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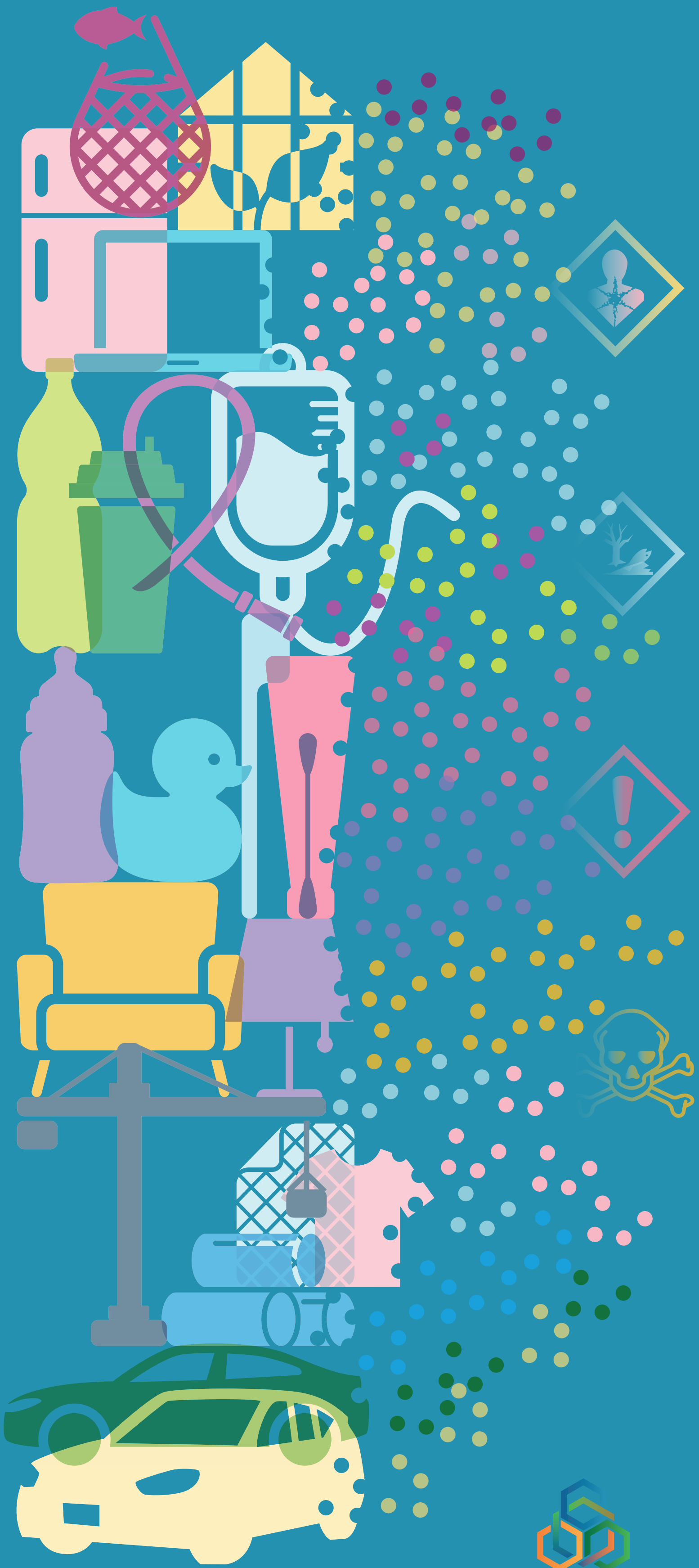
CHEMICALS IN PLASTICS

A TECHNICAL REPORT



BASEL / ROTTERDAM / STOCKHOLM
CONVENTIONS

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Authors: Roland Weber (International Panel on Chemical Pollution, Switzerland, and POPs Environmental Consulting, Germany), Narain M. Ashta (ETH Zurich and Empa - Swiss Federal Laboratories for Materials Science and Technology, Switzerland), Nicolò Aurisano (Technical University of Denmark, Denmark), Zhanyun Wang (International Panel on Chemical Pollution and Empa - Swiss Federal Laboratories for Materials Science and Technology, Switzerland), Magali Outters and Kimberley De Miguel (MedWaves, UNEP/MAP Regional Activity Centre for Sustainable Consumption and Production and Stockholm Convention Regional Centre, Spain), Martin Schlummer (Fraunhofer Institute for Process Engineering and Packaging IVV, Germany), Markus Blepp (Büro für Umweltwissenschaften, Germany), Helene Wiesinger, Helena Andrade and Martin Scheringer (International Panel on Chemical Pollution and ETH Zurich, Switzerland), Peter Fantke (International Panel on Chemical Pollution, Switzerland, and Technical University of Denmark, Denmark).

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Layout, graphic design, infographics: Lowil Espada, Irina Roffe (UNOG), Malgorzata Stylo

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ABBREVIATIONS AND ACRONYMS

6PPD	p-Phenylenediamine (N-(1,3-dimethylbutyl)-N'-phenyl-p-phenylenediamine)
ABS	Acrylonitrile-butadiene-styrene
AOP	Adverse outcome pathway
ATSDR	Agency for Toxic Substances and Disease Registry
BAT	Best available technique
BAuA	German Federal Institute for Occupational Safety and Health
BBP	Benzyl butyl phthalate
BBzP	Butylbenzyl phthalate
BCF	Bioconcentration factor
BEP	Best environmental practice
BFR	Brominated flame retardant
BHET	Bishydroxyethyl terephthalate
BPA	Bisphenol A (4,4'-(Propane-2,2-diyl)diphenol)
BPB	Bisphenol B (4,4’-(1-methylpropylidene)bisphenol)
BPF	Bisphenol F (4,4'-Methylenediphenol)
BPS	Bisphenol S (4,4'-Sulfonyldiphenol)
BREF	BAT reference document
BTBPE	1,2-Bis (2,4,6-tribromophenoxy) ethane
BTEX	Benzene, toluene, ethylbenzene and xylenes
BZTs	Benzotriazoles
CENELEC	The European Committee for Electrotechnical Standardization
CFR	Chlorinated flame retardant
CiP	Chemicals in Products
CKBD	Cement kiln bypass dust
CLP	Classification, labelling and packaging (EU regulation)
CMR	Carcinogenic, mutagenic, or toxic for reproduction
CPCat	Chemicals and Product Categories database
CPPdb	Database of Chemicals associated with Plastic Packaging
CPSC	US Consumer Product Safety Commission
CSI	Cement Sustainability Initiative
DBDPE	Decabromodiphenylethane
DBP	Dibutyl phthalate
DCPS	4,4’-Dichlorodiphenyl sulphone
DecaBDE	Decabromodiphenyl ether
DEHP	Di-2-Ethylhexyl phthalate; Bis(2-ethylhexyl)benzene1,2-dicarboxylate
DIBP	Diisobutyl phthalate
DIDP	Diisodecyl phthalate
DINCH	1,2-Cyclohexane dicarboxylic acid diisononyl ester
DINP	Diisononyl phthalate
DnOP	Di-n-octyl phthalate
DPS	Diphenyl sulphone
DRS	Deposit-Refund Systems
DTSC	Department of Toxic Substances Control
EC	European Commission
ECETOC	European Centre for Ecotoxicology and Toxicology of Chemicals

ECHA	European Chemicals Agency
EDCs	Endocrine disrupting chemicals
EEA	European Environmental Agency
EEE	Electrical and electronic equipment
EFSA	European Food Safety Authority
EGIP	Environmental Genome of Industrial Products
EPS	Expanded polystyrene
ESM	Environmentally sound management
EU	European Union
EVA	Ethylene vinyl acetate
FCMs	Food contact materials
FTMS	Fourier transform mass spectrometry
GC	Gas chromatography
GHS	Globally Harmonized System of Classification and Labelling of Chemicals
GPWM	Global Partnership on Waste Management
HCl	Hydrogen chloride
HF	Hydrogen fluoride
HALS	Hindered amine light stabilizers
HBCDD[1]	Hexabromocyclododecane
HBB	Hexabromobiphenyl
HBBz	Hexabromobenzene
HBM4EU	Human Biomonitoring for Europe
HCB	Hexachlorobenzene
HCFCs	Hydrochlorofluorocarbons
HFCs	Hydrofluorocarbons
HCWH	Health Care Without Harm
HDPE	High-density polyethylene
HFCs	Hydrofluorocarbons
HIPS	High impact polystyrene
HP	Hewlett Packard
HT	High throughput
IAS	Intentionally added substances
ILO	International Labour Organization
ILSI	International Life Sciences Institute
IPCP	International Panel on Chemical Pollution
IPEN	International Pollutants Elimination Network
ISWA	International Solid Waste Association
JRC	Joint Research Center
LC	Liquid chromatography
LCA	Life cycle assessment
LCCPs	Long-chain chlorinated paraffins
LDPE	Low-density polyethylene
MARPOL	International Convention for the Prevention of Pollution from Ships
MCCPs	Medium-chain chlorinated paraffins
MEG	Mono ethylene glycol

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[1] The Stockholm Convention uses the abbreviation HBCD, which, however, is normally used for hexabromocyclodecane (Bergman et al. 2012). For hexabromocyclododecane, HBCDD is the abbreviation commonly used in the scientific literature and therefore also in this report.

MRSL	Manufacturing Restricted Substances List
MS	Mass spectrometry
MSW	Municipal solid waste
Mt	Million tonnes
NHANES	National Health and Nutrition Examination Survey
NIAS	Non-intentionally added substances
Norad	Norwegian Agency for Development Cooperation
OECD	Organisation for Economic Co-operation and Development
OPFRs	Organophosphorous flame retardants
OPTOCE	Ocean Plastic Turned into an Opportunity in Circular Economy
PA	Polyamides
PAHs	Polyaromatic hydrocarbons
PBB	Polybrominated biphenyl
PBDDs	Polybrominated dibenzo-p-dioxins
PBDEs	Polybrominated diphenyl ethers
PBDFs	Polybrominated dibenzofurans
PBT	Persistent, Bioaccumulative, and Toxic
PC	Polycarbonate
PCBs	Polychlorinated biphenyls
PCDDs	Polychlorinated dibenzo-p-dioxins
PCDFs	Polychlorinated dibenzofurans
PCNs	Polychlorinated naphthalenes
PE	Polyethylene
PEEK	Polyether ether ketone
PES	Polyethersulfone
PEX	Cross-linked polyethylene
PFASs	Per- and polyfluoroalkyl substances
PFCs	Perfluorocarbons
PFOS	Perfluorooctanesulfonic acid
PFOA	Perfluorooctanoic acid
PESTOOL	Plastics Exposure Scenario Tool
PET	Polyethylene terephthalate
PiF	Product intake fraction
PLA	Polylactic acid
PMMA	Polymethyl methacrylate
POM	Polyoxymethylene; polyacetal
POPs	Persistent organic pollutants
POPRC	POPs Review Committee
PP	Polypropylene
PPE	Poly(p-phenylene ether)
ppm	Parts per million
PPO	Plastic pyrolysis oil
PRTRs	Pollutant release and transfer registers
PS	Polystyrene
PTA	Purified terephthalic acid
PTFE	Polytetrafluoroethylene
PUF	PUR foam
PUR	Polyurethane
PVB	Polyvinylbutyral

PVC	Polyvinyl chloride
PVDC	Polyvinylidene dichloride
QSAR	Quantitative structure-activity relationship
RAPEX	Rapid Alert system for dangerous products (Europe)
REACH	Registration, Evaluation, Authorisation and Restriction of Chemicals
RIVM	National Institute for Public Health and the Environment of The Netherlands (Rijksinstituut voor Volksgezondheid en Milieu)
RoHS	Restriction of Hazardous Substances in Electrical and Electronic Equipment
SAICM	Strategic Approach to International Chemicals Management
SAN	Styrene-acrylonitrile resin
SCCPs	Short-chain chlorinated paraffins
SCCS	Scientific Committee on Consumer Safety
SCENIHR	Scientific Committee on Emerging and Newly Identified Health Risks
SCHER	Scientific Committee on Health and Environmental Risks
SCIP	Substances of Concern in articles as such or in complex objects (Products)
SETAC	Society of Environmental Toxicology and Chemistry
SINTEF	Stiftelsen for industriell og teknisk forskning (Norwegian research organization)
SVHC	Substance of Very High Concern
TBBPA	Tetrabromobisphenol A
TBOEP	Tris(2-butoxyethyl) phosphate
TBT	Tributyltin
TCEP	Tris(2-chloroethyl)phosphate
TCPP	Tris(2-chloroisopropyl) phosphate
TDCPP	Tris(1,3-dichloroisopropyl)phosphate
TDI	Tolerable daily intake
Tg	Glass transition temperature
TG	Test guideling
TOF	Time-of-flight
TPE	Thermoplastic elastomers
TPhP	Triphenyl phosphate
TTC	Threshold of toxicological concern
UN	United Nations
UNEA	United Nations Environment Assembly
UNEP	United Nations Environment Programme
UNITAR	United Nations Institute for Training and Research
UPOP	Unintentional persistent organic pollutants
US	United States
USD	United States Dollar
US EPA	United States Environmental Protection Agency
US FDA	United States Food and Drug Administration
UTC	Unintentional trace contaminant
UV	Ultraviolet
VOCs	Volatile organic compounds
vPvB	Very persistent and very bioaccumulative
WEEE	Waste electrical and electronic equipment
WHO	World Health Organisation
WWTP	Wastewater treatment plant
XPS	Extruded polystyrene
ZDHC	Zero Discharge of Hazardous Chemicals

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ABOUT THIS REPORT

This report aims to inform the global community about the chemical-related issues of plastic pollution and options to address them. It also aims to support the negotiation process to develop the instrument on plastic pollution based on United Nations Environment Assembly resolution 5/14. The report outlines a set of credible and publicly available scientific studies and initiatives focused on chemicals in plastics and the science-policy interface. However, the report is not exhaustive given the large and rapidly growing volume of global research on the topic, nor does it comprehensively review the environmental and occupational epidemiological literature. As the evidence base grows, future efforts will build on this report to further provide support and take actions to reduce the impacts of plastic pollution and the associated chemicals affecting individuals, communities, and ecosystems.

EXECUTIVE SUMMARY

All plastics are made of chemicals, including basic polymers and solvents; additives such as plasticizers, flame retardants, stabilizers or pigments used to deliver the material's functionality; and unintentional chemical residues resulting from incomplete processing during the chemical synthesis and plastic manufacturing stages.

With the increased production and consumption of plastics worldwide, the production of plastic-associated chemicals has also increased, both in quantity and diversity. More than 13,000 chemicals are associated with plastics and plastic production across a wide range of applications, of which over 3,200 monomers, additives, processing aids and non-intentionally added substances are of potential concern due to their hazardous properties. These properties include carcinogenicity, mutagenicity, reproductive toxicity, specific target organ toxicity, endocrine disruption, ecotoxicity, bioaccumulation potential, environmental persistence and mobility, including potential for long-range environmental transport to remote locations.

In addition to certain monomers, ten groups of chemicals associated with plastics have been identified in this report as being of major concern due to their known toxicity and potential to be released from plastics. They include specific flame retardants, ultraviolet light stabilizers, per- and polyfluoroalkyl substances (PFASs), phthalates, bisphenols, certain alkylphenols and alkylphenol ethoxylates, biocides, metals and metalloids, polycyclic aromatic hydrocarbons (PAHs), and non-intentionally added substances such as polychlorinated dibenzo-p-dioxins and furans (PCDD/Fs) or other contaminants in new products made from recycled plastics.

The report further identifies ten priority use sectors where chemicals of concern have been found in plastic products and their use can lead to either high likelihoods of ecosystem and human exposures or exposures in vulnerable populations such as children. These include toys and other children's products, packaging (including food contact materials), electrical and electronic equipment, vehicles, synthetic textiles and related materials, furniture, building materials, medical devices, personal care and household products, and agricultural/aquaculture/fisheries plastics.

Hazardous chemicals can be emitted and released at all stages of the life cycle of plastics, leading to ecosystem and human exposures. Workers in the plastic or chemical sector may be exposed to hazardous chemicals during the production of polymers, additives, and plastic products, or during plastic waste management, including recycling. In the use phase, consumer exposure is particularly relevant since consumers come in direct contact with plastic-based food contact materials, textiles, building materials, furniture, vehicles, electronics, toys, as well as personal care and household products. In addition to exposure to chemicals via mouthing of products and direct dermal contact, humans and biota can also be exposed to chemicals indirectly when plastic-associated chemicals are released into air, soil, or water, via inhalation of air or consumption of contaminated food and drinking water. Organisms can also be exposed to plastic-associated chemicals after ingesting plastic debris or following exposure in waterways and in terrestrial environments.

Susceptibility to the effects of chemical exposure can differ by gender and age. Women and children are particularly vulnerable to the toxicological hazards of chemicals. Men are also at risk, with research documenting detrimental effects on male fertility resulting from cumulative exposures to hazardous chemicals, many of which are also associated with plastics.

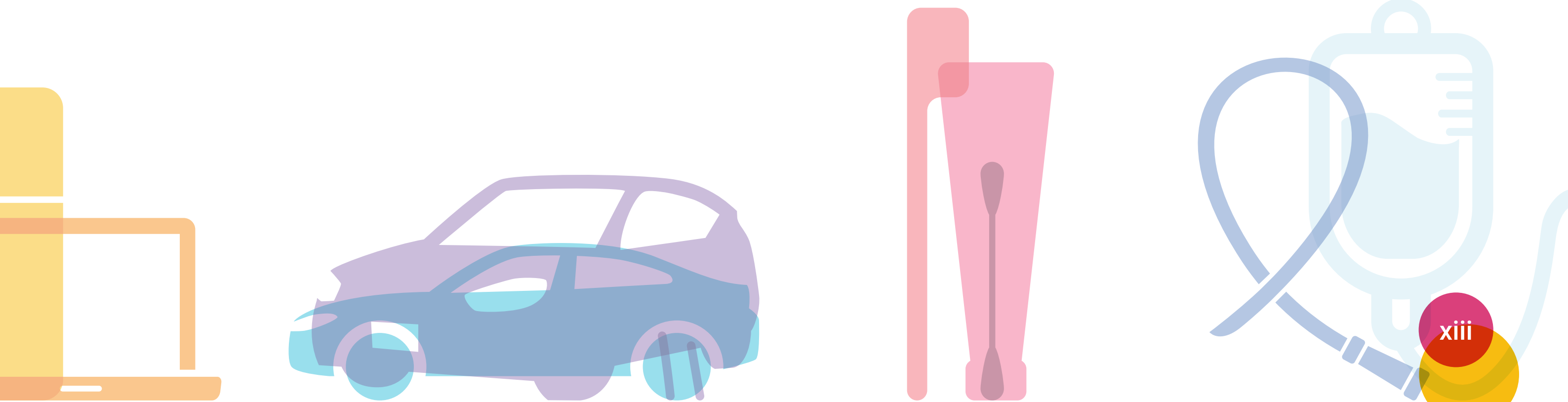
The sheer volume and variety of plastics and associated chemicals produced, used, and released to the environment worldwide have resulted in transboundary pollution that can have adverse effects on human health and the environment, including by exacerbating climate change and biodiversity loss. Without the implementation of globally coordinated measures, the increasing production of plastics and associated chemicals will result in increasing pollution levels and associated environmental, social, and economic costs.

Global action is needed. However, a number of key challenges need to be overcome, including:

- Fragmented policies for regulating plastics and chemicals at local, regional, and international levels;
- Unsustainable plastic production and consumption patterns;
- Lack of access to basic, yet crucial information on the identity, quantity, and hazards of chemicals used or found in different plastic products, as well as the exposure pathways along the plastic value chain;
- Identifying and implementing safer and more sustainable alternatives to hazardous chemicals in plastics; and
- Lack of capacity, particularly in developing countries, to manage plastic waste and to identify and eliminate hazardous chemicals from the life cycle of plastics, including from recycling loops.

The following actions will help reduce the chemical-related impacts of plastic pollution:

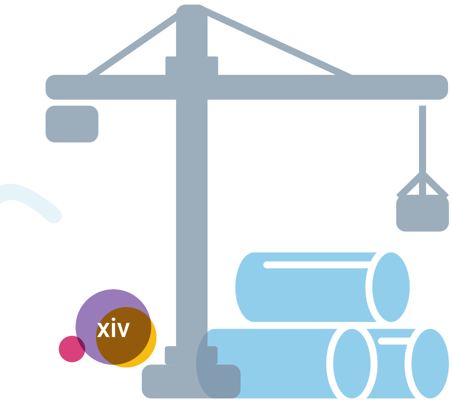
- Reduce plastic production and consumption, starting with non-essential plastics.
- Design and manufacture plastics that are free of chemicals of concern. Chemicals already restricted internationally (e.g., persistent organic pollutants) or identified as issues elsewhere (e.g., PFASs, endocrine disrupting chemicals) should be prioritized for phase-out. Furthermore, the concept of "essential use" can help restrict hazardous chemicals or chemical groups to only specific applications where they cannot yet be substituted.
- Avoid regrettable substitutions (i.e., replacing a chemical/material with an equally hazardous alternative) and shifting burdens (e.g., in terms of resource and energy consumption). Conducting chemical alternatives assessments and life cycle assessments, as well as following Green and Sustainable Chemistry objectives and guiding considerations, can help identify safer substitutes.
- Improve transparency along the entire plastics value chain. Manufacturers should fully disclose the identity and quantity of all chemicals used and found in plastic products, product use patterns, and release of chemicals. This would allow (i) regulatory authorities to better assess exposure and risk to derive science-based policy decisions; (ii) recyclers to identify hazardous chemicals in plastic waste, thereby mitigating occupational exposure and preventing cross-contamination of recycled products; and (iii) businesses and consumers to make informed decisions about the products they buy or use.
- Update regulatory testing guidelines, for instance by including rapid and cost-effective approaches such as bioassays and computational tools. This serves as a prerequisite for research to fill existing knowledge gaps, e.g., regarding the mixture toxicity of leachates from plastics or the toxicity and bioaccumulation potential of hydrophobic chemicals.
- Develop robust regulatory waste management frameworks that align with the waste hierarchy and consider chemicals of concern present in plastic waste. The protection of the informal sector should be given specific attention.



Significant capacity-building efforts are also needed worldwide, and in particular in developing countries, for the following:

- Development, implementation, and enforcement of national legal frameworks and policies that are synergistic with international efforts;
- Competencies in identifying and implementing solutions/chemical substitutions, especially among small- and medium-sized enterprises;
- Supply chain management, including concerning transparency and information sharing;
- Competencies in conducting chemical hazard, exposure, and risk assessments;
- Waste management capacity and technology, including to support the identification of products/waste streams containing hazardous chemicals and to separate and destroy plastics containing hazardous chemicals; and
- Local and international networks to support companies or sectors that cannot develop capacity themselves.

By outlining the chemical-related issues of plastic pollution and providing recommendations to address them, this report aims to inspire and catalyze immediate action to transition to safe and sustainable material cycles, with the overall goal of protecting human health and the environment. This transition only succeeds through the combined effort of all stakeholders along the plastics value chain, including regulatory authorities, industries involved in plastic manufacturing and use, waste managers and recyclers (including the informal sector), scientists from multiple disciplines, and the general population.



1

INTRODUCTION

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1 / INTRODUCTION

Plastics have become ubiquitous in modern life because they are lightweight, cheap, and versatile materials. However, with the increasing production and consumption of plastics, the world has also seen rampantly increasing releases of plastics to the environment and transboundary pollution. Concerned by the global scale of the issue, the United Nations Environment Assembly (UNEA) adopted a historical resolution in March 2022, calling for the development by the end of 2024 of an international legally binding instrument on plastic pollution (UNEA resolution 5/14). The instrument is to be based on a comprehensive approach that addresses the full life cycle of plastics.

All plastics are made of chemicals, including basic polymers and solvents; additives such as plasticizers, flame retardants, stabilizers or pigments used to deliver the material's functionality; and unintentional chemical residues resulting from incomplete processing during the chemical synthesis and plastic manufacturing stages (Geueke 2013a; Geiser *et al*. 2015; Geueke 2018; Pan 2018).

With the continuous increase in plastic production worldwide, the production of plastic-associated chemicals has also increased, both in quantity and diversity. In 2017, the annual global primary plastic production was 438 million tonnes, of which 27 million tonnes (6%) were additives, and primary plastic production is projected to reach 1.1. billion tonnes in 2050 if the current trend continues (Geyer 2020).

While the adverse physical impacts of plastics in the environment are often visible, less apparent are the health risks associated with the chemicals used to produce or found in plastics and subsequently released into the environment. Latest research has identified over 13,000 chemicals associated with plastics and plastic manufacturing across a wide range of applications such as packaging, building and construction, consumer and institutional products, automotive and transportation, electrical, and many more (Annex 1, Aurisano *et al*. 2021b; Wiesinger *et al*. 2021). Amongst these, 7,000 chemicals have been screened for their hazardous properties, of which more than 3,200 plastic monomers, additives, processing aids, and non-intentionally added substances have been identified as chemicals of potential concern based on their hazardous properties (Annex 1; Aurisano *et al*. 2021b; Wiesinger *et al*. 2021).

Plastics may for instance contain persistent organic pollutants (POPs) such as short-chain chlorinated paraffins (SCCPs), polybrominated diphenyl ethers (PBDEs), polychlorinated biphenyls (PCBs), polychlorinated naphthalenes (PCNs), perfluorooctanesulfonic acid (PFOS) and perfluorooctanoic acid (PFOA) and related chemicals, as well as other potentially hazardous chemicals. Plastic-associated chemicals also include those identified as emerging policy issues or issues of concern under the Strategic Approach to International Chemicals Management (SAICM), such as endocrine disrupting chemicals (EDCs), hazardous substances within the life cycle of electrical and electronic products, and per- and polyfluoroalkyl substances (PFASs), as well as those identified as issues under the Global Chemicals Outlook II (UUNEP 2019a), such as bisphenol A (BPA), cadmium, lead, microbeads in personal care products and cosmetics, polycyclic aromatic hydrocarbons (PAHs), phthalates and triclosan, for which available evidence indicates a risk to human health and the environment

Many of these chemicals of concern are used, emitted, and released throughout the plastic life cycle – from the extraction of oil and gas and the production of polymers and chemicals to the manufacturing, use, and end-of-life management of plastics. These chemicals have been found to be associated with a wide range of acute, chronic, or multi-generational toxic effects, including specific target organ toxicity, various types of cancer, genetic mutations, reproductive toxicity, developmental toxicity, endocrine disruption and ecotoxicity (Carlisle *et al*. 2009; Shaw *et al*. 2010; International Agency for Research on Cancer [IARC] 2019; Chen *et al*. 2019; Brinkmann *et al*. 2022; ECHA 2023a).

However, information on chemicals in plastics is rarely transmitted along the plastics life cycle and is therefore unavailable to regulatory authorities, consumers, and waste managers. This lack of information hampers risk assessments and product safety. It further complicates recycling of plastic waste because some additives can negatively impact recycling processes (Hahladakis *et al*. 2018; Azoulay *et al*. 2019; UNEP 2021c) and are dangerous for recyclers. The recycling of plastic products containing hazardous chemicals can transfer these chemicals into consumer products such as food contact materials (FCMs) and children's toys (Chen *et al*. 2009; Samsonek and Puype 2013; Ionas *et al*. 2014; Puype *et al*. 2015; Guzzonato *et al*. 2017; Kuang *et al*. 2018), with potential for high levels of exposure or exposure in vulnerable populations. These issues hamper the much-needed transition to a toxic-free circular economy.

Many chemicals of concern also exhibit high environmental persistence, bioaccumulation potential, and mobility in the environment, including the potential for long-range transport to remote locations. Global action on chemicals in plastics is therefore urgently needed to protect human health and the environment (Carney Almroth *et al*. 2022, Dey *et al*. 2022, Persson *et al*. 2022, Wang and Praetorius 2022).

In the following chapters, the various chemical-related issues of plastic pollution are described. Chapter 2 provides an overview of the different types of plastics and the chemicals used to produce or found in these plastics. It identifies ten groups of chemicals that are of concern due to their toxicity and other hazardous properties, or due to a complete lack of information about them. It further pinpoints ten priority use sectors where there is evidence of the presence of chemicals of concern in their plastic products, and where plastic applications result either in relatively high likelihoods of chemical exposure or exposures in vulnerable populations. Chapter 3 provides an overview of the environmental release of plastics and the fate of chemicals in plastics. It further presents the current knowledge on the different pathways of ecosystem and human exposures to plastic-associated chemicals, as well as on the related adverse impacts. Chapter 4 is targeted towards practitioners and researchers working on risk assessments. It discusses the existing challenges related to the hazard, exposure and risk assessments of individual chemicals and chemical mixtures. It describes various methods that have been developed or are under development to improve such assessments. Future directions for research and method development are also identified. Chapter 5 provides options for different stakeholders – including regulatory authorities, manufacturers, businesses, and consumers – to address the chemicals of concern in plastics in the context of a toxic-free circular economy. Chapter 6 addresses some of the concerns related to substituting hazardous chemicals with other chemicals, or plastics with other materials. In doing so, it describes approaches such as chemical alternatives assessment and life cycle assessment that could guide decision-making processes when alternatives to chemicals or plastics are being considered. Chapter 7 describes the various challenges and options for the end-of-life management of plastics that contain hazardous chemicals in the context of transitioning towards a toxic-free circular economy.

