Spain was used. It covered the production and disposal stages while neglecting the use stage, assuming similar impacts due to negligible differences during greenhouse use. The manufacturing stage was also identified as the largest contributor to the carbon footprint. Similar findings were observed in the assessment of PLA/PBAT versus LDPE by Nessi et al. [66,74,75], which further indicated that PLA-based BDM manufacturing generally exhibits higher environmental impacts than starch-based BDMs. Likewise, in Xiong et al. [34], the manufacturing stage of PBAT/PLA BDMs exhibited higher impacts across all impact categories, which offset the benefits attributed to in-situ biodegradation (Fig. 3). However, when assessed on a cradle-to-grave basis, the overall impact of PBAT/PLA films was lower than that of PE films in OLD, ADP, ionizing radiation, HTP, and terrestrial ECP. The average impact reduction of ten categories was, therefore, only 2.6 % better than PE films.

The film production stage has a similar impact contribution. One of the earliest comparative LCA studies of TPS vs. LDPE films was conducted by Dinkel et al. [76], as reported in Patel et al. [38]. The study examined the impact of 150 μm-thick mulch films covering a 100 m^2 farmland in Switzerland, assuming an EOL scenario of 80 % incineration and 20 % landfilling. The study indicated that TPS films outperformed LDPE films in NREU, GWP, and HTP. However, LDPE exhibited better performance in EUP and deposited waste by 85 % and 27 %, respectively, while ACP and ECP were similar [38,39]. ACP was also the same for TPS/PBAT vs. LDPE films in the comparative assessment of Nessi et al. [74]. EUP was primarily attributed to fertilizer and chemical use in cultivating renewable raw materials for bioplastic production [39]. However, it is important to note that the high material use attributed to TPS films per the functional unit (about 60 % higher than PE) could have influenced the EUP results and may not reflect current practices, as material use of TPS films is now lower [38]. TPS film production also imposed additional stress on agricultural land use if virgin lands were displaced to grow starch crops, but using current agricultural lands had no additional negative impacts. Further details on impact causes and mitigation strategies are discussed in Section 4.

In another cradle-to-grave LCA case study comparing starch-based plastic mulch films to LDPE films reported by Shen [41], the material manufacturing phase was found to dominate, accounting for 80 % of the overall environmental footprint. The study used a functional unit of 1 ha of agricultural land mulched for six months, with an EU-mix EOL scenario comprising 5 % recycling, 53 % municipal incineration, and 42 % landfilling. Direct or indirect land-use change effects were not considered in the analysis. Results indicated that starch plastic generally had lower environmental impacts than LDPE films, except for particulate emissions from post-harvest field burning. Notable impact reductions by starch plastics were 81 % in GWP, 79 % in POF, and 72 % in ADP-fossils. Surprisingly, the data shows a 77 % reduction in terrestrial EUP by starch plastic compared to LDPE films, although no specific cause was identified for this. Similarly, biogenic carbon credit was assigned to the carbon footprint of starch plastic material production, reducing global GWP impact during raw material acquisition. Likewise, embedded biogenic carbon emissions were accounted for at the EOL stage during biodegradation, where the carbon is fully or partially oxidized to CO_2 and/or CH_4 or not emitted if stored in compost. This resulted in a relatively high GWP impact for the EOL of starch plastic films, but still 81 % lower than LDPE films. Overall, the environmental impacts were dominated by the material manufacturing phase of both starch plastic and LDPE films.

BDMs made from polyester displayed a higher environmental burden than starch-based ones due to substantial material and energy consumption. In Tan et al. [77], the environmental impacts of PE (10- and 14-μm thickness) and PBAT mulch films were compared, covering film production, transportation, and EOL. The functional unit was kg of mulch film covering 1 km^2 of agricultural land. PBAT film demonstrated significant net impact reductions over PE film in HTP, EUP, freshwater ECP, and marine ECP by 89.8 %, 96.4 %, 99.8 %, and 99.6 %, respectively. This suggests that PBAT minimizes toxicity to humans and aquatic life. Notable here is the low PBAT EUP. These lower impacts result from eliminating additional energy inputs and secondary pollutant outputs associated with the collection and waste management processes of PE mulch films, as soil biodegradation was considered for the EOL of PBAT. Pelleting plastic granules was the major source of eutrophication, accounting for 53.5 % in the manufacturing stage. Eutrophication was attributed to VOC emissions into the air [78] and organic matter release into freshwater. Nonetheless, PBAT film was unfavorable in terms of GHG emissions and fossil resource consumption owing to high energy and raw material consumption during manufacturing, specifically in the production and esterification of petroleum chemicals, e.g., basic raw materials like adipic acid, butanediol, and terephthalic acid. A similar conclusion was reported by Schrijvers et al. [25], where PBAT films had higher NREU and GHG emissions than LDPE films. Conversely, PE film showed significant environmental benefits with 54.0 %, 54.9 %, and 49.6 % reductions in GWP, ACP, and ADP-fossils, respectively, compared to PBAT film [77]. Nova Institute [79] reported similar results of substantially higher GWP for PLA/PBAT blend production than PE film, given the benefit of energy and material recovery of PE film at the EOL stages. Additionally, PBAT film's biodegradable nature contributes to higher waste volume and microplastic generation [80], although at a significantly shorter residence time than PE.

These findings stress the substantial environmental burdens associated with the material manufacturing stage of mulch films. The environmental impacts of specific BDM manufacturing processes vary, with starch-based BDMs exhibiting lower impacts compared to PLA-based and polyester-based BDMs. Additionally, the manufacturing stage of starch-derived mulch films generally demonstrates advantages, particularly in GWP and NREU, but presents major concerns in LUP compared to their petrochemical-based counterparts. Conclusions regarding EUP and ACP are less consistent and depend on the specific types of BDM feedstock and the scope of the LCA study. Addressing these challenges is crucial for the optimal environmental benefits of PLA and TPS films, as discussed in Section 4.

