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Plastics used in agriculture and for food:
uses, properties and impacts

Condensed report of the collective scientific assessment – July 2025

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Condensed report of the collective scientific assessment

Plastics used in agriculture and for food: uses, properties and impacts

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Abbreviations and acronyms

ACTA	Agricultural Technical Coordination Association
AD	anaerobic digestion
ADIVALOR	*Agriculteurs, Distributeurs, Industriels pour la Valorisation des déchets agricoles*
AGEC	against waste and for a circular economy / *anti-gaspillage pour une économie circulaire*
APCA	Permanent Assembly of Chambers of Agriculture or *Assemblée permanente des Chambres d'agriculture*
ASTM	Advancing Standards Transforming Markets
ATBC	acetyl tributyl citrate
ATR-FTIR	attenuated total reflectance with Fourier transform infrared spectroscopy
BADGE	bisphenol A diglycidyl ether
BAU	business-as-usual
BBP	butylbenzyl phthalate
BET	Brunauer-Emmett-Teller
BFR	brominated flame retardant
bio-PET	bio-polyethylene terephthalate
BPA	bisphenol A
bw	body weight
CAP	Common Agricultural Policy
Cd	cadmium
CICES	Common International Classification of Ecosystem Services
CPA	French Committee for Plastics in Agriculture or *Comité des plastiques agricoles*
CTC	chlortetracycline
CTCPA	*Centre technique agroalimentaire*
DBP	di-n-butyl phthalate
DEHP	di(2-ethylhexyl) phthalate
DfS	design for sustainability
DIBP	di-isobutyl phthalate
DIDP	di-isodecyl phthalate
DINP	di-isononyl phthalate
DRS	deposite-return schemes, also known as deposit-refund schemes
dsDNA	double stranded deoxyribonucleic acid
EC	European Commission
EDC	endocrine disrupting chemical
EFSA	European Food Safety Authority
EG	ethylene glycol
EKC	environmental Kuznets curve
EPR	extended producer responsability
EPS	expanded polystyrene
ESI	electrospray ionisation
EU	European Union
EVA	ethylene-vinyl acetate
EVOH	ethylene-vinyl alcohol
FCM	food contact material
FDP	fossil depletion potential
FFF	field-flow fractionation

FNE	*France Nature Environnement*
FTIR	Fourier transform infrared spectroscopy
GABA	gamma-aminobutyric acid
GC	gas chromatography
GDP	gross domestic product
GPI	genuine progress indicator
GWP	global warming potential
HDI	human development index
HRMS	high resolution mass spectrometry
IAS	intentionally added substances
IBD	inflammatory bowel disease
ICP-MS	inductively coupled plasma mass spectrometry
ICP-OES	inductively coupled plasma optical emission spectroscopy
INC	International Negotiation Committee
IPC	Industrial Technical Centre for Plastics and Composites
IR	infrared
ISO	International Organisation for Standardisation
LC	liquid chromatography
LCA	life-cycle analysis
LNE	*Laboratoire national de métrologie et d'essais*
LOAEL	lowest observed adverse effect level
MALS	multi-angle light scattering
MaPL	macro-plastic
M-ARCOL	mucosal artificial colon
MNPL	micro- and nano-plastic
MPL	micro-plastic
MRF	material recovery facility
MS	mass spectrometry
M-SHIME®	Mucosal-Simulator of the Human Intestinal Microbial Ecosystem
MSW	municipal solid waste
NGO	non-governmental organisation
NIAS	non-intentionally added substances
NIR	near infrared
NP	nonylphenol
NPL	nano-plastic
OECD	Organisation for Economic Co-operation and Development
OPECST	Parliamentary Office for evaluation of scientific and technological options
ORP	organic residual product
PA	polyamide
PAA	primary aromatic amines
PAH	polycyclic aromatic hydrocarbon
PAN	polyacrylonitrile
PAR	photosynthetically active radiation
PBAT	polybutylene adipate terephthalate
PBDE	polybrominated diphenyl ether
PC	polycarbonate
PCB	polychlorinated biphenyl
PCL	polycaprolactone

PE	polyethylene
PES	poly(ethylene succinate)
PET	polyetehylene terephthalate
PHA	polyhydroxyalkanoate
PHB	polyhydroxybutyrate
PHBV	polyhydroxybutyrate-co-valerate
PHE	PET hydrolic enzyme
PLA	polylactic acid
PMMA	poly(methyl methacrylate)
POP	persistent organic pollutant
PP	polypropylene
PPH	polypropylene homopolymer
PPP	plant protection product
PRO	producer responsibility organisation
PS	polystyrene
PTFE	poly(tetrafluoroethylene)
PUR or PU	polyurethane
PVAc	polyvinyl acetate
PVC	polyvinyl chloride
PVDC	polyvinylidene chloride
PVOH	polyvinyl alcohol
Py-GC/MS	pyrolysis-gas chromatography/mass spectrometry
QA/QC	quality assurance and quality control
QTOF	quadrupole time-of-flight
ROS	reactive oxygen species
$scCO_2$	supercritical CO_2
SRP	the French national Syndicate of Plastics Regenerators or *Syndicat National des Régénérateurs de matières Plastiques*
STIRPAT	STochastic Impacts model by Regression on Population, Affluence, and Technology
SUP	single-use plastic
TBBPA	tetrabromobisphenol A
TED-GC/MS	thermal extraction desorption-gas chromatography/mass spectrometry
T_g	glass transition temperature
U.S.	United Sates
UDW	unsound disposal of waste
UK	United Kingdom
UNEA-5	Fifth United Nations Environment Assembly
UNEP	United Nations' Environment Program
USA	United States of America
UV	ultraviolet
VALHOR	French organisation for ornamental horticulture and landscape
VOC	volatile organic compound
WVP	water vapour permeability

1. Why a collective scientific assessment on plastics used in agriculture and for food?

Because of low cost and many other interesting properties, plastics production and consumption have increased since the 1950s in many economic sectors (Figure 1A). Plastics production reached 5.5 million tonnes (Mt), 54.0 Mt and 413.8 Mt in 2023 in France, Europe and worldwide, respectively, with plastics in food value chains accounting for about a fifth of plastics consumed (Table 1; Yates *et al.*, 2025). Based on available data, plastics are mainly and widely used downstream of food value chains, *i.e.*, beyond the farm gate, to package food and drink (91, 80 and 86% of plastics consumed in food value chains in France, Europe and worldwide, respectively). On farms, plastics mainly contribute to (i) forage conservation as stretch films to wrap haylage bales, silage films to cover silage, twines and bale nets to tie hay bales; and to (ii) protected cultivation as greenhouses, small tunnels, mulching films, pipes and frameless direct covers (2.4.3). More than half of agricultural plastics were used in cattle systems in Europe in 2019, the other 45% being used in horticultural systems. The proportion of agricultural plastics used in cattle systems is even higher in France and reached 73% in 2023.

The growing production and use of plastics has been accompanied by increases in plastic waste generation and pollution (Figure 1B to 1D). This pollution is much more than visual pollution and is not only due to mismanaged plastic waste. It also stems from the ability of plastics to deteriorate, fragment, disperse in the form of particulate plastics (Lebreton *et al.*, 2019; Magnin *et al.*, 2020), transfer their constituents, and adsorb and then desorb contaminants (Cook *et al.*, 2023) from their production, during and after their use.

The awareness of plastics and plastic pollution not being biochemically inert, of the extent of marine plastic pollution and its mainly continental origin (Winterstetter *et al.*, 2023), as well as of resources needed and emissions associated with plastic production and plastic waste management is rising. In response, plastic regulations evolved in an attempt to protect human health and the environment in addition to economic freedom (Figure 2).

Among international agreements addressing plastic impact on the environment are the London Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter of 1972, replaced by the London Protocol in 1996, MARPOL International Convention for the Prevention of Pollution from Ships of 1973 and the Paris Agreement on Climate Change of 2015. Among international agreements addressing plastic impact on human health explicitly and the environment are the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal of 1989, the Stockholm Convention on Persistent Organic Pollutants (POPs) of 2001, aiming at eliminating the production and use of certain persistent toxic chemicals, and the Minamata Convention on mercury of 2013, restricting the use of mercury and mercury compounds including in processes required for plastic production. More recently, in 2022, the United Nations Environment Assembly adopted a resolution to convene an Intergovernmental Negotiating Committee and develop an international legally binding instrument on plastic pollution, noting with concern its negative impact on the environmental, social and economic dimensions of sustainable development, including its related risks to human health and adverse effects on human well-being (UNEP, 2022).

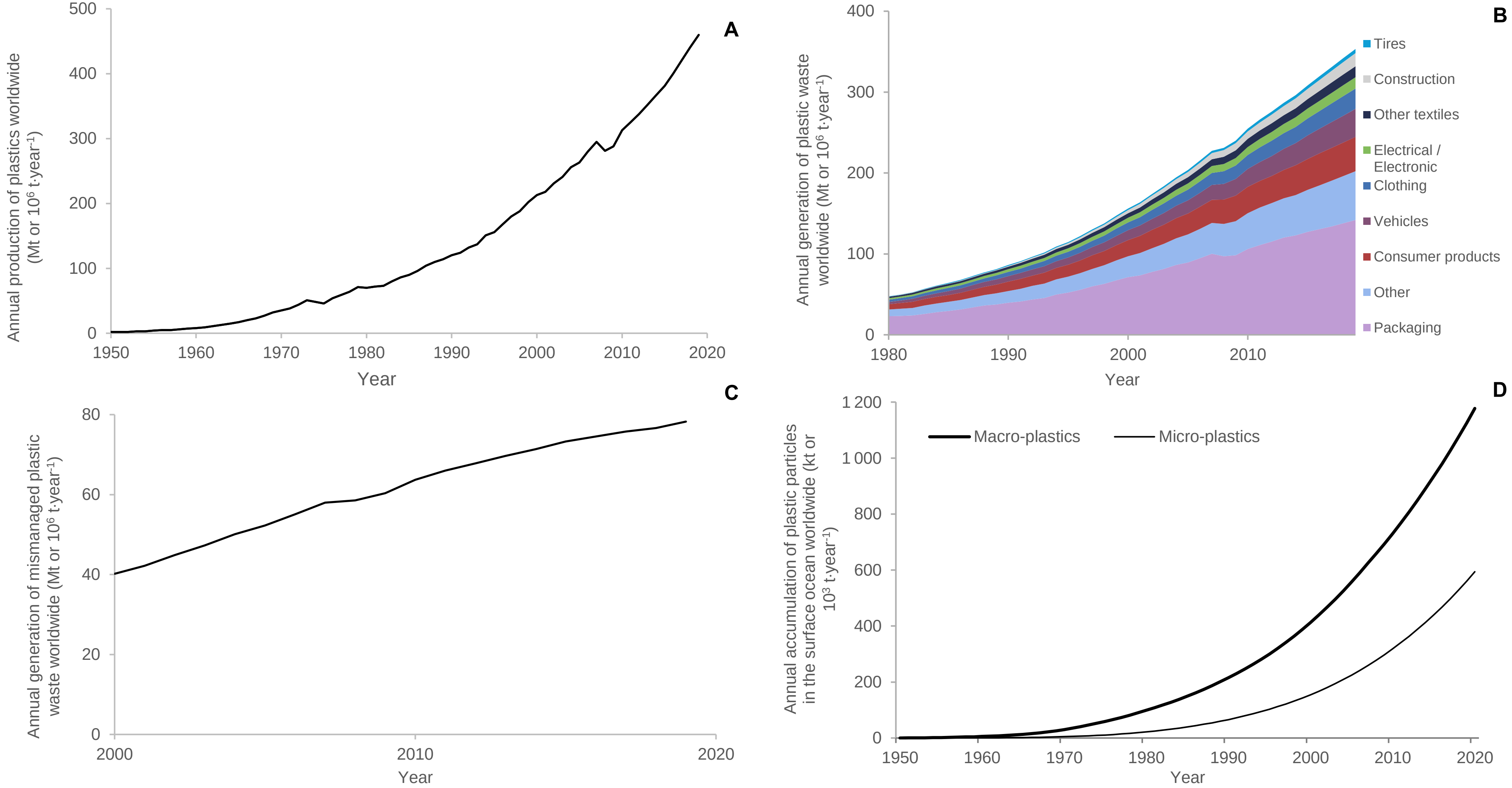
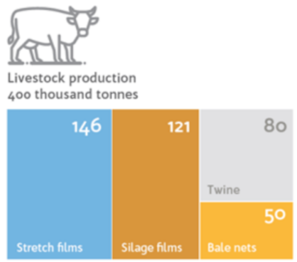
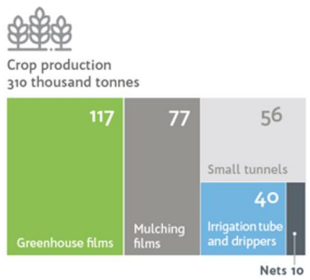


Figure 1. A. *Annual production of plastics worldwide from 1950 to 2019. Data: Geyer et al. (2017); OECD (2022) – with major processing by Our World in Data.* ***B.*** *Annual generation of plastic waste worldwide from 1980 to 2019, by application. In this dataset, applications in agriculture and for food are not made specific categories. Plastic food packaging waste are included in the 'Packaging' category. Waste from other uses for food may be included in the 'other' or 'electrical/electronic' category. Waste from uses in agriculture may be included in the 'other' category. Data: OECD (2022), retrieved by Statista, 2024.* ***C.*** *Annual generation of mismanaged plastic waste worldwide from 2000 and 2020. In this dataset, mismanaged plastic waste includes materials burned in open pits, dumped into seas or open waters, or disposed of in unsanitary landfills and dumpsites. Data: OECD (2023) – processed by Our World in Data.* ***D.*** *Annual accumulation of macro- (>0.5 cm) and micro-plastics (<0.5 cm) in the surface ocean worldwide from 1950 to 2020. In this dataset, nano-plastics (< 1 µm) are not distinguished from micro-plastics. Data: Missing plastic budget (Lebreton et al., 2019) – processed by Our World in Data.*

***Table 1.** Contextual data on plastics, mostly from the grey literature and estimated by actors of plastic value chains. As from different references, data are not necessarily consistent with each other. Because of limitations (3.3.1), these 'grey' data are not validated. However, in the absence of other data, they were used to contextualise the content of the scientific literature. Data on plastics used for food other than packaging (e.g., plastics coming into contact with food during food production or processing, kitchenware and tableware) were not available.*

	Global	Europe	France
Plastic production, *i.e.*, manufacture of plastics in primary forms, before the manufacture of plastic products (Mt·year^{-1})	In 2023: Total **plastic production**[1]: 413.8 of which: Petroleum-based: 374.2 Mechanically and chemically recycled (post-consumer): 36.5 including 0.3 chemically recycled (post-consumer) Bio-based and bio-attributed: 3 Carbon-captured: 0.1 In 2019: Primary **plastic production by polymer**[2]: Polyethylene (PE) 109.84 including High-density polyethylene (HD-PE) 55.54 and Linear low-density polyethylene and low-density polyethylene (LLD-PE, LD-PE) 54.30 Polypropylene (PP) 72.81 Polyvinyl chloride (PVC) 51.39 Polyethylene terephthalate (PET) 24.92 Polystyrene (PS) 21.12	In 2023: Total **plastic production**[1]: 54 of which: Petroleum-based: 42.9 Mechanically and chemically recycled (post-consumer): 7.2 including 0.1 chemically recycled (post-consumer) Bio-based and bio-attributed: 0.8	In 2023: Total **plastic production**[3]: 5.51 Petroleum-based[1]: 4.80 Mechanically and chemically recycled (post-consumer)[1]: 0.63 Bio-based and bio-attributed[3]: 0.08
Plastic consumption (or marketed) (Mt·year^{-1})	Total plastic consumption: not available In 2023: **Packaging**[4]: 144.83 **Food and beverage packaging**[5]: 78.25 In 2019: **Plastic products in plant and animal production**[6]: 12.5	In 2022, in the EU27 + Norway + Switzerland + the United Kingdom: Total **plastic consumption**[7]: 53.3 34.7% for **packaging**[7]: 18.5 18.8% for **food and beverage packaging**[8]: 10 4.7% for **agriculture, farming and gardening**[7]: 2.5 In 2019: Plastic used for **livestock and crop production**[9, 6]: 0.72, *i.e.*, 28.7% of plastic consumption in agriculture, farming and gardening, and 1.4% of total plastic consumption in Europe. 	In 2018: Total **plastic consumption**[10]: 5.6 In 2022: Total **plastic consumption**[12]: 6.38 of which: 37.2% for **packaging**[12]: 2.38 20.1% for **food and beverage packaging**[8]: 1.28 3.2% for **agriculture, farming and gardening**[12]: 0.207 (12%, 12.9%, 12.8%, 8.3% of the total consumption, consumption for packaging, consumption for food and beverage packaging, and consumption for agriculture, farming and gardening in Europe, respectively) In 2023: **Packaging and non-packaging agricultural plastics**[14]: 0.125, *i.e.*, 61.8% of plastic consumption in agriculture, farming and gardening, and 2% of total plastic consumption in France. **Non-packaging agricultural plastics**[14]: 0.103, *i.e.*, 49.8% of plastic consumption in agriculture, farming and gardening, and 1.6% of total plastic consumption in France.

	Global	Europe	France
		 In 2017: **MPLs intentionally added to products used in agriculture**[11]: 0.02	
Plastic waste generation ($Mt \cdot year^{-1}$)	In 2016: **Municipal plastic waste**[15]: 231.3 In 2019: Total **plastic waste**[16]: 353.31 of which: 40% **packaging waste**: 141.96 Plastic food packaging waste: not available Agricultural plastic waste: not available In 2019: Primary **plastic waste by main polymer produced**[17]: Polyethylene (PE) 93.92 including High-density polyethylene (HD-PE) 44.71 and Linear low-density polyethylene and low-density polyethylene (LLD-PE, LD-PE) 49.21 Polypropylene (PP) 62.00 Polyvinyl chloride (PVC) 21.20 Polyethylene terephthalate (PET) 24.81 Polystyrene (PS) 21.12	In 2018, in the EU28 + Norway + Switzerland: Total **plastic waste** (post-consumer)[18]: 29.1 of which: 5% **agricultural plastic waste**: 1.455 In 2022: **Plastic packaging waste**[19]: 16.16 Plastic food packaging waste: not available	In 2020: Total **plastic waste** (post-consumer)[20]: 3.76 of which: 3% **agricultural plastic waste**: 0.113 In 2022: **Plastic packaging waste**[19]: 2.43 Plastic food packaging waste: not available
Plastic waste collected ($Mt \cdot year^{-1}$)	In 2016: **Municipal plastic waste**[15]: 179.5 Plastic packaging waste: not available Plastic food packaging waste: not available Agricultural plastic waste: not available	In 2022, in EU27 + Norway + Switzerland + the United Kingdom: **Plastic waste** (post-consumer)[7]: 32.3 of which: 57.3% **plastic packaging waste**: 18.5 4.6% **agriculture, farming and gardening waste**: 1.5 Plastic food packaging waste: not available	In 2018: **Plastic waste**[10]: 3.6 In 2023: **Plastic waste in agriculture**[21]: 0.098 In 2022: **Plastic packaging waste**[19]: 2.43 of which: 53.9% **food packaging plastic waste**[22]: 1.31

	Global	Europe	France
Municipal solid waste generation, *i.e.*, all municipal waste, excluding industrial waste (Mt·year^{-1})	In 2020: All **municipal solid waste**[23]: 2126, of which: 11.5%[15] **plastic waste**[24]: 244.5	In 2022: **Municipal solid waste**[25]: 230.18 of which: 7.0%[26] **plastic packaging waste**[19]: 16.16	In 2022: **Municipal solid waste**[25]: 36.42 of which: 6.7% **plastic packaging waste**[19]: 2.43
Plastic sent to landfill, to incineration, to recycling (Mt·year^{-1})	In 2019: **Plastic waste**[27]: 63.6% landfilled 24.7% incinerated 11.7% collected for recycling	In 2020, in the EU27 + Norway + Switzerland + the United Kingdom: **Plastic waste** (post-consumer) collected[28]: 29.5 23.4% sent to landfill: 6.9 42.0% sent to energy recovery: 12.4 34.6% sent to recycling: 10.2 In 2022, in the EU27 + Norway + Switzerland + the United Kingdom: **Plastic waste** (post-consumer) collected[7]: 32.3 23.5% sent to landfill: 7.6 49.6% sent to energy recovery: 16.0 26.9% recycled: 8.7 In 2022, in the EU27 + Norway + Switzerland + the United Kingdom: **Plastic packaging waste** (post-consumer) collected[7]: 18.5 17.3% sent to landfill: 3.2 44.9% sent to energy recovery: 8.3 37.8% recycled: 7.0 **Plastic waste from agriculture, farming and gardening** (post-consumer) collected[7]: 1.5 33.3% sent to landfill: 0.50 46.7% sent to energy recovery: 0.70 20.0% recycled: 0.30	In 2018: **Plastic waste** collected[10]: 3.6 32.2% sent to landfill: 1.16 32.6% sent to energy recovery: 1.17 35.2% collected for recycling: 1.27 In 2022: **Plastic packaging waste**[29]: 25% recycled In 2020: 56% of agricultural plastic waste recycled[30]

In black: data from the grey literature
In blue: data from the scientific literature
In grey: calculated data

Note: Post-consumer recycling does not include pre-consumer recycling, *i.e.*, the recycling of waste originating from the manufacture of plastic products and the manufacture of plastics in primary forms (polymerisation) to a lesser extent.

Note: In line with Directive (EU) 2018/852, estimated data of Plastics Europe and Eurostat for recycled plastics in 2022 in Europe and France, respectively, refer to plastics entering a recycling factory.

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[3] Based on: Plastics Europe, 2024. The fast facts 2024. https://plasticseurope.org/knowledge-hub/plastics-the-fast-facts-2024/ (Plastics Europe, 2024b)

[4] Based on: Dow, 2023. 2023 Progress Report. Midland, USA: Dow Inc., 206 p. https://corporate.dow.com/content/dam/corp/documents/about/066-00469-01-2023-progress-report.pdf (Dow, 2023) and Plastics Europe, 2024. The fast facts 2024. https://plasticseurope.org/knowledge-hub/plastics-the-fast-facts-2024/ (Plastics Europe, 2024b)

[5] Based on [4] and FBI, 2025. Plastic Packaging Market Size, Share & Industry Analysis, By Material (Polyethylene [High-Density Polyethylene (HDPE), Low-density polyethylene (LDPE)], Polypropylene (PP), Polyethylene Terephthalate (PET), Polyvinyl Chloride (PVC), Polystyrene (PS), & Bioplastic), By Product Type (Rigid Plastic [Bottles & Jars, Containers, Trays & Pallets, IBCs & Drums, Caps & Closures] and Flexible Plastic [Bags, Pouches & Sachets, Films & Laminates, Tapes & Labels, Tubes]), By End-use Industry (Food & Beverage, Healthcare, Home Care & Personal Care, Industrial, E-commerce), and Regional Forecast, 2024-2032: Fortune Business Insights, (FBI110535), 220 p. https://www.fortunebusinessinsights.com/plastic-packaging-market-110535 (FBI, 2025a)

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[8] Based on: FBI, 2025. Plastic Packaging Market Size, Share & Industry Analysis, By Material (Polyethylene [High-Density Polyethylene (HDPE), Low-density polyethylene (LDPE)], Polypropylene (PP), Polyethylene Terephthalate (PET), Polyvinyl Chloride (PVC), Polystyrene (PS), & Bioplastic), By Product Type (Rigid Plastic [Bottles & Jars, Containers, Trays & Pallets, IBCs & Drums, Caps & Closures] and Flexible Plastic [Bags, Pouches & Sachets, Films & Laminates, Tapes & Labels, Tubes]), By End-use Industry (Food & Beverage, Healthcare, Home Care & Personal Care, Industrial, E-commerce), and Regional Forecast, 2024-2032: Fortune Business Insights, (FBI110535), 220 p. https://www.fortunebusinessinsights.com/plastic-packaging-market-110535 (FBI, 2025a) and Plastics Europe, 2024. The Circular Economy for Plastics. A European Analysis 2024, 112 p. https://plasticseurope.org/wp-content/uploads/2024/05/Circular_Economy_report_Digital_light_FINAL.pdf (Plastics Europe, 2024c)

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[21] Data 2023 provided by ADIVALOR
[22] Based on: FBI, 2025. Plastic Packaging Market Size, Share & Industry Analysis, By Material (Polyethylene [High-Density Polyethylene (HDPE), Low-density polyethylene (LDPE)], Polypropylene (PP), Polyethylene Terephthalate (PET), Polyvinyl Chloride (PVC), Polystyrene (PS), & Bioplastic), By Product Type (Rigid Plastic [Bottles & Jars, Containers, Trays & Pallets, IBCs & Drums, Caps & Closures] and Flexible Plastic [Bags, Pouches & Sachets, Films & Laminates, Tapes & Labels, Tubes]), By End-use Industry (Food & Beverage, Healthcare, Home Care & Personal Care, Industrial, E-commerce), and Regional Forecast, 2024-2032: Fortune Business Insights, (FBI110535), 220 p. https://www.fortunebusinessinsights.com/plastic-packaging-market-110535 (FBI, 2025a) and Eurostat, 2024. Packaging waste by waste management operations. https://ec.europa.eu/eurostat/databrowser/product/page/ENV_WASPAC [Accessed 2 April 2025] (Eurostat, 2024b)
[23] UNEP, 2024. Beyond an age of waste: Turning rubbish into a resource. Global Waste Management Outlook 2024. Geneva: United Nations Environment Programme, Technical Reports, 114 p. http://dx.doi.org/10.59117/20.500.11822/44939 (UNEP, 2024)
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[26] Based on: Eurostat, 2024. Packaging waste by waste management operations. https://ec.europa.eu/eurostat/databrowser/product/page/ENV_WASPAC [Accessed 2 April 2025] (Eurostat, 2024b) and Eurostat, 2024. Municipal waste by waste management operations. https://ec.europa.eu/eurostat/databrowser/product/page/ENV_WASMUN [Accessed 2 April 2025] (Eurostat, 2024a)
[27] OECD, 2023 – processed by Our World in Data. Share of plastic waste that is recycled, landfilled, incinerated and mismanaged, 2019 https://ourworldindata.org/grapher/share-plastic-fate?tab=table [Accessed 2 April 2025] (Our World in Data, 2023b)
OECD, 2022. Global Plastics Outlook. Plastics waste by region and end-of-life fate, polymer and application. https://ourworldindata.org/grapher/share-plastic-fate?tab=table [Accessed 21 September 2023] (OECD, 2022a)
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[29] Eurostat, 2024. Packaging waste by waste management operations. https://ec.europa.eu/eurostat/databrowser/product/page/ENV_WASPAC [Accessed 2 April 2025] (Eurostat, 2024b)
[30] Data 2020 provided by ADIVALOR

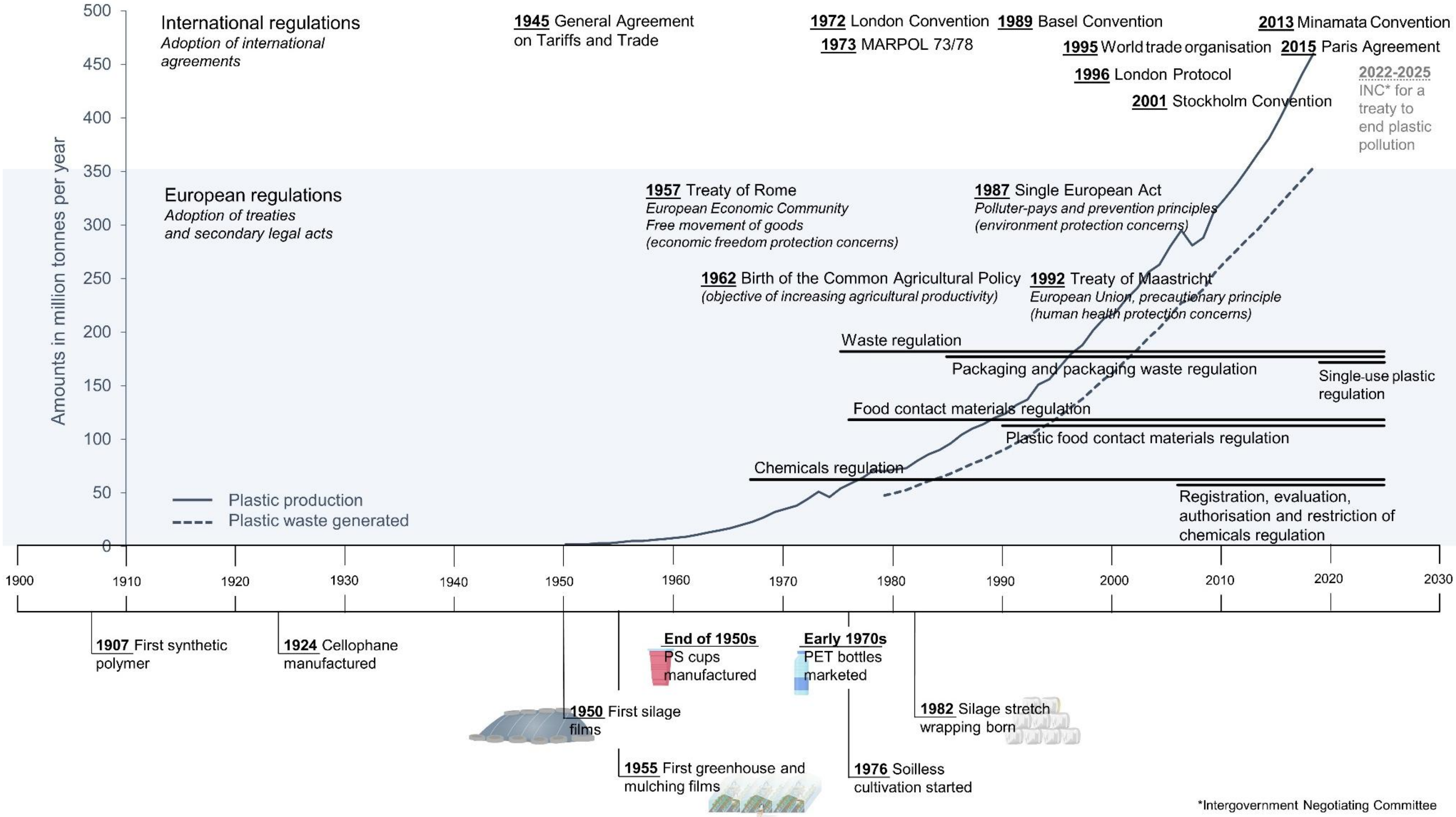

Figure 2. A. Iconic innovations in plastics used in agriculture and for food, annual production of plastics worldwide from 1950 and 2019, annual generation of plastic waste worldwide from 1980 to 2019, and major international and European regulations (blue area).

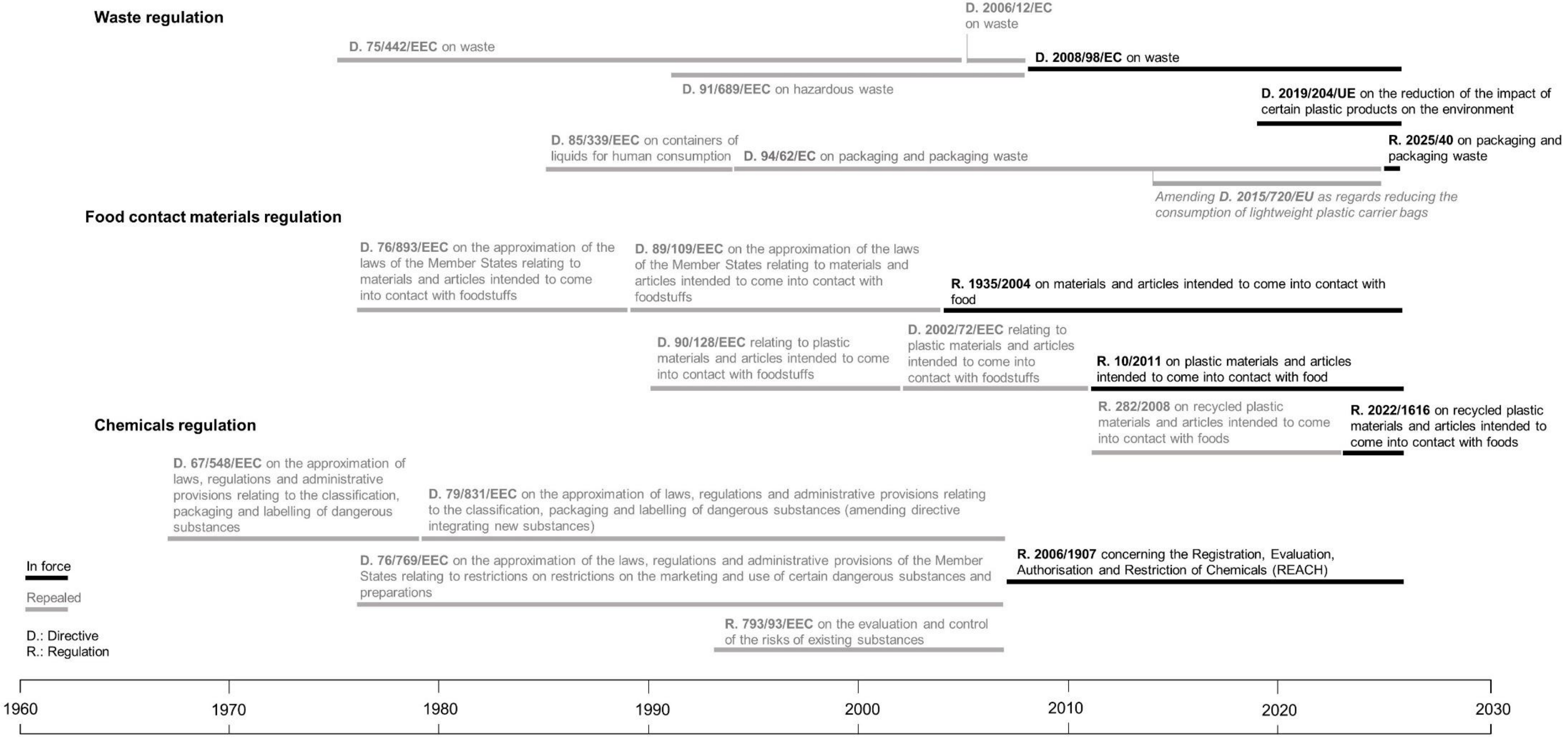

Figure 2. B. *Main European legal acts regulating plastics used in agriculture and for food.*

Plastics, the free movement of which - like any other goods - was ensured under international trade agreements and European Community law, thus also became subject to rules based on the polluter-pays principle, and principles of prevention and precaution in the late 1980s and early 1990s (Figure 2A). At the European level, this shift and structuring of the human health and environmental law translated for plastics into the implementation, extension and reinforcement of (i) the dangerous substances regulation from 1967 (replaced by the Registration, Evaluation, Authorisation and Restriction of Chemicals regulation (REACH) in 2006); (ii) the waste, packaging and packaging waste and single-use plastic (SUP) products regulations from 1975, 1985 and 2019, respectively; and (iii) food contact materials (FCMs) and plastic FCM regulations from 1976 and 1990, respectively (Figure 2B). At the French level, these European regulations of plastics directly applied or were transposed into law.

Until now, however, international, European and French regulations have failed to bend the plastic production, waste generation and pollution curve (Figures 1 and 2). Moreover, scientific knowledge continues to accumulate on the impact of plastics on the environmental, social and economic dimensions of sustainable development as defined by the United Nations, including on human health. This scientific knowledge may inform on the suitability of plastics with a development that meets the needs of the present without compromising the ability of future generations to meet their own needs. It may further call into question the way plastics are currently regulated, and especially plastics used in agriculture and for food. As these plastics are in close contact with soils, feed and food in the making (crops, livestock and poultry), and food that humans ingest, their sustainability and regulation raise particular concerns.

The French Agency for Ecological Transition, the French Ministry of Agriculture and Food Sovereignty, and the French Ministry of Ecological Transition requested a Collective scientific assessment (CSA) to INRAE and the CNRS. Such a CSA is conducted following principles of competence, plurality of disciplines and approaches, impartiality, and transparency.

In this CSA, 30 French and European experts from different disciplines reviewed knowledge from the peer-reviewed scientific literature on the sustainability of plastics used in agriculture and for food, in all its dimensions, and considering their whole life, from production, use, waste management to environmental fate.

More specifically, this CSA aimed at (i) understanding uses of plastics in agriculture and for food; (ii) investigating their properties from their production, during and after their use; (iii) examining their waste management strategies; (iv) assessing their presence and impacts on continental ecosystems (including terrestrial, freshwater, estuarine ecosystems but excluding marine ecosystems) and humans; (v) analysing the way in which trade-offs in environmental, social and economic impacts are taken into account in a sustainable approach to the design of the system of plastics used in agriculture and for food.

Whenever possible and relevant, experts isolated plastics used in agriculture and for food from other plastics in their analysis. Given the broad scope of the CSA and abundance of the literature, per- and polyfluoroalkyl substances (PFAs), some of which are plastics, and silicones, although they are plastics, have not been addressed. Regarding uses in agriculture and for food, experts focused on Metropolitan France and Europe. However, broader geographical boundaries were considered regarding production, waste management, and impacts.

In Section 2 of this condensed report, materials and methods used to provide the state of knowledge are first described: CSA principles followed (2.1), expert committee composed (2.2), literature reviewed (2.3) and system and definitions considered (2.4).

Section 3 is divided into five subsections. Findings of the CSA - all based on available literature and expert analysis of this literature – are presented and structured around four key messages and an open question. In 3.1, a systemic and historical approach was used to understand the role of plastics in food value chains, and show why and how plastics have spread throughout food value chains, in particular downstream of production. In 3.2, relationships between the petroleum- and bio-based origin of plastics, their formulations, design and properties, were investigated, emphasising how the plasticity of plastics encourages complexity in formulations and design. In 3.3, plastic waste management strategies were examined: reuse, recycling and biodegradation in particular, and difficulties to monitor and implement plastic waste management in practice are demonstrated. In 3.4, the dispersion of plastics in continental ecosystems, contamination and contamination routes to agricultural soils, living organisms and humans in particular were characterised, and consequences on ecosystem functioning and human health were assessed. Results highlight the ubiquity, health hazard and multiscale impacts of plastics on living organisms, humans and continental ecosystems. Finally, in 3.5, limitations of approaches to a sustainable system of plastics used in agriculture and for food in terms of understanding of sustainability, sustainability assessment methods and resulting changes proposed were pointed to. Beyond providing knowledge on the sustainability of plastics used in agriculture and for food, our CSA results call into question the very possibility of making the associated sociotechnical system sustainable. The way in which objectives relating to various dimensions of sustainability and actors' interests influenced the regulation of the system of plastics used in agriculture and for food was analysed. Resulting regulations are shown to overlook key aspects to protect human health and the environment.

2. Materials and methods

2.1. Collective scientific assessment principles

In France, public actors request CSAs as part of public research missions. CSAs aim at informing public policy, identifying research needs and feeding the public debate on complex interdisciplinary issues. However, they do not aim at making policy recommendations. In CSAs, the peer-reviewed scientific literature is reviewed in order to identify consensual, still debated and missing knowledge needed to address questions raised by public actors.

CSAs are conducted following four principles: competence, plurality of disciplines and approaches, impartiality, and transparency. Principles and methodology used are further detailed in the guidelines for collective scientific assessments and advanced studies at INRAE[1], and the CNRS institutional scientific expertise charter[2].

Three public deliverables are produced: an extended report, including the collective review of the literature and overall conclusions, and a condensed report and summary report, including main findings of the extended report. A public symposium is held in Paris, and the video is made available[3].

CSAs are monitored by an oversight committee. In this specific CSA, the oversight committee was composed of representatives of the French Agency for Ecological Transition, the French Ministry of Agriculture and Food Sovereignty, the French Ministry of Ecological Transition, and of INRAE and the CNRS.

A stakeholder advisory committee provides a place for the expression of stakeholders' view. In this specific CSA, the stakeholder advisory committee was composed of 19 representatives of industries, professional organisations, organisations at the interface between research and stakeholders, civil society organisations, and public authorities. Members of the committee were selected to cover the production, use, post-use and environmental fate of plastics used in agriculture and for food. They included representatives of: Plastics Europe – France, the French Committee for Plastics in Agriculture (CPA), the Industrial Technical Centre for Plastics and Composites (IPC), the Polymeris competitiveness cluster, the Agricultural Technical Coordination Association (ACTA), the French organisation for ornamental horticulture and landscape VALHOR, the Permanent Assembly of Chambers of Agriculture (APCA), the Centre technique agroalimentaire (CTCPA), the consumer organisation UFC – Que Choisir, the Réseau Vrac & Réemploi, the producer responsibility organisations (PROs) CITEO for household waste and ADIVALOR (Agriculteurs, Distributeurs, Industriels pour la Valorisation des Déchets Agricoles) for agricultural plastics, the national network AMORCE, the Team2 competitiveness cluster, the Parliamentary Office for evaluation of scientific and technological options (OPECST), the Alliance des collectivités pour la qualité de l'air, the Laboratoire national de métrologie et d'essais (LNE), France Nature Environnement (FNE), and Break Free from Plastic.

2.2. Expert committee composition

A committee of experts from public research organisations writes the extended report, in a collective and iterative process, based on the review of the knowledge available in the peer-reviewed scientific

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[1] https://www.inrae.fr/sites/default/files/pdf/2023_Guide_ESCo_DEPE_Web_VA.pdf
[2] https://www.cnrs.fr/sites/default/files/download-file/CharteExpertise_EN_web.pdf
[3] https://esco-plastiques-agri-alim.colloque.inrae.fr/

literature. Experts of this committee are identified based on their scientific publications on topics relevant to the CSA.

In this specific CSA, experts were first selected among more than 100,000 authors from an exploratory corpus of 72,751 publications indexed in the Web of Science (WoS). They were then ranked based on additional criteria of competence, including the range of topics addressed, the plurality of approaches and disciplines used, teamwork skills, the experience in reviews, the involvement in relevant research projects, etc.). Some experts, particularly in the humanities and social sciences, had little or no publications indexed in the WoS and/or had published little or nothing on the specific topic of 'plastics'. They were therefore identified based on their disciplinary expertise.

A group of 30 experts plus three scientific leads from 24 public research organisations (Figure 3) in 8 European countries (Figures 3 and 4) was set up. A fifth of the group (five experts and one of the scientific leads) was affiliated with INRAE. The group gathered 21 macro-disciplines and mainly four types of macro-disciplines: life sciences, applied sciences, physics, chemistry and materials science, and humanities, economics and social sciences (52%, 45%, 40% and 39% of the group, respectively) (Figure 5).

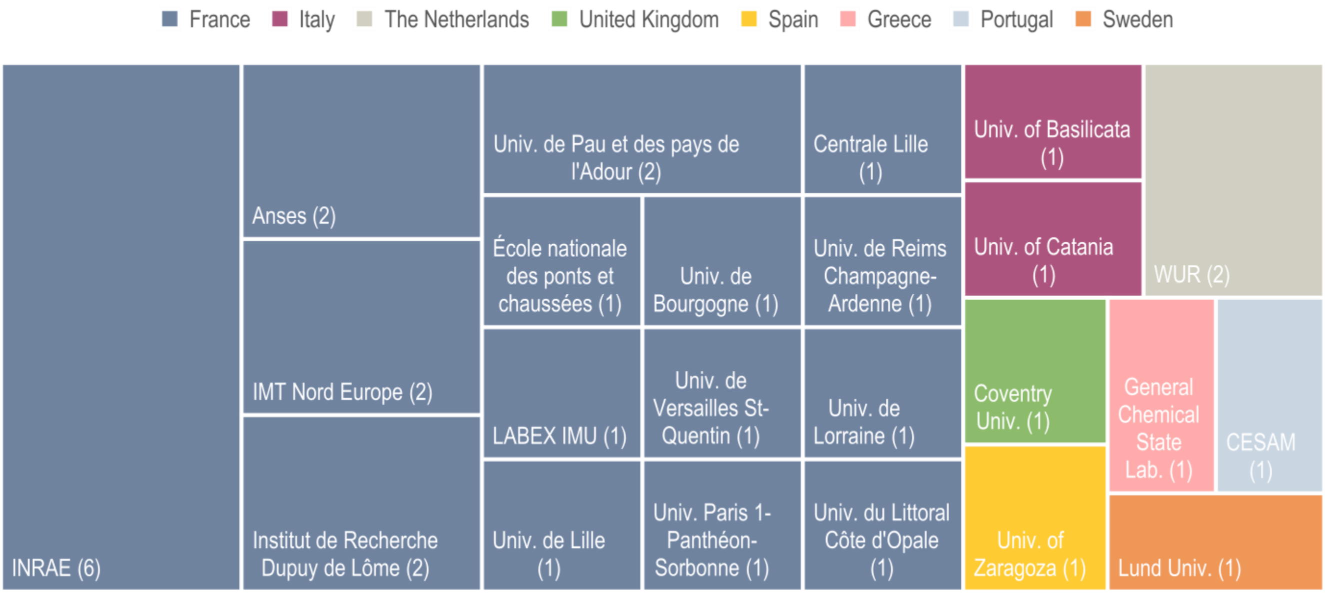

Figure 3. *Affiliations of the 30 experts and three scientific leads during the assessment. The number of experts and/or scientific leads affiliated with each organisation is indicated between brackets. Countries are represented by different colours.*

Figure 4. *Countries represented in the group of 30 experts plus three scientific leads. The number of experts and/or scientific leads is indicated for each country.*

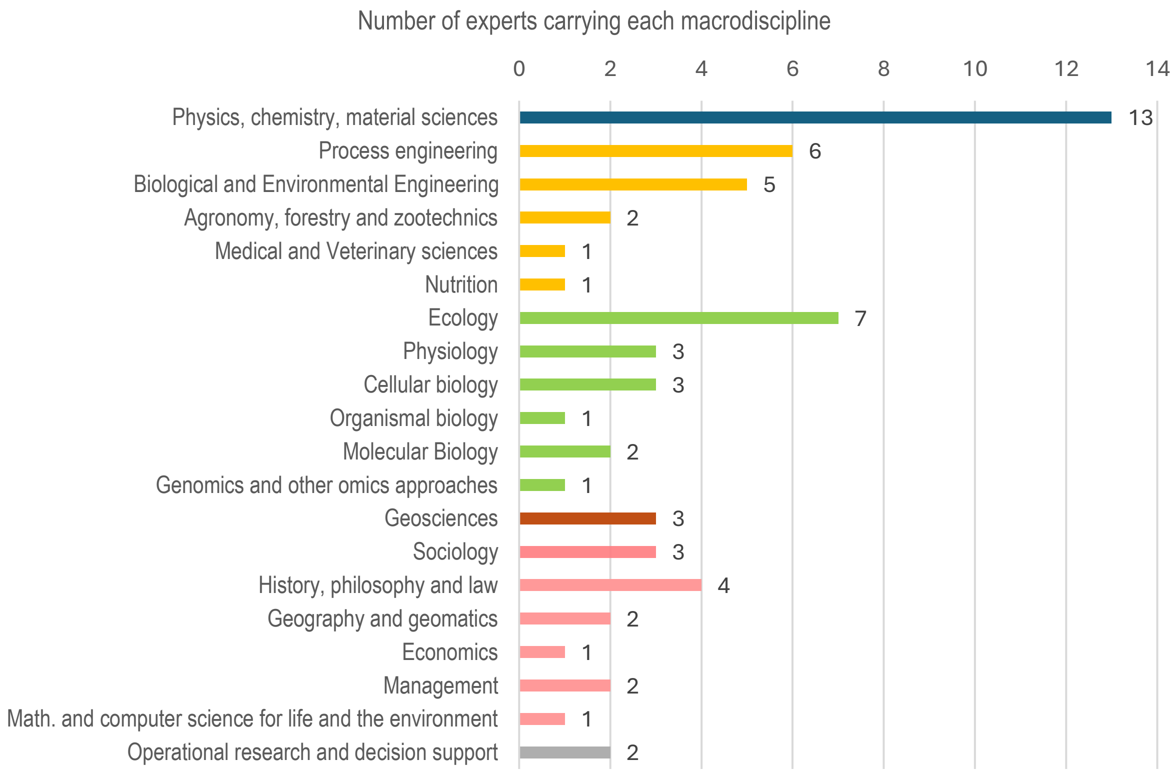

Figure 5. *Macro-disciplines gathered in the group of 30 experts and three scientific leads. The number of experts and/or scientific leads is indicated for each macro-discipline. The same expert can be competent in several macro-disciplines. Types of macro-disciplines are represented by different colours. Physics, chemistry and materials science, applied sciences, life sciences, geosciences, humanities, economics and social sciences, and digital sciences and modelling are represented in blue, yellow, green, brown, pink and grey, respectively.*

2.3. Literature reviewed

Experts critically analysed 4342 references (Table 2), some of which are cited in this condensed report. The working group, *i.e.*, the expert committee and the coordination team composed of three scientific leads and the project manager, was supported by two librarians to build search queries and collect records from database searches (mainly on the WoS, and to a lesser extent on Scopus for economics and social sciences).

The thousands of references collected were screened by the expert committee based on their title, abstract and author keywords. Only references relevant to the CSA were selected for full text analysis. Experts further enriched this initial corpus with additional references not indexed in databases or difficult to capture using systematic search queries.

The resulting final corpus was mainly composed of academic references (92%) and scientific articles in particular (84%), but also ‘grey’ references such as reports and legal texts (Table 2 and Figure 6). Of the 3657 scientific articles, most (85%) were primary research articles (Figure 7). Of the 140 legal texts cited, 102 were specifically analysed in a dedicated section of the extended report (V.1).

Table 2. Detailed composition of the final corpus analysed in the CSA.

	Number of references	% of total
I. Academic references and among these references:	**3983**	**92%**
1. Scientific articles published in peer-reviewed journals and among these publications	3657	84%
a. (Primary) Research articles	*3083*	*71%*
b. Narrative reviews, systematic reviews and meta-analyses	*556*	*13%*
c. Other	*18*	*0,4%*
2. Scientific books or chapters of books	125	3%
3. Thesis	10	0%
4. Conference proceedings or conference papers	191	4%
II. 'Grey' references and among these references	**359**	**8%**
1. Reports	158	4%
2. Legal texts	140	3%
3. Magazines (*e.g.*, Plasticulture Magazine), online articles or manuscripts	61	1%
Total	**4342**	**100%**

Journal articles
Conference proceedings
Reports
Legal texts
Books
Other
Thesis

Type of references	Total	%
Journal articles	3 657	84%
Conference proceedings	191	4%
Reports	158	4%
Legal texts	140	3%
Books	125	3%
Other	61	1%
Thesis	10	0%
Total	**4 342**	**100%**

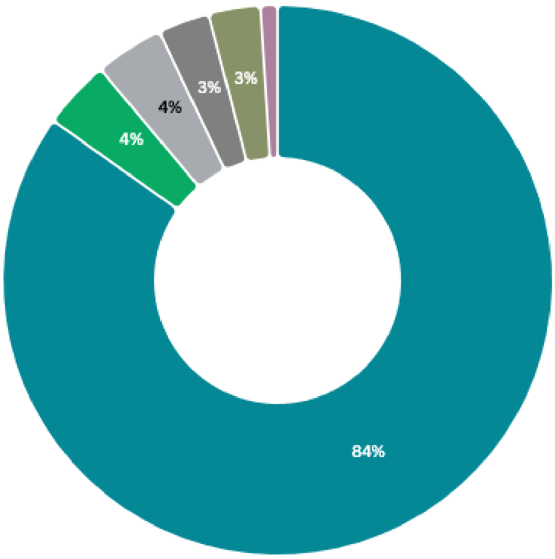

Figure 6. Distribution of references by main type in the final corpus of the CSA.

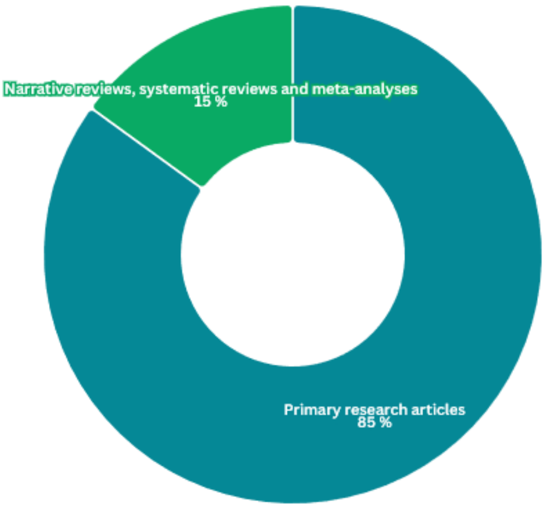

Figure 7. *Distribution of scientific articles published in peer-reviewed journals: primary research articles vs. narrative reviews, systematic reviews and meta-analyses).*

2.4. Analytical framework: system and definitions considered

2.4.1. Plastics in their diversity

Plastics in general and plastics used in agriculture and for food in particular are diverse. They can be characterised, among others, based on their composition, intermolecular forces and origin, as materials or objects, or as particles in the environment based on their size. In following sections, we will also address the characterisation of plastics based on their properties, functions and objectives of use (2.4.2), as well as uses and life stages (2.4.3).

Regarding their composition (Figure 8A), not all the polymers are plastics, but all plastics are essentially polymers and are generally named after the main polymer they are made from. Polymers are composed of repeating carbon-based units called monomers connected to one another in a chain-like structure. Additives are added to polymers to achieve certain properties and give them the versatility they are known for. Among others, they soften (plasticisers), stiffen (fillers), colour (colourants and pigments), prevent or slow the further development of flames (flame retardants) or oxidation (antioxidants), stabilise (ultraviolet (UV) or heat stabilisers) or improve the processability (processing aids) (Law *et al.*, 2024). Additives are thus referred to as intentionally added substances (IAS), together with residual substances from the different steps to prepare polymers (*i.e.*, polymerisation and/or polymer processing), *e.g.*, unreacted monomers. On average, plastics contain 93% polymers and 7% additives by mass (Geyer *et al.*, 2017) but the composition of plastics and proportion of additives in plastics vary widely for a given application and according to applications (Hahladakis *et al.*, 2018; Groh *et al.*, 2019; Wiesinger *et al.*, 2021; UNEP, 2023a; Law *et al.*, 2024; Wagner *et al.*, 2024). In addition to additives and IAS, plastics contain non-intentionally added substances (NIAS) accumulated from their production, during and after their use (Geueke, 2018; Groh *et al.*, 2019; Cook *et al.*, 2023; Law *et al.*, 2024), *i.e.*, impurities, reaction by-products, and degradation products, and chemical (including heavy metals, pesticides, pharmaceuticals) and/or microbial (including pathogens) contaminants.