Definition of key terms

Plastics are solid materials that contain as an essential ingredient one or more high-molecular-mass polymers, and which are formed (shaped) by heat and/or pressure during either the manufacture of the polymer or the fabrication into a finished product (MARPOL 2011). Plastics have material properties ranging from hard and brittle to soft and elastic (MARPOL 2011). A more waste management-related definition of plastics can be found in the Basel Convention technical guideline (UNEP 2021h). There are two main types of plastics: thermoplastics and thermosetting polymers (thermosets). Thermoplastics can be melted and moulded various times since they do not undergo chemical changes in their composition when heated. In contrast, thermosets can be melted and shaped only once; after they have solidified, they tend to stay solid, due to an irreversible chemical reaction that creates a tight crosslink between the plastic molecules during melting/solidification (Hansen *et al*. 2013). During the manufacturing of thermoplastic, pre-production plastic pellets or "nurdles" (1–5 mm) are melted and moulded into the desired plastic product along with various additives (Hunter *et al*. 2022).

Microplastics refers to plastic particles <5 mm in diameter. They are either used intentionally in certain products and industrial processes, or formed unintentionally when larger plastic products break down into smaller fragments. Microplastics are also generated as fibres from synthetic textiles, as paint flakes, or as tire wear particles due to the abrasion of rubber tires from vehicles in contact with road surfaces (UNEP 2021d).

Nanoplastics are plastic particles <1 µm in diameter (Gigault *et al*. 2018).

Plastic debris includes plastic items or particles that have been released to the environment.

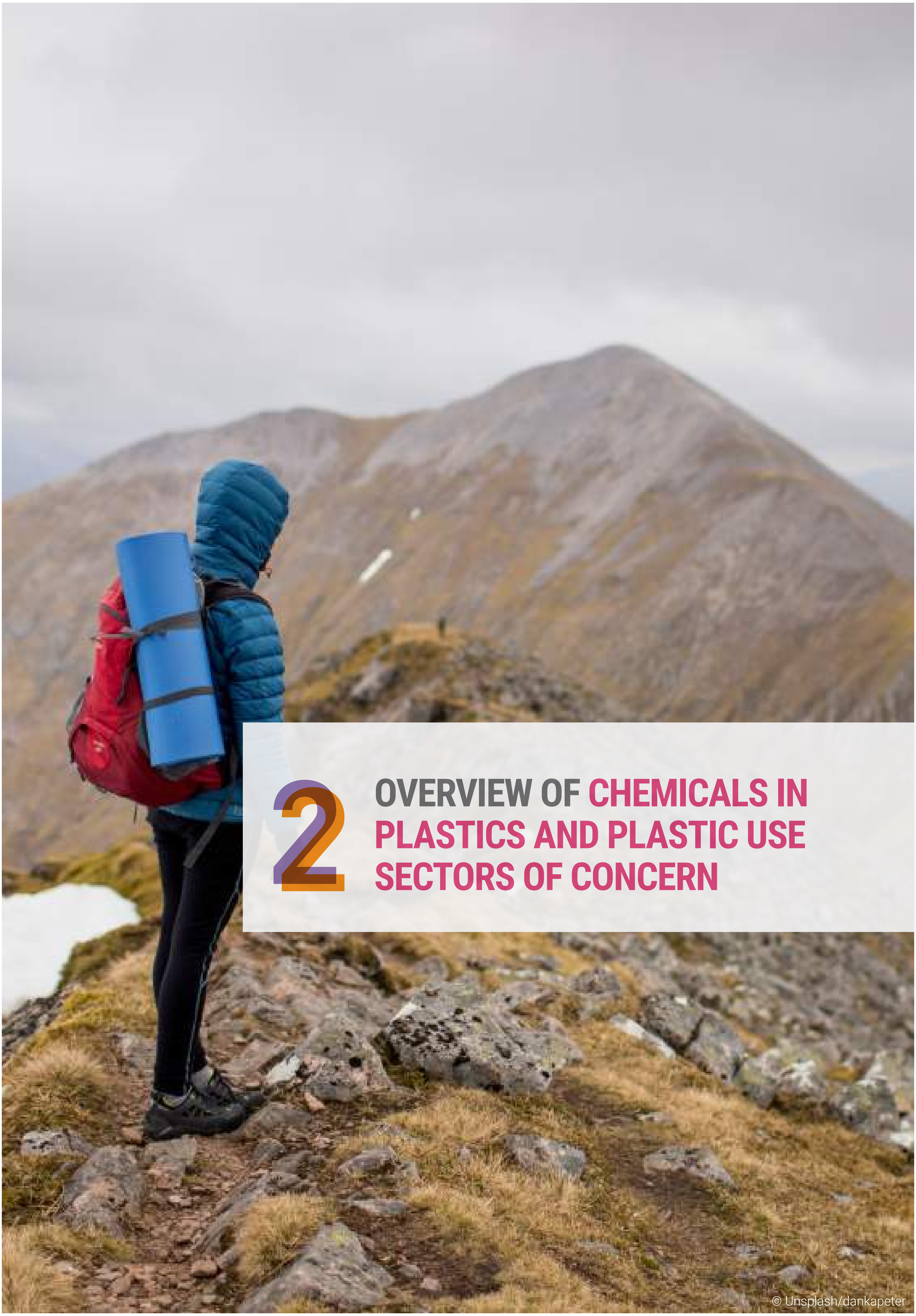
Bioplastics are plastics that are **bio-based** (i.e., made from renewable feedstock sources such as corn and sugar cane) or **biodegradable** (i.e., undergo microbial conversion into CO_2, methane, biomass, and mineral salts) (van den Oever *et al*. 2017).

Polymers are natural or synthetic long-chain substances consisting of sequences of one or more types of monomers (European Commission 2006, Rossi and Blake 2014). They form the backbone of a plastic material.

Monomers are molecules that can undergo polymerization, thereby contributing constitutional units to the essential structure of a macromolecule (Jenkins *et al*. 1996).

Additives are chemical compounds added during plastic compounding (the process of mixing or blending polymers and additives in a molten state) to fulfill specific desired functional properties in the production process or in the final plastic product (Hansen *et al*. 2013; Rossi and Blake 2014). Additives can be divided into four different categories: functional additives, colorants, fillers, and reinforcements (Hansen *et al*. 2013). Typical additives include plasticizers, flame retardants, heat and ultraviolet (UV) light stabilizers, antioxidants, biocides, pigments (colorants), antistatic and blowing agents, impact modifiers, and lubricants. The function usually determines the weight fraction range of an additive in the given plastic application. For example, the softer the plastic, the higher the plasticizer content in that plastic.

Non-intentionally added substances (NIAS) are chemical compounds present in plastic but not intentionally added for a technical reason during the production process. NIAS include impurities in the substances used, reaction intermediates formed during the production process, as well as decomposition or reaction products (European Commission 2011a). The European food contact regulation also includes any unwanted reaction product, impurity, or breakdown product (Framework Regulation EC 1935/2004, Art. 3). Monomers remaining in plastic from production or formed from degradation are considered intentionally added substances (IAS).



2

OVERVIEW OF CHEMICALS IN PLASTICS AND PLASTIC USE SECTORS OF CONCERN

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2 / OVERVIEW OF CHEMICALS IN PLASTICS AND PLASTIC USE SECTORS OF CONCERN

Depending on their specific applications, plastic products are designed by selecting one or more polymers and several additives (e.g., plasticizers, flame retardants, UV light stabilizers, pigments) used to impart functional properties such as pliability, resistance to thermal or UV degradation, and color. Table 1 provides an overview of different polymer types and their main plastic applications. Table 2 provides an overview of different types of additives, compatible polymers, chemical classes, and typical application amounts.

Most of the plastic typically consists of the polymer matrix, followed by fillers and plasticizers. Other additives such as pigments or stabilizers are required in smaller amounts compared to the overall polymer mass. On average, 4% of the weight of plastics consists of additives (Bouwmeester *et al*. 2015), but different polymers use different amounts. For instance, plasticisers can make up over 50% of the total weight of a polyvinyl chloride (PVC)-based plastic (Murphy 2001; Net *et al*. 2015).

Figure 1 presents the share of various additive types in global plastics production during 2000–2014 (Geyer *et al*. 2017). Plasticizers are the most widely used additives, followed by fillers and flame retardants. Due to their high applications and volumes, these two types of additives have been receiving more attention compared to others, for example in the Commission Regulation (EU) 2018/2005 (European Commission 2018b).

In addition to polymers, fillers, and additives, other chemicals might be present in plastic, including other intentionally used substances such as solvents, unreacted monomers, starting substances, and processing aids, as well as non-intentionally added substances (NIAS), which include impurities, reaction by-products, and breakdown products (Groh *et al*. 2019). NIAS alone may account for more than half of the number of chemicals in plastics (Grob *et al*. 2006; Geueke 2018; Geueke *et al*. 2022). Thus, from a chemical perspective, plastic products are complex mixtures of one or more polymers, fillers, several additives, and many (often unidentified) NIAS. Figure 2 provides an overview of chemicals in plastics, including ten groups of chemicals identified as of potential concern. Annex 1 provides a comprehensive list of more than 13,000 plastic-associated chemicals, together with information on their functions, compatible polymer types, typical industrial sectors where they are used, hazard classifications, regulatory status, and more.

Figure 1. Share of main additive types in the global plastics production from data covering the period 2000–2014 (modified from Geyer *et al*. 2017). ▼

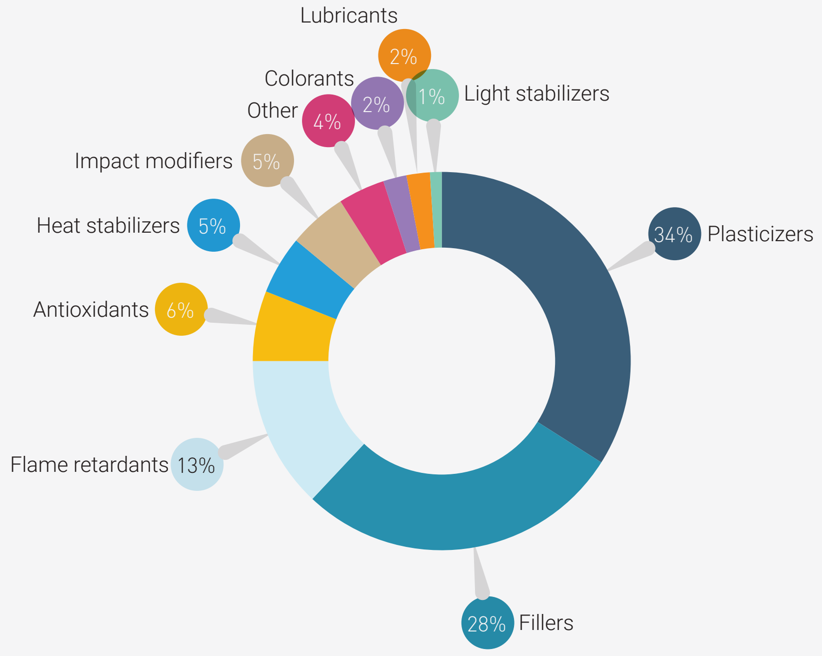
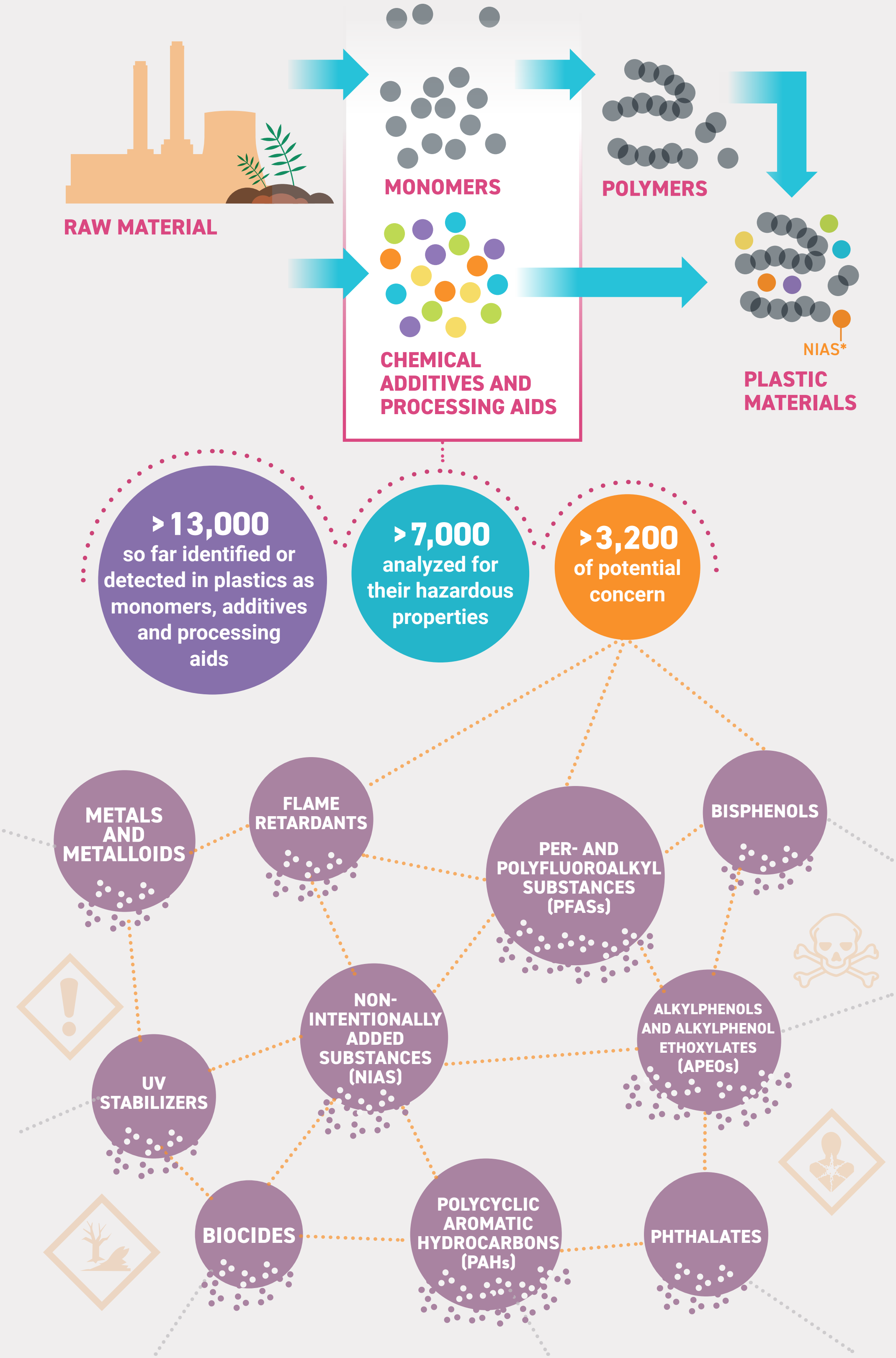


Figure 2. Overview of chemicals associated with plastics, including ten groups of chemicals identified as of concern. ▼

CHEMICALS IN PLASTICS OVERVIEW



RAW MATERIAL

MONOMERS

POLYMERS

CHEMICAL ADDITIVES AND PROCESSING AIDS

NIAS*

PLASTIC MATERIALS

>13,000 so far identified or detected in plastics as monomers, additives and processing aids

>7,000 analyzed for their hazardous properties

>3,200 of potential concern

METALS AND METALLOIDS

FLAME RETARDANTS

PER- AND POLYFLUOROALKYL SUBSTANCES (PFASs)

BISPHENOLS

NON-INTENTIONALLY ADDED SUBSTANCES (NIAS)

ALKYLPHENOLS AND ALKYLPHENOL ETHOXYLATES (APEOs)

UV STABILIZERS

BIOCIDES

POLYCYCLIC AROMATIC HYDROCARBONS (PAHs)

PHTHALATES

Table 1. **Overview of plastic categories and their main application** (list from PlasticsEurope (2019), applications from various sources).

Polymer type	Examples of main plastic applications
THERMOPLASTICS	
Polyethylene (PE)	packaging, consumer goods, fibers and textiles, pipes and fittings
Polypropylene (PP)	packaging, consumer goods, automotive applications
Polyvinyl chloride (PVC)	building and construction, health care, electronics, automobile and other sectors, piping and siding, blood bags and tubing, wire and cable insulation, windshield system components
Polyethylene terephthalate (PET)	plastic bottles, tape applications, packaging, rigid cosmetic jars, microwavable containers, transparent films
Polystyrene (PS)	protective packaging (such as packing peanuts and CD and DVD cases), containers, lids, bottles, trays, tumblers, disposable cutlery, in the making of models
Expanded polystyrene (EPS)	building materials, white foam packaging, surfboards
Acrylonitrile butadiene styrene (ABS)	drain-waste-vent pipe systems, musical instruments (recorders, plastic clarinets, piano movements), golf club heads, automotive trim components, automotive bumper bars, medical devices for blood access, enclosures for electrical and electronic assemblies, protective headgear, whitewater canoes, buffer edging for furniture and joinery panels, luggage and protective carrying cases, small kitchen appliances, toys (including Lego and Kre-O bricks), household and consumer goods
Styrene-acrylonitrile resin (SAN)	food containers, water bottles, kitchenware, computer products, packaging material, battery cases, plastic optical fibers
Polyamides (PA)	textiles, automotive industry, carpets, kitchen utensils, sportswear
Polycarbonate (PC)	electronic components, construction materials, data storage, automotive, aircraft, railway, security components, medical applications, phones
Poly methyl methacrylate (PMMA) (also known as acrylic, acrylic glass, or plexiglass)	transparent glass substitute, daylight redirection, medical technologies and implants, artistic and aesthetic uses
Thermoplastic elastomers (TPE)	automotive sector, household appliances sector
Polyarylsulfone	membranes, fuel cells
Fluoropolymers, e.g., polytetrafluoroethylene (PTFE)*	automotive and aircraft bearings and seals, flame retardants, cookware coatings, linings of piping and chemical tanks, packing for lithium-ion batteries, cable coating in telecommunications and computer industries, implants and catheters for bio-medical applications
Polyether ether ketone (PEEK)	bearings, piston parts, pumps, high-performance liquid chromatography columns, compressor plate valves, electrical cable insulation, ultra-high vacuum applications, medical implants
Polyoxymethylene (POM)	in precision parts requiring high stiffness, low friction, and excellent dimensional stability
Polybutylene terephthalate	housings in electrical engineering, in automotive construction as plug connectors, showerheads, irons, processed into fibers in toothbrushes and false eyelashes, in the keycaps of some high-end computer keyboards
Polylactic acid (PLA)	plastic film, disposable tableware, cutlery bottles, 3D printing, medical implants, monofilament fishing line
Polyhydroxyalkanoates (PHA)	packaging, medical implants
Polybutylene succinate (PBS)	packaging, tableware, mulching films, release materials for pesticide and fertilizer
Polybutylene adipate terephthalate (PBAT)	cling wrap for food packaging, compostable plastic bags, and water-resistant coatings such as on paper cups
Polycaprolactone (PCL)	medical implants; drug delivery device

continued on next page

Table 1 continuation

Polymer type	Examples of main plastic applications
THERMOSETTING POLYMERS	
Polyurethane (PUR)	cleaning products, furniture, automobile seats, hoses, sculptures, decorations, filling of spaces and cavities
Unsaturated polyester	sheet moulding compound, bulk moulding compound and the toner of laser printers, as overlays on roads and bridges
Epoxy resins	paints and coatings, adhesives, industrial tooling and composites, wind turbine technology composites, electrical systems and electronics, petroleum and petrochemical, consumer and marine applications
Melamine resin	construction material, kitchen utensils and plates, cabinet and furniture making
Vinyl ester	as an alternative to polyester and epoxy materials, as thermoset polymer matrix in composite materials, manufacture of fiberglass reinforced plastics (FRP), tanks and vessels
Silicone	automotive, aerospace, coatings, cookware, defoaming, dry cleaning, electronics, firestops, lubricants, medicine, mold making, ophthalmology, personal care, plumbing and building construction, toys and hobbies
Phenol – formaldehyde resin	production of molded products including billiard balls, laboratory countertops, and as coatings and adhesives
Urea – formaldehyde	adhesives, finishes, particle boards, medium-density fiberboard, and molded objects.
Acrylic resins	ingredient in paints

*PTFE cannot be remoulded due to its high melt viscosity

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Table 2. **Examples of additives for the four main additives categories** (Zweifel *et al*. 2009; Hansen *et al*. 2013; Drobny 2014; Hahladakis *et al*. 2018; Andrade *et al*. 2021).

Additive category	Specific additive group	Example application	Chemical classes	Typical amount (%)
Functional additives influence specific properties such as stability against UV light and heat, resistance to microbes, flame retardancy, durability, softness, hardness, aesthetics, etc.	Light stabilizers	PE, PP, PVC	phenolic benzotriazoles (e.g., UV-328), cadmium compounds (e.g., cadmium oxide), lead and lead compounds	0.1-10
	Heat stabilizers	PVC	cadmium compounds, lead compounds, nonylphenol, barium and calcium salts	0.1-3
	Antistatic agents	PE films, PE and PP foams, PVC, PP injection moulding applications	long-chain alkyl phenols, ethoxylated amines, glycerol esters such as glycerol monostearate	0-1
	Antioxidants	LDPE, HDPE, high impact polystyrene (HIPS) and ABS	phenolic antioxidants, phosphites	0.05-3
	Flame retardants	Foam; plastic	brominated and chlorinated flame retardants (BFRs/CFRs)	2-28
	Plasticizers (softeners)	PVC, cellulose plastic	phthalates and short- and medium-chain chlorinated paraffins (SCCPs/MCCPs)	10-70
	Lubricants	PVC, PS/ABS, PP, PE	fatty acid esters, hydrocarbon waxes, metal stearates, amide waxes, ester waxes	0.1-3
	Slip agents	LDPE, PP	erucamide, oleamide, stearamide	0.05-0.15
	Curing agents (hardeners)	Epoxy resins	4,4’-Diaminodiphenylmethane, 2,2’-dichloro-4,4’-methylenedianiline	0.1-2
	Foaming agents			
	Nucleating agents			
	Blowing agents	PVC, PE, epoxy resins	C,C’-azodi(formamide) (ADCA); fluorinated greenhouse gases: hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulfur hexafluoride (SF6)	0.05-20
	Dispersing agents	Dispersing agent for the production of special plastics, e.g., PTFE	PFOA and similar compounds	
	Biocides	PUR; PVC	organic tin compounds, arsenic compounds; triclosan	0.001-1
	Fragrances	PE, PP, PS, PC, PET, EVA, PLA, nano particles	natural extracts; synthetic fragrances	
	Catalyst	PVC, PE, PP and other non-specified plastics, PUR	chromium and chromium compounds (e.g., chromium trioxide), mercury and mercury compounds	0.1-0.3
	Cross-linker	PE, PP	organic peroxides, BPA, formaldehyde	

continued on next page

Table 2 continuation

Additive category	Specific additive group	Example application	Chemical classes	Typical amount (%)
Colorants give color to the final plastic product	Pigments	PVC, PE, PP and other non-specified plastics	cadmium compounds (e.g., cadmium sulfide), chromium compounds; lead chromates	0.01-5
	Soluble azocolorants	polyethersulfone (PES), PA, acrylic	organic-based colorants (e.g., 573-58-0, 1937-37-7)	
Fillers occupy space in plastic materials without changing the functional properties; they replace expensive resins to reduce costs	Mica			
	Talc			
	Kaolin			
	Clay			
	Calcium carbonate			
	Barium sulphate			
Reinforcements are used to enhance mechanical properties such as the strength and elasticity of plastics	Glass fibers		glass fibers	15-30
	Carbon fibers		carbon fibers	

2.1 Chemicals of concern in plastics

Of the more than 13,000 chemicals associated with plastics and plastic production, 7,000 have been screened for their hazardous properties (Annex 1). Of these 7,000, more than 3,200 have been identified as substances of potential concern based on their hazardous properties (Aurisano *et al*. 2021b; Wiesinger *et al*. 2021). Hazardous properties considered include carcinogenicity, mutagenicity, reproductive toxicity, endocrine disruption, and ecotoxicity to aquatic organisms as classified according to the United Nations (UN)'s Globally Harmonized System of Classification and Labelling of Chemicals (GHS) and the European Union's Classification, Labelling and Packaging Regulation (CLP) (European Commission 2008a). Detailed reasoning for identifying the individual chemicals can be found in Annex 1.

Further studies, not included in Annex 1, found that over 1,000 different chemicals can migrate from plastic food contact materials into food or food simulants (Geueke *et al*. 2022). A related study by Zimmermann *et al*. (2022) identified 197 chemicals intentionally used in plastic food contact materials and hazardous according to the criteria set out by the European Union's Chemicals Strategy for Sustainability, including some considered carcinogenic, mutagenic, or toxic to reproduction (CMR). In addition, the chemical ingredients of more than 50% of plastics are considered hazardous according to GHS and CLP, with a range of monomers classified as mutagenic and/or carcinogenic (category 1A or 1B) (Lithner *et al*. 2011). Moreover, plastic debris is associated with 78% of the priority pollutants listed by the US EPA and 61% of those listed by the EU (Rochman *et al*. 2013a).

Described below are the most relevant groups of chemicals of concern due to their toxicity and potential migration from plastics (Rijk and Ehlert 2001; Noguerol-Cal *et al*. 2011; Mercea *et al*. 2017; Hahladakis *et al*. 2018; UNEP 2019i). An emphasis here is on persistent chemicals and related groups and on substances that are regulated either globally or in some countries. Additional IAS and NIAS are used and present in plastics, but have received less scientific and regulatory attention, often due to their better degradbility in the environment. However, they can also have hazardous properties and pose risks to human health and the environment. Of note, chemicals of concern can be present in both petroleum-based and bio-based/biodegradable plastics (Zimmermann *et al*. 2020).

For certain chemicals of concern in these groups, regulations have been developed in some countries or regions. The Stockholm Convention lists several additives as POPs, including some flame retardants used in plastics and other polymers (PBDEs, HBCDD, and HBB), as well as some PFASs (PFOS and PFOA and related precursor substances) that have been used in side-chain fluorinated polymers and in the treatment of synthetic carpets and textiles, for which the Convention provides a partial exemption (UNEP 2022a). Most recently, the POPs Review Committee (POPRC) of the Stockholm Convention adopted the decision to recommend the listing of UV-328, a UV stabilizer used widely in plastics, in Annex A to the Convention (elimination) (decision POPRC-18/2). However, developing countries often lack a regulatory framework to assess and restrict chemicals in consumer goods, including plastics (Weber *et al*. 2018a). In many countries, this is even true for the POPs listed with exemptions.

2.1.1 Flame retardants

Flame retardants are a class of additives used to reduce flammability and the spread of fire in plastics (e.g., EPS/PUR insulation; polyester textiles in vehicles) and other flammable materials (Shaw *et al*. 2010; Weil and Levchik 2016). They are used in many varied consumer products, such as electronic devices, vehicles and other transportation media, furniture, insulation foams, and certain textiles.

The main flame retardants used in plastics include:

- Brominated and chlorinated flame retardants (BFRs/CFRs), often used with antimony (Sb) as a synergist. BFRs include for example polybrominated diphenyl ethers (PBDEs), decabromodiphenyl ethane (DBDPE), hexabromocyclododecane (HBCDD), and tetrabromobisphenol A (TBBPA) (Shaw *et al*. 2010; Hahladakis *et al*. 2018); examples of CFRs include short-, medium- and long-chain chlorinated paraffins (SCCPs/MCCPs/LCCPs) (Shaw *et al*. 2010; Hahladakis *et al*. 2018) and dechloranes (Sverko *et al*. 2011);

- Organophosphorous flame retardants (OPFRs) such as tris (2-chloroethyl)phosphate (TCEP), tris(1,3-dichloroisopropyl)phosphate (TDCPP), tris(2-chloroisopropyl) phosphate (TCPP), tris(2-butoxyethyl) phosphate (TBOEP), triphenyl phosphate (TPhP) (Shaw *et al*. 2010; Van der Veen and de Boer *et al*. 2012; Chen *et al*. 2019);

- Inorganic flame retardants such as aluminum and magnesium hydroxides and boron.

Many organic flame retardants have documented toxicity concerns (Shaw *et al*. 2010; Van der Veen and de Boer 2012) and risk/benefit assessments favor the elimination of halogenated FRs (Shaw *et al*. 2010). For instance, PBDEs are of great concern for human health due to being ubiquitous, persistent, bioaccumulative and exhibiting various modes of toxicity including endocrine disruption, developmental neurotoxicity, and reproductive disorders (Z. Wu *et al*. 2020). TBBPA is a carcinogen (Norwegian Environment Agency 2020) and the most commonly produced BFR in the world (>50% of the BFR market) (EFSA 2011a). It is typically used as a reactive flame retardant and chemically bound to the polymer (Morris *et al*. 2004) but might leach out in small amounts (Brandsma 2014).

OPFRs such as TBOEP, TDCPP, and TPhP exert several toxicological effects and are associated with health problems, including adverse neurological (Dishaw *et al*. 2011), reproductive (Meeker and Stapleton 2010), and endocrine-disrupting effects (Xu *et al*. 2015). TDCPP was determined to be a carcinogen in mammals based on increased incidence of liver adenomas and carcinomas in rats chronically exposed to TDCPP for two years (Freudenthal and Henrich 2000). TDCPP is also genotoxic (Chen *et al*. 2018a). The metabolic products of TDCPP, including 1,3-dichloro-2-propanone and 1,3-dichloro-2-propanol, are powerful direct-acting mutagens (Gold *et al*. 1978; Andres, Appel, and Lampen 2013). TDCPP exposure substantially altered the pathways related to DNA damage, including the cell cycle, DNA replication, and various DNA repair pathways, and induced apoptosis (Chen *et al*. 2019).