3.2.2. Field use stage

At the film application stage, the focus shifts to assessing the environmental impact of mulch use. Dong et al. [3] conducted a comparative LCA between biodegradable mulch (PLA/PBAT) and traditional PE mulch during a one-year maize planting cycle in

China, with a scope covering film application, post-use collection and transportation, and EOL. The functional unit was 1 ha of maize cropland covered. GHG emission was the main environmental impact during the film use stage in all scenarios and was greater than in the post-use stage. Nonetheless, PLA/PBAT had about 22.6 % lower GWP than PE mulch (Fig. 3) despite requiring about 2.4 times more material (by weight) for mulching per functional unit area, and biodegradation was assumed to be complete after one year with the release of CO_2 and water. Post-use impurities in PE mulch exacerbated environmental impacts, including GWP and POF, with about 80 % of collected PE mulch comprising soil-type impurities. An estimated 18.2 % PE mulch residue was reported. Although quantifying the impact of PE mulch residue using current LCA methodologies is challenging, its environmental implications are well known. PE mulch residue remaining in the field after removal is a concern as this could alter soil physical properties like bulk density, porosity, field capacity, etc. [81], alter soil water distribution and permeation [82], alter nutrient uptake and water flow [83], modify soil microbial activities [30], alter soil microbial community structure and stimulate soil CO_2 emissions [84], lower soil macroaggregates [85], and reduce plant growth and crop yield [86]. PE microplastics could selectively alter microbial abundance and seriously threaten terrestrial biogeochemical cycles [87]. Specifically, the presence of 5 % microplastics facilitated CO_2 release and altered the abundance of microbes responsible for N_2O emissions and CH_4 uptake. In addition, the additives in the micro- and nanoplastics become mobile through the action of wind and water, which can end up in ground and surface water [88] or taken up by plants [89].

In the LCA of strawberry cultivation conducted by Girgenti et al. [90] in Northern Italy, the environmental impacts of starch-derived BDM (Mater-Bi®) and traditional PE films were assessed, although limited to NREU and GWP. Using BDM reduced overall impacts by 10–15 % compared to PE films and outperformed PM in crucial categories of GWP and NREU. However, this is less than the overall 25–80 % reported by Razza et al. [68], probably due to the broader scope [40]. Razza et al. [68] reported that using BDM instead of PM can reduce GWP and NREU by 60 % and 80 %, respectively. Despite higher EUP and ACP benefits in PE mulch production, the overall environmental performance was worse due to higher plastic requirements and the compulsory removal and disposal of PE mulch waste. Therefore, adopting BDM over traditional PM for field applications is environmentally justifiable. Nonetheless, the environmental benefits of mulch application also depend on climate and production region.

In terms of crop yield, BDM has a comparable yield to PM. Numerous studies have shown this. For instance, similar yields were observed in tomato tunnels in northern Washington State, USA [91]; winter melon in Sicily, Italy [36]; strawberry production in Portugal [35]; and eggplant, pepper, and cherry tomato cultivation in the Hainan Province of China [34], among others. A study by Gao et al. [92], conducted on maize production systems, assessed the carbon footprint and GWP of no-mulch, PM, straw mulch, and BDM (mainly composed of PBAT). The results show that PM and BDM increased yield by 11.3–13.3 % and 9.4–10.6 %, respectively, compared to the no-mulch scenario. However, there was no significant net return due to extra mulch materials and labor expenses. Despite yield improvements, both PM and BDM significantly increased net GWP and carbon footprint compared to the no-mulch scenario, although BDM had a significantly lower carbon footprint than PM. The increase in GWP was attributed to higher CO_2 emissions, promoted by the increased quantity and activity of soil microorganisms that decompose organic carbon and enhance soil-atmosphere carbon exchange [92]. The study also highlighted the unsuitability of mulching practices in rain-fed semi-arid regions with excessive rainfall (>550 mm).

Similar results were reported for wheat cultivation under the same rain-fed conditions in the Loess Plateau, China, by Lin et al. [93]. Compared to the no-mulch scenario, the study examined the effects of black and clear PE mulches and BDMs (primarily composed of PBAT) on wheat yield and carbon footprint. The findings revealed that black BDM increased annual wheat yield by 60 % compared to no-mulch and PE mulches (clear PE: 40 % and black PE: 47 %). Clear BDM achieved a 47 % yield increase, comparable to PE mulch. In contrast to the findings of Gao et al. [92], who reported no significant net return with mulches, Lin et al. [93] found substantial net returns with black BDM, clear PE, and black PE, ranging from 17 % to 124 %. Regarding GWP, clear PE, and clear BDM increased CO_2 emissions by 43 % and 52 %, respectively, and net GWP by 49 % and 17 %, respectively. Overall, black BDM performed the best, enhancing wheat grain yield and net returns while reducing net GWP and carbon footprint compared to PM. These findings indicate that mulch color can significantly affect both yield and carbon footprint.

In summary, GWP is the primary environmental impact during the mulch use phase. Most LCA studies focus predominantly on GWP, where BDMs demonstrate lower GWP compared to PMs (Fig. 3). The environmental benefits of mulch applications also depend on climate and mulch color; dry climates benefit more from the microclimate effects of mulches. In terms of crop yield, BDMs perform comparably to PMs. These findings highlight that adopting BDMs over traditional PMs is environmentally and agronomically advantageous for field applications.

3.2.3. Post-use stage

3.2.3.1. End-of-life strategies and practical considerations. The EOL strategy adopted for PM waste is crucial for overall environmental impact. Priority should be given to material, heat, and energy recovery while minimizing waste streams like toxic wastewater and particulate matter. Commonly adopted EOL strategies for PMs include incineration, recycling, landfilling, on-site stockpiling, and burning, while soil biodegradation and composting are common for BDMs [94].

Incineration offers heat and energy recovery [95], yet its efficiency relies on the calorific value of the plastic waste [27]. Mulch waste lowers calorific value due to moisture and impurities (e.g., soil) [96], necessitating removal before incineration. However, the washout of adsorbed chemicals, such as pesticides from plastic waste, may cause environmental toxicity and eutrophication. Incinerating impurity-laden plastic waste generates CO_2, sulfur dioxide (SO_2), and nitrogen dioxide (NO_2) [3]. Incineration with and without energy recovery produces different environmental impacts. Energy recovery results in avoided burdens from energy production, leading to lower fossil resource use, GHG emissions, and human toxicity potential [3,20,23,25]. However, the feasibility of energy recovery from mulch films may be questioned due to the high soil contamination in mulch waste, which can reach up to 80 % [3]. For plastic mulch waste to have a good calorific value, the soil contaminant content should not exceed 50 wt% [25,97]. Problems may also arise if the incinerator is not designed to handle such impurity levels [13]. Regarding environmental impact, the combustion of organic carbon in incinerators emits CO_2, which is released into the atmosphere, resulting in a net positive contribution to GWP, especially when the carbon is fossil-based, as opposed to bio-based carbon [98].

Landfilling is a common method of mulch waste disposal [99] due to limited mulch waste processing facilities. However, it poses environmental concerns, including land and groundwater pollution from toxic substance leaching, marine pollution from toxic wastewater runoff into water bodies, and air pollution from particulate

matter emissions, odor pollution, and emissions of gases like CH_4, CO_2, NO_x, hydrogen sulfide (H_2S), and ammonia (NH_3) [100–104]. Ultimately, landfilling causes ecotoxicity and poses critical risks to soil, plants, microbiota, animals, and humans. Although landfilling ordinarily does not recover energy, methane can be captured for energy [105]. Landfilling with energy recovery can reduce environmental impacts, such as GWP [70,105] and human health impacts [20] by offsetting the burdens of energy production. Similar to landfilling, stockpiling mulch waste can lead to adsorbed agrochemical migration into soil and groundwater through seepage triggered by irrigation or rainfall [13]. Additionally, microplastics from stockpiled waste breakage deteriorate soil health [106] and reduce the functional diversity of soil microorganisms [107].