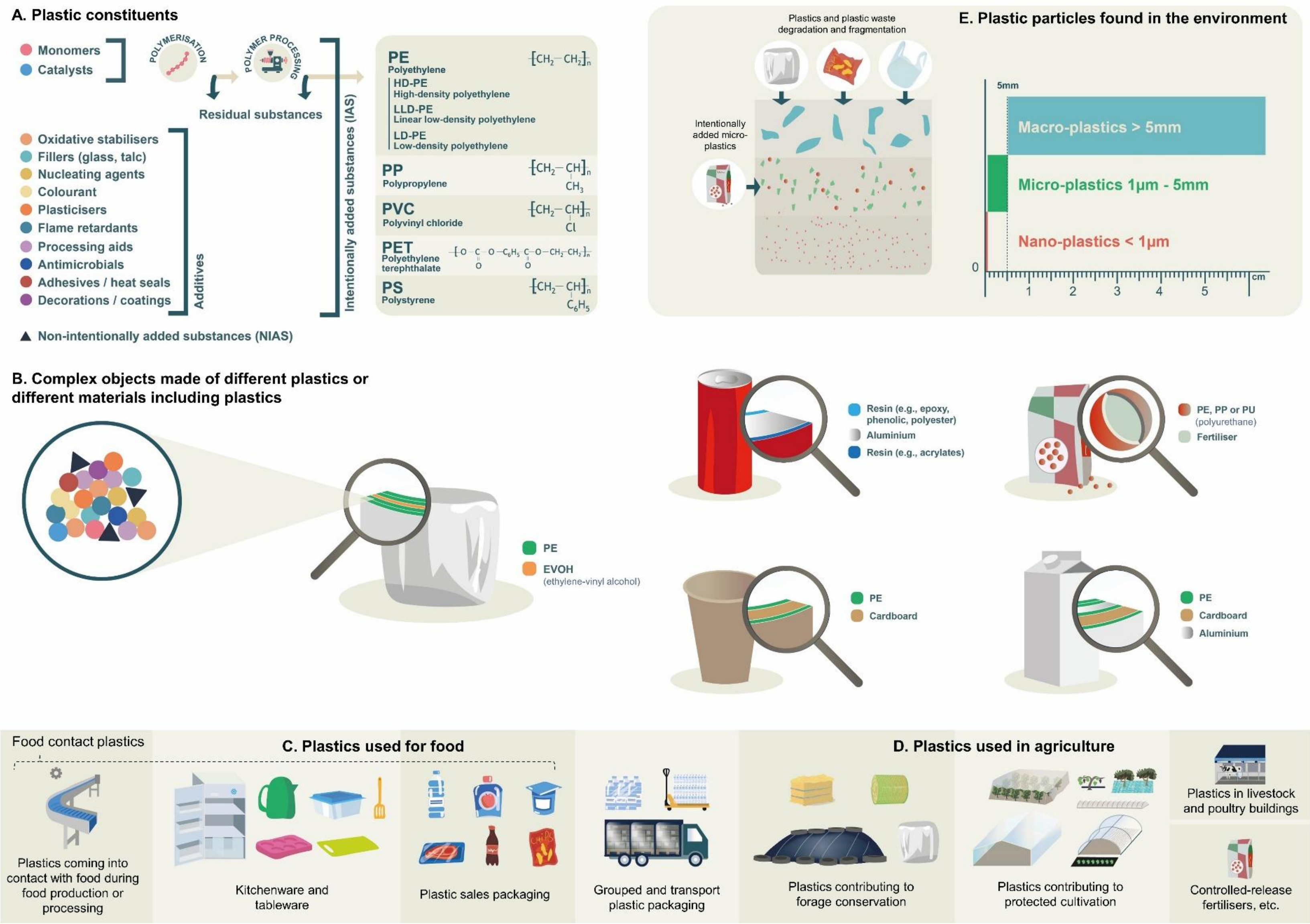

Figure 8. *Plastics in their diversity: from constituents (**A**), materials and objects (**B**, **C**, **D**) and to particles (**E**).*

Based on the magnitude of intermolecular forces of attraction that hold polymer chains together, plastics can be classified into four categories, in descending order: thermosets, fibres, thermoplastics and elastomers. Unlike thermosets, thermoplastics can soften when heated and become hard when cooled (Deb *et al.*, 2019). The five plastics most produced worldwide in 2019 (Table 1 and Figure 8A), *i.e.*, polyethylene (PE) including high-density polyethylene, linear low-density polyethylene and low-density polyethylene (HD-PE, LLD-PE and LD-PE, respectively), polypropylene (PP), polyvinyl chloride (PVC), polyethylene terephthalate (PET) and polystyrene (PS) are thermoplastics. In the case of elastomers, *e.g.*, silicones, intermolecular interactions that hold polymer chains together are weak enough for them to temporarily deform under stress and return to their original shape once the stress is released (Bensalem *et al.*, 2024).

Definitions considered

Plastics (adapted from Regulation (European Union (EU)) 10/2011 (2011) and Directive (EU) 2019/904 (2019a))
Plastics were defined as polymers to which additives (including, e.g., processing aids, fillers) may have been added.
Coatings, paints, inks, adhesives, composites and elastomers were considered as plastics as long as they are made of polymers to which additives may have been added. Moreover, plastics manufactured with modified natural polymers, or plastics manufactured from bio-based, fossil or synthetic starting substances (e.g., cellulose-based, starch-based or chitosan-based polymers) are not naturally occurring and were also considered as plastics.

Polymer (adapted from Regulation (EU) 10/2011 (2011) and ISO (ISO, 2013))
Polymer means any macromolecular substance obtained by:
(a) a polymerisation process such as polyaddition or polycondensation, or by any other similar process of monomers and other starting substances (in the case of, e.g., starch-based polymers); or
(b) chemical modification of natural or synthetic macromolecules; or
(c) microbial fermentation.
A polymer is composed of molecules characterised by the multiple repetition of one or more species of atoms or groups of atoms (constitutional units) linked to each other in amounts sufficient to provide a set of properties that do not vary markedly with the addition or removal of one or a few of the constitutional units.

Additive (adapted from Regulation (EU) 10/2011 (2011)
Additive means a substance that is intentionally added to polymers to achieve a physical or chemical effect during processing of the plastic or of the final material or article (e.g., mould release agent); it is intended to be present in the final material or article.

Regarding their origin, plastics and IAS they are made from are almost entirely sourced from fossil feedstocks (oil and natural gas). In 2023, only 1.5% (0.08 out of 5.51 Mt), 1.5% (0.8 out of 54 Mt) and 0.7% (3 out of 413.8 Mt) of plastics were produced from bio-based feedstock in France, Europe and worldwide, respectively (Table 1). Bio-based plastics are fully or partially made from biological resources (EC, nd). They contain bio-based polymers (Barretto *et al.*, 2024) and/or bio-based additives (Marturano *et al.*, 2023). Bio-based polymers can be extracted from biomass (alginate, starch, pectin, cellulose, hemicellulose, protein, chitin, lignin), produced from biomass monomers (polylactic acid, bio-based polyethylene terephthalate abbreviated PLA and bio-PET, respectively) or by microorganisms (polyhydroxyalkanoate abbreviated PHA, bacterial cellulose) (Barretto *et al.*, 2024). Biomass can include grown crops, such as maize, or organic residuals and waste, such as agricultural or food waste. However, there are no 'natural' plastics. Bio-based plastics are not naturally occurring and are thus not necessarily biodegradable (3.3.4). For this reason, and as the terms biopolymer or bioplastic often used in the literature may refer to bio-based or biodegradable or both, they were considered misleading and were not used in this condensed report.

Definitions considered

<u>Bio-based or bio-sourced plastic</u> (EC, nd)
Bio-based plastics are fully or partially made from biological resources, rather than fossil raw materials. They are not necessarily biodegradable as petroleum-based plastics are not necessarily non-biodegradable.

Plastics make up all or part of materials and objects, materials being what objects are made from (Figure 8B, C and D). They can be linked or layered with distinct materials in composites or multi-layered materials (de Mello Soares *et al.*, 2022; Wang *et al.*, 2022b). Coatings (*e.g.*, in the case of plastic-coated paper cups), paints, inks and adhesives were considered as plastics as long as they are made of polymers to which additives may have been added.

Definitions considered

<u>Composite</u> (adapted from Wang et al. (2022b)
A material composed of more than one distinct materials that are linked together by covalent or noncovalent bonds, such as electrostatic, hydrophobic, hydrogen bonding, or van der Waals interactions.
Multi-layered materials fall within the scope of composite definition but since they have specific characteristics that need to be considered, they are named multi-layered materials and not composites in this condensed report.

<u>Multi-layered material</u> (de Mello Soares et al. (2022)
Multi-layered materials consist of more than one layer of distinct materials where the components are layered to form flexible packaging (pouches, bags, shrink films, other pliable products) or rigid ones (trays, cups, containers, other rigid plastic sheets).

As particles in the environment, plastics correspond to both primary and secondary plastics (Figure 8E). Primary plastics are plastic objects in the same form as they were produced, either lost (*e.g.*, packaging waste) or intentionally added to some products and used in the environment (*e.g.*, controlled release fertilisers and fertiliser additives). Secondary plastics result from the degradation and fragmentation of primary plastics under the effects of mechanical or physico-chemical processes (triggered by UV radiation and oxygen in particular) (Lebreton *et al.*, 2019, Magnin, 2020 #107). Plastic particles with size larger than 5 mm, ranging from 1 µm to 5 mm, and smaller than 1 µm are referred to as macro- (MaPLs), micro- (MPLs) and nano-plastics (NPLs), respectively (Frias et Nash, 2019; ISO, 2020).

Definitions considered

<u>Micro-plastic</u> (Frias and Nash (2019) Micro-plastics are any synthetic solid particles or polymeric matrices, with regular or irregular shape and with size ranging from 1 µm to 5 mm, of either primary or secondary manufacturing origin, which are insoluble in water.

<u>Nano-plastic</u> (ISO, 2020)
Nano-plastics are plastic particles smaller than 1 µm.

2.4.2. Distinguishing materials properties from users' objectives

Materials and plastics in particular are able to be adapted to many different properties that make them useful as all or part of objects to perform certain functions. Properties describe how the material or object behaves. They vary depending on the structure of the material, itself depending on its formulation and processing. Functions refer to the actions for which plastic materials or object are specially formulated and processed, or in other words specially fitted or designed for.

Beyond intrinsic properties and functions of plastic materials and objects are their relationships with humans. Actors of food value chains, whether agrichemical, food processing, retail, and food service companies, farmers or final food consumers use plastic objects for different reasons that go beyond their properties and functions. Plastics objects have affordances (Babri et al., 2022) and actors have objectives of use.

For example, oxygen permeability is a property used to describe silage films. Silages films can be adapted to decrease oxygen permeability. This makes them useful to perform the function of sealing forage in suitable anaerobic conditions. Beyond this function, silage films may allow farmers to achieve different objectives such as guaranteeing the nutritional quality of conserved forage, improving livestock productivity, farm income and/or farm labour productivity.

Similarly, barrier properties of food packaging are desired to protect food. In this way, food packaging may allow retailers to extend the shelf-life of food products and reduce losses in the distribution process.

Several properties can contribute to meeting the same function. Several functions can contribute to meeting the same objective. Several objectives can be associated with the same function and objectives can be nested.

Definitions considered

Functions of plastic materials and objects (adapted from Merriam-Webster dictionary[4])
By function, we mean the action for which plastic material or object is specially fitted or designed for. Function is also referred to as application in the literature.

Objectives of use of plastic objects (adapted from Cambridge dictionary[5])
By objective of use, we mean something that a user (either intermediate or final user) aims to achieve when using a plastic object.

Affordances of plastic objects (adapted from Babri et al. (2022)
Affordances refer to the uses that plastics make possible for their users. The affordances of plastic are not intrinsic qualities of plastics, as chemical or mechanical traits. Affordances are relational. They are expressions of what materials afford those interested in using them. The affordances of plastics are how humans perceive and exploit the possible uses that plastics make possible, in other words how humans relate to plastics.

2.4.3. Plastics used in agriculture and for food, from production to environmental fate: a sociotechnical system at the crossroads between food and plastic value chains

This CSA focused on impacts of plastics used in agriculture and for food from production, use, waste management to environmental fate. This focus implied considering a system at the crossroads between food and plastic value chains, even organic waste value chains (Figure 9), within their ecological, social, economic, cultural and political environments, and analysing how parts and wholes of this system are related.

[4] https://www.merriam-webster.com/dictionary/dictionary
[5] https://dictionary.cambridge.org/dictionary/

Food value chains are the network of actors and activities that bring a basic agricultural product from inputs and infrastructures supply and production in the field to final consumption (FAO, 2005; IPES-Food, 2017; Ingram, 2019; BASIC, 2024). They include activities of storing, transporting, processing, retailing and disposing.

Plastics used in agriculture are those used to grow crops and raise livestock and poultry on farms, upstream of food value chains (Figures 8C and 9). They encompass non-exhaustively (i) plastics contributing to forage conservation, *i.e.*, stretch films to wrap haylage bales, silage films to cover silage, twines, and bale nets to tie hay bales; (ii) plastics contributing to protected cultivation, *i.e.*, greenhouses, small tunnels, mulching films, pipes and frameless direct covers; (iii) controlled-release fertilisers, fertiliser additives, coated seeds and controlled-release plant protection products (PPPs), which are considered as intentionally added MPLs; (iv) plastic sales packaging for, *e.g.*, fertilisers, seeds, PPPs, and hygiene and veterinary products; and (v) plastics in livestock and poultry buildings such as plastic floors and building envelopes.

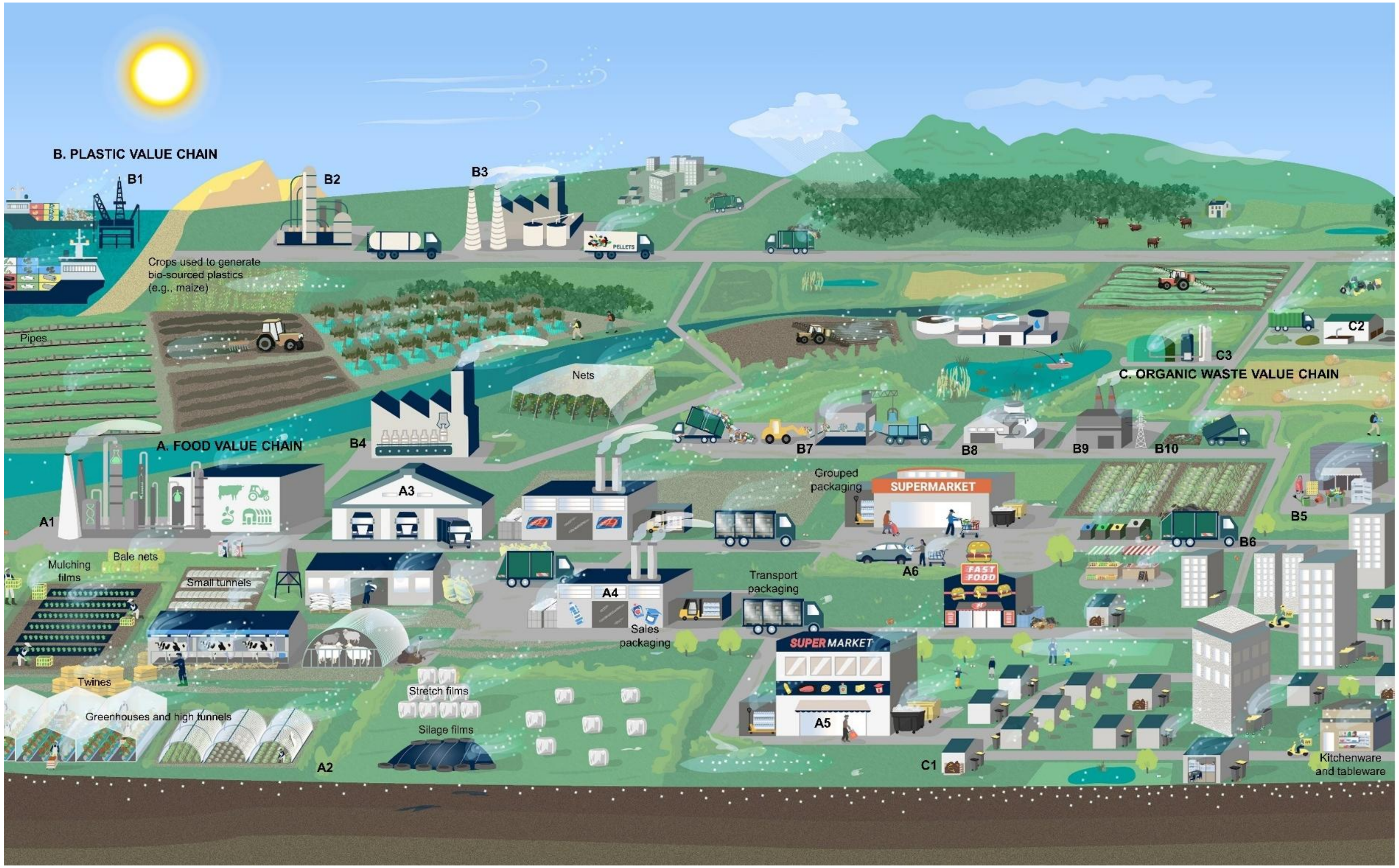

Figure 9. *System of plastics used in agriculture and for food considered in the CSA. Visible macro-plastics, sources and dispersion of invisible micro- and nano-plastics are represented.* ***A. Food value chain****. A1. Agrochemical inputs & infrastructures. A2. Producing crops & livestock and poultry. A3. Food storing & transporting. A4. Processing. A5. Retailing. A6. Consuming.* ***B. Plastic value chain****. B1. Raw material extraction. B2. Refining. B3. Polymerisation & processing. B4. Plastic product manufacturing. A1 to A6. Uses in agriculture and for food. B5. Reuse (*réemploi *and/or* réutilisation*). B6. Collection. B7. Sorting. B8. Recycling. B9. Waste-to-energy incineration. B10. Landfilling.* ***C. Organic waste value chain****. C1. Home composting. C2. Industrial composting. C3. Anaerobic digestion.*

Plastics used for food are those used for food in the regulatory sense, *i.e.*, including drinks and excluding crops prior to harvesting and animals prior to slaughtering (Regulation (EC) 178/2002 (2002b). They are used in logistics (storing and transporting), processing and retailing activities, downstream of food value chains (Figures 8D and 9). They include (i) food contact plastics that are plastic sales packaging (primary plastic packaging) and any other plastics coming into contact with food during food production, processing, storage, preparation and serving, such as machinery to process food, and kitchenware and tableware; and (ii) grouped and transport plastic packaging, also referred to as secondary and tertiary plastic packaging, respectively (Directive 94/62/EC (1994). Food contact plastics require specific properties for food preservation and safety, framed by a specific FCM legislation (Regulation (EU) 10/2011 (EU, 2011)).

The five plastics most produced worldwide in 2019 (Table 1 and Figure 8A), *i.e.*, PE including HD-PE, LLD-PE and LD-PE, PP, PVC, PET and PS are also the most used in agriculture and for food (FAO, 2021; Plastics Europe, 2022).

Similar to food value chains, plastics value chains are the network of actors and activities that bring a plastic product from raw material extraction or generation from natural resources (for petroleum- and bio-based plastics, respectively) to final consumption (Hsu *et al.*, 2022). They include activities of reusing plastics, and post-using plastics, *i.e.*, managing plastic waste (Figure 9). Both plastics used in agriculture and for food include single-use (used once) and multiple-use or reusable plastics (with reusable not necessarily equivalent to durable). By reuse, we mean *réemploi* and/or *réutilisation* as defined in the French Environmental Code (Code de l'Environnement français, art. L541-1-1) and not recycling, these strategies being often confused in the literature, although different in many ways including impacts (3.5). By plastic waste management, we mean the collection and sorting of plastic waste as well as their treatment: reuse (*réutilisation*), closed-loop and open-loop recycling, waste-to-energy incineration and landfilling of plastic waste, but not their 'end-of-life', supposing it exists.

The so-called life of plastics or 'life cycle' of plastics – the appropriateness of the term cycle being debated as circularity in the system of plastics is (3.5) – starts and continues in different forms, as plastic particles and plastic-related compounds (IAS and NIAS), long after their intended uses and well beyond the implementation of plastic waste management strategies. Plastics show up in organic waste and thus in the organic waste management stages that are home composting, industrial composting and anaerobic digestion (AD) (Figure 9), and disperse in the environment. Studying impacts of the system of plastics used in agriculture and for food, taking the whole life of these plastics into account, thus implied studying plastic materials and added substances flows throughout food and plastics value chains, organic waste value chains and their ecological environment.

Finally, to understand uses, resulting impacts and analyse the way in which these are assessed and taken into account, the sociotechnical nature of the system of plastics used in agriculture and for food and of the broader system of plastics it is embedded in was looked at. The interrelatedness of social and technological aspects of these systems was recognised, incorporating artefacts (material technologies) and technical knowledge in the concept of technology (Bergek *et al.*, 2008). The system of plastics used in agriculture and for food is in fact shaped by and shapes network of actors that make it up: petrochemical, plastic (product designers and manufacturers), agrichemical, food processing, retail, and food service companies, farmers, final food consumers, waste collectors, sorters and recyclers, but also scientific and political actors.

Definitions considered

<u>Food contact materials</u> (EC, nd)
Materials and articles are called food contact materials (FCMs) when they come into contact with food during food production, processing, storage, preparation and serving. Such materials and articles include food packaging and containers, machinery to process food, and kitchenware and tableware.

<u>Packaging</u> (Directive 94/62/EC (1994)
The European Union defines packaging as consisting of:
a) sales packaging or primary packaging, i.e., packaging conceived so as to constitute a sales unit to the final user or consumer at the point of purchase;
b) grouped packaging or secondary packaging, i.e., packaging conceived so as to constitute at the point of purchase a grouping of a certain number of sales units whether the latter is sold as such to the final user or consumer or whether it serves only as a means to replenish the shelves at the point of sale; it can be removed from the product without affecting its characteristics;
c) transport packaging or tertiary packaging, i.e., packaging conceived so as to facilitate handling and transport of a number of sales units or grouped packages in order to prevent physical handling and transport damage. Transport packaging does not include road, rail, ship and air containers

<u>Waste</u> (Directive 2008/98/EC (2008a)
'Waste' means any substance or object that the holder disposes of or is required to dispose of pursuant to the provisions of national law in force.

<u>Reuse</u>
By reuse, we mean réemploi *and/or* réutilisation *as defined in the French Environmental Code (Code de l'Environnement français art. L541-1-1) and not recycling.*
Whenever possible, the distinction between réemploi *and* réutilisation *is made in the text. French terms were used to limit confusion and because satisfactory English translation were not found in the literature.*

<u>Réemploi</u> *(Code de l'Environnement français art. L541-1-1)*
Means any operation by which products or components that are not waste are used again for the same purpose for which they were conceived.
Examples for plastics: multiple-use plastics, LDPE shopping bag, in, e.g., Germany, returnable PET bottles that are cleaned to be reused.

<u>Réutilisation</u> *(Code de l'Environnement français art. L541-1-1)*
Means any operation by which products or components that are waste are used again.
Examples for plastics: repaired disposed small kitchen appliances (e.g., repaired disposed plastic kettle)

<u>Recycling of plastics</u>
'Recycling' of plastics means any recovery operation by which plastic waste materials are reprocessed into products, materials or substances whether for the original or other purposes. It does not include biological conversion, energy recovery and the reprocessing into materials that are to be used as fuels or for backfilling operations.

<u>Closed-loop recycling</u> (adapted from Larrain et al. *(2020))*
In closed-loop recycling, the recycled material can substitute the virgin material and be used to create a new version of the same product.

<u>Open-loop recycling</u> (adapted from Larrain et al. *(2020))*
In open-loop recycling, the properties of the recycled material differ from those of the virgin material, so that it cannot be used to create a new version of the same product.

<u>Life cycle of plastics</u> (adapted from ISO14040/2006 (2006))
Consecutive and interlinked stages, from raw material extraction or generation from natural resources (for petroleum- and bio-based plastics, respectively), use and waste management to their environmental fate.

Sociotechnical system (Gille, 1979; Rip et Kemp, 1998; Simondon, 2012)
A sociotechnical system refers to the combination of social practices, technologies, regulations, and cultural norms that collectively shape and govern a particular domain or sector. In essence, a sociotechnical system encompasses both the social and technical dimensions of a system, recognising that they are deeply intertwined and mutually constitutive. Sociotechnical systems are not static entities but are subject to evolution and adaptation over time in response to internal and external pressures, technological innovations, policy interventions, and shifts in societal values or preferences.

3. Key messages

3.1. Plastics have spread throughout food value chains, in particular downstream of production

3.1.1. Historical perspective. The Plastic Age: from drivers of plastics introduction in food value chains to the changes they drove

(based on Part I of the extended report)

Given the intertwined growth of the oil industry, agricultural intensification, and consumerist infrastructure that characterise the spread of plastics during 20th century, these interconnected systems reinforce each other in a form of sociotechnical lock-in (Rip et Kemp, 1998)[6]. Consequently, transitioning away from current consumerist lifestyle or disposable culture that have been driven by plastics uses would require more than just reducing or banning plastics; it necessitates a comprehensive shift in societal values and practices.

To explain why and how plastics have spread through the food value chains during the 20th century, historical literature provides valuable context for assessing the drivers associated with the extensive use of plastics today. Plastics' pervasive presence in contemporary society is the result of an intricate interplay between technological advancements, economic imperatives, and cultural shifts over time. By tracing the evolution of plastics from their inception to the present day, this CSA gains insights into the factors that have shaped their development, influenced their applications, and contributed to their ubiquity in the food value chain.

If major inventions and innovations occurred in the first half of the 20th century, the second half of the 20th century can be called the 'Plastic Age' because of the acceleration of plastic production, after World War II, with a sharp rise of production in the 1970s that continues until today. Most of the identified historical literature is focused on North America, where plastics were first produced and consumed at an industrial scale.

3.1.1.1. From mass production to mass consumption: plastics changed everything

The integration of plastics into food value chains has accompanied a transformative phase that reshaped their dynamics from the interdependency between production sites of food products and their consumption sites to a metabolic breakdown, between growing cities and peripherical agriculture (Barles, 2005). Where at the very end of 19th century, cities and their agricultural suburbs were still following synergistic material flows exchanges, demographic growth in cities has led to an increase in demand for food, unable to be followed by proximity producers, resulting in an increase in the distance between production and consumption sites, and growing thus transportation issues. Consequently, historical literature shows that before the advent of plastics, food value chains provided favourable contexts for their introduction. Driven notably by demographic dynamics, the evolving needs of agricultural productivity, coupled with the demand for efficient food storage and transportation solutions, created a conductive environment for plastics.

[6] Sociotechnical lock-in refers to a situation in which a particular technology, in this case plastics, becomes dominant and resistant to change, when social and technical factors reinforce each other, creating barriers to the adoption of new technologies or practices, such as established infrastructure, vested interests, standardisation, users habits and cultural norms.

3.1.1.2. Plastics changed food value chain starting with downstream

Historical literature shows that the advent of plastics has profoundly reshaped the food system, altering both its structure and the nature of the food we consume. Traditionally, paper, glass and metal were the materials of choice for ensuring the safe circulation and conservation of food. These materials had established a reliable system where urban regions efficiently metabolised material flows, and waste was not really existing (Barles, 2005). Plastics not only changed existing practices but also gave rise to new ones, changing various aspects of the continuum going from the farm to the fork (especially farming, food storage, distribution and retailing).

Plastic packaging made its debut in the 1930s, coinciding with the rise of supermarkets and advancements in refrigerated storage and transportation systems (Hachez-Leroy et Fridenson, 2019). This trio of developments – plastic packaging, supermarkets, and refrigeration – catalysed a significant shift in the food system. Its lightweight, durable, and versatile nature made plastics an attractive choice for packaging, leading to its widespread adoption across various sectors of the food industry, notably from logistics to retailing. They not only encouraged but also widened the gap between food producers and consumers, as well as between rural and urban citizens. Urbanisation and expansion of packaging are intrinsically linked processes that have been clearly influenced by the rise of plastics.

The packaging industry has demonstrated great skills in exploiting the technical potentials of plastics to develop packages that are attractive to food producers, distributors, and consumers. Contemporary uses of plastics in packaging are the result of over a hundred years of developments in knowledge, technique, production of polymers, additives and other chemical substances, consumption, and waste management, developments that have actively participated in shaping the supply, demand, and consumption of food. Omnipresent at every stage of food chains, plastics have actively participated in the development of food value chains and markets at all scales (Figures 9A and 10).

Figure 10. *Food value chain in the system of plastics used in agriculture and for food considered in the CSA: from inputs and infrastructures supply and production in the field to final consumption. Activities of storing, transporting, processing and retailing are also shown.*

Plastic packaging emerged within a specific sociotechnical and economic context, quickly establishing itself as a pivotal market device in new systems of mass consumption: plastics contributed to global commodification processes (Hawkins, 2011). The expansion of mass consumption increased the economic pressure on packaging materials and food transport. In bottled water industry for instance, because deposit system represented an important activity requiring employees and financial resources for all the marketers, disposable plastic bottles were quickly adopted between the 1970s and the 1980s.

The introduction of plastics had a profound impact on the balance between private and public costs in the distribution of food. For instance, the rise of fast-food chains, which heavily rely on plastic packaging, allowed private sectors to save on wages but resulted in increased waste treatment costs for the public sector. This shift in cost dynamics highlights the complex economic implications of plastic's integration into the food system.

Plastic has undeniably revolutionised the food system, changing our eating practices, redefining the relationships between consumers and products – standardised pieces of meat and fish are no longer viewed as animal parts but can be visually inspected by customers, preventing the use of other senses like touching or smelling (Hisano, 2019) – while also modifying the very nature of the food itself: for instance, composition of processed food has been modified for the purpose of adoption to the plastic wrap (wrapped bread for instance in Hachez-Leroy (2019). Plastic packaging has been instrumental in the development of new food products, such as minimally processed fruits and vegetables preserved under modified atmospheres (Del Nobile *et al.*, 2007). It has also supported alternative food formats like multi-compartment containers for ready-to-eat meals then has been essential for enabling new modes of distribution, such as fast-food restaurants, and new social habits (for instance on-the-go consumption practices).

PET Bottles as a disruptive innovation

As Marty (2014) showed, the transition from glass to plastic bottles in the bottled water industry was a pivotal moment in the development of mass consumption and the socio-economic organisation of beverage distribution. Prior to the 1970s, glass bottles were the dominant packaging material for bottled water. This limited distribution to high-end venues like hotels and restaurants, as deposit system required significant financial resources and personnel.

A French leading bottled water brand sought to expand its market reach by replacing glass bottles with plastic. This shift aimed to streamline distribution and eliminate the need for return policies. However, initial concerns about the health implications of PVC led to its replacement with PET, a safer and more versatile plastic. PET bottles offered several advantages, including lightness, strength, and clarity. These properties made them more suitable for transportation, particularly in the aviation industry, where weight reduction was crucial for fuel efficiency. In 1986, DuPont's engineer who invented the PET bottle estimated fuel savings alone around "$25,000 a year or more for the typical passenger airliner" carrying beverages in PET packaging (Wyeth, 1986).

The introduction of plastic bottles also reshaped the consumption of water. By making water more accessible and convenient, PET bottles contributed to the 'commodification of water' and the integration of packaged water into everyday life (Hawkins, 2011). While the switch to plastic bottles provided economic benefits, it also introduced new problems, particularly in terms of waste management. The shift from glass to plastic generated a new type of waste that local communities had to manage and finance.

In brief, the evolution of bottled water packaging from glass to plastic not only transformed the distribution and consumption of bottled water but also played a role in shaping the socio-economic landscape of the beverage industry and public waste management.

3.1.1.3. Plastics changed agriculture

Two parallel trends that emerged in the post-World War II era drove plastic integration into agriculture: the reconversion of the plastic industry in search of new markets and the intensification of agriculture fuelled by the productivity fervour of the post-war decades. Plastics production increased rapidly during the Second World War, and the industry sought outlets for its products, particularly in the food value chain.

In France, the adoption of plastics in agriculture was not merely a random occurrence but was strategically facilitated by policy frameworks such as the European Economic Community's 1962 agricultural policy. This 'green revolution' policy aimed at enhancing agricultural productivity by promoting technical progress and by ensuring the rational development of agricultural production and the optimal utilisation of the factors of production, in particular labour. This policy thus aimed at ensuring a stable supply of affordable food, making the diffusion of plastic greenhouses, tunnels, and mulches an integral part of its implementation. While agricultural plastics may not have originated from ground-breaking inventions, they spurred technological innovations through efforts to involve together plastic manufacturers, farmers, and research institutes. This collaborative approach also extended to public administration, which played a crucial role in standardising and certifying plastic products to meet quality and safety standards (CIPA, 2000).

Plastics have deeply transformed agricultural practices that existed before their introduction essentially through a substitution process to other materials. Initially emerging in the United States of America (USA) in the 1950s, their use in agriculture expanded to Japan and Northwestern Europe in the 1960s, the Mediterranean in the 1970s, and China and Southeast Asia in the 1980s. Before the 1960s, irrigation and drainage relied on furrows, open canals, and metal or rubber pipes, with drainage using terracotta or porous concrete: the advent of plastics offered cheaper and more long-lasting alternatives. Concerning plant protection, traditional materials such as glass greenhouses, oiled paper, and organic covers like straw and bark were progressively replaced by plastics, which provided more effective solutions. In Northern Europe, steam heating and fumigation methods were enhanced by plastics for better frost protection. In cattle breeding, plastics replaced metal or wood containers and improved silage methods, making fermentation more efficient with plastic-wrapped systems. This transition began in the 1950s in the USA and the United Kingdom (UK), and later spread to France, where it gradually replaced traditional grass silage methods by the 1990s.

Plastics not only transformed agricultural practices, they also generated new ones. The introduction of these agricultural plastics went along with the adoption of new intensive agricultural practices and production systems. The massive use of plastic tunnel greenhouses, for instance, supported a *Californian agricultural development model* focused on productivist innovations that transform the land. Intensive greenhouse and mulch cultivations yield products tailored to meet the stringent requirements of large-scale buyers in export markets, driving productivity gains and facilitating globalisation within a liberalised economic system (Davis, 2022). Plastic mulches have emerged as cost-effective tools, promoted for their dual benefits of sheltering and moisturising plants while simultaneously curbing pesticide usage. By the 1970s, plastic mulching had taken off, leading to a rapid expansion of plastic use in agriculture worldwide as both new inputs (mulches) as a new material for infrastructures building (greenhouses, tunnels or irrigation systems). By 2013, 3.6 Mt of plastic were used in agricultural films, predominantly in Asia and Europe (Orzolek, 2017).

In France, PE plastic mulches contributed to intensive cultivation of fruits and vegetable, notably melon and strawberries for the global market. Similarly, plastic silage developed in central regions and spread across the country, with PE being the most common plastic used for this purpose, and PP and PVC used for tying bales and irrigation, respectively.

This transition also widened the gulf between peasants and city dwellers, reflecting broader socio-economic disparities and contrasting lifestyles between rural and urban communities. Moreover, the development of these 'plasticultures' – as promoters of plastics uses in agriculture (*e.g.*, the CPA) first referring to the use of plastic infrastructures to grow crops, plastic greenhouses in particular – intensified the need for seasonal human labour force, especially during harvest time (Domingo, 2007; Hellio, 2008; Avola, 2022). Limiting year-round agricultural employment, plastic greenhouses worsen the conditions of employment in agriculture (*e.g.,* favouring low skill and temporary jobs).

The incorporation of plastics into agriculture exemplifies the intricate interplay between technological advancements, policy frameworks, and socio-economic dynamics within the food value chain.

3.1.1.4. Petrochemical and packaging industries promoted a new way of life, based on plastics uses

The rise of plastics is intricately linked to the strategies and growth of the petrochemical industry, representing a symbiotic relationship that has fuelled the rapid expansion of plastic uses. As by-products of oil cracking for fuel production, mass production of plastics has been seemingly abundant and cost-effective. This synergy between the oil and plastics industries was further amplified by substantial investments in industrial and public research aimed at developing new chemical substances, manufacturing processes and thus new markets. Industrial research and development laboratories played a pivotal role in the synthesis and development of plastics, driven by profit motives and commercial competition (Davis, 2022). During the first half of the 20th century, while academic research circles were embroiled in theoretical debates about the existence and nature of polymers, industrial research was driven by the need to solve practical problems and meet market demands (Hargittai, 2012). Petrochemical industry pushed for the expansion of plastics uses, while more and more retaining under the principle of industrial secrecy a set of data, particularly resin formulation data, capable of informing both public opinion and legislators about the composition and potential adverse effects of these new materials (Markowitz et Rosner, 2013). Additionally, marketing divisions were established to enhance the sociotechnical properties of plastics, with narratives that positioned plastics as innovative, versatile, and indispensable commodities for modern living.

The entanglement of plastics with modern societies is evident in the profound impact it has had on consumption practices, production processes, and commercialisation strategies. Plastic industries introduced new consumers' practices that significantly boosted the production of single-use commodities (Guien, 2021; Ghosh, 2022). Furthermore, the food and packaging industry played a pivotal role in developing innovative products (for instance packaged single-use and individual portions) and pushing them into the market through strategic marketing campaigns. These campaigns propagated a sociotechnical imaginary where plastics were portrayed as agents of affluence, democratisation, economic growth, and modern way of life, especially for women (Meikle, 1997). During the 1950s and 1960s, plastics played a pivotal role in shaping a culture of disposability. The mass consumption of single-use disposable commodities became synonymous with convenience and modernity (Guien, 2021). Plastics were deliberately designed to be 'made to be wasted' (Meikle, 1997; Hawkins, 2011), further reinforcing their association with consumerism and the accumulation of waste (Gabrys *et al.*, 2017).

Through the 20th century, plastics have not only contributed to shape consumerist practices but also transformed the relationship between humans, time, and space. The convenience offered by plastic packaging has conditioned consumers to expect quick and easy access to food, regardless of geographical or temporal constraints. This has fostered a culture of instant gratification and convenience, further distancing consumers from the origins and processes involved in food production.

The widespread consumption of plastics and the disposable culture it fosters often divert attention away from the historical origins (oil extraction) and future consequences (waste) of these ubiquitous materials. Today's consumption patterns are shaped by the remnants of past fossil fuel use, while the environmental impacts of current plastic production will leave a lasting imprint on future generations. As plastics production erodes the Earth's natural archives, the waste it generates becomes 'technofossil', a lasting legacy of the 20th century (Bensaude-Vincent, 2022; Davis, 2022).

3.1.2. Today, downstream food value chain and mainstream agriculture rely on plastics

(based on Chapter II.1 of the extended report)

3.1.2.1. Plastic packaging is a market device that makes food manageable in downstream food value chain

Packaging is a coordinated system of preparing goods for transport, distribution, storage, retailing and end use, a means of ensuring safe delivery to the ultimate consumer in sound condition at optimum cost and a techno-commercial objective aimed at optimising the costs of delivery while maximising sales and thus, profit (Coles et Kirwan, 2011). Commission Regulation (EU) 10/2011 of 14 January 2011 (2011) distinguishes primary packaging (*i.e.*, packaging conceived so as to constitute a sales unit to the final user or consumer at the point of purchase, this category includes FCM packaging); secondary packaging (*i.e.*, packaging conceived so as to constitute at the point of purchase a grouping of a certain number of sales units whether the latter is sold as such to the final user or consumer or whether it serves only as a means to replenish the shelves at the point of sale; it can be removed from the product without affecting its characteristics); tertiary packaging (*i.e.*, packaging conceived so as to facilitate handling and transport of a number of sales units or grouped packages in order to prevent physical handling and transport damage. Transport packaging does not include road, rail, ship and air containers). Most of the identified literature focuses on FCM issues, thus on primary packaging.

Considering scientific literature dedicated to logistics and value chains, plastic packaging operates as a market device (Callon *et al.*, 2007), serving as material artefacts that facilitate and enable market exchanges. In this capacity, plastics play a key role in transporting agricultural and food products to consumers – however far they are from production sites – through logistics, transportation and commercialisation processes, ensuring that goods can be efficiently distributed, sold, and finally consumed. These packaging materials play thus a structuring role in driving food chain activity. Plastic packaging materials are not just passive containers: they not only have actively participated to the construction of contemporary food value chains but they also are active participants in the intricate web of market exchanges that characterise contemporary consumption. Plastic uses are now embedded in the food value chain practices, today's food distribution is thoroughly plasticised.

Beyond their function as market devices, plastic packaging thus stands as the 'skin of commerce' (Hawkins, 2018). This metaphorical description underscores their ubiquity as organic component of every stage and scale of food value chains. From the initial food transport stages to the final user consumption, plastic packaging is omnipresent, acting as a convenient protective barrier that safeguards food products during transit, storage, and display. In other terms, plastic packaging makes food manageable. In a world where convenience and efficiency are paramount, whether it is the resealable pouch that keeps snacks fresh or the durable container that protects delicate produce, plastic packaging plays a pivotal role in delivering food products to consumers in optimal condition. Its contributions are not limited to a single stage but span the entire lifecycle of food products, from storage and distribution to consumption and disposal. These packages streamline various handling activities in stores, such as receiving, storing, putting away, and order picking, thereby playing a key role in optimising time and cost of human labour across the value chain.

Overall, plastic packaging serves as a linchpin that holds together the complex network of activities and stakeholders involved in bringing food from farm to table and thus contributes to shape both processes and plasticity of contemporary global food system (from production to consumption).

Corporate strategies are drivers of plastic packaging uses

Considering literature dedicated to historical drivers and value chain analysis, the use of plastic packaging in food value chains extends beyond the realm of consumer choice, reflecting a broader industry-driven evolution (both agri-food and plastic packaging industries) beyond than individual preferences.

At a global scale, most plastic packaging are single-use, meaning they are discarded after one use. This results in a significant loss of value to the global economy, estimated in 2018 by the Institute of European Environmental Policy, a sustainability think tank, at around 100 billion euros annually (Schweitzer *et al.*, 2018). While paper and cardboard are materials still frequently used for packaging in general, plastic industry declares that even if plastic is accounting for more than half of all goods packaging in Europe, it represents only 17% of the total packaging weight, a weight that has decreased by 28% over the past decade (Plastics Europe, 2023).

Despite the ubiquitous presence of plastic packaging in our daily lives, most of their effective objectives of use are often poorly documented in scientific literature, which often studies physico-chemical properties without liking them to their effective uses. Instead, much of what is known about practicality of using plastic packaging comes from plastic packaging industry assertions rather than academic research.

Contrary to many consumer goods that are deliberately purchased, plastic packaging are rarely the object of intentional buying. Instead, consumers predominantly encounter plastics when purchasing packaged food, making their presence more a consequence of choices made by manufacturers and retailers than of consumer demand.

The increased prevalence of plastic in packaging is a deliberate and industry-driven evolution, responding to the objectives, constraints, and practices that the industry sets for itself (3.1.1.4), a value chain corporates' calculated strategy aimed at optimising logistics efficiency, reducing costs, and meeting regulatory requirements.

Combined qualities of plastic packaging – lightweight, mechanically strong, cheap, and convenient – are unique and partially explain the increase in plastic packaging uses

The distinction between the objectives of use for food packaging in general and plastic food packaging in particular is often blurred in the literature. While the general objectives of packaging – whether it is to protect, preserve, or promote – are generally understood, the specific role and attributes of plastic as a material are not always clearly documented. This ambiguity can lead to misconceptions and misunderstandings about the unique benefits and challenges associated with using plastic for food packaging purposes.

Nevertheless, literature points that the combined qualities of plastic packaging, which include being lightweight, mechanically strong, cheap, and convenient, set them apart from other packaging materials: plastic packaging are thus highly versatile and well-suited for a wide range of applications in the food industry where preserving freshness and ensuring product integrity are crucial. These combined qualities make plastics incomparably competitive. Interestingly, the fact that plastics are cheap and available is a selling point for the plastic industry that is abundantly communicated to commercial players in the value chain, but not to consumers. Economic competition among corporate actors, enhanced mobility of products (*e.g.*, the development of online shopping) and users (*e.g.*, consumption on the move), demographics changes, risk management by regulators and users and corresponding offering of safety by the industry, and a sustained interest among consumers for convenience and novelty are just a few of the drivers that can explain the increase of uses of plastics in packaging today (Simoens *et al.*, 2022).

Plastic food packaging plays an important role in food safety, as rigorously defined and regulated by FCM regulations (notably Regulation (EU) 10/2011). Thus, beyond these regulatory obligations, plastic food packaging serves practical purposes that directly benefit value chain actors, notably retailers, through its ability to extend a product's shelf life. Plastic packaging helps prevent cross-contamination, spoilage, and degradation, safeguarding the integrity and safety of the food contained within. By creating a protective barrier that seals out contaminants and minimises exposure to environmental factors like moisture and air, plastic packaging helps maintain the quality and freshness of food products, thereby reducing losses in the distribution process. This is particularly valid for highly perishable food such as soft fruits and vegetables, dairy, fish, and meat products. For other less perishable food, this argument needs to be moderated. In any cases, as pointed by the Institute of European Environmental Policy, a sustainability think tank, plastic packaging inevitably contributes to increase packaging waste. Interestingly, in Europe, food waste have increased alongside the increase use of plastic packaging (Schweitzer *et al.*, 2018). Moreover, size and shape of plastic packaging may be inadequate and then induce the generation of food waste (Robertson, 2012).

One of the other notable roles that plastic packaging plays is as a marketing tool. In a competitive marketplace where brands are looking for consumers' attention, the design, colour, and even texture of plastic packaging can be strategically leveraged to communicate brand values, product attributes, and unique selling propositions. They also are medium to informative and legal mentions, such as ingredients or expiration date. Whether it is through graphics, informative labels, or innovative shapes, plastic packaging serves as a visual and tactile interface that also engages consumers and influences purchasing decisions.

3.1.2.2. In upstream food value chain, among agricultural systems, mainstream cattle and horticultural systems' practices rely on plastics as an agrochemical input and infrastructure

Among agricultural systems, plastics are particularly deeply embedded in mainstream cattle and horticultural systems. In 2019, more than a half of agricultural plastics quantified by Agriculture Plastic & Environment (APE Europe) were marketed as agrochemical input in forage conservation, *i.e.*, as stretch films to wrap haylage bales (20%), silage films to cover silage (17%), twines (11%) and bale nets (7%) to tie hay bales (Table 1). Sales as packaging plastics (*e.g.*, for fertilisers, pesticides, hygiene products, veterinary products and seeds) and for livestock buildings (*e.g.*, floors and building envelopes in livestock systems) are not included in assessments. The other 45% were marketed in horticultural systems mostly as greenhouse, small tunnel and pipe infrastructures (30% with 17%, 8% and 6% of sales, respectively), followed by mulching films (12%).

Plastics in mainstream cattle systems contribute to forage conservation

As a result of agriculture intensification that began in the 1960s (Sailley *et al.*, 2021); 3.1.1.3 above), cattle systems now heavily rely on conserved maize and grass, and dairy and mixed systems on maize silage in particular. Plastics deeply transformed forage conservation with the use of stretch films to wrap haylage bales and silage films to cover silage in silos. However, although they are major consumers of plastics, livestock production systems received little attention as compared to crop production systems.

The scientific literature on this topic focused on comparing the nutritional quality of maize or grass silage for different types of plastics. Only two experiments studied the effects of plastics used to conserve forages as silage on livestock animal productivity, focusing on dairy heifers (Parra *et al.*, 2021) and feedlot calves (Neumann *et al.*, 2021b). Actual needs and practices of farmers were not considered. Alternative system level solutions that may avoid or limit the need for plastics were not investigated.

Plastics in mainstream horticultural systems improve short-term yields of single crops

Assessments of plastics in crop production systems focused on horticultural crops (almost 90% of publications) and tomatoes in particular (14% of publications) – the area (planted mostly in Italy and Spain) and market value of which was the highest among fresh vegetables in the European Union (EU) (Eurostat, 2022)[7]. Most assessments (three quarters) were concerned with the use of plastic mulches followed by plastics used as tunnel coverings (one fifth). Paradoxically, the widespread use of plastics can make them invisible. Their use is not always explicitly mentioned in the literature, as in the case of irrigation systems.

Studied objectives of plastics use were chosen by academics and actual reference to the needs of farmers was rarely made. Publications considered common agronomic objectives with the general objective of improving yield using plastics being the most investigated, followed by improving water use, modifying the temperature conditions and contributing to weed control. Although the high labour intensity of horticultural products likely drove the replacement of labour using plastics technological solutions (Pekkeriet et van Henten, 2011), farmers' socio-economic objectives and constraints were overlooked in the literature.

A range of agrochemical inputs and infrastructures technologies, among which plastics, applied to a single crop were usually compared at the plot or field level, based on on-station experiments conducted in one or two growing seasons rather than under on-farm conditions and for extensive periods.

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[7] https://ec.europa.eu/eurostat/statistics-explained/index.php?title=The_fruit_and_vegetable_sector_in_the_EU_-_a_statistical_overview#Fruit_and_vegetable_production%20r

Landscape, farm and cropping system level solutions that may avoid or limit the need for plastics were not investigated, *e.g.*, diversified rotations to control soil-borne pests and diseases as an alternative to plastic solarisation. Such limitations raise the question of results extrapolation over the long term and to real-life conditions, and imply that they need to be put into perspective.

Within these limitations, in mainstream monocropping or low diversified cropping systems and at least in the short-term, the use of plastics often increased yields by means of controlling crop environmental conditions in particular temperature, water supply and weeds. Plastic mulches can raise or lower the soil and canopy temperature to make it more favourable to crop growth. Plastics used as tunnel coverings can also protect crops from low temperatures, extend the growing season or provide shade to lower temperature around the crop. For crops grown under tunnel coverings, which are entirely reliant on piped water, combining drip irrigation with plastic mulches can improve water use efficiency. Plastic mulches can inhibit the growth of weeds physically or prevent their germination or growth through the exclusion of light or by heating the soil beneath to destroy seeds or vegetative propagules. However, their potential to substitute for herbicides and hand weeding remains under-researched. It also remains that trade-offs between short-term yield benefits and environmental impacts of plastic production, use and disposal on, *e.g.*, soil and water quality are not analysed in the literature.

3.2. The plasticity of plastics encourages complexity: lots of ingredients to reach and balance various properties, functions and uses

(based on Chapter II.2 of the extended report)

3.2.1. Playing with polymers and additives in formulations

The uses and objectives of use of plastic materials in agriculture and for food strongly influence the type of plastics to use, *i.e.*, the nature of the polymer, the types and quantity of additives including fillers, processing aids and functional additives as well as the processes used to process the materials into an object (a film, a bottle, etc.).

The properties and characteristics of all polymers are linked to several factors, such as nature, chemical composition, and arrangement of their constituents (monomers), and can be ad hoc designed for specific applications. The structure of the polymer is controlled by its synthesis, also called polymerisation. In some cases, grafting of functional moisture or reaction between two polymer chains are carried out to obtain specific properties. The properties of polymers are thus linked with their chemical composition and structure.

The five plastic polymers most used in agriculture and for food include PE, PP, PVC, PET, and PS. Many other polymers are also used, including, *e.g.*, polycarbonates (PC), polyamides (PA, Nylon®), acrylics, PLA, polyurethane (PUR or PU), ethylene-vinyl alcohol (EVOH), and ethylene-vinyl acetate (EVA). Their chemical composition and structure are described in Table 3. The example of PE (HDPE, LDPE and LLDPE) is further detailed to illustrate how the properties of a polymer containing the same monomer (ethylene) can change when the structure is different. As a consequence, considering the polymer as the main ingredient of a plastic, it is not so obvious to replace one polymer by another one because different uses require different properties.

Table 3. *Chemical composition and structure of plastic polymers commonly used in agriculture and for food.*

Polymer name	Chemical formula	Structure	Chemical family
LDPE (d =0.910–0.925)	-[CH2]n1-[CH-(CH2)p1)-CH3]q1-	Highly branched Semi-crystalline	Polyolefins
LLDPE (d=0.910 to 0.940)	-[CH2]n1-[CH-(CH2)p2)-CH3]q2- with p1>p2	Highly branched Semi-crystalline	
MDPE (d =0.926–0.940)	-[CH2]n3-[CH-(CH2)p3-CH3]q3- with $n_1<n_3$ and $p_1>p_3$ and $q_1>q_3$ (less branching)	Branched Semi-crystalline	
HDPE(d =0.941–0.959)	-[CH2-CH2]n-	Linear Semi-crystalline	
PP	-[CH2-(CH-CH3)]n-	Linear Semi-crystalline	
EVOH	-[CH2-CH2]n-[CH2-CHOH]p	Semi-crystalline	Vinyl polymer
EVA	-[CH2-CH2]n-[-[CH2-CH(OCOCH$_3$)]p	Semi-crystalline	
PVC	-[CH2-CHCl]n-	Linear Syndiotactic	
PS	-[CH2-(CH-Φ)]n-	Syndiotactic or isotactic – semi-crystalline Atactic - amorphous	
PA66	H-[NH-CO-(CH2)4-CO-NH-(CH2)6]n-NH2	Semi-crystalline	Polyamides
PC or BPA-PC	HO-[Φ-CH3CCH3)-Φ-O-COO]n	Amorphous	Polycarbonates
PET	HO-[C2H4-OOC-Φ-COO-CH2H4]n-OH	Semi-crystalline	Polyester

The example of polyethylene (PE)

Different types of polyethylene (PE) exist and are used in agriculture and for food: LDPE (Low density PE), LLDPE (Linear Low Density PE), and HDPE (High density PE). All of them are produced from the same monomer, namely ethylene. Other alpha-olefins (such as 1-butene, 1-hexene, 1-octene and 1-decene) can be co-polymerised with ethylene, mostly to introduce short side-chains in PE-LLD to modulate its properties, i.e., its density, flexibility, and strength. The density of LDPE varies from 0.910 to 0.925 g/cm³, that of LLDPE from 0.910 to 0.940 g/cm³ and that of HDPE varies from 0.941 to 0.965 g/cm³ (Table 4). The density varies depending on crystallinity rate, i.e., thermal history when processed. This property (and the associated properties of the polymer) is linked to its structure that can be linear or branched (Figure 11). Processing methods used to formulate the polymer into a plastic and process the plastic into an object also affect the properties of the materials.