There are also concerns about the efficacy of flame retardants because their use in some applications does not provide additional protection. Certain classes of flame retardants may not be effective, but rather counterproductive (DiGangi *et al*. 2010; Babrauskas *et al*. 2012). For example, BFRs can increase the toxicity of smoke due to smouldering, which releases more carbon monoxide (CO) (Di Gangi *et al*. 2010; McKenna *et al*. 2018)

In recent years, several flame retardants were listed as POPs under the Stockholm Convention, including PBDEs, HBCDD, hexabromobiphenyl (HBB), and SCCPs. Dechlorane plus was proposed for listing by the POPs Review Committee (POPRC) (UNEP 2019c; International Institute for Sustainable Development 2022), and MCCPs with carbon chain lengths C14–17 and chlorination levels at or exceeding 45% chlorine by weight are currently being evaluated by the POPRC (UNEP 2021f). Other emerging BFRs such as 1,2-bis (2,4,6-tribromophenoxy) ethane (BTBPE), decabromodiphenyl ethane (DBDPE), and hexabromobenzene (HBBz) have received increased attention, as they have been identified in many environmental compartments, in biota, and in humans (Covaci *et al* 2011; EFSA 2012). Most novel BFRs lack assessments for PBT properties (Hennebert 2021).

2.1.2 Per- and polyfluoroalkyl substances (PFASs)

The class of PFASs is composed of thousands of synthetic organic fluorinated substances that contain at least one fully fluorinated methyl or methylene carbon atom (without any H/Cl/Br/I atom attached to it) (OECD 2021). Prominent examples of long-chain PFASs are the perfluorocarboxylic acids (PFCAs) with carbon chain lengths C8 and higher, including perfluorooctanoic acid (PFOA), and the perfluoroalkane sulfonic acids (PFSAs) with carbon chain lengths C6 and higher, including perfluorohexane sulfonic acid (PFHxS) and perfluorooctane sulfonate (PFOS). A major use of PFOS, PFOA, and related substances (precursors) was in the manufacturing of side-chain fluorinated polymers such as fluorinated (meth)acrylate polymers, fluorinated urethane polymers, and fluorinated oxetane polymers (Buck *et al*. 2011; Henry *et al*. 2018). These polymers are used as linings for food containers and wrappers, and as surface treatments for plastic fibers in carpets, furniture, and other textiles to repel water, oil, and dirt (Glüge *et al*. 2020). They can be released as microfibers/microplastics during use and laundry (Wijnhoven *et al*. 2010; Schellenberger *et al*. 2022). Degradation of side-chain fluorinated polymers can release other PFASs, including PFOA or PFOS, depending on their synthesis pathway (Wang *et al*. 2010; Washington and Jenkins 2015; Henry *et al*. 2018). PFOA and increasingly other PFASs like GenX and ADONA have also been used as a surfactant in the production of fluoropolymers such as PTFE and polyvinylidene fluoride, with associated releases and contamination (Lohmann *et al*. 2020).

PFOS and PFOA do not follow the pattern of a classic POP – they do not accumulate in fatty tissues but instead bind to proteins (Beesoon and Martin 2015; Xu *et al*. 2018). Therefore, they accumulate mainly in organs such as the liver, kidney, brain, and spleen. In animal studies, PFOS has been shown to cause cancer, neonatal mortality, delays in physical development, and endocrine disruption (Luebker *et al*. 2005; Fuentes *et al*. 2007; Coperchini *et al*. 2016; Arrieta-Cortes *et al*. 2017). Higher levels of PFOS and PFOA are associated with delayed pregnancy (Fei *et al*. 2009) and reduced human semen quality and penis size (Joensen *et al*. 2009; Di Nisio *et al*. 2018). Toxicity data for most other PFASs are insufficient (Blum *et al*. 2015).

PFOS and related compounds have been listed under the Stockholm Convention since 2009. PFOA and related compounds and PFHxS and related compounds were listed in 2019 and 2022, respectively (Pinas, Van Dijk, and Weber 2020; UNEP 2022b). Long-chain PFCAs, their salts, and related compounds are currently being evaluated by the POPRC and were found to meet the POP criteria (UNEP 2022c). Furthermore, all PFASs are an issue of concern under SAICM. As a progressive regional approach, the EU has committed to phasing out the use of PFASs unless proven essential (European Commission 2020). A related proposal for a general restriction of PFASs under REACH is currently being evaluated (ECHA 2023b).

2.1.3 Phthalates

Phthalic acid esters or phthalates are a family of additives used as plasticizers, mainly in PVC production (Lowell Center for Sustainable Production 2011). PVC can contain 10-60% phthalates by weight, which are not covalently bound to the polymers (Net *et al*. 2015). Phthalates also add fragrance to products. An estimated 6 to 8 million tonnes of phthalates are consumed every year (Net *et al*. 2015).

Some phthalates can easily leach into the environment during plastic manufacturing, use and disposal, and have been found in a wide range of environments (Net *et al*. 2015). A review integrating results from *in silico*, *in vitro*, *in vivo*, and epidemiological studies concluded that phthalate exposure is associated with reproductive disorders with potential transgenerational or multigenerational effects (Hlisníková *et al*. 2020). Several phthalates are known endocrine disruptors that can interfere with the production of androgen (testosterone), a hormone critical to male development and also relevant to females (Legler *et al*. 2015; UNEP 2017d; UNEP 2017e). Di-2-ethylhexyl phthalate (DEHP) is classified as reprotoxic (category 1B) in the EU. An observation of 207 elderly men from Taiwan showed a positive association between the levels of DEHP metabolites in urine samples and benign prostate hyperplasia and prostatic enlargement (Chang *et al*. 2019). Concerning respiratory outcomes, phthalate concentrations in dust collected from children's bedrooms were associated with asthma and allergies in children (Bornehag *et al*. 2004). A study of over 1,500 Australian men found a positive association between total urinary phthalates and cardiovascular disease, type 2 diabetes, hypertension, and increased levels of biomarkers indicative of chronic low-grade inflammation (Bai *et al*. 2017). Legler *et al*. (2015)

calculated that in the EU each year phthalate exposure causes 53,900 cases of obesity in older women, with € 15.6 billion in associated costs, and 20,500 new-onset cases of diabetes in older women, with € 607 million in associated costs. Phthalate exposure in US adults was calculated to lead to 90,761–107,283 deaths and costs of $39.9–47.1 billion due to lost economic productivity (Transande, Liu, and Bao 2021).

The EU has restricted DEHP, dibutyl phthalate (DBP), benzyl butyl phthalate (BBP), and diisobutyl phthalate (DIBP) in toys, childcare articles, and all indoor and outdoor articles with prolonged contact with human skin (European Commission 2018b). Additionally, di-isononyl phthalate (DINP), Di-isodecyl phthalate (DIDP), and Di-n-octyl phthalate (DNOP) are restricted in toys and childcare articles that children can place in their mouths (ECHA 2014). Also, Brazil, Canada, Israel, and the US have restrictions for phthalates in children's toys or articles that young children may put in their mouths (US CPSC 2008; UNEP 2017e; Canadian Minister of Justice 2022). However, current regulatory limits for some phthalates are insufficiently protective, as they are based only on male reproductive toxicity and do not consider other, more sensitive health endpoints (Maffini *et al*. 2021). Although restrictions or labelling requirements for phthalates are in place in many developed countries, there are hardly any in most developing countries or countries with economies in transition (UNEP 2017e).

2.1.4 Bisphenols

Bisphenols are a group of aromatic compounds with two hydroxyphenyl functionalities. They are the monomers of polycarbonate plastic products (including water bottles, food storage containers and packaging, sports equipment, and compact discs) and of the epoxy resin liners of aluminium cans. They are also used as a developer in thermal paper products such as cash register receipts (Michalowicz 2014; EFSA 2015; UNEP 2019i). Polymer degradation is the dominant mechanism responsible for bisphenol releases from products and subsequent exposure (Mercea 2009; Danish Ministry of Environment 2015).

BPA is the most widely produced and representative member of the bisphenol chemical class. It is one of the most commonly produced chemicals worldwide, with over three million tonnes annually (Laing *et al*. 2016). 65% of the volume of BPA produced was used as a monomer for PC plastics, while 30% was used for epoxy resins (Crain *et al*. 2007). BPA can also be used as an additive, for instance as an antioxidant or plasticizer in other polymers (e.g., PVC) (France 2016). Other bisphenol analogues such as bisphenol F (BPF) and bisphenol S (BPS) are also used in plastics and have been detected in foodstuffs, personal care products, house dust, sediments of rivers and lakes, and in human biological specimens. Accordingly, they are directly available for exposure and may also represent a threat to human health and the environment (Pelch *et al*. 2019).

BPA is a known reproductive toxicant that impacts female reproduction and has the potential to affect male reproductive systems in humans and animals (Michałowicz 2014; Peretz *et al*. 2014; EFSA 2015, EFSA 2021; Kortenkamp *et al*. 2022). BPA also contributes to obesity (Kim *et al*. 2019; X. Wu *et al*. 2020). Although less well-studied, BPB, BPS, and BPF also appear to have BPA-like effects. The hormonal pathways disrupted by BPS manifest in many ways in animal studies: changed uterine growth, shifts in both male and female sex hormone concentrations, and reproductive disruptions including changes to egg production and sperm count (Naderia *et al*. 2014; Rochester and Bolden 2015), as well as statistically significant weight gain and altered hormone metabolic profiles (Del Moral *et al*. 2016). BPS alters maternal behaviour and the brain in mice exposed during pregnancy and lactation, as well as the behavior and brain of the daughters of exposed mothers (Catanese and Vandenberg 2017).

Recent receptor-binding studies indicate that BPS is about as potent as BPA when acting through at least one of the estrogen receptors (Chen *et al*. 2016). These studies are complemented by animal tests that show the effects of BPF on uterine growth and testes weights, demonstrating impacts on the estrogen and androgen pathways, respectively (Higashihara *et al*. 2007). BPF, like BPA, also appears to disrupt thyroid pathways (Kim and Park 2019).

A recent review concluded that hazard and risk characterizations should consider all BPA structural and functional analogues (Pelch *et al*. 2019). The EU Commission developed a restriction roadmap including bisphenols as a group (European Commission 2022a) and ECHA compiled a related document on the assessment needs (ECHA 2021a). BPA and BPB are listed as Substance of Very High Concern (SVHC) under REACH (ECHA 2018a; ECHA 2021b). Some plastic products containing BPA, in particular baby bottles, are restricted in several countries, for instance in Brazil, the EU, India, Israel, and South Africa (European Commission 2011b; UNEP

2017e). In December 2021, the European Food Safety Authority (EFSA) called into question the current safety limits for consumer exposure to BPA, and it is considering recommending that the allowed daily intake of BPA should be reduced by a factor of 100,000 to sufficiently protect public health (EFSA 2021).

2.1.5 Certain alkylphenols and alkylphenol ethoxylates (APEOs)

Nonylphenol (NP) is commonly used as a stabilizer and intermediate in plastics production. 4-tert-Octylphenol has major uses in formaldehyde resins.

NP is an endocrine disrupting chemical and suspected to be toxic to reproduction (ECHA 2022a). Biomonitoring studies have demonstrated the widespread human exposure to NP (Ademolla *et al*. 2008; Lopez-Espinosa *et al* 2009; Ringbeck *et al*. 2022). 4-tert-Octylphenol is a potential environmental pollutant and exhibits estrogenic and anti-androgenic activity in human reporter cell lines (Paris *et al*. 2002).

In 2021, the US state of Maine published a list of ten food contact chemicals of high concern, including NP and 4-octylphenol. Furthermore, 4-tert-octylphenol as well as branched and linear 4-nonylphenol are included on ECHA's SVHC list (ECHA 2011; 2022b).

2.1.6 Polycyclic aromatic hydrocarbons (PAHs)

Polycyclic aromatic hydrocarbons (PAHs) are a class of chemicals that occur naturally in coal, crude oil, and gasoline and are formed during pyrolysis or incomplete combustion processes. PAHs are organic compounds consisting of multiple aromatic rings and refer to a mixture of over one hundred substances. They can be found in direct-contact consumer goods made of rubber or elastomers, including children's toys (BfR 2009). This is due to the use of PAH-containing plasticizer/extender oils or carbon black, which are added during the production of rubber or elastomers to provide the materials with different mechanical and process-related properties (BfR 2009). Also, EPS contains non-intentionally added PAHs, with the highest anthracene levels in new EPS, likely from production (Yeo *et al*. 2022). Apart from production, PAHs can also absorb to PS during use (Li, Ni, and Zeng 2017). For instance, in virgin PS pellets, 79 – 97 ng PAHs/g pellet were measured (Rochman *et al*. 2013b) while the concentrations for finished food contact articles reached up to 475 ng/g (Li, Ni, and Zeng 2017).

Numerous PAHs are classified as carcinogenic, mutagenic, or toxic for reproduction (CMR substances) (BfR 2009; RIVM 2018). Eight PAHs are classified as carcinogens category 1B (JRC 2018) and restricted in the EU under REACH in plastic and rubber parts of articles that can be used by consumers, particularly by children (ECHA 2018b). Under the EU REACH restriction, each of these eight listed PAHs has an allowable limit of 0.5 mg/kg in plastic and rubber components in toys and childcare articles, and 1 mg/kg in all other articles (ECHA 2018b).

2.1.7 Biocides

Biocides are antimicrobial substances classified by their target organism e.g., bacteriocides, fungicides, insecticides, and rodenticides (Geueke 2014a). They are applied to plastic materials as additives in the formulation to protect the plastic from attack and degradation by microorganisms. Examples of biocides applied as additives are organic tin compounds (e.g., tributyltin (TBT) and bis(tributyltin)oxide), usually applied in PUR foam (PUF) and PVC; arsenic and arsenic compounds (e.g., 10,10'-oxybisphenoxarsine), mostly applied in plasticized PVC, PUR, LDPE and polyesters; and triclosan, usually applied in PE, PP, PVC, polyester and polyamide fibers (Hansen *et al*. 2014; Davoodi *et al*. 2016). Furthermore, they are used in plastic FCMs to control and reduce the number of microorganisms on the food itself and on materials coming into contact with the food (Geueke 2014a). Biocides applied in or on FCMs include alcohols, organic acids and their esters, aldehydes, amines, quarternary ammonium compounds (QACs), halogen compounds, ionic silver and nanosilver, oxidizing agents, isothiazolones, phenols and biguanides (Geueke 2014a).

Biocides are also used as additives in bioplastic materials (Gómez-Heincke *et al*. 2016). Other polymers are often resistant to attack by microorganisms. Nevertheless, some microorganisms are capable of using additives like plasticizers as an energy source in the presence of water. Thus, antimicrobial substances can also be added to plastics to prevent microbial degradation of some of the other additives (Hansen *et al*. 2014). In addition, biocides are used to boost the antimicrobial properties in new-generation products such as functional, smart, or active packaging (e.g., Nguyen Van Long *et al*. 2016), or more generally to prevent the degradation of the plastic material (Groh *et al*. 2019). The major fields of application are deemed to be plasticized PVC material (e.g., synthetic leather, plastisol prints, or synthetic shoe insoles), followed by PUR coating and binders, PE and polyester.

Most biocides pose environmental hazards since they strongly interact with living organisms, but some are also classified as hazardous to human health (Groh *et al*. 2019). Many biocides are irritants and sensitizers and act on the skin, eyes and mucous membranes. They can, for instance, lead to allergic contact dermatitis and asthma, in particular for occupational users (Geueke 2014a). TBT has very high toxicity for aqueous organisms. It is also genotoxic and shows endocrine effects.

Biocides in packaging and other uses are regulated in industrial countries. In the US, antimicrobials used in or on food packaging, material preservatives, and non-functional antimicrobial components in food contact articles are regulated as food additives by the US Food and Drug Administration (US FDA) under the Federal Food, Drugs, and Cosmetics Act, § 348 (US FDA 2018). While in Europe, a number of biocides are restricted by the Biocidal Products Regulation (Regulation (EU) 528/2012) (European Commission 2012b).

2.1.8 UV stabilizers

UV stabilizers prolong the lifespan of plastics by protecting them from photo-degradation initiated by UV light, absorbing radiation before it reaches photosensitive moieties in the polymer (Zweifel and Amos 2001; Tylkowski *et al*. 2017). They decrease weatherability through light stabilization or UV absorption. Benzophenones, benzotriazoles (BZTs), hindered amine light stabilizers (HALS), and their combinations are the most common UV stabilizers for the plastic industry (Zweifel and Amos 2001; Cangelosi *et al*. 2002; BASF 2019).

Benzophenones are natural components of plants that absorb light in the range of 290 – 340 nm. Different benzophenones have different characteristics related to varied substitutions (Heurung *et al*. 2014). The most popular derivative of this class is oxybenzone or 2-hydroxy-4-methoxybenzophenone (BP-3) (Asimakopoulos *et al*. 2016). Benzophenones are used in polyolefins and PVC. BZTs absorb a wide range of UV light; UV-320, UV-327, UV-328, and UV-350 absorb strongly in the range between 290 and 350 nm. BZTs dissipate absorbed energy without oxidizing the polymer. Due to their mechanism of action, their use is reversible and non-destructive. Like benzophenones, BZTs are technically important UV absorbers in transparent plastic materials (Kirk *et al*. 2007). They are used in many consumer plastic products (Nakata *et al*. 2009; Kim *et al*. 2011a; Avagyan *et al*. 2015). In general, BZTs are used in ABS, HIPS, PVC, PES, PC, POM, PMMA, polyvinylbutyral (PVB), and PUR fibers (Mikkelsen *et al*. 2015). HALS are used in polyolefins, styrenics, polyamides, polyurethanes, polyacetals, adhesives, and sealants.

Benzophenones are of concern due to their potential ecotoxicity. Some derivatives exhibit estrogenic and hormonal activities *in vitro* (Zhao *et al*. 2013). BZTs are also potential EDCs (Sakuragi *et al*. 2021). UV-328 is classified as having specific target organ toxicity upon repeated exposure (STOT RE 2), due to its ability to cause liver toxicity in mammals (Stockholm Convention 2022). The most commonly used HALS, Tinuvin 770 and Chimassorb 944, are classified as having high toxicity (class III) according to Cramer rules.

Benzophenones were named Allergen of the Year in 2014 by the American Contact Dermatitis Society due to their potential to induce allergies (Heurung *et al*. 2014). BP-3 is listed in the REACH database as a possible EDC (ECHA 2015b). UV-320, UV-327, UV-328, and UV-350 have already been classified as SVHC in the EU and are on the list of substances subject to Authorisation under EU REACH due to their PBT/vPvB properties (ECHA 2019b). UV-328 was recently evaluated by the POPRC of the Stockholm Convention (UNEP 2020a), which affirmed that UV-328 meets the Annex D criteria for POPs properties (UNEP 2021a) and recommended its listing as a POP in Annex A to the Convention (elimination) (International Institute for Sustainable Development 2022). Chimassorb 944 is regulated in food contact plastic under EU legislation, with a limit of 3 mg/kg foodstuff. Tinuvin 770 is not allowed in food contact packaging in the EU, but the US FDA authorizes restricted use, below 0.1% by weight (Noguerol-Cal *et al*. 2010).

2.1.9 Metals and metalloids

A particularly prominent group of hazardous additives consists of substances containing metals, including antimony, cadmium, chromium, lead, mercury, cobalt, tin, and zinc. Cadmium- and zinc-containing substances have been used as stabilizers in PVC and some other plastics (zinc is safer but less efficient than cadmium). Heavy metals have been used to dye polymers because they can form complexes with strong colours (Groh *et al*. 2019). Some examples of heavy metal-based compounds used as colorants are cadmium compounds (e.g., cadmium sulphide and cadmium selenide), which can convert plastics of all colors, especially yellow, orange, and red, into green, brown, beige, etc. Chromium and chromium compounds can impart yellow, red, and green colours in PVC, PE, and PP. Lead chromate molybdate sulphate red can be applied to all types of plastics where red pigments are used (Hansen *et al*. 2014). The typical amount of heavy metal-based colorants in plastics ranges between 0.01 and 10% (Hansen *et al*. 2013). Metal-containing substances are also used as antimicrobials, accelerators, and catalysts (e.g., chromium trioxide) (Zweifel *et al*. 2009). Mercury catalysts were and may still be used in the production of PUR coatings, adhesives, sealants, and elastomers, with a final mercury content of 0.2% in PUR (COWI 2008).

Apart from frequent classifications for aquatic toxicity, some of these chemicals are also ranked high for human health hazards, mostly because of CMR properties along with a range of severe effects on multiple organs, including the brain and nervous system (Jaishankar *et al*. 2014). For instance, occupational antimony exposure may cause respiratory irritation, pneumoconiosis, and gastrointestinal symptoms (Sundar and Chakravarty 2010). Antimony trioxide is classified as possibly carcinogenic to humans (Group 2B) (IARC 1989). Methylmercury, the organic form of mercury, is associated with neurocognitive deficits in human fetuses and cardiovascular effects in adults (Axelrad *et al*. 2007; Roman *et al*. 2011). Cobalt exposure may result in systemic health effects including neurological, cardiovascular, and endocrine deficits (Leyssens *et al*. 2017).

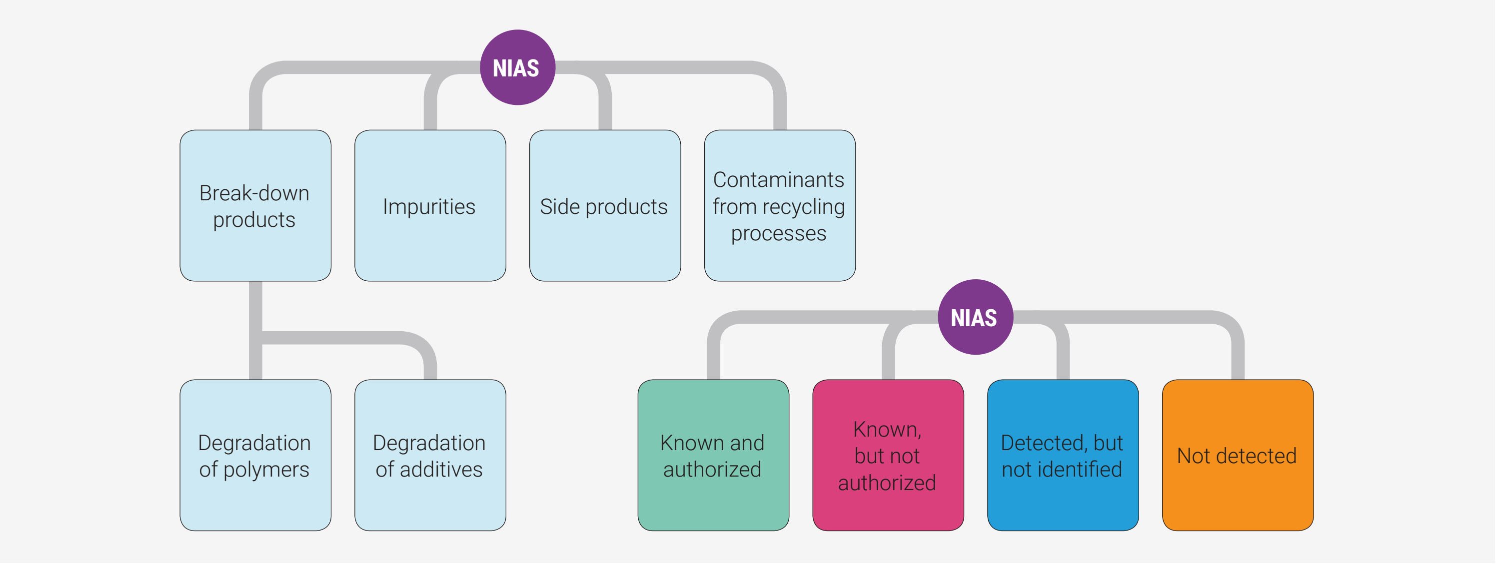
Due to their severe health impacts, heavy metal compounds in plastics are considered an issue of concern to authorities. For example, use of the four most hazardous metals (cadmium, hexavalent chromium, lead, and mercury) in plastic packaging has been regulated since 1994 under European Commission (EC) Directive 94/62/EC on packaging and packaging waste (European Commission 1994). In 2022, the Food Safety and Standards Authority of India (FSSAI) amended its Food Safety and Standards (Packaging) Regulations to limit the overall migration of antimony from plastic packaging to 0.04 mg/kg food (FSSAI 2022).

2.1.10 Non-intentionally added substances (NIAS)

NIAS are specifically mentioned in the EU's plastic FCM Regulation (EC 10/2011). As such, their evaluation has become increasingly of interest, but remains analytically challenging (Nerin *et al*. 2013; Kato and Conte-Junior 2021). NIAS originate from different sources and include breakdown products of polymers, impurities in starting materials, unwanted side-products, and various contaminants from recycling processes (Figure 3) (Geueke 2013a; Geueke 2018):

- **Breakdown products:** Plastic materials are subjected during their lifetime to a combination of heat, oxygen, light, radiation, moisture, and mechanical stress. Plastic degradation is mainly caused by photooxidation processes (Rabek 1996; Yousif and Haddad 2013) and results in the formation of NIAS. The breakdown products can be separated into degradation products from polymers, processing aids, and additives

Figure 3. Sources and categories of non-intentionally added substances (Geueke 2013a). ▼



(Geueke 2013a). A range of hazardous volatile organic compounds (VOCs) such as toluene, xylene, or ethylbenzene are degradation products, for instance from PET/polyester (Kato and Conte-Junior 2021).

- **Impurities:** Raw materials and additives that are used to produce plastic often contain impurities. The quality of the starting materials can vary from batch to batch. Main impurities are generally known to the producer, whereas minor impurities are often unknown (Bradley and Coulier 2007).
- **Side reactions:** Plastics are often produced in several steps, and they usually contain additives such as UV stabilizers, antioxidants, and pigments. Side-reactions occur during the production of the starting substances, materials, and additives. Besides the main chemical reaction pathways, other reactions and transformations are possible, resulting in the formation of NIAS (Geueke 2013a).
- **Contaminants from recycling:** Products made from recycled materials are a special source of NIAS. First, plastics entering recycling processes usually contain a variety of substances such as additives, dyes, and degradation products that can contaminate products made from these recycled materials. Recycling can thus also cause banned hazardous chemical additives (legacy additives) to enter new products. For example, the recycling of pesticide packaging can transfer the pesticides to new plastic products (Eras *et al*. 2017). Second, impurities from waste management may still be present in the recycled product. Moreover, chemicals present during the recycling process can react, forming additional NIAS. With each recycling cycle, the amount and diversity of chemicals may increase (Geuke 2013a).

A specific NIAS group consists of unintentional POPs such as polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans (PCDD/PCDFs), PCBs, PCNs, and hexachlorobenzene (HCB), which are regulated in products by some countries (Japanese Ministry of Economy, Trade and Industry 2013; UNEP 2017c; UNEP 2021b) and in wastes by the Basel Convention's low POP content limit values (UNEP 2019d). Unintentional POPs in plastics can originate from additives such as chlorinated paraffins (Takasuga *et al*. 2012; Takasuga *et al*. 2013), from color pigments (Anezaki *et al*. 2015; Government of Japan 2006; Government of Japan 2007), from degradation products of polymerization catalysts (Herkert *et al*. 2018), or as byproducts of biocides (Halden *et al*. 2017). Polybrominated dibenzo-p-dioxins and polybrominated dibenzofurans (PBDD/PBDFs) are NIAS in plastics that contain PBDEs or other BFRs (Weber and Kuch 2003; Sindiku *et al*. 2015a) and are transferred via recycling into new products, including toys (Petrlik *et al*. 2018; Budin *et al*. 2020) and food packaging (Miralles *et al*. 2021).

In the EU, NIAS have been explicitly mentioned and defined in the regulation on plastic food contact materials since 2011 (EC 10/2011). The current European legislation acknowledges that NIAS are present in food contact materials. According to Article 3 of the Framework Regulation EC 1935/2004, the manufacturer is obliged to ensure the safety of NIAS. To achieve this, NIAS have to be identified, quantified, and assessed toxicologically. However, this is practically impossible (Félix *et al*. 2012). Similarly, the Chinese Standard GB 4806.1 includes a definition of NIAS. Although this standard requires manufacturers to confirm the safety of NIAS in food contact materials, their explicit approval is not required (Manoukian *et al*. 2017).

2.2 Priority use sectors

The sectors discussed in the following sections are of particular concern because one or more of the chemicals of concern identified above are known to be present in plastic products made by these sectors, and because these specific plastic applications result in either relatively high likelihoods of chemical exposure in humans and ecosystems or exposures in vulnerable populations. These sectors include consumer goods such as food and beverage packaging, personal care and household products, and textiles; long-term applications such as in buildings, vehicles, and furniture; applications associated with sensitive exposures such children's toys and medical devices; and plastics in agriculture, aquaculture, and fisheries that are directly released in the environment (Figure 4).