Recycling can be either mechanical or chemical. Chemical recycling involves depolymerizing polymer structures into monomers or producing secondary valuable chemicals [108], which can be used as feedstock for new polymer production [109]. This process includes pyrolysis, gasification, and catalytic cracking [3,110]. However, chemical recycling is less commonly reported for post-use mulch waste processing than mechanical recycling. Mechanical recycling is a promising EOL strategy that involves reusing plastic film waste. It is most practical for homogeneous waste streams [111], but extensive cleaning is required to remove physical impurities (soil, dust, grease, and organic matter) [112] and chemical impurities (from fertilizers and pesticides) [113]. This cleaning process generates wastewater streams that may be hazardous due to pesticide impurities, necessitating further treatment [97]. The cleaning also requires additional energy and resources, increasing costs. Failure to treat the impurities properly may lead to ecotoxicity, eutrophication, and acidification. Moreover, mechanical recycling becomes impractical if mulch waste contains more than 5 wt% impurities [13]. Even with washing and cleaning, the adhered impurities (which depend on soil type and texture) make it difficult to produce high-quality plastic resins [3]. Therefore, mechanical pelletizing is often the final process in most mechanical recycling LCA studies.

On-site burning is an environmentally harmful practice [114], but some farmers consider it economical due to the elimination of transport costs and tipping fees [13], especially when there are no laws prohibiting open burning. Additionally, burning mulch waste with impurities such as sand, agrochemicals, and water at low temperatures (200–315 °C) results in incomplete combustion [13], releasing harmful pollutants like dioxins, particulate matter, carbon monoxide (CO), polycyclic aromatic hydrocarbons (PAHs), and furans [27,29,115]. These pollutants are highly toxic to both the environment and human health. Dioxins are carcinogenic and disrupt reproductive and immune systems [116], while particulate matter increases the risks of lung and heart diseases and causes environmental damage [117]. This highlights the need for a ban on on-site burning of mulch waste.

Soil-biodegradation and composting involve biodegradation facilitated by microorganisms, converting organic matter to CO_2 and biomass [118]. The CO_2 generated is part of the biogeochemical carbon cycle and does not contribute to increased environmental GHG emissions [119]. Rich in organic matter, compost and humus are byproducts used as soil amendments [120]. Composting may be utilized for BDMs as most BDM feedstocks are compostable under specific environmental conditions [118,121]. However, composting requires costly removal and transport to composting facilities, just like PM. Alternatively, tilling BDM waste into the soil in soil biodegradation eliminates these costs.

3.2.3.2. Environmental burdens of end-of-life strategies. Landfilling is unfavorable where there is neither energy nor material recovery [68], while incineration with no energy recovery raises GHG concerns but can mitigate ADP with energy and heat recovery [24]. Several studies have shown that mechanical recycling is preferable to mitigating major environmental impacts rather than landfilling and incineration. In Razza et al. [68], only 10 % of the PE films were considered recycled, while the rest was disposed of according to the Italian scenario of 84 % landfill and 16 % incineration with energy recovery. Mineralization was considered complete without ecotoxicity and hazardous substances for starch-based mulch. Hence, the EOL stage of PE films posed a higher negative environmental impact than BDM films. Mechanical recycling of PE mulch films had the lowest impacts in POF and NREU, while incineration showed the lowest impact in ACP, EUP, and ADP. Landfilling had the lowest impact only in GWP. These results are reasonable, as are the contexts given in Section 3.2.3.1. Overall, landfilling was identified as the least favorable EOL due to no energy or material recovery, while mechanical recycling was preferred.

Recycling was also the best scenario in Hottle et al. [70], followed by composting, and landfilling was the worst, specifically in GWP and ADP-fossils. Biopolymer (PLA and TPS) landfilling raises GWP concerns due to methane emissions. The most notable biopolymer EOL impacts resulted from material degradation, equipment, and diesel fuel used at landfills and compost facilities. The landfilling scenarios considered complete degradation of both TPS and PLA, which resulted in higher GWP due to the rise in methane emissions than composting. Meanwhile, landfilling with energy recovery could reduce this emission [105]. Other than GWP, composting of PLA and TPS showed higher impacts than landfilling in seven impact categories, including smog, acidification, carcinogenic, non-carcinogenic, respiratory effects, ecotoxicity, and fossil fuel depletion due to diesel fuel use for heavy machinery and water use associated with the composting process. PLA and TPS waste collection, transportation, and handling also substantially impacted ozone depletion and respiratory effects. For mechanical recycling, benefits over landfilling were only found in GWP and ADP-fossils from the offset from virgin plastic production, but not for ozone layer depletion and smog due to additional contributions from transportation. This shows that EOL choices matter in biopolymer environmental impacts and are influenced by energy and equipment used for the process of collection, transportation, and other operations.

Similarly, due to reduced GWP and ADP fossils, Tan et al. [77] preferred mechanical recycling over landfilling and incineration for PE films. Mechanical recycling of 14-μm PE film avoided almost half of the inherent environmental impacts, resulting in lower GWP and ADP-fossil than 10-μm PE film. However, toxicity and eutrophication are notable in recycling and littering scenarios. Incineration was better for co-generation, but POF must be managed to reduce NO_2 releases, which is one of the important constituents of photochemical smog [3]. Also, the incineration of PE films had significantly higher GWP compared to recycling and soil biodegradation [79]. Considering avoided burdens, thicker PE film offered greater benefits in material recovery from the recycling process but with higher ACP from electricity consumption in polymerization and extrusion during recycling processes. The material recovery of granules from the mechanical recycling of PE could not offset the electricity input of the mechanical recycling process but could avoid energy use in the primary granule production process.

In the study of Dong et al. [3], landfilling, incineration, recycling, and biodegradation were considered for PE film wastes. Mechanical recycling (pelletizing) was preferable due to net environmental benefits in all impact categories, except for human toxicity concerns arising from high energy use and wastewater production. In recycling processes, extensive cleaning is often required to remove major impurities, including physical impurities (soil and organic

matter) and chemical impurities (from fertilizers and pesticides) [113]. This results in additional energy consumption and wastewater generation, which may require further treatment, especially in wet cleaning processes. The wastewater often contains SS, NH_3-N, COD, and NO_x, while exhaust gas from energy consumption contains VOCs, CO, and soot. Human toxicity may also result from the release of additives into wastewater while cleaning degraded mulch waste, which can spill into waterways and be consumed by marine life, birds, and ultimately humans [24]. As a result, human toxicity is higher than for other EOL strategies. Mitigation measures must, therefore, be implemented to reduce human toxicity potential in recycling processes and meet required emission standards. These measures can include adopting cleaner energy options and treating process wastewater to standard thresholds before discharge into the environment.