Table 4. *Properties of LDPE, LLDPE and HDPE from various producers. (Shen et al., 2009)*

a) Braskem data sheet for polymer grade HF0150; b) Braskem data sheet for polymer grade IE59U3; c) Braskem data sheet for polymer grade BS002; d) Brasken data sheet for polymer grade BF-0323 HC; e) Braskem data sheet for polymer grade BI-818; f) Brasekm data sheet for polymer grade EG 0921; g) Dow Technical Information DOWLEX2045; h) Braskem data sheet for polymer grade Braskem FA31; k) Melt Flow Index, g/10 min (190° C/5.0kg) l) ASTM D1238; m) ASTM D1505; n) ASTM D792; o) ASTM D638; p) ASTM D822; q) ASTM D790

	HDPE (Pchem)	LDPE (Pchem)	LLDPE (Pchem)	LLDPE (Pchem)	HDPE (Pchem)	LDPE (Pchem)	HDPE (Pchem)	LDPE (Pchem)
Application	Film extrusion				Injection moulding		Blow moulding	
Polymer type	Braskem HF0150[a]	Braskem BF0323HC[d]	DOWLEX™ 2045 LLDPE[g]	Braskem FA31[h]	Braskem IE59U3[b]	Braskem BI818[e]	Braskem BS002[c]	Braskem EG0921[f]
Melt flow index (190C/2.16kg), g/10min	0.45[k,l]	0.32[l]	1.0[l]	0.75[l]	5.0[k,l]	7.5[l]	0.29[l]	0.9[l]
Density (g/cm³)	0.948[m,n]	0.923[m]	0.920[m]	0.919[n]	0.959[n]	0.918[m]	0.954[m,n]	0.921[m]
Tensile Strength at Yield, MPa	23/27[p]				28[o]	9[o]	27[o]	12[o]
Tensile Strenght at Break, MPa		19[o]		26[o]	26[o]	11[o]	24[o]	
Elongation at Break, %	571/832[p]	715[o]	827[p]	900[o]	>1000[p]	641[o]	>1000[p]	600[o]
Elongation at yield (%)					12[p]		8[p]	
Flexular Modulus (MPa)					1200[q]		1300[q]	

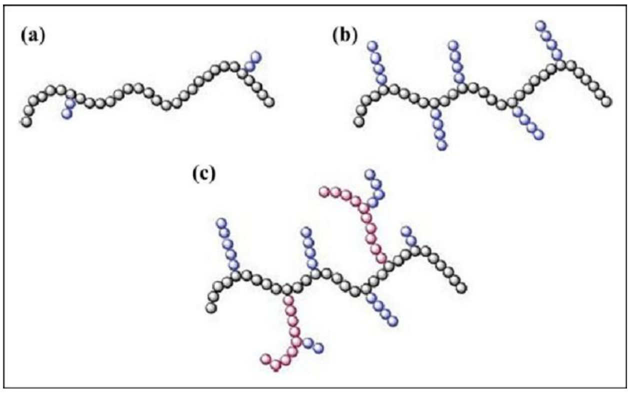

Figure 11. *Structures of HDPE (a), LLDPE (b), and LDPE (c). (Graziano et al., 2019)*

Polymers used in agriculture and for food are mainly sourced from fossil feedstocks (Table 1; 2.4.1). Bio-based polymers can be classified as proposed in Figure 12.

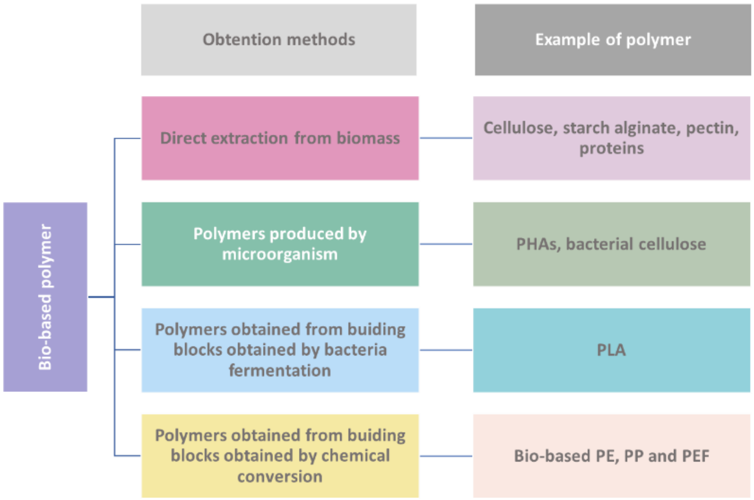

Figure 12. *Classification of bio-based polymers.*
(adapted from Barretto et al. (2024), and Ibrahim et al. (2021))

Regarding synthetic bio-based polymers, two approaches are used. Either the monomer obtained is synthesised from renewable resources and thus the polymer that is obtained is similar to the petroleum-based polymer, or new monomers are used to produce new type of polymers. In the case of common polymers used in agriculture and for food, the first approach mainly concerns bio-PE and PET. For this latter polymer, ethylene glycol (EG), one of the monomers of PET, is readily available starting form bio-based ethanol. Substitution of petroleum-based EG with bio-based EG leads to bio-PET 30, as it contains 30% bio-based share. In 2016, the worldwide production capacity of bio-PET 30 was ca. 95 ktyear. This data has to be compared with global uses of PET packaging (excluding fibres) amounted to a total of 21.7 Mt the same year (Welle, 2018). PLA is a polymer that was developed following the second approach. The primary raw material for PLA production is maize or other industrial plants with a high starch content.

Additives are used in polymers to modulate their properties (and sometimes their cost) as well as to facilitate their processing. These additives are essential to reach properties that the polymer itself does not have and offer the versatility to plastics they are known for. Additives used are classified as plasticisers, flame retardants, pigments, antioxidants, (heat, light, etc.) stabilisers, nucleating agents, antistatic agents, processing aids and additives for other functions (Figure 13). On average, plastics contain 93% polymer resin and 7% additives by mass (Geyer *et al.*, 2017). It is important to note that this data is not limited to the agri-food sector and the content of additives can vary from one application to another one as well as within the same application. As an example, studies report the use of phthalates in agriculture film with amount ranging from tens of milligrams per kilogram to a very high amount of 140 g/kg (Xu *et al.*, 2023). The majority of the agricultural plastic waste in Europe, consisting of LDPE and in some cases LLDPE films coming from protected cultivation, shrink films and silage applications, with thickness ranging from 20 to 250 μm, may contain (reaching up to 15% for some of them) a significant amount of dyes or colourants, additives for UV protection and/or fillers (*e.g.*, mulching films and irrigation pipes and tapes, which contain carbon black).

Similarly to what is observed for polymer, there is a trend concerning additives for plastics where more and more bio-based substitution are considered. Literature review ((Marturano *et al.*, 2023); Figure 14)) shows that a constantly growing number of substances from natural sources and their waste are considered as alternatives to synthetic additives in polymer formulations for specific applications.

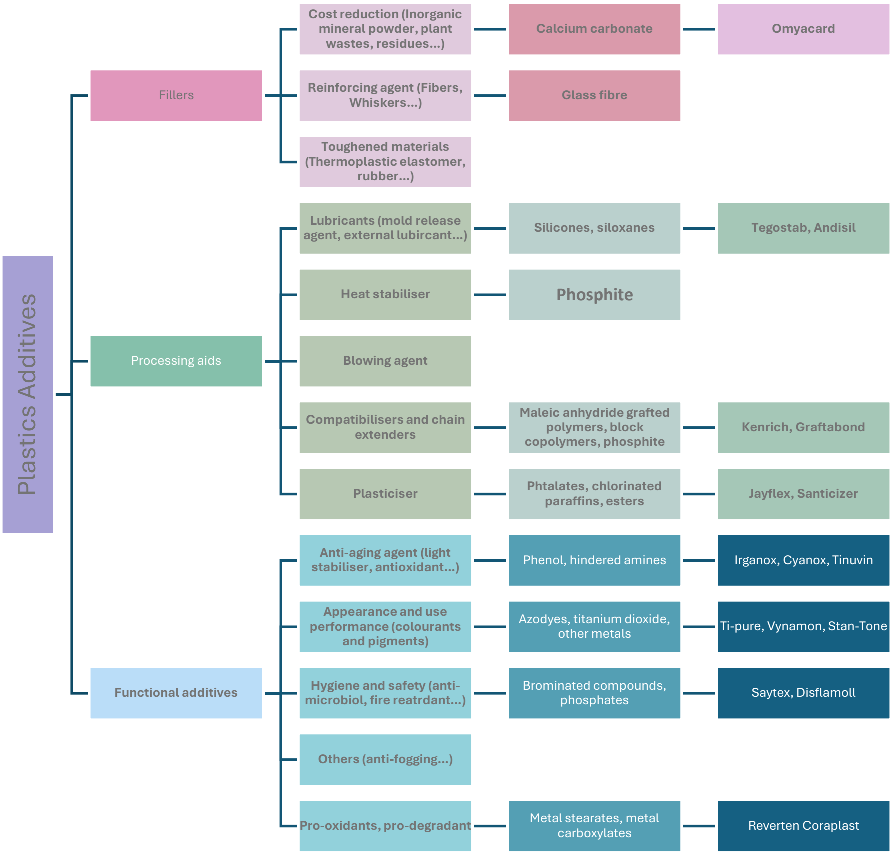

Figure 13. *General classes of plastics additives commonly used in plastics manufacturing. (adapted from Law* et al. *(2024))*

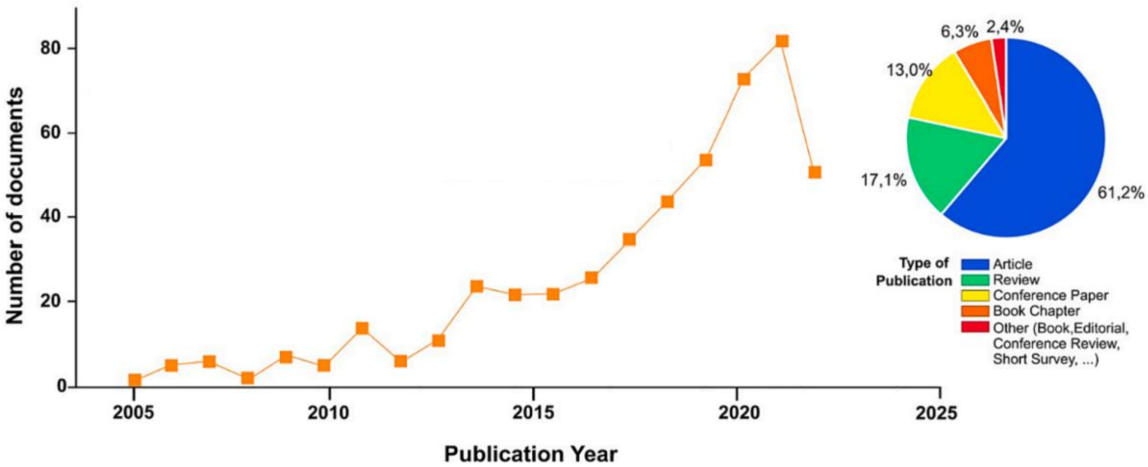

Figure 14. *Number of yearly scientific publications since 2015 regarding the topic of 'bio-based polymer additives'. (adapted from Marturano* et al. *(2023))*

However, in most cases, there is still a need to adapt functionality and improve efficiency. It has to be noted that the development of bio-based solutions does not simplify the formulation of plastics.

Plastics are not only composed of polymer plus additives. They may contain both residual starting substances (*e.g.*, solvent or unreacted monomers), known as IAS, and impurities, unwanted side-reactants and degradation products (*e.g.*, oligomers), known as NIAS (Groh *et al.*, 2019). This is particularly important in the case of FCMs since these substances may migrate to the food. NIAS originate from break-down products of FCMs, impurities of starting materials, unwanted side-products, and various contaminants (Geueke, 2018). It is generally accepted that only compounds below 1000 Da are considered as NIAS. Substances with a higher molecular weight are regarded as inert towards migration due to their larger sizes. Oligomers, defined as molecules consisting of a few monomer units, are regarded as the primary form of NIAS or IAS in polymers. They arise from the production of polymer or from its degradation either during its process or its service life. They may be present in significant quantities within the polymer matrix. Studies have reported that total oligomers can constitute as much as 0.5~1.3% in PET (Shi *et al.*, 2023).

Plastics in general and plastics used in agriculture and for food are based on a century of research of development of polymer formulations. The chemical agents, including additives, are incorporated into the polymer from its manufacture and throughout its life cycle (Figure 15). NIAS are also potentially present at various stages of the life cycle of plastics. The chemical complexity of the materials is thus increasing within the life time of plastics leading to a difficulty to track its composition along its whole life cycle. This unknown composition directly affects recycling processes.

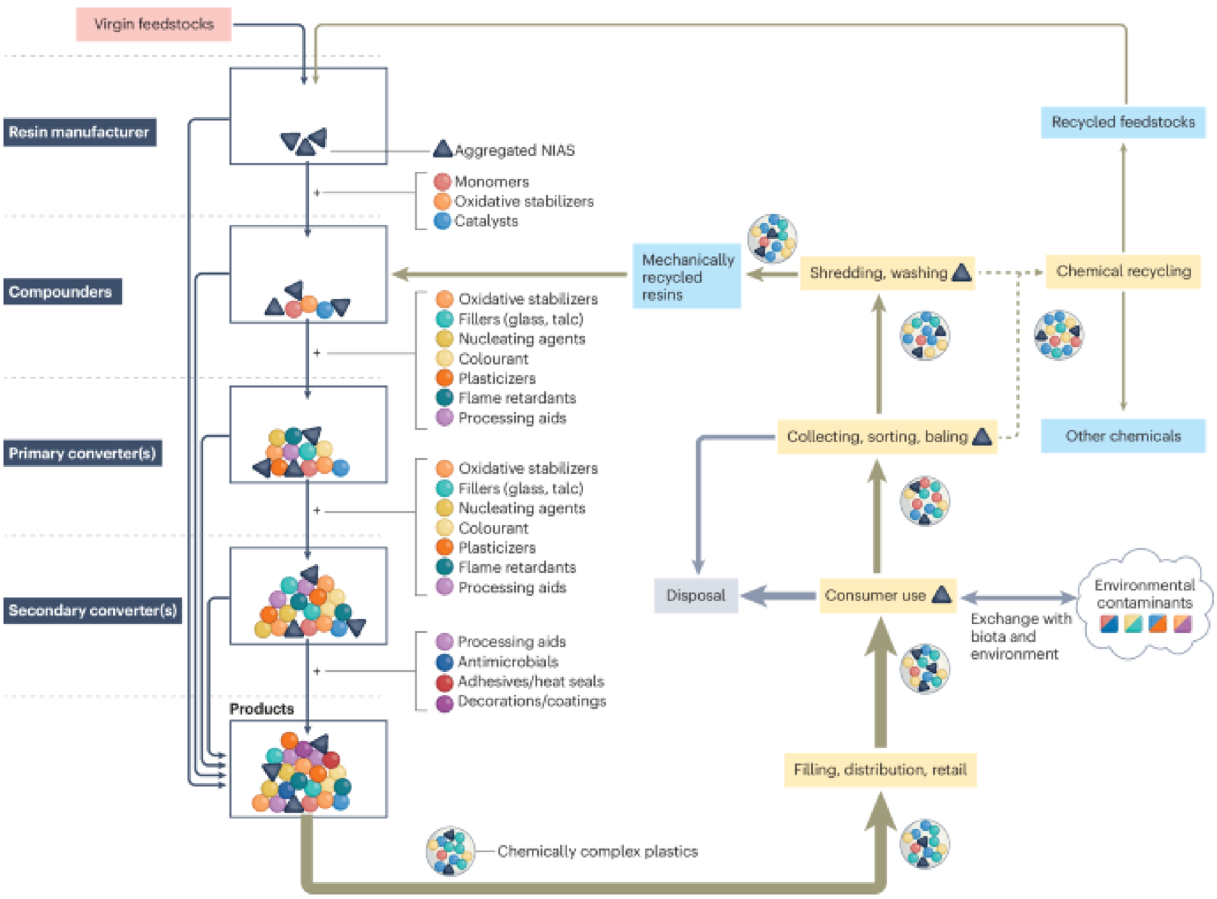

Figure 15. *Increasing chemical complexity of plastics over their life cycle. (Law et al., 2024).*

3.2.2. Plastic formulations and design fit for properties of use

The existing literature in the field of polymer formulation is abundant and reports on formulations and design made complex to reach specific properties. The analysis of the literature carried out as part of this CSA did not reveal any major studies on the simplification of formulations, the majority of which focused on the objective of meeting required specifications, and this objective is achieved by adding specific additives.

In the field of **crop production**, the review of the literature shows that research focus is done on radiometric properties, surface properties, mechanical properties, permeability, biodegradation as well as water absorption (Table 5). This is linked to the fact that publications mainly considered films (79% of the publications) and to a lesser extent encapsulant for fertilisers and seeds (14%) and irrigation systems (7%). Films are used in crop production for the following applications: (i) greenhouse or tunnel films and nets - *i.e.*, impermeable (film) or permeable (net) plastic membranes, used to protect crops; and (ii) mulches, used to protect soils.

Table 5. *Main properties used to characterise agricultural plastic materials. (adapted from Picuno (2014))*

Physical/chemical properties	Mechanical properties	Spectro-radiometrical properties
• density; • gas and steam permeability; • thermal expansion coefficient; • thermal conductivity; • chemical neutrality; • low- and high-temperature stability; • melting point; • electrostatic properties; • surface adhesion of motes	• tensile stress at yield; • tensile stress at break; • percentage elongation at break; • tear resistance; • elastic limit; • young modulus; • compression and bending properties (for rigid sheet).	• transmissivity and reflectivity in the solar band [0 ÷ 3,000] nm, mainly in the photosynthetically active radiation (PAR) wavelength [400 ÷ 700] nm; • transmissivity and reflectivity in the Long Infrared (IR) band [7,500 ÷ 12,500] nm; • transmissivity and reflectivity under stress condition.

For films, radiometric properties refer to the interaction between polymers and electromagnetic waves. Polymers used in agriculture must have specific levels of permeability to various wavelengths of solar radiation depending on their application and according to climate conditions (Katsoulas *et al.*, 2020). Photosynthetically active radiation (PAR), which largely overlaps with the visible spectrum (400-700 nm), is essential for photosynthesis and is identified as the paramount parameter for greenhouse applications, particularly in Western Europe (Pollet et Pieters, 2000). The review of the literature shows that for mulching films, blocking the transmission of PAR is a desirable feature and that for nets, while the material certainly plays a role, the woven porosity emerges as a key determinant of these properties. Consequently, the radiometric behaviour of shading nets is influenced not solely by inherent properties of the material but also by the construction of the product.

The PAR transmittance varies from one polymer to another one. As an example, Tantau *et al.* (2012) conducted a study to measure the PAR transmittance of 20 different cladding materials used in greenhouse applications. Their findings revealed that ethylene tetrafluoroethylene (ETFE)-film outperformed conventional glass in terms of PAR transmittance. LDPE-based commercial solutions exhibited slightly lower PAR transmittance compared to glass and PC-based double cladding materials exhibited the lowest transmittance among the tested materials.

The PAR transmittance of polymer may be balanced by adjusting the formulation. Around ten studies dealt with the addition of additives. To block the transmission of PAR, lignine, UV-reflective additive, chitosan have been proposed. Other studies also considered nanofillers such as hydroxide-based nanofillers, nano-oxides such as titanium dioxide or zinc oxide and clay at a content lower than 10 wt%. These nanofillers affect the radiometric properties. For example, UV-A transmittance from 90% for pure LDPE can be decreased to less than 10% in the nanocomposite without significantly altering the transmittance in the visible light spectrum. Additionally, infrared (IR) retention efficiency can be enhanced from 39% to 50% for the nanocomposite (Monzó *et al.*, 2019). In some of the scientific papers dealing with the use of nanofillers to adapt the radiometric properties, the modification of the mechanical properties of such materials is shown as a limitation for film application.

Regarding surface properties, antifogging is addressed in the literature as a desirable property for greenhouse applications. This need arises from the significant temperature difference between the inside and outside of greenhouses, combined with the typically high humidity levels within them, which often leads to water condensation on the interior surfaces of the greenhouses. Water surface free energy is about 70 dynes/cm and can be compared to the one of PE, the most widely used polymer for manufactured plastic film used in agriculture that has a much lower surface free energy of approximately 30 dynes[8]/cm (Mansoor *et al.*, 2021). Consequently, water tends to form droplets on greenhouse covers made of PE, leading to a modification of the radiometric properties due to the diffraction of light by the droplets. Uncontrolled, localised water dripping can also lead to plant rot and the proliferation of germs (Yang *et al.*, 2020).

To control the surface properties of plastic films used in agriculture, altering the surface through micro or nano-indentation can be a strategy. However, modifying the composition of PE-based cladding is considered less technically and financially challenging (Irusta *et al.*, 2009). The addition of a surfactant (such as lauric acid diethanolamide; Yang *et al.* (2020)) or the addition of a PVA-based layer that can include surfactant or silica nanoparticle (Mansoor *et al.*, 2021) are also proposed in the literature. Surfactant additives can migrate to the film surface and partially dissolve in the condensed water, reducing the water surface tension. However, due to the continuous dissolution of the surfactant, the anti-fogging effect weakens over weeks or months. In addition to this, some degradation of the tensile or optical properties of the material are mentioned in the literature with such modifications (Ge *et al.*, 2020).

Mechanical properties receive extensive attention in the literature, likely due to their universal significance across various applications and the ease of measurement, with testing facilities (*i.e.*, universal testing bench) available in most laboratories. The focus of many studies revolves around evaluating the modulus, stress, and strain at break of films. Other metrics or tests can provide a more comprehensive understanding of film behaviour but are less documented, *e.g.*, the dart drops test measures resistance to perforation or tear strength tests, which require specific die cutting tools. The literature review shows that for the same application, significant variability in mechanical properties can be observed. For instance, in the case of greenhouse applications, Franco *et al.* (2022) show that the Young's modulus ranged from 12 to 75 MPa, the strain at break from 32 to 1535%, and the resistance at break from 10 to 34 MPa. For mulch (Wang *et al.*, 2020), conventional commercial solutions typically demonstrate a stress at break averaging of about 1.5 MPa and a strain at break surpassing 200%.

The analysis of the literature carried out in this CSA also shows that there is limited research dedicated to modifying the mechanical behaviour of PE-based films, as PE inherently possesses suitable mechanical qualities for agricultural applications. The ductility of PE-based film is often seen as a major advantage, and there is typically no need for stiffness improvement. It is important to note that this observation does not apply to bio-based plastics. For such plastics, there is an explicit need for dedicated additives or blend that would enhance the ductility of the film. In contrast, the effect of plastic additives on mechanical properties is a widely researched topic in the literature but in most studies, these additives were used for another purpose and enhancing the mechanical properties of plastics was not their primary objective. Results vary, with some studies demonstrating an improvement, others a decrease and still others no effect on the mechanical properties when additives are used. Regarding radiometric additives, it is generally accepted that optimal radiometric properties can be achieved with a low additive concentration (5 wt%), without significant adverse effects on mechanical properties. Bio-based and/or biodegradable plastics do not necessarily exhibit adequate mechanical behaviour for use

[8] Dyne is a unit of force in the CGS system. It is defined as the force required to accelerate a mass of one gram by one centimeter per second squared. One dyne is equal to 10^{-5} newton.

in agricultural films. PHA, polyhydroxybutyrate (PHB), and PLA typically demonstrate an elongation at break of about 15% and stress at break ranging from 20 to 30 MPa, resulting in inherently brittle behaviour (Shen *et al.*, 2009). It is therefore generally necessary to specifically 'additivate' bio-based and/or biodegradable polymers to modify their mechanical behaviour. Similar limitations were found for starch-based materials.

The analysis of the literature carried out in this CSA also shows that water vapour permeability (WVP) is of critical importance for mulching film application. The permeability of CO_2 and O_2 is considered less critical as evidenced by the limited number of studies addressing these properties. The majority of the research is focused on the comparison between bio-based films to conventional PE-based solutions. These bio-based alternatives often exhibit higher WVP, which must be reduced in order to be competitive with PE films. Indeed, since most bio-based polymers are hydrophilic (compared to petroleum-based solutions such as PEs), they are likely to exhibit a high permeability. The WVP of polymer can indeed be modified by incorporating additives that increase the path length for H_2O molecules to pass through the film, thus enhancing tortuosity. This principle has been well understood for decades and remains central to various industrial applications. However, this approach faces limitations due to its impact on optical properties, particularly transparency. While this issue may be less significant for mulching film applications, it poses a greater challenge for greenhouse applications where transparency is a critical concern.

Finally, it is intriguing that despite the spotlight on bio-based polymers in the literature, the biodegradation of films used for agricultural applications is not a major focus in current discussions in particular for mulching film application. Publications on biodegradable polymers often overlook the actual degradation process, typically mentioning biodegradability in the introduction as an uncontested advantage of these polymers over polyolefins and justifying the use of such polymers as a sign of progress toward more environmentally friendly materials.

It has to be noted that in most of the studies on required properties and developed formulations for crop production, properties of plastics films and nets are determined by laboratory tests, which are conducted on a new material. This is a serious limitation, since a constant exposure to the atmosphere can rapidly alter some of them. The technical aspects about the performance of material during its working life are usually poorly or not considered at all during the design and production phase. Plastic materials have a limited service life, even if they are produced with considerable quantities of appropriate additives with the aim of prolonging their useful lifetime. A lowering below the limit of 50% of the elongation at break, compared with the value at new, is usually considered as an indicator that the material has reached the end of its useful life and that it has to be removed, before its mechanical characteristics become so poor that its removal and mechanical recycling can become impossible (Briassoulis, 2005). In practice, this control is not carried out by anyone (farmer, technician, agronomist, etc.), the only reference being the signal received by farmers (or, in any case, by the person responsible for the periodic re-stretching of the plastic film), when the plastic film does not homogeneously rewind.

Regarding encapsulants, applications focus on controlled-release fertilisers and coated seeds. Plastics used for such applications are different from those used in films because their expected properties are different.

For controlled-release fertilisers application, solutions aimed at slowing down the release of nitrogen-rich fertilisers to mitigate greenhouse gas emissions and promote optimal absorption of fertilisers by crops emerge. PE, PP and PUR are the most widely used polymers for this type of application (Yang *et al.*, 2017), raising questions about the pollution generated. Bio-based formulations (such as lignin-based composites) have been proposed, and by adapting the formulation, the release duration may vary from 4 to 44 days (Wei *et al.*, 2021). Other authors proposed polyester/lignite coating solution or PUR-based solution (Abhiram *et al.*, 2023). The analysis of the literature shows that the release properties are linked to the properties of the coating polymer, such as crosslinking level, presence of additives, porosity, and water absorption. While releasing properties remain a major performance factor in such applications, the behaviour of coatings left in fields should also be considered. Once the fertiliser has been released, because common polymers such as polyolefins and PUR used in coatings are not photodegradable, coatings accumulate on the ground, potentially hindering the permeation of molecules or ions into the soil. Given that most fields are ploughed, photodegradation of the residual coating can also be a challenge. Research efforts aimed at developing more environmentally responsible solutions regarding the end-of-life management of these products and assessing their impact on soil quality.

For seeds, coating facilitates their homogenisation and protection making them easier to handle for machinery and often leading to higher germination rates. The specific properties required for such applications include the need for porosity to air, the biodegradability in contact with soils and a certain degree of water solubility, and specific mechanical properties to prevent dusting and breakage. These properties are not extensively discussed in the literature, which tends to focus more on germination rates and the effects of additives such as fungicides and herbicides in seed coatings. Polymers commonly used for such applications include polyvinyl alcohol (PVOH), polyvinyl acetate (PVAc), methyl and carboxymethyl cellulose, starches, or gum Arabic (Afzal *et al.*, 2020).

In the field of water management, peer-reviewed studies on irrigation are limited although over 90 kt of plastics are used annually in Europe for micro-irrigation systems. Two scientific papers investigate irrigation pipes and four super absorbent polymers for agricultural applications. Micro-irrigation is studied, but discussions often focus on the architecture of products rather than on the material they are made of. Hiskakis *et al*. (2011) explored alternatives for micro-irrigation, such as PLA as a biodegradable drip irrigation system. They concluded that despite challenges in production, it is feasible to create efficient products with bio-based plastics. Super absorbent polymers (SAPs) are used to regulate soil moisture levels during drought conditions. These polymers can absorb significant quantities of water and subsequently release it gradually, thereby optimising water consumption. Analysis of the peer-review papers on the topics show that acrylate-based SAPs are often used. Bio-based alternatives currently appear to be less effective than traditional synthetic options (Orts *et al.*, 2000).

In **livestock production systems**, plastics are widely used for forage conservation. For silage conservation in particular, the principal function of the film is to seal the forage and allow to establish anaerobic conditions during at least 12 months (Borreani et Tabacco, 2014). An ideal film should have (i) high mechanical properties to resist to wind, hail, frost and handling (sufficient thickness), (ii) physical strength properties that can be maintained over a long period, (iii) UV protection, and (iv) low oxygen permeability (Borreani et Tabacco, 2017). Plastic films with low oxygen permeability aim to decrease gas exchange between the interior of the silo and the external environment, and thus maintain the required anaerobic conditions inside the silo. Research evolved in the comparison of plastics materials for silages: first, co-extruded, mono-extruded and films with three layers were compared (Snell *et al.*, 2002); then, films with different thickness (Snell *et al.*, 2003). Finally, PE films having generally a high oxygen permeability were compared with films with a low one (containing PA, EVOH or PVC; Dolci *et al.* (2011); Bernardes *et al.* (2012); Orosz *et al.* (2013); Borreani *et al.* (2014); Parra *et al.*, (2021); Neumann *et al*. (2017 ; 2021a, 2021b). For example, a co-extruded oxygen barrier film (45-µm thickness) provides 100 times more barrier to oxygen than a standard PE film (125-µm thickness) (Orosz *et al.*, 2013). Literature reviews show that there has been an increase in the development of plastic films that are less permeable to oxygen and UV rays and more mechanically resistant.

In the **food sector**, plastics are used especially as packaging materials. The main properties that have been studied in this field of plastic food packaging are the thermal, mechanical, barrier and optical properties (Table 6).

Table 6. *Main properties used to characterise plastic materials used for food packaging.*

Barrier Properties	gas permeability, gas solubility and diffusivity, moisture permeability, moisture solubility, diffusivity, water vapour permeability
Mechanical properties	tensile strength (stress/strain) and elongation, impact strength, tear strength, bursting strength, flexibility/moldability, folding endurance, tearing resistance, seal strength, heat sealability, coefficient of friction, blocking, rubproofness, coefficient of linear thermal expansion, torque test of caps, compression test, vacuum leakage test
Thermal Properties	melting temperature, glass transition temperature, enthalpy of melting associated to the crystallinity/amorphous phase determination, heat capacity, thermal conductivity
Optical properties	clarity, transparency, haze, colour, transmittance, reflectance, gloss, refractive index
Others	melt flow index, thickness, density, recyclability, antimicrobial properties, biodegradability, chemical safety, migration test, drop test

The barrier properties have been extensively studied for food packaging. Numerous factors influence these phenomena, so it is very important to take them into account when selecting and designing a material for these applications. Water vapour, CO_2 and O_2 are generally the gases of greatest interest for food preservation. The barrier properties are strongly related to the chemical and molecular structure of the polymer. To be considered a good all-round barrier material, a polymer must possess the following properties (Robertson, 2012):

- A certain degree of polarity;
- High polymer chain stiffness;
- Be inert towards permeating gases to avoid the phenomena of swelling or plasticising observed when polymers absorb moisture from the atmosphere or liquids in contact with the polymer itself.
- High packing capacity between polymer chains due to symmetry or molecular order, crystallinity or orientation in space. Linear polymers with a simple molecular structure lead to good chain packing and lower permeability than polymers whose main structure contains bulky side groups leading to poor packing capacity;
- Presence of cross-linking between the polymer chains; the effect of cross-linking is more pronounced for large molecular sized permeants;
- High glass transition temperature (Tg), higher than the use temperature, since below the Tg, the segments have little mobility, the permeant molecules diffuse through a much more tortuous path and the permeability is lower. The absorption of humidity by the polymers creates a lowering of the Tg value, increasing the movement of the polymer chains and therefore the permeability value.

It has to be mentioned that it is very difficult to generalise about the permeability of polymers because there are many factors that come into play (*e.g.*, the temperature). Furthermore, there are different ways to measure the permeability of materials, as well as different units of measurement. It is therefore not possible to report a unique permeability value for every known polymer (Table 7).

Table 7. *Permeability to oxygen and water vapour for various plastic polymers used in packaging. (Bai et al., 2024)*

Types	Monomer	Po_2 [cm^3.mm/(m^2. day.atm)]	Water permeability [g.mm/m^2.day]
LDPE	Low-density polyethylene	98–453	0.39–0.59
HDPE	high-density polyethylene	26.3–453	0.1–0.24
LLDPE	linear low-density polyethylene	252.62	0.55–1.15
PS	polystyrene	4350–6200	109–155
PP	polypropylene	35–377	3.9–6.2
PET	polyethylene glycol terephthalate	55	0.28
PA	polyamide	1.8–7.6	0.01–0.28
PVDC	vinyl chloride	0.00425–0.57	0.025–0.913
EVOH	ethylene vinyl alcohol copolymer	6–38	320–560
PVOH	polyvinyl alcohol	0.1–45	41.904
PLA	polylactic acid	0.038–0.042	1.34
PVC	polyvinyl chloride	3.28–394	0.94–0.95

As a complementary information, Wu *et al.* (2021) report the oxygen and water vapour barrier requirements for different food packaging and the corresponding barrier values for different polymers (Figure 16) demonstrating the limitation of the considered polymers.

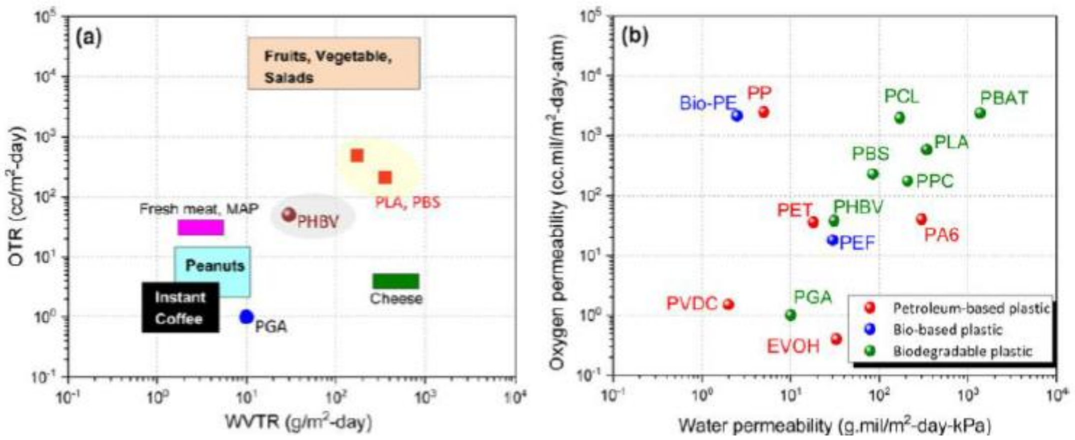

Figure 16. a. *Requirements of oxygen and water vapour barrier properties for different food packaging applications (MAP: modified atmosphere packaging).* ***b.*** *Oxygen and water permeability of different polymers. (Wu et al., 2021).*

To improve gas barrier performance of polymers, multi-layered films are produced made up of different polymers that have an excellent barrier property to oxygen or water vapour such as PET and PP.

Regarding the mechanical properties of plastics for food packaging, tensile strength (stress/strain), impact strength, tear strength, bursting strength and flexibility/moldability tests are studied. Often, to overcome disadvantages of low mechanical properties, reinforcing agents in the film-forming matrix are used.

Clarity, haze, colour, transmittance, reflectance, gloss and refractive index are optical properties of thermoplastic polymers that have been studied for food packaging.

Finally, antimicrobial properties of films used for food packaging have also attracted the attention of the scientific community. Different technologies can be used to achieve such properties. Among them, the addition of nanoparticles (including silver, titanium oxide, zinc oxide, silica, clay nanoparticles) or antimicrobial agents (*e.g.*, beeswax, chitosan, vanillin, natural rosin, nisin and nanolignin), into different biodegradable matrices such as cellulose films, thermoplastic starch, PLA, PHA, polybutylene adipate terephthalate (PBAT) and their mixtures such as PLA/PBAT and PHB/PCL has been studied.

3.2.3. Plastic formulations and design fit for properties of use but not for objectives of plastic uses

Complex formulations of plastics and plastic materials are designed to meet desired properties and be adapted to existing common processing lines. However, the link with objectives of using plastics in agriculture and for food, *e.g.*, increasing crop yields, guaranteeing the nutritional quality of the conserved forage, protecting or preserving food, etc. (2.4.2, 3.4.2), is usually not considered. Indeed, publications exploring relationships between the chemical structure of macromolecules, formulation, properties, together with benefits associated with objectives of using plastics in agriculture and for food are lacking. There seems to be a notable gap between polymer-oriented research and application-oriented studies.

However, a few studies investigate the link between properties and objectives of using plastics in crop production and livestock systems. For example, in the case of coated seeds, Vanangamudi *et al.* (2010) investigated the relationship between coating properties and germination rates (3.2.2). In the case of mulching films, Miceli *et al.* (2002) studied the effect of two PE-based films (one with anti-UV additives and mineral compounds and a second one with anti-UV additives only) and an EVA film, which showed different optical properties and mechanical characteristics, on the growth and yield of pepper, zucchini and string bean. In the case of plastics used to conserve forage, the relationship between film properties (in terms of thickness, colour and structure) and forage nutritional quality is considered, but the effect of plastic films on livestock animal productivity is less documented. Two experiments studied the effects of different types of plastics used to conserve forage as silage on livestock animal productivity, focusing on dairy heifers (Parra *et al.*, 2021) and feedlot calves (Neumann *et al.*, 2021b).

3.2.4. Trade-offs and adaptation strategies are needed along the whole life cycle of plastics

Formulations and processing of plastics are adapted in order to reach specifications along their life cycle. Trade-offs and adaptation strategies are needed. Additives are usually used to obtain such trade-offs leading to complex formulations (3.2.1).

Such trade-offs may concern the service life of plastics. As an example, several solutions based on the addition of nanofillers are reported in the literature to modify the properties of films. Monzó *et al.* (2019) conducted research aiming at enhancing the UV-A reflectance of LDPE by developing hydroxide-based nanocomposites, LDPE serving as the primary matrix. Their findings indicated that LDPE, in its original form, does not effectively block radiation in the UV-A range. However, by incorporating hydroxide nanoparticles along with a surfactant and a whitening agent, they managed to reduce the UV-A transmittance from 90% for pure LDPE to less than 10% in nanocomposite, without significantly altering the transmittance in the visible light spectrum. The authors argued that this formulation is particularly beneficial for greenhouse applications, given the detrimental effects of UV-A radiation on plants. However, this enhancement was achieved at the expense of diminished mechanical properties of the films. Such an example, among a number of others, show that the use of additives to fit one property of plastics can negatively influence another property and balancing both properties is a challenge.

Finally, some strategies aim to strike a balance between in-use and post-use properties. As outlined by Briassoulis (2004) a significant portion of mulching or tunnel films remains on fields or is directly burned by farmers (fin France, this practice has been banned), making biodegradability desirable, especially in areas lacking proper waste management solutions. However, biodegradable plastics usually suffer from lower mechanical properties and other disadvantages (Table 8).

Table 8. *Comparison of conventional and biodegradable polymers widely used in agriculture. (Lewicka et al., 2024)*

Polymers	Application	Advantages	Disadvantages
Conventional Polymers			
PE	Mulching films; Greenhouse covers; Irrigation pipes	Durability; Chemical resistance; Cost-effective	Environmental pollution due to non-biodegradability
PVC	Agrivoltaic systems; Irrigation pipes	High durability; weather resistance	Recycling difficulties; Cost issues
PP	Agrofabrics; Packaging	Flexibility; UV resistance	Environmental hazards due to non-biodegradability

Polymers	Application	Advantages	Disadvantages
Biodegradable Polymers			
PHA	Controlled release systems	Biodegradability; environmental safety	High production cost
PBS	Controlled release systems; Mulching films; Composites	Biodegradability; suitable for blends	High crystallinity reduces degradation
PLA	Controlled release systems; Agricultural films; Containers	Biodegradability; Flexibility	Hight cost; Brittleness

The blending of biodegradable polymers with other petroleum-based polymers and/or fillers is widely used to regulate the properties of the final material (such as water or gas permeability, flexibility, strength, temperature resistance, etc.). This change in the composition of the plastics can affect the material biodegradability. These aspects are fully detailed in 3.3.4. As an example, Narancic *et al.* (2018) tested the biodegradation of six major biodegradable polymers and blends in seven environments: industrial and domestic composting, high solid AD, soil, marine, aerobic and anaerobic aquatic conditions. They showed that, while some polymers and their blends showed good biodegradation in soils and water, the majority of polymers and their blends tested in this study failed to achieve International organisation for standardisation (ISO) and Advancing standards transforming markets (ASTM) biodegradation standards, and some failed to show any biodegradation.

On the other hand, when it comes to recycling, the multiplicity of plastic constituents and products (*e.g.*, multi-layered plastics, flexible plastics, coloured plastics) that are used in the global market become a very complex problem. The 'design for recycling and from recycling' is the new concept-strategy put forward by the Ellen MacArthur foundation in the report on 'rethinking the future of plastics' (Ellen MacArthur Foundation et World Economic Forum, 2016). According to this new challenge, the recyclability aspect should be incorporated as one of the top performance criteria, together with, *e.g.*, product safety, performance, marketing and branding. A balance between all these objectives, prioritising the recyclability of a plastic product, should be achieved. To illustrate this aspect, among plastics, thermosets typically exhibit higher rigid, as well as dimensional, thermal, and chemical stabilities. Indeed, in contrast to thermoplastics, which can be melted and reshaped, thermosets form a three-dimensional network upon curing, are resistant to dissolution and reprocessing. As a recycling method, they can only be subjected to harsh chemical degradation or pyrolysis. Consequently, an overwhelming majority of thermosets are relegated to incineration or landfill disposal after use. In an effort to address the recycling challenges associated with thermosets, researchers have been striving to design dynamic covalent bonds into their highly intertwined networks, aiming to enable chemical recycling and/or reprocessing. An array of dynamic covalent bonds, including ester, diketoenamine, disulfide, imine, silyl ether, vinylogous urethane, dioxaborolane, boroxine, acetal and so forth, have been explored as dynamic modules in the production of these materials. However, at present and despite ongoing efforts, the trade-offs between performance and recyclability remain a challenge in the design of recyclable thermosets: the majority of recyclable thermosets reported so far suffer from inferior properties, such as reduced strength, modulus, and dimensional stability, compared to the conventional thermosets.

3.2.5. 20 years of research focusing on bio-based and/or biodegradable plastics as well as on nanocomposites

There is a clear bias in the literature towards bio-based and/or biodegradable plastic formulations over conventional petroleum-based plastics. Consequently, there is a notable lack of studies examining the properties of the latter, with the exception of PE-based cladding films. This can be linked to the fact that in the methodological approach of the CSA, literature review was carried out from 2000s at the exception of the historical part (Part I of the extended report). This can also, at least partially be explained by the difficulty to access data on commercially available petroleum-based products due to undisclosed compositions including additives. It significantly limits discussion on the origins of product properties. Moreover, even if studies on petroleum-based products exist within the industry, they are likely to be sensitive due to concerns about product performance and market share.

On the other hand and as reported previously, within the period covered by the CSA, the addition of nanoparticles into polymer to design nanocomposites was widely considered for both agriculture (Upadhyay *et al.*, 2022) and food applications (de Sousa *et al.*, 2023; Ghosh *et al.*, 2025). This technology is used to achieve lower relative humidity, oxygen permeability, high strength, etc., and various types of nanoparticles are used (including silver, titanium oxide, zinc oxide, silica, clay nanoparticles). However, despite the numerous studies dealing with the use of nanoparticles as additives for agricultural plastics (mainly for crop production) and food packaging (in particular in active packaging), questions arise from their human health and environmental impact.

3.3. Plastic waste management is difficult to monitor and implement in practice

3.3.1. Plastic waste flows are difficult to track and quantify

(based on Chapters III.1 and III.2 of the extended report)

3.3.1.1. Reliability of data is uncertain because of complexity in collection

The published scientific data related to the mapping of plastic waste flows are scarce both quantitatively and qualitatively. The available data are mainly based on statements, brochures, and press releases by industrials and/or recyclers. Both at the European and French levels, data providers include PROs that supervise Extended Producer Responsibility (EPR) schemes; trade associations, such as Plastics Europe as well as plastics recyclers such as the National Syndicate of Plastics Regenerators (*Syndicat des Régénérateurs de matières Plastiques* (SRP)). These organisations produce data on waste flows that corresponds to a need and are thus included in the compilations of public institutions, such as ADEME, Eurostat, and the Organisation for Economic Co-operation and Development (OECD). As their reliability remains uncertain with regard to scientific criteria (*e.g.*, lack of peer review), their use may be questionable. Scientific literature critically appraising these data and their modes of production is lacking in spite of their widespread use in numerous works, including scientific manuscripts, organisational reports, and official documents prepared by national and supra-national institutions.

The complexity in data collection of plastic waste and their uncertainty rely on several aspects. On the one hand, the manufacturers make effort to protect the information deemed confidential and crucial for their business strategies. On the other hand, many of the reported values provided by associations, companies, and groups with vested interests are only declarative and are usually based on estimates. Additionally, data are often collected, processed, and stored in a fragmented manner, lacking comprehensive approach. This fragmentation can be geographical (no centralised data), sectorial (map of plastic waste by sector, by resins, etc.), methodological (limited information on the methodology used to obtain values, making data gathering, comparison, and analysis rather cumbersome) or even linked to the regulation (regulations and legislation may differ even within the EU). Indeed, food packaging waste in particular are local authority-dependent. There is thus no centralised data available neither for Europe nor for France. In the field of agriculture, ADIVALOR, as the national voluntary organisation fully dedicated to the recovery of agri-supplies waste and supported by the Ministry in charge of the Environment's services, produce data related to collection or treatment internally but the data concerning the volumes placed on the market are produced by another organisation (CPA) and are based on estimates. It leads to uncertainty when dealing with rate of plastic from recycling. Finally, some waste may be routed through intermediary countries before reaching their final destination, which can distort trade statistics and the fate of plastic waste. Underreporting and illegal trade cannot also be ignored. An accurate depiction of the plastic waste flows within the European, more generally, and French territories, in particular, is not feasible. It must also be highlighted that in most of the case, the definition of what is considered in the studies is rarely accurate. As an example, the data considering 'recycled materials' may concern the materials that are sent to recycling but not necessarily recycled. It should be underlined that these observations do not call into question the methods and/or quality of the data gathered and made available by the different organisations involved, but rather underscore the difficulties in understanding these figures and establishing valid and thorough plastic waste flows based on these public data.

The complexity of data collection leads to inconsistencies between reported values. To illustrate this, it can be noted that comparing data published in the Déchets chiffres-clés 2023 (ADEME, 2022) and database consultations and verifications, large differences exist. The aforementioned report stated that total French exports of plastic waste amounted to 800 kt in 2021, according to public customs database. However, when consulting the UN COMTRADE databases (https://comtradeplus.un.org/), French exports amounted to 342 kt (HS code 3915) during that year, nearly a 2.5-fold difference. This example highlights the challenge of evaluating publicly available data when the methodology used for its collection and processing is not clearly explained.

Regarding the agricultural sector, few countries have implemented a national organisation to collect plastics. This is the case of France (ADIVALOR, a national voluntary organisation) and partially of Italy, Germany, and the Netherlands. ADIVALOR PRO centralises collection and redirection channels for reprocessing or disposal of agricultural plastic waste. For this CSA, ADIVALOR provided two sets ofdata on the plastic waste that it is mandated to manage: the first by use and the second by resin. However, classification was different and it was not possible to determine specifically the quantities of resins for each use, or the types of treatment for each waste (incineration, recycling, landfilling).

In the field of food industry, the problem is even more complex. Plastic packaging are considered in the EPR for household packaging which applies to all packaged products consumed or used by households on the French market. It is thus very difficult to quantify the share of plastic packaging dedicated to food among other uses, such as personal care, or more broadly, all packaging for non-food consumer goods. Scattered information suggests a varying proportion of food packaging waste in municipal solid waste (MSW), ranging from 78 (Lee Hae-rin, 2021)[9] to 83% (Greenpeace, 2022); however, these estimates are based on surveys and concern different nations – in this particular case, UK and South Korea – again making comparisons and extrapolations somewhat difficult. It is therefore impossible to determine the proportion of plastic packaging waste from the food sector in relation to total packaging in France and even in Europe, and no scientific work identified during this study has been able to determine the specific characterisation of plastic packaging waste.

We can also mention that a new EPR scheme for catering packaging came into force on 1 January 2024 (article L.541-10-1 of the Environment Code)[10]. This new EPR scheme is governed by two pieces of legislation: the implementing decree published on 7 March 2023, which defines the main principles of EPR, and the order of 20 July 2023, which sets out the precise rules for identifying packaging covered by EPR, based on the volume or weight of the packaged food product.

We can thus conclude that even if the existence of EPR scheme facilitate data collection, lack of information on the methodology to collect them and on the information regarding plastic composition still remains a limitation to map plastic flows. Moreover, there is almost no scientific literature available on the subject. As a result, reference is often made to literature from stakeholders, associations, etc., whose primary objective is not necessarily to develop methodologies that are universally recognised as scientific.

Data were analysed in Chapter II.1 of the extended report taking limitations described above into account. The total plastic waste collected has steadily increased, as evidenced in Figure 17A, and numbers indicate that the total quantity of waste sent to recycling facilities has more than doubled in 15 years, likely owing to higher volumes of plastic packaging waste separately collected, improvements in waste collection systems, as well as in sorting technologies and the expansion of recycling initiatives for plastic waste from agricultural, farming and gardening applications (Antonopoulos *et al.*, 2021). Moreover, although landfilled plastic waste has decreased significantly, just under 7 Mt of plastics ended up in European landfills in 2020 (Figure 17B), a yet considerable amount. In France, figures indicate that plastic waste are collected for recycling (35%), sent to incineration (33%) or landfill (32%) in similar amounts, *i.e.*, 1.27, 1.17 and 1.16 Mt, respectively (Table 1; Perez *et al*. (2025)).

[9] https://www.koreatimes.co.kr/www/nation/2024/03/113_318974.html [Consulted the 26th of February, 2025].
[10] https://www.legifrance.gouv.fr/codes/article_lc/LEGIARTI000021660934/2010-07-14

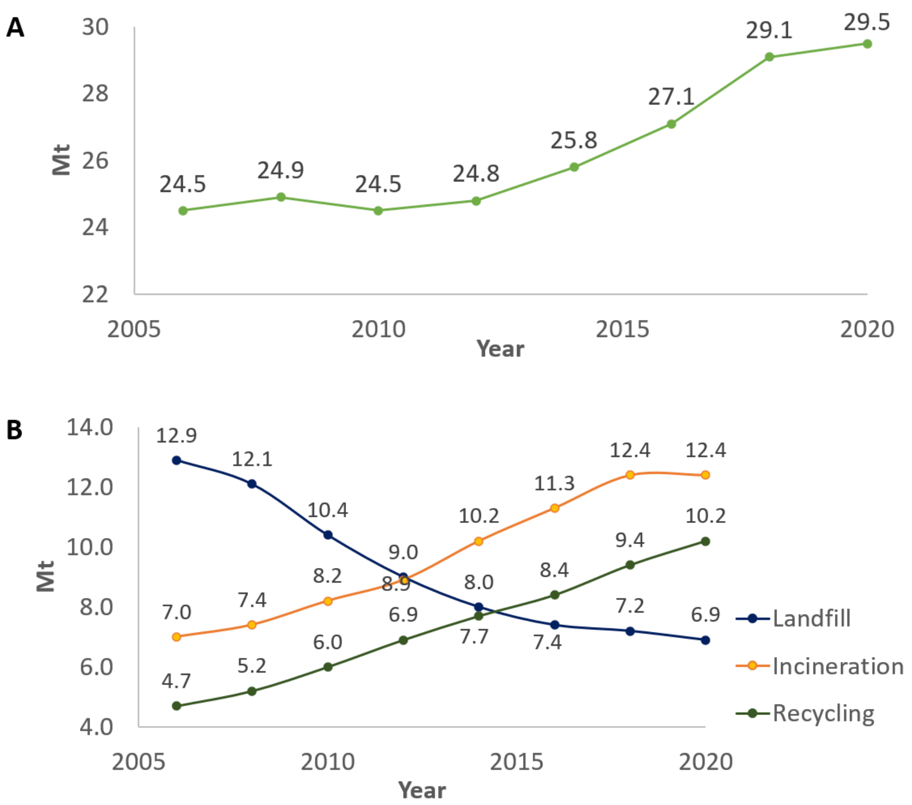

Figure 17. A. *Total plastic waste collected at the EU (27+3) between 2006 and 2020, in million tonnes (Mt).* ***B.*** *Breakdown of the fate of this plastic waste. (Plastics Europe, 2022. The facts 2022. https://plasticseurope.org/knowledge-hub/plastics-the-facts-2022/)*

Regarding plastic packaging waste (Figure 18), Deloitte and ADEME (2022) estimates that: (i) of the 1.27 Mt of plastic waste directed to recycling channels, 0.89 Mt, *i.e.*, approximately 24% of the total French plastic waste, is post-consumer waste, with the remainder consisting of manufacturing waste; (ii) 70% of this 0.89 Mt comes from packaging, *i.e.,* 0.62 Mt; and that (iii) 50% of this 0.62 Mt comes from household packaging, totalling about 0.31 Mt. It should be noted that these data do not reveal what proportion of this packaging stems from the food sector.