2.2.1 Toys and other children's products

An estimated 90% of toys available on the market are made of plastic (UNEP 2020b), which may contain chemicals of concern. The polymers most commonly used in toys including PVC, ABS, HIPS, PP, PE, and PUR (Aurisano *et al*. 2021a). Children often put toys, fingers, and other objects in their mouths as part of their normal development (Inter-Organization Programme

Figure 4. Priority use sectors where chemicals of concern have been found in plastic products and where plastic applications can lead to relatively high likelihoods of chemical exposure, exposures in vulnerable populations, or environmental pollution. ▲

for the Sound Management of Chemicals [IOMC] 2019). This mouthing behavior in combination with low body weight can lead to higher chemical exposure. Additionally, due to their ongoing developmental processes, children are generally more vulnerable to chemical risks.

Children's plastic toys and other children's products can contain EDCs such as phthalates or BPA (Ionas *et al*. 2014; Weber *et al*. 2018a). POPs have also been detected in those types of products, such as SCCPs in PVC toys and rubber parts (Miller and DiGangi 2017; UNEP 2019e), and PBDEs likely stemming from the recycling of PBDE-containing plastic (Chen *et al*. 2009; Ionas *et al*. 2014). PBDEs and other plastic additives in toys made of recycled plastic can lead to exposures above the reference dose or Tolerable Daily Intake (TDI) for children (Chen *et al*. 2009; Ionas *et al*. 2016; DiGangi *et al*. 2017). PBDE-containing plastic toys can additionally contain brominated dioxins in high concentrations (Petrlik *et al*. 2018; Budin *et al*. 2020).

Furthermore, studies have found concerning concentrations of a wide range of heavy metals (e.g., lead, cadmium, arsenic, chromium, zinc, selenium, mercury, barium), sometimes exceeding the limits set by the former European toys regulations (European Council Directive 88/378/EEC on toy safety). These studies tested vintage plastic toys from the 1970s and 1980s (Miller and Harris 2015), second-hand children's plastic toys in the UK (Turner 2018a,b), low-priced toys in Malaysia (Ismail *et al*. 2017), and new children's plastic toys and modelling clays (Korfali *et al*. 2013). Heavy metals used as additives in PVC were detected in 30% of toys tested from different world regions (IPEN and GRID-Arendal 2013).

The US Consumer Product Safety Improvement Act of 2008 prohibits concentrations over 0.1% for three plasticizers in children's toys and childcare products (US CPSC 2008), and the European Commission Regulation (EU) 2018/20051 of December 2018 restricts several phthalates (DEHP, DBP, BBP, DIBP, DINP, DIDP, DNOP) listed in REACH Annex XVII. European Directive 2009/48/EC also restricts CMRs in children's toys. However, developing and emerging economies often do not regulate chemicals in toys (Weber *et al*. 2018a).

2.2.2 Packaging, including food contact materials

Both intentionally and non-intentionally added substances (i.e., IAS and NIAS) present in plastic-based food contact materials (FCMs) can transfer into foodstuff, likely contributing substantially to human exposure to chemicals (Grob *et al*. 2006; Rudel *et al*. 2011; Hutter *et al*. 2016; Biryol *et al*. 2017). The extent of chemical migration from food packaging into foodstuff depends on the physicochemical properties of the chemical and the plastic FCM, the nature of the food (i.e., fat-soluble chemicals will migrate faster into fatty foodstuff than into aqueous foodstuff), as well as the temperature and storage time (i.e., migration increases with longer storage time and higher temperatures). Packaging materials are also a major contributor to marine plastic litter and marine pollution (UNEP 2016), including due to the additives they contain (Gallo *et al*. 2018; see Chapter 3).

Groh *et al*. (2021) identified and prioritized 325 hazardous substances known to be potentially used in the manufacture of plastic FCMs, and Geueke *et al*. (2022) found that over 1,000 different chemicals are known to migrate from plastic food contact materials into food or food simulants. The exact number of hazardous chemicals in plastic FCMs is likely much higher since most IAS have not yet been fully tested for their toxicity and many NIAS are still unknown. Legacy additives such as PBDEs from flame retarded plastic (from e-waste) were found in FCMs as NIAS due to recycling (Samsonek and Puype 2013; Rani *et al*. 2014; Ionas *et al*. 2014; Puype *et al*. 2015). The high exposure potential for humans makes controlling potentially harmful IAS and NIAS in plastic FCMs a high priority.

In the EU, for example, chemicals in plastic FCMs are regulated by the Commission regulation EU No 10/2011 on plastic materials and articles intended to come into contact with food (European Commission 2011a). Annex I to this regulation established a list of authorised substances (Union List) that may be used in the manufacture of plastic materials and articles intended to come into contact with food. Furthermore, the European Commission (2013a) developed dedicated guidance for plastic FCMs to help stakeholders with the interpretation and implementation of the EU No 10/2011 regulation.

2.2.3 Electrical and electronic equipment

A large proportion of high-quality plastics is used in electrical and electronic equipment (EEE). About 10 million tonnes of plastic waste are contained in the 50 million tonnes of e-waste generated each year (World Economic Forum 2019). While the quality and value of the plastic may be high, the recycling rates are low due to the presence of legacy additives listed as POPs (in particular, PBDEs, but also HBB and HBCDD) for post-consumer plastic and the complexity of the EEE plastic mixture, which needs advanced separation techniques (UNEP 2021c). Hazardous substances within the life cycle of EEE were highlighted as an issue of concern under SAICM (SAICM 2012a,b).

Inefficiently and poorly sorted waste electrical and electronic equipment (WEEE) plastic may introduce restricted and potentially harmful substances into the recyclate. Plastics containing restricted legacy additives and other potentially harmful chemicals are frequently recycled into consumer products – often into black plastic (Turner 2018a,b) – including sensitive product categories such as toys, kitchen tools and other FCMs (Chen *et al*. 2009; Ionas *et al* 2014; Puype *et al*. 2015; Kuang *et al*. 2018). In addition to POP-BFRs, other chemicals can also be reintroduced into consumer products by recycling, e.g., antimony (a flame retardant synergist) and heavy metals including cadmium, chromium, mercury and lead (Turner 2018a,b; Turner 2019).

The Commission Regulation (EU) 2019/2021 that establishes ecodesign requirements for electronic displays banned all halogenated flame retardants in the enclosure and stand of electronic displays (European Commission 2019a).

2.2.4 Transport

The plastic and polymer content of the transport sector has been increasing. While in the 1980s a passenger car contained an average of 100 kg of plastic, this increased to 200 kg around 2014 and is expected to further increase (Szeteiova 2010; IHS Markit 2016). Although a wide range of different polymers may be used in a single car, three types of plastics make up approx. 66% of the total plastics used in cars: PP (32%), PUR (17%) and PVC (16%) (Szeteiova 2010).

The emission of VOCs from automobile cabin materials, in particular plastics including synthetic textiles, is one of the main causes of poor vehicular air quality (Fedoruk and Kerger 2003; Xiong *et al*. 2015; Zulauf *et al*. 2019). Aromatic compounds are found at especially high levels in automobile cabin air, particularly toluene, ethylbenzene, o-xylene, m-xylene, p-xylene, styrene, and 1,2,4-trimethylbenzene (Yoshida 2010), which are NIAS for instance in polyester, or IAS from PS (Kato and Conte-Junior 2021). One study found that the amount of styrene in car indoor air was approximately 1.5 times the tolerable daily intake (7.7 µg/kg per day) recommended by the World Health Organization (Yoshida 2010). Pollutants in automobile cabin air can lead to sick car syndrome, particularly in new cars (Kim *et al*. 2011b). The car age is the most important factor influencing airborne VOC pollution, followed by in-cabin temperature. This indicates that outgassing from plastics and other polymers decreases over time (Chen *et al*. 2014; Zulauf *et al*. 2019).

Major plastic additives detected as indoor pollutants inside cars are BFRs, OPFRs and phthalates. For decabromodiphenyl ether (DecaBDE) and other PBDEs listed as POPs, ingestion/inhalation of dust inside cars is considered a major exposure pathway (Imm *et al*. 2009; Harrad and Abdallah 2011). Novel BFRs have also been detected inside cars (Besis *et al*. 2017). A wide range of OPFRs was identified in dust samples gathered from the car interior (Tokumura *et al*. 2017; Christia *et al*. 2018). Tris (2-ethylhexyl) phosphate and TCPP had the highest concentrations, with 390 mg/kg (in dust from car seats) and 640 mg/kg (in dust from car floor mats) (Tokumura *et al*. 2017). Various phthalates were detected in indoor air in cars and on the seats, but at levels below reference doses (RfDs) (Geiss *et al*. 2009; Perez *et al*. 2017).

Major environmental releases include microplastics and related chemicals from tires (Tamis *et al*. 2021), such as the antioxidant 6 p-phenylenediamine (6PPD) that forms 6PPD-quinone, which is acutely toxic to coho salmon and other fishes (Brinkmann *et al*. 2022). At end-of-life, plastic waste including additives is largely released to the environment at shredding facilities (including through fires), via disposal of light car shredder residues, or through open burning of plastic parts in developing countries (Zimring 2011; Takahashi *et al*. 2017; California EPA 2021). Upon burning, plastic particles and chemicals can end up in the air, as does street dust (including tire wear particles) when re-entrained. This way, plastic-associated chemicals may be transported over long distances and redeposited, leading to their widespread distribution. Furthermore, tire wear and associated chemicals are resuspended in stormwater runoff, contaminating adjacent watersheds (Calvillo, Williams, and Brooks 2015, Müller *et al*. 2020)

In airplanes, flame retardants are widely used in many plastic/polymer components and all airplane types (Allen *et al*. 2013). Most flame retardants, including TDCPP, were detected in 100% of dust samples collected from 19 airplanes. The concentrations of BDE-209 were approx. two orders of magnitude higher compared with residential and office environments (Allen *et al*. 2013).

The automotive industry (including its supplier base in the chemicals and plastics industries) has developed the Global Automotive Declarable Substance List (American Chemistry Council n.d.). The list covers substances that are addressed by substance-specific legislations worldwide and are expected to be present in materials and parts in vehicles at point-of-sale. This approach is a voluntary industry initiative with the purpose to facilitate communication and exchange of information regarding the use of certain substances in automotive products throughout the supply chain.[2]

2.2.5 Synthetic textiles

Polyester, nylon, acrylic and other synthetic fibers are all different forms of plastic that now account for over 60% of the material that makes up our clothes worldwide. In 2016, 65 million tonnes of plastic were produced for textile fibers (The Fiber Year 2017). Synthetic plastic fibers are cheap and extremely versatile, providing stretch, breathability, warmth and sturdiness. On the other hand, as synthetic fibers burn more readily than natural ones, they require more flame retardants (Horrocks *et al*. 2005; Shaw *et al*. 2010; Weber *et al*. 2018a).

Several POPs (e.g., PBDEs, HBCDD, SCCPs, PFOS, and PFOA) have been used for textiles, seating and other components in vehicles, trains, ships, offshore installations, as well as for textiles such as curtains, carpets, and mattresses for use in public spaces like hotels, healthcare institutions, jails etc (Puzyn and Mostrag 2012). DecaBDE and PFOA have received exemptions for use in textiles in Stockholm Convention. Specifically, PFOA can be used to impart oil and water repellency in textiles if needed for the protection of workers from dangerous liquids that pose risks to their health and safety. DecaBDE can be used in textile products that require anti-flammable characteristics, excluding clothing and toys.

These chemicals are not labelled on textile products, rendering it virtually impossible for consumers to make informed purchasing decisions and for recyclers to recycle these products safely. As a result, consumers have no information about the POPs content of the textiles they purchase, while governments and textile recyclers might not know whether they are complying with the Stockholm Convention's prohibition on recycling products containing POPs.

2.2.6 Furniture

The polymers used in furniture can be a significant source of human exposure to POPs and other hazardous additives. The use of PBDEs and OPFRs in furniture is thought to contribute to high levels of exposure among the general population, particularly for children, and to increased risks for adverse health outcomes (Hale *et al*. 2003; Birnbaum and Staskal 2004; Cooper *et al*. 2016). Dermal contact with furniture fabrics can become a significant pathway of human exposure to HBCDD, exceeding the exposure from most other sources (Abdallah and Harrad 2018).

BFRs and OPFRs have been added to furniture PUF to meet required state and federal flammability standards. PUF in furniture was a main source of PBDE exposure for the North American population, which is likely different in other regions with different flammability standards (Charbonnet *et al*. 2020). However, some flame retardants associated with health hazards (e.g., PBDEs and TDCPP) are now restricted from use in some regions.

Flammability standards for furniture are driving the use of flame retardants. The US and UK had particularly high use of flame retardants in furniture (UNEP 2017a; Charbonnet *et al*. 2020). PUF, in addition to being flammable, forms carbon monoxide, hydrogen cyanide, and other toxic products during decomposition and combustion (McKenna and Hull 2016). The use of flame retardants in furniture produced significantly greater quantities of carbon monoxide and hydrogen cyanide, and therefore increased smoke toxicity due to smouldering (McKenna *et al*. 2018). California amended its fire safety law TB-117 to require a smolder test instead of an open flame test for compliance purposes, for which enforcement began in 2014 (Charbonnet *et al*. 2020). Due to the change in this flammability standard, the use of flame retardants in furniture has decreased in the US (Cooper *et al*. 2016). Furthermore, California requires that furniture products containing flame retardants are labelled as such (State of California 2019).

Furniture can have a long service life that can be prolonged by second-hand use. The reuse of furniture containing commercial PentaBDE or TDCPP with known exposure risk should be avoided or restricted (Stapleton *et al*. 2008; Imm *et al*. 2009; Betts 2015).

2.2.7 Building materials

A major use of plastic and other polymers is in construction. Large volumes of polymer foams are used as insulation in buildings and other construction. The main polymer foams used are expanded and extruded polystyrene (EPS/XPS), PUR and polyisocyanurate. These foams are often flame retarded with OPFRs, and sometimes with BFRs, to meet flammability standards (Charbonnet *et al*. 2020). HBCDD, used in EPS/XPS until recently, was listed as a POP under the Stockholm Convention in 2013, with a specific exemption for use in insulation in construction. DecaBDE was listed as a POP in 2017, with an exemption for use in PUF in building insulation. Other polymers used in construction and treated with decaBDE or other flame retardants include PE insulating foam, PE plastic sheeting and PP plastic sheeting.

PVC, a major polymer used for instance in flooring, roofing and cables, contains plasticizers such as phthalates or SCCPs/MCCPs (Petersen 2012; Just *et al*. 2015; UNEP 2021g). PVC in flooring was associated with increased levels of butylbenzyl phthalate (BBzP) in indoor air and children's urine. Higher BBzP metabolite concentrations were associated with an increased incidence of childhood asthma (Whyatt *et al*. 2014).

SCCPs/MCCPs are also still used in the sealants and paints (UNEP 2019e; UNEP 2021g). PCBs used in older constructions until the 1970s in cables, sealants and paints still pose significant exposure risks in impacted buildings (Meyer *et al*. 2013; Lehmann *et al*. 2015; UNEP 2017c). The presence of PCBs in legacy polymer sealants and coatings impacts the recycling of construction and demolition waste (Weber *et al*. 2018b). Newer sealants can also contain hazardous chemicals such as 2,4-di-tert-butyl-6-(5-chlorobenzotriazole-2-yl)phenol in silyl-modified polyether, as well as TCPP and SCCPs/MCCPs in spray PUF (Nilson *et al*. 2004; Bello *et al*. 2018; Brandsma *et al*. 2021; Chen *et al*. 2021).

Plastic additives and other chemicals in building materials can be emitted indoors, deteriorating the indoor air quality, accumulating in house dust and resulting in human exposure (Lucattini *et al*. 2018). A recent high-throughput screening of chemicals used in building materials identified 55 chemicals of high concern mainly from plastics and other polymeric materials. In particular, formaldehyde exceeded the maximum allowable content in different applications (Huang *et al*. 2021).

These plastics used as building materials and their additives have a service life of several decades up to a century, which poses long-term challenges to waste management (Li *et al*. 2016). Paints and coatings often contain polymers with plasticizers such as SCCPs, or formerly PCBs. The removal of PCB-containing coatings and paints from buildings and bridges has contaminated the environment with PCBs, including several hundred kilometers of sediments in the Elbe River in the Czech Republic and Germany, as well as several fjords in Norway (Jartun *et al*. 2009; ELSA 2016; Weber *et al*. 2018b).

2.2.8 Medical devices

In hospitals and health care centers, numerous single-use medical devices made of plastics are widely used, with a wide range of applications from screening and diagnosis to treatment and care. These include, for example, intravenous tubing, bags, catheters, nasogastric tubes, dialysis bags and tubing, blood bags, transfusion tubing and air tubes (Weber *et al*. 2018a).

PVC is estimated to account for approx. 40% of all plastics-based medical devices used (PVC Med Alliance 2019). The most widely used plasticizer in PVC medical devices is DEHP, which has been classified as toxic to reproduction (class 1B) under the EU CLP regulation since it might decrease fertility and damage unborn babies (SCENIHR 2016). The use of DEHP in medical devices has recently been regulated in Europe (European Commission 2017a) and South Korea (South Korea's Ministry of Food and Drug Safety 2014). The highest exposures to DEHP result from medical procedures such as blood transfusions or hemodialysis, during which DEHP leaches from the plastic equipment into biological fluids (US FDA 2001). Exposures of babies to DEHP can be especially high as a result of some medical procedures, such as total parenteral nutrition (e.g., estimated upper bound limit of 2.5 mg/kg/day), and extracorporeal membrane oxygenation (e.g., estimated upper bound limit of 14 mg/kg/day) (US FDA 2001).

Furthermore, PVC can release high amounts of PCDD/PCDFs during the burning of medical waste in incinerators that do not comply with best available technique (BAT) in developing countries (Ikeguchi and Tanaka 2000; Coutinho *et al*. 2006; Zhang *et al*. 2015), leading to contamination of the food chain (Petrlik *et al*. 2022). Also, PFOA and related substances are used in certain medical devices and implants and have received an exemption under Stockholm Convention for these uses.

2.2.9 Personal care and household products

Personal care products (e.g., cosmetics, personal hygiene products) and household products (e.g., household cleaning products, arts and crafts) can expose humans and the environment to chemicals of concern via multiple pathways. The chemicals of concern in these products include plastic-related additives directly used in personal care and household products, such as microbeads made from PE, PP or PS, solid silicone-based resins containing polymerized siloxanes used for instance in lipsticks due to their film-forming properties, or plastic-based propylene or styrene copolymers used to control the viscosity of various cosmetics (Leslie 2014). Depending on the physicochemical properties of such chemical ingredients and the use patterns of the products (e.g., dermal application of hand creams and shampoos, spraying of deodorants, ingestion of toothpastes), humans can be directly exposed to these plastic-associated chemicals at high doses via dermal uptake, inhalation and ingestion. For example, a total of 714 tonnes/year of intentionally added plastic particles with various additives (e.g., phthalates or other plasticizers, stabilizers, colorants) were used in products made by the cosmetics and personal care industry, with associated releases (European

Commission 2017c). These releases can be reduced by bans (e.g., US Congress 2015; European Commission 2022b) and voluntary phase-outs.

The plastic packaging of personal care and household products is another source of chemical exposure, especially in household cleaning agents and liquid soaps, shampoos, or toothpastes. Both chemical additives and NIAS in these plastic packaging materials can migrate into personal care and household products and may lead to high levels of exposure in humans, while the fraction that is washed down the drain may reach and expose ecosystems (Grob *et al*. 2006; Rudel *et al*. 2011; Hutter *et al*. 2016; Biryol *et al*. 2017). In addition to increasing the risk of chemical exposure, personal care and household packaging materials are major contributors to marine plastic litter and marine pollution (UNEP 2016) (see Chapter 3).

Chemicals in personal care and household products are regulated in the EU by different regulations such as the Commission Regulation (EC) No 1223/2009 on cosmetic products (European Commission 2009), the Commission Regulation (EC) No 648/2004 on detergents (European Commission 2004), and the Directive 2001/95/EC on general product safety (European Commission 2001b). In China, the Cosmetic Supervision and Administration Regulation (CSAR) was implemented on 1 January 2021. Brazil, Canada and the US also have some relevant regulatory frameworks (Ferreira *et al*. 2022).

2.2.10 Agriculture, aquaculture and fisheries

Plastics are used widely in agriculture to improve crop yields, reduce food loss and waste and improve food security. In 2018, an estimated 12.5 Mt of plastics were produced for agricultural purposes (FAO 2021). For crop production, plastic applications include polymer coatings for fertilizers, fertilizer sacks, flexible intermediate bulk containers, seedling plug and nursery pot trays, mulching films and greenhouse films (FAO 2021). In livestock production, plastics are used in ear tags for animals, bunker covers, bale nets and twines, silage tubes, etc. (FAO 2021). In forestry, plastic applications include tree guards, tree labels and support ties, and chainsaw fuel containers.

In fisheries and aquaculture, plastics are used in insulating crates, ropes and fishing nets (FAO 2021). Plastics used in fisheries and aquaculture contain additives that can be released to the marine environment, exposing aquatic organisms. For instance, nets used in aquaculture are frequently coated with copper-containing paints and other biocides to prevent being covered with algae, mussels or barnacles. This practice releases hazardous chemicals into the water (Kartal and Sarıısık 2020).

The most common polymers in agricultural plastics include PE, PP, EPS, EVA, PVC and PET. Biodegradable polymers such as polylactic acid (PLA), polyhydroxyalkanoates (PHA), polybutylene succinate (PBS), polybutylene adipate terephthalate (PBAT), polycaprolactone (PCL) and starch blends are also being used, especially as replacements for non-biodegradable polymers in single-use applications (FAO 2021). Their biological and environmental impacts, as well as industry labelling and certification are rapidly expanding areas of research. The results of field studies show that when these plastics are outside industrial or controlled composting conditions, some (e.g., PLA) can persist for many years once they are in marine environments (Greene 2012; UNEP 2021d).

The widespread use of agricultural plastics is a cause for concern because many of these plastics, especially mulching films, are disposed of in the terrestrial environment after use due to difficulties in collection and recycling (Sintim and Flury 2017). This results not only in the contamination of soil with plastics and microplastics (J. Wang *et al*. 2019), but also in the release of chemicals of concern such as phthalates and bisphenols to the environment (FAO 2021; Y. Wang *et al*. 2021). The common practice of openly burning farmlands to clear space can result in the formation of unintentional POPs such as PCDD/PCDFs when plastics, particularly PVC, are present in soil or are intentionally burnt (Ikeguchi and Tanaka 1999).

Biosolids, i.e., the solid by-products of wastewater treatment systems, are regularly used as fertilizers, potentially contaminating agricultural soils with microplastics (Mohajerani and Karabatak 2020; Okoffo *et al*. 2021). Another major concern is the likelihood of empty pesticide containers to contaminate the plastic recycling stream (Eras *et al*. 2017). Additionally, US EPA testing showed contamination with PFOA and other PFASs from fluorinated pesticide containers such as fluorinated high-density polyethylene (HDPE) containers and similar plastics (i.e., fluorinated polyolefins) (US EPA 2021a).

3

ECOSYSTEM AND HUMAN EXPOSURES TO CHEMICALS IN PLASTICS

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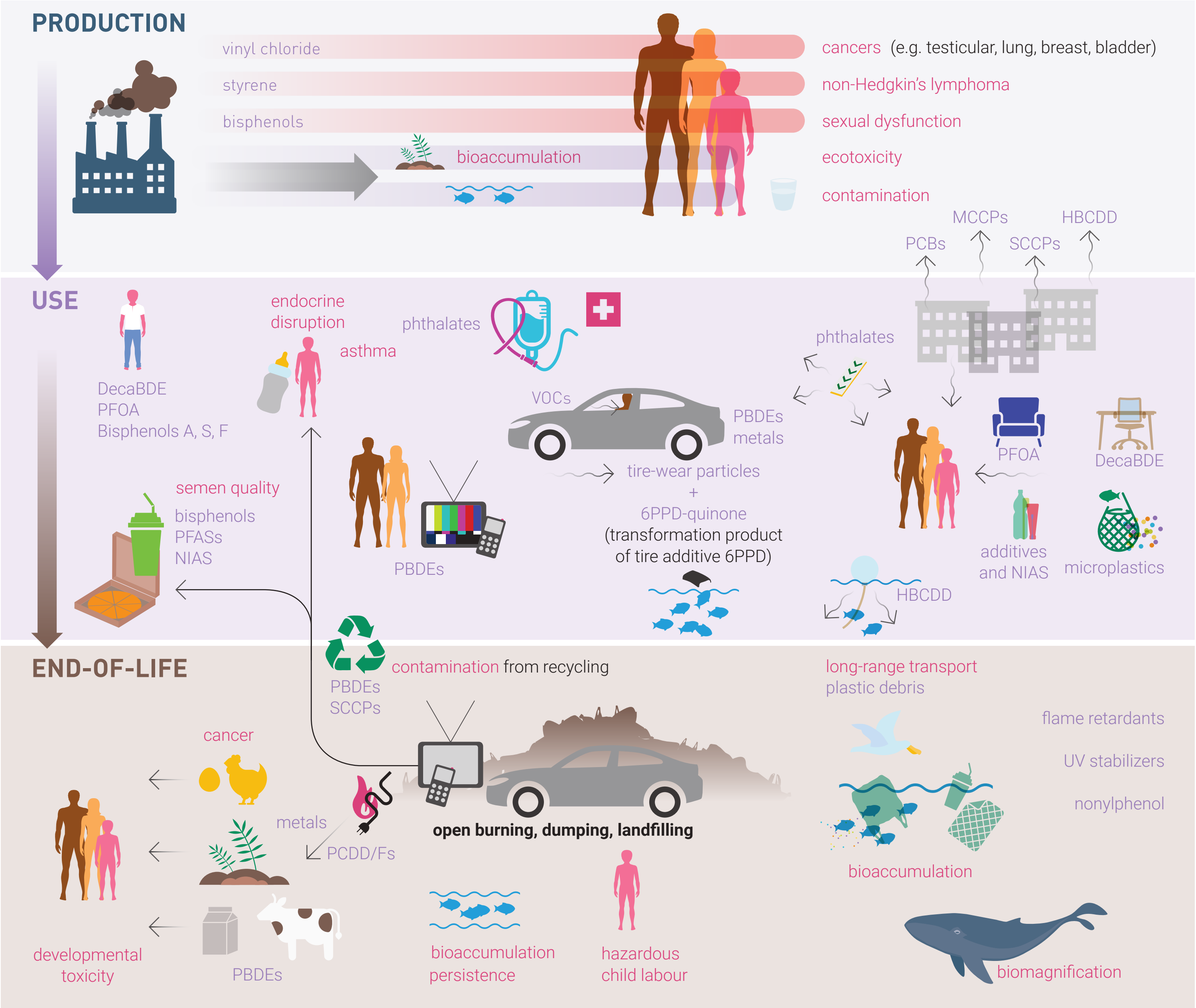
3 / ECOSYSTEM AND HUMAN EXPOSURES TO CHEMICALS IN PLASTICS

Ecosystem and human exposures to plastic-associated chemicals are influenced by a variety of factors including the properties of the plastic and chemicals, whether the plastic is in use or in the environment, the receptor, and frequency and level of exposure (Figure 5). Understanding these factors and the relevant exposure pathways is key to assessing the risks of plastic-associated chemicals.

3.1 Environmental releases and fate of plastics

Globally, approximately 22 million tonnes of plastics, along with the chemicals present in them, are released to the environment every year (OECD 2022a). Sources of release can be both land- and sea-based, and include agriculture, fisheries and aquaculture, personal care, pharmaceutical and healthcare products, building and construction, transportation and shipping, offshore operations, ship-based tourism, plastic production and transport, littering and illegal dumping of waste (GESAMP 2015; GESAMP 2016;

Figure 5. Pathways for ecosystem and human exposures to plastic-associated chemicals along the plastics life cycle. ▼



Science Advice for Policy by European Academies [SAPEA] 2019; UNEP 2021d). Plastic debris can also be transferred across environmental compartments by wind and water currents (e.g., land-sea transfer, surface runoff, atmospheric deposition) (SAPEA 2019; Sugiura *et al*. 2021).

Due to exposure to sunlight, wind and waves, plastics in the environment undergo weathering and break down into smaller fragments to form microplastics and nanoplastics. Plastics may also be decomposed by various other processes including biodegradation, photo-degradation, thermal degradation and hydrolysis (Gregory and Andrady 2004; Andrady *et al*. 2011). Plastics are more likely to degrade on surface soil, surface water or in air, as compared with underground, underwater (e.g., in the deep sea) or in non-surface polar environments where UV radiation and mechanical erosion are minimal (Gregory and Andrady 2004, Barnes *et al*. 2009; Corcoran *et al*. 2009; Jahnke *et al*. 2017; Biber *et al*. 2019). However, as the majority of plastics are made from non-biodegradable polymers and contain additives such as UV stabilizers, antioxidants and heat stabilizers specifically intended to make them durable, plastics are extremely persistent in the environment (Hong *et al*. 2018; ECHA 2020).

Plastic debris, along with the associated chemicals, can also be transported over long distances by river and ocean currents or in the atmosphere, including to remote locations such as the Arctic, ocean gyres and uninhabited islands, where subsequent deposition and uptake by biota can occur (Andrade *et al*. 2021; Yamashita *et al*. 2021). Ocean currents transport an estimated 1,900 to 7,400 tonnes of hazardous additives along with floating plastic debris to the Arctic every year (Andrade *et al*. 2021). Therefore, plastic-associated chemical pollution is a transboundary issue of global concern.