Furthermore, the benefit of mechanical recycling was significant, mitigating 18.8 % of GWP, 8.99 times the ADP-fossils, and 6.40 times POF per functional unit [3]. Additionally, the net benefit of fossil fuel consumption in mechanical recycling was 2.3 times greater than incineration, although incineration (with energy recovery) still had the lowest GHG emissions and human toxicity potential due to energy recovery. Without energy recovery, however, incineration would result in very high GHG emissions since it is the only EOL process where embedded fossil carbon is released as GHGs. GHGs and photochemical pollutants, such as CO_2, SO_2, and NO_2, are also released from the combustion of PE mulch films containing impurities. Incineration caused the highest POF emissions due to the release of NO_2. Overall, mechanical recycling yielded the maximum benefit in ADP-fossils and POF but had worse human toxicity potential, while incineration had the maximum benefit in GWP and human toxicity but was worse in POF. Similar results were reported by Xiong et al. [34], where recycling had the most environmental benefits in GWP and ADP compared to landfilling and incineration due to the avoided burdens of material and energy requirements for manufacturing virgin PM films. Incineration with energy recovery also had a lower net impact than landfilling in ACP, EUP, aquatic and terrestrial ECP, and LUP due to the avoided burdens from energy production (Fig. 3).

With increasing environmental concerns and higher costs associated with EOL and disposal, complete soil biodegradation remains the most viable option for BDMs. However, the uncertainties surrounding biodegradation and soil health concerns need to be addressed to ensure optimal environmental sustainability. This is also supported by the findings of Xiong et al. [34], where in-situ biodegradation of PLA/PBAT BDMs had the best EOL scenario, showing reduced environmental impacts across all ten categories assessed (Fig. 3). Moreover, since BDMs are already more costly, avoiding additional EOL costs would make their use more economically sustainable.

From the foregoing, polymer type and impurity content are crucial for selecting sustainable EOL strategies. Mechanical recycling is generally preferred for PMs, though eutrophication and toxicity must be addressed. Human toxicity can be addressed by treating wastewater to standard limits before discharge and adopting cleaner energy options. GWP and POF in incineration must be addressed for sustainable deployment. Energy recovery in incineration is crucial to mitigating environmental impact. Soil-biodegradation is recommended for BDMs, given its low costs and the potential of composting to further exacerbate the overall environmental impact, as well as its favorable environmental impact reductions compared to PM EOL options [34]. There is, however, uncertainty about BDM degradation and its impact on soil microbial communities [122]. This is due to different feedstock combinations and additives used in BDM production, resulting in varying biodegradation periods depending on climate and soil conditions [14] and leaving in the soil mulch residues that may affect soil health to a similar degree as PM would. While some studies confirm this in the short-term, for instance, Bandopadhyay et al. [122] concluded that BDMs had a comparable impact on soil microbial communities as PE films, other studies, however, showed minor effects on soil health and that effect is time-dependent in the short-term but recommend long-term studies to ascertain the long-term effects of BDM on soil health [123–125]. No consistent effects were observed in another study across different periods and site locations [126]. Therefore, long-term studies are necessary to understand biodegradation dynamics, as the microbial community composition is influenced by the composition of BDMs [127] and under diverse natural conditions and climates. Demonstrating favorable long-term effects could encourage the widespread adoption of BDMs [124]. Advanced EOL options like pyrolysis, asphalt binding, and composite boards can also be explored.

4. Causes and impact mitigation strategies in agroecosystems

This review identifies EUP, ACP, ECP, and LUC impacts as significant limitations of BDMs. This section identifies the causes of these environmental impacts and provides feasible solutions. Fig. 4 shows an overview of the causes and sources of major environmental impacts associated with the use, manufacture, and disposal of agricultural mulch films.

4.1. Eutrophication potential mitigation

In general, EUP is often higher for bio-based materials than for conventional fossil-based materials [128], but this depends on the specific type of BDM and the life cycle stage being considered. This is primarily due to the bioplastic feedstock production processes, which involve industrial farming practices that release nutrients such as nitrogen (N) and phosphorus (P) from nitrogen fertilizers and chemical inputs, including pesticides [70,71,129–131]. Additional sources include ammonia emissions from manure applications [128] and nitrate and phosphate runoff from farmland [129,130]. Another contributing source is the EOL stages involving effluent release from municipal and industrial waste discharge [129,132], such as wastewater from mulch waste washing and landfill seepage. Organic matter release into freshwater and emissions of VOCs into the air also contribute [77].

Eutrophication leads to algal blooms in aquatic ecosystems due to nutrient (N and P) enrichment in water bodies, threatening ecosystem services [130,133–137] by depleting oxygen levels and creating dead zones in water bodies [129]. The study of Schindler et al. [138] identified P as the primary nutrient for controlling eutrophication and emphasized that reducing P levels is critical to managing algal blooms. In contrast, decreasing N alone without also reducing P is insufficient to control aquatic eutrophication [129,138]. Similarly, in the field application of mulch films, soil P levels must be maintained within the critical agronomic P threshold for optimal crop growth, which varies depending on crop type and soil conditions [139]. Exceeding this threshold can negatively impact crop yield and increase the risk of surface water eutrophication through P losses from leaching and surface runoff [140], leading to elevated environmental soil P levels [139]. Soil P levels can rise due to fertilizer application [141]. The optimal agronomic P threshold ranges between 9.30 and 30.04 mg kg^{-1}, depending on the crop type, while the environmental soil P threshold is approximately 48.56 mg kg^{-1}, depending on soil texture [139]. A eutrophic state is indicated for water bodies when total P and N levels exceed 20 and 300 mg m^3, respectively [142].