It is not possible to map the plastic packaging waste flows when considering the type of resin (Figure 18). Data show that 31% of plastic waste (0.75 Mt) were collected selectively, and among them 2% went to landfill (0.05 Mt), 4% to energy recovery (0.11 Mt) and 25% to recycling (0.60 Mt). However, it is stated that the industrial and commercial plastic packaging waste figures given relate primarily to waste not sent for recycling, rendering obscure the details of industrial and commercial plastic packaging waste sent for energy recovery or landfill, included in the 'uncollected/unknown' flow. The absence of an EPR scheme for industrial and commercial packaging waste (planned to be implemented in 2025) means that these figures are highly uncertain. Moreover, it has to be noted that data are from 2022 at a time when the extension of sorting instructions was not fully deployed. On 30 December 2020, it was estimated that 30.5 millions of French people were concerned by this extension (CITEO, 2021) and it should have concerned all the French population from 1 January 2023.

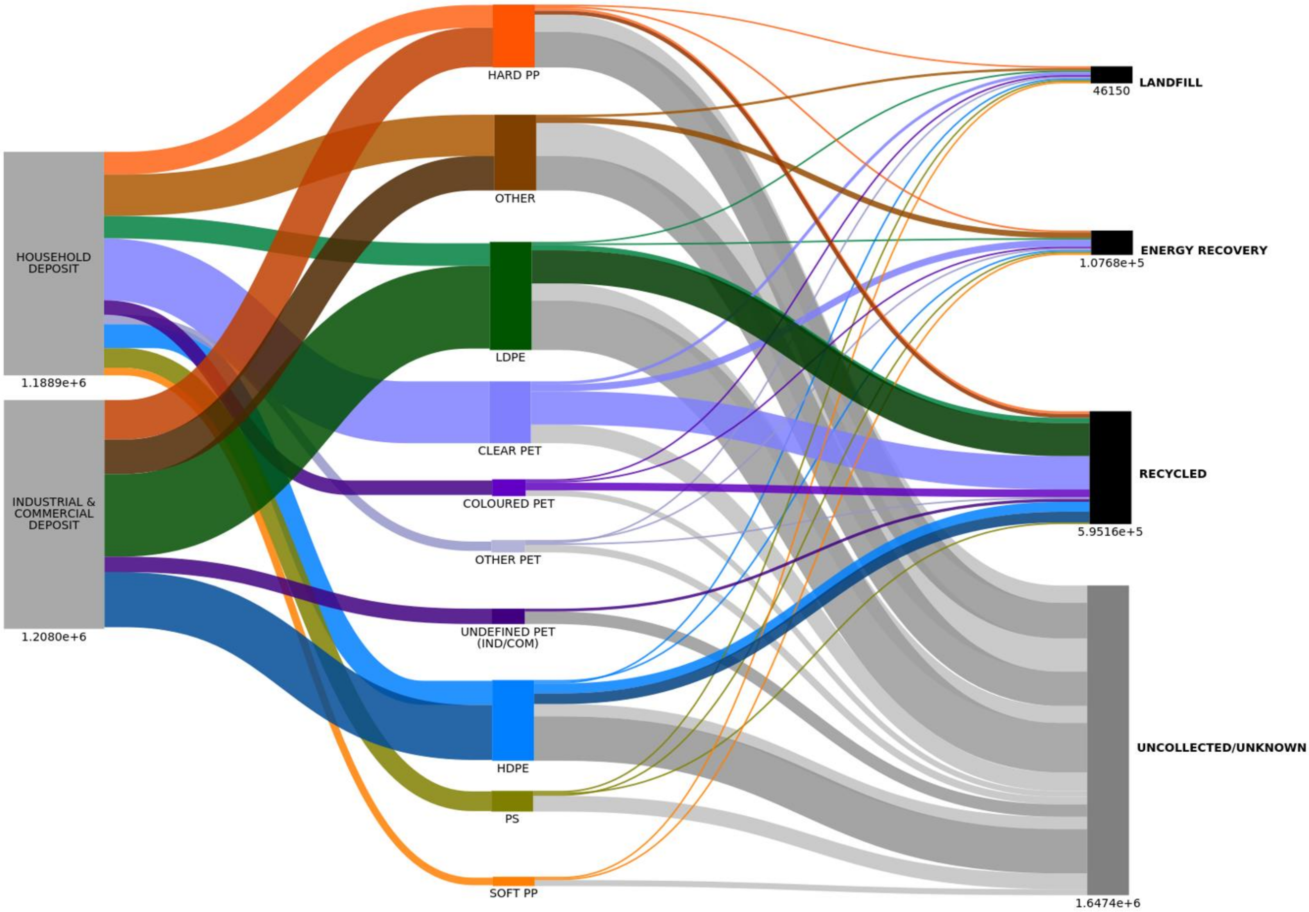

Figure 18. *Plastic packaging waste flows by resin in France in 2022. Values are in tonnes. Please note that for industrial and commercial waste, only flows destined for recycling are mentioned; flows destined for incineration or landfill are included in the uncollected/unknown flow. (Deloitte and ADEME (2022))*

Regarding agricultural plastic waste, PE (HDPE and LDPE), and LDPE in particular, is the main resin found (Figure 19; ADIVALOR (2023); Deloitte and ADEME (2022)). This is consistent with the large quantities of LDPE used for plastic films and HDPE for drums and bale nets. It is important to highlight that livestock farming account for 59% of total agricultural plastic waste in France.

According to data shown in Figure 19, almost all plastics collected by ADIVALOR are recycled, with the exception of certain HDPE and LDPE fractions. However, no information is available regarding the fate of this non-recycled fraction, nor about the uncollected tonnages.

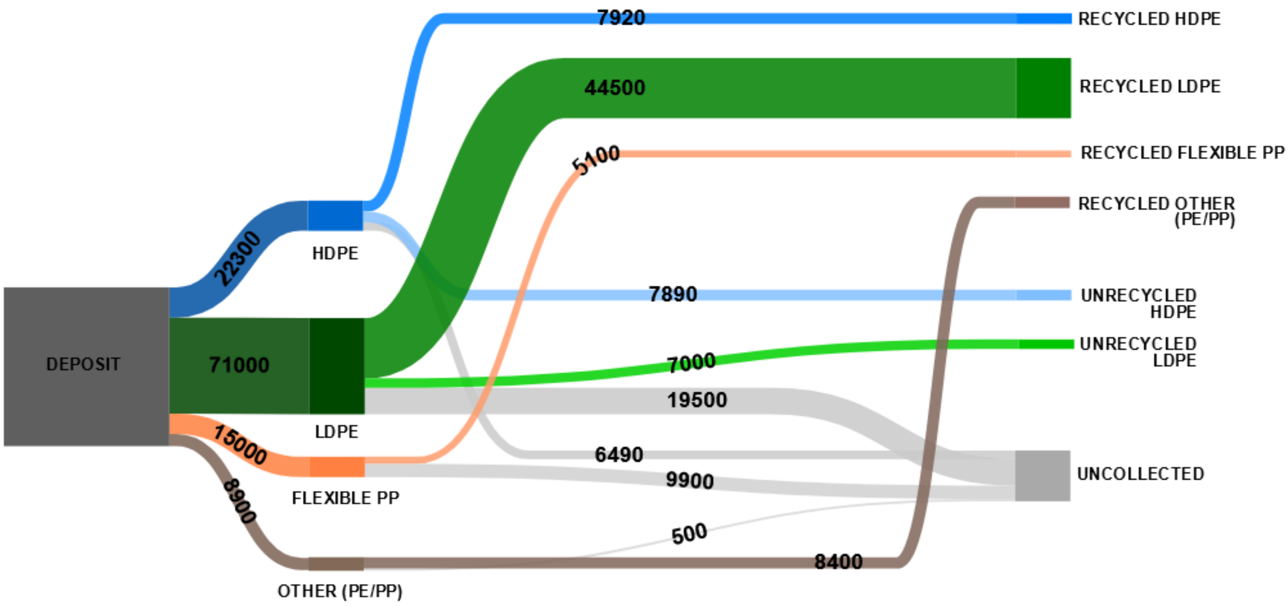

Figure 19. *Agricultural plastic waste flows in France in 2020. Values are in tonnes. (adapted from ADIVALOR (2023) and Deloitte and ADEME (2022))*

Finally, data regarding the economic sectors of origin of plastic waste and the economic sectors of destination of recycled polymers were analysed. Figure 20 based on SRP data (SRP, 2022) shows that packaging account for 47% of outlets for recycled plastics and that the agricultural sector accounts for only 0.03%. According to figures for plastic consumption in France (in 2018), this reincorporation represents approximately 11.4% of plastics consumed in the packaging sector and 10% of plastics consumed in agriculture. However, these figures are of limited relevance, as recycled materials are not necessarily used in French production (they may be exported), the packaging category is again undefined in terms of the details of their use and there is no detail regarding the type of resin(s) used. Again, the ‘packaging’ category does not capture the specificity of packaging linked to the food industry, falling into three categories: ‘industrial packaging’, ‘industrial and commercial packaging’ and ‘household packaging’.

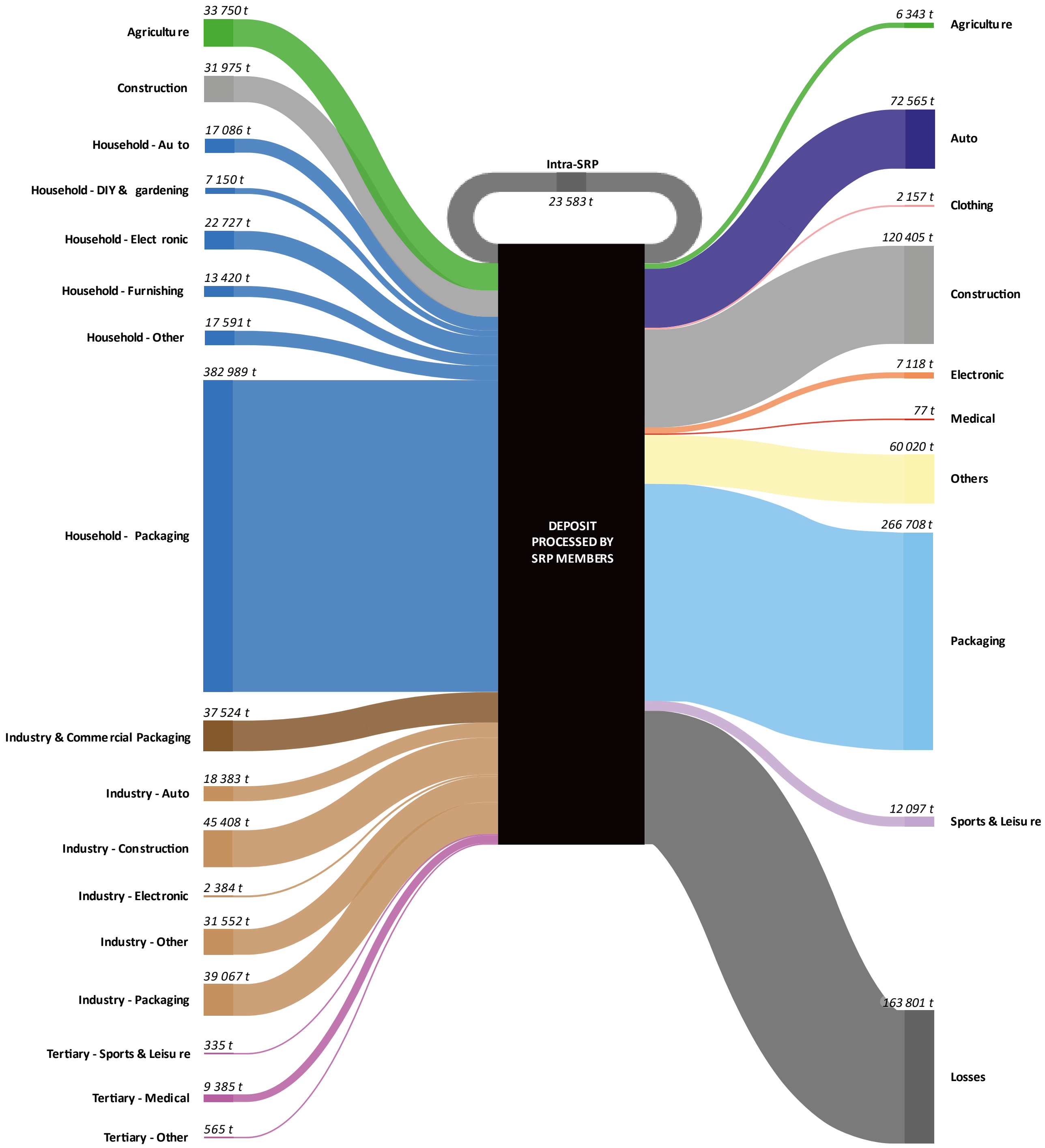

Figure 20. *Economic sectors of origin of plastic waste collected by SRP members and economic sectors of destination of recycled resins by SRP members. Values are in tonnes. (SRP (2022)).*

3.3.1.2. Plastic recycling: many actors, technologies, materials, and few comprehensive studies

Improving the recycling system would require all independent organisations (Figure 21) to work in unison. The production of petroleum-based plastics begins with oil and natural gas, which undergo a cracking process to generate feedstock and monomers. These monomers are then polymerised to form polymers, which are subsequently combined with additives to create compounds. These compounds are further processed and assembled into end products. In the end, plastic products become waste and are, for part of them, directed to various recycling methods, such as mechanical recycling, solvent-based purification, chemical recycling (*i.e.*, depolymerisation), and feedstock recycling. Mechanical recycling remains the primary method used in Europe, while other processes are still in pilot phases or nearing commercialisation. Numerous stakeholders are involved at different stages of the plastic life cycle, including petrochemical producers, manufacturers, retailers, and waste management companies (Hsu *et al.*, 2022). Defining a value chain acceptable to all actors is not an easy task.

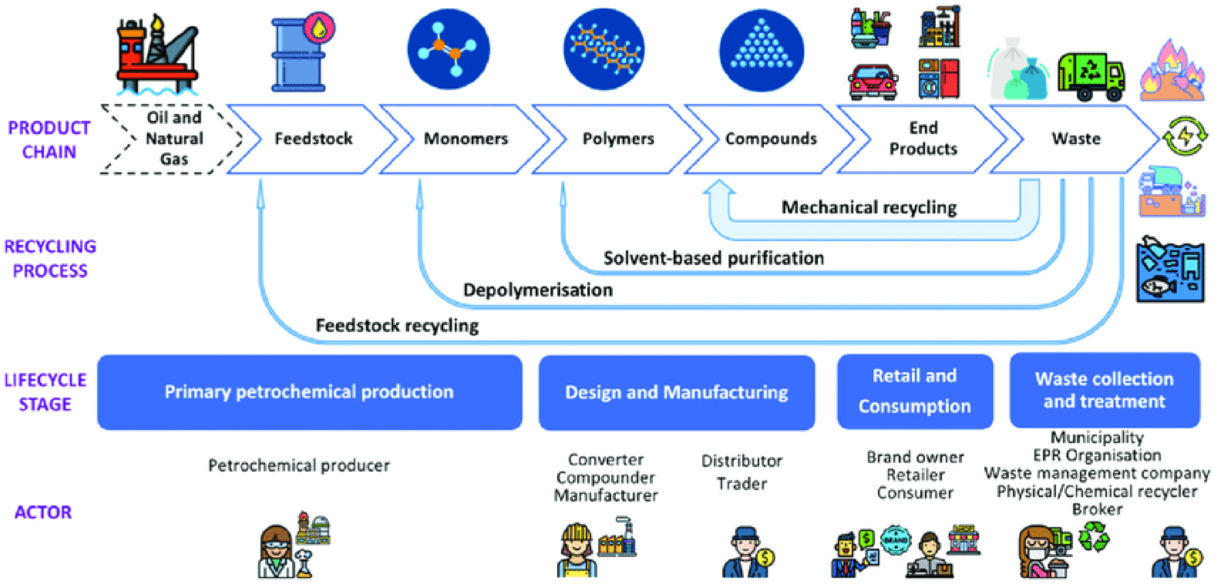

Figure 21. *The plastic value chain. Actors, life-cycle stages, and key routes for secondary materials are shown. (Hsu et al., 2022)*

In addition, recycling is polymer-specific and depends on the waste streams (3.3.3). No generic solution exists to handle the wide diversity of polymer matrices and additives used industrially, and mixed waste streams. The presence of additives in plastics make their recycling more complex, in particular for FCMs, and no recycling method ensures a 100% reuse of the carbonaceous matter. Indeed, plastic products contain a vast array of chemicals, designed to improve the overall characteristics of the final products and/or reduce costs (3.2.1), making their treatment more complex (Barra et González, 2018). Following a 'design for recycling' approach, would make recycling more feasible by eliminating the various additives that may cause degradation during melting and re-extrusion or deactivate catalysts in thermal processes. Studies on this topic are rare. For example, Hees and al. (2019) explored this 'design for recycling' strategy, particularly for polyolefins. Their work, along with others, discusses the production of 100% polyolefin plastics and composites without additives. This is made possible through the use of multisite polymerisation catalysts and specialised injection-moulding techniques like oscillating packing injection moulding. This method eliminates the need for producing different polymers at separate facilities, avoids the use of additives such as glass fibres, and creates products that can potentially be depolymerised using polymer synthesis catalysts like Ziegler-Natta. The production of PP or PE-based plastics and composites using this approach may significantly improve production efficiency and reduce environmental impact. Such an approach, even if it exists in the literature, is rare in comparison to the literature available on recycling technology of plastics. This may be because today's plastic waste are made up of materials produced in the past, and the transition in plastic formulation is a long process.

A few studies investigate the circularity of plastics from a holistic approach and at a mesoscale. Paletta et al. (2019) considered the case of Italy. They reported some of the barriers (technical-technological, legislative, economic, and socio-cultural) to a better circularity of plastics (Table 9). They concluded that 'companies cannot operate as isolated organisational silos; they need more collaboration with all the stakeholders involved in the co-production of a complex outcome such as the reduction of environmental pollution'. This concerns the greater integration in business-to-business relations, between consumer goods companies, manufacturers of plastic packaging, plastics manufacturers, and companies involved in the collection, sorting and reprocessing. This issue is emphasised in the conversion industry, where small and medium-sized enterprises are the majority.

Table 9. *Some of the barriers to plastic circularity – mesoscale analysis. (Paletta et al., 2019)*

BARRIERS at mesoscale level	Plastic production	Plastic compound	Plastic convertion	Plastic distribution and use	Plastic recycling
Techical-technological barriers				Contamination of post consumer plastic and secondary materials	Lack of innovation on recycling plant
				Low efficiency on collection system	Low efficiency on sorting system
			Lack of mechanical performance in recycled plastic-based products		Recycling challenge for LD polimer-based films
			Incompatibility of recycled plastics with product manufacturing process		Recycling challenge for composite materials
	Lack of information on recycled plastic composition, including hazardous substances				Recycling challenge for thermoset polymers
Legislative barriers	Unclear definition and management for waste (pre and post consumer waste), by-product and secondary materials				
	Lack of harmonization on evaluating safety of recycled process for plastic food contact materials				
	Safety requirements for highly technical products and hazardous components and material issue				
			Rigid certification for EE products		
			Rigid food contact legislation		
	Missing guidance for ecodesign and lack of support for scaling up circular models				
Economic barriers			Resistance to change among product manufaturers		High costs of collecting, sorting and processing waste plastics (influencing the secondary plastics costs)
				Irregular and significant peaks of waste market	
	Vulnerable markets for recycled plastics (depending on oil price and recycled plastics availability)			Limited resilience of the sector to market shocks	Competition between recycling and energy from waste
	Lack of incentives for performing new circular materials and products				
	High risk for moving from linear to circular production process				
	Lack of support for scaling up circular eocnomy models, especially for SMEs				
Socio-cultural barriers	Hostility in the direction of innovative materials			Unsustainable cultural behaviour	
			Inert attitude toward the client request	Low consciousness on single-use packaging consumption	
				Low consciousness on correct waste disposal	
				Business as usual (BAU) model diffusion	
				Customers' disinterest toward secondary plastics	Illegal waste trafficking

3.3.2. Collection and sorting are key steps scarcely studied

(based on Chapter III.2 of the extended report)

Very little literature is available (39 studies identified) on the topic of collection and sorting. As a consequence, most of the information and data comes from authorities or professional literature. The conclusions reported below are thus mainly based on this grey literature. In addition, very few studies refer to waste, plastic collection systems and uses in France. Most of the examples and data concern Western European countries (the UK, Norway, Germany, Spain, etc.) and USA. Moreover, no scientific studies are related to the collection of post-industrial plastics from agriculture or the food industry for France, nor for the EU.

3.3.2.1. Collection and sorting influence subsequent treatment, including recycling

The initial phase of the recycling process concerns collection. Within this step, the collection rate (and among them the collection rate of plastics) appears as a key parameter to optimise (Figure 22).

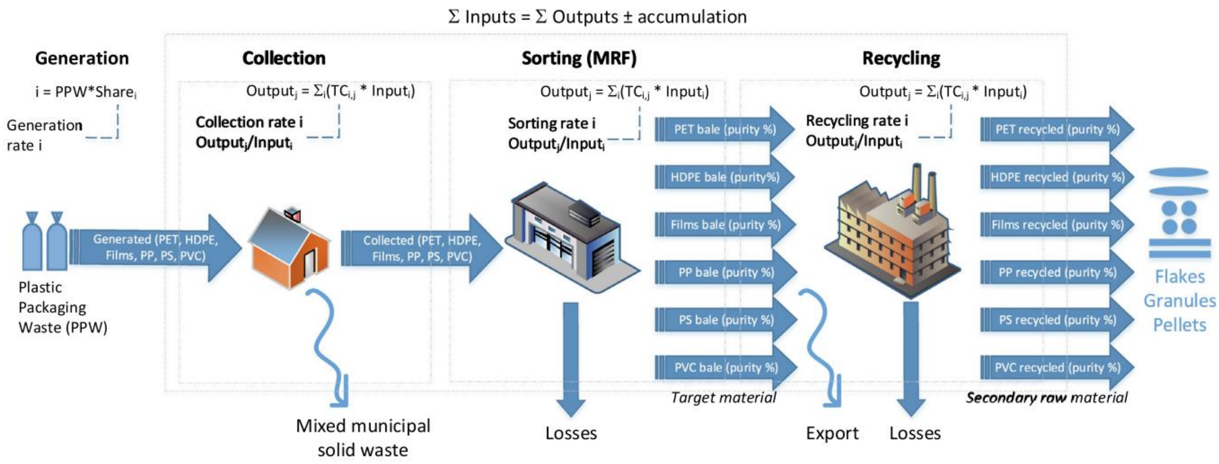

Figure 22. *Conceptual diagram depicting the waste management phases. (Antonopoulos et al., 2021).*

In EU countries, plastic waste collection can take various forms (European Environment Agency, 2024) including: curbside door-to-door pick-up; drop-off centers; recycling; Deposit Refund Systems (DRSs); mobile collection units; informal waste pickers; clean-up campaigns; commercial collection services and community-led initiatives. Only curbside pick-up, drop-off centres and DRSs are documented in the scientific literature. Two main ways are actually involved in France: municipal drop-off centres (or municipal waste collection centres), and curbside collection using trucks that can contain different compartments lining both sides of the vehicle (Cimpan *et al.*, 2015).

Drop-off recycling is developed mainly for glass, paper, metals in the EU and France, but remains very scarce for plastics, except in the USA where its efficacy is demonstrated compared to curbside in cities where the ‘pay as you throw’ tax is applied. In this case, people deposit their waste in centres to avoid throwing it in their rubbish bin, which has to be paid for by weight. Regarding DRSs, the group of private operators communicates on a mostly only glass bottle system (France Consigne, 2024). Although prioritised by EU for 2029 (European Court of Auditors, 2020; Meissner *et al.*, 2022), DRSs are a no-question for plastics for now in France for both local authorities and end-users (final consumers, MSW). This is mainly because of the costs induced and destination of returned bottles mainly for recycling and not reuse (*réemploi* and/or *réutilisation*) (Que Choisir, 2023; Trellevik *et al.*, 2023; Messad, 2024). Setting up a deposit system requires significant investment: recovery machines (such as automatic deposit machines), logistics and flow management and these costs are deemed unjustified since recycling facilities are already existing and efficient. Only few scientific studies illustrate the feasibility of DRSs in countries like United Arab Emirates or China (Ajaj *et al.*, 2022; Abu Jadayil et Aqil, 2023; Zhou *et al.*, 2023). Such a system implies a sufficient size/volume of packaging and uniformisation of both polymer types and volume/shape with improved mechanical resistance for reuse (*réemploi* and/or *réutilisation*). The chemical risks of reused (réemployées and/or réutilisées) PET bottles for consumers should be highly considered whatever the systems for collection (Gerassimidou *et al.*, 2022).

The efficacy of the collection depends on several factors and among them the education of consumers (Bing *et al.*, 2014). Another parameter to consider is the gross domestic product (GDP). Indeed, the higher the GDP per capita, the higher is the demand for plastics but the more efficient the collection (Lebreton et Andrady, 2019). When GDP per capita is lower than 5,000 US$, the MSW generation is about 200 kg/year/capita and the unsound disposal of waste (UDW) of about 95%, whereas for GDP per capita higher than 40,000 US$ (like for France), the MSW is about 580 kg/year/capita and UDW of less than 3%. Literature also shows that, in country where collection systems are developed (such as Austria), the rate of plastic waste collection decreases as the density of population increases, *i.e.*, in the metropolitan and big cities (Schuch *et al.*, 2023). Finally, the plastic rate collection increases with the combination of collecting systems available for end users for rural areas (values of separate collection rates increases to 78-83% compare to 74-77%) (Schuch *et al.*, 2023). The results for urban areas showed a lower separate collection rate of 56%. In the case of separate collection targeting plastic bottles only, maximum collection rates of around 50 % were observed.

Regarding the cost of the collection and considering curbside, it has been shown that when the weight-based invoicing is applied, various positive effects are observed, *e.g.*, a significant decrease in the amount of unsorted waste by an average of 91 kg/year/inhabitant, a lower collection and sorting costs for the local authorities, an increase in recyclable materials including plastics, and an increase in sorting efficacy by consumers (*i.e.*, separate stream) by 30% (Cassette *et al.*, 2022). However, negative effects such as unpaid bills or illegal dumping were also noticed. They remain marginal and diminished after the local authority has taken preventive action or issued fines). Prevention action are carried out by local authorities that communicate and raise awareness of waste issues, particularly on waste reduction and sorting instructions, but prevention initiatives do not consider the impact of littering and illegal dumping on the environment and society. On the other hand, in The Netherlands, it was also shown that the cost efficiency of plastic collection-separation from solid waste is improved using single stream collection systems (Feil *et al.*, 2017).

Regarding regulation in France, the extension of sorting instructions is part of the French Law on Energy Transition for Green Growth no. 2015-992 (République française, 2020). It consists of enabling residents to put all packaging in the sorting bin and to increase the collected and the recycling rate of plastic packaging that were not previously recycled (films, pots and trays, etc). The objective was to encourage French people to sort packaging by dispelling doubts about packaging sorting, and increase the recycling of plastic packaging other than bottles and flasks, which were already recycled. It may permit recycling companies to have access to more material for the development of processes and technologies needed for large-scale recycling. CITEO reports (2021) that in towns with extended sorting instructions, people sort 4 kg more per inhabitant per year than in towns that are not: 2 kg more new plastic packaging (the equivalent of 26,000 t in 2018 - 16,000 t of pots and trays and 10,000 t of films) plus 2 kg of cardboard, glass, metal and plastic bottles, which have historically been included in the sorting instructions. These figures confirm that simplifying the sorting process increases its frequency and thus the collection rate of waste. It was also shown in Belgium that, when sorting instructions are extended, the collection rate of plastic packaging increased from 33.6 to 64.4% (Roosen *et al.*, 2022). It results in a net recovery rate (*i.e.*, fraction of a desired product that is captured in the correct sorted fraction taken into account the collection and sorting process; Table 10) estimated to be around 49.7%. However, real recycling rates (*i.e.*, amount of plastic that is actually reprocessed into products) will actually be lower as certain sorted fractions such as PET trays and other films are still very challenging to be effectively recycled and since losses in the mechanical recycling process occur. This is an important result, highlighting the extra effort that is still necessary to be able to meet the European recycling targets of 50% by 2025 and 55% by 2030.

The total sorting recovery is used to determine the quality of sorting. This parameter is calculated by dividing the total mass of waste that is correctly sorted by the total amount of waste entering Municipal Recovery facilities (MRFs; Table 10). In addition, the composition of sorted plastic in bales dedicated to recycling is often given to evaluate the polymer contamination prior to the recycling process. It corresponds to the quality of plastics sent to recycling.

***Table 10.** Overview of the applied indicators and corresponding definitions and equations, comprising sorting recovery R_T , net recovery R_N, product grade G_T, polymer grade G_P, SDI, total analysed metal content C_M, total analysed halogen content C_H, and C, H, N, O levels X_{CHNO}. (Roosen et al., 2022)*

Performance Indicator	Definition	Equation
Sorting recovery	Fraction of a desired product that is captured in the correct sorted fraction taken into account the sorting process	$R_T = \frac{f_z^T}{\mu^{TS}}$
Net recovery	Fraction of a desired product that is captured in the correct sorted fraction taken into account the collection and sorting process	$R_N = \frac{f_z^T}{\mu^{TO}}$
Product grade	Purity level of a desired product in a sorted fraction	$G_T = \frac{f_z^T}{\sum_{m=1}^{M} f_z^m}$
Polymer grade	Purity level of a desired polymer in a sorted fraction	$G_P = \frac{f_z^P}{\sum_{m=1}^{M} f_z^m}$
SDI	Abundancy of different substances in a sorted fraction	$\mathrm{SCI} = \left(1 - \frac{\sum_{i=1}^{S} (n_i - 1) n_i}{(N-1)N}\right) \times 100\%$
Total analysed metal content	Sum of the concentrations of Cd, Cu, Co, Zn, Fe, Mn, Pb, Li, Mg, Sr, Tl, Sb, Ti, Ca, Mo, V, As, Ni, Al, Be, Na, and Se present in a sorted fraction	$C_M = \frac{\sum_{i=1}^{M} f_z^i}{\sum_{m=1}^{M} f_z^m}$
Total analysed halogen content	Sum of the concentrations of Cl, Br, and F present in a sorted fraction	$C_H = \frac{\sum_{j=1}^{M} f_z^j}{\sum_{m=1}^{M} f_z^m}$
C, H, N, O levels	Mass fractions of C, H, N, S and O present in a sorted fraction	$X_{CHNO} = \frac{f_z^{CHNO}}{\sum_{m=1}^{M} f_z^m} \times 100\%$

Various systems have been settled for the plastic sorting according to the collection systems and type of stream (mixed or separate). Sorting centres may be included in municipal waste collection centres, as it is the case in many French cities, but this sorting remains coarse (Figure 23). Plastics are first often separated from other waste (metals, paper, organic waste) and in some cases, sorting centres also consider separation of flexible plastics from rigid plastics. Most small plastic objects go to incineration. This generally concerns waste measuring less than 7 cm (Natura Sciences, 2020).

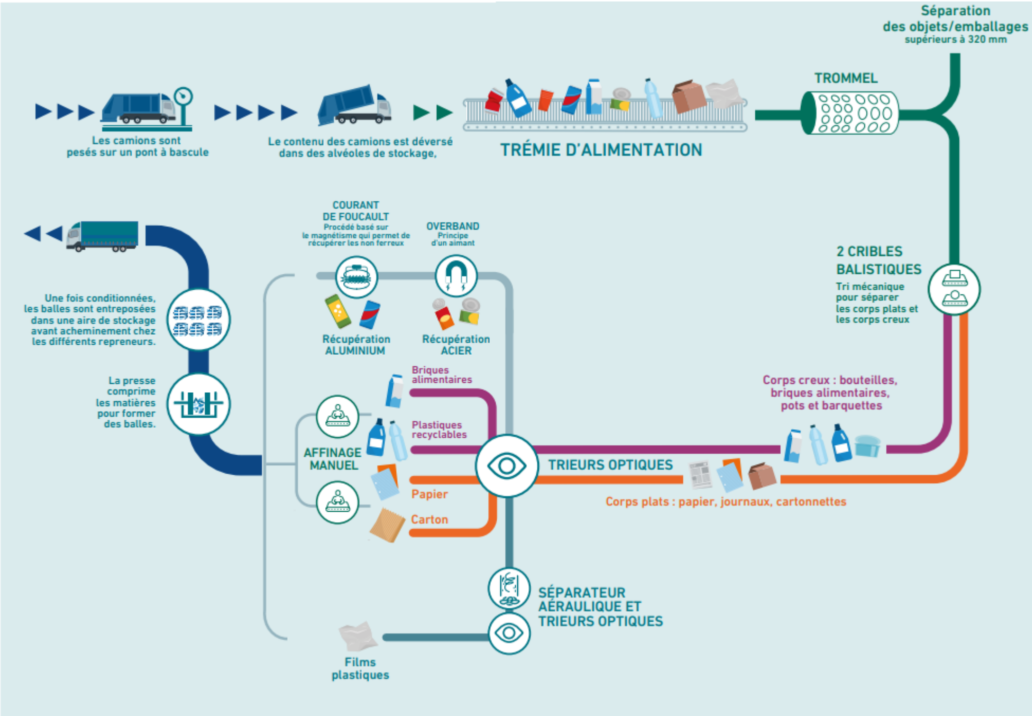

Figure 23. *Steps in the sorting process of a sorting centre in France.*
(https://www.syctom-paris.fr/actualites/paris-xv-un-centre-de-tri-modernise-pour-mieux-trier.html)

Plastics recovered for recycling are mainly rigid containers corresponding to PET (clear coming from bottle), HDPE and a rest fraction corresponding to a mixture of polymers. In some cases, the PP fraction can be sorted out as well (Gundupalli *et al.*, 2017; Ragaert *et al.*, 2017). They are prepared as bales to be sent to recycling centres. Poor bale quality (evaluated by the purity level of products) is often associated with the presence of labels on plastics, multi-layered films, mixed polymers and poorly sorted individual polymer types. Plastic films, wraps, and shopping bags from commingled waste are usually considered contaminants and sent to landfills (however, not in France) or energy recovery (Horodytska *et al.*, 2022). For now, almost no industrial technologies are settled for sorting composites, multi-layered plastics, coated plastics and flexible plastics.

Very few studies in the scientific literature relate the impact of the sorting technologies on the quality of plastics recovered for recycling. Depending on the origin of plastic waste (post-industrial or post-consumer), there can be major differences in the quality of sorting, and consequently in their possible recyclability or reusability. Effective recovery of sorted waste is highly dependent on the quality of sorting and material identification (Ragaert *et al.*, 2017) (Figure 24). The pre-sorting prior to collection (single stream collection with dedicated packaging and/or plastic waste containers) is the most efficient (Woidasky *et al.*, 2020; Schuch *et al.*, 2023). Finally, the collection method for plastic packaging waste has virtually no influence on the final quality (% impurity) of the recycled product; however, the sorting and reprocessing stages and processes do influence the final quality of the recycled product (Luijsterburg et Goossens, 2014).

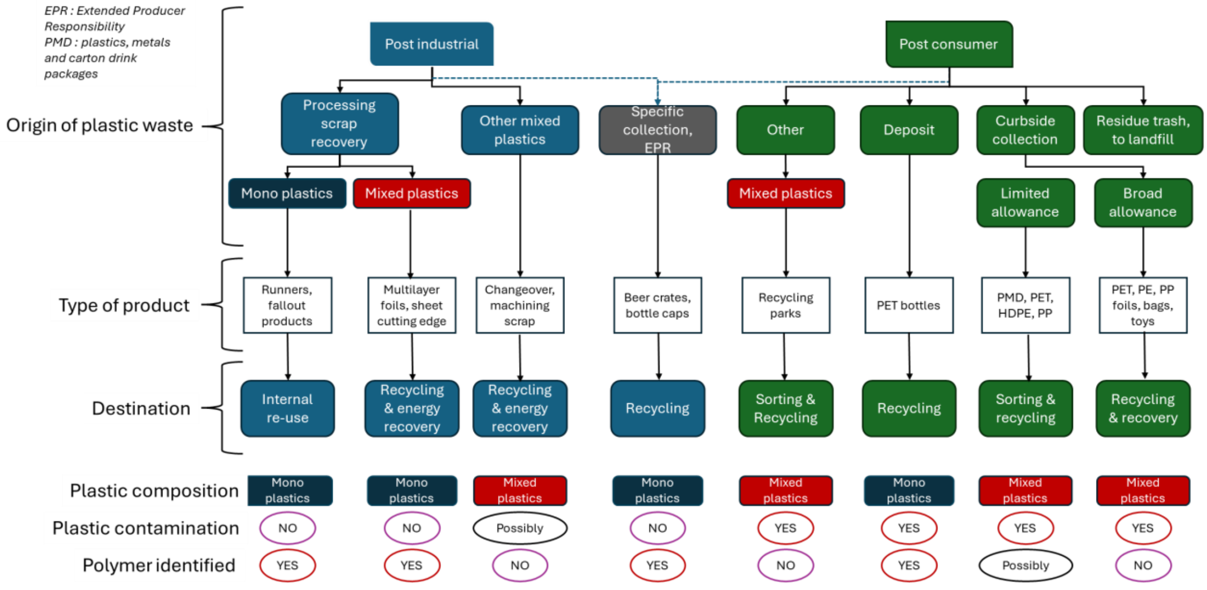

Figure 24. *Impact of origin and collection system on sorting and destination of plastic waste. Internal reuse is in fact reuse* (réemploi)*. (adapted from Ragaert et al. (2017))*

3.3.2.2. Current collection and sorting systems are not designed for biodegradable plastics

Efforts are made to consider biodegradable plastics with the objective of making the management at the end-of-life of objects easier, *i.e.*, avoiding recycling, incineration or landfilling. However, for now in France, separate collection systems do not include the separation of biodegradable plastics, which are dispersed within the other plastic and/or organic waste. Such plastics are not yet identified to be treated separately, either in composting ways or biorefineries, and very often are considered as contaminants in recycling units or composting units (Rosenboom *et al.*, 2022; Ortiz, 2023). However, when separate collection of biodegradable plastics was included in the collected streams, the overall sorting of recyclable plastics increased, meaning that separate processing of biodegradable plastics seems to facilitate the sorting of other recyclable plastics (Chioatto et Sospiro, 2023).

Similarly, compostable plastics often contaminate the recycling of conventional plastics such as HDPE and PET, reducing their value. Due to the low proportion of compostable plastics in the market, recyclers have not yet invested in equipment to sort and separate adequately compostable plastics (Hopewell *et al.*, 2009). However, several sorting technologies can be used to separate compostable and other biodegradable plastics from other plastics, such as centrifugal sorting, flotation sorting, flotation and triboelectric and near infrared (NIR) sorting (Taneepanichskul *et al.*, 2022).

3.3.3. Plastics may be recyclable but most are not recycled

(based on Chapter III.3 of the extended report)

Various technologies exist for the recycling of polymers (Table 11 and Figure 25). Among them, two categories are usually described: mechanical recycling and chemical recycling. New emerging technologies (such as biological recycling or dissolution/precipitation) exist.

Table 11. Recycling process description. (adapted from Vollmer et al. (2020))

Process name	Description
Mechanical recycling (also: secondary recycling)	Physical treatment of the plastic to achieve a consumer product from plastic waste. The most common mechanical recycling process involves melting and re-extruding the plastic.
Chemical recycling (also: tertiary recycling, feedstock recycling)	Instead of merely physically transforming the shape and macroscopic properties of the plastic, chemical changes are made through breaking bonds. Often the goal is to depolymerise the polymers into monomers. These can be used to synthesise new polymers, but other chemical building blocks can result as well. Feedstock recycling is used to describe the recycling back to feedstocks used to make new polymers that is either monomers directly or a crude oil resembling product that can be fed to steam-crackers to produce monomers.
Biotechnological recycling	It refers to the use of biological systems — such as microorganisms, enzymes, or metabolic pathways — to break down plastic materials into their monomers or other valuable compounds.
Dissolution/precipitation	In this process a plastic containing additives and impurities of other polymers or materials is dissolved. A solvent is chosen to selectively dissolve the desired polymer. Unwanted additives are filtered out and the desired polymer is precipitated. Strictly speaking dissolution/precipitation is not a chemical recycling process as usually no bonds are cleaved. However, since chemical fundamental knowledge is needed to understand the solvent/polymer interaction, solvent design and solvent recovery this process is covered in this perspective and is often considered chemical recycling.

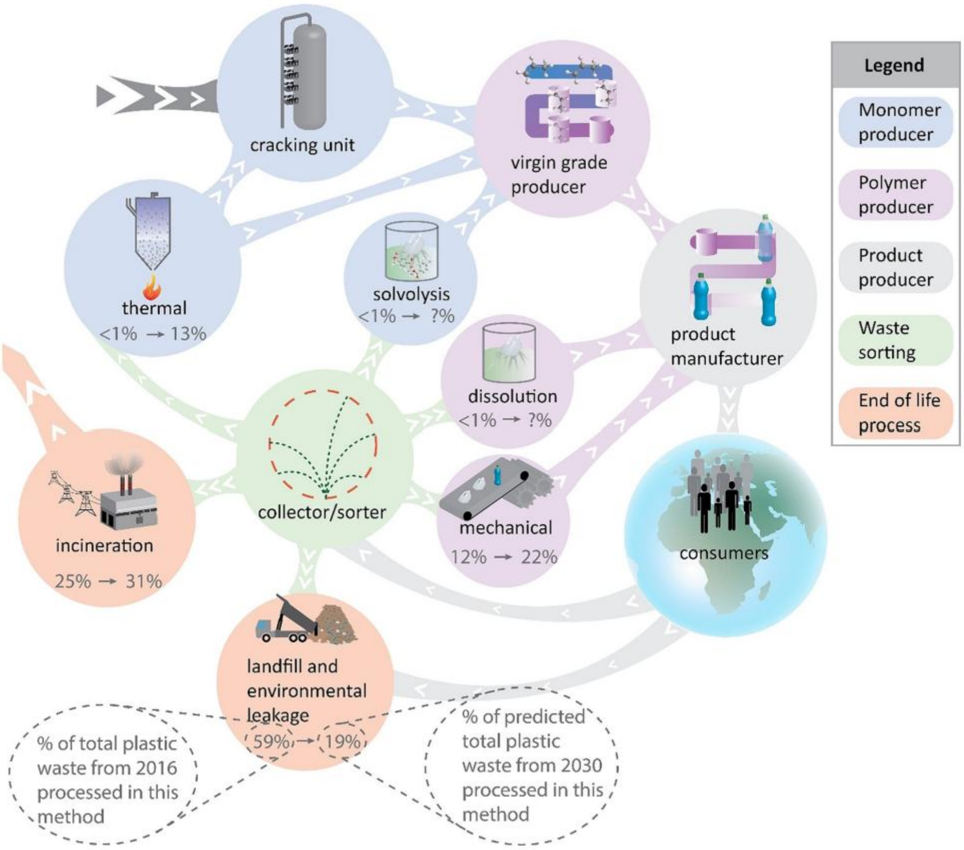

Figure 25. Plastics value chain in a circular economy. (Vollmer et al., 2020)

3.3.3.1. Mechanical recycling, the shortest loop to recycling, presents limitations

Mechanical recycling of plastics refers to the processes (primary and secondary) where plastic waste are reprocessed into secondary raw materials and products through physical methods that avoid degradation of plastics during processing in order to maintain the chemical structure of the polymers. Mechanical recycling of plastics is well researched (Figure 26). The existing literature primarily emphasises packaging overall, with a predominant focus on municipal waste and food sectors. There is limited coverage of the mechanical recycling of agricultural products despite its industrial development. Additionally, research efforts predominantly target rigid packaging, whereas attention to flexible film packaging remains minimal (Cabrera *et al.*, 2022).

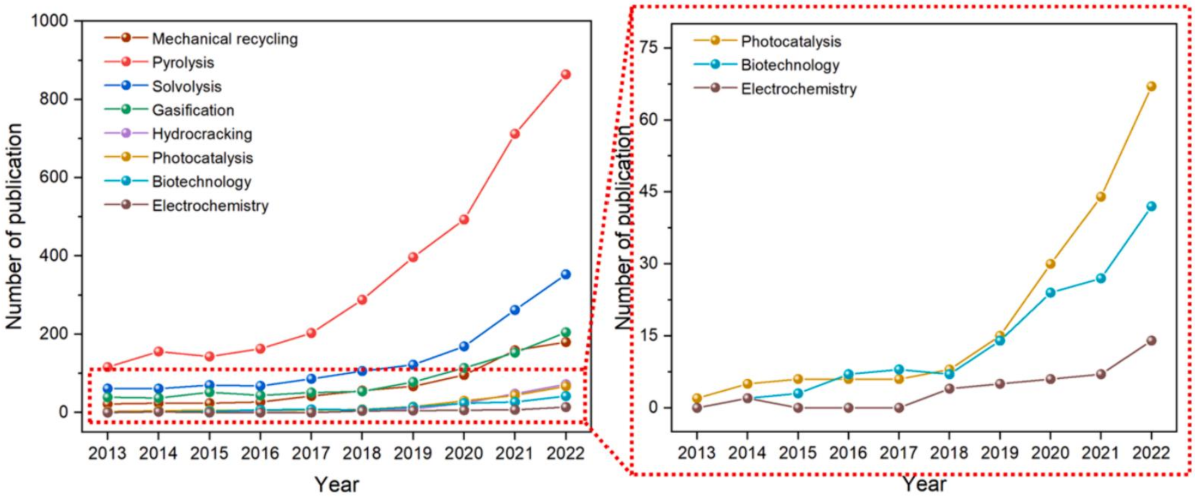

Figure 26. *Number of publications of various plastic waste recycling approaches in recent ten years (2013-2023). (Chen et Hu, 2024)*

The mechanical recycling of plastics is practically the only mature industrial solution actually used on a large industrial scale. While it is generally carried out by melt extrusion, the sorting, grinding and cleaning of materials are essential preliminary stages. Indeed, the recycling process involves carefully sorting clean and mono-stream fractions (high-quality products) and compounding it to pellets.

Recycling plastics through mechanical approach typically faces three primary challenges: pre-existing damage from ageing, the presence of mixtures (*e.g.*, multi-layered films or blends) and the inclusion of impurities or contaminants, including heterogeneous mixed plastics due to sorting issues.

When appropriate processing parameters are used and the formulation of plastics adapted, most of the polymers have the capability to go through multiple cycles of primary and secondary mechanical recycling without compromising their performance (Ragaert *et al.*, 2017). However, considering the processes put in place at an industrial level, mechanical recycling is generally limited to a few cycles using only the purest waste stream. Indeed, the extrusion process and the ageing of the polymer during its previous life can lead to degradation of the materials. While degradation mechanisms vary among different polymers, a common issue is the reduction in chain length (PP) and branching (PE) and thus to the alteration of mechanical properties. As an example, mechanical processing of polyolefins induces degradation at the molecular level and the formation of aldehydes, ketones, and short-chain hydrocarbons as well as branching leading to changes in properties. Another limitation regarding mechanical recycling is the presence of inherited substances as well as contaminants. First, they can have direct impact on the recyclability of plastics or even might support the degradation of plastic. In addition, during mechanical recycling, several potentially harmful substances, such as metals, brominated flame retardants (BFRs), POPs, and polycyclic aromatic hydrocarbons (PAHs), can be released. A portion of the plastic waste in Europe originates from products manufactured outside the continent, where the enforcement of European regulations on materials and production standards may be insufficient. This can lead to the presence of toxic substances in imported products, which then persist in the waste at the end of their life cycle. Furthermore, differences in regulatory standards between products made in Europe and those produced elsewhere can also contribute to the unintended presence of these hazardous substances (Hahladakis *et al.*, 2018). The fact that the collection system does not separate FCMs from other packaging materials can lead to cross-contamination. Coatings, labels or inks and adhesives, closures and pumps can also contaminate the stream making the mechanical recycling less efficient or increasing the treatment cost. Although labels and inks make up a small proportion of packaging, they significantly impact the PET recovery process. Inks are responsible for the formation volatile organic compounds (VOCs) and smell of recycled plastics (Cabanes *et al.*, 2020), and also cause undesirable colours during processing (Awaja et Pavel, 2005). They can affect the sorting step: to guarantee precise automated sorting of PET bottles, it is advised that label size remains below 40% of the bottle's surface area (Zheng *et al.*, 2023). Dealing with products constructed from various types of plastics poses a significant challenge in the field of recycling (Zheng *et al.*, 2023). Different types of plastics are often combined in a manner that makes mechanical separation very difficult or even impossible. These products are difficult to sort and can pollute single-material streams. As a result, they are discarded and sent to incineration. If they are properly isolated, however, they remain impossible to recover today, due to the difficulty of separating the materials at a reasonable cost. Recovery as a blend is difficult due to the incompatibility of plastics when mixed. The case of multi-layered films is a good illustration of this problem, as they are not currently recycled. As an example, polyvinylidene chloride (PVDC) offers excellent barrier properties and is commonly used as bottle's barrier layer Nevertheless, the melting temperature of PVDC is much lower than that of PET which leads to inevitable thermal degradation during the melting processing phase for PET (Ding et Zhu, 2023).

Due to these limitations, mechanical recycling is mainly open-loop (Lange, 2021). However, a critical element in promoting and sustaining plastic recycling is the ability of recyclers to create products that meet market demand and command premium prices (Bernat, 2023). A first approach to go beyond the limitations of mechanical recycling is to blend recycled stream with a virgin polymer of the same family eventually with compatibilisers and additives. In addition, many efforts have been done using antioxidants, chain extenders, blending technologies, fillers, and plasticisers (Figure 27) to counterbalance the limitations previously described. Antioxidants, along with their combinations with co-stabilisers, are crucial in mechanical recycling, enabling efficient processing without damaging the polymer chains and preserving the long-term properties of recycled materials. Additionally, they often enhance the mechanical properties of recyclates (Pfaendner, 2022). Compatibilisers are commonly used to obtain the desired performance for polymer blends starting from resins that would otherwise be incompatible. This is particularly used considering mixture of polymers corresponding to the presence of impurities in the stream obtained after collection and sorting (Zhan *et al.*, 2025) but also considering polymer mixtures and/or multi-layers.

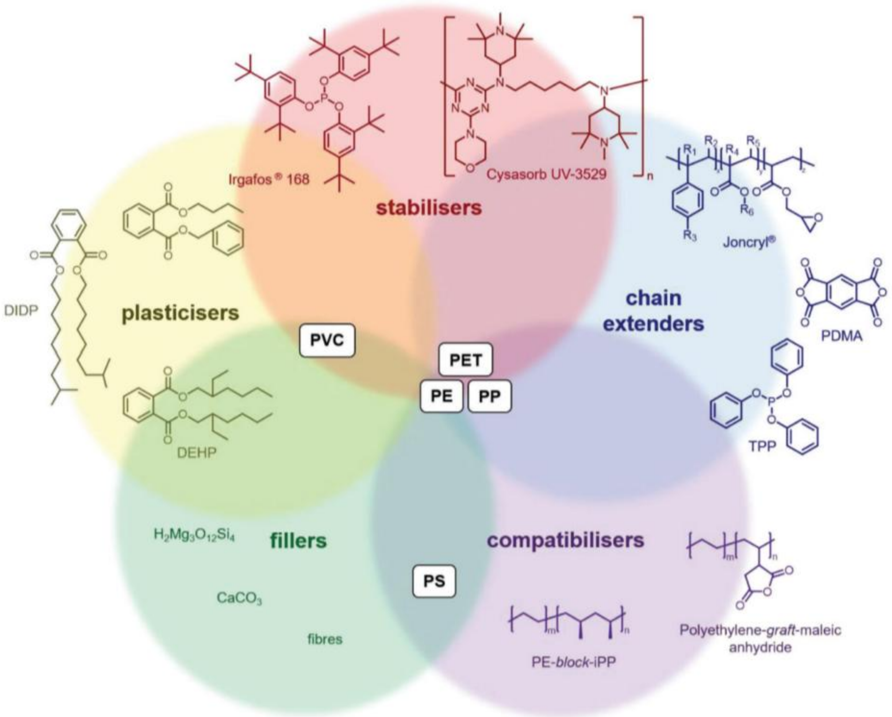

Figure 27. *Common polymer additives used to improve polymer recyclates. (Schyns et Shaver, 2021)*

Moreover, decontamination processes have been proposed in the literature in order to permit the use of recycled plastics in various fields of application and among them FCMs or agricultural films. Several technologies have been developed, including the super-clean process, which is a specialised washing method. This technique involves thoroughly cleaning materials using various approaches such as high temperatures, vacuum or inert gas environments, and non-toxic chemical agents applied to the surface (Garcia *et al.*, 2014). This approach was recently applied to recycled HDPE in a context of FCMs. In the analysis of recycled HDPE flakes and pellets, 67 compounds were identified, including additives, degradation products, and contaminants from previous uses. Thermal drying at 120°C under vacuum for 5 hours was found to be the most effective reducing volatile content by over 50% for 42 of the compounds in the case of pellets. Although decontamination was faster for flakes, it was less effective. The efficiency of decontamination decreased with increasing molar mass of the contaminants. Migration tests on treated and untreated samples showed a significant reduction in compound concentration for decontaminated materials. Similar migration values were observed in food simulants like 3% acetic acid and 10% ethanol, but higher values were found with 95% ethanol, suggesting caution for recycled HDPE use with fatty foods (Pérez-Bondía *et al.*, 2024). Another technique reported for the decontamination of plastic waste is the extraction with supercritical CO_2 ($scCO_2$). This technique has been used as a decontamination technology for recycling polyolefins such as PP, HDPE, LLDPE and LDPE. A theoretical model was developed to analyse extraction yields based on diffusion through the polymer matrix and solubility in $scCO_2$. A recent patent outlines an industrial process for polyolefin recycling using supercritical fluids, particularly CO_2, although it does not address FCMs. Studies on PP and LLDPE examined the effects of material shape (pellets or films), thickness, and operational conditions (time, temperature, pressure) on process performance. The antioxidant Irganox 1010 was effectively extracted, with better results in LLDPE than in PP due to differences in molecular structure and Tg. For PP, $scCO_2$ was also tested on model contaminants (stearin, dilaurin, trilaurin, and tripalmitin), showing reduced extraction yields as the molar mass increased (from 358 to 807 g/mol) (Singh *et al.*, 2023).