Although the Stockholm Convention regulates several POPs that have uses as plastic additives, many other additives do not necessarily meet *all* the Convention criteria to be listed as POPs. These additives can nonetheless persist for a long time in plastics, travel long distances along with the plastics, and upon ingestion of plastics and microplastics by organisms, result in exposure and potential toxicity, and therefore be of similar concern to POPs (Gallo *et al*. 2018; Andrade *et al*. 2021). Similarly, EDCs in plastics and microplastics may be as harmful as acknowledged POPs due to their potential for long-range environmental transport via plastic debris and due to their activity levels, distribution, toxicity, and bioaccumulation potential comparable to those of listed POPs (UNEP 2017b; Gallo *et al*. 2018).

It should be noted that plastic debris can carry not only the additives intentionally applied to the original plastic product but also hydrophobic environmental pollutants (e.g., POPs and PAHs) that adsorb to plastic debris due to its hydrophobicity (Teuten *et al*. 2009; Gallo *et al*. 2018; Andrade *et al*. 2021). The concentrations of environmental pollutants adsorbed to plastic debris are orders of magnitudes lower than those of the intentional plastic additives (Gallo *et al*. 2018), and recent studies consider the plastic-mediated exposure to such adsorbed pollutants to be of low relevance compared to other sources of exposure for those pollutants (Koelmans *et al*. 2016; WHO 2019). Therefore, the topic of pollutants adsorbed to plastics and related exposure is not discussed in further detail in this report. This comes with an important caveat that for remote ecosystems such as oceans or islands, which receive large amounts of plastic debris, plastic-mediated exposure to adsorbed pollutants may be significant. Sporadically high concentrations of POPs have been observed in plastic debris on remote beaches (Yamashita *et al*. 2018). In future scenarios with higher levels of marine plastic pollution, where relative amounts of plastic debris compared to natural media will be greater, the relative importance of plastic-mediated exposure to adsorbed pollutants will also increase.

3.2 Releases of chemicals from plastic products and plastic debris

Most chemicals used or found in plastics (i.e., additives and NIAS) are typically not covalently bound to the polymer matrix. Therefore, they can migrate or leach out of plastics over time, leading to human and ecosystem exposures to chemicals during the use and end-of-life stages of the plastics' life cycle. Methods to determine the relative potential of additives to leach out of plastic matrices have recently become available (ECHA 2019a).

Some additives are strongly bound to the polymer. For example, TBBPA is typically used as a reactive flame retardant and chemically bound to the polymer (Morris *et al*. 2004) but might still leach out in small amounts (Brandsma 2014). Heavy metal and other metal compounds are normally strongly bound to the plastic polymer chains, and therefore tend to undergo limited migration (Hansen *et al*. 2014).

Nevertheless, high levels of metals can be released from plastics under certain conditions. For instance, antimony (Sb) can leach from PET bottles into water and other beverages under improper storage conditions, particularly at higher temperatures (Xu *et al*. 2021). Migration can be enhanced by acidic beverages, which can actively leach the antimony out of the PET (Hansen *et al*. 2010; Sanchez-Martinez 2013). Mercury also tends to migrate from plastics and evaporates easily, thereby being released to the environment (Hansen *et al*. 2014).

Chemicals leaching out of plastic products can be relevant for direct human exposures. DEHP, for example, has been shown to migrate from medical PVC tubing to patients or other plastic products into food and water (Erythropel *et al*. 2014; Eckert *et al*. 2016). In the US, different types of baby teethers were shown to leach EDCs such as parabens and their transformation products, bisphenols, and benzophenones (Asimakopoulos *et al*. 2016). Leaching of chemicals from plastic toys due to contact with saliva has also been observed (Noguerol-Cal *et al*. 2011). A wide range of studies has demonstrated the transfer of additives and NIAS from plastic-based packaging to food (Bhunia *et al*. 2013; Muncke 2014, Hahladakis *et al*. 2018). Microwaving of different plastics increases the migration of plastic additives and their degradation products into food (Alin and Hakkarainen 2011; Cai *et al*. 2014).

Chemicals leaching out of plastics in the environment can be relevant for environmentally-mediated human and ecosystem exposures. In a study in South Korea, HBCDD was found to be enriched in sediments near aquaculture areas where EPS buoys were used (Al-Odaini *et al*. 2015). Buoys had high amounts of HBCDD, with lower levels in the surface layer compared to the inner layer/bulk, suggesting leaching (Jang *et al*. 2016). A study that compared the levels of additives present in marine plastic debris found at a beach on Geoje Island to those present in newly produced counterparts of the debris (e.g., packaging, fishing nets), found lower concentrations of additives in plastic debris than in new products, also indicating leaching (Rani *et al*. 2017). Nonylphenols found in Tokyo Bay have been suspected to originate from PP pellets (Mato *et al*. 2001).

In the aquatic environment, the leaching potential of chemicals is influenced by several factors, including the chemical's molecular weight, concentration and physicochemical properties, the crystallinity of the plastic, surface-area-to-volume ratio of the plastic particle, duration of exposure to water, the extent of weathering, and environmental conditions such as pH and temperature (Teuten *et al*. 2009; Suhrhoff and Scholz-Böttcher 2016; Hahladakis *et al*. 2018; Luo *et al*. 2019; Xu *et al*. 2020; Andrade *et al*. 2021).

In general, small, low molecular weight chemicals can leach faster than larger, heavier chemicals. Chemicals with a molecular weight higher than 600 g/mol have a low tendency to leach from plastic (Hansen *et al*. 2013). Depending on the chemical properties, the leaching rate is determined by the internal diffusion of chemicals within the plastic matrix or by aqueous boundary layer diffusion, i.e., diffusion from plastic to water (Zweifel and Amos 2001). For hydrophobic chemicals, which have high n-octanol/water (K_{OW}) and plastic/water (K_{PW}) partition coefficients, the leaching process is dominated by aqueous boundary layer diffusion (Kwon *et al*. 2017). The leaching half-life of hydrophobic chemicals can be on the order of years to decades (Endo *et al*. 2013; Andrade *et al*. 2021).

The crystallinity of the polymer is also crucial for determining the dynamics of chemicals dispersed within the plastic structure (Teuten *et al*. 2009). Polymers have sections with closely packed polymer chains (crystalline) and loosely packed chains (amorphous); additives can only occupy the amorphous sections (Andrade *et al*. 2021). Above a polymer's glass transition temperature (Tg), the amorphous regions tend to be in a more rubbery, flexible state (Shrivastava 2018), where diffusion of chemicals is faster (George and Thomas 2001; Sun *et al*. 2019; Andrade *et al*. 2021). Below the Tg, amorphous regions tend to be in a glassy, rigid state. At ambient temperatures, PE and PP are above their Tg, while polymers such as PET, PVC and PS are below their Tg. Therefore, the leaching of chemicals is faster from PE and PP plastics than from PET, PVC and PS (O'Connor *et al*. 2016; Andrade *et al*. 2021).

Leaching may be accelerated at high temperatures and pH, as well as due to large surface-area-to-volume ratios, long duration of exposure to water, and high turbulence in water (Nakashima *et al*. 2016; Sun *et al*. 2019). Leaching of fluorescent additives from PUR microplastics in simulated and natural waters was shown to increase with increasing pH and time (Luo *et al*. 2019). Leaching of PBDEs and BTBPE from microplastics pellets made of ABS increased as particle size decreased; their leaching rate was controlled by their diffusion within the plastic matrix (Sun *et al*. 2019).

Ingestion of plastics by organisms such as seabirds and fish can also enhance leaching (Tanaka *et al*. 2019b; Coffin *et al*. 2019). Hydrophobic fluids like stomach oil and fish oil present in seabirds' stomachs have been shown to substantially accelerate the leaching of hydrophobic additives such as PBDEs, BTBPE, DBDPE and UV stabilizers out of ingested plastics, leading to an accumulation of these additives in the liver, abdominal adipose tissue and preen gland oil of seabirds (Tanaka *et al*. 2015; Guo *et al*. 2019; Kühn *et al*. 2020; Tanaka *et al*. 2020).

3.3 Ecosystem exposures to plastics and their associated chemicals

3.3.1 Animal exposure

Due to the ubiquitous presence of plastics and microplastics of varying sizes in different environmental compartments, organisms across the globe are exposed to plastics and their associated chemicals. Plastic ingestion has been documented in 1,288 marine species and 277 freshwater and terrestrial species across all trophic levels (Gall and Thompson 2015; Santos *et al*. 2021). While the adverse physical impacts of plastic ingestion and entanglement are well-documented, the exposure of organisms to plastic-associated chemicals is an often hidden yet additional cause for concern (UNEP 2021d). Several studies have shown a potential for (eco)toxicity via leachates from plastics (Bejgarn *et al*. 2015; Zimmermann *et al*. 2019; Zimmermann *et al*. 2021) including tire wear particles (Hiki *et al*. 2021; McIntyre *et al*. 2021; Rauert *et al*. 2022).

The relevance of plastic-mediated chemical exposure depends on location, background pollution, individual chemicals and their physicochemical properties, species of biota and their plastic ingestion rates. In the North Pacific Ocean Gyre, a known accumulation zone of marine plastic debris, several PBT substances (PAHs, PCBs, PBDEs and HBCDD) were found in samples of floating plastic debris, 84% of which had at least one chemical at concentrations exceeding its threshold effect level (Chen *et al*. 2018b). In addition, the mass of floating plastic debris was estimated to total 180 times the mass of all marine biota in that area. This indicates a significant risk of chemical exposure for organisms such as sea turtles and seabirds that feed in the region and may ingest large loads of plastic debris instead of their natural prey (Chen *et al*. 2018b).

In a global biomonitoring study of additives in seabirds covering all major ocean basins, BFRs (BDE-209 and DBDPE) and UV stabilizers (UVP, UV-234, UV-326, UV-327, UV-328, UV-329) were detected in seabirds in both populated and remote regions (Yamashita *et al*. 2021). UV stabilizers were detected in almost half of the seabirds sampled in the study, whereas BDE-209 and DBDPE were detected in 5% and 6% of the seabirds, respectively (Yamashita *et al*. 2021). Some of the highest concentrations of UV stabilizers were found in seabirds residing on remote islands located close to oceanic plastic gyres (Yamashita *et al*. 2021). To assess whether the seabirds could have been exposed environmentally or via their natural diet, the concentrations of additives found in those seabirds (UV stabilizers and BFRs) were compared to the concentrations of PCBs in the same seabirds (representative of background pollution from feed) and the concentrations of plastic additives found in their typical prey. As these concentrations did not correlate, and plastics with high concentrations of additives were present in the seabirds' stomachs, plastic-mediated exposure to additives was determined to be likely (Yamashita *et al*. 2021). BP-12, styrene, and UV-328 have also been detected in ingested plastics present in seabird stomachs at concentrations up to 1,700 µg/g, 3,200 µg/g, and 1.4 µg/g, respectively (Tanaka *et al*. 2019a,b).

Several other organisms have been found to accumulate chemicals through plastic-mediated exposure. Mussels inhabiting EPS marine debris containing HBCDD were found to contain both micro-sized EPS particles and high HBCDD concentrations, indicating a transfer of the additive from plastics to the mussels (Jang *et al*. 2016). Fish in the Antarctic Ocean have been found to contain BPA, nonylphenols and PBDEs, with plastics being a probable source (Rochman *et al*. 2014). In a laboratory experiment it was demonstrated that additives in plastics (BDE209, DBDPE UV-234, UV-327 and BP-12) can be transferred through the food chain (from mysids to fish) along with microplastics (Hasegawa *et al*. 2022). The trophic transfer of microplastics had a higher contribution to the accumulation of highly hydrophobic additives (BDE209 and DBDPE) in fish tissue than direct ingestion from the water column (Hasegawa *et al*. 2022).

On land, plastics and microplastics present in soil may be picked up by livestock such as chickens, goats, and sheep, and the associated chemicals can be transferred into meat, milk and eggs. The transfer of

HBCDD to eggs due to chickens picking flame retarded polystyrene has been documented (Hiebl and Vetter 2007). Monitoring data have shown that chicken eggs and milk produced around landfills in Nigeria contain high levels of PBDEs (Oloruntoba *et al*. 2019). House crickets can accumulate PBDEs after feeding on PUF for 28 days (Gaylor *et al*. 2012). In earthworms, PVC microplastics can enhance the uptake and reproductive toxicity of PFOS and PFOA (Sobhani *et al*. 2021). Plastics may also release other hazardous chemicals that inhibit soil-dwelling organisms (Teuten *et al*. 2009).

The relevance of microplastic-mediated exposure to additives is unclear as studies to date have generally focused on pollutants adsorbed to microplastics (Gouin *et al*. 2011; Besseling *et al*. 2013; Herzke *et al*. 2016; Bakir *et al*. 2016; Koelmans *et al*. 2022). A recent study investigated additives in weathered PUR and PVC microplastics and detected several plasticizers (Fauser *et al*. 2022). Microplastic-mediated exposure to these additives was found to pose a low risk to copepods, cod and fulmar (Fauser *et al*. 2022).

3.3.2 Plant exposure

Plants can be exposed to plastics, microplastics and nanoplastics present on or in soil. They are not expected to take up plastics or microplastics due to the high molecular weight and size of the particles (Teuten *et al*. 2009). However, they may take up nanoplastics depending on their size (Zhang *et al*. 2022). For example, 20 and 40-nm plastic particles were found to translocate into plant cells via endocytosis, although 100-nm particles did not (Bandmann *et al*. 2012).

Plastic-associated chemicals present in soil can be taken up by plants, including food and forage crops (Eggen *et al*. 2013; Chae and An 2018). The uptake and translocation of OPFRs in wheat have been observed (Wan *et al*. 2017; Wang *et al*. 2019). OPFRs with relatively higher hydrophobicity were shown to be taken up by the roots, while those with lower hydrophobicity were more likely to be translocated (Wang *et al*. 2019). UV stabilizers and synthetic musks present in sludge-amended soils have also been shown to be taken up and translocated into edible parts of tomatoes (Ramos *et al*. 2021). PFASs can also be readily taken up by plants, especially short-chain PFASs, and evoke metabolic responses and phytotoxic effects at high concentrations (Costello and Lee 2020).

3.4 Human exposure to plastic-associated chemicals

Humans can be exposed to plastic-associated chemicals by (i) direct inhalation; (ii) ingestion of contaminated food, water, and dust; (iii) dermal contact; and (iv) mouthing (most relevant for children). Maternal transfer of chemicals to offspring can also occur via breastmilk or in the placenta (Lite *et al*. 2022). Women of fertile age (Box 1) and children are particularly susceptible to toxicological effects, but men are also at risk, with the latest research documenting substantial detrimental effects on male fertility due to cumulative exposures to hazardous chemicals, many of which are associated with plastics (Kortenkamp *et al*. 2022). The following sections discuss human exposures to plastic-associated chemicals, focusing on occupational, consumer, and environmentally-mediated exposures. Where available, related health and economic impacts are also provided.

3.4.1 Occupational exposure

Occupational exposure to plastic-associated chemicals is particularly relevant during the production and end-of-life stages. Workers can be exposed to chemicals via various pathways, including inhalation, ingestion, and dermal contact (Herber *et al*. 2001). Many chemicals used to make plastics are highly hazardous (Lithner *et al*. 2011; Wiessinger *et al* 2021) and therefore of significant concern for occupational health (Montano 2014; Fucic *et al*. 2018; Box 1).

Several polymers with production volumes above 1 million tonnes per year are synthesized from monomers classified as mutagenic and/or carcinogenic (category 1A or 1B), including the polymer families of polyurethanes, polyacrylonitriles, PVC, epoxy resins, and styrenic copolymers (ABS, SAN, and HIPS). Of the 55 polymers examined by Lithner *et al*. (2011), 31 were found to be made of monomers belonging to the highest two of the ranking model's five hazard levels. Workers are also exposed to additives at levels at which adverse health effects may occur (Väisänen *et al*. 2019; Stefaniak *et al*. 2021).

A review of occupational exposures in Europe listed the plastics, rubber, and textile industries as common industrial sectors associated with increased exposure rates to hazardous chemicals (Montano 2014). For instance, workers from industries that use or manufacture BPA were found to have 70 times

Box 1. Some gender considerations for hazardous chemicals in plastic production, use and disposal

Throughout their lives, women and men are exposed to numerous hazardous chemicals at all stages of the plastic life cycle. Women and men both experience occupational exposure to plastic-associated chemicals and exposure through plastic-based consumer products. Still, these exposures may differ based on occupation, access to information, and type of products. Women workers dominate some professions, including those directly linked to plastic production and use.

While men and women may be exposed to the same toxicants, several factors make women more vulnerable to chemical exposures—including biological factors. Vulnerable windows include puberty (also relevant for males), pregnancy, lactation, and menopause, especially for EDC exposure. During these periods in women's lives, chemical exposures can have especially severe or long-lasting adverse effects and may even impact the next generation. In addition, women have on average more fatty tissue where many toxic chemicals (e.g., POPs) tend to accumulate. Moreover, chemicals in women's bodies can be transferred across the placenta during fetal development and through breast milk to the nursing baby. Exposures during fetal development can cause lifelong diseases and disabilities and increase the risks of irreversible harm. Adverse effects can also be carried across multiple generations.

Consumer products targeting women are an additional significant source of exposure to chemicals. These include cosmetics and feminine hygiene products, which are made of plastic or may contain plastics with hazardous chemicals. Consumers often lack information about what is in these products. Even though they may be present in small amounts, many additives and residues have been linked to health impacts ranging from irritation to long-term health problems.

Thus, women have different susceptibilities and likelihoods of exposure to hazardous chemicals. However, gender differences are not always considered when analyzing biomonitoring data for known plastic chemicals, including phthalates, bisphenols, PBDEs, and PFASs, nor when setting standards for these chemicals. Therefore, women's occupational health and safety risks are often underestimated. That is why gender-disaggregated exposure data are especially needed. There is not enough information about the chemicals women and men are exposed to in the workplace in many sectors, including plastic production and plastic product manufacturing.

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higher urinary BPA levels compared to the general public (Hines *et al*. 2017). An occupational cohort study reported that BPA-exposed workers had a consistently higher risk of male sexual dysfunction than unexposed workers (Li *et al*. 2010). Moreover, in the plastics industry, increased mortality from non-Hodgkin's lymphoma among workers compared to the average population has been associated with exposure to styrene at the mean level found in the air in these occupational settings (Loomis *et al*. 2019). The 15th Report on Carcinogens of the National Toxicology Program found styrene "reasonably anticipated to be a human carcinogen" (United States Department of Health and Human Services 2021), and the World Health Organization's (WHO) International Agency for Research on Cancer (IARC) concluded that both styrene and styrene-6,8-oxide are "probably carcinogenic to humans." Elevated testicular cancer rates have been documented for workers involved in PVC production (Beranger *et al*. 2013). Among employees of the synthetic rubber industry with worktime of a minimum of 10 years, standardized mortality ratios were elevated for leukemia, non-Hodgkin lymphoma, bladder cancer and, for women only, lung cancer (Sathiakumar *et al*. 2019). Women working in industrial sectors with exposure to carcinogens and EDCs, such as the automotive plastics sectors, are also found to have elevated risks of developing breast cancer (Brophy *et al*. 2016).

Occupational exposures at the end-of-life stage (e.g., during recycling), are also of concern, especially for workers in informal sectors, due to a lack of regulatory frames and protection measures. In developing countries, a large proportion of waste management and recycling is often conducted by the informal sector, including child labourers (ILO 2011). Plastic waste is typically collected and separated by the informal sector on and around dump sites or transfer stations, resulting in workers' exposure and environmental contamination (Lundgren 2012; Zheng *et al*. 2017; Yang *et al*. 2018). The recycling process is frequently conducted in a semi-open environment and often the workers do not use protective clothing or equipment while working, not even when handling plastic containing hazardous chemicals such as

PBDEs or other BFRs (Shaw *et al*. 2010; Toxics Link 2011; Haarman 2016). Such activities are considered examples of hazardous child labour (ILO 2011). The uncontrolled end-of-life management of WEEE, including open burning, in developing countries and the resulting contamination from chemicals such as PBDEs and other BFRs, PFASs, and SCCPs further raise serious concerns (Wong *et al*. 2007; Leung *et al*. 2010; Shaw *et al* 2010; Yu *et al*. 2010; Labunska *et al*. 2014; Iqbal *et al*. 2017; Ma *et al*. 2021; Petrlik *et al*. 2021). Informal workers at e-waste recycling sites have been found to experience changes in thyroid function, changes in cellular expression and function, adverse neonatal outcomes, changes in temperament and behaviour, and decreased lung function (Grant *et al*. 2013).

In developed countries with improved sorting, shredding/grinding, and extrusion technologies, occupational exposures to particulates, PAHs, and heavy metals remain in most cases below allowable workplace limits (BAuA 2012). However, other studies show that recycling plastics can result in significant occupational exposure during plastic sorting, shredding/grinding, extrusion, and possibly other processes (Stapleton *et al*. 2008; Hartikainen 2015). A study of e-waste workers at a formal recycling facility in Canada (Nguyen *et al*. 2019) found that the workers' exposure to most flame retardants exceeded the levels reported in other e-waste worker studies from China (Zheng *et al*. 2015) and Thailand (Muenhor *et al*. 2010), possibly due to certain flammability standards in the US that have resulted in higher use of flame retardants in North America (Charbonnet *et al*. 2020). Another study revealed high workplace exposure to PBDEs and other hazardous flame retardants at two facilities recycling PUF in the US (Stapleton *et al*. 2008). The PUF recyclers and carpet installers handling the recycled products had high PBDE levels in their blood (Stapleton *et al*. 2008).

3.4.2 Consumer exposure

Plastics are omnipresent in daily life. This means consumers are continuously exposed to a wide variety of plastic-associated chemicals released from plastic products, including BFRs (Kim *et al*. 2006; Chen *et al*. 2009; Ionas *et al*. 2016; Bohlin-Nizzetto 2022), OPFRs and TBBPA (Bohlin-Nizzetto 2022), SCCPs/MCCPs (Yuan *et al*. 2017; Gallistl *et al*. 2018), phthalates (Rijk and Ehlert 2001), BPA (Brede *et al*. 2003), bisphenol-A dimethacrylate, lead, tin, cadmium, formaldehyde, acetaldehyde, 4-nonylphenol, methyl tert-butyl ether, benzene, and many other VOCs (UNEP 2019i). Such exposures can be of concern, as several studies have shown (US FDA 2001; Brede *et al*. 2003; Bradley and Coulier 2007; Meeker *et al*. 2009; Ionas *et al*. 2016; Koster *et al*. 2016; Geueke 2018).

Direct consumer exposure (i.e., near-field exposure) mainly takes place during the use phase of plastic products (Fantke *et al*. 2016) and can occur via different pathways. Possible exposure pathways include inhalation (e.g., of indoor air and dust contaminated with chemicals from buildings and vehicles), consumption (e.g., of foodstuffs contaminated with chemicals from FCMs), and dermal contact (e.g., with chemicals contained in textiles, furniture, toys, or personal care products). Some consumer subgroups can have unique exposures when using certain products, such as children using plastic toys (Ionas *et al*. 2016) and women using personal care products (Dionisio *et al*. 2015).

More than 1,000 different chemicals have been found to migrate into food or food simulants, leading to direct consumer exposure via consumption of foodstuffs, including beverages. A recently published database compiled all the scientific data on chemicals found to be extractable or to migrate from FCMs into food or food simulants based on scientific studies published as of 2021 (Geueke *et al*. 2022). Frequent consumption of food packaged in plastic or microwaved in plastic packaging has been shown to increase the levels of plastic additives in blood and urine (Rudel *et al*. 2011; Dong *et al*. 2017; Sessa *et al*. 2021).

Plastic building materials are also a source of exposure to many hazardous chemicals. For instance, the presence of PCBs in certain plastic-based construction materials (e.g., sealants and coatings) still leads to significant exposure to PCBs among the inhabitants of impacted buildings (Meyer *et al*. 2013; Lehmann *et al*. 2015). PVC in flooring was associated with increased levels of BBzP in indoor air and children's urine, and higher BBzP metabolite concentrations were associated with an increased incidence of childhood asthma (Whyatt *et al*. 2014). Other hazardous chemicals include 2,4-di-tert-butyl-6-(5-chlorobenzotriazole-2-yl)phenol in silyl-modified polyether sealants; TCPP in spray PUF; phthalates in acrylic latex, silyl-modified polyether, and PUR sealants; and SCCPs in polysulphide, PUR, acrylic, and butyl sealants. SCCPs and other chlorinated paraffins are also used in construction materials such as PVC flooring, PUF, and rubber. Therefore, they may be present in and released from a variety of polymers (Petersen 2012; Brandsma *et al*. 2021).

The concentrations of chemicals released from plastic products can exceed established health risk levels. For example, in 2005 the EFSA Panel set a TDI of 0.05 mg/kg body weight for DEHP (EFSA 2005). However, exposure levels to DEHP in PVC medical devices exceeded this TDI in adults following transfusions of blood components (8-10 mg/kg body weight per day), in trauma patients (2.2 mg/kg body weight per day), and in premature neonates in intensive care units (up to 6 mg/kg body weight per day) (SCENIHR 2016).

Recently, Aurisano *et al*. (2021a) analyzed data on chemical functions and amounts found in plastic toys, quantifying the exposure and related health risks in children using a series of models and a coupled near-field and far-field exposure assessment framework. Chemicals were ranked according to their non-cancer and cancer risks and compared with existing regulatory prioritization lists. Out of 419 chemicals found in different plastic materials and used in children's toys, 126 were identified as of potential concern, including 31 plasticizers, 18 flame retardants, and 8 fragrances. Of concern are also the high concentrations of pigmented cadmium (up to 2% by weight) in older products, particularly in older children's toys that remain in circulation. In some products, cadmium migration levels exceed the Toy Safety Directive limit by an order of magnitude (Turner 2019).

The Danish EPA analyzed the release of four phthalates (DIBP, DBP, BBP, and DEHP) from plastic sandals. Although they estimated no high risk for children from only wearing plastic sandals, the authors emphasized that the results were concerning since phthalate exposure comes from different routes and "the migration of phthalates in plastic sandals presents a significant contribution to the total phthalate exposure for consumers" (Tønning *et al*. 2010). This highlights another important aspect, also pointed out by other authors, which is that the existing guideline values do not consider the toxicity of mixtures, nor the low levels at which EDCs may display effects (Kortenkamp 2008; Vandenberg *et al*. 2012; Kortenkamp *et al*. 2022). For instance, tolerable exposures to mixtures of substances associated with deteriorations of semen quality are exceeded by a large margin, with the highest hazard level from bisphenols and a lower hazard level from phthalates (Kortenkamp *et al*. 2022).

Consumer exposure to chemicals of concern is not only associated with health risks but also with high costs. For instance, the costs of PBDE-related loss of IQ points and intellectual disability were estimated at $266 billion in 2010 for the North American population (Attina *et al*. 2016), and at 9.6 billion Euro per year for the European population (Trasande *et al*. 2016). In the US, the highest PBDE exposures were found to occur indoors (due to consumer goods) and in vehicles (Imm *et al*. 2009). As another example, Legler *et al*. (2015) calculated that prenatal BPA exposure causes 42,400 cases of childhood obesity in the EU every year, with associated annual societal costs of €1.54 billion. The primary source of exposure to BPA for most people is through food and beverages, due to migration from containers (Carlisle *et al*. 2009; Michałowicz 2014).

3.4.3 Environmentally-mediated exposure

As compared to direct exposure to chemicals, such as via dermal contact with a plastic product or consumption of foodstuffs contaminated with chemicals from FCMs, environmentally-mediated exposure or far-field exposure refers to pathways that involve different environmental media or compartments. This includes chemicals emitted to air, water and soil from plastic production sites, from products during use (e.g., agricultural/aquaculture/fisheries plastics) and from waste management sites to the outdoor environment, which can be taken up directly by humans in surrounding communities or by organisms (e.g., fish, crops) that are then consumed by humans.

Exposure to the chemicals used to produce plastics have been found to be harmful, even lethal, for communities living near plastic production sites. For example, in 2020, a polystyrene production plant in Andhra Pradesh, India experienced a leak of ~800 tonnes of toxic styrene gas, which spread to the surrounding community and killed 12 civilians and several animals, hospitalized hundreds of civilians, and contaminated soil, water and food (IPEN 2020). In Louisiana, US, the use of a known carcinogen chloroprene at a neoprene production plant, and of vinyl chloride for PVC production has resulted in high levels of exposure in surrounding communities that may cause adverse health effects, including cancer (US EPA 2010b; CIEL 2021; US EPA 2022a). Formal and informal recycling activities are also important sources of chemical exposure, not only to workers but also to the environment, particularly for plastics from e-waste and end-of-life vehicles. High concentrations of plastic additives including PBDEs, BPA and nonylphenol have been detected in leachate from landfills and dumpsites in developed and developing countries, as reviewed by Kwan and Takada (2016) and Teuten *et al*. (2009). PBDE levels were elevated in the blood of residents living within 10 km of landfill sites in California (Liu *et al*. 2016).