Considering the eutrophication sources discussed—particularly from fertilizer use in BDM feedstock farming processes—the

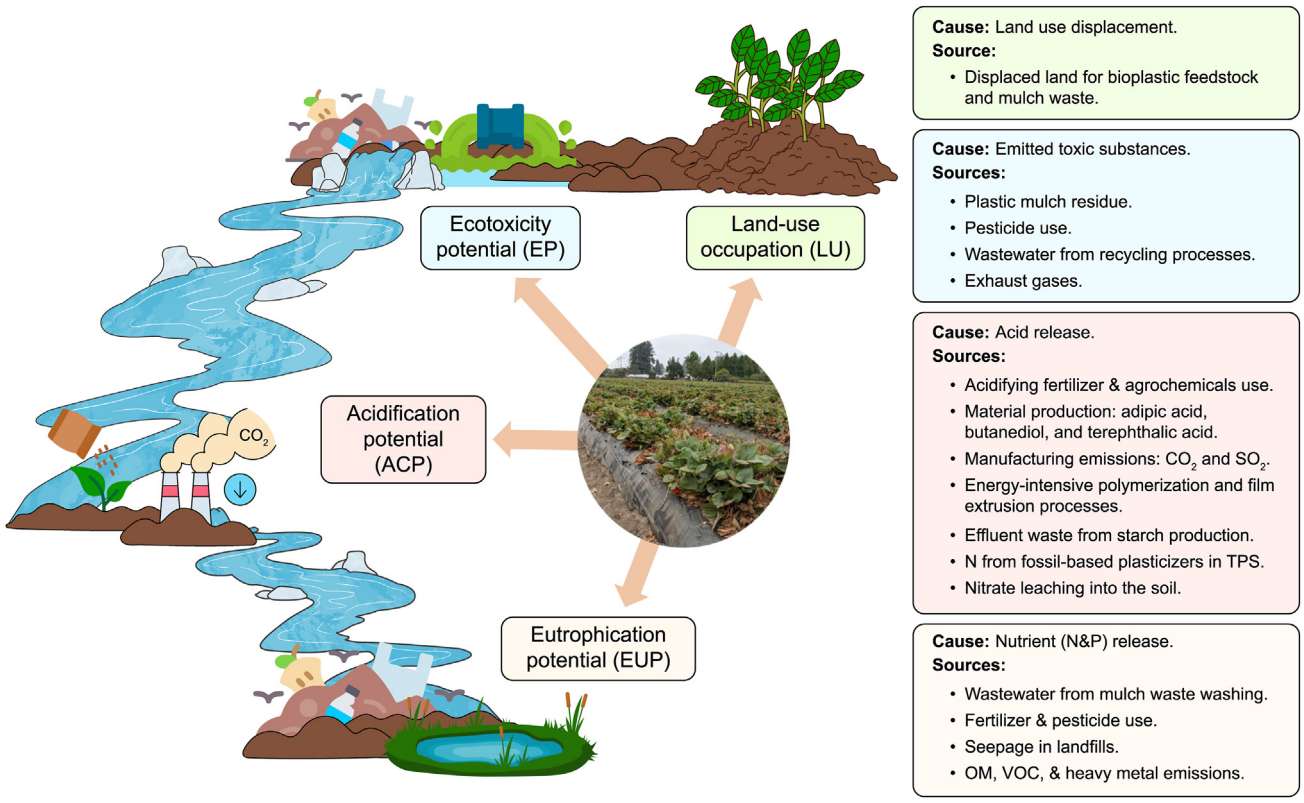

Fig. 4. Overview of the causes and sources of major environmental impacts associated with mulch film manufacture, use, and disposal/end-of-life. The colors of the text boxes correspond to different environmental impacts. CO_2: carbon dioxide; SO_2: sulfur dioxide; N: nitrogen; P: phosphorus; TPS: thermoplastic starch; OM: organic matter; VOC: volatile organic matter.

potential for eutrophication (EUP) can only be managed, not eliminated. Addressing eutrophication at both the EOL stage of PM and the manufacturing stage of BDMs is crucial for maximizing environmental benefits. The ultimate solution lies in reducing nutrient emissions, specifically N and P [129]. Proposed strategies for mitigating EUP include:

(a) Reducing P levels in bioplastic feedstock production farming processes [138].
(b) Optimizing fertilizer application to match crop-specific nutrient requirements, soil conditions, and their assimilation capacity.
(c) Minimizing wastewater emissions by increasing water reuse cycles and promoting cleaner production practices during plastic granule pelleting. Granule pelleting contributed significantly to eutrophication emissions, accounting for 53.5 % of the total [77].
(d) Blending starch polymers with biodegradable polyesters, as suggested by Shen [41], can mitigate eutrophication in TPS film production. This is probably due to the lower EUP of polyester-based polymers like PBAT [77]. However, non-biodegradable copolymers in bioplastics may increase energy demand and CO_2 emissions [39].
(e) Utilizing reclaimed starch instead of virgin starch for granule production [69].
(f) Adopting EOL strategy with limited secondary wastewater stream, such as incineration with energy recovery. Compared to mechanical recycling or landfilling, this approach can lower EUP and ACP impacts, which generate secondary wastewater. Incineration was reported to have the lowest EUP and ACP [68].
(g) Improving EUP modeling accuracy in LCAs by employing consistent inventory modeling practices and enhancing methodological transparency [69].

4.2. Acidification potential mitigation

The causes of acidification are closely related to those of eutrophication but stem from acids released during mulch production, use, and waste management. Point sources include fertilizer manufacturing [70], emissions from field use of fertilizers, agrochemicals, and manure applications [39,128,131], material production, energy-intensive polymerization, and film extrusion processes [77], and biomass combustion for cogeneration [131]. Fertilizer is manufactured using chemicals like phosphorous, diammonium phosphate, urea, and ammonium nitrate, which can contribute to acidification [70]. Additionally, effluent wastes from starch production and additional nitrogen from fossil-based plasticizers in TPS further exacerbate acidification [70]. Acidifying fertilizers like ammonium sulfate and urea contribute to soil acidification [143]. Likewise, acid gas emissions such as CO_2 and SO_2 from manufacturing processes, fossil fuel and energy consumption during film production and esterification of petrochemicals, and basic raw material manufacturing (e.g., adipic acid, butanediol, and terephthalic acid) collectively contribute to acidification [77]. SO_2 drives acid rain formation, which has negative impacts on agroecosystems [144].

In the context of agroecosystems, acidification can occur in soil and freshwater. Soil acidification can hamper plant growth if the species are not adapted to or require acidic soil conditions (e.g., blueberry, cranberry), which can, in turn, limit root growth and

uptake of water and nutrients, lower soil fertility, diminish rhizobia populations, and mobilize toxic metals into the food chain depending on soil pH [143]. ACP can be mitigated by minimizing nitrate leaching into the soil, preventing soil erosion, limiting acidifying fertilizer use unless required by the crop, minimizing plasticizers/additives in BDMs, and adopting energy- and material-efficient film production processes. Suitable EOL strategies, such as incineration, also mitigate acidification by reducing the release of acidifying substances from mulch waste.

4.3. Ecotoxicity mitigation

Plastic toxicity to human, marine, and freshwater life has been attributed to the substantial volume of plastic residues contaminating soil and water bodies [77], particularly with PMs, which take hundreds of years to fully degrade, causing long-term adverse effects on terrestrial and aquatic agroecosystems and organisms within them [6,81–83,113]. Importantly, standards like EN 17033 [145] require meeting certain criteria for ecotoxicity, e.g., toxic effects on plants, invertebrates, and microorganisms. Ecotoxicity has been linked to pesticide use [76], exhaust gases containing CO, VOCs, and soot [3], and wastewater from recycling processes containing suspended solids (SS), ammonia nitrogen (NH_3-N), NO_x, and chemical oxygen demand (COD).