Extrusion is the main process used in the field of mechanical recycling. However, an alternative route is the dissolution/precipitation process, where plastics are dissolved in a suitable organic solvent and subsequently precipitated by introducing a non-solvent. Undissolved materials like pigments are separated from the polymer solution. This method thus enables the recovery of polymers from plastic waste devoid of additives like pigments. An optimal solvent for dissolution/precipitation can significantly enhance mechanical recycling efficiency by facilitating both the separation of desired plastics and the removal of contaminants. In some cases, additives, such as flame retardants, can even be recovered for reuse (Vollmer *et al.*, 2020). The solvent and anti-solvent mixture must however be separated for reuse, a process that is both time- and energy-intensive, especially for solvents with high boiling points. Additionally, complete removal of residual solvents is critical, as any remaining solvent can impact the properties of the polymer. This process becomes easier when one of the solvents is a supercritical fluid, as it readily evaporates when the pressure is reduced (Vollmer *et al.*, 2020). Several dissolution/precipitation processes have been implemented on an industrial scale such as the NewCycling process (8000 Mt/year capacity for PP and PE) or additionally, PolyStyreneLoop (capacity of 3300 t in Terneuzen applied to PS foam).

Pin *et al*. (2023) compared, in the case of expanded polystyrene (EPS), the two mechanical recycling technologies (solvent-based and mechanical recycling technology approaches). The comparison focused on how each method affected molar mass, contaminants in the recycled material, and the resulting melt viscosity and mechanical properties. Mechanical recycling through extrusion showed limitations due to the quality of the feedstock, with significant reductions in molecular weight, changes in colour, and the release of volatile chemicals. This method can thus only be used for very clean, pre-washed feedstock and typically white EPS. It also requires the addition of stabilisers and radical scavengers to maintain material properties over multiple reprocessing cycles and minimise degradation during extrusion. At the opposite, the solvent-based technology developed in this study showed that it can preserve the molar mass of PS feedstock. This method is also effective at removing both soluble and non-soluble contaminants, including dust and flame retardants. Additionally, it reduces residual styrene monomer content by 75%.

3.3.3.2. Chemical and enzymatic recycling routes are at their early stage

Chemical recycling is the process of converting polymeric waste by changing its chemical structure through breaking bonds. It turns polymer back into substances that can be used as raw materials (monomers or other molecules) for the manufacturing of plastics or other products. It is well described in the literature (Figure 26) and among the existing processes, pyrolysis was the most studied one.

Research has been focused on virgin or single types of plastics. However, actual plastic waste, which contains various impurities, is significantly different from these pristine feedstocks. This is particularly true for chemical recycling and it is generally accepted that these processes can treat various types of plastic waste with high tolerance to contaminations due to high operating temperatures. However, this point is rarely discussed in the literature (Chen et Hu, 2024).

Traditional chemical recycling approaches, including thermal processes (pyrolysis, gasification, and hydrocracking) and chemical processes (solvolysis), operate under relatively harsh operating conditions. These conditions allow high degradation rate and capacity for large industrial scales. As a result, they have been only successfully implemented on a medium-industrial scale and for some applications on a large industrial scale (Solis et Silveira, 2020). These processing conditions make these processes cost-effective only with large volumes of plastic waste.

The categories of polymers that are very promising candidates for chemical recycling include the following (Podara *et al.*, 2024):

- Polyolefins (such as PE, PP) that can be chemically recycled via thermal processes (Table 12);
- Polycondensation polymers such as PET, PA, PLA, and PC that can be recycled by solvolysis processes;

- Polyaddition polymers with charge de-localising side groups (*i.e.*, alkenyl, phenyl, ether groups, halogens) such as poly (methyl methacrylate) (PMMA) and poly (tetrafluoroethylene) (PTFE) can also be recycled via chemical processes.

Table 12. *Suitable plastics for pyrolysis. (adapted from Miandad* et al. *(2016))*

Resin	Suitability for pyrolysis	Remarks
Polystyrene (PS)	Very good and gives excellent fuel properties	(1) Low temperature required as compared to PP and PE. (2) Produce Less viscous oil as compared to PE and PP
Polyethylene (HDPE, LDPE, LLDPE)	Very good for pyrolysis	(1) Required temperature is high >500 ◦C due to its branched chain structure (2) In thermal pyrolysis it converts into wax instead of liquid oil (3) In catalytic pyrolysis wax formation is occurred on external site of catalyst while further cracking of wax into gases and liquid occurred in internal site of catalyst
Polypropylene (PP)	Very good	(1) Required high temperature. (2) After PE it is difficult to degrade thermal pyrolysis of PP. (3) In catalytic pyrolysis produces liquid yield with high aromatic compounds.
Polyvinyl chloride (PVC)	Not suitable. Only few studies were carried out by different scientists	(1) Produce hazardous Chlorine gas (2) Dechlorination via low temperature (250–320 ◦C) or physical or chemical adsorption. (3) In catalytic pyrolysis presence of chlorine and deposition of coke affect the catalytic activity of catalyst.
Polyethylene terephthalate (PET)	Not suitable	(1) It contains heteroatoms

Pyrolysis is the process of thermally degrading long chain polymer molecules into smaller, less complex molecules through heat and pressure. The process requires intense heat with shorter duration and in absence of oxygen. The three major products that are produced during pyrolysis are oil, gas and char. At high temperatures, polymers undergo random scission due to heat-induced radicals, resulting in low selectivity and quality of the end products. Catalytic pyrolysis is considered to improve these parameters (Miandad *et al.*, 2016). A catalyst plays a key role in enhancing both the quality of the product and the overall efficiency of the pyrolysis process. Various researchers have experimented with different catalysts, including ZSM-5, HZSM-5, FCC, natural zeolite, activated alumina, and red mud (Miandad *et al.*, 2016). Catalytic pyrolysis operates at lower temperatures, requiring less energy. The use of catalysts, particularly those with high Brunauer–Emmett–Teller (BET) surface area, helps to reduce impurities in the liquid oil and shift the product distribution toward low molecular weight hydrocarbons in the gasoline range. Although this process may decrease the total liquid yield, the produced liquid has a higher octane number, making it more valuable for transportation fuels. Catalytic pyrolysis also promotes the formation of C_3 and C_4 gases, which are important intermediates in petrochemical industries. Additionally, the presence of catalysts lowers char production, reducing solid waste generation. Overall, catalytic pyrolysis offers greater selectivity, cleaner products, and better fuel quality, while thermal pyrolysis is more straightforward but less efficient. The choice between the two depends on the desired product profile and available processing infrastructure. Thermal pyrolysis usually proceeds through a free-radical mechanism whereas catalytic pyrolysis proceeds through a combination of free-radical and carbocation mechanisms. High monomer yields can be obtained from PMMA pyrolysis because of the dominance of chain-end scission reactions as opposed to random scission in polyolefins (Figure 28). With an increase in the number of functional groups and heteroatoms in the backbone of the polymer, the distribution of products and the pyrolytic mechanisms become less complicated. Process parameters provide a higher degree of control over the product distribution for polyolefins, in this example HDPE, although the ultimate monomer yield is lower than for PS and PMMA. A common theme of all pyrolysis is that an excessively high temperature leads to coke formation given that the residence time is long enough.

Although, most reaction steps occur at lower temperatures, similar trends are observed for catalytic processes (Vollmer *et al.*, 2020).

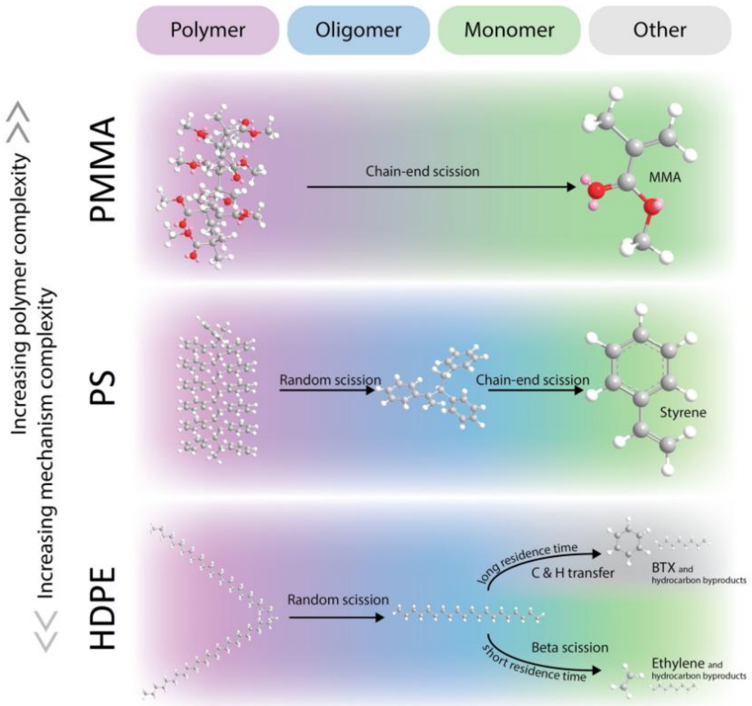

***Figure 28**. Comparison of pyrolysis products from HDPE, PS and PMMA. BTX denotes benzene, toluene, and xylene. (Vollmer et al., 2020).*

The information on contamination and the need to upgrade pyrolysis products is rarely discussed in the literature (Belbessai *et al.*, 2022). In the few studies investigating this aspect and considering pyrolysis of polyolefin, it was shown that in incorrectly sorted plastics, the stream can contain polymers with heteroatoms, such as PET, PVDC, PA, and PUR, along with organic residues, additives, and pigments, which are significant sources of heteroatoms and metals. These contaminants can partially be transferred into pyrolysis products, contributing to the wide variability in hydrocarbon composition and contaminant levels. Moreover, the presence of contaminants such as heteroatoms and metals leads to operational problems such as corrosion, process fouling and downstream catalyst poisoning. This leads to the need of pre-treatment processes like melt filtration and thermal dehalogenation to remove contaminants before pyrolysis (Kusenberg *et al.*, 2022). This study concludes that to achieve good-quality products, such as fuels and chemical precursors, the pre-treatment of the feedstock is necessary and satisfactory products cannot be obtained from a heterogeneous mixture of waste.

Solvolysis is a chemical process that breaks chemical bands through reactions with solvents (Figure 29). This method is particularly effective for plastics containing C–X bonds, as breaking C–C bonds via solvolysis is highly difficult. Much research and development activities focus on PET and PUR, and to a lesser extent on PA, PC and PLA depolymerisation. Several reasons may explain the high interest of researcher towards solvolysis of PET (Vollmer *et al.*, 2020). PET is a unique example of a relatively pure mono-stream with minimal contamination. Mechanical recycling of PET from bottles can achieve rates as high as 15 wt%. However, the recycled material is typically blended with virgin resin to preserve both colour and structural integrity. Alternatively, highly sorted polyester streams can undergo depolymerisation through solvolysis, producing monomers that can be used by virgin-grade manufacturers to create new polymer resins (Vollmer *et al.*, 2020). Common solvolysis reactions for PET include glycolysis, hydrolysis, methanolysis, and ammonolysis, where PET interacts with EG, water, methanol, and ammonia, respectively. These reactions lead to the formation of monomers or their derivatives (Chen et Hu, 2024). Challenges common to all solvolysis processes include separating the solvent (liquid cleavage agent) from by-products, the limited contact area between the liquid cleavage agent and the solid polymer, and the retrieval of dissolved catalysts (Vollmer *et al.*, 2020). Solvolysis operates at relatively lower temperatures (50–300°C) than other approaches such as pyrolysis, which requires lower energy input. Additionally, solvolysis usually processes C–X-containing plastics, such as PET, resulting in higher selectivity of products (Chen et Hu, 2024).

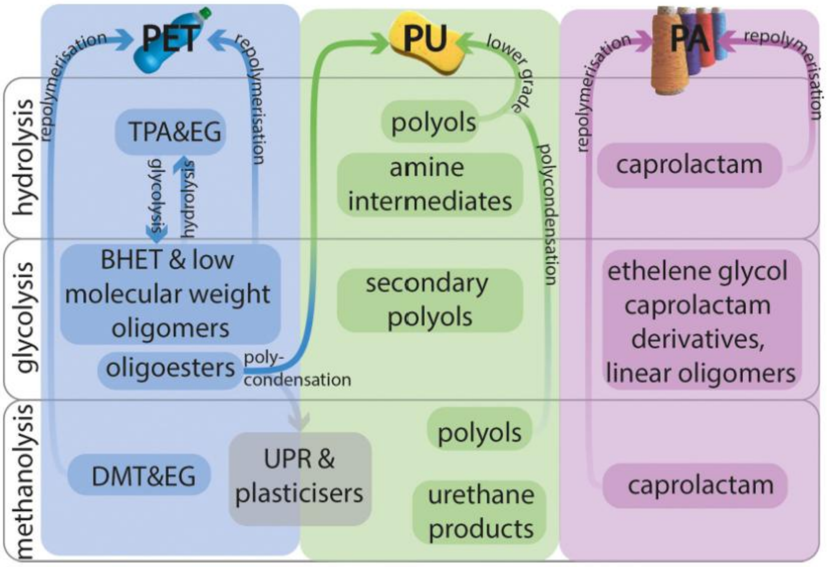

Figure 29: *Products obtained through the different solvolysis pathways of PET, PUR, and PA, and how these products can be used to recycle back to the polymer or to obtain valuable products. (Vollmer et al., 2020).*

Pyrolysis and solvolysis represent the two most studied chemical recycling methods for plastic waste and are applied industrially. Pyrolysis, applicable to all plastic types, offers the advantage of producing energy and a broad range of products — solids, liquids, and gases — while minimising the release of toxic emissions. However, it involves complex reaction chemistry, it is only cost-effective at large scales, and it suffers from reduced product quality due to heteroatoms. In contrast, solvolysis is more selective and effective for specific polymers like PET, PLA, and PC. It enables extensive depolymerisation reactions using solvents, making it suitable for producing low molecular weight compounds. Still, this process is challenged by long depolymerisation times, sensitivity to additive-rich plastics, and difficulties in impurity separation. Therefore, while both techniques have distinct strengths, their applicability depends heavily on feedstock composition and desired end products.

Beyond these two techniques, innovative plastic recycling methods, such as photocatalysis, biotechnology, and electrochemistry, are attracting growing interest as potential future solutions. However, these emerging technologies face the challenge of low conversion efficiency, driving the need for more effective catalysts and advanced reaction systems. Despite their promise, practical implementation remains limited due to low economic viability (Chen et Hu, 2024). Among them, enzymatic recycling routes were particularly studied.

Enzymatic recycling routes can selectively break down biodegradable plastics into valuable compounds. However, the commercialisation of this method is still limited due to low yields and high downstream costs, necessitating the development of new technologies to address these challenges. We can however mention the company Carbios in France. Carbios is a French biotechnology company specialised in enzymatic recycling of plastics, particularly PET. It has developed proprietary enzymes that break down plastic waste into its original monomers for recycling in new products. Literature on enzymatic recycling routes focuses on recycling of PET. A significant number of PET hydrolytic enzymes (PHEs) have been identified, characterised, and optimised through various protein engineering approaches for practical use (Shi et Zhu, 2024). They have successfully achieved the complete depolymerisation of post-consumer PET waste under mild conditions. The monomers resulting from enzymatic depolymerisation have been recycled to regenerate PET or metabolised by microbes to create other valuable polymers. In addition to improving the intrinsic properties of PHEs, considerable efforts have been made to tackle external challenges in the enzymatic PET depolymerisation process, such as the cost of enzyme production, the high crystallinity of certain PET waste, and the costs associated with product separation and recovery (Shi et Zhu, 2024). Significant advancements have also been made in the biological degradation of other plastics materials (Table 13). As an example of interest, a novel bacterial consortium has been identified, comprising four specific microorganisms — *Bacillus vallismortis*, *Pseudomonas protegens*, *Stenotrophomonas* sp., and *Paenibacillus* sp. — which work synergistically to effectively degrade both LDPE and HDPE. When tested individually on LDPE, the isolated strains exhibited degradation efficiencies ranging from 15.5 to 19.3% over 120 days at 55°C and pH 8.0. However, when combined, the consortium dramatically increased degradation efficiency to 75%. Polymers such as polycaprolactone (PCL) and PLA containing less than 2 wt% enzymes can be depolymerised within days, resulting in up to 98% conversion of polymer to small molecules in standard soil composts and household tap water (Chen et Hu, 2024).

Table 13. *Biological destruction* of different types of plastics. (Chen et Hu, 2024)*
*(*Biological destruction involves the use of enzymes or microorganisms to break down plastics, effectively degrading them).*

Plastics	Microorganisms/Enzyme	Condition	Duration	Degradation efficiency
PE	Acinetobacter pitti IRN19	30 °C	4 weeks	26.8 %
	Cupriavidus necator H16	30 °C	21 days	33.7 %
	Bacterial consortia	55 °C, pH 8.0	120 days	75.0 %
	Enterobacter spp nov., *Pseudomonas aeruginosa* nov.	37 °C	160 days	64.3 %
PP	*Enterobacter* spp nov., *Pseudomonas aeruginosa* nov.	37 °C	160 days	63.0 %
	Bacillus thuringiensis	37 °C	15 days	12.0 %
PS	*Mealworms*	Not mentioned	60 days	7.4 %
	Enterobacter sp.	30 °C	30 days	12.4 %
PVC	Pseudomonas citronellolis	30 °C	30 days	6.88 %
	Marine consortia	20 °C	7 months	11.7 %
PET	Engineered *Clostridium thermocellum*	60 °C	14 days	62.0 %
	Glycosylated LCC	70 °C, pH 8	48 h	95.0 %
	Tfcut2	70 °C	120 h	97.0 %
	Ideonella sakaiensis 201-F6	70 °C	6 weeks	~100 %
	LCC – ICCG variant	65 °C, pH 8	14 h	92.3 %

In conclusion, chemical recycling has an advantage over mechanical recycling in that the recovered polymer can be repolymerised an indefinite number of times to reproduce the corresponding polymers (Chen et Hu, 2024). However, chemical recycling aims at overcoming limitations of mechanical recycling but this is not totally true (Table 14) and it is rather a complementary approach to mechanical recycling (including dissolution-precipitation). The chemical and biological (enzymatic) recycling can enable an extended recycling of the carbon content.

Table 14. Advantages, disadvantages, and economic feasibility of plastic recycling approaches. (Chen et Hu, 2024)

Approaches	Advantages	Disadvantages	Economic feasibility
Mechanical recycling	• Relatively simple process • Energy efficiency • Low cost • Closed-loop recycling	• Quality degradation • Contamination issue	High
Traditional chemical recycling (pyrolysis, solvolysis, gasification, hydrocracking)	• Broad versatility and tolerance to raw materials • High degradation efficiency • Ability to deal with complex plastics	• High energy cost • Harmfulness to environment • Low product selectivity and quality	High for solvolysis, medium for others
Innovative chemical recycling (photocatalysis, biotechnology, electrochemistry)	• Low energy consumption • Environmentally friendliness • High product selectivity	• Low degradation efficiency • Limited applicability	Low, potentially high in the future

3.3.3.3. Recycling and food contact plastics: stringent regulations and the state of knowledge only allow for mechanical recycling of food contact PET and food contact plastics in a closed and controlled chain

FCMs include all materials and articles intended to come into contact with food and drink, such as containers, packaging, kitchen utensils, cutlery and dishes. They also include materials used in food processing equipment, such as coffee machines or food production machinery, as well as containers used to transport food. FCMs can be made from a variety of materials such as plastic, rubber, paper, ceramic or metal. In the case of FCMs, the use of recycled plastics represents a major challenge because chemical substances contained in materials in contact with food can migrate into the food and drinkable fluids we consume, and conversely, chemical substances in food and drink can migrate into plastics. During recycling, these substances and others contained in plastics (NIAS and inherited substances) are not necessarily eliminated. In the case of new use in FCMs, these substances can pose health risks if they migrate. The use of recycled materials in FCMs should thus be fully controlled. Moreover, FCMs represent a large amount of plastic waste and their recovery after the end-of-life of objects, in particular in closed loop, have to be considered.

The fundamental requirement of Regulation (EC) No 1935/2004 (2004) applied to FCMs is that materials or articles, including recycled, intended to come into contact with food must be sufficiently inert. This means that substances from the article/material do not migrate into food in quantities that may endanger human health or bring about an unacceptable change in the composition of the food or a deterioration in its properties. Apart from chemical hazards, microbiological contamination of food shall be precluded. Provided that the input material for recycling is a waste, it is much more likely to be chemically or microbiologically contaminated than materials and articles that have been newly manufactured from virgin substances. The use of recycled materials for FCMs is thus challenging.

The European Food Safety Authority (EFSA) Scientific Committee (2012) established that for substances with a structural alert[11] for genotoxicity, a threshold exposure of 0.0025 µg/kg body weight (bw) per day should be applied. This value is calculated for the value of 0.15 µg/person for an adult weighing 60 kg. Other threshold values are reported as followed: 18 µg/person per day for organophosphate and carbamate substances with anti-cholinesterase activity, 90 µg/person per day for Cramer Class III and Cramer Class II substances, and 1800 µg/person per day for Cramer Class I

[11] Parts of organic molecules which are believed to be responsible for adverse effects (e.g. genotoxicity) and can be used to predict the toxicity of similar compounds. https://www.efsa.europa.eu/en/glossary/structural-alert#:~:text=Description%3A,the%20toxicity%20of%20similar%20compounds

substances, but for application to all groups in the population, these values should be expressed in terms of bw, *i.e.*, 0.0025, 0.3, 1.5 and 30 µg/kg bw per day, respectively. Based on this assessment, the threshold value obtained for genotoxicity is taken into account when applying the decontamination efficiency and calculating the migration of the substance under examination from the plastic article to the food.

To protect consumers from a chemical hazard, the recycled plastic material has to follow specific procedures of collection, sorting, and decontamination. EFSA is entitled to perform the risk assessment of a recycling process if it is required by the legislation. For this purpose, EFSA published guidelines with the requirements for the submission of a dossier to evaluate a recycling process (European Food Safety Authority, 2008). EFSA has published more than 200 opinions on recycling processes based on the requirements of the new Regulation (EU) 2022/1616 (2022) which updates and simplifies existing regulations on the development, certification, and use of FCMs, including innovative novel plastic recycling technologies under development. With this, the previous Regulation (EC) No 282/2008 (2008c) is repealed. These opinions mainly concerned recycled PET, a few closed loops and HDPE.

Considering chains where materials do not pass through the hands of the consumers and according to the Regulation (EU) 2022/1616 (2022), there is no requirement for safety assessment by EFSA. This is based on the presumption that a controlled and closed chain of circulation of the products prevents contamination, and subsequently any contamination of the plastic input can be removed with the simple cleaning and heating processes applied when the material is recycled. However, the recyclers are obliged to apply a quality management system, perform quality control, and keep records. In addition, Member States authorities should perform audits on a regular basis (European Union, 2022).

In case the material passes to consumers, it is required to demonstrate the safety of the process. For a recycling process for the production of FCMs, it is required that the input materials originate from previous food contact uses, which can be challenging considering the collection systems. Specifically, for PET, a 5% content of non-food contact articles is permitted. The articles are then ground, washed and dried. However, this step is not considered sufficient to remove contaminants that may have migrated inside the plastic mass. Additional process steps are required to achieve to the quality required for a use as FCMs. A scientific opinion on the criteria that are used for the safety evaluation of PET mechanical recycling processes was published in 2011 (EFSA Panel on food contact materials enzymes et processing aids, 2011). In that case, a step of polycondensation is included either in solid state or in melt state. This step is performed in specific reactors operating in batch, semi-continuous or continuous mode. The operating conditions shall be kept under continuous control. PET mechanical recycling processes are the only one that have been granted positive evaluations for the production of FCMs. The first PET bottle-to-bottle recycling plant in Europe was installed in Beaune, France, in 1998 (Franz et Welle, 2022). Then, numerous recycling installations were established in Europe focusing mainly on post-consumer PET bottles while recycling of PET trays does not find significant applications. This canbe due to the fact that trays were not collected before the sorting instructions extension and thus that there was no material and no associated market. There are several reasons why PET recycling has advanced more than other materials. PET has low diffusivity and absorbs minimal amounts of compounds during use, with only low molecular weight compounds (up to approximately 200 g/mol) needing removal during decontamination. Unlike other plastics, PET does not require common additives like plasticisers, or slip agents, which can break down during use and recycling leading to the presence of NIAS. Additionally, residual polymerisation catalysts in recovered PET help maintain the polymer chain length, increasing viscosity and regulating its physical and mechanical properties. Advanced decontamination methods, such as solid-state polycondensation, have been developed to thoroughly cleanse post-consumer PET using high temperatures, vacuum, and gas flow, effectively reducing potential contaminant levels (Singh *et al.*, 2023). At the opposite, in the case of polyolefins according to the challenge tests provided, limited decontamination efficiency for the installations under evaluation was demonstrated. From a review paper (Palkopoulou *et al.*, 2016), it is demonstrated that, in comparison to PET, PE and PP post-consumer waste may be contaminated with a wide range of chemical substances, *e.g.*, with higher molecular masses, which may not satisfactorily be removed during decontamination steps of recycling processes. It may mean that further research is needed in order to achieve appropriate decontamination of mechanical recycling processes or gain data on contamination of polyolefin waste. Some fields of research focus on different ways of decontamination of polyolefins waste in order to be used for food contact but they are not at present applied successfully at industrial scale. Even if studies regarding biodegradable materials are scarce, research was carried out on PLA reprocessing but it was demonstrated that the mechanical properties are affected, limiting its use for new articles. Moreover, it was demonstrated that the decontamination approach that is used for mechanical recycling of PET cannot be used for biodegradable polymers (Dedieu *et al.*, 2022b). On the other hand, the mechanical recycling of another biodegradable polymer, polyhydroxybutyrate-co-valerate (PHBV), for food contact purposes, does not currently seem effective due to issues of thermal stability and the sorption/desorption properties of the material (Dedieu *et al.*, 2022a). In addition, PHBV can be contaminated during its use by chemical substances with a wide range of molecular mass and polarity. Therefore, once placed on the market and collecting systems established, studies on PHBV post-use contamination will be required (Dedieu *et al.*, 2023).

To use non-decontaminated post-consumer recycled plastics for FCMs, an alternative approach consists in the use of functional barrier. According to Commission Regulation (EU) 10/2011 (2011) (EU, 2011) a ‘functional barrier’ means a barrier consisting of one or more layers of any type of material, which ensures that the final material or article does not transfer its constituents to food in quantities which could: endanger human health, bring about an unacceptable change in the composition of the food, or bring about a deterioration in the organoleptic characteristics thereof. The kinetics of contaminant transfer from plastic layer to a functional barrier have been studied (Feigenbaum *et al.*, 1997). In general, however, the use of functional barrier in plastic films prevents further recycling the material because it leads to multi-materials (3.3.2).

3.3.4. Most plastics are neither biodegradable plastics nor biodegraded

(based on Chapter III.4 of the extended report)

To limit environmental pollution by conventional plastics, new plastic materials are developed that are called either bioplastics, compostable or biodegradable plastics. However, for many scientists the term bioplastics should be avoid since it encompasses plastics of bio-based origin whatever their biodegradability, and biodegradable plastics whatever their origin. This confuses users to have under the same name bio-PE that is bio-based but not biodegradable, PCL that is petroleum-based but biodegradable, and PHAs that are both bio-based and biodegradable. While non-biodegradable plastics should be sorted, collected and if possible recycled, the compostable and biodegradable plastics are supposed to be disposed of along bio-waste to end their life in composting or AD plants, avoiding the need to develop a new waste management network. However, it is important to ensure that their degradation is complete and safe for human health and the environment.

3.3.4.1. Composting and anaerobic digestion are different processes

Composting and AD are two natural biological processes that break down putrescible organic matter into stabilised products, namely compost and digestate, that are spread on the soil as organic fertilisers. These processes can be carried out at different scales and under different temperature conditions. In industrial composting, the organic waste is managed in tunnels or windrows of several cubic meters and the aeration is generally boosted mechanically. In home composting however, the input of fresh organic matter and the compost aeration are too low to promote a strong heat production and sanitisation of the pile. The compost temperature and moisture are not controlled and the mixing relies on the operator willingness. AD exists today primarily at industrial scale and is performed at mesophilic (35 to 40°C) or thermophilic (50 to 58°C) temperature. Most of the facilities installed in Europe are run at farm in mesophilic processes to valorise livestock effluents, either alone or with urban or agri-food organic co-substrates. During composting, the organic matter is mineralised by bacteria and fungi into principally carbon dioxide (CO_2), water (H_2O), mineral salts (nitrate, and others) and heat. It produces dried and stabilised compost rich in humic substances. During AD, the organic matter degradation is performed by bacteria and archaea that produce a biogas composed principally of methane (CH_4) and CO_2, and a liquid or pasty undigested residue enriched in ammonium called digestate. The high temperature (70°C for several days) and the presence of fungi with strong enzymatic capacities make composting a more efficient process for breaking down recalcitrant organic matter.

3.3.4.2. Plastics enter the composting and anaerobic digestion chains

Beside atmospheric deposit and accidental discharge, the two major routes of plastics to end up in compost and digestate are the valorisation of wastewater sewage sludge and bio-waste by composting and AD processes. Indeed, sewage sludge retains small plastic particles from wastewater (from laundry and personal care products) and carries them with it when it is composted or digested. Similarly, bio-waste can bring with it varying quantities of plastics, depending on how it is collected and sorted.

In European countries, two strategies exist for bio-waste management. In the first one, household waste or packaged food waste are collected without sorting by the waste producer, and the mixed waste is sorted in specific facilities where it undergoes several manual and mechanical sorting steps to remove large solid materials (wood, paper, plastics, etc.) and metals. The resulting fraction, enriched in organic residues, is then screened before undergoing a composting or AD step or both (AD followed by composting of digestate). The technology used to sort unwanted materials from the waste is a key determinant of the final product quality, since it may fragment the plastic materials (Sholokhova *et al.*, 2022b; Porterfield *et al.*, 2023). The second strategy, called selective collection, relies on waste producers to sort the organic fraction of their waste on site, using specific collectors or bags made of either kraft paper or compostable plastic bags. The bags and bio-waste are then collected in a door-to-door process or brought to street collection bins. The collected bio-waste is then sent to waste treatment centres to undergo composting or AD. Since 1 January 2024, sorting bio-waste is mandatory in France for all waste producers — professionals and private individuals — in accordance with the law No 2020-105 against waste and for a circular economy (AGEC law) (République française, 2020).

3.3.4.3. Plastic degradation in the environment is a complex process

The mechanism of plastic degradation is a complex process involving several steps influenced by many parameters related to plastic properties and environmental conditions (Figure 30).

The linear carbon structure of the polymers, their high length and molecular weight, low chain flexibility, high crystallinity and the presence of side chains, influence their biodegradation. Many environmental factors can increase material biodegradation, *i.e.*, sun exposure, temperature, repeated freezing/thawing, moisture, high acidic or alkaline conditions, and high oxidative conditions (Figure 31). Some biological factors (such as the nature of the microorganisms, their abundance, etc.) are also important for plastic degradation.

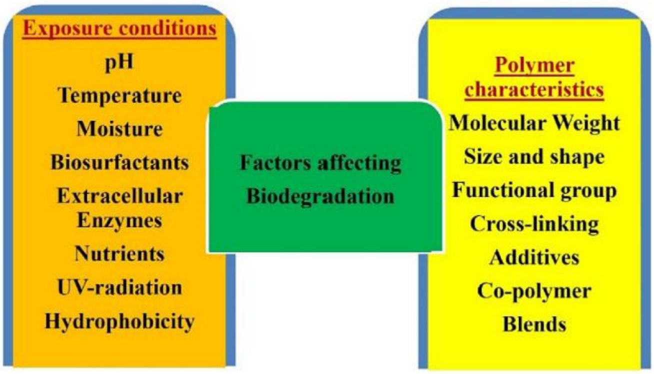

***Figure 30**. Factors affecting polymer biodegradation. (Kumar et al., 2023)*

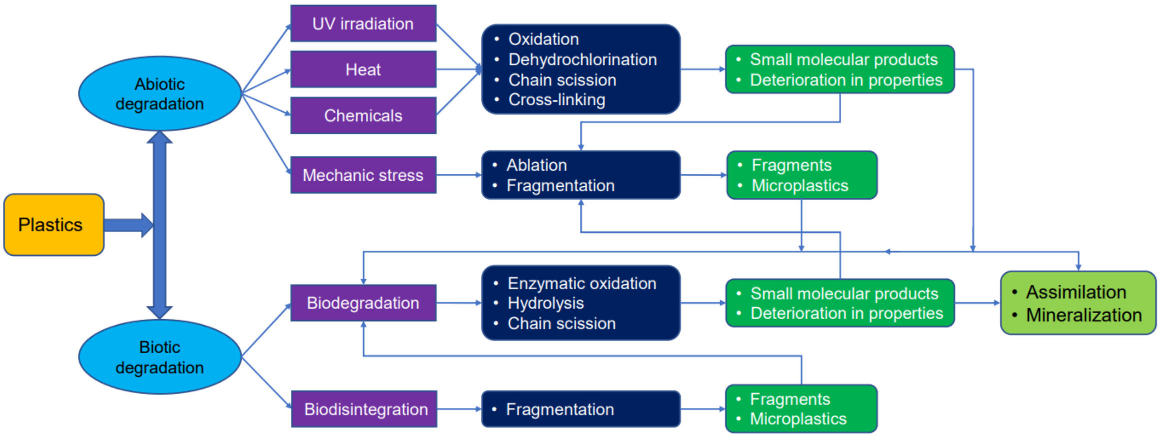

***Figure 31**. A schematic diagram showing the general processes involved in the degradation of plastics. (Zhang et al., 2021b)*

Fragmentation, erosion and degradation of polymers

Fragmentation refers to the process by which the polymer splits into several pieces (Fanon, 2019) Fragmentation results from structural changes due to degradation in the material leading to cracking. and to a loss of mechanical integrity. When these cracks propagate and join, fragmentation is achieved.

Bulk erosion refers to degradation of the material (e.g., hydrolysis) that occurs within the material, leading to a reduction in polymer molecular weight and a decline in mechanical properties, although mass loss occurs later (Figure 32). The overall dimensions of the polymer stay relatively constant until fragmentation reaches a critical point. Most currently available biodegradable polymers primarily degrade through a bulk erosion mechanism. In surface erosion, degradation initiates at the outer layer of the material, with the inner material remaining intact until the surrounding area has completely eroded. It is important to note that many materials experience a mixture of both surface and bulk erosion, suggesting that these processes exist on a spectrum rather than as distinct categories.

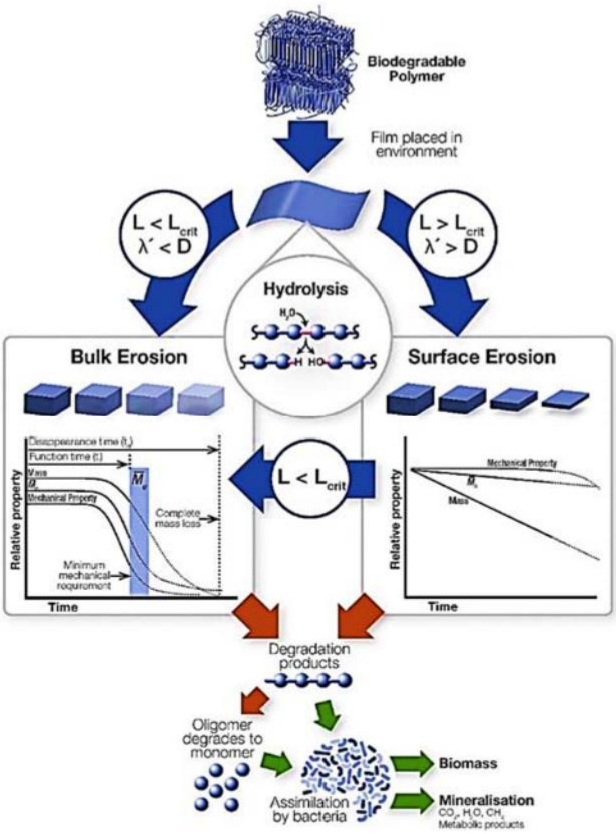

Figure 32. Steps involved in polymer degradation. (Sazali et al., 2019)

In the context of polymer science, degradation describes a complex phenomenon where a polymeric material, when subjected to environmental factors and stress, gradually loses its original characteristics (Figure 33). Although degradation is often an undesirable process, controlled degradation can be beneficial in certain situations, such as enhancing the processability of the polymer or facilitating the recycling and natural breakdown of polymer waste. In most instances, the breakdown of macromolecules plays a crucial role in polymer degradation. More specifically, this process of breaking down macromolecules into smaller fragments of varying structures and sizes is frequently identified as polymer degradation. However, when the end-products are monomers, this process is termed depolymerisation, as it is essentially the reverse of polymerisation. When the cleavage of macromolecules is not the primary mechanism, the process is typically termed polymer ageing or corrosion.

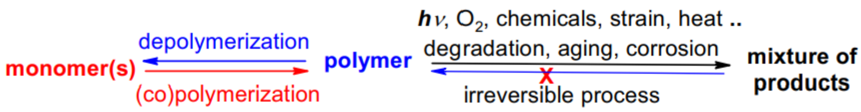

Figure 33. Relation among polymerisation, depolymerisation, and degradation. (Vohlídal, 2021)

3.3.4.4. Plastics, including the so-called biodegradable plastics, are not always biodegraded in soils, composting and anaerobic digestion facilities

Most studies on the degradation of petroleum-based plastics were carried out in the last century, whereas those on bio-based plastics dominate the last 10 years. Studies clearly confirm that conventional non-biodegradable plastics (namely PE, PP, PVC, PET and PS) do not degrade under composting and AD conditions but rather age and fragment until they are invisible to the naked eye, making MPLs that will be spread onto the soil along with the compost and digestate (3.4.1.1). The other petroleum-based polymers (PBS, PBAT, and PCL), too often claimed biodegradable, are actually efficiently biodegraded only in industrial composting conditions for PBS and PBAT (Table 15). PCL appears much more biodegradable than the two others, exhibiting good degradation in both home and industrial composting, and correct degradation in soils and under thermophilic AD conditions. Finally, while the behaviour of PLA is clear with an efficient biodegradation only at high temperature, the behaviour of the other bio-based biodegradable polymers (PHA, PHB, PHBV) like cellulose and starch-based materials, is usually good but dependent upon the environmental conditions, the material size and their blending with other polymers. Moreover, the studies done at pilot or real-scale are much less prolific than those done with lab-scale processes.

Table 15. *Expected degradation of the most studied polymers under soil, composting and AD conditions in the time required by standards (in brackets) with: Null or negligible = 0-15% degradation, Low = 15-40%, Medium = 40-60%, Good = 60-90% and OK = higher than 90% degradation.*

Polymers	Soil (6-24 months)	Mesophilic AD (60-90 days)	Thermophilic AD (60-90 days)	Home composting (12 months)	Industrial composting (6 months)
* PE, PP, PS, PET, PVC, PUR, PA	Null or negligible degradation				
** PBAT	No data	Null or negligible	Null or negligible	No data	OK
** PBS	Null or negligible	Null or negligible	Low and slow	Null or negligible	OK
** PCL	Medium to good	Low and slow	Good to OK	OK	OK
PLA	Null or negligible	Low Disintegration > degradation	Good	Null or negligible	OK
PHA, PHB, PHBV	Variable	OK	Good but inhibition problems	OK	OK
PHO	Null or negligible degradation			Low	OK
Starch-based	Usually good but depends strongly of the blend				
Mater-Bi®	Low ≤ 6 months Good ≤ 24 months	Low Disintegration > degradation	Low Disintegration > degradation	Good disintegration Variable degradation	OK
Cellulose-based	Usually good but depends strongly of the blend				

*conventional petroleum-based non-biodegradable polymers, **new petroleum-based biodegradable polymers (or said so by European Bioplastics, 2019), ***bio-based biodegradable polymers.

3.3.4.5. The poor biodegradation of plastics has consequences

The low biodegradability of most plastics leads to the presence of MPLs in compost and digestate. This is a recent subject of investigation. Seventy nine percent of the data analysed here were published between 2018 and 2023. The literature can be divided into two periods separated by the discovery of the danger posed by MNPLs to the environment and health. In the 1980s and 1990s, the plastic was considered as an inert undesired contaminant, such as stones or glass, which 'visually' degrades compost quality and soils (Brinton, 2005). However, the increasing awareness about plastic accumulation in soils and the potential danger of MNPLs for the environment and health (3.4) have led researchers to look for these particulate plastics in compost and digestate. Many publications over the last 10 years are thus focusing on this new hazard of MPLs. The detection of NPLs is to our knowledge still out of reach with the present methodologies. Literature review from 34 publications from 16 different countries shows the presence of plastic and MPLs in almost every waste, compost and digestate but with a very high variability that often span 3 to 5 orders of magnitude (Porterfield *et al.*, 2023). As such, plastic abundance in compost and digestate ranges from 310 to 82,800 particles/kg of raw compost and 2 to 38,667 particles/kg of raw digestate. In publications reporting the abundance of particles per kg of dry matter, the values range from 5 to 37,000 particles/kg of compost and 0 to 21,000 particles/kg of digestate. This high level of variability is partly due to the absence of an international standard protocol to extract and identify MPLs in organic waste and fertilisers. Many protocols start with different amounts of initial product, target different plastic sizes, and use different extraction and identification methods, each one inducing different experimental biases.

Interestingly, whatever the origin of the waste treated, the most frequent and abundant polymers detected were PE, PP, and PS, and to some extent PET, *i.e.*, the most common polymers used in food packaging (Porterfield *et al.*, 2023). The presence of biodegradable plastics in compost or digestate is cited in only three publications (Huerta-Lwanga *et al.*, 2021; Ruggero *et al.*, 2021; Porterfield *et al.*, 2023). It concerns Mater-Bi®, PBAT (as a constituent of Mater-Bi®) and PLA. The impact of the waste management process on the production or reduction of MPL concentrations is not clear. It may depend on the quantity and type of plastic present in the initial waste and the successive waste management steps. However, many studies report that composts from municipal mixed solid waste have higher levels of plastic (around 2% of the physical contaminants) than composts from waste separated at home (less than 1%) (Dimambro *et al.*, 2007).

3.3.4.6. Can we improve plastic biodegradation?

To improve plastic degradation, several strategies have been followed. Among them, the addition of additive or filler can be used to promote plastic fragmentation. However, it is not always accompanied by a better mineralisation of the material (mineralisation indicates the conversion of polymers into water, carbon dioxide, biomass and/or minerals). As an example, oxo-degradable plastics are materials to which pro-oxidants have been added to make them sensitive to abiotic oxidation and induce faster cleavage of the long polymer chains (Yousef, 2023). Historically, they have been used to produce oxo-fragmentable materials from conventional polymers (PE, PP, PS and PET). More recently, various additives were also tested in PLA materials to either improve their degradation or change their physical properties. In France, since 19 August 2015, the production, distribution, sale, making available and use of packaging or bags made, in whole or in part, from oxo-fragmentable plastics have been banned, because of the lack of conclusive evidence of a total mineralisation of the plastic polymers. By contrast, some additives are included in plastic formulations to increase their resistance to photo- and thermal-oxidation and thus reduce their biodegradability (Hahladakis *et al.*, 2018).

The blending of petroleum-based and bio-based biodegradable polymers with other plastic or natural polymers is also a practice that has various effects on the degradation of the final material, from positive, to null, or negative. Actually, only one study reported the increased degradation in soils of a PLA blended with 20% PCL. While in most other studies, in the given conditions, the blending of a non-biodegradable polymer with another biodegradable either had no effect, or only artificially increased the final material disintegration by the mineralisation of the filler alone. It should also be noted that some natural polymers, such as wood derivatives, can undergo good biodegradation under aerobic conditions but remain intact under AD. Consequently, all new material have to be tested to validate its degradation in each specific environment.

On the other hand, pre-treatments can be used to enhance plastic degradation during composting or AD, yet few studies addressed this, focusing primarily on less favourable AD conditions. The slow degradation rates of certain polymers, especially under mesophilic AD, highlight the need for effective treatments. Four main types of pre-treatments — mechanical, thermal, chemical, and biological — are commonly employed. Mechanical grinding, for example, has been shown to accelerate degradation without affecting ultimate degradation levels. Various studies have demonstrated that reducing particle size enhances degradation kinetics. Thermal treatments, particularly at high temperatures, have also proven effective, while acidic treatments appear less beneficial. Other authors have explored different combinations of pre-treatments, yielding mixed results; some treatments significantly improved biodegradation, while others showed minimal or negative impacts. Overall, grinding and thermo-alkaline treatments emerged as the most promising methods. However, the environmental impact, safety of handling chemicals, and processing time remain critical considerations for real-scale applications.

3.3.4.7. Conclusions

The biodegradation of plastics is a complex process that depends on one side on the plastic constituents and structure - including additives and other molecules - and on the other side on many environmental factors, physic, chemical and biological (microorganisms and macro-organisms). While the non-biodegradability of conventional petroleum-based plastics is acknowledged in almost all natural environments, the biodegradability of the so-called biodegradable plastics is not systematic in all environments. It depends on each formulation and should be carefully tested.

Numerous standards and labels regulate the assessment of the biodegradability of plastics in natural and industrial environments. In the scientific literature, they have been reviewed by many authors (Briassoulis *et al.*, 2010a; 2010b; Ruggero *et al.*, 2019; Folino *et al.*, 2020; Cazaudehore *et al.*, 2022; Pires *et al.*, 2022). It is admitted that the multitude of highly technical normative references makes it difficult to identify the requirements and therefore the guarantees associated with the 'biodegradable plastics' designation, for both producers and users. Moreover, the multiplicity of standards and private labels, as well as natural, compostable and biodegradable labelling voluntarily (green-washing) or involuntarily confuses the consumers. Nevertheless, despite this profusion of standards, there is still no clear specification standard to define the expectations required for the full biodegradation of plastics in AD processes. In fact, each plastic material should advertise clearly its composition and end-of-life disposal. An alternate possibility would be to produce materials fully biodegradable in the less efficient conditions, which is not the case at this time.

The quality of organic fertilisers (compost or digestate) depends on the quality of the incoming waste. The will to valorise the organic fraction of urban waste and packaged food waste in composting and AD facilities, strongly promoted by current policies (*i.e.*, Bioeconomy strategies, Carbon storage in soils, Ban of landfilling and incineration of organic waste), has increased the contamination of composts and digestates with non-biodegradable MPLs of fossil origin (Kawecki *et al.*, 2021; Sholokhova *et al.*, 2022a; ADEME *et al.*, 2023; Porterfield *et al.*, 2023). A real evaluation of the benefit-risk of such practices for the environment and health should be considered.

3.4. Plastics are ubiquitous, hazardous and have multi-scale adverse impacts on living organisms, humans and continental ecosystems

3.4.1. Plastics have the ability of fragmentation and dispersion in the environment, living organisms and along food chains

(based on Chapters IV.1, IV.2 and IV.3 of the extended report)

3.4.1.1. Soil is the poor relation when it comes to studying plastic pollution in the environment, and yet...

Soils are contaminated by particulate plastics that breakdown from macro- to micro- and nano-plastics

In soils, plastics exist as a continuum of sizes, from MaPLs to NPLs, including MPLs. Research on MPLs in soils is much more extensive than that on MaPLs. This observation aligns with what is known for other environmental matrices such as fresh- and marine waters. The fragmentation process, especially in natural environments, is poorly understood and documented. However, as a first piece of evidence, Berenstein *et al.* (2024) have shown that effective and rapid fragmentation processes reduce the size of plastic fragments in agricultural soils over time.

The literature on soil contamination by plastics is recent, following work on the marine and freshwater environment and the atmosphere. All terrestrial surfaces are contaminated by plastics in their particulate form, including the most remote areas from human activity such as the Himalayan peaks - though they are affected by tourism - (MPLs; Napper *et al.* (2020)), deserts (MPLs; Abbasi *et al.* (2021)), primary and secondary tropical forests (MPLs; Xu *et al.* (2022)) and remote conservation areas (MaPLs; Cyvin *et al.* (2021)). For example, the contamination of soils in remote desert areas has been estimated to be 10^2 MPLs/$kg_{dw\ (dry\ weight)}$. Such ubiquity shows the high ability of plastics to disperse in the environment.

Agricultural soils are heavily contaminated by MPLs

First results of plastic contamination on samples taken from different types of soil were published in 2018, as highlighted by Büks and Kaupenjohann (2020) in their review. Since then, three main types of soil have been studied: (i) reference soils (*i.e.*, with only background atmospheric exposure), often desert or forest soils; (ii) non-agricultural, urban or industrial soils (associated to intense human activity in the surrounding area); and (iii) agricultural soils. The latter are the most studied for MPL contamination, mainly in China (60% of publications), followed by Germany (7% of publications); France is in third place, with 6% of publications. Agricultural soils studied are very diverse, including market gardening, arable crops, plantations (bananas, rubber) and grasslands soils.

MPL abundances are expressed as the total number or total mass of MPLs within a given size range. For analytical reasons, the majority of studies use the first option. Abundances are then reported, either relative to a mass of soil (dry weight (dw)) or relative to an area of soil. In line with its widespread use in agriculture (3.1), PE appears to be the most abundant polymer in terms of number of particles per unit mass (proportions between 60 and 80%), followed by PP in all available studies worldwide.

A preliminary estimate of global soil contamination with MPLs in agricultural soils from different geographical origins (however, 84% of the data come from analyses of Chinese soil samples) was made in 2023 (Kedzierski *et al.*, 2023). It is estimated to be between 1.5 and 6.6 Mt in the top metre of soil, which is one to two orders of magnitude greater than the MPL stock at the ocean surface (290 to 800 kt according to the most recent estimates; Lebreton *et al.* (2019). This should therefore be considered as a matter of serious concern. It should be noted that this estimate only considers contamination associated with agricultural mulching and sewage sludge or wastewater application. Another estimate was made for French soils, especially in the context of agricultural soils (Palazot *et al.*, 2024). An average contamination of 244 kg MPLs per ha was estimated for the top 20 cm of soil. Based on the world's agricultural land area, this was extrapolated to the global scale and an estimate of 664 Mt of MPLs worldwide was arrived at in Chapter IV.1 of the extended report - an estimate 100 times larger than the previous one. The difference of two orders of magnitude between these two estimates is likely to reflect uncertainties in sampling, analysis and methodology, as discussed in section I.4.4.2.

With these uncertainties in mind, estimates of soil contamination for the different soil types were proposed in Chapter IV.1 of the extended report. This corresponds to 10^2 MPLs/kg_{dw}, 10^3 MPLs/kg_{dw} and 10^4 MPLs/kg_{dw} for background contamination in remote deserts, median contamination in agricultural soils and contamination in non-agricultural soils, respectively (urban or industrial, European context). However, these are rough estimates and there may be considerable variability between sites within each category.

Plastic inputs and infrastructures used for crop and livestock production, organic residual products and irrigation are direct human sources of micro-plastics in agricultural soils

Soil contamination can result from the direct addition of plastics by human activities. For agricultural soils, there are other sources of contamination that combine agricultural plastic inputs and infrastructures, including agricultural mulches, application of organic residual products (ORPs, *e.g.*, compost and sewage sludge) and irrigation (with surface water, untreated or treated wastewater).

Plastics used in agriculture, including intentionally-added MPLs, contribute to soil contamination by MPLs. Considering the different agricultural practices (*e.g.,* mulching but also greenhouse cultivation) described in Chapter II.1 of the extended report, the contribution of mulching to plastic soil contamination in Europe is lower than in China, the country where plastic mulch is most widely used. In fact, the quantities of plastics used in China are much higher and the widespread use of thin and transparent films makes their removal from the soil very difficult. Globally, the contribution of plastic mulching to soil contamination by MPLs has been estimated to be between 0.5 and 2.3 Mt (Kedzierski *et al.*, 2023).

In Europe, approximately 750 kt of plastics are used annually in agriculture, of which 23.5 kt (ECHA (2019), Tab. 15 p.74-75) are intentionally-added MPLs (*e.g.*, controlled-release fertilisers, fertiliser additives, coated seeds, capsule suspension PPPs (or controlled-release PPPs). It is important to note that the use of these MPLs is expected to decrease following the entry into force of Regulation (EU) 2023/2055 of 25 September (2023), which bans the marketing of products containing MPLs for agricultural and horticultural uses from 17 October 2028. MPLs are completely transferred to the soil matrix. Even in the case of collected agricultural plastics (*e.g.*, by ADIVALOR in France), transfer to the soil matrix still occurs. This is due to carelessness or the formation of MPLs during use. Few studies have focused exclusively on soil contamination from greenhouses. In fact, in many cases, greenhouse crops are also grown under mulch, and the contamination observed is mainly the result of both greenhouse and mulch elements, making it difficult to distinguish between the two sources.

Other sources of soil contamination by MPLs are ORPs, especially compost, and irrigation water. In France, according to ADEME (Mortas *et al.*, 2023), the main source of MPL contamination in agricultural soils (high-end value) is by far compost produced by mechanical-biological treatment (mean value of 126.21 kg MPLs or 1150 10^6 MPLs per ha and per year). This is in line with soil contamination from compost applications (Colombini *et al.*, 2022). This exceeds the values obtained for composts and digestates from kitchen and table waste (mean value of 9.66 kg MPLs or 72.4 10^6 MPLs per ha and per year, and 13.79 kg MPLs or 112 10^6 MPLs per ha and per year, for composts and digestates, respectively). In comparison, the flux from sewage sludge (0.29 kg MPLs or 3.36 10^6 MPLs per ha and per year) appears relatively low. For bio-waste, standards NFU 44-051 on organic soil improvers and NFU 44-095 on sludge composts set maximum permissible size and percentage criteria for plastics. An organic amendment must not contain more than 0.3% by mass of film and EPS over 5 mm, and 0.8% of other plastics over 5 mm (AFNOR, 2006). At present, there is no standard for MPLs in France (Chapter V.1 of the extended report).

Data were collected from the literature to assess the MPL contamination levels for irrigation water. These range from [1-1000] 10^6 MPLs per ha and per year for untreated wastewater irrigation, and from [0.01-100] 10^6 MPLs per ha and per year for treated wastewater irrigation. The treatment process results in a reduction in MPLs of between 90% and 99%. Groundwater irrigation, unlike surface water irrigation, cannot currently be considered as a source of MPL contamination for agricultural soils. For surface water irrigation, the contamination level is significantly lower compared to untreated wastewater (by a factor of 10^6) and treated wastewater (by a factor of 10^4 to 10^5). The contamination is therefore [0.001-100] 10^2 MPLs per ha and per year.