For some persistent chemicals that bioaccumulate up the food chain, such as PBDEs and HBCDD, environmental contamination has been documented as a significant source of dietary exposure (Shaw and Kannan 2009; Schecter *et al*. 2010; EFSA 2011b; EFSA 2012). Indigenous peoples in the Arctic have an especially high exposure through the consumption of their traditional foods (AMAP 2017). Near dumpsites in developing countries, human exposure to PBDEs is known to occur via food, such as eggs and milk (Oloruntoba *et al*. 2019; Petrlik *et al*. 2021). For PAHs, the major source of exposure is through food consumption and the dietary intake of PAHs can be up to 22.5 µg/day (Domingo and Nadal 2015). Especially for high molecular weight PAHs besides ingestion also inhalation represents a major exposure route (Yu *et al*. 2015).

Chlorinated paraffins, used in large quantities as additives in PVC, rubber and polymer coatings/ paints (UNEP 2019e), have contaminated the outdoor environment and food chains (Zhou *et al*. 2018; Krätschmer *et al*. 2019), as well as the indoor environment (Brits *et al*. 2020). However, since chlorinated paraffins are also used in dispersive non-plastic applications such as lubricants and metal working fluids (UNEP 2019e), it is currently difficult to determine the relative contributions of plastic-mediated human and environmental contamination versus other exposure sources.

For the less persistent chemicals, the extent of human exposure from environmental sources is less clear. For example, DEHP is found to bioaccumulate in aquatic organisms, with the highest bioconcentration factor (BCF) values observed for invertebrates, e.g., 2,700 for Gammarus (whereas the BCF for fish is 840) (JRC 2008). This indicates that uptake via the food chain might be an important exposure route (secondary poisoning). However, the BCF and monitoring data for different trophic levels indicate that DEHP does not bio-magnify (JRC 2008). This may in part be due to a more effective metabolism in higher organisms. Due to its high affinity for organic matter, DEHP is expected to show only limited bioaccumulation in plants (JRC 2008). Therefore, environmental exposure via food is limited. A food basket study in the US showed DEHP contamination of food, including meat (Schecter *et al*. 2013). However, due to DEHP's limited bioaccumulation potential, the presence of DEHP in these food items stems from direct contamination of the food (e.g., from the food packaging) rather than from environmental contamination.

There is a lack of data for most other additives. For instance, due to very limited information on the occurrence of and exposure to "emerging" and "novel" BFRs, the EFSA Panel on Contaminants in the Food Chain could not perform a risk characterisation for any of the BFRs considered (EFSA 2012). More studies on other major plastic additives and the relevance of far-field/environment-mediated exposure are needed (Fantke 2016).

4

GAPS AND IMPROVEMENTS IN HAZARD, EXPOSURE, AND RISK ASSESSMENTS

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4 / GAPS AND IMPROVEMENTS IN HAZARD, EXPOSURE, AND RISK ASSESSMENTS

Authorities typically conduct risk assessments, often followed by risk management, to regulate chemicals and reduce their risks to humans and the environment to an acceptable level. Carrying out risk assessments requires two key pieces of information: (i) hazard data for the chemical and (ii) the actual concentrations at which humans and wildlife will be exposed to the chemical. These concentrations depend on factors such as chemical use patterns, amounts used and emissions from products. The actual risk assessment then consists of a comparison between the expected exposure concentrations (derived from exposure assessments) and those considered safe (derived from hazard assessments).

Risk assessments have conventionally been carried out on a substance-by-substance basis (i.e., one chemical at a time). However, in reality, humans and wildlife are cumulatively exposed to mixtures of chemicals over their lifespans. PBT/vPvB properties are also known to increase uncertainties when applying quantitative risk assessment methodologies. Further, for EDCs, the associated health risks are typically underestimated, less well characterized, and not regulated to the same extent as those of some other chemicals of concern, such as carcinogens (Fucic *et al*. 2018). These discrepancies mean that existing risk assessment frameworks are systematically under-protective and therefore need to be improved.

This chapter discusses existing gaps and challenges associated with the hazard, exposure and risk assessments of both individual chemicals and chemical mixtures. It further provides information on the approaches available or under development to improve such assessments.

4.1 Hazard assessments

4.1.1 Gaps in hazard assessments of individual chemicals

The hazards of a chemical (e.g., toxicity, bioaccumulation potential) are related to its inherent properties. These properties are substance-specific and independent of how a chemical is used. Many chemicals in plastics, including additives and NIAS, are poorly characterized and have large data gaps regarding their chemical properties and toxicity (Nerin *et al*. 2013; Groh *et al*. 2019; Zimmermann *et al*. 2019; Kato and Conte-Junior 2021).

These challenges are not unexpected, as testing chemicals for their various physicochemical and (eco) toxicological properties is demanding and requires substantial resources in terms of time, money, personnel and availability of adequate methods (van Leeuwen and Vermeire 2007), as well as suitable test species (Sorgog and Kamo 2019). This problem has often been discussed and was one of the major drivers that led to the EU REACH regulation (EEA 1998; European Commission 2001a). REACH is based on the basic concept of “no data, no market” and has improved the situation to some extent. However, even under REACH the problem of missing or highly uncertain data on the properties of many chemicals persists and has been identified as a major challenge to the successful implementation of the regulation (Stieger *et al*. 2014; Springer *et al*. 2015; BfR 2018; EEB 2019; ECHA 2023c).

Chemicals with high octanol-water partition coefficients (K_{ow}) (i.e., log K_{ow} >5) are particularly difficult to test. Many additives fall into this group (see Table 3). Chemicals with high K_{ow} pose special challenges when conducting relevant ecotoxicological tests for bioconcentration, bioaccumulation and toxicity to aquatic organisms, including OECD test guideline (TG) 305 (for bioconcentration in fish) and 203 (for acute toxicity in fish). They have low water solubilities and are therefore difficult to measure at low concentrations. This leads to studies arbitrarily reporting nominal concentrations such as “100 mg/L” as actual concentrations in the test system cannot be tested. There are also differences in water solubility between the solid substance and sub-cooled liquid (Mayer and Reichenberg 2006).

Overall, the first important consequence of these testing difficulties is a tendency to underestimate the bioconcentration, bioaccumulation and toxicity of high-K_{ow} chemicals (Jonker and van der Heijden 2007). Stieger *et al*. (2014) showed that results from toxicity tests for many BFRs with high K_{ow} were physically and biologically meaningless. A second consequence is that for many high-K_{ow} chemicals no test data are available at all, and this in turn often leads to incorrect conclusions: the absence of

Table 3. Plastic additives with high K_{ow}.

Chemical	Application	CAS No.	logK_{ow}[1]
UV-320	UV stabilizer	3846-71-7	6.27
UV-327	UV stabilizer	3864-99-1	6.91
UV-328	UV stabilizer	25973-55-1	7.25
UV-350	UV stabilizer	36437-37-3	6.31
DBDPE	Flame retardant	84852-53-9	13.6
BTBPE	Flame retardant	37853-59-1	9.15
TDBP-TAZTO	Flame retardant	52434-90-9	7.37
DINCH	Plasticizer	166412-78-8	9.82

[1] estimated with KOWWIN (US EPA 2019).

meaningful toxicity data is taken for evidence of the absence of toxicity. This logical fallacy is common for data-poor chemicals.

4.1.2 Improving hazard assessments of individual chemicals

Several approaches to fill in knowledge gaps for data-poor chemicals have been developed and considered for use within regulatory contexts. Examples include new approach methodologies (ECHA 2016; Gwinn 2020), read-across (ECHA 2017a) and adverse outcome pathways (AOP). AOP is a conceptual framework that organizes existing knowledge, based on mechanistic data, concerning the biologically plausible and empirically supported links between molecular-level disturbances caused through direct interaction with a chemical, and adverse outcomes of regulatory relevance (Villeneuve *et al*. 2014; Wittwehr *et al*. 2017). It was used recently for the identification of molecular targets, health outcomes and some potential AOPs of novel flame retardants for which limited data were available (Bajard *et al*. 2019).

For high-K_{ow} chemicals, new testing methods for bioaccumulation and aquatic toxicity are required. Importantly, the uptake of chemicals sorbed to particles (food) ingested by aquatic organisms should be systematically included in the test methods. The latest revision of OECD TG 305 (III – Dietary Exposure Bioaccumulation Fish Test) addresses this by explicitly mentioning the use of aquatic dietary exposure for very hydrophobic chemicals. However, OECD TG 305-III has yet to be widely applied in regulatory testing. New testing methods for high-K_{ow} chemicals have also been developed (Kwon *et al*. 2016, Stibany *et al*. 2017), but it will take several more years before they can be applied more widely and routinely.

4.1.3 Gaps in hazard assessments of chemical mixtures

Currently, toxicological testing of chemicals in relevant plastic products such as toys and FCMs is focused on individual substances. However, plastic products can contain hundreds of substances that might migrate or leach out of a product. NIAS can represent more than half of the compounds that migrate out of plastics (Grob *et al*. 2006; Geueke 2018). Many of the NIAS are unidentified or even completely unknown (Qian *et al*. 2018). The presence and toxicity of a chemical can only be tested if its chemical identity is known and if it is available as a standard (i.e., pure chemical), which is often not the case (Geueke 2014a; Groh and Muncke 2017; Qian *et al*. 2018; Zimmermann *et al*. 2019). Accordingly, NIAS are rarely analyzed at all and rarely detected. Besides, toxicity data of these substances are often limited or not available, making hazard identification and characterization impossible (Koster *et al*. 2016).

Plastics undergoing recycling also often contain intrinsic contaminants such as legacy additives, dyes and degradation products. The mixtures of chemicals present in the recycling processes can react and form new unwanted compounds that add to the list of NIAS. NIAS in recycled materials are particularly challenging to identify and manage because of the difficulty in tracing their origin (Geueke 2013a; Geueke 2018).

It is increasingly being acknowledged that a traditional approach based on the identification and quantification of all substances in plastic together with their full toxicological characterisation is impractical (highly resource intensive) (Koster *et al*. 2016) and may not address the effects of mixtures. Yet, the toxicological effects of such complex mixtures, including EDCs, need to be considered and assessed (Geueke 2018; Science Council of Japan 2020; Kortenkamp *et al*. 2022) to protect consumers and improve or substitute the products (see Chapter 6).

4.1.4 Improving hazard assessments of chemical mixtures

The assessment of complex chemical mixtures in plastics needs to consider the toxicity of such mixtures. In the last decade, scientific approaches and best practices for the assessment of chemical mixtures have been developed (Fenner *et al*. 2002; Backhaus and Faust 2012, EC Scientific Committees 2012; Altenburger *et al*. 2013; Kortenkamp *et al*. 2022). The EU's Chemicals Strategy for Sustainability is also considering the integration of mixture assessment factors for the safety assessment of chemicals under REACH and has made progress in this regard (European Commission 2020).

One option to assess the overall toxicity of chemicals released from plastic products is to test the overall extract/leachate using *in vitro* bioassays, which are rapid and cost-effective (Groh and Muncke 2017), and use other screening technologies that measure relevant toxicological effects such as cytotoxicity, genotoxicity and endocrine effects (Table 4) (Koster *et al*. 2016). The AOP approach (see section 4.1.2) can also be used to predict the *in vitro* toxicity of chemical mixtures (Lichtenstein *et al*. 2020).

Industry is already conducting assessments of the toxicity of additives or plastics. For instance, the Swiss multinational company Nestlé has developed a recommendation document for their assessments, which describes the research and development process to perform chemical and biological characterization of coated metal packaging materials for safety prioritization purposes (Nestlé Research Center 2018). This document is aligned with the guidance document developed by the International Life Sciences Institute (ILSI) on best practices for the risk assessment of NIAS in Food Contact Materials and Articles (Koster *et al*. 2016), which describes a combination of bioassays, analytical chemistry and risk assessment. However, the recommendations have not yet been endorsed by competent authorities and should therefore be considered a research/management tool for prioritization purposes only, for example in the context of developing new packaging (Nestlé Research Center 2018). Further discussions amongst experts are necessary to identify the best sample preparation steps (Wagner *et al*. 2013) and select the relevant *in vitro* assays (Groh and Muncke 2017). Additionally, the assumption that only molecules smaller than 1,000 Daltons are toxicologically relevant needs to be revisited because there is evidence that larger molecules can be absorbed in the intestinal gut tract, especially if so-called permeation enhancers are present (Groh *et al*. 2017).

While ILSI and Nestlé's approach was mainly developed for FCMs, it may be applied analogously to other plastic materials with significant exposure potential (e.g., toys, kitchen tools, water pipes and tanks). The toxicity screening methodology developed for FCMs could also be used to identify the toxicity of plastics and other polymers at the production stage, including of those plastics that might subsequently be recycled.

Table 4. A selection of *in vitro* bioassays that have been used in NIAS research (Koster *et al*. 2016).

Potential endocrine activity	Cytotoxicity	Genotoxicity/potential carcinogenicity
Oestrogen receptor (ER) redistribution Androgen receptor (AR) redistribution	Cell Organelle Health (COH); end points: DNA content, cytochrome C, mitochondrial membrane potential, RNA synthesis kinetic inhibition.	Indicator assays for genotoxicity (PARP, GADD45,…), Comet-FPG assay
Transcriptional activation assay Oestrogen receptor (ER) (anti) Androgen receptor (AR) Glucocorticoid receptor (GR) Progesterone receptor (PR) Thyroid recepted (TR) Peroxisome Proliferator Activated Receptor (PPARγ)	Cell Proliferation and Cell Death (CPD); end points: apoptosis – caspase3, p53; DNA content, DNA proliferation – BrdU	Mutagenicity test (Ames test, mammalian cell gene mutation tests, micronucleus (MN) test) Potential carcinogenicity Cell Transformation Assay (detection of both geno- and non-genotoxic carcinogens)
H295R Steroidogenesis assay (changes in hormone production)		

Regulatory requirements calling for safety assessments of all substances, including NIAS that may migrate out of plastic products, could also address these challenges and gaps. As a first step, such requirements may focus on products with high exposure risk or exposure to vulnerable populations. For instance, the EU framework regulation on FCMs (EU 2004 Regulation 1935/2004) requires assessing migration as well as genotoxicity and mutagenicity of all starting substances and additives. This approach neglects many substances present in the final article, such as NIAS, as well as other toxicological effects besides mutagenicity and genotoxicity, such as endocrine disrupting effects (Muncke *et al*. 2017). A specific requirement to assess NIAS is included in the FCM plastics regulation (European Commission 2011a, Regulation 10/2011).

Recently, the EU legislation for biocides included a testing requirement for endocrine disrupting activity. To support this requirement, a guidance for the identification of endocrine disruptors in the context of Regulations (EU) No 528/2012 and (EC) No 1107/2009 was developed (ECHA and EFSA 2018). A similar requirement to test for endocrine disrupting properties could be established for chemicals used in products with high exposure potential such as FCMs, kitchen tools, toys, water tanks and pipes, as well as for plastics possibly recycled into such products. The OECD Conceptual Framework for Testing and Assessment of Endocrine Disrupting Chemicals (OECD 2018; e.g., TG 455, TG 456 and TG 458 guidelines) could be considered in the development of a framework for assessing EDCs in plastics and plastic additives.

4.2 Exposure assessments

To reliably estimate the concentration of a chemical in plastics to which humans and the environment may be exposed, the following types of data are needed: (i) the identity of the chemicals used in each plastic on the market; (ii) the amounts of chemicals present in a given plastic material; and (iii) the leaching and evaporation rates of chemicals out of the plastic matrix.

However, data on chemical uses, emissions and resulting concentrations and exposure are scarce. In many cases, these data are not available or highly fragmented. The chemicals used in individual polymers are not normally labelled. Moreover, while the general product categories in which some hazardous additives like POPs have been used are known, the exact products that contain these chemicals are unknown (UNEP 2021b). Therefore, uncertain estimates are the only basis for current exposure assessments.

These challenges are difficult to overcome because many plastics would have to be tested for their composition and leaching behavior, which is hardly feasible. Monitoring of chemicals in products, including restricted chemicals, is expensive and time-consuming and can only be done to some extent. Monitoring capacity is limited or non-existent in developing countries.

Fully understanding the plastic material and product compositions along the entire life cycle is, however, fundamental to successfully assessing, managing and minimizing exposure to chemicals of concern in plastics. This includes knowledge of how plastic-associated chemicals are synthesized, how they are integrated into plastics, and at what levels they end up in plastic products. This further includes and enables an understanding of emissions and exposures along the supply chain and complex consumer exposure to the marketed materials and products, and informs recyclability and recovery potentials at end-of-life. The next step is identifying the populations at risk of exposure (e.g., workers, consumers, vulnerable populations, different environmental compartments) and the exposure pathways.

4.2.1 Gaps in occupational exposure assessments

While a range of studies has been published on the exposure of workers in the production of plastic and related health impacts, there are hardly any data on occupational exposure from plastic production in developing countries and only limited robust studies at other stages of the plastic lifecycle, including recycling (Section 3.4.1).

Overall, there are several gaps in the assessment of occupational exposure. Currently available tools for assessing workers' exposure are not suitable for substances such as EDCs or chemical mixtures. While approaches to assess occupational exposure from hydrocarbon solvent mixtures exist (McKee *et al*. 2017), such assessments and tools have not yet been extended to the production of plastics or the use of plastic resins and additives. Methodologies and tools are hence needed to assess the mixture effects of exposure considering not only CMR effects and the like but also the effects of the wide range of EDCs used in plastic production and recycling. Existing health surveillance of workers in the plastic industry and construction sector gives very limited insight into the health risks associated with exposure to EDCs (Caporossi and Papaleo 2017).

Overall, many thousands of substances are present in occupational settings, but occupational exposure limits (i.e., workplace exposure limits or threshold limit values) to protect workers are available for only approx. 500 of these substances in developed countries (UK Health and Safety Executive 2018). Most of the thousands of additives used in plastics lack occupational exposure limits, including halogenated flame retardants, which are present in plastics and therefore in plastic production and recycling work environments (Nguyen *et al*. 2019), and PBDD/PBDFs generated at plastic production and recycling sites (Ota *et al*. 2009; Sindiku *et al*. 2015b). In developing countries, such occupational exposure limits are either lacking or are rarely implemented due to a lack of enforcement or monitoring capacity.

4.2.2 Improving occupational exposure assessments

The EU REACH Regulation requires developing exposure scenarios, which should describe how a substance can be handled safely during its manufacture and use and should bring this information to the users. Due to the REACH requirements, the plastics industry developed the Plastics Exposure Scenario Tool (PESTOOL), which is a Chemical Safety Assessment specifically developed for the needs of the plastics industry. (Polymer Comply Europe n.d.). Exposure scenarios and material safety data sheets (MSDS) are only transferred among professional users (Stenmarck *et al*. 2017). Most developing countries do not compile and transfer such information (i.e., exposure scenarios and MSDS), and many do not even implement GHS (United Nations Economic Commission for Europe [UNECE] 2019). Therefore, further promoting and implementing GHS with its labelling requirement for hazard communication is an important step towards improving the understanding of worker exposure and reducing harmful exposures.

Specific biomarkers of occupational exposure to EDCs should be introduced in occupational health surveillance, such as the collection of anamnestic data on fertility problems, miscarriages and early menopause (Fucic *et al*. 2018). Such a study was conducted for different kinds of occupational exposures by using effect-based biological methods for male and female hormones (Brouwers *et al*. 2011). In addition to measuring EDCs or their metabolites in biological samples, the levels of hormones such as estradiol and testosterone, which can be measured simultaneously with blood screening, should also be assessed in occupational exposure studies, as they can provide insights into possible hormonal disturbances without a significant increase in biomonitoring costs (Fucic *et al*. 2018).

Consideration should be given to the development of a personal biomonitoring tracking system linked to personal medical records that enables more accurate preventative measures and diagnostics for chemical mixtures (Fucic *et al*. 2018). To achieve this, Fucic *et al*. (2018) suggest:

1. Fostering collaborative studies to collect and share data on biomarkers of occupational exposure to ECDs, initially in the plastics manufacturing industry;
2. Promoting studies to get further insight into oestrogen and testosterone levels and fertility problems in workers occupationally exposed to EDCs;
3. Including gender-specific analyses of occupational health risks (see also Box 1);
4. Educating occupational physicians and hygienists, employers, employees and other key stakeholders about co-exposures to various EDCs.

4.2.3 Gaps in consumer exposure assessments

Consumer exposure to chemicals from plastic products depends on the chemicals present in the plastic, the migration potential of these chemicals out of the plastic, as well as the exposure routes (such as ingestion and skin contact) and pathways (such as mouthing or release into indoor dust that can be inhaled or ingested) (Imm *et al*. 2009; Rudel and Perovich 2009; Abdallah and Harrad 2018). However, fully assessing and quantifying consumer exposure remains one of the most difficult challenges across different science and policy fields (e.g., risk assessment or life cycle impact assessment) for several reasons.

First, obtaining reliable data on plausible ranges of chemical levels in products is a difficult task and thus still a key issue. Manufacturers tend to divulge neither the identity of the chemicals in their products nor their specific concentrations. Consequently, very few publicly available databases can provide this type of information. One example is the Pharos database, which provides hazard, use and exposure information for over 174,700 chemicals across 202 different kinds of building products[2]. Another is the EU **S**ubstances of **C**oncern **i**n articles as such or in complex objects (**P**roducts) (SCIP) database (see section 5.1.7). The few available databases focus on specific sectors (e.g., building products (Healthy Building Network n.d.), cosmetics (European Commission n.d.), food contact materials (Food Packaging Forum (a) n.d.), toys (Interstate Chemicals Clearinghouse n.d.). Finding data for other sectors without available databases (e.g., textiles) is a nearly impossible task.

Second, still missing are reliable product use data (e.g., how and how often consumers use a specific product) and exposure scenarios (e.g., how consumers behave, both indoors and outdoors). These data are of fundamental importance when building models for consumer exposure (Jolliet *et al*. 2015).

Third, there is little to no information on several key exposure pathways (e.g., dust ingestion), which should be coupled with the traditional exposure pathways (food ingestion, inhalation, direct dermal contact) to fully capture the consumer exposure scenario. While there are some available studies on chemical exposure from food packaging (Arvanitoyannis and Kotsanopoulos 2014; Biryol *et al*. 2017), there is still a lack of sufficient data on chemical migration from plastic-based FCMs into food. Furthermore, artificial sweat and saliva simulants are currently not harmonized across different standardization bodies and their respective technical committees, which is important for ensuring the comparability of results regarding chemical migration from FCMs.

Finally, there is a lack of information concerning the general exposure of citizens to chemicals due to gaps in biomonitoring. Such data are critical to assessing the potential health risk of chemicals in plastics.

4.2.4 Improving consumer exposure assessments

Several authors have recommended developing a systematic exposure assessment methodology that considers chemical mixtures (Backhaus and Faust 2012; van Broekhuizen *et al*. 2016; Drakvik *et al*. 2020; Swedish Chemicals Agency 2021) and the complexity of NIAS (Bradley and Coulier 2007; Kato and Conte-Junior 2021; Nerín *et al*. 2022; Box 2), and that includes integrative toxicity assessments such as bioassays (Table 4). This section presents key elements of such a methodology.

Comprehensive analytical screening including non-target analysis is another key tool that could be used to identify the plethora of chemicals present in plastics (Tian *et al*. 2020; Zimmermann *et al*. 2021; Tisler and Christensen 2022). The case study comparing baby bottles made from alternative plastics to those made from PC demonstrates the importance of analytical screening of additives and NIAS in alternatives to restricted polymers. This study also showed that it is possible to produce polymers with low levels of additives and NIAS (Box 3).

For PAHs, the knowledge gap regarding their migration from plastics and rubber and the appropriateness of deriving migration-based limits for their restriction[3] was acknowledged in paragraph 8 of entry 50 of

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[2] https://pharosproject.net

[3] The exposure of the general population to hazardous PAHs has since long been recognised as a matter of concern (ATSDR 1995; WHO 2000).

Box 2. A comprehensive study of NIAS in six different plastics (Bradley and Coulier 2007)

Bradley and Coulier systematically investigated the NIAS present in six different plastics typically used as FCMs (PP, HDPE, PS, PET, PVC and PA) (Bradley and Coulier 2007). Their report gives an overview of the chemical composition of major food contact polymers. The authors combined a thorough literature search with theoretical predictions and detailed analytical studies. They used thermodesorption GC-MS, GC-MS, GCxGC-TOF-MS, LC-MS, LC-FTMS, LC-TOF-MS and NMR to analyze the different materials and showed that GC-MS was able to detect the highest number of compounds. All studied plastics contained unknown chemicals, and the most unknowns were in PVC > PP > HDPE > PS > PET. Although the authors proposed identities for newly detected compounds, many of the peaks were not assigned to any structures (Bradley and Coulier 2007).

Box 3. **A comparison of chemicals released from baby bottles made of alternative plastic versus PC**

The EU Joint Research Center (JRC) laboratory detected a wide range of chemicals released into milk substitutes in an assessment of 449 alternative plastic baby bottles on the European market. 29 benzophenones were detected and quantified in 20 out of the 27 polyamide bottles, with concentrations ranging between 0.5 and 250 μg/kg in 19 of the bottles. One bottle released more than 1,000 μg/kg in the first extraction (Simoneau *et al*. 2012), exceeding the levels extracted from PC bottles (Simoneau *et al*. 2011). None of the 30 polyethersulphone (PES) bottles tested released any detectable amounts of 4,4’-dichlorodiphenyl sulphone (DCPS) or 4,4’-dihydroxydiphenyl sulphone (BPS) and only two bottles released a very low amount of diphenyl sulphone (DPS) (~1 μg/kg compared to a regulatory limit of 3,000 μg/kg). The levels of contaminants found in polypropylene bottles were compliant with the limits set in the EU food contact legislation. No hazardous compounds were detected in bottles made from copolyester (Simoneau *et al*. 2012).

Box 4. **Systematic assessment of the migration of PAHs from rubber and plastic** (JRC 2018)

The main goal of the “STANPAHs project” was to provide support for the implementation and potential amendment of the restriction on PAHs in rubber and plastic, with the specific objectives to (i) gather a better understanding of the migration behaviour of certain PAHs in plastic and rubber components of articles; and (ii) develop a reliable methodology to determine PAH migration from these matrices, under conditions simulating dermal contact (including in the oral cavity).

The methodology needed to be scientific, robust, reproducible and capable of being applied by commercial and service laboratories at an acceptable cost. Furthermore, the methodology would ideally be developed to the point where it could become the basis for a European Standard.

The following activities were undertaken: (i) a scoping study comprising a comprehensive review of the literature and the development of the experimental design concept; (ii) the selection of representative rubber and plastic materials; (iii) the development and optimization of a migration method; (iv) in-house validation of the migration test method; and (v) an initial collaborative trial.

Overall, the study and resulting report make available new data and scientific information on the migration behaviour of certain PAHs from selected plastic and rubber polymeric matrices in support of the European Commission’s legal obligation to review the PAH restriction under REACH. Standard operating procedures for quantifying the content of each of the eight restricted PAHs and their migration into 20% ethanol were developed. Moreover, the information gathered as part of STANPAHs (e.g., the literature search), the manufactured rubber materials still available, and the JRC in-house analysis method for PAH content can help accelerate the standardization of PAH content analysis in consumer products (JRC 2018).

Annex XVII to REACH for articles during normal conditions of handling (ECHA 2013). Guidelines regarding the scope of the restriction were developed at the EU level (ECHA 2018c). To provide support for the implementation of this PAH regulation and identify potentially needed amendments, the EU JRC conducted a state-of-the-art systematic assessment of PAH migration from rubber and plastic (JRC 2018; Box 4). More migration studies for other plastic-associated chemicals of concern or product categories (e.g., FCMs) along with standardized procedures for migration tests are needed for effective exposure assessment.

4.2.5 Gaps in environmentally-mediated exposure assessments

Several environmental exposure assessments of plastics are available. Nevertheless, reliable models for estimating not only the amounts of plastics entering the environment but also their fate are still missing. There is a lack of information and methods regarding:

- the environmental distribution profiles of various types of plastics (where the plastic goes in the environment and from where);

- the leaching of plastic additives in different environments (what proportion of the additives present in the plastic migrates to the environment);
- the ecotoxicity effects of plastics and related chemicals across different ecosystems (environmental impacts resulting from plastic pollution); and
- how the environmental impacts of plastics compare to other environmental issues and forms of pollution.

A broad overview of biomonitoring data for plastics and their associated chemicals for most species is missing (e.g., for seabirds, see Yamashita *et al*. 2021). Furthermore, there is little to no ecotoxicity information for plastic additives and most species across various ecosystems (e.g., terrestrial soil and groundwater).

4.2.6 Improving environmentally-mediated exposure assessments

Consistently evaluating exposure pathways along product life cycles requires a flexible mass balance-based assessment framework that structures chemical transfers across indoor and outdoor environmental compartments in a consistent set of chemical mass transfer fractions (Fantke *et al*. 2016). For example, PCB releases from polymers such as sealants and coating into the environment and the related accumulation in animals, including human food, and human exposure have been documented along the product life cycle (Weber *et al*. 2018b). A better understanding of the environmental releases and fate is needed for other major plastic additives, with greater priority for POPs and other PBTs.

Gallo *et al*. (2018) reviewed the relevant existing knowledge about the fate of chemicals in marine plastic litter. More recently, Andrade *et al*. (2021) published an overview of oceanic long-range transport of organic additives present in plastic products. Furthermore, guidance on long-range environmental transport is currently being developed by a POPRC working group, which will also address the topic of persistent chemicals in plastics as raised during the POP nomination process of UV-328 (UNEP 2021e). The many existing data gaps regarding the concentrations of additives present in the different polymers hampered a more accurate estimation of the mass flows of hazardous additives in mismanaged plastics (Andrade *et al*. 2021). Therefore, more transparent data on additives and other chemicals in plastics are needed to better assess and model their environmental fate.