Efficient PM removal after cultivation and use of fast-degrading BDMs are vital to reduce plastic residues and mitigate ecotoxicity in agroecosystems. Optimizing the biodegradable component of copolymer blends and pesticide use is necessary to minimize ecotoxicity impacts. Also, adopting EOL strategies that reduce wastewater production is effective for toxicity mitigation.

4.4. Land-use occupation mitigation

Another significant environmental footprint is land-use occupation potential (LUP), which can occur as either direct land use change (dLUC) or indirect land use change (iLUC). dLUC results from the conversion of preserved lands, such as forests, into arable land for biomass feedstock production. This conversion releases soil organic carbon as CO_2. However, as per PAS2050 [146], CO_2 emissions from dLUC are only considered in LCA if the land-use change occurred within 20 years of the study. Land-use changes older than this threshold are considered negligible. In contrast, iLUC refers to land-use changes driven by the expansion of current farmland to accommodate the growing demand for food, feed, and bioplastic feedstock.

Displacing agricultural land for bioplastic feedstock cultivation leads to land-use change, risking global food, water, and energy security [147,148], and contributes to climate change emissions [149]. This is in addition to the agricultural land use stress already being imposed by bioenergy feedstocks. Such land-use change is a significant contributor to the negative LUP impact of starch-derived BDMs, as highlighted in Dinkel et al. [76]. LUP impact can be mitigated by (a) utilizing existing agricultural lands productively, (b) producing non-food bioplastic feedstock on degraded lands [128], and (c) producing bioplastic feedstock from high-yield crops [150]. These strategies can minimize the strain on agricultural land requirements. A careful balance is therefore needed to ensure sustainable practices that address agricultural mulching needs while conserving essential resources for food and energy security.

5. Factors affecting environmental burdens of mulch films

The environmental burdens of mulch films vary by polymer type, life cycle stages, farming practices, additive content, film thickness, and climate (Fig. 5).

5.1. Life cycle stages

Various LCA studies reveal that specific life cycle stages disproportionately contribute to environmental impacts. The manufacturing stage, covering resin, granule, and film production, stands out as a dominant contributor to the overall environmental impact. Razza et al. [68] and Shen [41] highlighted the critical contributions of the granule production and polymer/material manufacturing phases to environmental impacts, particularly regarding NREU and GWP. Yield is often comparable for PM and BDM use stages [92,151]. Plastic residues after harvest pose risks to soil health, plants, and organisms, as well as other organisms that ingest them [30,32,81,82,86,89]. The post-use stage, particularly for PMs, plays a notable role in the overall impact. The impact at the post-use stage is influenced by several factors, including the type of mulch waste, the level of impurity content, the methods used for cleaning mulch waste and treating impurities, and the chosen EOL strategies.

5.2. Farming practices

Farming practices significantly contribute to the overall field use stage of mulch films, particularly to EUP, ACP, ECP, and stratospheric ozone depletion [128]. These practices include, for example, fertilizer, manure, and pesticide applications. Fertilizer application is crucial to crop yield and soil improvement but contributes significantly to the environmental impact of both PMs and BDMs. However, as noted in Section 4, fertilizer production and excessive application contribute to eutrophication [39,70,71,129,130], acidification [39,70], and GHG emissions [152,153]. Nitrate and phosphate emissions from nitrogen fertilizers, along with ammonia emissions from manure, contribute to EUP, while manure and mineral fertilizers contribute to ACP [128]. Additionally, pesticide use contributes to ECP [76]. Additionally, according to Xiong et al. [23], chemical fertilizer input accounted for about 60 % of the total environmental impact due to N_2O and ammonia (NH_3) emissions into the air, nitrate (NO_3^-) and phosphorus (P) release into groundwater, and heavy metal release (e.g., As, Cu, Cd, Zn, Pb) into the environment from fertilizer applications [23,154,155]. Growers

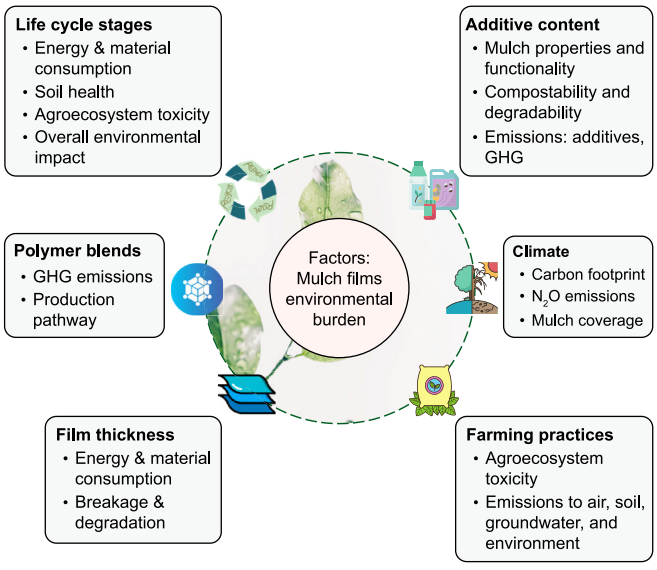

Fig. 5. Factors affecting environmental burdens associated with mulch films. Bulleted points indicate where these factors impose an environmental burden throughout the mulch film life cycle. GHG: greenhouse gas, CO_2: carbon dioxide, N_2O: nitrous oxide.

should optimize fertilizer use to only apply what is needed and assimilated by the crop to mitigate EUP, ACP, and N_2O release.

5.3. Polymer blend grades

The composition of plastic polymers in polymer blends contributes to environmental burdens by influencing the overall production pathway. For example, Broeren et al. [69] investigated various film grades of starch blends with PLA, PBAT, PBS, PHB, recycled PLA (re-PLA), and fiber and reported that minimizing PBAT and PBS while maximizing starch, natural fibers, and mineral fillers resulted in an optimal 85 % GHG emission reduction. However, certain grades showed an 80 % increase in environmental impact compared to their petrochemical counterparts on an equal-weight basis. Consequently, the environmental implications of starch plastic blend production vary significantly from one grade to another, emphasizing the need to limit PBAT and PBS in co-polymer blends for environmental sustainability. Polymer blend grades also contribute to production costs. Xiong et al. [34] observed that blending PLA with PBAT can reduce both the biodegradability and production costs of PBAT-based products.