Plastic pollution does not stop at borders, it travels through the air: atmospheric deposition results in high MPL contamination levels of agricultural soils

In addition to direct contamination by human activities, soil contamination can result from the transfer of plastic particles between environmental compartments. Exchanges from the atmosphere to soils (*e.g.*, atmospheric deposition), from continental waters to soils (*e.g.*, inputs from river floods), but also from soils to the atmosphere and from soils to surface water or groundwater, are currently poorly understood. From the current state of the literature, it is not possible to quantify and distinguish stocks and fluxes of particulate plastics from plastics used in agriculture and for food, and other plastics within and between the different environmental compartments.

Atmospheric fallout of MPLs is an indirect source of contamination. The dynamics of atmospheric transport of MPLs remain to be characterised and quantified. Larger particles would tend to be redeposited close to their emission source, while smaller particles (<100 µm) may be transported over longer distances. Reported orders of magnitude for deposition rates of MPLs range from 10 to 1000 MPLs per m^2 and per day (Beaurepaire *et al.*, 2021), corresponding to 0.36 to 3.6 10^9 MPLs per ha and per year. This upper limit is surprisingly much higher than for other sources of contamination and should be investigated further.

Figure 34 depicts the processes that potentially affect the amount of particulate plastics in soil systems, including sources and fate processes.

In addition to particulate plastics, agricultural soils are contaminated by plastic-related compounds, at least plastic intentionally added substances known to pose a health risk to humans

In agroecosystems, IAS play a crucial role in determining environmental risks (3.4.2.1). Typically, IAS can leach from the polymer surface under certain conditions through fragmentation and degradation. The leaching process depends on a number of factors such as the type of plastic product, the chemical structure and properties of IAS, and environmental factors. In soils, the behaviour of IAS is mainly determined by complex dynamic physical, chemical and biological processes, including adsorption and desorption, volatilisation, leaching, uptake by plants or animals and runoff (Cao *et al.*, 2023). These processes determine the distribution of IAS in the soil, the transfer from the soil to water and air and the eventual uptake into the human food chain.

Cao *et al.* (2023) investigated the presence of IAS in agricultural soils, including plasticisers, antioxidants and stabilisers. Among investigated IAS, phthalates, Bisphenol A (BPA), nonylphenol (NP) and polybrominated diphenyl ethers (PBDEs) were the most commonly detected. Overall, phthalates had the highest concentration in agricultural soils, with an average concentration of 1523 µg/kg, followed by NP, BPA and PBDEs with concentrations of 65.30, 15.87 and 12.24 µg/kg respectively. Among the typical phthalates, Di(2-ethylhexyl) phthalate (DEHP) had the highest average concentration, *i.e.*, 670.80 µg/kg. For France, the level of BPA contamination was reported in one study. A BPA concentration of 0.42 µg/kg was found in an agricultural soil in the Paris area (near Fontenay-les-Briis, Essonne). This compares with 4-14 µg/kg in American farmland, 0.40-99.89 µg/kg in farmland near rivers in various regions of India, and 1.1-17.9 µg/kg in several Spanish farmlands (a higher value of 55.9 µg/kg was reported for a soil sample irrigated with effluent from a wastewater treatment plant).

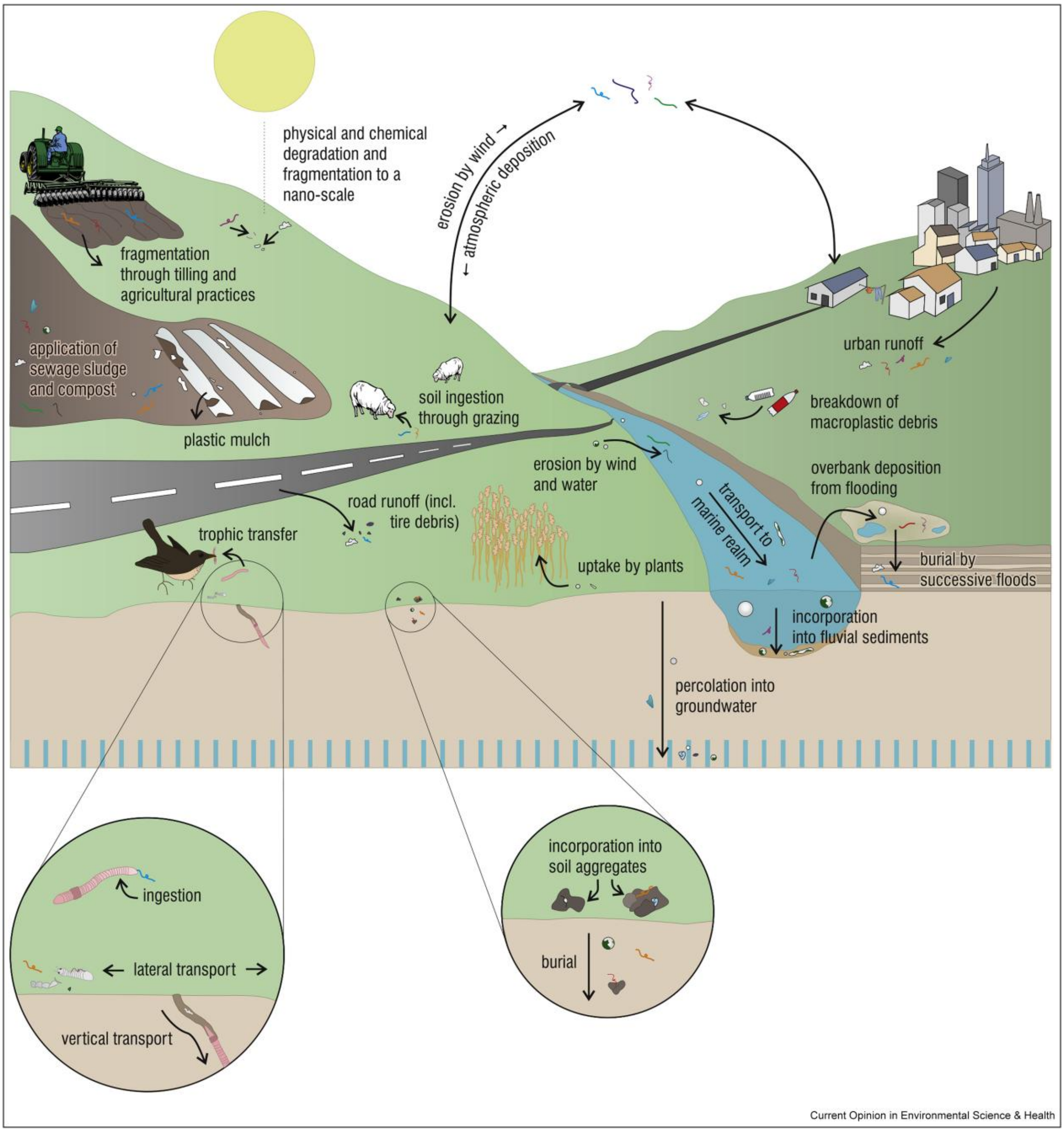

Figure 34. *Sources and fate processes of particulate plastics in soil systems.*
The sources include inputs from agricultural practices, the influence of runoff and deposition, and the fragmentation of larger plastic debris. (Hurley et Nizzetto, 2018)

3.4.1.2. Food and drinks are contaminated by plastics

The presence of plastics in food and drinks contributes to their dispersion through the food chain as plastic-related compounds and/or particulate plastics. It should be noted that of the large number of plastic-related compounds potentially used or unintentionally present in plastics, only a few are currently covered by regulation, which does not fully reflect the pace of scientific knowledge.

A huge number of plastic-related compounds, including unregulated ones, find their way from food contact plastics into food and drinks

There are a large number of plastic-related compounds that are present in FCMs. In fact, more than 16,000 chemical substances are potentially used or unintentionally present in plastic materials and products. More than a quarter of these are known to be hazardous because they are persistent, bioaccumulative, mobile and/or toxic, and hazard data are lacking for 66% of them (Wagner *et al.*, 2024). Only 6% are currently covered by multilateral environmental agreements (Stockholm Convention, Basel Convention, Minamata Convention and Montreal Protocol, Chapter V.1 of the extended report).

For FCMs, of which plastics are the most important, the number of chemical substances present and potentially released can exceed 10,000. Only a few have been evaluated for their potential impact on human health 3.4.3.1). The existing regulation on FCMs has established a positive list of certain families of chemical substances that can be used in the formulation of food contact plastics (Chapter V.1 of the extended report). It should be noted that ageing of FCMs over time is not currently a criterion considered by the regulation. The Food Packaging Forum[12] has published several databases containing all the substances (Food Contact Chemicals database (n.d.-b) – FCCdb, version 5 (Groh *et al.*, 2020))[13], the migration compounds (FCCmigex database (n.d.-a))[14], the toxicity level and all relevant information dealing with the substances and their dispersion.

Migration from plastic materials that come into contact with food involves a variety of plastic materials, food simulants and plastic-related compounds. Different types of food contact plastics, including single use (*e.g.*, bags, flexible food wraps, trays and bottles) and multiple use (*e.g.*, houseware, kitchenware, and tableware utensils) are being investigated. The term 'migration' in this context refers to the transfer of compounds, IAS or NIAS, to the food or drinks in contact with plastics. The migration process takes place either in the vapour phase or in direct contact with the food and it is influenced by the experimental conditions applied to the plastic-food pair, including the type of plastic, the characteristics of the food and drinks (*e.g.*, pH, fat content), temperature and contact time.

For plastic materials, PET, PVC, and polyolefins such as PP and PE (LDPE, HDPE), and PS, are the most commonly investigated food contact plastics in the literature (19%, 12%, 12%, 11% and 7% of published studies, respectively) as potentially releasing plastic-related compounds. However, other materials containing plastics in their composition are also considered (*e.g.*, paper and paperboard coated with epoxy or acrylic coatings). Studies mainly address migration from conventional petroleum-based plastics and less on bio-based plastics (88% as compared to 12% of reported studies).

Another point to mention is the matrices used for such migration studies. The EU legislation (2011) has proposed food simulants (10%, 20% and 95% ethanol (instead of edible oil), 3% acetic acid, Tenax® that is a trademark of poly(2,6-diphenyl-p-phenylene oxide)) to mimic different foods and beverages. The migration values can then be compared between laboratories and used to certify compliance. As a result, migration analysis was mainly performed with these simulants (56% of published studies) and only in a few cases (16% of reported studies) real foods (*e.g.*, cakes, bread, cereals, rice and pasta) or beverages (*e.g.*, milk, cola-type drinks) were used (Nerin *et al.*, 2012). Some studies compared the migration

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[12] https://foodpackagingforum.org/
[13] https://zenodo.org/records/4296944
[14] https://foodpackagingforum.org/resources/databases/fccmigex

capacity using simulants and real food (Aznar *et al.*, 2011; Otoukesh *et al.*, 2020; Canellas *et al.*, 2021; Vera *et al.*, 2023). They showed that simulants maximised migration values when used in liquid form, whereas when Tenax® was considered as a solid simulant, the final values were not so different (Vera *et al.*, 2018).

Plastic-related compounds may or may not be volatile. Volatile compounds (*e.g.*, solvents, monomers and other low molecular weight compounds) can migrate from plastic packaging into food, especially at high temperatures. Their volatility also means that they can contribute to off-flavours or off-tastes in food. Due to their intrinsic nature, non-volatile compounds (*e.g.*, plasticisers, antioxidants and oligomers) may have a greater likelihood of migrating through direct contact with food and beverages, especially fatty food. Nanoparticles can also be used in plastics to improve mechanical properties or as UV blockers, antimicrobials or antioxidants. Their small size allows them to migrate easily into food (Paidari *et al.*, 2021). Finally, metals and metalloids (*e.g.*, tin, lead, cadmium (Cd) and antimony), which can be incorporated into plastics as catalysts, stabilisers or colourants, have a migration capacity, especially under acidic conditions or at high temperatures.

The migration of phthalates, antioxidants, bisphenols and oligomers has been the main focus of attention in plastic food packaging. Phthalates, antioxidants (*e.g.*, butylated hydroxytoluene (BHT) and butylated hydroxyanisole (BHA)), oligomers and bisphenols are the most studied plastic-related compounds in food packaging studies. Among the bisphenols (BPA but also BPB, BPC, BPF, BPM, BPP, BPS, BPZ, BPAF, BPFL and BPC12), BPA is undoubtedly the most studied one because of the public debate on it, the risk it poses to human health, and also because bisphenols have become the subject of regulation at European level (Chapter V.1 of the extended report). In France, the law suspended 'the importation and placing on the market, whether free of charge or against payment, of any packaging, container or utensil containing BPA and intended to come into direct contact with any food as from 1 January 2015'. In December 2024, the EC adopted a ban on the use of BPA in FCMs.

Oligomers are released from all plastics and it is worth noting that not only conventional polymers (*e.g.*, PET, PA6 and PS) but also bio-based materials such as PLA are capable of releasing such chemicals (Gavril *et al.*, 2019; Ubeda *et al.*, 2019). Over the last decade, special efforts have also been made to identify and evaluate the migration capacity of NIAS from FCMs. They represent a broad class of chemicals, including impurities from plastic ingredients, degradation products and contaminants. Unlike IAS, which are theoretically known for FCMs and can therefore be assessed, NIAS are often complex and difficult to assess. Among the NIAS identified so far, primary aromatic amines (PAAs) as degradation products of PUR adhesives and azo dyes in printing inks are a group of particular interest. They have been reported to migrate from, *e.g.*, PP packaging (Kolado et Balcerzak, 2009). Other migrating compounds, including NPs and photoinitiators (considered as NIAS and IAS, respectively), fluorescent whitening agents and stabilisers, have been studied. UV absorbers (*e.g.*, Cyasorb®, Chimassorb®, Tinuvin®) are also potential migrating compounds, especially under direct sunlight. Finally, styrene can migrate into food (Gelbke *et al.*, 2019), especially in the presence of fatty substances or at high temperatures when PS starts to degrade.

For plastic tableware and kitchenware utensils, heating or contact with acidic or fatty foods increases the migration of chemicals into food. Research on tableware and kitchenware utensils focuses on items used daily for food preparation and consumption, and of particular concern are materials that can release harmful compounds when heated (*e.g.*, PC containers, melamine bowls). In some cases, they contain additives not authorised in Europe (*e.g.*, bamboo powder) or low quality plastics (*e.g.*, melamine).

The migration of melamine derivatives from plastic utensils and formaldehyde from melamine tableware has been assessed (Poovarodom *et al.*, 2014), as these reusable tableware items often contain bamboo powder as a load in the plastic item (Wrona *et al.*, 2023). Although they are advertised as bamboo utensils, they are still melamine-based and contribute to the dispersion of plastics. As mentioned above, the use of bamboo powder is not authorised by EU regulation 10/2011 (2011). A note from the European Commission (EC) has been circulated to all Member States in 2020 stating that bamboo or other such plant fibres are not wood fibres (that are authorised by the regulation for use in plastic FCMs) and, like any additive not listed in the Annex to the regulation on plastic FCMs, cannot be added to plastics for use as FCMs. In addition, the study of additives in kitchenware is important because they can migrate into food, especially under certain conditions such as heating or contact with acidic or fatty foods. For example, acetyl tributyl citrate (ATBC) and amines have the potential to migrate into food. As with food packaging, NIAS and oligomers have also been detected and quantified as migrating compounds (Canellas *et al.*, 2021; Asensio *et al.*, 2022; Wrona *et al.*, 2023).

Particulate plastics including MPLs, NPLs and possibly oligomer submicrometre particles are additional food and drink contaminants

The potential contamination of food and beverages by MPLs is a significant and growing concern. Data on NPLs are less abundant (but more recent), accounting for about 10% of the total number of scientific publications.

Converging experimental evidence is consistent with the presence of MPLs in a wide range of foods, including meat, honey, sugar, salt, fruit, vegetables, processed foods and beverages such as drinking water, tea, beer and milk. Among foods, most studies to date have focused on seafood. While MPLs have been found in the muscles of commercially valuable fish and crustaceans, the concentration of MPLs is generally higher in their digestive tracts, which are usually removed before consumption. Therefore, the presence of plastics in the gastro-intestinal tract of fish does not necessarily indicate direct human exposure. On the contrary, juvenile fish, dried or salted fish, crustaceans and bivalves (*e.g.*, oysters, mussels) are usually consumed whole, with their digestive tract (EFSA Panel on Contaminants in the Food Chain, 2016; Dawson *et al.*, 2021), and may pose a higher risk to human health. With regard to beverages, numerous studies have investigated MPL contamination in drinking water treatment plants, tap water and bottled water from different locations around the world. The reported values vary by several orders of magnitude (from 0 to 5.42 10^7 MPLs/L), probably due to differences in water sources, geographical locations, quantification methods and possibly inadequate analytical procedures or contamination precautions (3.4.4.2).

In order to understand the origin and contamination pathways of particulate plastics in edible products, it is important to trace their potential sources. Typically, foods and beverages can be contaminated via the food chain, the environment (3.4.1.1), farming, fishing and aquaculture practices, processing (trays, filtration processes, manufacturing, air deposition, clothing worn by operators, etc.), transport, storage, but also food preparation, including cooking methods, choice of kitchen utensils (Habib *et al.*, 2022; Yadav *et al.*, 2023), and contamination by airborne particles in indoor or outdoor environments (Dris *et al.*, 2017; Catarino *et al.*, 2018).

With regard to packaging, products such as beverages, meat, yoghurt, take-away food and rice have been described as containing MNPLs from packaging materials when subjected to mechanical stress (*e.g.*, opening and closing of bottle caps; Winkler *et al.* (2019) or thermal stress, *i.e.*, heat, or freezing). However, the chemical composition of nanoparticles released from polyester textiles has recently been the subject of debate (Yang *et al.*, 2024). The possibility has been raised that the so-called NPLs, as submicron particles, may actually consist of oligomers, as has been suggested for tea bags (Busse *et al.*, 2020). The presence of oligomers is rarely reported in the current literature in this field, but all plastics release oligomers as mentioned above. It is important to note that although Fourier Transform IR Spectroscopy (FTIR) and Raman spectroscopies, which are classically used for the chemical identification of MNPLs (3.4.4.2), detect functional groups (*i.e.*, the main bonds in molecules). These functional groups are the same in oligomers as in the polymers from which they are derived. In the case of PE and PET, ethanol pre-treatment of samples has been recommended to differentiate between NPLs and oligomer aggregates, as oligomers are soluble in ethanol, unlike NPLs (Li *et al.*, 2022; Yang *et al.*, 2024). However, the applicability of this method to other polymers requires further investigation. Therefore, there is a critical need to delineate the boundary between oligomers and NPLs as one of the bases for proper human health risk assessment.

3.4.1.3. Biota are contaminated by plastics in their various forms and contribute to their dispersion

Living organisms studied to date in the literature, *i.e.*, soil invertebrate organisms (*e.g.*, earthworms, nematodes), soil microorganisms, terrestrial plants, wild terrestrial vertebrates (mammals and non-mammals), livestock animals and poultry, as well as humans, are contaminated by plastics of different sizes (MaPLs, MPLs and/or NPLs, according to the target organism; Figure 35 for MPLs) and plastic-related compounds (in particular IAS) and contribute to their dispersion.

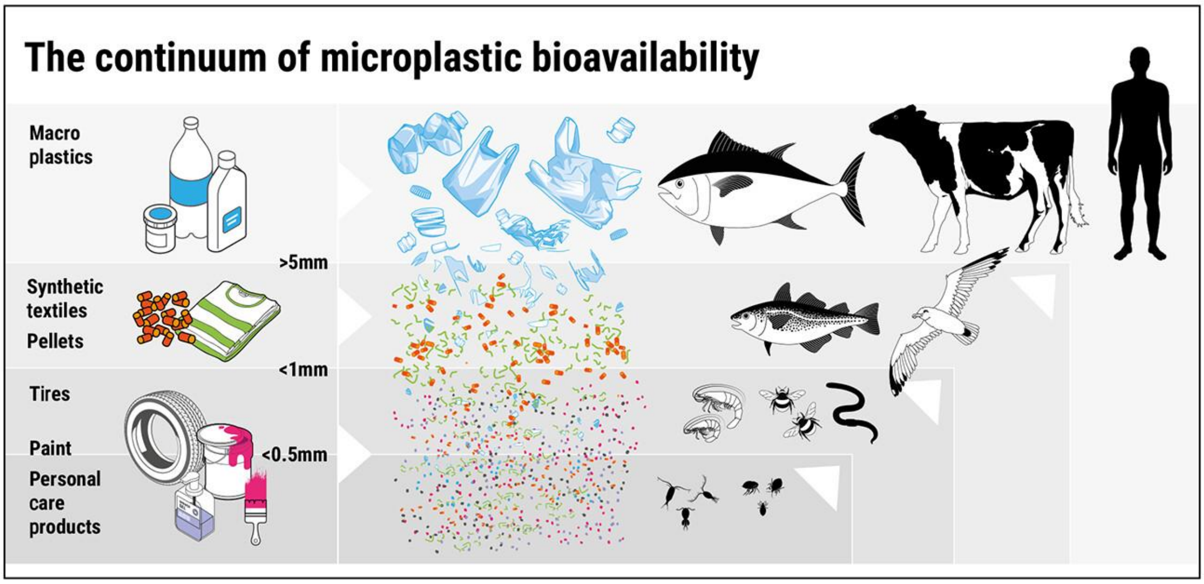

Figure 35. *The continuum of MPL bioavailability in living organisms. As plastics fragment into smaller pieces, they become available to a wider range of organisms (descending horizontal lines) and the potential for transfer along food chains also increases (diagonal arrows). (Napper et al., 2020)*

Soil (micro)organisms and terrestrial plants are exposed to plastics

As reported above (3.4.1.1), plastic contamination is generally present in all soil types, leading to potential changes in soil physical and chemical properties, and affecting soil invertebrates, microorganisms, and terrestrial plants.

Studies on invertebrate organisms are limited to earthworms. Earthworms are the most widely studied soil organisms for exposure to MPLs or NPLs. Most earthworms have a mouth size of about 3 mm, and so MPLs smaller than 3 mm and NPLs can be ingested. These animals can be grouped according to their burrowing behaviour (epigeic, anecic, and endogeic) and feeding behaviour (detritivores and geophages). These characteristics highlight earthworms as key players in the biogenic transport of MNPLs into the soil (Ju *et al.*, 2023) and in their fragmentation (Meng *et al.*, 2023). Nematodes are also considered as model organisms due to their widespread distribution in soils and their key roles in biological processes (*e.g.*, plant-, bacterial-, fungal-feeders, carnivorous and omnivorous feeders). For example, in a laboratory-scale study, MPLs (PS) were found throughout the digestive tract, lumen, pharynx, gut lumen, and rectum of *Caenorhabditis elegans* (Lei *et al.*, 2018). Due to their abundance, ecological functions, and their role in nutrient cycling and decomposition, collembolans are another group of soil organisms considered in the literature, although less so than earthworms and nematodes. Interestingly, these microarthropods may contribute, at least to some extent, to the dispersal and transport of MPLs through the soil profile, as MPLs can attach to their cuticle (Maass *et al.*, 2017). Other soil organisms (*e.g.*, snails, protists) may bioaccumulate MPLs, although knowledge in the field is clearly in its infancy.

Focusing on soil microorganisms introduces the concept of 'plastisphere'. The soil microbiome plays an important role in all soil functions. Most articles investigating the soil microbiome consider bacteria, fungi, protists and viruses, with bacteria being by far the most studied. The soil microbiome is a complex community of interdependent organisms, which is influenced by many soil properties. In addition, the soil properties are very heterogeneous and depend on many different factors. Therefore, the soil microbiome is most often described in terms of key drivers and/or key properties of interest. The plastisphere is the plastic surface and/or soil under the influence of plastics. The rhizosphere is the soil under the influence of roots. The aggregatosphere is the soil aggregates as the main building blocks of soil structure. The drilosphere is the soil under the influence of earthworms. The detritusphere is the soil under the influence of dead organic matter. The gut microbiome of soil organisms is also part of the soil microbiome, in the gut conditions of the specific organism. There is still an open definition of the plastisphere: from including only those organisms that remain strictly on the plastic surface, to organisms living in the surrounding soil under the influence of plastic contamination. As it is more widely accepted, the latter was used in the report (Rillig *et al.*, 2024). More details on the colonisation of plastic debris by microorganisms and plastisphere microbial communities are given in section 3.4.2.1.

Terrestrial plants are contaminated by plastics to varying degrees depending on their characteristics, particularly their size. MaPLs and some fibrous MPLs have been shown to have the potential to entangle plant roots and inhibit their development (Elbasiouny *et al.*, 2022). In addition, MPLs can physically block the pores in the seed capsule, preventing the seeds from absorbing water. In contrast, NPLs can penetrate the seed coat and induce up-regulation of water channel gene expression by creating small pores. Similarly, plastics can penetrate the plant tissues to varying degrees depending on their size. For example, plants are not expected to be able to take up MPLs because their size prevents them from penetrating the cellulose-rich plant cell walls. However, if MPLs degrade to NPLs, they can cross biological membranes and enter plant cells (Li *et al.*, 2020). Plastic residues (*e.g.*, NPLs and oligomers) have a tendency to enter root cell walls and be transported to the shoot. This has been demonstrated for NPLs (PS, 200 nm) taken up by wheat and lettuce roots. NPLs may therefore enter the wider food chain by ending up in parts of plants intended for human or animal consumption. In addition, plastic-related coumpounds such as BPA, NP and phthalates can end up in wheat grains (Wang *et al.*, 2018; Shi *et al.*, 2019).

Wild vertebrates are contaminated by plastic waste in their habitats

The contamination of wild terrestrial mammals by particulate plastics and/or plastic-related compounds has been poorly documented in the literature to date. Nevertheless, published results converge on the habitat contamination by plastic waste as a major source of exposure. For example, plastic ingestion has been reported in the semi-aquatic coypu (*Myocastor coypus*; Meyer *et al.* (2023) and wild brown-nosed coatis (*Nasua nasua*; Rodrigues *et al.* (2021)). MPLs (composed of PUR, PA, PET, PS, PE, and PP as well as polyethoxylated tallow amine residues) have been found in the intestines and faeces of free-ranging hares (*Lepus europaeus*) and housed hares (Hornek-Gausterer *et al.*, 2021). MPLs (polyester, polynorbornene, PE) have also been found to accumulate in the faeces of terrestrial wild mammals such as European hedgehogs, wood mice, field voles and brown rats (Thrift *et al.*, 2022; Liu *et al.*, 2023b). Less is known about non-mammalian terrestrial vertebrates (*i.e.*, fish, amphibians, reptiles and birds). The uptake of MPLs by freshwater fish is much less studied than for their marine counterparts. However, fish have been shown to ingest and bioaccumulate MPLs in lotic, lentic and controlled environments. Other existing studies have focused on turtles and tadpoles, and less on snakes, lizards and crocodiles (Hou et Rao, 2022). For example, turtles can become entangled in plastics and ingest large pieces of plastics such as plastic bottles, bags or straws, which can result in physical injury, including suffocation or organ damage. Well-known examples from the marine environment include turtles with a blocked nasopharynx, which increases breathing difficulties, and turtles entangled in plastic nets, which reduces their mobility. However, similar to fish, plastic ingestion by freshwater turtles is relatively understudied (Clause *et al.*, 2021) compared to their marine counterparts. Like turtles, snakes have been documented to become entangled in plastic bags and to ingest plastic bottles, bags and bottle caps, which can be fatal. Finally, ingestion and contamination of birds by particulate plastics has been documented. For example, ingestion of MPLs by grey heron (*Ardea cinerea*) has been reported in Lake Geneva (Switzerland) (Wang *et al.*, 2019). Among raptors, MPL contamination (with PET, polyacrylonitrile (PAN) and PA as the most abundant polymers) has been found in barn owl (*Tyto alba*) pellets (Nessi *et al.*, 2022). Importantly, mobile organisms such as fish, mammals and birds may contribute to the long-range dispersal of MPLs following ingestion and subsequent egestion.

Livestock animals and poultry are exposed to plastics via ingestion of contaminated feed and water, and inhalation

The routes of exposure to particulate plastics in livestock and poultry production systems have been identified, and converging literature has reported the same routes of entry for MPLs and NPLs into the animals. These routes include: (i) ingestion of feed, including through grazing on contaminated farmland, conserved forage, and other feedstuff such as grains contaminated with, *e.g.*, MPLs in storage tanks (especially in large-scale facilities) or in feedbags; (ii) ingestion of water from contaminated surface water and groundwater and from plastic-contaminated containers or taps; and (iii) exposure through air (Ramachandraiah *et al.*, 2022). As described above, NPLs can be taken up by plant roots and transported to edible parts of plants and thus potentially be present in forage and crops intended for animal consumption. With regard to dermal exposure, it has been shown that, in the barn, MPLs in the air or in the water during cooling by spraying can come into contact with the skin of the animals. The abundance of fur on the surface of the animal's body protects the skin and reduces the accumulation of MPLs on the skin surface (Dong *et al.*, 2023). The routes of exposure and their importance differ depending on whether the livestock production system is grazing, mixed or industrial, with animals in grazing and mixed systems having access to farmland for feeding and drinking. Priyanka and Dey (2018) reported that in developing countries, particularly in urban areas, domestic ruminants (cattle followed by buffaloes, sheep and goats) that are allowed to graze freely ingest plastic waste that accumulates in open areas. In addition to exposure through feed and water, airborne exposure should not be neglected for terrestrial animals, especially in livestock and poultry buildings, where MPLs and NPLs from dust, feed, animal manure, bags and ageing equipment can accumulate in the air if ventilation is inadequate. Inhaled NPLs can be absorbed and accumulate in the lungs, while most larger plastic particles are coughed up and can enter the digestive tract if swallowed (Dong *et al.*, 2023). Overall, given these routes of exposure, grazing or outdoor animals are likely to be more contaminated than others.

For domestic ruminants, MPL intake has been estimated to be between 3 and 677 mg per week (Urli *et al.*, 2023). Once entered in the mouth, due to their small size and chemical stability, MPLs are unaffected by dental chewing and saliva and therefore easily reach the stomach. The fate and absorption of MPLs in the digestive tract of ruminants may differ significantly from that of monogastric animals due to the presence of the rumen (in these monogastric animals, effects of MPL exposure, *e.g.*, transfer from feed to muscle, may be similar to those in rodents; 3.4.3.1). In the rumen, the unique anaerobic environment and microbiota structure allows microorganisms to be potential degraders of MPLs. MPLs have been observed in raw milk obtained from a milking machine at levels ranging from 204 to 625 particles per 100 mL (Da Costa *et al.*, 2021). PE, the main polymer present in the milking machine, was probably the main contributor to MPL contamination of raw milk. The presence of other polymers such as PP, poly(ethylene succinate) (PES) and PTFE in raw milk may be explained by their ubiquitous occurrence in the farm environment and along the milking process including storage containers. MPLs have also been found in the faeces of sheep, dairy cows, pigs and poultry (Dong *et al.*, 2023). In chickens, the presence of MPLs (PVC, LDPE, PS and polypropylene homopolymer (PPH)) has been assessed in the crop (pouch-like structure at the beginning of the digestive system) and gizzard of animals from different farms in Pakistan (Bilal *et al.*, 2023). MPL contamination was also detected in skeletal muscle (PS and PA). Very little data is available on NPLs.

For plastic-related compounds, diet (fresh and conserved grass and processed food through migration from packaging), drinking surface water and contaminated soils were identified as exposure routes. For example, the daily intake via surface water and pasture was estimated to be 119 and 21.6 µg/day for BPA and 4465 and 786 µg/day for phthalates for cattle and sheep, respectively. In addition, bisphenols were found in plastic feed packaging (PP-based woven bags and PE-based films; Wang *et al.* (2021b)). BPA was the predominant analogue with a wide range of concentrations in both PP- and PE-based packaging. These compounds may then migrate into the solid feed of ruminants. BPA was found in cattle urine (229-305 ng/L) (Zhang *et al.*, 2014). For pigs, BPA levels were assessed in fresh pork meat collected immediately after death from animals raised under standard conditions. The mean concentration of BPA in loin meat was 13.77 µg/kg$_{dw}$ of meat (Makowska *et al.*, 2022), suggesting that pork meat may be a source of BPA in human food. Another study extended the analysis to other bisphenols present in different food matrices, including meat. Contamination concentrations of up to 104.0 µg/kg of BPS in pork and 36.4 µg/kg of BPF and 326.0 µg/kg of BPA in canned meat were found. This bioaccumulation in animals and animal products may be due to the highly lipophilic properties of bisphenols (Urli *et al.*, 2023).

Humans are exposed to and contaminated by plastics, as are other living organisms

Plastics, in their various forms, enter the food chain and humans are therefore exposed to contaminated food and drinks, with increasing evidence of accumulation in the human body. This section focuses mainly on particulate plastics. For MPLs, Figure 36 summarises where they are found and how much occurs in the human body. **There is a diversity of polymer types found in human samples, all of which have so far been of petroleum origin. It should be noted that there is a lack of data on bio-based and/or biodegradable plastics and this needs to be further investigated. The presence of these petroleum-based polymers has only been linked in a few cases to lifestyle habits (*e.g.*, working and living conditions, drinking habits of bottled water, consumption of fast food packaged in plastics, smoking or exposure to dust, infant feeding practices). It is likely that because of this diversity of polymers, the effects on human health will be different, although the main results in this area, as described in section 3.4.3.1, have been obtained using PS as a polymer model.**

Routes of exposure to plastics have been identified. For particulate plastics, where the focus to date has been on MPLs, as for the other living organisms studied, ingestion of contaminated food and drinks (3.4.1.2) and inhalation are generally considered to be the main routes of entry into the human body. Inhalation has been studied in the context of occupational exposure to airborne MPLs. Different types of workers have been affected in the synthetic textile, flock and VC/PVC industries (Prata, 2018). More occasional routes of exposure have also been described, for example with the use of protective masks. The third possible route of exposure is dermal contact, which is poorly documented. Initial contact may occur through interaction with water contaminated with rinse-off products such as facial or body scrubs, or through shedding of fibres from clothing. Particle uptake through the skin requires penetration of the stratum corneum, which is limited to plastic particles smaller than 100 nm (Aristizabal *et al.*, 2024). For plastic-related compounds, in the case of food contact articles used to store, process, package and serve foodstuffs, a recent study reported that evidence of their presence in human samples was found for a total of 3601 (or 25%) of the 14,402 known food contact chemicals (Geueke *et al.*, 2025).

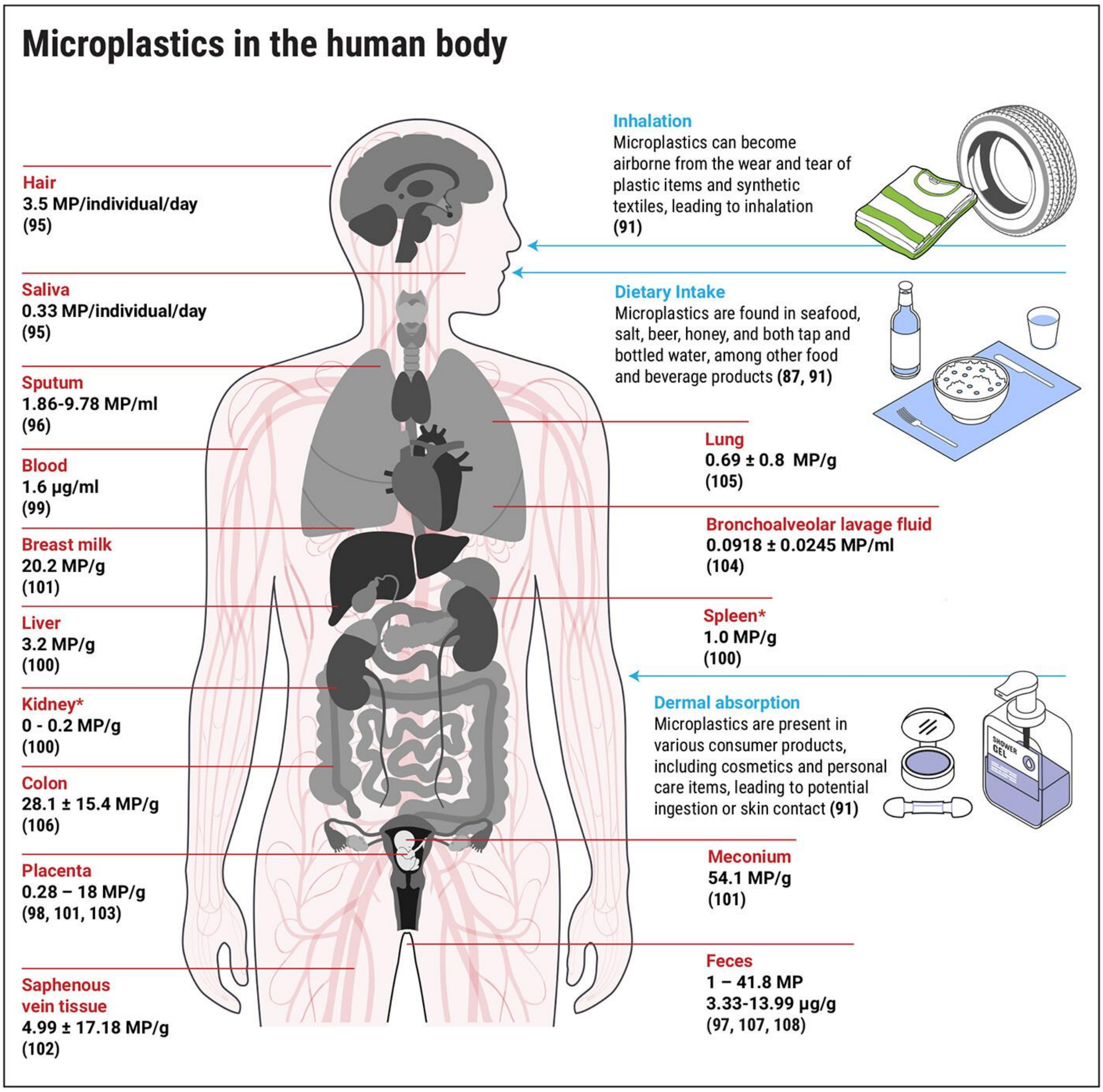

Figure 36. *Locations and amounts in the human body where MPLs have been reported. Exposure pathways (turquoise labels) and reported quantities (red labels) are shown, with * quantities reported as being around the limit of detection. The authors noted that quantities were as reported in each study and had not been further Quality Assurance and Quality Control (QA/QC) screened for their review. They emphasised that inter-study comparisons should be made with caution due to variation in methods and units of reporting between studies (3.4.4.2). (Thompson et al., 2024)*

The levels of MPL contamination, which are still poorly understood, appear to be linked to our lifestyles. Although the literature on the different dietary matrices is not equally developed (3.4.1.2), current studies emphasise that humans are mainly exposed to MPLs orally through food products of aquatic origin and drinking water (Jin *et al.*, 2021). Contamination also depends on factors such as food origin, local consumption habits and regional characteristics (*e.g.*, seafood consumption), but also on lifestyle (*e.g.*, living near or far from a major road, personal smoking habits, health status) in addition to occupational exposure. Furthermore, both children and adults accumulate MPLs, but the rate and amount can vary considerably depending on age, lifestyle and diet, with children potentially more susceptible due to higher intakes relative to their bw.

It is important that exposure data are interpreted with extreme caution. Some studies suggest extrapolation based on MPL occurrence and dietary habits to define human contamination levels (Cox *et al.*, 2020). Pletz (2022) looked at the highly publicised scientific statement that humans consume up to 5 g of MPLs per week (Senathirajah *et al.*, 2021), the equivalent of the weight of a credit card, and suggested that these previous calculations may be overestimated, pointing to methodological flaws. In fact, the actual amounts ingested by humans are still uncertain and likely to vary widely. Further research is therefore necessary for the refinement of these estimates, so that a health risk assessment is not possible at this time.

Human stools were the first evidence of MPL contamination in humans, including those at risk. Due to the analytical challenges posed by the various methods currently available for the quantification of MNPLs, no large-scale epidemiological study has yet been conducted in humans. However, converging evidence and an increasing number of studies in different types of samples (stools, organs, biological fluids) show that the human body is contaminated with MPLs. Following the seminal prospective study by Schwabl *et al.* (2019) on eight adult volunteers from Europe and Asia, several studies have consistently reported the presence of MPLs in human stools. For example, MPLs were found in the stools of Chinese student volunteers with an abundance ranging from 1 particle/g to 36 particles/g, with particle sizes ranging from 20 to 800 µm (Zhang *et al.*, 2021c). One to eight types of polymers were detected in each sample, with PP being the most abundant. Several types of plastic particles (PA, PUR, PVC, PTFE, PET) were detected in meconium. MPLs (PET and PC) were also found in meconium, infant and adult faecal samples collected in New York State (Zhang *et al.*, 2021a). For PET MPLs, a higher level was found in infants compared to adults, although for PC MPLs the overall occurrence was the same between the two. There is a need to focus on at-risk populations (infants but also people with susceptibility factors such as lifestyle, diet, comorbidities or stress, or patients with obesity or inflammatory bowel disease (IBD)). In IBD patients, the faecal MPL concentration was significantly higher than that in healthy individuals (Yan *et al.*, 2022a). A positive correlation was found between the faecal MPL concentration and the severity of IBD symptoms, suggesting that MPL exposure may be related to the development of the disease (3.4.3.1).

Body fluids and organs provided confirmatory evidence of MPL contamination in the human body. In addition to faecal samples, the presence of MPLs has been demonstrated in other human samples, including body fluids (*e.g.*, blood circulatory system). A first detection of MPLs (composed of PVA, PVC, PP and PE, 4-15 µm in size) was obtained in human urine (Pironti *et al.*, 2022), despite a small Italian cohort size of six volunteers. In addition, MPL contamination was found in a number of organs such as colonic tissue (mainly fibres of PC, PA and PP), male reproductive system (semen and testis), and respiratory tract (lung, bronchoalveolar fluid and sputum). Notably, in lung tissue samples collected from non-smoking patients, a median concentration of 2.19 particles/g was found, with most MPLs (mainly from PP, PET and PS) being between 20 and 100 µm in size (Wang *et al.*, 2023b). Compared with men and those living far from a major road, women and those living close to a major road had higher levels of MPLs in their lung tissue. Several converging studies have indicated the presence of MPLs (*e.g.*, from PVC, PP and PBS, mainly sizes smaller than 100 µm) in placentas (Ragusa *et al.*, 2021; Zhu *et al.*, 2023). Breast milk has also been found to be contaminated with MPLs (Ragusa *et al.*, 2022; Liu *et al.*, 2023c). Finally, the presence of MPLs in the human liver has been assessed, but only in patients with cirrhosis (Horvatits *et al.*, 2022).

3.4.2. Plastics pose a health hazard to living organisms and affect ecosystems through widespread contamination

(based on Chapters IV.2 and IV.3 of the extended report)

Most living organisms have similar sources of exposure to plastics, including air, food and water. The (eco)toxicological impact induced by exposure to MNPLs and plastic-related compounds shows similarities among organisms as described at different scales, *i.e.*, from the sub-individual (molecular, cellular) to the population level. Extrapolation from studies on marine organisms suggests possible adverse effects on freshwater (micro)organisms, which are so far limited to a few model species (*e.g.*, freshwater invertebrates such as *Daphnia magna*, algae and other aquatic plants). For MNPL-related studies, some of the experimental conditions currently tested have limitations, including the few types of polymers considered (with an over-representation of PS), the use of monodisperse, spherical and pristine particles, mainly from commercial sources, the partial data on NPLs and associated chemicals, but also the limited number of endpoints and species considered, mainly under laboratory-scale conditions. The effects described from these commercial particles result from a confounding influence of polymers and other potential chemical substances (*e.g.*, surfactants, additives) that cannot currently be distinguished. Despite these limitations, **adverse effects on behaviour, biological activity, growth, reproduction and metabolism have been convergently described in the literature. There are common players, in particular microbiota (gut microbiota, soil microbiota, plastisphere), between the different organisms. Similarly, common mechanisms of action have been highlighted, in particular oxidative stress.**

3.4.2.1. Plastics have (eco)toxicological effects on living organisms at multiple scales

This section provides a synthesis of the (eco)toxicological effects of particulate plastics and plastic-related compounds on soil invertebrates, soil microorganisms, terrestrial plants, wild terrestrial vertebrates, and livestock and poultry[15].

The effects of plastic-related compounds are most well-known for a limited number of chemicals

The literature to date has focused on a limited number of plastic-related compounds, most of which are endocrine disrupting chemicals (EDCs). These include plasticisers such as phthalates, octylphenol, NP and BPA and its analogues. In soil invertebrates, exposure of the earthworm *Eisenia andrei* to di-n-butyl phthalate (DBP) at low concentrations induced some adverse effects, including reduced cholinesterase activity at 1 mg/kg and reduced number of juveniles at 0.1 mg/kg (Berenstein *et al.*, 2022). DBP has also been reported to induce lesions in the reproductive system of rabbits and to affect spermatogenesis in frogs (Tan *et al.*, 2023). The only studies available to date on wild terrestrial mammals have focused on BPA and examined its effects on field voles (*Microtus agrestis*) or polecats (*Mustela putorius*). In field voles but not in polecats, exposure to subcutaneously injected BPA increased testosterone levels (Flint *et al.*, 2012). In wild amphibians, exposure to high doses of BPA was shown to affect the sexual and physical development of tadpoles.

Overall, the literature on livestock animals and poultry converges on the sensitivity of the female and male reproductive axis to EDCs, with results mainly derived from laboratory-scale conditions. In livestock ruminants, the majority of publications concern the effects of EDC exposure on reproductive function. Exposure to EDCs can affect the reproductive health of animals by acting directly on the gonads or through the hypothalamic-pituitary system. In sheep, octylphenol was reported to accelerate puberty (Majdic, 2010), and maternal exposure to a complex cocktail of EDCs, including BPA, induced reproductive health effects later in life. BPA exposure in lambs resulted in changes in follicular ovarian reserves (Urli *et al.*, 2023). In cattle, BPA exposure increased apoptotic gene expression in oocytes (Urli *et al.*, 2023). Both BPA and BPS disrupted oocyte-secreted proteins, altered the gap junctional intercellular communication of cumulus-oophore complexes and impaired the prophase I-to-MII transition in oocytes. BPS exposure induced changes in protein secretion, distribution of oestrogen receptors α and ß and aromatase in oocytes. BPA reduced the development of embryos at the blastocyst

[15] For humans, who are fully part of the living organism continuum, a separate section (3.4.3) is provided to present *in vitro* and *in vivo* results (including preclinical and clinical information) as a necessary step in assessing the risk of plastics to human health.

stage (Nandinee *et al.*, 2021). In addition, at the molecular level, some comparative studies have concluded that BPA causes meiotic and spindle fibre abnormalities and congenital defects in cattle.

The effects of several EDCs have been studied in poultry, including phthalates (DBP, DEHP), organophosphate flame retardants, BPA and its analogues (*e.g.*, BPS, bisphenol AF, and tetramethyl bisphenol F (TMBPF)). Particular attention has been paid to the effects of EDCs on the reproductive function, as for ruminants. For example, adverse effects of BPA have been reported in chickens, in particular increased embryo mortality and malformations of the reproductive organs (Urli *et al.*, 2023). Similarly, the effects of EDC exposure on reproductive function have been studied in pigs. NP and other additives (surfactants such as Surfynol®) have also been identified as having a high potential for reproductive toxicity (Nerín *et al.*, 2020). Importantly, in intensive pig production systems, fertility is achieved by artificial insemination. Semen from boars is collected, diluted with an appropriate aqueous solution and finally placed in high gas barrier plastic bags until final use. During storage, compounds (*e.g.*, cyclic phthalate, octyl phthalate, bisphenol A diglycidyl ether (BADGE), docosenamide, cyclic lactone) present in these plastic bags can leach into the semen solution and affect the functionality of the spermatozoa, causing reproductive failure (Nerin *et al.*, 2014).

The main focus to date has been on MPLs in relation to the impact of particulate plastics

Studies on soil organisms are limited to a few model species. Earthworms are the most widely studied species in ecotoxicological testing in response to MNPL exposure, with a particular focus on *Lumbricus terrestris* and *Eisenia fetida*. Tissue and DNA damage, increased burrowing activity, altered gut microbiome, neurotoxicity, oxidative stress and increased mortality are commonly reported adverse effects (Wang *et al.*, 2022a; Guo *et al.*, 2023). To date, studies evaluating the effects of MNPLs on nematodes have mainly been conducted using the model species *C. elegans*. MNPLs bioaccumulate and have a wide range of adverse effects (Lei *et al.*, 2018), including intestinal damage, oxidative stress, abnormal gonadal development and reproduction (with multi-generational and transgenerational effects), neuronal damage (cholinergic and gamma-aminobutyric acid (GABA) neurons), altered behaviour and energy metabolism, reduced body length and lifespan, and increased mortality. Overall nematode abundance may be affected. Soil parameters (*e.g.*, clay content and cation exchange capacity, porosity, moisture and pH) may influence the toxicity of MNPLs on nematodes. Finally, there is currently no scientific consensus on whether or not MNPLs are harmful to collembolans. Some preliminary studies have shed light on the possible adverse effects of MPLs, including behavioural changes, altered feeding behaviour, inhibited growth and reproduction. However, the effects of particulate plastics on collembolans have not been studied in the same detail as for earthworms and nematodes. Similarly, knowledge of potential effects on other soil organisms is clearly in its infancy.

Plastics serve as a novel ecological habitat for soil (micro)organisms, influencing their composition and functions at different trophic levels. Microbial communities and their activities are key indicators for assessing the impact of plastic contamination in soils (Figure 37). However, given the diversity of organisms, soil conditions and plastics, it is currently not possible to draw general conclusions about the overall impact of plastic contamination. Several studies have described that plastics serve as a novel ecological habitat for microorganisms living at the soil-plastic interface, allowing the formation of unique microbial communities, the plastisphere (Rillig *et al.*, 2024). In addition to their role as a carrier, plastics can be considered as an additional carbon source, and may provide diverse habitats for more rare microbial species (Zhang *et al.*, 2022). Overall, the plastisphere community is generally less diverse than the bulk soil and is enriched in microorganisms with specific traits and genes, such as pathogenic bacteria and bacteria with antibiotic resistance genes, or plastic-degrading microorganisms, including fungi (Rillig *et al.*, 2024). Changes in the microbial community were also observed in the soil surrounding the plastic debris (Wu *et al.*, 2024). From a functional point of view, plastic contamination can lead to either activated, suppressed or unchanged soil microbial enzyme activities (Zhou *et al.*, 2021).

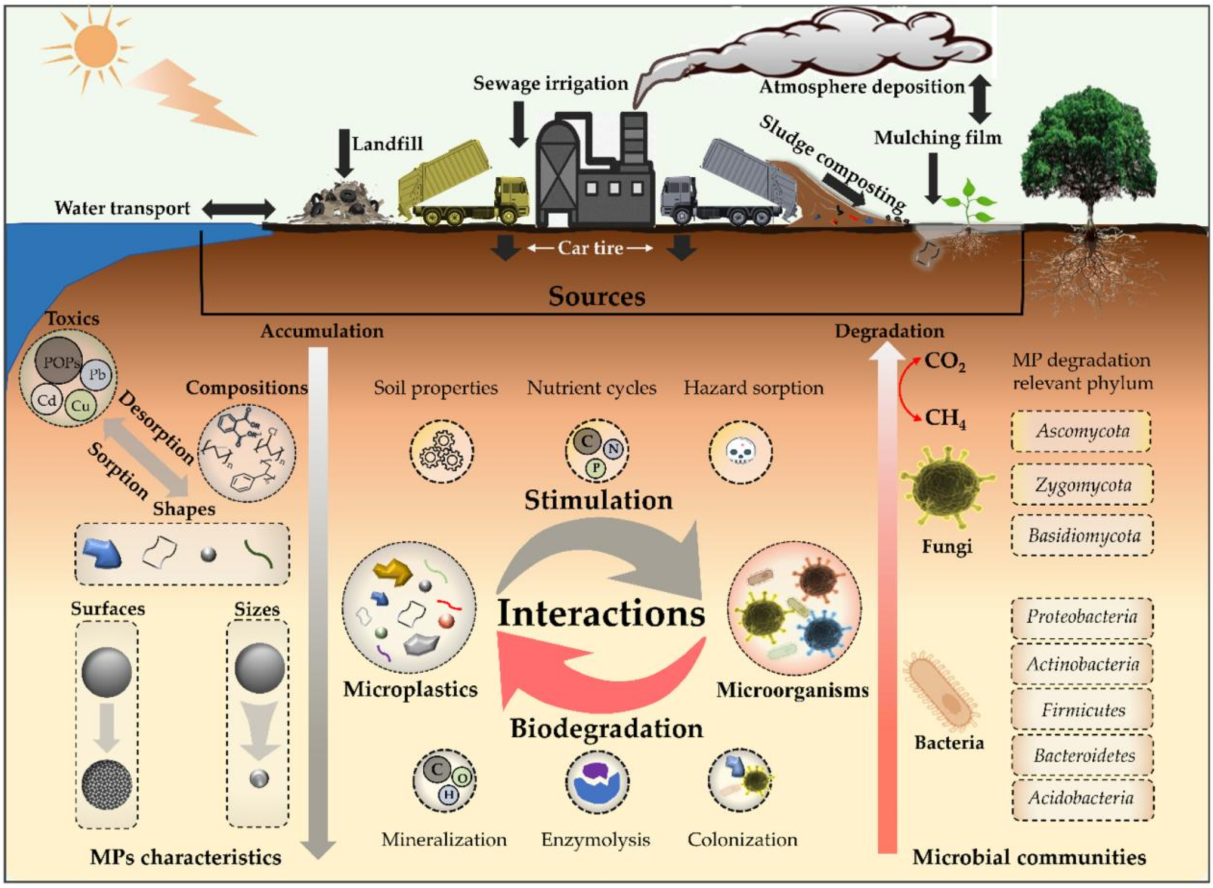

Figure 37. *Interactions between MPLs and microorganisms in soil systems. Arrows represent the directional processes of the interactions and the double-headed arrow exhibits bi-directional processes. Black arrows show potential sources of MPLs in soils, grey arrows show ageing processes of accumulation in soils and red arrows represent biodegradation processes of MPL-related microbial communities. (Zhang et al., 2021d)*

The effects of MPLs are transmitted through the soil-food web. Despite small effects on soil physico-chemical parameters, an altered and less diverse soil invertebrate community and reduced soil microbial activity have been observed in plastic-mulched fields. Interestingly, the responses of the soil fauna to MPL additions cascaded through the soil-food web, resulting in stronger indirect effects on soil microbial functioning than the direct effect induced by the MPLs themselves. This highlights the importance of considering the effects of MPLs at different trophic levels to elucidate the mechanisms involved in the ecological impact of MPL contamination on soil functioning.