Also relevant to the understanding of the environmental fate of plastics and their associated chemicals are the environmental degradation rates of plastics and the chemicals released from them into the environment. Data on the environmental degradation and pathways for major types of thermoplastic polymers have recently been compiled (Chamas *et al*. 2020). The authors used the specific surface degradation rate as a metric to harmonize disparate types of measurements to extrapolate half-lives (Chamas *et al*. 2020). They highlighted the need for better experimental studies under well-defined reaction conditions, standardized reporting of rates and methods to simulate polymer degradation (Chamas *et al*. 2020). Such studies should also assess the release of additives and degradation products such as nanoparticles, monomers, oligomers and other degradation products of the polymer chain.

Another area in need of research is the release of chemicals from plastic in landfills, which contain the largest stock of plastic slowly degrading and releasing additives, monomers such as BPA (Gomi *et al*. 2022) and likely other degradation products. Some information on the release of additives from plastic in landfills is compiled in Section 7.2.6. More research is needed, including on the long-term degradation of individual polymers and related releases.

Pollutant release and transfer registers (PRTRs) may provide some valuable information about the management, release, and transfer of chemicals, including multimedia release information from manufacturing facilities (OECD 2022b). While not specific to plastic monomers and additives, some of these chemicals are included in the chemical inventories of PRTRs. Additionally, sectors that participate in the plastics value chain are often required to report to PRTRs.

The Risk-Screening Environmental Indicators (RSEI) model incorporates release data from the US PRTR (the Toxics Release Inventory) along with the chemical's fate and transport, each chemical's relative toxicity, and potential human exposure (US EPA 2022b). RSEI is a screening-level tool designed to help identify geographic areas, industry sectors and chemical releases that may be associated with significant potential human health risks.

4.2.7 Spatiotemporal considerations in exposure assessments

In addition to human variability in exposure due to differences in product use patterns, age, sex, cultural background, socioeconomic status, etc., spatiotemporal aspects also need to be considered in exposure estimates, as they play a role in the emissions of chemicals from plastics into the environment.

Spatial aspects mainly relate to differences in the availability of certain plastic products and additives on the market across regions and countries, but also regionally varying consumer preferences (relevant for consumer exposure), safety standards at work places (relevant for occupational exposure), fire safety standards (relevant for exposure to flame retardants), environmental standards (relevant for far-field/environment-mediated exposure), consumer safety standards (relevant for consumer exposure) and climatic conditions influencing the environmental distribution of chemicals in the environment (relevant for population exposure).

Spatial differences in emissions of chemicals are mainly influenced by differences in climatic conditions such as wind speed and direction, temperature and humidity. Several available models deal with the spatial aspects of emission patterns (e.g., Hollander *et al*. 2012) and chemical distribution, including for plastic additives (e.g., Wegener Sleeswijk 2006; Wannaz *et al*. 2018). While several available modelling approaches consider spatial differences in environmentally-mediated exposures (i.e., emission-based), there are hardly any approaches available for considering spatial differences in exposures associated with chemicals in plastics.

4.2.8 Aggregate and cumulative exposure assessments

Humans and wildlife are exposed to multiple stressors via multiple pathways. Considerable attention has been dedicated to assessing how different stressors interact and how overall exposure is influenced by uptake via different routes and pathways. Nevertheless, quantifying and assessing exposure to plastics and associated chemicals is an emerging field of exposure assessment, where both aggregate and cumulative exposure must be considered. Aggregate and cumulative assessments represent the two major categories of combined exposure assessments currently in use.

Aggregate exposure assessment is the analysis of exposure to a single chemical (stressor) via multiple pathways and routes of exposure affecting a single biological target (US EPA 2003). Some additives are unique to certain plastic product categories and are unlikely to be used in multiple products. On the other hand, some additives are present in multiple consumer products (e.g., widely applied plasticizers, flame retardants and fragrances), and failing to consider consumers' aggregate exposure to these chemicals from all product types would lead to less reliable results (Safford *et al*. 2015). Therefore, it is important to assess exposure to the same plastic additive from different indoor and outdoor sources such as building materials, toys and packaging. However, data on additives application areas (i.e., sources) and quantities across different products and sectors, which are necessary for estimating aggregate exposure, are often lacking.

Cumulative exposure is the analysis of exposure to multiple chemicals (stressors) via multiple pathways and routes of exposure affecting a single biological target (US EPA 2003). Cumulative assessments more realistically depict real-world exposure, but also introduce a layer of complexity not found in traditional exposure assessments, which evaluate stressors individually (US EPA 1992). In addition to the lack of data on additives applications and quantities, information about how the biological target (e.g., humans) might react to exposure to multiple additives and other chemicals is also missing. More precisely, data on the interactions of various chemicals are missing (e.g., synergies or antagonistic effects). Thus, there is a lack of data on the different sources and quantities of specific substances and on the potential interactions between different substances, all of which are necessary for estimating cumulative exposures.

Under the ExpoCast program, US EPA researchers have developed a high-throughput (HT) framework for estimating aggregate exposures to chemicals from multiple pathways to support rapid prioritization of chemicals, including a definition of chemical-product categories (Dionisio *et al*. 2015) and consumer exposure predictions, such as for chemicals in FCMs (Biryol *et al*. 2017). While these HT methods have large uncertainties (and thus may not be appropriate for assessments of single chemicals), they can provide a critical refinement to aggregate exposure predictions used in risk-based chemical priority-setting (Biryol *et al*. 2017; Ring *et al*. 2019). While the HT approach has been developed initially for plastic-based and other types of FCMs, it could also be applied to leachates from other plastic products with high exposure potentials, such as toys and medical devices.

Another emerging approach to address screening-level exposure to chemicals in various plastics and other applications is the product intake fraction (PiF) framework, which integrates consumer near-field exposure with environmental emission-based far-field exposure into a consistent, mass balance-based framework. The product intake fraction represents the cumulative intake of chemicals by consumers (e.g., children for plastic toys) from a given plastic or other application via all relevant exposure pathways. This metric is built on a consistent mass balance (Jolliet *et al*. 2015; Fantke *et al*. 2016) and is fully compatible with existing population exposure frameworks, such as the UNEP/SETAC scientific consensus model USEtox (http://usetox.org). This allows application not only for chemical prioritization, but also for evaluating alternatives to harmful chemicals and for integrating comparative risk and life cycle impact assessment methods, as already shown in examples for personal care products, cosmetics, FCMs and building materials (Csiszar *et al*. 2016; Ernstoff *et al*. 2016; Fantke *et al*. 2016; Csiszar *et al*. 2017; Ernstoff *et al*. 2017; Huang *et al*. 2019). Being based entirely on mass balance, the PiF framework can be extended to all relevant applications of plastics and applied to estimate occupational exposure to chemicals, such as during plastic manufacturing processes (Fantke *et al*. 2018).

The risk assessment community in the US has also been developing methodologies for aggregate risk and exposure assessment, such as US EPA's cumulative framework of risk (developed for pesticides) (US EPA 2002) and the cumulative impact assessment, which seeks to address both chemical and non-chemical impacts (US EPA 2022c).

4.2.9 Human biomonitoring

Human biomonitoring data are of fundamental importance to understanding exposure and assessing related risks. The Human Biomonitoring for Europe (HBM4EU) project has recently gathered data on human exposure to chemicals in Europe, including several plastic additives, to better understand the associated health impacts and to improve chemical risk assessment (HBM4EU n.d.). In 2022, the EU launched the European Partnership for the Assessment of Risks from Chemicals (PARC), an initiative aimed at improving knowledge about chemical substances (German Federal Institute for Risk Assessment and German Environment Agency 2022). Similarly, the US National Health and Nutrition Examination Survey (NHANES) provides nationally representative biomonitoring data for chemicals such as phthalates, BPA and nonylphenol (Calafat *et al*. 2005; US EPA 2017). Such initiatives are urgently needed in developing countries, especially for vulnerable groups.

4.3 Risk assessments of chemical mixtures

As humans and wildlife are always exposed to multiple chemicals in combination, risk assessment schemes that aim to be realistic and protective need to consider the exposure to and toxicity of complex chemical mixtures rather than those of individual chemicals alone. This is highlighted by evidence that even mixtures in which individual chemicals are present at low concentrations, i.e., below the chemicals' respective no-observed-effect levels, can cause significant toxic effects, also in humans (Silva *et al*. 2002; Altenburger *et al*. 2013; Kortenkamp and Koch 2020; Tanner *et al*. 2020; Caporale *et al*. 2022; Kortenkamp *et al*. 2022). The risk assessment of chemical mixtures has therefore been an important topic for method development among scientists and regulatory authorities (Kienzler *et al*. 2017).

Several publications provide an overview of the regulatory context and methods available for the assessment of chemical mixtures, as well as experimental data that support these methods (see e.g., Backhaus and Faust 2012; van Broekhuizen *et al*. 2016; Sweden, Ministry of the Environment 2019; Kortenkamp and Koch 2020; European Commission 2020; Science Council of Japan 2020; Swedish Chemicals Agency 2021). The principles according to which the toxic effects of a mixture are derived from the potencies of the individual components of the mixture are well understood and have been known since the 1930s (Backhaus and Faust 2012). The two main concepts for describing how chemicals cause mixture effects are concentration addition and independent action.

Concentration addition applies to chemicals with the same mode of toxic action; all components of a mixture of chemicals with the same mode of action contribute to the same effect according to their relative potency. There is an important practical implication of the concentration addition concept:

the risk quotient of the mixture is the sum of the risk quotients of the individual mixture components.

Independent action applies to chemicals with different modes of toxic action; the basic idea is that these chemicals exert their different toxic effects independently of each other. Calculating the joint effect of the components of a mixture of chemicals with independent action requires information about the effect caused by each component of the mixture.

Both concepts, concentration addition and independent action, have been confirmed experimentally in setups where the individual potencies or, if needed, the complete dose-response curves of all mixture components were known. The mixture effects measured experimentally were then compared with the effects calculated according to concentration addition and independent action (see for example Silva *et al*. 2002). These experiments have shown that concentration addition often provides a good or even very good estimate of the effect of the mixture (Backhaus and Faust 2012; Altenburger *et al*. 2013; Kortenkamp *et al*. 2022).

For regulatory risk assessments, this has the important implication that the risk quotients of the individual mixture components can be used to estimate the risk quotient of the mixture (see for example Fenner *et al*. (2002)). Martin *et al*. (2021) performed a systematic review of 10 years' worth of experimental mixture studies, which confirmed that the concentration addition concept is a suitable default concept for anticipating the combined effects of chemicals.

The concepts of concentration addition and independent action are both based on the assumption that the chemicals in the mixture do not interact in their toxicokinetics, i.e., during uptake by, distribution in, and elimination from the organism, nor in their toxicodynamics, i.e., in their interactions with the receptor where the toxic effect is caused. In the vast majority of cases, this assumption is justified (European Commission 2012a). However, interactions between the components of a mixture, if present, can lead to mixture effects that are stronger than expected (synergism) or weaker than expected (antagonism). For specific classes of chemicals, such as EDCs and metals, new evidence of synergism makes it necessary to assess their mixture effects in more detail.

In addition to considering the toxicity of chemical mixtures, an adequate risk assessment requires a detailed understanding of all possible exposures, including those of the workers involved in the extraction of the fossil fuels, in the production of the resins, additives, and plastic products, as well as in the waste management stages, including recycling; of consumers exposed through the various plastic product applications; and of vulnerable populations and the environment exposed via emissions and releases along the entire life cycle of plastic products.

Acquiring such life cycle-based knowledge requires exploring several enabling factors. One such factor is an increased uptake of tools for digitalization and management of big data (e.g., data related to complex supply chain information on chemicals in specific plastic materials) that can help generate relevant knowledge and track chemicals along life cycles, including reuse and recycling (Fantke *et al*. 2021). Another factor is the development and broad implementation of a fully transparent data flow management system across chemical, material, and product supply chains that also allows for compliance with data and ownership rights protection. Most important, however, is the development and application of standards for data and information sharing to allow for consistent tracking of information on chemicals and material composition along life cycles and recycling cycles, and to move away from "no data, no harm" to "no data, no market" management. The combination of these advancements will finally enable the transition from a market driven by the widespread use of numerous chemicals across plastic applications towards a market in which chemical complexity is reduced (allowing for better separation for recycling) and exposure becomes manageable along a complex global supply chain network of chemicals and plastic products. The better reuse and recycling of products will also help minimize the use of new resources (i.e., reduce virgin materials manufacturing).

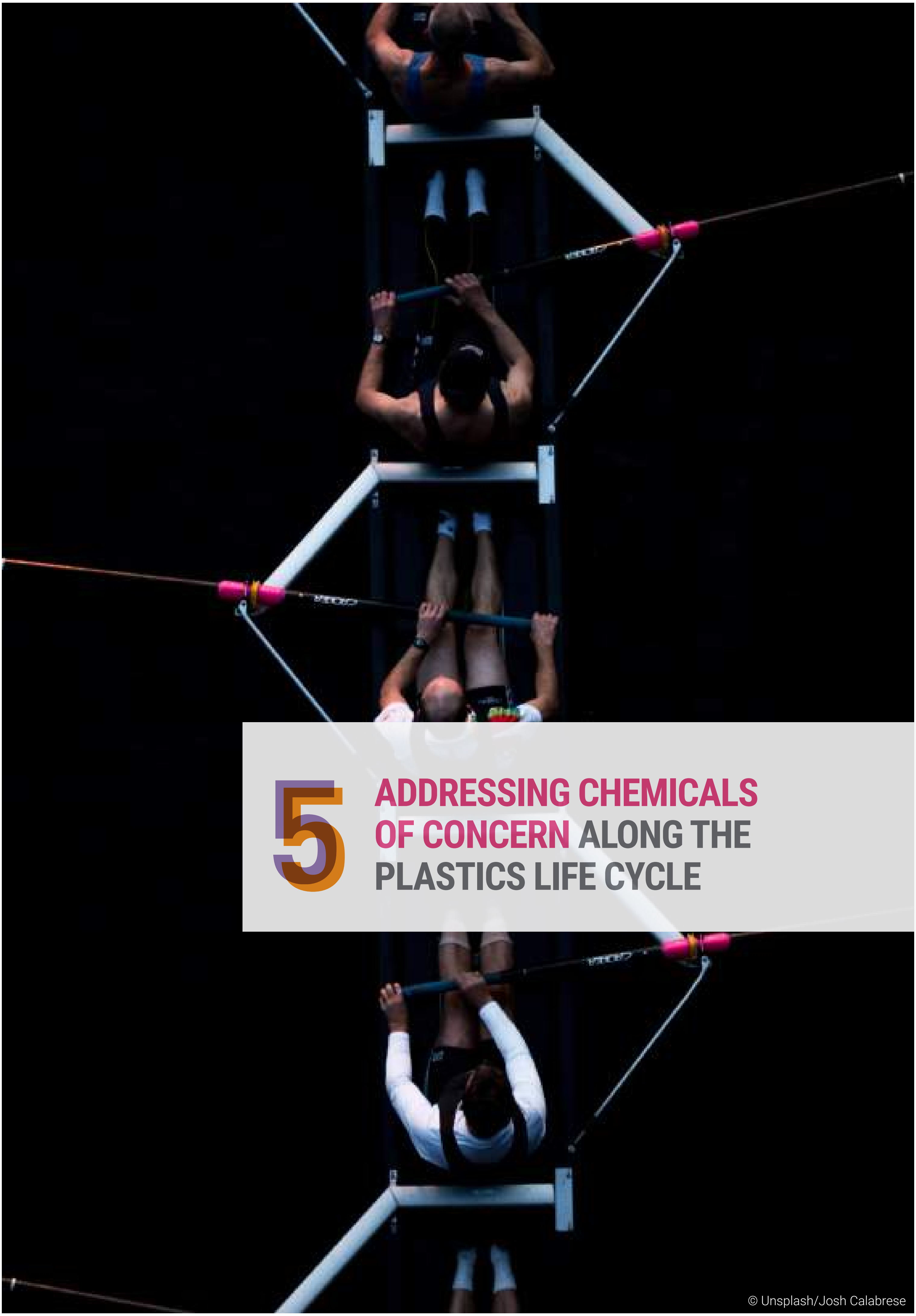
One approach for the prioritization of safety assessment for chemicals lacking toxicity data is the threshold of toxicological concern (TTC) approach. According to the TTC, a chemical's toxicity is estimated based on toxicity data from structurally-related chemicals. This means that by knowing the structure and exposure potential of a chemical, its risks can be estimated in the absence of appropriate toxicological data. This enables prioritizing for testing those chemicals estimated to have the highest risk. The TTC is based on the assumption that it is possible to derive exposure threshold values below which there is a very low probability of adverse health effects.

The TTC approach was originally applied to substances known to be found in food (Kroes *et al*. 2004). Since then, Koster *et al*. (2011) have also proposed its application to unknown substances found in food. The following are the most important aspects to be considered when using this approach (Koster *et al*. 2011; Geueke 2013a):

- As many as possible of the substances present in the sample must be detected by using several analytical techniques.
- The chemicals identified in the sample are classified according to the Cramer rules. They are sorted into three classes, with a maximum intake level applied for each class (class I – 1.8 mg/person/day; class II – 0.54 mg/person/day; class III – 0.09 mg/person/day).
- Substances that lead to exposure of less than 0.09 mg/person/day and belong to Cramer classes I-III are regarded as safe. In this step, the exposure estimate determines whether the TTC is exceeded or not.
- CMRs, suspected CMRs and those that accumulate (log Kow >3, EFSA 2008) are excluded from this classification and considered high-risk compounds irrespective of their concentrations. Koster *et al*. (2011) published a detailed protocol that helps detect these chemicals and includes analytical methods for structural alerts and bioassays. Choosing suitable chromatographic techniques and sample preparation and detection methods are considered especially important for (partially) identifying unknown compounds. When these approaches do not rule out the presence of toxic compounds, specific screening methods and targeted analyses have to be performed.

Although analytical techniques are steadily improving, certain compounds might not be assigned to the proper risk class, or might not even be detected at all. Furthermore, the TTC approach does not cover the cumulative effects of mixtures nor non-monotonic dose-responses, which often exist for EDCs. In addition, the TTC only allows for evaluating substances of known chemical structure and with known exposure data. However, for many NIAS, the structure and exposure data are unknown. A solution could be the combination of the TTC approach with bioassays. While the TTC approach has been developed for food and plastic-based FCMs, it could also be applied to leachates from other plastic products with high exposure potentials, such as toys and medical devices. Importantly, TTC is a proxy tool for estimating risk in the absence of empirical hazard data, and it is associated with large scientific uncertainty (Bschir 2017).

In addition to assessing the risks of chemical mixtures, risk assessments of polymers are also needed. While the registration of polymers is still exempt from the EU REACH regulation, the European Commission is currently developing a proposal on how and which polymers to register under REACH. The main challenges for polymer registration under REACH are the very high number of polymers on market (estimated at 200,000 in the EU alone) and the lack of hazard data for most polymers (IPCP 2021). The plastic industry has developed a Conceptual Framework for Polymer Risk Assessment (CF4Polymers), which is presented in a technical report by the European Centre for Ecotoxicology and Toxicology of Chemicals (ECETOC) Polymers Task Force. The work of the Polymers Task Force was motivated by the versatility and complexity of polymers, which are generally not present as mono-constituent substances, but as complex products consisting of the polymeric substance (polymeric macromolecules), IAS (e.g., stabilizers), and NIAS (e.g., impurities). Furthermore, polymer products can change their form during different life cycle stages. For these reasons, conventional risk assessment approaches used for chemicals may need to be modified to match the specificities of polymers (ECETOC 2019). The framework (Version 1; ECETOC 2019) does not however specifically address the issue of plastics recycling, which is a major challenge for additives and degradation products (Geueke 2013a; Geueke 2018). The report also commented that there is no ‘one size fits all’ polymer hazard and risk assessment process. In the same way, there is no ‘one-size-fits-all’ approach to determine if any given tool, test method or model is applicable to the assessment of all polymers.



5

ADDRESSING CHEMICALS OF CONCERN ALONG THE PLASTICS LIFE CYCLE

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5 / ADDRESSING CHEMICALS OF CONCERN ALONG THE PLASTICS LIFE CYCLE

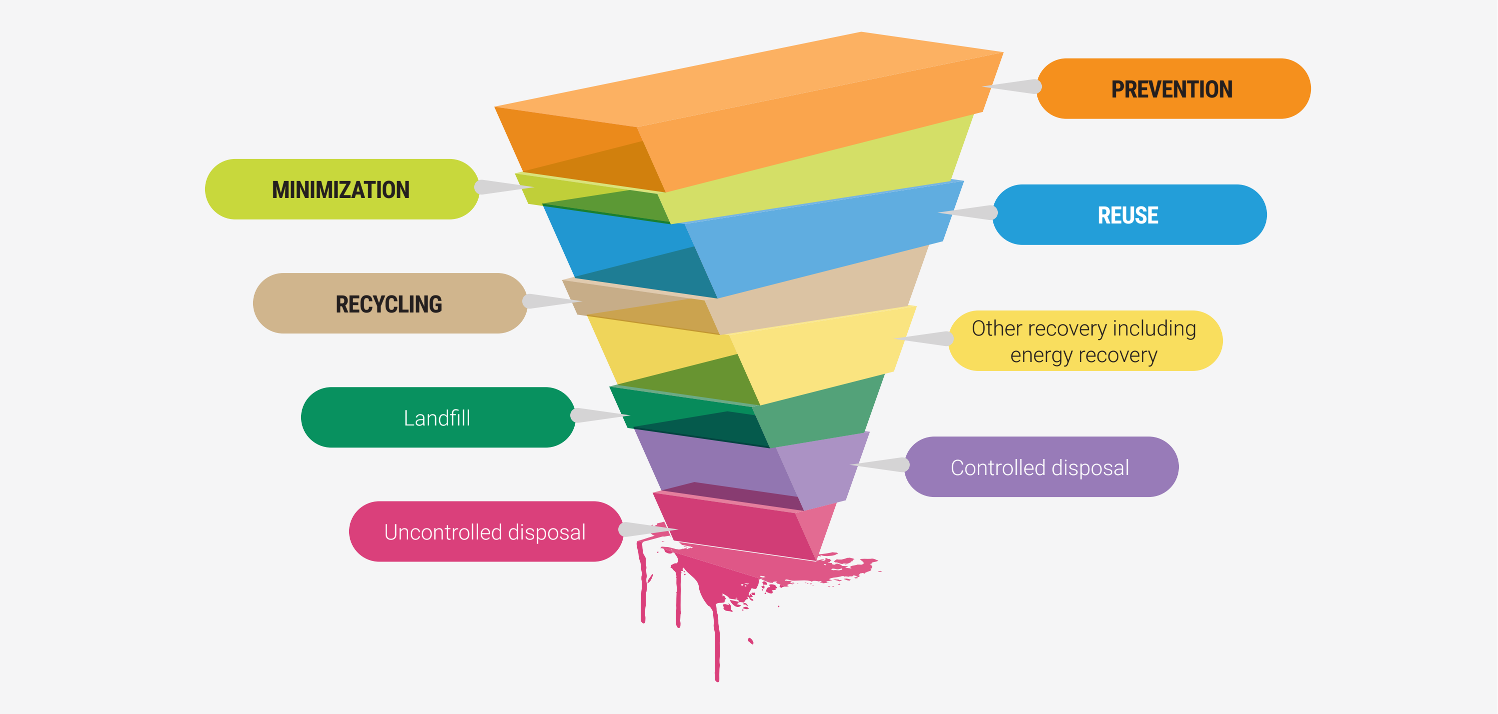
Given the complexities surrounding plastic pollution and the interlinkages with planetary crises such as climate change and biodiversity loss, solutions for addressing chemicals of concern in plastics should be developed based on systems thinking principles (UNEP 2021d). System thinking is a powerful approach that supports decision-making and effective actions for complex and interlinked problems by looking at a system as a whole rather than at its individual parts (Meadows 2009). This entails moving away from looking only at pieces of data or single events, to identify trends and patterns of behaviour over time and understand the systemic structures and mental models that drive such events and patterns.

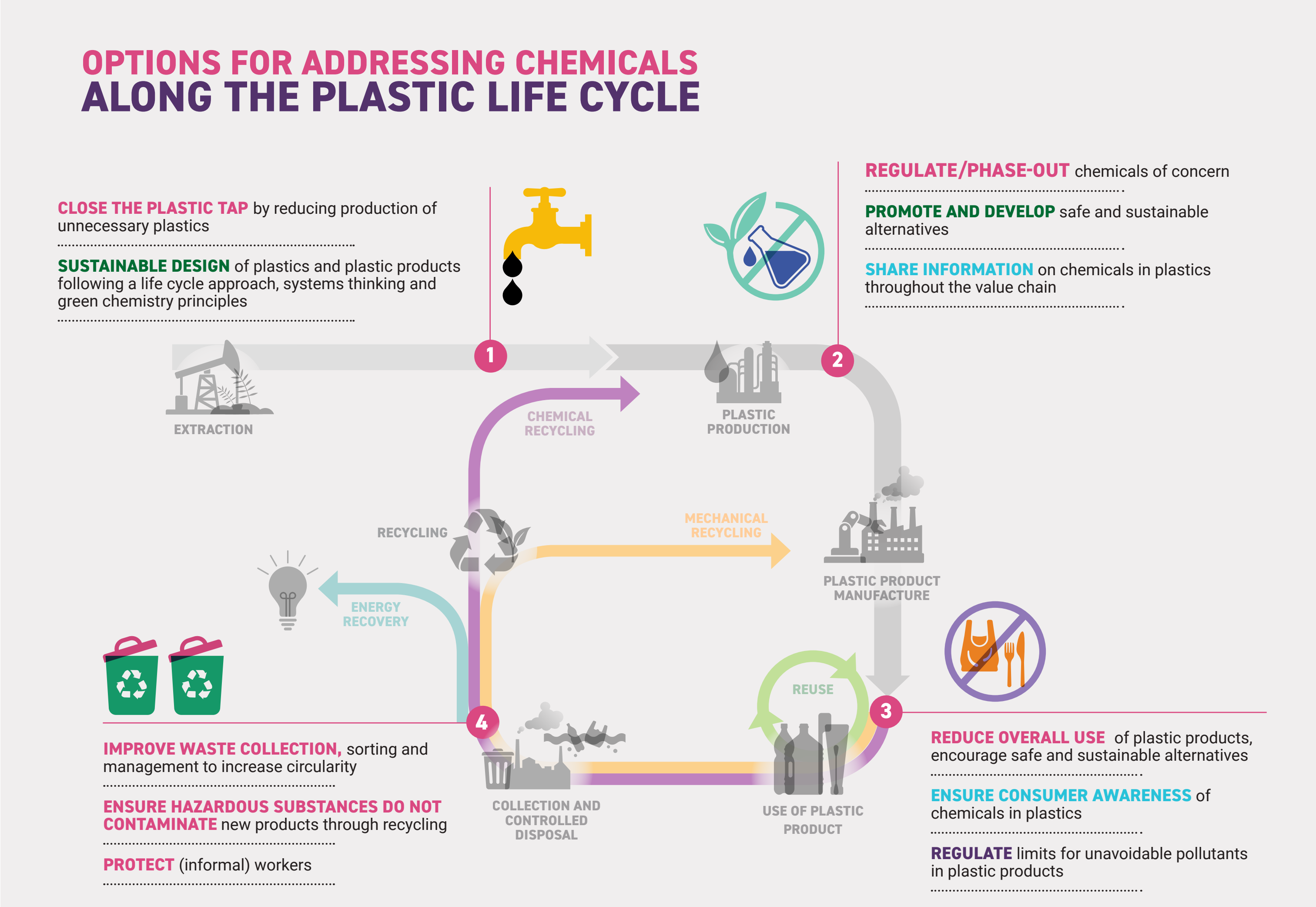
For example, as the public debate has focused largely on plastic litter, much attention has been placed on removing plastic debris from the environment. While this certainly has benefits (e.g., reducing environmentally-mediated exposures to plastic-associated chemicals and creating public awareness), clean-up measures may not be as effective in the long term as “closing the plastic tap” in the first place. Regional and global organisations have therefore expressed an urgent need to reduce the production and consumption of plastics and associated chemicals (OECD 2012a; OECD 2012a; UNEP 2013a; European Commission 2017b; National Geographic 2018; UNEP 2018a; UNEP 2018b; UNEP 2019a; Science Council of Japan 2020; Australian Government 2021).

A first step is the phase-out of plastic uses that are not essential for human health, safety, or the functioning of society (e.g., as applied under the Montreal Protocol; see section 5.1.3 for further discussion). Reducing the overall production and use of plastics would cut down resource use, chemical use, and plastic waste generation. It would reduce the pressure on overstrained waste management systems and enable better end-of-life management, thereby further reducing the environmental releases of plastics and associated chemicals. In plastics deemed to have essential uses, hazardous chemicals need to be replaced with safer and more sustainable alternatives.