5.4. Additive content

Mulch additives play a crucial role in enhancing film properties and overall functionality [156]. Additives include UV stabilizers, which protect against UV-induced polymer breakdown; anti-fog agents, which minimize the formation of water droplets and improve light transmission and transparency; colorants and pigments to achieve specific material colors or film properties; etc. [157–159]. Film color influences solar radiation absorption or reflection, affecting mainly soil temperature as well as weed suppression and canopy temperature [160]. Additionally, compatibilizers are employed to improve the compatibility between different polymers in a polymer blend [161], while plasticizers, e.g., polyols, amine, amide, citrate, etc., enhance flexibility, durability, and workability [43,44,162]. The selection of additives is contingent upon the desired characteristics of the mulch film and the environmental conditions it will encounter in agricultural settings.

Despite the specific roles of additives, minimizing additive use in BDM production is recommended, as additives may promote or reduce the compostability and degradability of mulch film [39,163,164]. For example, the incorporation of natural fillers (carbon black, silica rice ash, and organic fertilizers) was found to improve the biodegradation of PBAT/PLA BDM [165], while UV stabilizers can reduce the photodegradation of PBAT BDMs [166,167]. Moreover, additives can contribute up to 46 % of GHG emissions in starch plastic production and represent a substantial portion of starch plastic weight [69]. Specific emphasis has therefore been placed on reducing reliance on them, especially compatibilizers [69]. While rapid fragmentation in BDMs facilitates additive release [168], this concern is not limited to BDMs but also exists for PMs [88]. Also notable is the concern about the migration of additive chemicals from films into agroecosystems [42,169] and the absorption by crops and migration into the food chain [24,88]. An example is phthalates (PAEs), an additive that can be released into the soil as a contaminant through leaching, migration, and abrasion [88,170]. Meanwhile, phthalates were once used but are no longer used due to human health concerns [171].

5.5. Film thickness

Mulch film thickness was found to be a crucial factor affecting GWP, ADP-fossils, and ACP [77]. 14-μm PE film showed higher contributions than a 10-μm film due to increased energy and raw material consumption in manufacturing. Notably, covering 1 km^2 of land with a 14-micron-thick film requires 1.4 times the mass of a 10-micron-thick film, leading to elevated energy and material input during manufacturing and transportation processes [77]. Specifically, the 14-μm PE film consumed 48.9 % more fossil resources. Additionally, 70 % of 10-μm PE film residue remained in the soil, posing additional threats to soil and environmental health as a pollutant, whereas only 10 % of waste from 14-μm films went uncollected. This difference is because thicker films are less prone to breakage and possess higher strength and suitability for recycling, although energy and material requirements are much higher [3,34].

5.6. Climate

Environmental impacts of mulch films can vary depending on the farming season, particularly for seasonal crops like maize and wheat [92,152,172]. Climate, or the overall weather patterns in a geographic region, is important. In semiarid regions experiencing heavy rainfall, it is advised to avoid mulching practices for maize cultivation due to the significant increase in net GWP and carbon footprint associated with PM and BDM compared to scenarios with no mulch [92]. Emissions tend to be higher during summer compared to winter, primarily due to elevated nitrogen fertilizer applications [152]. Furthermore, regions with higher humidity typically exhibit less mulch coverage, thereby influencing the degree of environmental impacts related to the utilization and disposal of mulch films [3]. Climates with dryer seasons during key times of crop production and/or cooler soil temperature conditions are more likely to benefit from the effects mulches have on soil microclimates.

6. Agronomic, environmental, and economic considerations of mulch choices

6.1. Environmental and agronomic tradeoffs in long- and short-term

From an agronomic perspective, both BDM and traditional PM have comparable performance in improving crop yield relative to unmulched crops [35,36,91,173]. Due to these agronomic benefits, farmers base their decisions on short-term benefits rather than long-term consequences [88]. From an environmental standpoint, BDMs offer some inherent advantages over PMs despite some negative impacts that need addressing. Meanwhile, the major environmental concerns currently hindering the widespread adoption of BDM are the uncertainty of degradation in agricultural soils and its effect on soil health [122,174–176]. There is, therefore, a consensus for long-term studies on the effect of BDM microplastics on soil health [123–127].

6.2. Economic considerations

On the economic front, the higher perceived cost of BDMs [39] makes traditional PMs more appealing to farmers, hindering the widespread adoption of the former [77,175]. The purchase cost of BDMs was reported to be approximately three times higher than that of PMs [92] or twice as expensive [177,178], depending on factors such as polymer feedstock, film thickness, location, supplier, and availability [164,179,180]. However, BDMs may be more economical than PMs when a holistic economic feasibility analysis is conducted, taking into account costs associated with labor, removal, and disposal of PMs. Labor costs are influenced by geographical location and vary within and between countries. For instance, wage rates in the United States differ across states, meaning the significance of labor costs in reducing overall BDM

costs can vary. Disposal costs in the United States and Europe are minimal, representing only a small percentage of the total PM use cost [164,180,181]. However, the removal costs for PMs are significant [164], particularly in regions with high labor costs. Thus, the removal and disposal costs could provide a major cost-saving benefit for adopting BDMs. Disposal costs are similarly minimal in China, as evidenced by the industry disposal cost data from Dong et al. [3]. However, the data indicates that even when considering the costs of landfilling, incineration, and recycling for PE films, it remains more affordable than BDM in Beijing, China, although the removal costs were not included. This highlights the importance of geographical factors in evaluating the cost feasibility of BDMs, which should be considered in economic feasibility analyses.

Despite the cost difference between BDM and PM, survey data indicates that some consumers, particularly in the United States, are willing to pay a 10 % premium for a 0.5-kg box of strawberries grown with biodegradable mulch if specific criteria are met [7,182]. However, farmers prioritize price premiums over soil health improvement advocated by crop advisors [182]. Insufficient knowledge, the unfavorable aesthetics of BDM fragments on farms, and durability concerns also contribute to risky perceptions of BDMs among growers [174,175,183]. In addition, concerns about field contamination regarding food safety compliance and by rotation partners could be other barriers.

It is, therefore, clear that BDMs appear less economically attractive, especially to growers. Besides addressing the environmental impact contributors, the significant limiting factor of the perceived cost of BDMs must also be addressed. Until the cost becomes viewed more favorably, farmers may find it economically unjustifiable to adopt BDMs, considering that yield is often comparable with PM. Legislative policies, government subsidies, and financial incentives supporting the production and use of BDMs can help address cost barriers, promote widespread adoption, and offset initial costs for farmers. Cost-minimizing efforts should also be put in place by manufacturers without compromising effectiveness and quality. Lastly, major awareness campaigns are needed to educate farmers about the cost-saving benefits of BDMs in relation to the labor, removal, and disposal costs associated with PMs.