The environmental conditions largely influence the effect of plastics on terrestrial plants. Many studies have shown that plastic contamination can have direct and indirect effects on terrestrial plants (Figure 38). Plastic contamination has positive, negative or negligible effects on plants, depending on the specific environmental conditions, the type of plastic, its size, quantity and the plant species considered. Endpoints studied to date include germination, root growth, elongation growth, biomass and photosynthesis. Germination is the most widely used, with inhibition or no effect observed under the various conditions studied. For example, a significant reduction in seed germination of garden cress (*Lepidium sativum*) was observed for PE and PP in a MPL-based assay (Pignattelli *et al.*, 2020), whereas NPLs had negligible effects on germination rates of onion (*Allium cepa*). Effects on germination may be caused by obstruction of root pores. The type of polymer may also influence seed germination.

In addition, the concentration of MPLs appears to be a determining factor and the germination rates of certain seeds can be restored and even enhanced at higher concentrations of MPLs (Zhang *et al.*, 2022). The effect of plastics on photosynthesis is also controversial. Some studies have shown that MPLs do not interfere with plant photosynthesis and even enhance it, probably by stimulating plant nutrient uptake and conversion, which facilitates the rate of photosynthetic carbon reaction (Lian *et al.*, 2020). Other studies suggest that disruption of photosynthesis is one of the main mechanisms contributing to the effects of particulate plastics on terrestrial plants (Zhang *et al.*, 2022). Finally, the degradability of plastics is an important property that influences their effect on plants. For example, biodegradable starch-based MPLs had a greater negative effect on wheat height and biomass than non-biodegradable petroleum-based MPLs. In addition, biodegradable MPLs (*i.e.*, from PHBV) caused wheat death (Zhou *et al.*, 2021), which may be attributed to the intermediate and/or final metabolites produced during PHBV degradation. Nevertheless, it should be mentioned that the degree of oxidative stress on rice shoot and root caused by PE-based mulching film MPLs was higher than that of MPLs from PBAT-based biodegradable mulching film.

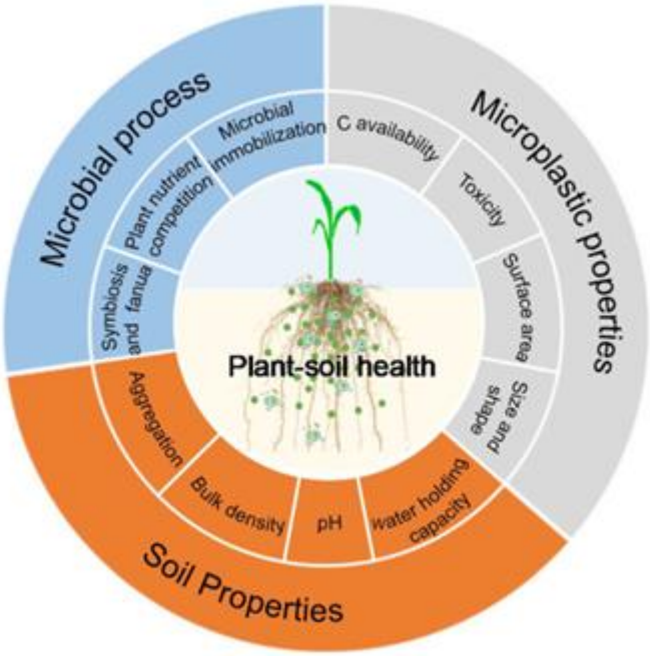

Figure 38. *Schematic overview of the connection between MPL properties, the soil processes they may influence, and the plant-soil health. (Zhou et al., 2021)*

The detailed molecular mechanisms underlying the phytotoxicity of MNPLs are still under investigation, but oxidative stress has been consistently implicated. When plants are exposed to MPLs or NPLs, an excess of reactive oxygen species (ROS) present in a variety of plant organelles including chloroplasts, mitochondria, peroxisomes and the endoplasmic reticulum, can cause irreversible damage to plant cells (Maity *et al.*, 2020; Li *et al.*, 2021). MPLs have also been shown to cause inhibition of plant growth-related gene expression and negative regulation, as well as reduced activity of key enzymes involved in carbohydrate metabolism. In addition, MPLs were found to induce chromosomal aberrations or oncogenic aberrations in onion, resulting in DNA and spindle damage and genotoxicity (Maity *et al.*, 2020). NPLs were also able to alter the expression of genes that specifically regulate resistance-related functions in rice roots and affect some important metabolic pathways against oxidative stress (Zhou *et al.*, 2021). Catalase activity in *Vicia faba* root tips, but also in cucumber and rice leaves, was shown to be dependent on the concentration and type of particulate plastics. The same diversity of response was observed in *Vicia faba* root tips and rice roots for superoxide dismutases.

The very few studies on wild terrestrial vertebrates have been conducted in laboratory conditions. The effects of exposure to particulate plastics on wild terrestrial mammals are to date largely unknown. The effects on non-mammalian vertebrates are also poorly understood. The few available studies have been conducted under laboratory conditions, mainly using PS particulate plastics. For example, exposure to MPLs has been found to induce neurotoxicity and alter several functional traits in freshwater fish, such as growth rate, early life stage development and behavioural patterns. After administration through an aquatic food chain (*i.e.*, from algae and via Daphnia), NPLs were shown to induce adverse effects on feeding and shoaling behaviour as well as on metabolism (Mattsson *et al.*, 2015). *Physalaemus cuvieri* tadpoles showed reduced locomotor ability, anti-anxiety behaviour and anti-predator responses following exposure by direct contact or ingestion of MPLs. Although the ingestion and contamination of birds by particulate plastics has been reported in many cases (3.4.1.3), the resulting effects are poorly known. As a first insight in this field, ingestion of MPLs by small terrestrial birds induced biochemical changes that were particularly predictive of oxidative stress and reduction-oxidation imbalance (de Souza *et al.*, 2022).

Livestock animals are less studied than poultry. The ingestion of MaPLs by livestock ruminants can result in a variety of symptoms, including suspended rumination, ruminal impaction, indigestion, partial or complete anorexia, recurrent bloating, rumen pain and inflammation, reduced milk yield and weight loss (Priyanka et Dey, 2018). Their persistence in the rumen interferes with normal digestion, fermentation and rumen microbiota function. Exposure of chickens to MPLs had negative effects on markers of energy metabolism. It also decreased total cholesterol and triglyceride concentrations, induced oxidative stress and neurotoxicity in skeletal muscle, altered the metabolomics profile and reduced meat quality (Chen *et al.*, 2023). Reduced growth performance and impaired antioxidant capacity were reported, along with liver inflammation, renal glomerular hypoplasia, altered kidney function and changes in intestinal villus structure. Gut microbiota composition and function were also affected. As a first insight into NPLs, exposure of poultry was found to cause structural changes in the intestinal villi with impaired absorption of elements such as copper, zinc, calcium but also vitamins (Kik *et al.*, 2020).

3.4.2.2. Particulate plastics have a ‘Trojan horse’ effect

Particulate plastics can adsorb and desorb inorganic and organic contaminants (*e.g.*, polychlorinated biphenyls (PCBs), PAHs, heavy metals, antibiotics, pesticides) and modify their fate (bioaccumulation, bioavailability) and toxicity. In addition, they can promote the spread of antibiotic resistance genes and microbial contaminants (*e.g.*, pathogens) through ecosystems. Importantly, due to their higher surface to volume ratio, NPLs have the capacity to adsorb and desorb larger amounts of chemical substances than MPLs at equivalent mass, potentially increasing their role as a ‘Trojan horse’.

Some evidence illustrates this ‘Trojan horse’ effect. In livestock, results obtained using a ruminant *in vitro* digestion model showed that the adsorption capacity and release of Cd occurred mainly in the rumen and abomasum phases (Liao et Yang, 2022). In poultry, the effects of co-exposure to PAS MPLs and an antibiotic (chlortetracycline (CTC)) were studied. It reduced the gut damage caused by MPL exposure by regulating the gut microbiota, but increased the total number of antibiotic resistance genes (Liu *et al.*, 2023a). First insights into the potential toxicity (carcinogenicity, disruption of early development, bone and liver metabolism, endocrine disruption) of PAHs attached to MPLs on fish were provided. In freshwater mussels, NPLs have been found to accumulate and alter the toxicity profile of PAHs, which in turn may facilitate the uptake of NPLs. It has also been hypothesised that MPLs not only act as a carrier for chemical substances, but also influence their bioavailability. This was demonstrated in common carp exposed to PE MPLs and glyphosate (Chen *et al.*, 2022). In a study on collembolans (*Folsomia candida*), exposure to PS MPLs loaded with the antibiotic sulfamethoxazole showed a distinct gut microbiota and antibiotic resistance gene profile compared to those exposed to MPLs alone (Xiang *et al.*, 2019). In terrestrial plants, most studies observed a decrease in contaminant bioaccumulation in the presence of plastics (*e.g.*, PS and heavy metal in wheat). In addition, the toxicity of arsenic(III) to rice was increased or decreased when MPLs and arsenic were co-exposed, depending on the MPL concentration. PS NPLs could accelerate the formation of long-lived radicals in wheat leaves after exposure to Cd, and improve carbohydrate and amino acid metabolism, thus partially reducing Cd levels in leaves and alleviating Cd toxicity to wheat (Lian *et al.*, 2020). A final example of this 'Trojan horse' effect involves MPLs and their interaction with pesticides in soils. MPLs have the ability to retain pesticides, especially the more hydrophobic ones, thereby reducing their immediate availability for leaching, degradation, uptake by plants or uptake by other organisms present in soils. As a result, pesticides may not reach target pests at recommended doses, contaminated plastics may be ingested or taken up by non-target organisms, or contaminated plastics may desorb pesticides over time, resulting in long-term release of pesticides into the ecosystem (Peña *et al.*, 2023).

3.4.2.3. The accumulation of particulate plastics has an impact on ecosystem functioning and is likely to affect the provision of ecosystem services

MPLs and, although less well documented, NPLs, affect the growth, development and physiology of most of microbial, plant or animal species including keystone species. By altering biodiversity, plastics can have higher-level effects on ecological processes (*i.e.*, activities that result from interactions between organisms and between organisms and their environment), thereby disrupting the processes underlying ecosystem functions and ecosystem services.

Knowledge of the impacts of plastics on ecosystem functions in realistic agro-environmental situations remains fragmentary. However, the vast majority of available studies indicate that plastics have a wide range of detrimental effects on the chemical, physical and biological properties of the ecosystems, thereby affecting their functions, with a lack of consensus in some cases. Based on Pesce *et al.* (2023), twelve ecosystem functions (EFs) were considered as potentially affected by plastics (Table 16). The effects of plastics on these functions are diversely documented with 'Formation and maintenance of soil and sediment structure', 'Water retention in soil and sediment' and 'Production and input of organic matter in terrestrial and aquatic ecosystems' being the most well documented.

Overall, particulate plastics are found to affect the balance and stability of ecological interactions and disrupt ecosystem functions that underpin the provision of ecosystem services. As such, they may affect the provision of these ecosystem services. To address this question, the Common International Classification of Ecosystem Services (CICES) scheme (version 5.1; Haines-Young and Potschin (2018) was used. Of the three categories of services considered (*i.e.*, provisioning services, regulating and maintaining services, and cultural services), only the first two have received some attention. In particular, the potential alteration of provisioning services (*e.g.*, drinking water and soil quality, feed and food quality and productivity, biomass and keystone species) due to plastic contamination has been addressed. MPLs have been shown to reduce the biomass and productivity of many crops, and their accumulation can reduce crop yields in the long term. In addition, plastics affect the digestive tract of livestock and poultry, leading to indigestion or anorexia, weight loss and reduced growth, thereby reducing meat production. Finally, the regulation and maintenance services related to the quality of ecosystems are affected by the impact of plastics, especially MPLs, on several functions (*e.g.*, degradation of organic matter, alteration of soil structure, nutrient and energy reserves, habitats for numerous (micro)organisms, disruption of the food chain).

Table 16. *Main results presented in the literature and associated number of reviews and articles for each of the twelve ecosystem functions considered.*

Ecosystem function	Main results	Number of reviews + articles		
		MPLs	NPLs	Other forms of plastics
EF1 - Gas regulation	Plastics of different types and sizes affect greenhouse gas emissions (NO_2, NH_3, CH_4 and CO_2) with a possible impact on global warming. Effects on soil N and C transformations have been reported, with decreases or increases in gas emissions depending on specific pedo-climatic situations, plastic types, media or amendments. Similar effects have been reported in marine ecosystems. Studies in freshwater and estuarine ecosystems are lacking.	1 + 9	1 + 1	0 + 1
EF2 - Dissipation and mitigation of contaminants and wastes in terrestrial and aquatic ecosystems	Plastic additives and environmental pollutants that plastics can adsorb are transferred through food webs, affecting microflora, flora and fauna. Effects on ecosystem functioning need to be investigated.	3 + 3	0	0 + 3
EF3 - Resistance to disturbance	Effects on ecosystem resistance to disturbance are poorly documented.	0 + 1	0 + 1	0
EF4 - Water retention in soil and sediment	The presence of plastic particles in soil systems alters water distribution, infiltration pathways and subsequent water holding capacity, depending on the nature of the polymer. Plastic films, and probably fibres, can alter water flow in soils by affecting water infiltration and absorption, with possible implications for plant germination and growth, and reduced soil microbiological activity.	5 + 1	0	0 + 2
EF5 - Water flow regulation				
EF6 - Albedo and reflection	Albedo and reflection help regulate soil temperature, which in turn affects soil biophysical processes. Plastic mulches can indirectly affect soil ecosystems by modifying the soil microclimate and atmosphere.	1 + 0	0	0 + 2
EF7 - Production and input of organic matter in terrestrial and aquatic ecosystems	Despite the low contribution of MPLs to soil carbon content, plastics affect the environmental carbon cycle by reducing soil organic matter concentrations and altering the molecular composition of soil dissolved organic matter. Mulching films and MPLs affect both the composition and activity of microbial communities, with consequences for geochemical cycles. However, there is no consensus on the effects of plastics on the regulation of nutrient cycling.	4 + 5	0 +1	0 + 2
EF8 - Nutrient regulation in terrestrial and aquatic ecosystems		3 + 4	0 + 2	0 + 5
EF9 - Formation and maintenance of soil and sediment structure	The presence of MPLs generally affects soil physical properties and parameters by reducing soil bulk density and aggregate stability to water. The effects of plastics on sediment structure are not known.	7 + 2	0	0
EF10 - Dispersion of propagules in terrestrial and aquatic ecosystems	Plastics affect the transport of propagules (spores, cysts) by wind, water or animals, although this is poorly documented in terrestrial ecosystems.	0	0	0 + 3
EF11 - Provision and maintenance of biodiversity and biotic interactions in terrestrial and aquatic ecosystems	Particulate plastics affect individual organisms as well as populations and communities (*e.g.*, arbuscular mycorrhizal fungi in soils with contrasting effects depending on the type of polymer), with consequences for the provision and maintenance of biodiversity and biotic interactions.	2 + 4	0	0
EF12 - Provision and maintenance of habitats and biotopes in terrestrial and aquatic ecosystems	The reported effects of plastics are diverse. For example, plastics in soil can entangle soil particles, leading to clumping, which ultimately affects soil-dwelling microorganisms in the vicinity. The presence of fibres reduces the bulk density of the soil and can promote plant growth, probably by reducing the resistance to root growth.	2	0	0 + 2

3.4.3. A focus on human health from a translational research perspective provides a way to better assess the health risks of plastics

(based on Chapter IV.3 of the extended report)

3.4.3.1. The role of plastics in promoting human disease is known to a different extent for plastic-related compounds and particulate plastics

The role of plastic-related compounds in promoting human disease is limited and only fully elucidated for bisphenol A and phthalates

BPA and phthalates harm human health. For these plastic-related compounds, there is preclinical (*i.e.*, rodent studies) and clinical (*i.e.*, epidemiological studies) evidence of their role in promoting human disease. They are the most studied EDCs to date, because of the public debate on them, the risk they pose to human health, and because they are regulated at European level (Chapter V.1 of the extended report). Possible effects of other compounds introduced by plastics have not yet been thoroughly investigated.

Adverse effects of BPA on human health at very low concentrations have been convergently demonstrated and toxicological reference values have been adjusted over time. In 2023, EFSA (2023) established a tolerable daily intake TDI of 0.2 ng/kg bw/day, replacing the previous provisional level of 4 µg/kg bw/day. This TDI is based on adverse immune effects, reproductive and developmental toxicity, metabolic effects, but also neurotoxicity. In addition, accumulating evidence implicates BPA in cancer, polycystic ovarian syndrome, obesity, type 2 diabetes and cardiovascular disease (Table 17). Its substitutes, *e.g.*, BPF, BPS, bisphenol AF, have also been identified as endocrine-disrupting chemicals and have similar overall adverse effects (den Braver-Sewradj *et al.*, 2020; Catenza *et al.*, 2021; McDonough *et al.*, 2021). The same regulatory update strategy has been implemented for phthalates. In 2019, EFSA (2019) re-evaluated the risk assessments of DBP, DEHP, butylbenzyl phthalate (BBP), di-isononyl phthalate (DINP) and di-isodecyl phthalate (DIDP). Based on a plausible common mechanism (*i.e.*, reduction in foetal testosterone) underlying the reproductive effects (Table 17) of DEHP, DBP and BBP, a group-TDI of 50 µg/kg bw/day, expressed as DEHP equivalents, was established. Moreover, EFSA highlighted the adverse immunotoxic, metabolic and neurotoxic effects (Table 17) and called for further studies on these endpoints that could be more sensitive. The immunotoxicity of phthalates was based on epidemiological studies reporting associations of several phthalates (DEHP, DBP, di-isobutyl phthalate (DIBP), DINP, and DIDP) with respiratory allergy, asthma and atopic dermatitis. In addition, recent evidence has suggested that phthalates may be involved in cardiovascular disease, childhood eczema, early and delayed puberty, and placental outcomes.

A synthesis of the estimated preclinical and clinical level of evidence on the toxicity of bisphenols and phthalates was provided (Chapter IV.3 of the extended report). All preclinical studies including adult and perinatal exposure were included. This synthesis showed that **the toxicity of bisphenols and phthalates is associated with a wide range of human diseases.** Similar data are not available for the wide range of other potentially hazardous chemical substances associated with plastics (3.4.1.2). Early action based on a **hazard-based approach** to address some of the shortcomings of 'traditional' health risk assessment strategies (*e.g.*, time and resource intensive, lack of information) may tackle this challenge (Wagner *et al.*, 2024).

The burden of human disease has a cost. The Endocrine Society has been at the forefront of documenting the disease burden and associated costs of hazardous chemical substances used in plastics. Indeed, conservatively estimated disease costs have been assessed in the European Union (Trasande *et al.*, 2015), the USA (Attina *et al.*, 2016) and Canada (Malits *et al.*, 2022). Trasande *et al.* (2015) also estimated cardiovascular mortality due to phthalates in the USA. Table 18 shows the data obtained for phthalates and BPA. The burden of human disease represents an enormous economic cost to society, equivalent to 33.23 billion euros at 2025 prices in the EU. Importantly, estimates are conservative because they are limited to a subset of chemicals and diseases in the EU, Canada and the USA.

Table 17. *Synthesis of the estimated preclinical and clinical levels of evidence for human toxicity of bisphenol A and phthalates. The level of evidence was considered high if the toxicity was recognised either by EFSA or by at least one meta-analysis. The level of evidence was considered medium if the toxicity was recognised by at least one systematic review or by more than three consistent publications. The level of evidence was considered low if the toxicity was found in one to three publications. If no study was found, 'not studied' (NS) was reported.*

	Bisphenol A		**Phthalates**	
	Preclinical level of evidence	Clinical level of evidence	Preclinical level of evidence	Clinical level of evidence
Asthma/allergy	Medium	High	High	High
Cancer	Medium	Medium	High	High
Cardiovascular diseases	Medium	High	Medium	High
Metabolic diseases	Medium	High	High	High
Developmental toxicity	High	High	High	High
Male reproductive toxicity	High	High	High	High
Female reproductive toxicity	High	High	High	High
Gastro-intestinal diseases	High	Low	Medium	Low
Hepatic diseases	Medium	NS	High	NS
Immune diseases	High	Low	Medium	Low
Neurological diseases	High	Medium	Medium	Medium
Pulmonary diseases	Low	Low	Low	Low
Renal diseases	Medium	High	Low	High
Thyroid diseases	High	High	Medium	High

***Table 18.** Estimates of the costs of diseases associated with phthalates and bisphenol A for the USA, Canada and the European Union. (adapted from Trasande et al. (2015))*

Contaminant	Life stage of exposure	Outcome	USA		Canada		European Union	
			Disease burden (# cases)	Economic cost (billion US$)	Disease burden (# cases)	Economic cost (billion US$)	Disease burden (# cases)	Economic cost (billion US$)
Phthalates	Adult	Obesity	5,900	1.7	2,093	0.6848	53,900	20.8
	Adult	Type 2 Diabetes	1,300	0.0914	225	0.0258	20,500	0.8072
	Adult females	Endometriosis	86,000	47.0	10,151	5.7	145,000	1.7
	Adult males	Male infertility	240,100	2.5	1,395	0.017	618,000	6.3
	Adults	Cardiovascular mortality	90,800	39.9				
Bisphenol A	Prenatal	Childhood obesity	33,000	2.4	1,023	0.059	42,400	2.0

The role of particulate plastics in promoting human disease is based on preclinical evidence only

Based on consistent conclusions across publications, the toxicity of MNPLs to human health appears no longer to be questioned by the scientific community, with the gut and lung being the primary target organs. In contrast to BPA (and its substitutes) and phthalates**, there is currently preclinical but no clinical evidence of human disease promotion by particulate plastics, except for gastro-intestinal disease (and with a limited number of studies)** (Table 19). Toxicological reference values are not available or discussed in preclinical studies. As a first attempt in this direction, consolidated data on spherical PS MNPLs only (distinguishing between MPLs and NPLs) were collected and the lowest observed adverse effect level (LOAEL) was estimated. **Developmental toxicity, male or female reproductive toxicity and gastro-intestinal toxicity of MPLs or NPLs were observed from 20 µg/kg bw/day. The same threshold was established for hepatic and renal toxicity of MPLs. MPLs induced neurological toxicity from 6.5 ng/kg bw/day. The other toxicity thresholds were higher than 20 µg/kg bw/day.**

Table 19. *Synthesis of the estimated preclinical and clinical levels of evidence for MNPL human toxicity. The level of evidence was considered high if the toxicity was recognised either by EFSA or by at least one meta-analysis. The level of evidence was considered medium if the toxicity was recognised by at least one systematic review or by more than three consistent publications. The level of evidence was considered low if the toxicity was found in one to three publications. If no study was found, 'not studied' (NS) was reported.*

	MPLs		**NPLs**	
	Preclinical level of evidence	Clinical level of evidence	Preclinical level of evidence	Clinical level of evidence
Asthma/allergy	Low	NS	NS	NS
Cancer	NS	NS	Low	NS
Cardiovascular diseases	Low	NS	Low	NS
Metabolic diseases	Low	NS	Low	NS
Developmental toxicity	Medium	NS	Medium	NS
Male reproductive toxicity	Medium	NS	Medium	NS
Female reproductive toxicity	Medium	NS	Medium	NS
Gastro-intestinal diseases	Medium	Low	Medium	NS
Hepatic diseases	Medium	NS	Low	NS
Immune diseases	Medium	NS	NS	NS
Neurological diseases	Medium	NS	Medium	NS
Pulmonary diseases	Medium	NS	Medium	NS
Renal diseases	Medium	NS	Low	NS
Thyroid diseases	NS	NS	NS	NS

Animal studies converge on the putative toxicity of MNPLs to many organs. The main areas of concern in these preclinical rodent models are reproductive and developmental toxicity (Afreen *et al.*, 2023; Fard *et al.*, 2023; Hong *et al.*, 2023; Song *et al.*, 2023). In males, ingestion of MNPLs decreases sperm count and motility and increases sperm abnormalities. In females, ovarian function is reduced. MNPL-induced decreased female fertility is also associated with direct damage to the placenta and to uterus. MNPL ingestion impairs the immune response, alters the ability to defend against sepsis or infection, but also impairs the gut microbiota and has deleterious effects on the colon such as inflammation, weakening of the gut barrier, colitis and *ex vivo* exacerbation of tumorigenesis (Hirt et Body-Malapel, 2020; Turroni *et al.*, 2021; De Souza-Silva *et al.*, 2022; Santos *et al.*, 2022). Notably, there are consistent data showing that the microbial dysbiotic effect of MNPLs may be exacerbated in individuals with colitis or metabolic disorders, probably related to gut barrier dysfunction. In addition, MNPL ingestion induces inflammation and fibrosis in the liver, kidney and lung. Finally, MNPLs cause cardiac oedema and fibrosis, as well as behavioural and cognitive problems, including memory loss, learning difficulties, anxiety, depression, as well as autism spectrum disorder-like symptoms and Parkinson's disease-like symptoms (Wang *et al.*, 2022c; Guimaraes *et al.*, 2023; Kim *et al.*, 2023). However, most of these preclinical studies in mice have been conducted with commercially available PS MNPLs. Therefore, they only reflect the toxicity of spherical PS MNPLs. They do not take into account the mixture effects of different MNPL shapes and polymer types or the weathering and ageing of MNPLs. To date, only three studies (Deng *et al.*, 2022; Jia *et al.*, 2023; Yang *et al.*, 2023) have been carried out on more realistic MNPLs, *i.e.*, ground or crushed plastic particles.

In vitro models are a step forward in the identification of common mechanisms of plastic toxicity to the gut, lung and secondary organs

Human cancer or immortalised cell lines derived from gut or lung tissue are used as experimental models in most *in vitro* studies dealing with ingestion or inhalation route of exposure. Some insights into the human gut ecosystem have been provided using *in vitro* fit-for-purpose lower gut models (3.4.3.2). These models have consistently shown that exposure to MPLs (PE, PET or PLA) leads to perturbations in both the composition and metabolic activity of the human gut microbiota (Tamargo *et al.*, 2022; Jiménez-Arroyo *et al.*, 2023; Fournier *et al.*, 2023a; 2023b). Regarding the interplay between MPLs and plastic-related compounds, the microbial dysbiotic effects induced by plasticisers (phthalates) were found to be at least as important as those of MPLs alone, and these effects could be additive and even synergistic (Yan *et al.*, 2022b). For gut or lung cell models, while exposure at 1 µg/mL is relevant because it approximates environmental exposure measurements, concentrations above 100 µg/mL appear unrealistic (Barceló *et al.*, 2023). However, high concentrations are often used for dose-response assessment and to facilitate primary observation of some cellular and molecular effects. Cellular uptake of MPLs larger than 10 µm is limited, but can still cause cell membrane damage. Plastic particles smaller than 10 µm can be internalised by cells by endocytosis, with an efficiency that is inversely related to particle size (50-100 nm) in the gastro-intestinal tract or in the alveolar cells at the end of the lung tracts. After exposure, MNPLs can induce inflammatory responses related to the immune response to foreign agents. MNPLs can interact with blood and lymphatic immune cells and reach secondary organs (liver, brain, ovaries and testes, placenta, etc.) due to their ability to cross barriers. Translocation is likely to be facilitated during inflammation due to the increased permeability of epithelial barriers.

For the ingestion or inhalation route of exposure, cytotoxicity, oxidative stress, genotoxicity (*e.g.*, double stranded deoxyribonucleic acid (dsDNA) breaks) or release of pro-inflammatory cytokines have often been reported (although usually at very high, rather unrealistic, test concentrations) (Yong *et al.*, 2020). In *in vitro* models simulating secondary organs (liver, reproductive system, blood-brain barrier), cytotoxicity, oxidative damage, increased apoptosis and inflammatory response have also been reported. This highlights mechanisms common to all living organisms (3.4.2.1). It is noteworthy that, although the exposure scenarios are still highly variable (*e.g.*, type of *in vitro* model, exposure time), an increasing number of studies deal with MNPLs with variable composition, size distribution (polydisperse) and modifications (simulation of ageing, corona formation, gastro-intestinal digestion, etc.). For plastic-related compounds, studies are mostly carried out on bisphenols (BPA, tetrabromobisphenol A (TBBPA)) and phthalates (DEHP). Interestingly, in a few cases, these plastic-related compounds are studied in combination with MPLs. For example, exposure of a colon cancer cell line to TBBPA and high concentrations of PE MPLs resulted in mild joint toxicity (Huang *et al.*, 2021). In addition, the combined adverse effects of PS NPLs and BPA on a liver cell model were found to be synergistic with increased hepatotoxicity and alteration of lipid metabolism.

Improved human health risk assessment relies on moving forward in vitro: a necessary step from basic to more advanced dynamic models

In vitro cell-based assays, as currently performed under human-like conditions, allow a rapid screening of the effects of MNPL exposure as well as mechanistic studies; they are also compatible with multi-parametric and high-throughput screening. However, there may be a discrepancy between the concentration used *in vitro* and the 'real' concentration in contact with cells (due to the buoyancy of particulate plastics). In addition, different units are generally used (µg/mL, µg/cm^2, particles/mL), making comparisons and conclusions difficult. 2D culture systems, widely used to determine the effect of MNPLs on biological barrier integrity, cellular uptake and biological outcomes, appear to be the most standardised platforms, mainly because of their low cost, high reproducibility and ease of manipulation. While cell monocultures are basic models that are very useful for screening purposes, co-culture models combining different cell lines better recapitulate the physiological environment and the interactions that occur between the different cell types, and improve the predictive potential of *in vitro* models. More physiologically relevant *in vitro* models such as 3D organoids or organs-on-chips have been developed for the gut, lung, liver or brain (Miloradovic *et al.*, 2021; Forest et Pourchez, 2023; Wang *et al.*, 2023a). They are used for mechanism of action studies to better understand the complex issues associated with exposure to MNPLs and/or plastic-related compounds.

In order to assess the impact of particulate plastics (only MPLs have been studied so far) on the composition and functions of the human microbiota, *in vitro* digestion models that recapitulate the human upper and/or lower gut ecosystem are increasingly being used. Several models have been tested, inoculated with stools from healthy volunteers and exposed to MPLs under different conditions (*e.g.*, MPL size, polymer type or dose, acute or repeated exposure). For example, the effects of MPL exposure on infant microbiota have been reported using Toddler Mucosal Artificial Colon (Tm-ARCOL), an adaptation of M-ARCOL to infant gut physiology (Fournier *et al.*, 2023b). Tm-ARCOL is coupled with a co-culture of intestinal enterocytes and mucus-secreting cells to evaluate the host response. Other set-ups have been implemented, such as the combination of the gastro-intestinal INFOGEST system with the dynamic Simulator of Gastro-Intestinal tract (Simgi®) (Tamargo *et al.*, 2022) or, to further increase the physiological relevance, the dynamic Mucosal-Simulator of the Human Intestinal Microbial Ecosystem (M-SHIME®) system (Yan *et al.*, 2022b). It should be noted that, with the exception of infants, no at-risk populations have yet been simulated.

3.4.4. Experimental analytical methods for plastics exist but need to be improved for the wide range of matrices and plastic forms being considered

(based on Chapter IV.1 of the extended report)

To fully address the global problem of plastic contamination and its impact on all ecosystems, it is necessary to develop a holistic experimental strategy that breaks down barriers between scientific communities. For this reason, the focus of this final section is on analytical chemistry as a key transversal approach. Indeed, complementary analytical methods and tools are available to detect and quantify plastics (including MNPLs, plastic-related compounds and chemical contaminants) in environmental (3.4.1.1) and food/drink matrices (3.4.1.2), as well as in living organisms including humans (3.4.1.3). There is a strong focus in the literature on MPLs (only 17% and 7% of references deal with NPLs and chemical substances respectively).

3.4.4.1. A wide range of analytical techniques are suitable for use with plastic-related compounds

The analysis of chemical substances can be approached by various methods: directly from the plastic particles, by extracting the additives and contaminants using solvents or by assessing the residual concentration in the aqueous phase using a differential approach. For the analysis of organic molecules, gas chromatography (GC) and liquid chromatography (LC) techniques are preferred because of their precision in separating and analysing these compounds. For metallic contaminants and additives, extraction by acid digestion, using wet and microwave assisted methods, is commonly used. Sophisticated characterisation techniques including Inductively Coupled Plasma Optical Emission Spectroscopy (ICP-OES), Inductively Coupled Plasma Mass Spectrometry (ICP-MS), X-ray Fluorescence, and Atomic Absorption Spectroscopy, have been documented for their effectiveness in accurately assessing trace metal elements. In the specific case of food and drink matrices, the migration of plastic-related compounds from food packaging and/or kitchenware and tableware utensils is evaluated by using GC coupled to either Flame Ionisation Detector (FID) or Mass Spectrometry (MS) for volatile compounds and LC using different detection techniques (Wrona et Nerín, 2020). MS is always required for accurate compound identification, particularly High Resolution MS (HRMS; *e.g.*, quadrupole time-of-flight (QTOF) or Orbitrap). For oligomers (as they represent an emerging area of concern for toxicity), chromatography coupled to MS is commonly used to qualify and quantify them in food samples (3.4.1.2). LC separates oligomers with polar functional groups and is often coupled to MS in Electrospray ionisation (ESI) mode, while GC is suitable for volatile oligomers of polyolefin polymers such as PE and PP. The analysis of other matrices (*e.g.*, complex environmental samples) is challenging due to the lack of authentic standards and the low concentration of oligomers. Their physico-chemical properties also vary considerably, requiring different analytical methods. The synthesis of oligomer standards is essential but complex, often performed ad hoc (Shi *et al.*, 2023).

3.4.4.2. A dedicated workflow is proposed for the analysis of particulate plastics

The identification, quantification, and characterisation of MNPLs in various matrices present a major analytical challenge due to the intrinsic heterogeneity of these plastic particles. Their different dimensions, morphologies, densities, polymer compositions and surface properties require the development and application of advanced, targeted analytical methods. This need arises from the complexity of reliably detecting these particles in environmental matrices (atmosphere, hydrosphere - freshwaters and estuaries, pedosphere - soil, compost and sediments), food and drink matrices, and living organisms. A workflow was proposed in Chapter IV.1 of the extended report, which is organised around four main tasks (Figure 39): (i) sampling; this initial stage involves the collection of samples from the different matrices to be studied, (ii) sample preparation; this stage involves the pre-treatment, extraction/concentration, and cleaning/purification of the samples, (iii) analysis techniques; this stage involves the application of specific methods for the identification/quantification and characterisation of plastic particles, (iv) method validation, quality assurance and quality control (QA/QC); this final stage ensures the reliability and reproducibility of the results obtained. Such a workflow highlights the inherent complexity of each stage of the process, which has its own challenges, exacerbated by a wide variety of procedures. For example, for MPLs in soil matrices, it is important to note that because visual observation is the basis for shape categorisation, there is a bias towards underestimating small particles, which are often masked or difficult to see, and overestimating larger particles, particularly fibres. This reflects the complexity of the overall methodology and helps to understand and explain the current lack of harmonisation and standardisation of methods as suggested in the scientific literature.

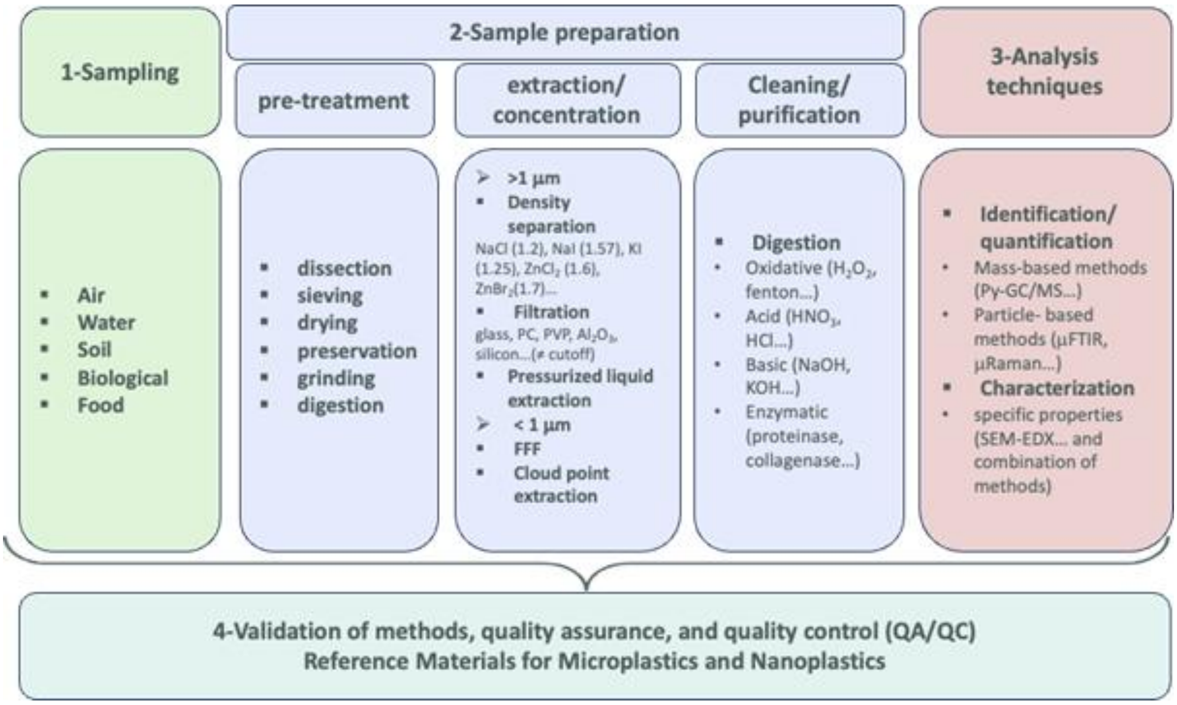

Figure 39. *Systematic workflow proposed for the methodological analysis of MNPLs: key stages from sample collection to result validation.*

Sensitive analytical methods are required for the chemical identification and quantification of MNPLs in complex matrices (Ivleva, 2021). Notably, this requirement does not apply to MaPLs, the quantity of which is sufficient to allow classical polymer analysis without raising specific methodological issues. Although there are several methods for the reliable identification of polymers present in MNPLs, quantification can be divided into two main categories: one based on mass and the other on the number of particles. In addition, characterising specific properties of MNPLs (*e.g.*, degradation state, surface properties, additives, alteration products, adsorbed chemicals) requires the use of one or more additional analytical methods. Currently, the most applicable mass-based methods include Pyrolysis-GC/MS (Py-GC/MS) and Thermal Desorption-GC/MS (TED-GC/MS). While the former has superior absolute sensitivity, the latter allows the analysis of relatively large sample masses (up to 100 mg, approximately 200 times more than for Py-GC/MS). Although all mass-based methods are destructive and provide information on the chemical composition and concentration of MNPLs regardless of the physical appearance of the particles, they should be considered complementary rather than competitive with particle-based methods (Seeley et Lynch, 2023). Of these, infrared spectroscopy is the most widely used due to the accessibility of the instrumentation (*e.g.*, Attenuated total reflection with Fourier transform IR spectroscopy (ATR-FTIR) available in the majority of analytical laboratories) and the reliability of the measurements obtained. Combined with microscopy (µ-FTIR), this technique can be used to identify and quantify particles, including information on their number, size/size distribution and shape, with a spatial resolution of around 10-20 µm. µ-Raman spectroscopy, another type of vibrational spectroscopy, is an effective method for the chemical identification and quantification of MPLs down to 1 µm (and even NPLs down to about 300 nm). Although this method is applicable to the analysis of MPLs in a smaller size range compared to µ-FTIR, µ-Raman analysis is generally more time-consuming and often affected by fluorescence interference, although correction procedures exist. (FT)IR and Raman spectroscopies require adapted spectral libraries. To ensure accurate identification, these libraries must include reference spectra not only for new plastics but also for altered plastics and non-plastic materials. In the specific case of NPL analysis, techniques already established in the study of MPLs can be adapted. In this respect, the combined use of Field-Flow Fractionation (FFF) – including particle size characterisation via UV and Multi-Angle Light Scattering (MALS) detectors – and the identification of NPLs by off-line approaches (*e.g.*, Py-GC/MS) has already shown promising results (Ivleva, 2021; Wahl *et al.*, 2021).

The implementation of specific QA/QC procedures is a critical step in ensuring the reliability of studies by standardising the materials and methods used in the analysis of MNPLs. The use of plastic reference particles and the use of blanks to estimate false positives and detect false negatives at different stages of analysis are recognised as necessary practices. Adjusting and correcting recovery rates also helps to provide reliable and verifiable data. Other essential measures include performing analyses in an environment free of plastic contamination, the preferred use of equipment made of inert materials such as glass, metal, or ceramic, the use of cotton clothing, a controlled work environment (*e.g.*, laminar flow box or clean room for sample preparation), the rigorous application of cleaning procedures for equipment and reagents used, and participation in interlaboratory comparisons (Schymanski *et al.*, 2021; Caldwell *et al.*, 2022).

3.5. Is a sustainable system of plastics used in agriculture and for food possible?

3.5.1. Plastics used in agriculture and for food are entangled in a complex plastic sociotechnical system

From a systemic point of view, plastics used in agriculture and food production cannot be considered in isolation from the broader plastic sociotechnical system. These plastics are deeply enmeshed within global supply, production, and consumption chains (3.1). They are intricately intertwined with other sectors through complex flows of raw materials (3.2), processed products, and waste (3.3). This entanglement means their management and environmental impact (3.4) are intrinsically linked to the dynamics of the entire plastic sociotechnical system.

The term ‘plastic sociotechnical system’ refers to the interconnected network of processes, actors, technologies, regulations, and socio-economic factors involved in the life cycle of plastics, from production and consumption to disposal and waste management. It encompasses various stakeholders, including manufacturers, consumers, policymakers, waste management facilities, and recycling industries, as well as the infrastructures and value chains that support the production, distribution, and disposal of plastic materials. The plastic system encompasses both the physical flow of plastics through society and the socio-cultural, economic, health and environmental implications associated with their use and management. Thus, the complexity of the plastic system is deeply rooted in its economic and sociotechnical structure, presenting significant challenges for regulation and sustainability.

At its core, the economy of plastics operates within a linear model, predominantly reliant on fossil feedstocks for production (3.1.1.4). This linear approach has led to a concerning situation in plastic waste management, with a significant portion ending up in landfills or the environment. Over the past 65 years, at a global scale, only a mere 9% of all plastics ever discarded have been recycled, while 12% have been incinerated, highlighting the ineffectiveness of current waste management practices (Geyer *et al.*, 2017).

Moreover, the plastics system is entrenched within a global network, characterised by inertia and sociotechnical lock-ins (3.1). These structural barriers pose substantial challenges to enacting meaningful change. Despite growing awareness of the environmental and health consequences of plastic pollution, the entrenched nature of the system makes it resistant to rapid transformation. Importantly, human health impact estimations associated with plastics are often conservative, as they focus on a subset of chemicals known to contribute to disease and disability (including phthalates and bisphenols depicted in section 3.4.3.1 regarding preclinical and clinical evidence of human disease promotion and estimated cost of human disease burden), while overlooking many others. This limited scope fails to capture the full extent of health risks posed by plastic pollution.

Plastic leakage and dispersion further exacerbate the complexity of the system, manifesting as structural issues that extend beyond individual actors or regions, revealing the pervasive nature of plastic pollution.

Indeed, plastics not only pose environmental and health risks but also negatively impact prosperity, thereby prompting key players to develop strategies to mitigate these impacts. As outlined by Sen (1999), prosperity encompasses various aspects of human well-being, including health, longevity, access to safe food, mobility, social relationships, and the ability to raise families. Plastic pollution undermines these values on a global scale, hindering the well-being and capabilities of many people around the world. Consequently, recognising the multifaceted impact of plastics on prosperity has spurred efforts among stakeholders to develop strategies aimed at addressing these challenges comprehensively.

3.5.2. Regulation of the system of plastics: a difficult pathway towards sustainability

(based on Chapter V.1 of the extended report)

Plastics have lately been considered as a specific object of regulation. The legal framework regulating plastics in agriculture and for food at a European level has initially focused on enabling the free movement of goods, then progressively evolved to incorporate health and environmental protection. It is now divided into three distinct areas: FCMs, waste management, and chemicals. Over time, the scope of plastics regulation has expanded from a narrow focus on specific life-cycle stages to a more integrated approach. Early regulations targeted most visible plastic objects and waste management but now encompass the complex nature of plastics as materials and mixtures of substances. This evolution highlights the necessity of addressing plastic pollution through systemic solutions that account for various stakeholders and materials.

Despite advancements, plastics regulation still focuses on the most materially and politically visible issues, such as a limited number of chemicals (*e.g.*, bisphenols), plastics iconic objects (plastic bags), and plastics in food processing or packaging. This leaves significant gaps, such as the regulation of polymers, MPLs, and plastics used in agricultural production. Notably, agricultural plastics, though integral to the food value chain, are insufficiently regulated under existing frameworks, leading to environmental and health risks. The regulatory approach to plastics remains predominantly curative, with an emphasis on waste management and recycling rather than preventive measures. Initiatives like the EU Green Deal and circular economy policies prioritise recyclability and the use of recycled plastics but do little to reduce overall plastic consumption. This downstream focus reinforces societal dependency on plastics while failing to address the root causes of environmental degradation associated with their lifecycle.

3.5.2.1. A legal framework driven by economic freedom, health and environmental protection objectives, now divided into food contact material, waste and chemicals regulations

Legal framework drivers evolving from economic freedom protection, to human health and environment protection objectives

The legal framework applying to plastics used in agriculture and for food aligns with a unified set of rules governing all activities and related products on the European market. It is built around three main objectives: safeguarding economic freedom through the free movement of goods, protecting human health, and preserving the environment. Initially focused on economic principles, this framework has progressively evolved to address environmental and health concerns, particularly since the 1980s, reflecting shifting priorities in regulatory policies.

Over time, a specific legal framework for plastics has been structured around three key legal areas: the FCM law, waste law, chemicals (or chemical products) law (Figure 40). These areas of regulation collectively address different stages of the plastics life cycle, integrating two or more of the core protection objectives to ensure a comprehensive approach to their regulation.

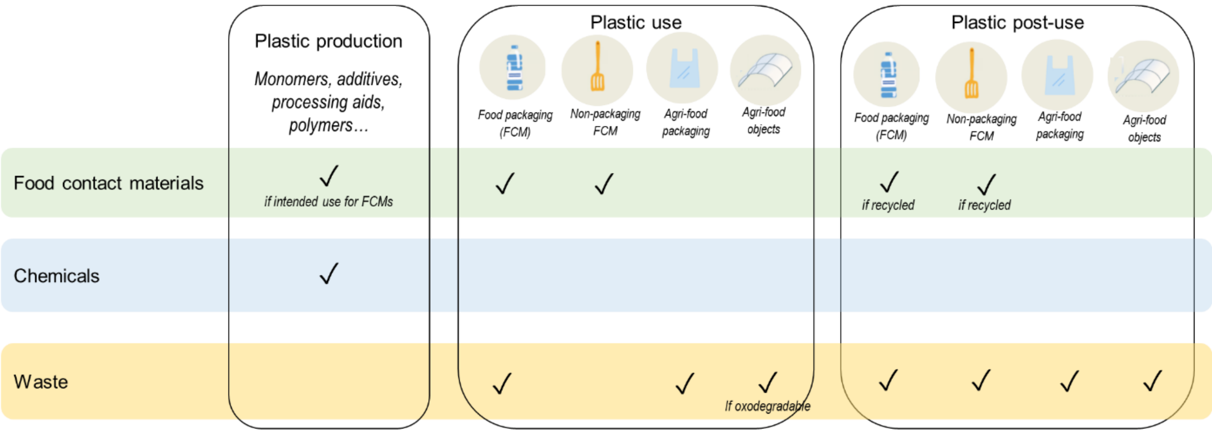

Figure 40. *Legal areas applying to plastic ingredients, plastic objects and plastic waste.*

A framework tending towards covering different stages of the life cycle of plastics

This regulatory framework now aims to address multiple stages of the life cycle of plastics. While regulations initially focused on specific stages of the plastics cycle in isolation, they have evolved to encompass more than one stage in most cases, reflecting a broader and more integrated approach.

Initially, apart from the plastic FCMs, plastic pollution was primarily regarded as an issue related to certain large-scale objects, such as plastic carrier bags with Directive (EU) 2015/720 (2015) and as pointed by Nielsen *et al.* (2020). However, this perspective has shifted, and plastics are now understood as complex materials containing various components (Barrowclough et Birkbeck, 2022). This change in understanding has highlighted that addressing plastic pollution cannot be limited to waste management alone. Instead, it requires a systemic and comprehensive approach that considers the stakeholders, materials, and multiple legal domains involved.

Perception of plastic pollution as a factor of change in regulation

The drivers behind plastics regulation remain relatively underexplored, though some studies have taken an interest to plastic bag regulation processes and try to understand the diffusion of norms and policies. First, plastic bag regulations have an uncommon diffusion pattern, from South to North, with norms emerging independently in the Global South from the late 1990s and spreading globally without a coordinated international movement or agreement (Clapp et Swanston, 2009; Knoblauch *et al.*, 2018).

In the global South, health and safety concerns have been an important factor, because plastic bags are linked with diseases linked to mosquitoes and stagnant water in discarded plastic bags and flooding caused by blocked drains (Clapp et Swanston, 2009). Additionally, the visibility of plastic litter in the global South with inadequate waste management systems has been a significant motivator (Clapp et Swanston, 2009). In contrast, in the Global North, public opinion, shaped by non-governmental organisation (NGO) campaigns and critical media coverage, has been a key driver for plastic bags' regulations.

Over time, the public awareness of plastic pollution has grown significantly, driven in part by the efforts of NGOs and media campaigns that have brought the issue into sharper focus. This increasing concern has pressured governments and international bodies to adapt regulations to address emerging environmental challenges (Knoblauch *et al.*, 2018). Additionally, deficiencies in waste management systems represent another factor for the diffusion of plastic norms and policies, particularly in the Global South, because plastic waste is made visible (Clapp et Swanston, 2009; Knoblauch *et al.*, 2018). This stark contrast in waste handling has underscored the urgency of connecting these localised issues to global regulatory frameworks.

The evolution of public perception and its influence is also evident in international law, where treaties and agreements increasingly reflect a recognition of plastic pollution as a global problem requiring systemic solutions.

3.5.2.2. Plastics lately addressed as specific mixture of chemical substances, material or objects, with evolving definitions, also depending on the legal area considered

Plastics as a mixture of chemical substances, a material or an object depending on the legal area

Plastics were defined relatively late in legal frameworks, and their definition varies depending on the legislative area in question, reflecting the distinct priorities and objectives of each of these.

In the context of FCM regulations, plastics were first defined in 1982. They are considered as materials capable of transferring their constituents into food. The regulations in this area focus on the composition of plastic materials and the migration of their constituents into food products.

In waste regulations, including packaging regulations, plastic waste was initially not distinguished from waste of other materials. In this area, the first legal definition of plastic appeared in 2015, within the directive on plastic bags (European Union, 2015) (an amendment to the packaging and packaging waste directive (European Commission, 1994)). Plastics were viewed as macro-objects destined to become waste.

In chemical regulations, and in REACH (European Commission, 2006), Registration, Evaluation, Authorisation and restriction of chemicals) in particular, plastics are approached through the lens of their constitutive ingredients. While the plastic materials resulting from this mixture of ingredients are not explicitly defined, their monomers and polymers are, and the other compounds are classified under the term ‘substance’.

An evolving (plastic) definition of plastics, also depending on the legal area considered

The definition of plastics has evolved over time and across different legal areas, making it challenging to determine what is regulated and whether certain materials are classified as plastics (Table 20). Two examples highlight the complexity of these definitions. In the case of silicones under FCM regulations, the initial definition of plastics explicitly included silicones, which were not considered elastomers at that time, while elastomers were excluded. However, in 2001, silicones were reclassified as elastomers and consequently excluded. Today, under Regulation (EU) 10/2011 (2011) on plastic FCMs, silicones are the only elastomers explicitly excluded from the directive's scope.

For coatings, the clarity of their definition depends on the regulatory context. In the FCM framework, coatings are ostensibly not covered under Regulation (EU) 10/2011 (2011), but the specifics are difficult to untangle. The introduction of multi-layered and multi-material structures in 2011 raised questions about whether certain coatings, such as those derived from paraffin or wax, varnished or unvarnished regenerated cellulose films, or paper and cardboard modified by adding plastics, fall under the regulation. These materials, previously excluded, are not explicitly addressed in the updated regulation, adding to the ambiguity. In the context of waste regulation, the definition of coatings is somewhat clearer in the SUP directive, which explicitly includes products partially made of plastic within its scope. This broader definition provides more certainty about what is considered a plastic under waste regulations, contrasting with the less definitive stance in the FCM context.

Table 20. *Evolution of plastic legal definition in major European directives since 1982.*

	Directive 82/711/EEC (1982)	Directive 90/128/EEC (1990)	Directive 2001/62/EC (2001)	Directive 2002/72/EC (2002a)	Regulation (EU) 10/2011 (2011)
Definition of plastic	= the organic macromolecular compounds obtained by polymerisation, polycondensation, polyaddition or any other similar process from molecules with a lower molecular weight or by chemical alteration of natural macromolecules. Other substances or matter may be added to such macromolecular compounds				= polymer to which additives or other substances may have been added, which is capable of functioning as a main structural component of final materials and articles
Elastomers	-	-	-	-	? (except for silicones and rubbers)
Silicones	+ Because not considered as elastomers at the time	+	- Because considered as elastomers	-	-

3.5.2.3. Plastics regulation focuses on the tip of the iceberg

Plastics regulation focuses on the ‘tip of the iceberg’, addressing only the most visible aspects, such as well-known chemical substances (rather than polymers), large and easily identifiable macro-objects, or plastics used in FCMs at the ‘process and package’ stage of their lifecycle, leaving many underlying issues unaddressed.

A large proportion of plastic-related chemicals, including hazardous ones, are unregulated

Several studies estimate between 10,000 and 16,000 chemical substances are found in plastics in general, and that at least a quarter of these substances are of concern due to their persistent, bioaccumulative, mobile, and/or toxic properties (Wiesinger *et al.*, 2021; Wagner *et al.*, 2024). However, only a small fraction of these substances are currently regulated at international level (Wagner *et al.*, 2024). Some authors emphasise that both research and regulation tend to focus on well-known molecules, such as phthalates and bisphenols, while the majority of chemicals remain insufficiently studied (Wiesinger *et al.*, 2021). For these lesser-known substances, there is a significant lack of data on their physicochemical properties, making it difficult to assess either their risks or their potential dangers effectively.