7. Future perspectives and recommendations

7.1. Recommendations

For better sustainability of BDM and PM films throughout their life cycle, the following recommendations are provided to address prevailing challenges.

(1) **Economic sustainability of BDM:** While the purchase costs of BDMs are generally higher than those of PMs, depending on polymer feedstock, film thickness, location, supplier, and availability, there are significant cost-saving benefits in terms of removal and disposal, particularly where labor is expensive, and tipping fees are high. We recommend conducting holistic, geographically specific economic assessments and launching awareness campaigns to highlight the cost-saving benefits of BDMs. Legislative policies, government subsidies, and financial incentives supporting the production and use of BDMs can help offset initial costs for farmers, address perceived cost barriers, and promote widespread adoption.

(2) **Eutrophication mitigation:** Eutrophication is a significant concern in the manufacturing of bioplastic feedstocks and the EOL stage of PMs. To mitigate the EUP impact in BDMs, we recommend the following practical applications: (a) reduce phosphorus (P) levels in bioplastic feedstock farming, (b) optimize fertilizer application, and (c) encourage the use of reclaimed starch instead of virgin starch for starch-based plastic production. For EUP mitigation in PM EOL, we suggest (a) increasing water reuse cycles in cleaning processes and treating toxic wastewater before discharge and (b) adopting cleaner EOL strategies with limited secondary wastewater streams.

(3) **Sustainable PM waste collection and management:** Traditional PM waste poses challenges in terms of collection and disposal costs and environmental impacts. In the absence of widespread facilities for agricultural plastics processing [7], there is a need to integrate PM waste collection and processing into existing municipal waste facilities to reduce costs and streamline logistics. Effective cleaning strategies are also needed to minimize ecotoxicity during waste mulch processing. This can involve preprocessing waste treatment to reduce the impact of waste impurities. To manage human toxicity in the mechanical recycling process, wastewater emissions should be minimized by treating wastewater to standard thresholds before discharge into the environment.

7.2. Future perspectives

While the complete adoption of BDMs could be more sustainable and desirable, perceived costs and uncertainties related to complete soil biodegradation and their impact on soil health remain significant barriers to widespread adoption. Long-term studies are needed to investigate biodegradation dynamics under diverse natural conditions and climates to assess the impact of BDM microplastics on soil health and the environmental health of surrounding landscapes. Research aimed at simplifying production costs and framing potential cost savings in the context of reduced expenses from mulch removal, transport, and disposal fees should be pursued and communicated to stakeholders. Additionally, research on accelerating rapid and complete biodegradation to minimize mulch residues in soil is necessary, particularly in systems with short rotations or where land is leased. Legislative policies and government incentives to support the production and use of BDMs are also essential to encourage their adoption among farmers.

Although starch-derived BDMs offer inherent benefits, some BDMs, such as PLA, are not ideal solely for mulch film production due to their high glass transition temperature, which hinders biodegradation under ambient conditions, and their hardness, which leads to brittleness and lower thermostability [184]. Structural enhancement through co-blending with other polymers or the use of additives is often required to improve their suitability. Further research is necessary to optimize the material properties of these biopolymers for BDM production. Additionally, technological advancements are needed to increase the bio-based content of polyester-based BDMs.

In the LCA of BDMs, the impact of mulch degradation is often neglected, or complete mineralization is assumed due to a lack of inventory data or for simplicity in defining system boundaries. However, for a holistic impact assessment, the consideration of plastic residues is essential, especially for BDMs that degrade within a short period. With the environmental impact of microplastics attracting widespread concern, incorporating microplastic quantification into LCA methodologies is crucial for a comprehensive evaluation of mulch film environmental impacts, but it remains a challenge. Currently, protocols are being proposed [185,186], and a methodology for the inclusion of plastic pollution in life cycle assessment has been developed [187]. Furthermore, research

aimed at reducing PM film fragmentation and the residence time of BDM microplastics in the soil should be encouraged.

8. Conclusions

A detailed review of comparative LCA studies has been conducted. This highlights the complex relationship between environmental and economic factors in the manufacture, use, and EOL of PMs and BDMs. While each reviewed LCA study strives to conduct a balanced comparative analysis, the lack of suitable data necessitates reliance on different data sources and causes wide uncertainties. This limits LCA scope and leads to varying assumptions. Differences in modeling approaches, geographical and climatic conditions, and the scope and context of various comparative LCA studies make it difficult to draw general conclusions for all impact categories. Here are the key findings of this study.

(a) The material manufacturing stage has a significant environmental impact contribution, with BDMs showing lower energy consumption and GHG emissions but higher land-use occupation potentials than PMs. Eutrophication and acidification potentials are less consistent across the board and depend on specific feedstocks and the scope of assessment of BDM and EOL of PM.
(b) Environmental burdens vary with polymer types, with polyester-based BDMs having higher environmental burdens than starch-based BDMs due to substantial material and energy consumption. Starch-based BDMs also exhibit lower impacts compared to PLA-based ones.
(c) Both BDMs and PMs increase yield in field applications compared to crops grown without mulch.
(d) EOL choices are critical, particularly for PMs. Optimal choices involve recovering heat, energy, and/or materials, with mechanical recycling typically preferred. However, addressing concerns such as eutrophication and toxicity is vital. GWP and POF in incineration must be addressed for sustainable deployment. Soil biodegradation is the most economical for BDM post-use, but the emphasis should be on reducing residue residence time in soil. Composting is not the intended EOL for BDMs as it adds costs associated with removal, and it has the potential to further exacerbate overall environmental impacts. Other traditional EOL methods of PM were also less environmentally sustainable for BDM post-use.
(e) Having displayed significant advantages, starch-plastic BDMs are recommended to replace PMs, while pure polyester-based bioplastics can be co-blended with other polymers.
(f) Despite BDMs appearing environmentally advantageous, their higher production costs make them less economically attractive. Meanwhile, the cost-reducing benefits from removal and disposal attributed to PM may offset costs, especially where labor is expensive and tipping fees are high. Therefore, the perceived costs of BDM must be addressed for widespread adoption.

The decision between PM and BDM, therefore, requires a holistic assessment that considers environmental and economic factors within specific contextual nuances linked to the particularities of given cropping and soil systems. Until perceived costs become favorable and the overarching uncertainties regarding the impact of mulch film residues on soil health and adjacent landscapes are addressed, the widespread adoption of BDMs will likely remain a topic of debate.

CRediT authorship contribution statement

Oluwatunmise Israel Dada: Writing – original draft, Visualization, Methodology, Formal analysis, Conceptualization. **Teshan Udayanga Habarakada Liyanage:** Resources. **Ting Chi:** Writing – review & editing, Investigation. **Liang Yu:** Writing – review & editing, Supervision. **Lisa Wasko DeVetter:** Writing – review & editing. **Shulin Chen:** Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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