It is worth noting that, to date, some substances have been classified as endocrine disruptors under the so-called CLP regulation (1272/2008 (2008b) on the classification, labelling and packaging of substances and mixtures). The list of molecules identified as potential endocrine disruptors is not exhaustive: some molecules that are still authorised may be banned in the future (the evaluation procedures for some molecules are ongoing and very lengthy). In fact, it is important not to misunderstand the term ‘endocrine disruptor-free’, as this term actually covers molecules with endocrine disrupting potential that are regulated under the CLP regulation. The other substances that require vigilance, in addition to those tested under the current regulations, can be found in Anses (2021).

Polymers are overlooked by regulations, for debatable reasons

Polymers are excluded from REACH regulations due to economic and technical feasibility concerns. Given the complex nature of polymers and their vast diversity, regulating them under REACH would require significant resources and pose considerable challenges. As a result, polymers are not subject to specific regulations, and their inclusion in broader chemical regulations remains limited. This exclusion stems from the practical difficulties of managing such a large and varied group of substances within a single regulatory framework, although their inclusion is currently under discussion to date.

More broadly, polymers are not the subject of specific regulations. For example, Regulation (EU) 10/2011 (2011) on plastic FCMs justifies their exclusion by stating that, due to their high molecular weight, polymers are unlikely to be absorbed by living organisms, thus presenting minimal potential risks to human health. This assumption has shaped the regulatory approach, reinforcing the idea that the risks associated with polymers are not significant enough to warrant detailed oversight.

Consequently, much of the focus in plastics regulation is directed at smaller molecules or additives, which are considered more likely to pose immediate health risks (although this is debatable, see 3.4).

Waste regulations focus on a handful of macro-plastic objects, overlooking other forms of plastic

Waste regulations have primarily targeted the most visible MaPL objects, beginning with plastic bags under Directive 2015/720 (European Union, 2015) and later expanding to include around ten common SUP items under Directive 2019/904 (European Union, 2019a). These regulations focus on consumer products that are part of daily life and make plastic pollution more tangible to the general public (Nielsen *et al.*, 2020). By addressing these highly visible items, the regulations aim to tackle sources of pollution that are easily recognised and widely understood by population.

However, this focus has rendered other types of plastic pollution less visible. Objects from less prominent sectors, along with MPLs and NPLs, receive less regulatory attention despite their significant environmental impact (3.4). These less apparent forms of plastic pollution are often overshadowed, highlighting a gap in the current regulatory approach that risks overlooking critical contributors to the global plastic dispersion in ecosystems and living organisms.

Regulations do not take account of plastics in the production part of the food value chain, even though they may come into contact with food

Plastics used in the upstream phase of the food value chain, in agriculture processes, are not comprehensively addressed by current regulations, even though they may come into indirect contact with food (Table 21). Since their introduction in the 1960s, agricultural plastics have not been as tightly integrated into the regulatory framework as plastics used directly in food processing or packaging.

Especially, food, as defined in the general food legislation, excludes crops prior to harvesting and livestock prior to slaughtering from its definition, meaning the rules on FCMs do not apply to agricultural plastics. Similarly, FCM regulations exclude feed, further limiting oversight in this area.

When agricultural plastics become waste, they fall under general provisions of Directive 2008/98 (European Commission, 2008a), which governs waste management. However, a specific EPR scheme, ADIVALOR, has been established to address agricultural plastics. Additionally, the SUP Directive (European Union, 2019a) bans oxo-degradable plastics, including those used in agriculture, signalling some regulatory attention to these materials.

In the context of chemical regulations, general rules apply to agricultural plastics, but since 2019, REACH has introduced restrictions (completing the measures taken by Regulation (EU) 2019/1009 (2019b)) on intentionally added MPLs in products such as controlled-release fertilisers, microencapsulated PPPs, and treated seeds. These measures will be enforced gradually between 2028 and 2031.

Despite these developments, broader frameworks like the Common Agricultural Policy (CAP) through the National Strategic Plans under CAP, as well as the regulations on organic farming, fail to address the issue of plastics in agriculture. This limited integration of agricultural plastics into existing regulation stands as an important regulatory gap.

Table 21. Inclusion of plastics used in agriculture in European main regulation areas.

	FCM		Waste			Chemicals
	R.1935/2004	R.10/2011	D.94/62	D.2008/98	D.2019/904	REACH
Plastics used in agriculture	No	No	Yes (general provisions)	Yes (general provisions)	Yes, for oxodegradable plastics	Yes (general + micro-plastics)

3.5.2.4. A legal framework for the regulation of plastics used in agriculture and for food that is mainly curative

The legal framework for regulating plastics, including those in the food and agriculture sectors, is primarily curative rather than preventive. While prevention, in the sense of reducing plastic waste production, is a stated priority in Framework Directive 2008/98/EC on waste (2008a) (Figure 41), most regulatory efforts focus on managing waste after their creation. These efforts include improving waste collection and recycling systems to prevent plastic from entering the environment, the emphasis remains disproportionately placed on curative measures, highlighting a gap between regulatory intent and implementation.

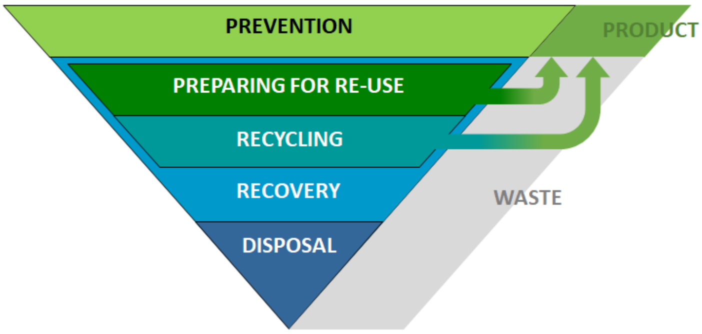

Figure 41. *Waste hierarchy among prevention and waste management options as it should apply to plastics. (adapted from https://environment.ec.europa.eu/topics/waste-and-recycling/waste-framework-directive_en, and definitions from Directive 2008/98/EC on waste (2008a))*

Focus on waste management and recycling in particular

The regulation of plastics largely focuses on the downstream stages of their lifecycle, particularly post-use waste management and recycling (Nielsen *et al.*, 2020; Barrowclough et Birkbeck, 2022). This emphasis overshadows the upstream steps, notably the production of virgin plastics, allowing the environmental impacts of their creation to remain underexamined, and aligning with the industry's interests. Policies like the EU Green Deal and the EU Strategy for Plastics in a Circular Economy for instance put forward recycling and recyclability targets. In the case of packaging, Directives adopted or amended according to these strategies encourage the incorporation of recycled materials (Packaging and packaging waste Directive or the SUP Directive (European Union, 2019a)), therefore promoting 'better packaging', rather than reducing overall plastic use in packaging (Peyen, 2023).

These downstream regulatory approaches perpetuate the ubiquity of plastics in society without significantly reducing their environmental footprint or challenging their central role in the food value chain. They align with the industry's interests by deflecting the attention from the production of virgin plastic (Nielsen *et al.*, 2019). Circular economy policies, by focusing on the integration of recycled materials, often reinforce the existing plastic system instead of promoting a substantial reduction in consumption, leaving the broader issue of plastic dependency unaddressed.

Additionally, as pointed in 3.5.2.1, while regulations attempt to address health concerns and tackle plastic reduction, these measures do not resolve the issue of recyclability. On the contrary, the use of more complex, multi-layered materials as a result of these regulations often complicates recycling processes, leaving a significant gap in achieving sustainable solutions.

No cap on the production of plastics

European regulations on plastics in the food value chain set ambitious targets but fail to fundamentally challenge the role of plastics or aim to reduce their production.

As plastics were initially not distinguished from other materials, recycling targets by weight set in the Waste Framework Directive (European Commission, 2008a), for instance, can be met without curbing overall plastic output. Now that specific plastic streams are incorporated into directives, targeted actions on plastics are needed. However, these actions often consist of packaging optimisations, such as thinner designs or combining thin plastic layers with other materials, which shift rather than solve the problem. In some cases, regulatory goals encourage substituting existing plastic products with alternatives that still incorporate plastics, although in reduced quantities. For example, the SUP Directive (European Union, 2019a) sets consumption reduction targets for some SUP items, such as plastic cups. However, as those reduction targets are set by weight and as it is possible to consider only the plastic part in the calculation of the attainment of the target, the targets can be reached with optimised designs, like paper cups coated with plastic. These composite materials, frequently presented as alternatives, are often a more complex material that still contains plastic and perpetuate its use and associated challenges.

In France, the AGEC law (République française, 2020) introduced a unit-based sales indicator in 2023 to address this limitation, although its effectiveness in reducing plastic production is yet to be assessed. Furthermore, regulatory bans focus on restricting certain plastic products from entering the market but rarely address production limits, likely due to economic considerations that prioritise industry growth over health and environmental concerns. As a result, production can continue, often for export purposes, even when domestic use is restricted. Lastly, while the SUP directive (European Union, 2019a) aims to reduce SUPs, it does not ban their replacement with other disposable materials that may pose similar risks: this is the case for the replacement of disposable plastic cups by disposable paper cups with plastic coatings. This approach sustains a culture of single-use consumption without questioning or altering underlying practices.

By failing to impose a cap on plastic production or demand substantial shifts in consumption habits, these measures may ultimately maintain the status quo. They emphasise managing plastics post-production rather than addressing systemic issues like overproduction and overreliance on disposable goods, leaving the broader environmental and societal impacts of plastics unchallenged.

Preventive measures taken on behalf of the precautionary principle? Regulation and scientific knowledge

Current regulatory measures for plastics are largely grounded in the principle of prevention, which relies on demonstrated risks supported by hazard and exposure data. For example, REACH (European Commission, 2006) imposes restrictions only when risks are scientifically established. While this approach provides a structured response to known threats, it leaves gaps in addressing hypothetical or emerging risks that may not yet be fully understood or quantified.

Adopting preventive measures under the precautionary principle could offer greater protection by proactively managing uncertain or potential risks. This approach prioritises safety and environmental health even in the absence of complete scientific certainty. While there are precedents, such as France's temporary restrictions on BPA based on NGO studies or foreign research while awaiting formal risk assessments, the precautionary principle remains underutilised.

From the legal literature point of view, shifting towards precautionary principle could enhance regulatory effectiveness, enabling swifter responses to potential dangers and fostering a more robust safeguarding of public health and the environment.

3.5.2.5. Corporate leverage on regulations

Corporate influence on plastic regulation is clearly documented in the literature through classic lobbying tactics and strategic diversions. Economic studies suggest industries with higher profits and more polluting activities are more likely to form lobby groups. Mah (2022) underlines this, noting industry actively opposes restrictions on SUPs and uses global events such as the COVID-19 pandemic to boost SUP usage. Additionally, commercial marketing can be considered as another lobbying strategy, flooding markets with diverse plastic products that make recycling efforts more difficult.

Literature also points that industry employs diversionary tactics to promote weaker regulations. One common strategy shifts public attention to final consumer responsibility, focusing on waste management rather than addressing plastic production issues (Bartow, 2001; Chamayou, 2018). Another diversion promotes recycling, even though supply cannot meet demand, thereby prolonging the production of virgin plastic (Gontard et Seingier, 2020; Mah, 2022; O'Neill, 2022). Lastly, industrial actors often promote the concept of circular economy (see below) through voluntary commitments, which, while promising, often lack ambitious targets or binding regulations (Corvellec *et al.*, 2022). The sustainability of plastic packaging is, for instance, promoted by packaging industry that, by its very nature, defends a strategy of endless growth of its activities. Industrial discourses underscore, for instance, reduction of energy use for transportation, prevention of food contamination in distribution processes or extension of shelf life, while downplaying health and environmental issues associated with plastics, by treating them essentially as a waste management issue (Lisiecki *et al.*, 2023).

While the literature consistently highlights corporate influence on plastic regulation, empirical evidence remains often difficult to obtain, notably because of limited access by researchers to industrial strategies through interviews or access to archives. In the case of France, which is concerned by this CSA, it is notable that no study explores the role of French key players of plastic industry in the manufacture of the law. Finally, other influencing factors may be overlooked, in particular the role of public expertise and science in regulatory decision-making, especially in the production of public expertise that leads to regulatory processes. While the thesis of corporate influence and lobbying suggests that public risk management is dominated by industry, this view oversimplifies the situation, given the significant contribution of public expertise to plastics regulation. The lack of social studies dealing with the interaction between public and corporate expertise means that it is not possible to assess how these dynamics influence the effectiveness and direction of regulatory frameworks.

3.5.3. The assessment of plastic sustainability in decision-making processes is too reductive, leading to the implementation of irrelevant mitigation strategies

(mostly based on Chapter V.2 of the extended report)

3.5.3.1. Distorted, blurred and misunderstood approaches: sustainability of the system of plastics is reduced to a promise to improve the circularity in economy, through recycling processes

Sustainability applied to the system of plastics remains a controversial issue. Regarding to the United Nation definition, sustainable development should encompass three objectives simultaneously:

(environmental quality, economic prosperity and social equity) 'without compromising the ability of future generations to meet their own needs' (World Commission of Environment and Development, 1987).

The issue stems from a common tendency among economic and regulatory practioners — including decision-makers — to reduce sustainability to circular economic practices. This narrow view prioritises recycling as a means of fostering economic growth, often at the expense of environmental protection and human health, by sidelining strategies focused on reducing and reusing plastics (Ghisellini *et al.*, 2016; Johansen *et al.*, 2022; Trasande, 2022; Dörnyei *et al.*, 2023). However, this recycling-centric approach contradicts the inherent hierarchy of the circular economy's 4-R framework, which unequivocally places reducing the consumption and production of plastics as the most critical step (Sihvonen et Ritola, 2015; Van Buren *et al.*, 2016; Kirchherr *et al.*, 2017; Potting *et al.*, 2017).

Kirchherr *et al.* (2017) revealed that two-thirds of the 114 circular economy definitions they analysed omit to address waste hierarchy. They showed that while some authors mistakenly equate the circular economy solely with recycling, others adopt a more comprehensive approach, emphasising the 'reduce, reuse, recycle, recover' framework. Notably, circular economy practitioners often omit 'reduce' from their definitions, probably because they have little interest in promoting a strategy that may limit consumption and economic growth.

While circular economy is broadly understood as a pathway to move on from a linear economic model of production-consumption-disposal, the prominence of economic prosperity in the definition of circular economy shows that its link to sustainability objectives remains weak. There is thus a need to harmonise understanding of circular economy integrating all three dimensions of sustainability – environmental quality, social equity and economic prosperity – beyond the sole interests of economic practitioners.

3.5.3.2. Prioritising plastic recycling over plastic production reduction is a counterproductive mitigation strategy as…

… plastic recycling is a mid-stream curative action to manage plastic waste

Despite the focus on recycling in many discussions of the circular economy, recycling alone is insufficient for achieving sustainability (Trasande, 2022). The technical limitations of recycling hinder its actual performances (Lau *et al.*, 2020; Volk *et al.*, 2021; Mayanti et Helo, 2022): (i) in practice, material loss (estimated by Shonfield (2008) at 15% of the weight after being manually sorted) occur during the recycling process due to shredding, separation, washing and extrusion (Mayanti et Helo, 2022); (ii) after several recycling loops, the material quality is so degraded that it completely loses its properties (Volk *et al.*, 2021), which implies, for example, that plastic covers used in horticulture are open-loop recycled to produce various products other than plastic covers, or simply incinerated (Montero *et al.*, 2012); and (iii) since the very first loop, and even if decontamination processes are eliminating known chemical substances, recycled plastic for food contact is still most of the time impossible or potentially dangerous for public health because plastic can be chemically or microbiologically contaminated, in particular by NIAS, thus contaminating food products with which it comes into contact (3.3.3.3).

In that sense, recycling does not prevent the production of raw plastics and is therefore a curative action to manage waste rather than a preventive one.

… plastic recycling cannot overcome the ever-increasing plastics consumption, mismanaged plastic waste and plastic pollution

Relying on the recycling of plastic waste as an overriding mitigation strategy is insufficient and even counterproductive to limit plastic pollution and achieve the sustainability of the system of plastics.

As a result of well-known rebound effect or Jevons paradox where resource efficiency gains lead to increased resource consumption (Lange, 2021), excessive reliance on recycling paradoxically leads to an increase in plastic production and consumption. The use of plastics in the food chain is often justified by the potential for proper management of plastics at the post-consumer stage, once they become waste. This rationale is linked to consumer perceptions associating recycled products to a so-called sustainable consumption: the expectation is that effective waste management and recycling systems can mitigate the environmental impact of plastics, thereby encouraging their continued use in food chains, and thus their consumption (see also Chapter II.1 of the extended report).

This was observed for plastic packaging and small non-packaging plastics item in Europe and in France in particular when comparing 2018 and 2012: despite improvements in waste collection coverage, recycling, and disposal among European Member States, the total annual amount of plastic waste generated continued to rise as a result of the exponential growth in total annual plastic production (Winterstetter *et al.*, 2023). In turn, the annual amount of plastic waste that is mismanaged continued to escalate, mismanaged waste corresponding to waste that is not collected, lost from collection, littered or not adequately disposed of in non-sanitary landfills. Such results contradict regulation efforts like the EC circular economy package (2014) (Chapter V.1.1 of the extended report), which aims to reduce plastic waste through recycling and point at the urgent need to prioritise plastic production reduction over recycling strategy, thereby limiting environmental impact of plastic production, use and waste (Winterstetter *et al.*, 2023).

… mechanical recycling of plastics is limited in practice

Mechanical recycling is the predominant method employed in plastic recycling, yet its success is hindered by challenges (3.3.1.2, 3.3.2 and 3.3.3) such as heterogeneous composition of waste (mixed collected waste in general and also mixed types of plastic waste); composite, multi-layered or coated plastics including functional barriers, plastic polymer blends, flexible and coloured plastics (3.3.1); polymer degradation during its lifespan and processing (Huysveld *et al.,* 2022); or even stringent regulations in the food industry (Regulation (EC) 1935/2004 (2004) applied to FCMs).

In the vast majority of cases, plastic recycling remains open-loop recycling and the only effective closed loop recycling process identified for plastics used in agriculture and for food concerns a single-use product: PET bottle-to-bottle process, developed since the end of the 1990s (Franz and Welle (2022); 3.3.1, 3.3.3). Interestingly, and regarding projections of PET production indicating that from US$ 52.94 billion in 2024 the global PET market size will grow to US$ 109.63 billion by 2032, growth in virgin PET production does not seem to be challenged by growth in recycled PET production (FBI, 2025b).

… profitability of mechanical recycling is debated

The economic viability of mechanical recycling as a solution to halt or limit plastic pollution is a topic of debate. On the one hand, in some specific conditions (achieved with a complex mechanical recycling system - including several waste sorting steps but not with simple mechanical recycling) and if recycling is operated and funded by public authorities or strongly regulated, estimations from Faraca *et al.* (2019), focused on Belgian context, and Volk *et al.* (2021), focused on German context, suggest economic viability of mechanical recycling (but not profitability). On the other hand, actual mechanical recycling clearly appears as economically challenging, especially for mixed plastic packaging waste: for plastic types other than rigid PET and HDPE, Larrain *et al.* (2021) showed mechanical recycling is currently unprofitable (this encompasses the following mixed plastic packaging waste: PP bottles and trays, PS trays, PE films and mixed polyolefin rigids (MPO rigids, which is a mix of PP and HDPE). The economic viability of mechanical recycling projects is influenced by factors including oil prices, market demand for recycled plastics, and government policies (Larrain et al. (2021); see also I.3.1.3).

Thus, reliance solely on market forces to drive progress in mechanical recycling is deemed insufficient. Without appropriate public policies and incentives, investors may find recycling ventures unattractive, further impeding progress in waste management efforts. Key policy measures to bolster the profitability of recycling include investments in advanced waste sorting technologies, climate policies to stabilise oil prices, and initiatives to ensure the cleanliness of plastic waste prior to recycling. As Tashkeel *et al.* (2021) explained, ’additional strategies beyond conventional recycling are necessary to improve circularity**,** including government intervention (implementing innovative policies such as environmental taxes, subsidies, and regulations to encourage companies to adopt circular practices)’.

… chemical recycling remains an unfulfilled promise

Chemical recycling is aimed to support mechanical recycling by recycling plastic waste, which cannot be easily mechanically recycled (*e.g.*, multi-layered, coloured or contaminated plastics). However, despite being an emerging sector with significant actual investments, it faces challenges in scaling up to meet recycling targets set by the EU (Davidson *et al.*, 2021).

Pyrolysis, a process that breaks down polymers at high temperatures, is a promising and mainly used chemical recycling technology. It produces pyrolysis oil, a valuable feedstock for various applications, including plastic manufacturing. While the global market for pyrolysis oil is still developing, it offers significant economic potential. Ma *et al.* (2023) estimated that chemical recycling of plastic waste in the USA could produce 6 Mt of virgin PE per year, worth an estimated US$ 13 billion annually. However, chemical recycling is more sophisticated and cost-intensive than traditional waste management technologies. Ma *et al.* (2023) simulated various chemical recycling scenarios in the upper Midwest region of the USA. Their results demonstrate that chemical recycling when combined with mechanical recycling, offers significant economic benefits, generating substantial revenues and profits. However, the profitability of chemical recycling can vary depending on factors such as the type of plastic waste processed, the products produced, and market conditions.

As in the case of mechanical recycling, the prevalence of open-loop recycling in chemical recycling can be explained by its economic viability that outperforms closed-loop recycling. But, Larrain *et al.* (2020) compared the economic performance of open-loop and closed-loop pyrolysis of polyolefin waste in Belgium, the Netherlands, and Luxembourg. They showed that open-loop pyrolysis, which produces both naphtha and wax, was economically more interesting due to the higher value of wax, yielding a net present value 34% greater than that of closed-loop recycling, which produces naphta only.

Larrain *et al.* (2020) examined the economic risks tied to oil price fluctuations and feedstock availability. Their model indicated that for both open-loop and closed-loop chemical recycling methods to remain profitable, oil prices must exceed €550-575/t (or €75-78/barrel). If plastic waste feedstock costs doubled, profitability would require oil prices to rise further to €625-653/t (€85-89/barrel). As an illustration, across the last year (from October 2023 to September 2024), oil prices varied from €67/barrel to €84/barrel (for the Brent Crude Oil). Moreover, open-loop recycling displayed greater economic resilience, remaining profitable even with reduced feedstock availability, while closed-loop recycling is depending on full feedstock capacity to avoid financial losses.

On a lab scale experimental basis, and in scenarios with innovative approaches such as alkaline hydrolysis showing high PET conversion rates, especially for challenging waste streams such as multi-layered or coloured plastics, research indicates that chemical recycling has a lower impact, in terms of resource consumption and terrestrial acidification, than traditional mechanical recycling (Ügdüler *et al.*, 2020; Huysveld *et al.*, 2022). However, in terms of global warming impacts, mechanical recycling gives better results than chemical recycling in all cases (Huysveld *et al.*, 2022). It is thus important to consider overall environmental impact, including energy consumption, water usage, and emissions from the production process (Ma *et al.*, 2023).

While chemical recycling holds promise for diversifying waste processing pathways and generating profits, its industrial implementation is still in its infancy. Challenges such as fluctuating oil prices and uncertainties surrounding market demand pose risks to the profitability of chemical recycling ventures.

3.5.3.3. Life cycle analysis as a hegemonic but limited tool: sustainability, including environmental impacts of the system of plastics, cannot be assessed and decisions cannot be made solely on its basis

Life cycle analysis (LCA) of plastics products does not provide a satisfactory assessment of sustainability or even of the environmental impacts themselves. Sustainability cannot be addressed solely on the basis of LCAs in decision-making processes considering what is not taken into account (social or economic impacts) or poorly taken into account (ecosystem quality, human health). Full awareness and transparency on LCA limitations are needed to properly support decision-making processes.

France is underrepresented in research utilising LCA-based decision-making tools, with most European scientific output in this field originating from Italy and Spain. Considering the selected corpus, environmental impact assessments of plastics predominantly focus on food packaging (77% of the articles), while only 19% of the articles address agriculture. Only **three studies (4%) consider the full farm-to-fork perspective, none of which were conducted in France.**

LCA has emerged as a prominent tool for the environmental assessment of plastics products. LCA is used to assess changes on the way plastics are sourced, produced, distributed, consumed and disposed of. However, there is a notable imbalance in the criteria considered in LCAs of plastics used in agriculture and for food.

By far, the most considered criteria were climate change and resources availability, with global warming potential (GWP) and fossil depletion potential (FDP) as the most evaluated impact categories, respectively.

However, these criteria are insufficient to assess impacts on continental ecosystems (3.4). Ecosystem quality has only recently been taken into account and is still a much less studied criterion. In particular, land use change, including for the production of bio-sourced plastics, biodiversity loss and consequences on ecosystem functions and services are not adequately addressed in LCAs. Moreover, LCAs do not take into account the accumulation of plastics in all their forms in environmental compartments and their long-term consequences.

Efforts are being made to extend the scope of LCA beyond environmental assessment and integrate human health impacts of plastics used in agriculture and for food. For example*,* Deeney *et al.* (2023) extracted and converted data on human health impacts from 49 LCAs into a single metric representing the loss of the equivalent of one year of life of full health.

However, because, by definition, LCAs are focused on environmental assessment of plastics, social and economic assessment are still absent from these analyses. Therefore, claims of sustainability assessment of plastic products using LCA are misleading. Concerning plastics in food value chains, Life Cycle Cost Analysis (LCC) and Social Life Cycle Analysis (S-LCA), are lacking.

Results and information drawn from LCAs vary according to the impact criteria and sustainability dimensions considered, as well as system boundaries considered, which makes it further difficult to interpret, compare and generalise their results. Not all LCAs cover the entire life cycle of plastic products. For example, cradle to grave LCAs cover raw material extraction (cradle) to waste treatment (grave) when cradle to gate LCAs cover only a portion of the life cycle from raw material extraction (cradle) to production in the factory (gate), *i.e.*, before transportation to the consumer. In addition, boundaries are either attributional or consequential. An attributional boundary focuses solely on the materials and energy directly associated with a specific product, whereas a consequential boundary accounts for indirect impacts that may arise from changes caused by the product's adoption (Miller, 2022).

The current lack of transparency along value chains also limits the reliability of plastic related LCAs results. Mainstream food value chains are global and long, involving multiple intermediaries. LCAs associated with these value chains require extensive and detailed information on inputs, outputs, and transportation routes, which remain difficult to obtain and model accurately.

LCAs do not sufficiently account for the context-dependent nature of impacts of food value chains. The literature review showed that LCAs of plastics used in agriculture and for food in France are scarce, most LCAs focusing on Spanish and Italian contexts. Yet, environmental impacts of agricultural activities can differ greatly based on factors such as location, climate, soil type, and management practices. Accounting for this variability in LCA models and incorporating territorial approaches is challenging, as it requires advanced modelling techniques and extensive data collection.

Finally, not only is the geographical context inadequately depicted in LCAs, but also parameter uncertainty and time. LCA methods are predominantly retrospective, relying on data from well-established industries with mature value chains. This is particularly challenging when assessing disruptive technologies for which historical data are lacking.

3.5.3.4. Design for sustainability promises are hampered by current critical limitations

DfS approaches as pathways to reach circular economy for plastics in agriculture and food also appear limited notably because the scale of the material, or even of the product or the processes alone are not yet sufficient to assess the sustainability of the plastic system as a whole.

Design for sustainability (DfS) is defined by Rocha *et al.* (2019) as a holistic design approach, developed since the early 1990s, that enables to integrate and assess the sustainability dimensions in different stages of the product or product-service development process towards the required scale of incremental innovations and/or systemic transformations. The integration of the sustainability dimensions in the design of a material or an industrial product has simultaneously evolved with the integration of the green design, biomimicry, cradle to cradle design, eco-design, to design for sustainability principles (Ceschin et Gaziulusoy, 2016). DfS initiatives are still at an emerging stage and mainly focused on the material product levels (*i.e.*, eco-design) and technocentric approach; they focus on incremental innovations rather than systemic transformations. Finally, because DfS initiatives are mainly based on LCAs results, the LCAs limitations that have already been detailed above also apply to DfS initiatives.

3.5.3.5. Deposit Return Schemes are recycling-oriented where they should be reuse (*réemploi*)-oriented

DRSs have proven to be an effective strategy for significantly increasing the collection and recycling rates of plastic waste. Kahlert and Bening (2022) found that EU countries implementing DRSs achieve an average collection rate of 90.8%, compared to 46.5% in countries without such schemes. This substantial difference highlights the potential of DRSs to enhance recycling efforts.

DRSs not only support recycling but also offers significant benefits when oriented toward reuse (*réemploi*), as opposed to recycling. Given the limitations of plastic recycling — such as the limited number of recycling cycles plastics can undergo and the environmental impact of recycling processes as detailed before — DRSs aimed at reusing (*réemployer*) containers can yield greater environmental benefits (Mazhandu *et al.*, 2020).. For example, refillable plastic and glass containers can be reused multiple times, with some glass containers being reused up to 50 times (Agnusdei *et al.*, 2022). This approach reduces the need for new materials and minimises the environmental footprint of packaging.

However, the effectiveness of DRSs for reuse (*réemploi*) depends on various factors, such as the distance between collection and washing facilities. Cottafava *et al.* (2021) showed that for reusable (*réemployable*) PP cups, the environmental benefits are maximised when the washing facility is within 33 kilometres. For glass cups, this distance is even shorter, at 15 kilometres. These considerations are crucial for designing DRS approaches that are both environmentally and economically viable.

Moreover, Lu *et al.* (2022) compared DRSs for recycling versus for reuse (*réemploi*) using plastic trays in Sweden and found that reuse (*réemploi*) scenarios offered better cost-benefit ratios. The reuse (*réemploi*) of trays generated benefits 1.67 times higher than the costs, whereas recycling scenarios resulted in benefits lower than the costs. This suggests that DRSs aimed at reuse (*réemploi*) can be more economically advantageous, particularly for sectors like hotels, restaurants, and cafés, which benefit from reduced packaging costs.

However, it is important to note potential challenges and limitations. Some environmental groups point that recycling-oriented DRSs may inadvertently encourage continued consumption of plastic bottles by reinforcing the idea that purchasing plastic is acceptable as long as it is recycled, potentially maintaining the plastic production cycle rather than reducing it (Cabot, 2024). Furthermore, the success of DRSs depends on the willingness of retailers and producers to participate, with some small retailers facing significant cost increases (Dāce *et al.*, 2013).

In conclusion, while DRSs have proven effective in enhancing recycling rates, its greatest potential lies in reuse (*réemploi*) schemes, which offer superior environmental and economic outcomes at least in the specific contexts that have been studied. However, the design and implementation of DRSs must be carefully considered to avoid unintended consequences and to maximise environmental benefits. More generally, the question of reuse seems to be the poor relation in sustainability research, as these issues are not given much attention in the literature.

3.5.4. Strategies of reduction: reliable options have yet to be found

As many scientific papers have already pointed out the necessity to combine curative strategies to preventive ones, Lau *et al.* (2020) emphasised the urgent need for coordinated actions combining pre- and post-consumption solutions to curb the rising trend of environmental plastic pollution. Their analysis suggests that reducing plastic production by 47% by 2040, along with substituting plastics with alternative materials, could significantly mitigate plastic pollution in terrestrial and aquatic ecosystems. They also advocate for additional measures like ocean, river, and beach clean-ups, as well as improving the collection, sorting, recycling, and disposal of plastic waste. Other researchers (*inter alia*, Richon *et al*. (2023); Griffin *et al*., (2024)) suggest complementing these strategies with responsible use of ocean and river clean-up technologies to address plastic debris already present in the environment, though Lau *et al.* (2020) do not advocate for this approach. Cordier *et al.* (2024) reviewed these findings and estimated that achieving zero plastic pollution by 2040 could cost US$18.3-158.4 trillion globally. This cost, which includes a 47% reduction in plastic production, is substantial – equivalent to the GDP of China at the lower-bound estimate or 1.6 times the world GDP.at the upper-bound estimate. However, the estimated costs of inaction range from US$13.7-281.8 trillion, suggesting that inaction could potentially be more costly than taking proactive measures.

Cordier *et al.* (2024) also analysed the net benefits of these actions, estimating that global income from plastic product sales could reach US$38 trillion from 2016 to 2040 if no actions are taken, and US$32.7-33.1 trillion if the proposed measures are implemented. The net benefits, calculated as income minus costs, show a wide range, with possible outcomes varying from a net loss of US$120.4 trillion to a gain of US$19.7 trillion if actions are taken, and from a loss of US$243.8 trillion to a gain of US$24.3 trillion if no actions are taken. These estimates suggest that while both scenarios could deliver positive outcomes at the higher estimates, the potential costs of inaction might be twice as high as those of taking action.

3.5.4.1. Substitution: most of the studied alternatives to plastics are… plastics, and not alternative material or practices

The principle of substitution of a material by another is dominant in the literature dedicated to sustainability approaches (LCAs, DfS, etc.). Nevertheless, in LCA studies on plastics used in agriculture and for food, plastics were scarcely compared to actual alternatives, whether in terms of materials (*e.g.*, glass to package drinks) or practice (*e.g.*, crop residues to mulching films).

The vast majority of scientific publications consider bio-based and/or biodegradable plastics as alternatives to plastics although these are obviously plastics. Over recent years, there has been a notable increase in research and development focused on bio-based and/or biodegradable polymers as substitutes for petroleum-based plastics.

There is a noticeable bias in publications supporting the development of bio-based plastic formulations over conventional petroleum-based plastics like PE, since the former represent only 1.5% (0.08 out of 5.51 Mt), 1.5% (0.8 out of 54 Mt) and 0.7% (3 out of 413.8 Mt) of plastics produced in France, Europe and worldwide, respectively (Table 1). This preference may stem from the pressure to publish, which often pushes researchers towards perceived innovations, especially in the field of bio-based polymers. However, these perceived innovations trying to reach more sustainability have paradoxically been found to raise economic (high investment and operating costs), structural (lack of standards and guidelines), psychological (political risks and financial concerns due to uncertainty), and behavioural tensions (conflicting views between actors - governments, industries, consumers) when it comes to their use and management by companies (Turkcu *et al.*, 2023).

In a large proportion of scientific papers, the term 'biopolymers' encompass both bio-based and/or biodegradable polymers. In these papers, 'biopolymers' properties are compared to petroleum-based plastics, in a substitution perspective. Bio-based polymers are derived from renewable biomass resources. However, not all bio-based polymers are biodegradable (3.3.4).

We cannot count on a process of substitution of petroleum-based by bio-based plastics

Bio-based and biodegradable plastics are still in development: a large scale process of substitution of petroleum-based by bio-based plastics in view of both their identified end-of-life and impacts (socio-economic and environmental) is not reliable.

In 2020, Europe accounted for 27.3% (551 kt/year) of global bio-based and biodegradable plastics production capacity, with projections to reach 32.0% (932 kt/year) by 2025, making it the second-largest producer after Asia, which held 46.9% in 2020 and is expected to have 43.2% in 2025 (Moshood *et al.*, 2022). Despite relatively low bio-based plastic production volumes and high costs compared to conventional plastics, consumer demand for eco-friendly products is expected to drive future growth (Moshood *et al.*, 2022).

One major challenge in PHAs production is high costs, which are greater than those of other bio-based polymers. Bassi *et al.* (2021) performed an economic analysis of second-generation PHA production using municipal food waste and wastewater sludge in five cities (Barcelona, Copenhagen, Lisbon, South Wales, and Trento). Their study showed that second-generation PHA outperforms petroleum-based PUR and first-generation PHA (from food crops) in terms of environmental impacts and societal costs, with four times lower impacts and eight times lower costs than PUR. However, further improvements in biorefinery processes, such as reducing methane leakage and upgrading biogas, are necessary for second-generation PHA to compete with LDPE (Bassi *et al.*, 2021). The production costs of second-generation PHA from urban bio-waste range from -0.6 to 2.0 €/kg, lower than those of PUR (2.6 €/kg) and first-generation PHA (3.1 €/kg) but higher than LDPE (1.5 €/kg). This approach, based on Bassi *et al.* (2021), quantifies the environmental and economic consequences of diverting municipal food waste and sewage sludge from traditional management to biorefinery-based production of PHAs. When PHA production in biorefineries is less costly or generates additional value compared to conventional treatment, it leads to a net benefit. The negative estimate of -0.6 €/kg thus indicates that, in this scenario, PHA production provides a financial advantage over conventional waste treatments. However, Bassi *et al.* (2021) performed an economic analysis of second-generation PHA production using municipal food waste found that production costs could be reduced to as low as -0.6 to 0.5 €/kg PHA when biorefinery residues are anaerobically digested and used as fertiliser, making second-generation PHA economically competitive with petroleum-based plastics. Utilising secondary feedstocks, such as biogas or wastewater, could further reduce competition with food for land (Bassi *et al.*, 2021). Second-generation PHA typically outperforms PUR and first-generation PHA in terms of societal costs but does not always perform better than LDPE. Bassi *et al.* (2021) finally recommend considering environmental hotspots throughout the biorefinery chain and avoiding first-generation biomass for bio-based plastic production to reduce impacts. The counterfactual waste management scenario and identification of competing fossil materials are crucial for comprehensive assessments. While PHA from second-generation biomass performs better than PUR, its advantage over LDPE depends on improvements in biorefinery processes and local conditions.

Jha *et al.* (2024) analysed the economic feasibility of bio-based plastics, focusing on PLA. While PLA production showed profitability compared to HDPE, transitioning from conventional to bio-based plastics presents challenges due to the significant land, water, and energy requirements. First-generation PLA is produced from food crops like maize and sugarcane, necessitating arable land and resources. Jha *et al.* (2024) suggested that advancements in second-generation PLA (from waste materials) and mechanical recycling could address these issues. Additionally, self-healing mechanisms in bio-based polymers, such as using Murexide salts to repair cracks, could extend the lifespan of bio-based plastics (Jha *et al.*, 2024).

LCA studies revealed mixed environmental impacts of bio-based and/or biodegradable plastics, with concerns about eutrophication, water use, and effects on biodiversity. Addressing these challenges requires a holistic approach, considering factors such as greenhouse gas emissions, recyclability, and compostability. Studies provide conflicting results. Bassi *et al.* (2021) found that bio-based plastics often have lower climate and fossil fuel impacts but higher eutrophication and toxicity levels. The climate benefits of bio-based plastics, particularly those made from first-generation biomass, may be offset by land-use change impacts. Most LCAs on PHAs have focused on first-generation biomass, with mixed results. Second-generation feedstock studies, mainly involving municipal and industrial wastewater, are limited.

Bassi *et al.* (2021) noted key weaknesses in life-cycle assessment (LCA) and life cycle costing (LCC) studies on bio-based plastics, including besides inconsistent methodologies, unclear handling of multi-functionality, and lack of regional context. Many studies fail to account for the environmental and economic consequences of diverting waste from traditional management to PHA production. They also highlighted that second-generation PHA production emits harmful substances, contributing 20-40% of societal costs, with production costs comprising the rest.

Moshood *et al.* (2022) emphasise that biodegradable plastics face technological and economic challenges, including competition with the bioenergy sector for feedstock and land. There is also a lack of consumer awareness of proper disposal.

3.5.4.2. Upstream strategies are a priority to reduce the production and consumption of plastics

Educate

Education plays a pivotal role in shaping ecological behaviours, particularly concerning plastic pollution and consumption. For instance, Nguyen *et al.* (2024) highlighted the significant influence of education on improving students' behaviour toward plastic waste management. Their study revealed that education has both direct and indirect positive effects on students' actions, adding new insights to the existing body of knowledge on ecological behaviour and offering practical implications for improving waste management among high school students. Nguyen *et al.* (2024) argued that to achieve the goals set by Vietnam's National Action Plan on Marine Plastic Waste Management – namely, reducing ocean plastic waste by 50% by 2025 and 75% by 2030 – education must play a central role in transforming plastic consumption habits.

As one of the many publications showing the importance of education in the shaping of ecological behaviours, Otto and Pensini (2017) insisted on connectedness to nature, fostered by direct contact with natural environments, and showed that it plays a more important role in promoting ecological behaviour than environmental knowledge alone.

Cordier *et al.* (2021) conducted a simulation study to assess the impact of education on reducing inadequately managed plastic waste globally. Their model predicts that by implementing education policies to ensure a minimum of 12 years of schooling for individuals in the 43 most polluting countries by 2050, the amount of inadequately managed plastic waste could be reduced by 34% compared to a business-as-usual (BAU) scenario. However, the study's reliance on the 'number of years spent at school' as an indicator of educational effectiveness and its focus on MSW limit its comprehensiveness. Yan *et al.* (2024) complemented these findings by emphasising the need for targeted policies that influence the behaviour of the working-age population, particularly in low- and middle-income countries. They advocate for a combination of regulatory, market-based, and behavioural instruments to achieve sustained reductions in plastic pollution, highlighting the importance of contextualising these measures to the specific needs and conditions of different regions.

Overall, these studies underscore the critical role of education in addressing plastic waste pollution. The findings suggest that while education is a powerful tool for promoting ecological behaviour, its effectiveness can be enhanced by integrating it with other policy measures and by addressing the limitations of current research approaches.

Public policies: challenges for regulation

Anti-corruption and lobby-control policies

Cordier *et al.* (2021) suggested that a 60% reduction in plastic waste by 2050 could be achieved by fighting against corruption and controlling lobby influences. This would reduce the annual amount of plastic waste to 44 Mt per year, from 110 Mt in the business-as-usual scenario. To achieve this, the top 43 biggest plastic polluter countries should gradually raise their corruption control policies, similar to countries like Uruguay and France. Yan *et al.* (2024) confirmed the outputs of this study and found a significant correlation between plastic pollution and policies controlling corruption and lobbies. France ranks 31st globally in terms of corruption control policies, with Denmark, Finland, New Zealand, Singapore, and Norway in the top five. The bottom five, *i.e.*, Venezuela, Yemen, Syria, Somalia, and South Sudan, have extremely low levels of control. Yan *et al.* (2024) found that if France improved its level of control of corruption and lobbies by 50% in 2024, it could reduce the annual discard of inadequately managed plastic waste by 15%.

Taxes and bonuses impacts are context-dependent

Studies on the effectiveness of taxes and incentives for plastic waste management show mixed and context-specific outcomes. Berk *et al.* (2016) found that a bottled-water tax in the United States (U.S.) state of Washington reduced sales by nearly 6%, with sales never fully recovering even after the tax's removal. The impact varied across income levels and tax rates, suggesting negative long-term economic outcomes.

Similarly, De Lucia *et al.* (2020) and De Lucia and Pazienza (2019) highlighted that market-based tools like tax credits and payback mechanisms have different impacts on agricultural plastic use in Italy. They found that policy choice depends on the type of plastic waste: subsidies are preferred for plastic packaging and films, while tax credits are favoured for plastics from cereal farming. Factors like proximity to waste collection sites and type of agricultural production also influence efficiency of such economic tools, indicating the need for tailored policies.

Conversely, Chen and Chen (2008) demonstrated the positive impact of a plastic bag disposal fee in Taiwan. Their statistical analysis showed the policy effectively reduced household waste, highlighting the role of income, policy design, and collection methods in determining success.

Colelli *et al.* (2022) analysed European packaging waste systems, emphasising the importance of EPR schemes. They found that higher recycling rates do not necessarily entail higher costs. Success hinges on local authorities' involvement in collection and sorting. Effective strategies include a mix of EPR, non-profit models, door-to-door collection, and DRSs.

In brief, while taxes and incentives can influence plastic waste management, their effectiveness on production and consumption reduction depends on factors like waste type, policy design, and local conditions.

Ban unnecessary plastics: the striking example of single-use plastics

Research on regulatory bans of SUP bags provides valuable insights into their potential impacts and challenges. Implementing such bans has shown promise in reducing plastic waste, cutting down on waste management costs, and curbing CO_2 emissions. In France, for instance, the ban on SUP bags (*LOI n° 2015-992 du 17 août 2015 relative à la transition énergétique pour la croissance verte (2020)*), aimed to reach substantial reductions in non-recycled plastic waste and associated costs, along with environmental benefits such as decreased pollution (Chapter V.1 of the extended report).

However, the enforcement of bans poses a significant challenge, particularly in regions where there is resistance from stakeholders and limited availability of alternatives. In France, where the ban on SUP products has been extended to various items, including cutlery, straws, and glass lids, ensuring strict enforcement becomes crucial. Few publications showed that regulations aimed at curbing plastic pollution can have a significant impact on the business plan of some industrial sectors (Convery *et al.*, 2007; He, 2012; Wang *et al.*, 2021a): industry changes its business plan and investment to adapt to the more stringent regulation, such as the SUP directive (European Union, 2019a) for instance by introducing recycled products. Without effective enforcement mechanisms, the ban may fail to achieve its intended goals, leading to continued plastic pollution and environmental degradation.

Furthermore, research highlights the importance of addressing unintended consequences of the ban, such as 'plastic leakage' where the elimination of SUP bags leads to increased consumption of alternative plastic products (for instance trash LDPE bags – Taylor (2019)9). Additionally, cultural and socio-economic factors may influence the effectiveness of such bans. Variability in consumer behaviour and access to alternatives across different regions and social groups may require tailored approaches to enforcement and compliance (Wang et Li, 2021; Arriagada *et al.*, 2022). Public education and outreach campaigns play a crucial role in fostering behavioural change.

In conclusion, while the regulatory ban on SUP bags in developed countries seems to hold its promise for reducing plastic waste, addressing enforcement challenges, unintended consequences, and cultural and socio-economic differences still remains a challenge.

Post-growth as a pathway to reduction: put a cap on Gross Domestic Product

In the context of ecological economics, the concept of sobriety challenges the traditional focus on GDP growth, advocating for a re-evaluation of economic activities based on their contributions to human well-being and ecological balance. Sobriety emphasises the quality of growth over quantity, aiming for long-term sustainability rather than mere increases in production and consumption (Haberi *et al.*, 2006; Krausmann *et al.*, 2009; Jackson, 2016; Victor, 2019). This perspective aligns with a broader understanding of progress that incorporates alternative metrics such as the Genuine Progress Indicator (GPI) or the Human Development Index (HDI) among others (van den Bergh, 2022; Wendling *et al.*, 2022). An analysis of GDP per capita alongside with other key prosperity indicators in a dashboard (Stiglitz *et al.*, 2018) – such as life expectancy at birth, educational attainment, infant mortality rates, employment rates, and self-reported life satisfaction or happiness indices – offers valuable insights for critically examining the growth paradigm. These data consistently reveal that, beyond a certain GDP per capita threshold, increases in prosperity stagnate, indicating that while average income growth (that is, GDP per capita) can enhance well-being and prosperity as a whole, this effect plateaus after a certain level (Jackson, 2016). Economic development will be insufficient, in itself, to bring about the ecological sustainability of societies (York *et al.*, 2004).

The idea of green economic growth driven by technological solutions is often proposed as a strategy to mitigate plastic pollution. However, studies by Cordier *et al.* (2021), Rom and Guillotreau (2024), and Yan *et al.* (2024) extended the findings of York *et al.* (2004) to plastic pollution, indicating that relying solely on GDP per capita growth within a BAU scenario will be insufficient to address global plastic waste management challenges by 2050. These studies demonstrated that unlimited economic growth is becoming less viable within the constraints of a limited global ecosystem, making it necessary to explore alternative approaches, including slow economic growth scenarios.

Interestingly, however, simulations based on data from Yan *et al.* (2024) indicated that, paradoxically, if in France the GDP per capita growth rate was to be halved by 2050, the annual discard of inadequately managed plastic waste would increase by 0.9% compared to the BAU scenario. This suggests that simply slowing economic growth without addressing other factors may not effectively reduce plastic pollution. Cordier *et al.* (2021) further demonstrated that a 1% increase in GDP per capita could decrease inadequately managed plastic waste globally by 0.28% but would also slightly increase total plastic waste by 0.02%, highlighting the ecological side effects of waste treatment processes.

Yan *et al.* (2024) used the STochastic Impacts model by Regression on Population, Affluence, and Technology (STIRPAT) model to assess the impact of socio-economic and political factors on future plastic pollution, identifying low- and lower-middle-income countries, especially in sub-Saharan Africa, as potential major contributors to plastic pollution by 2050, questioning the classical Kuznets Curve model. The Environmental Kuznets Curve (EKC) hypothesis suggests that environmental degradation increases with economic growth until a certain income level, after which it declines as wealthier countries invest in environmental protection (Barnes, 2019). This relationship is widely recognised in environmental economics (Cordier *et al.*, 2021; Rom et Guillotreau, 2024). Higher-income countries are better equipped to invest in technologies that reduce pollution, enhancing resource efficiency and minimising environmental impact (Barnes, 2019; Cordier *et al.*, 2021).

However, the EKC approach does not fully address the complexities of global plastic pollution. While global data indicates an inverted U-shaped curve — low-income countries generate minimal mismanaged plastic waste per capita, middle-income countries produce the most, and high-income countries produce less — this view is incomplete. It overlooks significant waste exports from high-income to lower-income countries, often excluded from official data due to their illegal nature (Yan *et al.*, 2024). In recipient countries, weak regulations and corruption lead to improper waste management, harming ecosystems (Yan *et al.*, 2024). Thus, while the EKC hypothesis may seem valid for high-income countries, it fails for lower-income nations that import plastic waste. These countries face escalating plastic pollution, which eventually impacts global ecosystems, including high-income nations, through ocean currents and contaminated food chains. Comprehensive policy interventions, such as banning plastic waste exports, strengthening global waste management systems, combating corruption, and promoting international cooperation for equitable waste handling, are essential to effectively address plastic pollution (Cordier *et al.*, 2021 ; Yan *et al.*, 2024). It is crucial not to rely solely on the EKC to design plastic pollution policies without prohibiting the outsourcing of externalities to lower-income countries.

In a word, technological innovations alone (such as investing in waste collection and treatment infrastructures), while important, should not be viewed as a panacea. Cordier *et al.* (2021) emphasised the importance of non-technological solutions, as those presented here, in reducing inadequately managed plastic waste. These human-centred strategies, also referred to as community-driven initiatives or socially-driven approaches, should be an integral part of efforts to combat plastic pollution.

Finally, a transition to a post-growth economy could be structured around four pillars. First, shifting from a product-based economy to a service-oriented model would reduce resource extraction and environmental degradation (Jackson, 2016). For example, DRSs promote the reuse and redistribution of packaging, creating more local jobs than the plastic industry. Second, improving the quality of work by prioritising stable and well-paid jobs in local service-based economic models (Jackson, 2016; de Jesus Pacheco *et al.*, 2019; Roman *et al.*, 2023). Third, redirecting investments by eliminating subsidies for fossil fuel-based industries, including plastics, could recover billions annually (Black *et al.*, 2023; Tilsted *et al.*, 2023; Eunomia, 2024; OECD, 2024a; European Environment Agency, 2025). Finally, rethinking the role of money and monetary policies to support a more sustainable economic model (Jackson, 2016). These strategies could help to minimise environmental impact of plastics while creating quality jobs and ensuring a fair economic transition.

Put a cap on plastic production

International negotiations for a global treaty on plastic pollution were initially expected to conclude by the end of 2024. The Fifth United Nations Environment Assembly (UNEA-5) made a significant step in March 2022 by adopting a resolution to develop a legally binding global treaty aimed at tackling plastic pollution, and addressing the full life cycle of plastics, from production to disposal.

The International Negotiation Committee (INC) has been working to finalise a draft by late 2024, with discussions and revisions ongoing. The revised zero draft (UNEP/PP/INC.4/3 (2023b)) presented two options for the treaty's objective: one focusing on plastic pollution, the other prioritising environmental and human health protection. These options underscore the diverse perspectives within the negotiations.

Several coalitions with varying interests have emerged among Member States. The ‘High Ambition Coalition’ led by Rwanda and Norway, gathers 66 members, including the EU and France, and advocates for reducing plastic pollution by curbing consumption and production, and by regulating plastic formulation. The ’Business Coalition for a Global Plastics Treaty’ consisting of businesses, financial institutions, and NGOs, supports a circular economy, and reducing reliance on petroleum-based plastics. Meanwhile, oil and plastic-producing nations, informally aligned as ’like-minded countries’ (*e.g.*, Saudi Arabia), prioritise improved waste management to address plastic pollution while protecting their economic interests. To date, and in the absence of a successful outcome, negotiations must continue beyond what was initially intended to be the last INC, which took place in South Korea in the beginning of December 2024. The INC-5.2 will take place in August 2025 in Geneva, Switzerland.

Overall, there is a significant lack of binding global policies with clear, measurable targets to reduce plastic pollution. While the International Treaty on plastic pollution could be a promising step, expectations for comprehensive international law should be tempered as international regulation always remains more flawed and superficial than European and French regulations (Peyen, 2023).

4. Conclusion and knowledge gaps

Conclusion

Because of their many uses, plastics are now structuring both technically and culturally food value chains, particularly downstream (3.1). Their versatility has encouraged greater complexity in formulation and design (3.2), making their waste management very difficult (3.3). Their ubiquity in continental ecosystems, living organisms and humans, and their proven impacts at all levels (3.4), therefore call into question the very possibility of making the use of plastics in agriculture and food sustainable (3.5).

Knowledge gaps and research needs

There is a pressing need for interdisciplinary research on plastics.

Strengthening field research to understand the usage of plastics in food value chains is essential, including surveys of actors involved in plastic consumption and production.

Tracking plastic flows from production and uses to their fate as waste and particles is key to assess associated impact on the environmental, social and economic dimensions of sustainable development, including its related risks to human health. This implies collecting transparent and traceable data.

Prevention of the hazards of plastics will progress by using a One Health research approach, acknowledging similarities, links and interdependence of the health of humans and other living organisms in ecosystems.

Developing scenarios for reducing plastic production and consumption (alternative systems and practices to plastics), while considering implications for both end-users and economic actors, is crucial.

In particular, research needs to clarify implications of increased or decreased plastic use in food value chains on food availability and safety, unravelling links with characteristics of food value chains (geographical spread, number of intermediaries) also influencing food losses and waste.

Furthermore, efforts focusing on simplifying plastic formulations, addressing both technical and regulatory aspects, are required.

Ultimately, these research priorities will contribute to scientific endeavours aimed at systemic transformation in food value chains.

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