The essential-use approach is in alignment with the waste management hierarchy (Figure 6), which emphasizes prevention and minimization as the most resource-efficient and sustainable solutions that protect human health and the environment. The next best solution is product reuse, followed by physical/

Figure 6. **The waste management hierarchy** (UNEP and ISWA 2015). ▼





OPTIONS FOR ADDRESSING CHEMICALS ALONG THE PLASTIC LIFE CYCLE

CLOSE THE PLASTIC TAP by reducing production of unnecessary plastics

SUSTAINABLE DESIGN of plastics and plastic products following a life cycle approach, systems thinking and green chemistry principles

REGULATE/PHASE-OUT chemicals of concern

PROMOTE AND DEVELOP safe and sustainable alternatives

SHARE INFORMATION on chemicals in plastics throughout the value chain

REDUCE OVERALL USE of plastic products, encourage safe and sustainable alternatives

ENSURE CONSUMER AWARENESS of chemicals in plastics

REGULATE limits for unavoidable pollutants in plastic products

IMPROVE WASTE COLLECTION, sorting and management to increase circularity

ENSURE HAZARDOUS SUBSTANCES DO NOT CONTAMINATE new products through recycling

PROTECT (informal) workers

Figure 7. Actions for addressing chemicals of concern along the plastics life cycle. ▲

mechanical recycling, then chemical recycling and thermal recovery, and finally controlled disposal. The lower on the hierarchy an action is, the lower the cost-efficiency and the greater the unwanted trade-offs (e.g., generation of unintentional POPs and additional waste streams in chemical recycling and energy recovery processes).

Especially as circular economy approaches are gaining momentum at the international level (e.g., through activities of UN organizations[4]), the use of chemicals of concern in plastics must be urgently addressed so as not to hinder the future reuse, repurposing, and recycling of plastic products and polymer streams. This requires measures to reduce the overall reliance on and use of plastics, stricter regulations that reduce the use, overall number, and variety of chemicals (particularly hazardous chemicals) in plastics (Fenner and Scheringer 2021), and requirements for industry to share information on the uses of all chemicals in plastics.

Since plastics and their associated chemicals can lead to adverse effects at all stages of the plastics life cycle – from resource extraction to end-of-life – actions should be based on a life cycle approach. This requires using insights from life cycle assessments (LCAs) or other tools, which address the entire plastics life cycle and consider how the different stages of product design, resource extraction, production, use, and post-use handling affect one another. This understanding can lead to actions upstream in the life cycle, such as changing service and business models to reduce the consumption of non-essential plastics. In cases where the use of plastics is unavoidable, innovative product designs can facilitate reuse and recycling, hazardous chemicals can be avoided or substituted with safer and more

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[4] https://www.unep.org/circularity;
https://www.unido.org/unido-circular-economy;
https://www.undp.org/publications/transitioning-circular-economy-through-chemical-and-waste-management

sustainable alternatives (see Section 6.2) and the challenges of hazardous chemicals in recycling can be overcome (see Section 7.2 and 7.3).

The following sections provide a series of options for different stakeholders, including regulatory authorities, manufacturers, businesses, and consumers, to address hazardous chemicals in plastics in support of a systemic change towards a toxic-free circular economy. An overview of such options is provided in Figure 7.

5.1 Options for regulatory authorities to address chemicals of concern in plastics

Worldwide there are numerous efforts on plastics and their associated chemicals, including national regulations, international actions (e.g., Stockholm Convention, SAICM), academic publications, and initiatives by companies and NGOs. Future actions should build on existing knowledge and avoid the duplication of efforts by identifying and harnessing national, regional and global synergies.

Regulatory instruments are a major tool for controlling and ultimately phasing out chemicals of concern (see Figure 9). Discussed below are several policy options that could be considered by regulatory authorities for addressing the various types of chemicals of concern used or found in plastics.

5.1.1 Phasing out the use of internationally restricted chemicals

Several chemicals used in plastics or other polymers have been listed in international legally binding instruments on chemicals such as the Stockholm, Basel, or Rotterdam Conventions. Examples include flame retardants such as PBDEs, HBCDD, HBB, and SCCPs, which have been listed as POPs under the Stockholm Convention (see Annex 1).

Listing chemicals as POPs leads to their reduction and eventual phase-out. However, all the above-mentioned POPs have received (many) exemptions under the Stockholm Convention for their continued time-limited use, including for their use as plastic additives in several applications UNEP 2019b; UNEP 2019f; UNEP 2019g). Also, several countries have yet to ratify and implement the new amendments related to new POPs. Therefore, these POPs are likely to continue to be used in plastic products or other polymers, increasing the stocks of plastic or other polymers containing these POPs. This will further increase the challenges of recycling these plastics and hamper the development of a circular economy in affected recycling flows (see Section 7.2). In this regard, it will be important to agree on criteria for environmentally sound management of waste (e.g., UNEP 2021c; UNEP 2021h; UNEP 2021i) and set sufficiently low POP content limit values under the Basel Convention to avoid the dispersion of legacy POPs through materials recycling.

Listed POPs used as plastic additives should be phased out as soon as possible at a global level. The Stockholm, Basel, and Rotterdam Conventions have developed various guidance documents and project activities to manage hazardous chemicals in plastics.[5,6] Their swift phase-out and replacement with sustainable alternatives are likely to require capacity-building and technology transfer in developing countries, with a particular focus needed on alternatives assessments and phase-in of safer and more sustainable alternatives to avoid regrettable substitution and burden shifting (see Section 6). To support such substitutions, draft guidance documents on alternatives to SCCPs, DecaBDE, and HBCDD have been developed (UNEP 2019b; UNEP 2019f; UNEP 2019g). Furthermore, a publication on the phase-out of POPs published under the framework of the Stockholm Convention includes information on alternatives to listed POPs as well as on tools for alternatives assessment (Stockholm Convention Regional Center Asia and the Pacific 2014).

5.1.2 Reducing the use of chemicals identified as emerging policy issues and issues of concern under SAICM

SAICM's International Conference on Chemicals Management has identified and called for appropriate action on several "Emerging Policy Issues" and "Other Issues of Concern", several of which are closely related to plastic additives. These include:

- Chemicals in products,
- EDCs,

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[5] A Plastic Waste Partnership (PWP) has been established under the Basel Convention. More information may be found here: http://www.basel.int/Implementation/Plasticwaste/PlasticWastePartnership/tabid/8096/Default.aspx

[6] Guidance on developing and updating National Implementation Plans (NIPs) under the Stockholm Convention may be found here: http://chm.pops.int/Implementation/NationalImplementationPlans/Guidance/tabid/7730/Default.aspx

- Hazardous substances within the life cycle of electrical and electronic products,
- Nanotechnology and manufactured nanomaterials,
- Perfluorinated chemicals and the transition to safer alternatives.

To address these issues, numerous efforts have been undertaken at the national, regional, and international levels (see Annex to UNEP 2020c). A policy brief on chemicals and plastics has been published under SAICM to address plastic-associated chemicals (UNEP 2020b). UNEP has also compiled a list of recognized EDCs (UNEP 2017b), many of which are additives in plastics (see Annex 1). Given its multi-stakeholder nature, further activities dedicated to hazardous chemicals in plastic could be considered within SAICM's "Beyond 2020" process.

5.1.3 Restricting chemicals of concern based on the concept of "essential use"

The use of hazardous chemicals in plastics may be restricted based on the concept of "essential uses." This concept has been applied in the context of the Montreal Protocol,[7] under which a use of a controlled substance should qualify as "essential" only if: (i) it is necessary for health and safety or critical for the functioning of society (which encompasses cultural and intellectual aspects); and (ii) there are no available technically and economically feasible alternatives or substitutes acceptable from the standpoint of health and the environment. Developing criteria for essential uses for chemicals is a key part of the European Commission's Chemicals Strategy for Sustainability adopted in October 2020. This is an ongoing process that will guide the application of essential uses in all relevant EU legislation for both generic and specific risk assessments. The concepts of "acceptable purpose" and "specific exemption" based on socio-economic analysis, as used in the Stockholm Convention, may also be considered.

Scientists have recently requested the application of the concept of "essential uses" to PFASs[8] (Blum *et al*. 2014; Cousins *et al*. 2019; Cousins *et al*. 2021; Glüge *et al*. 2021). The concept of "essential uses" could similarly be applied to the management of other chemicals or groups of chemicals of concern (Cousins *et al*. 2019; Cousins *et al*. 2021), including hazardous or otherwise problematic chemicals that are not yet completely phased out. This means that chemicals of concern would only be allowed in controlled material cycles for essential uses, where no safer substitutes are currently available. For uses that do not qualify as essential, hazardous chemicals would be phased out.

5.1.4 Minimizing and controlling unavoidable hazardous chemicals in plastics through regulatory limits

For some hazardous chemicals in plastics, a complete ban might not be feasible. This may be the case for example for PAHs in rubber tires, where PAHs are unintentionally present in extender oils used in tire production from refining in the petrochemical industry (ECHA 2017b). Also, NIAS are always present to some extent in plastics and their levels may only be minimized up to a certain point.

Unintentional POPs (UPOPs) such as PCDD/PCDFs, PCBs and HCB are formed in the production of some pigments and dyes used as additives in plastic. These UPOPs, as well as unintentional legacy additives from recycling, such as PBDEs, are collectively called unintentional trace contaminants (UTCs) (ECHA 2021c). UPOPs need to be controlled and minimized, as required by the Stockholm Convention. Establishing limit values for UTCs is important to avoid legal uncertainty and to ensure that product and articles are safe. In the absence of a harmonised limit value, some stakeholders may consider the limit value to be the limit of detection, which could be disproportionately low and thus prevent the placing on the market of any substance, mixture or article containing UPOPs or POPs unintentionally present from recycling (European Commission 2021a). Meaningful regulatory limits should be low enough to ensure the substance of concern cannot be intentionally used in products and to minimize its unintentional content in plastics or in additives, yet it should be above the detection limit of existing detection methods to enable control and enforcement. Such limits should be subject to periodic review and assessment considering all relevant impacts with regard to health and the environment. (Box 5 to Box 7).

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[7] Montreal Protocol on Substances that Deplete the Ozone Layer

[8] One of such essential uses is the production of fluoropolymers, for which PFOA listed as a POP has been exempted under the Stockholm Convention (UNEP 2019h), but has been substituted in developed countries with other PFASs such as GenX (which has been assessed for toxicity (e.g., US EPA 2021b)).

Box 5. Setting limits for PAHs in rubber

ECHA (2017b) and the Dutch National Institute for Public Health and the Environment (RIVM) (2018) assessed the risks associated with playing football on synthetic turf pitches on which end-of-life tire-derived rubber granules containing PAHs were used as infill. Both ECHA (2017b) and RIVM (2018) concluded that the presence of carcinogenic PAHs in granules at the concentration currently allowed in these mixtures in the EU poses an unacceptable risk for some athletes playing on synthetic turf pitches that surpass the acceptable risk level of 1 x 10^{-6} under specific exposure conditions. ECHA recommended initiating a restriction to lower the currently allowed concentration limit for PAHs in these granules, to ensure synthetic turf pitches are safe for their users. This restriction has now been adopted by the EU.

While the EU has the most comprehensive regulations regarding PAHs in artificial turf, also China passed a national standard in 2020 that stipulates a limit of PAHs amongst other substances that may be contained in vulcanized rubber powder (i.e., a product of waste rubber recycling). The 18 kinds of PAHs are divided into three levels, with limits of 150 mg/kg for level I, 200 mg/kg for level II, and 300 mg/kg for level III (Zuccaro *et al*. 2022). Generally, restrictions of PAHs in rubber turf and other recycled rubber products might be especially important for developing countries with hotter climates and high volatilisation rates of SVOCs, and where recycled tire rubber may be used in products with larger exposure potential such as shoes and furniture.

Box 6. Setting limits for unintentional POPs related to plastics or additives

Some unintentional POPs such as PCBs or HCB are restricted in certain applications, such as in pigment additives (UNEP 2013b; Weber 2015; UNEP 2021b). For example, Japan regulated the presence of unintentional PCBs in chemicals by setting a limit of 50 parts per million (ppm; mg/kg) which is relevant for certain pigments used in plastic or inks (Weber 2015; UNEP 2021b). Also, the European Commission has developed a UTC limit for HCB of 10 mg/kg in chemicals and products in a draft act to amend the EU POPs Regulation (EU) 2019/1021 (European Commission 2021a).

The UNEP unintentional POP Toolkit (UNEP 2013b) defines some BAT levels for unintentional HCB, for instance in tetrachlorophtalic acid, which is used for a range of plastic pigments such as SolventRed 135 (Government of Japan 2006). One may hence set limits for UPOPs in plastic additives considering the BAT concentration in the UNEP toolkit for these chemicals (UNEP 2013b).

Box 7. Establishing limit values for POPs in waste to ensure safe recycling

Legacy additives such as PBDEs or HBCD can contaminate new products through recycling (Chen *et al*. 2009; Kuang *et al.* 2018). The Basel Convention has set three provisional low POPs limits for PBDEs of 50, 500, and 1000 mg/kg in waste, which determine recyclability (UNEP 2021c). Furthermore, the EU has set a limit of 10 mg/kg for individual PBDEs in new plastic products (European Parliament 2019). A study for the European Commission compiled data on UTC levels in wastes to support the assessment of impacts associated with the review of POP limit values in waste (European Commission 2021b).

Additionally, PBDEs and some other BFRs contain brominated dioxins and furans (PBDD/PBDFs), which can contaminate plastic products (Sindiku *et al*. 2015b). PBDEs, as dioxin precursors, additionally form PBDD/PBDFs during recycling processes and extrusion (Ebert and Bahdir 2003; Weber and Kuch 2003; Zennegg *et al*. 2014). High levels of PBDD/PBDFs were detected in plastic toys containing PBDEs below 1000 mg/kg (POP limit of the Restriction of Hazardous Substances in Electrical and Electronic Equipment (RoHS)), highlighting that considerably lower PBDE limits are needed and that PBDD/PBDFs need to be considered when setting such limits (Petrlik *et al*. 2019c; Budin *et al*. 2020).

5.1.5 Considering hazardous chemicals in regulatory frameworks for plastic recycling

For POPs-containing waste, the Stockholm Convention establishes requirements (as set out in Article 6 paragraph 1(d)) to "take appropriate measures so that such wastes, including products and articles upon becoming wastes, are: (i) handled, collected, transported and stored in an environmentally sound manner; (ii) disposed of in such a way that the POP content is destroyed or irreversibly transformed so that they do not exhibit the characteristics of POPs or otherwise disposed of in an environmentally sound manner when destruction or irreversible transformation does not represent the environmentally preferable option or the POP content is low, taking into account international rules, standards, and guidelines, including those that may be developed pursuant to paragraph 2, and relevant global and regional regimes governing the management of hazardous wastes; (iii) not permitted to be subjected to disposal operations that may lead to recovery, recycling, reclamation, direct reuse or alternative uses of POPs; and (iv) not transported across international boundaries without taking into account relevant international rules, standards and guidelines." The Stockholm Convention also requires Parties to develop inventories of wastes and stocks containing POPs, along with action plans for managing them (e.g., plastics containing POPs such as WEEE plastic or HBCDD-containing EPS/XPS construction polymers).

The Basel Convention states that waste with POPs concentrations greater than the "low POP content" level must be destroyed or irreversibly transformed. If destruction is neither possible nor environmentally preferable, then an environmentally sound disposal method should be implemented. Waste containing POPs below the "low POP content" level is not considered POP waste, but also needs environmentally sound management (ESM). For PBDEs, the Basel Convention has currently listed three provisional low POP contents (50 mg/kg, 500 mg/kg, and 1000 mg/kg PBDEs) (UNEP 2019d). The lowest suggested limit (50 mg/kg PBDEs) is favoured for example by the African countries and civil society organisations, which would likely prohibit the recycling of WEEE plastic, since most recyclates produced by separating the BFR fraction cannot meet this strict limit (Straat and Nilsson 2018).

In the EU, these requirements translated into the revised EU POPs regulation (POPs Regulation (EU) 2019/1021, which repealed the previous POPs Regulation (EC) No 850/2004), addresses and implements the Stockholm Convention and considers the low POPs content set by the Basel Convention (UNEP 2018c; UNEP 2018d; UNEP 2018e; UNEP 2019d). Depending on the specific risk posed by individual POPs, different allowable levels have been stipulated for waste.

Some countries and regions (e.g., the EU) have set limits for the unintentional trace amounts of various POPs in products, including those made from recycled plastics. Control and inspection activities are conducted to monitor consumer products, and those with POP levels above regulatory limits are taken off the market (UNEP 2019e).

To protect consumers from exposure to additives and NIAS in FCMs, some countries and regions have set specific requirements for additives and the recycling of plastics used for food packaging. The Commission Regulation (EC) No 282/2008, for instance, requires that the recycling process must be authorized and managed by an appropriate quality assurance system, guaranteeing the quality of the recycled materials (European Commission 2008b).

Concerning FCMs, the most comprehensive EU-specific measure is Commission Regulation (EU) No 10/2011 on plastic materials and articles. It sets rules for the composition of plastic FCMs and establishes a Union List of starting substances and additives that are permitted for use in the manufacture of plastic FCMs. The Regulation also specifies restrictions on the use of these substances and sets rules to assess the compliance status of plastic materials and articles. This Regulation is regularly amended. In addition, Regulation EC No 282/2008 on recycled plastic materials intended to come into contact with foods sets requirements that recycled plastic has to fulfil before it can be used as an FCM. Reports on BFRs in kitchen utensils (Kuang *et al*. 2018) that found violations of these regulations indicate that regulatory control cannot rule out misuse of recycled plastics completely and that strict regulatory control of FCM materials is critical.

China has established a "Positive list" of 1,294 additives used for FCMs and articles, including 731 kinds of plastic (GB 9685-2016 Standard for the Uses of Additives in Food Contact Materials and Articles). This standard has an additional Specific Migration Limit for chemicals.

Several countries have also established targets for plastic recycling. For instance, in the EU, the 2018 Strategy for Plastics in a Circular Economy sets targets for plastic waste aiming to ensure that half of all plastic packaging used in the EU is either reusable or cost-effectively recycled by 2025 (European Commission 2018a). Packaging waste is controlled by Directive 94/62/EC, which was subsequently amended by Directive (EC) 2018/852. Furthermore, as of 2015, the End-of-Life Vehicle Directive (2000/53/EC) mandates increasing the reuse and recovery of all end-of-life vehicles to a minimum of 95%, including thermal recovery. Reuse and recycling must be increased to a minimum of 85% by weight per vehicle, excluding energy recovery. Consequently, a portion of the plastics used in vehicles must be recycled, while another portion is thermally recovered. Furthermore, the Restriction of Hazardous Substances in Electrical and Electronic Equipment (RoHS) Directive bans (with exemptions) EEEs containing polybrominated biphenyls (PBBs) and PBDEs (European Commission 2011/65/EU). For e-waste, the European Directive 2002/96/EC (WEEE Directive) and Directive 2002/95/EC (RoHS Directive) support the recycling of plastic by mandating collection targets (45-65%) and recovery rates (55-85%, depending on category) (European Commission 7/4/2012).

In Europe, plastics imported for plastic waste recycling that contain SVHCs must carry certain information if the SVHC content of the material exceeds 0.1%. Some, but not all, SVHCs are banned from use. SVHCs no longer allowed in new plastics products might still be introduced in the waste stream through older plastic products, or from plastics products manufactured outside of the EU. Competent authorities and research institutions monitor products for hazardous chemicals such as POPs and SVHCs. With the European Rapid Alert system for dangerous products (RAPEX) (European Commission – DIGIT n.d.), findings on hazardous additives restricted in products can be swiftly communicated to all 27 EU Member States to effectively remove such products from the market.

The presence of a restricted additive does not necessarily stop recycling, even if the additive cannot be separated. For example, although DEHP is an SVHC, to enable the recycling of soft PVC, the EC granted companies authorisation to recycle despite the presence of DEHP in recycled soft PVC, in compounds, and dry-blends (Stenmarck *et al*. 2017).

5.1.6 Building regulatory capacity for prioritization and restriction of hazardous chemicals

In recent years, public bodies have conducted assessments and regulatory risk management actions for various chemicals and groups of chemicals not addressed at the international level (UNEP 2019f; Aurisano *et al*. 2021b). As a result, a wide range of toxic chemicals in plastics are restricted in some countries or regions (Annex 1; Aurisano *et al*. 2021b), such as lead, DEHP and triclosan (Annex 1; Halden *et al*. 2017; UNEP 2017b; UNEP 2019a). BPA has also been restricted in baby bottles, for instance in the EU and Canada, but not in many developing countries (Weber *et al*. 2018a). Another restriction is a recent ban on halogenated flame retardants in the EU Ecodesign Directive's requirements for electronic displays, which restricts groups of chemicals for specific applications (European Commission 2019a). An EU Commission Staff Working Document on "Restrictions Roadmap under the Chemicals Strategy for Sustainability" has compiled a list of groups of substances for restriction, including several plastic additives (European Commission 2022a). There is proposed work on group restrictions for bisphenols as well as for PVC and its additives (ECHA 2021a; European Commission 2022c). A list of additives restricted for all or specific uses is included in Annex 1.

Developing countries often lack comprehensive and robust regulatory frameworks and restrictions on industrial chemicals, including hazardous chemicals that are already restricted or banned in developed countries (Wagner *et al*. 2013; Weber *et al*. 2018a; Chemical Watch 2019b). This creates challenges in terms of regulating the chemicals used in local plastic production as well as those contained in products or waste imported from developed countries. Testing for hazardous chemicals should be required before exporting waste to developing countries. Other possible areas of work include building regulatory capacities (e.g., for identifying chemicals of concern that require prioritization, for conducting chemical risk assessments, etc.) and developing plans for regulatory action in interested developing countries.

The science-based hazard criteria for chemicals in FCMs developed by Groh *et al*. (2018) or modified approaches could be used to prioritize action on

Table 5. Hazard grade scores assigned to different CLP classifications suggested by Groh *et al*. (2018).

Hazard class	Hazard category	Abbreviation	Hazard grade score	Contribute to sum hazard scores for
Carcinogenicity	1A	Carc. 1A	10,000	human health
	1B	Carc. 1B	10,000	
	2	Carc. 2	100	
Germ cell mutagenicity	1A	Muta. 1A	10,000	
	1B	Muta. 1B	10,000	
	2	Muta. 2	1000	
Reproductive toxicity	1A	Repr. 1A	10,000	
	1B	Repr. 1B	10,000	
	2	Repr. 2	100	
Lactation toxicity	n/a	Lact.	100	
Acute toxicity; oral, dermal, inhalation	1	Acute Tox. 1	1,000	
	2	Acute Tox. 2	1,000	
	3	Acute Tox. 3	100	
	4	Acute Tox. 4	10	
Respiratory sensitization	1	Resp. Sens. 1	1,000	
Skin sensitization	1	Skin Sens. 1	1,000	
Specific target organ toxicity – Single exposure	1	STOT SE 1	1,000	
	2	STOT SE 2	100	
	3	STOT SE 3	10	
Specific target organ toxicity – Repeated exposure	1	STOT RE 1	1,000	
	2	STOT RE 2	100	
Aspiration hazard	1	Asp. Tox. 1	100	
Skin corrosion	1	Skin Corr. 1	100	
	2	Skin Corr. 2	10	
Skin irritation	1	Skin Irrit. 1	100	
	2	Skin Irrit. 2	10	
Serious eye damage	1	Eye Dam. 1	100	
	2	Eye Dam. 2	10	
Eye irritation	1	Eye Irrit. 1	100	
	2	Eye Irrit. 2	10	
Hazardous to the aquatic environment – Chronic exposure	1	Aq. Chronic 1	1,000	environment
	2	Aq. Chronic 2	100	
	3	Aq. Chronic 3	10	
Hazardous to the aquatic environment – Acute exposure	1	Aq. Acute 1	100	

chemicals in plastics (Table 5). The sum of hazard scores from every applicable hazard class can be calculated for each chemical, and the chemicals with the highest scores may be prioritized for further action. However, the included hazard classes are not exhaustive and do not cover, for example, PBT/vPvB substances, EDCs, etc. Therefore, this approach should only be used to support prioritization.

The methodology and criteria suggested in the Global Chemicals Outlook II for "other issues where emerging evidence indicates a risk to human health and the environment" (UNEP 2019a) could also be considered. The criteria, i.e., entry points and necessary conditions for inclusion, are that "*at least two countries/regional economic integration organizations have recently (since 2010) undertaken one of these two types of action,* *including at least one regulatory risk management action (UNEP 2019a):*

- *There has been a regulatory risk management action on a chemical or group of chemicals, based on emerging evidence indicating a risk to human health and the environment.*
- *A full risk assessment or reassessment action for the same chemical or group of chemicals has been completed or initiated."*

It should be noted that there is no one-size-fits-all method for assessing and prioritizing chemicals, as each method has its shortcomings. Therefore, where feasible, more than one method should be explored and consensus scoring (i.e., combining results from multiple methods) could then be used to guide prioritization efforts.

5.1.7 Tracking chemicals in plastics to enable a toxic-free circular economy

The significant knowledge gaps regarding the use and presence of chemicals such as additives and NIAS in plastics not only hamper hazard, exposure, and risk assessments (see Chapter 4), but also cause problems for managing plastics in a circular economy (see Chapter 7). These knowledge gaps arise primarily due to a lack of transparency from industrial producers regarding the identities and quantities of chemicals used in different applications, which are typically held back as confidential business information. However, as stipulated in the Dubai Declaration on International Chemicals Management of 2006, "[…] information on chemicals relating to the health and safety of humans and the environment should not be regarded as confidential."

Transparency is crucial for tracking hazardous chemicals along the life cycle of plastics to promote a toxic-free circular economy. The hazardous additives that are deemed to be essential should be tracked and managed in controlled cycles. This can be achieved for instance by maintaining inventories and labelling products. Therefore, policy measures or incentives should be put in place to promote transparency (Wang and Praetorius 2022).

For certain chemicals, there are ongoing efforts to keep track of their uses and presence in products. For instance, the Stockholm Convention requires its Parties to develop inventories that compile information on past and present production and uses of listed POPs. Labelling requirements have been introduced by the Convention for a first plastic additive (HBCDD). In the EU, the SCIP database established under the Waste Framework Directive (2008/98/EC) compiles information on SVHCs present in products on the EU market. The SCIP database aims to ensure the availability of information on SVHCs throughout the whole life cycle of products and materials, including at the waste stage; encourages substitution of these hazardous chemicals; and contributes to a circular economy by helping waste operators ensure that such hazardous chemicals do not cross-contaminate recycled materials. The EU is also planning to implement a Digital Product Passport, based on blockchain technology, to store and share information throughout a product's life cycle. This is intended to provide industry stakeholders, businesses, public authorities, and consumers with a better understanding of the materials used in the product and the product's environmental impacts (University of Cambridge Institute for Sustainability Leadership and the Wuppertal Institute 2022).

UNEP's "Chemicals in Products (CiP) Programme" focuses on strengthening policy and access to information on chemicals contained in everyday products. The Programme aims to improve the availability and accessibility of information for stakeholders throughout the supply chain, enabling them to effectively manage these products and the chemicals they contain.

Another example is the European Plastic Additives Initiative, led by ECHA and more than 20 industry sector organizations (ECHA 2019a). This initiative explored how information on uses and exposures could assist regulatory work under the EU REACH Regulation. Outcomes included the development of a method to compare the release potential of different additives (ECHA 2019a; see section 3.2).

Inventories of plastic additives also provide a basis to understand the overall use of additives in plastics (Groh *et al*. 2018; ECHA 2019a; UNEP 2019f) and to determine whether further assessments and restrictions are needed. Such an inventory was established as part of developing the current report (Annex 1) and could be used as a starting point for future initiatives.

Some countries have screened the additives in certain plastic applications, such as CMRs in toys or SVHCs in consumer products, or the chemicals migrating from products (Danish EPA 2015; Stenmarck *et al*. 2017; Bohlin-Nizzetto 2022). The International Chemical Secretariat (ChemSec), an international NGO, established and maintains a Substitute It Now (SIN) List of about 1,000 substances that may qualify as SVHC under EU's REACH regulation, including EDCs and persistent, mobile, and toxic substances (PMTs) (ChemSec 2022).

5.2 Options for manufacturers and businesses to phase out chemicals of concern

A complementary approach to phasing out chemicals of concern in plastics is the phase-out of chemical groups through criteria in voluntary initiatives set by producers, eco-labels, and procurement schemes. While the grouping of substances is not yet included in regulatory schemes but is only suggested (e.g., European Commission 2022), it can be applied in voluntary schemes by companies, including retailers, as well as encouraged through Sustainable/Green Public Procurement by public authorities or Eco-labelling. Eco-labels and sustainable or green procurement, including by the public sector, can go beyond legal requirements and promote front runners. An example is the phase-out of chemical groups (e.g., phthalates, fluorinated compounds, EDCs) by the retailer Coop Denmark (Chemical Watch 2015). As another example, Greenpeace demanded in their "Detox campaign" that fashion brands commit to eliminating the use of all hazardous chemicals by 2020, including chemical groups like PFASs and phthalates.

The New Plastics Economy Global Commitment, launched by UNEP and the Ellen MacArthur Foundation in 2018, provides a list of criteria to help identify problematic or unnecessary plastic packaging and plastic packaging components. One of the criteria is whether the packaging or component "contains, or its manufacturing requires, hazardous chemicals that pose a significant risk to human health or the environment (applying the precautionary principle)." Hazardous chemicals are defined as those that show intrinsically hazardous properties (PBT, vPvB, CMR, EDC, or equivalent concern), and include not just those that have been regulated or restricted in other regions (Ellen MacArthur Foundation and UNEP 2019).

Industry has a range of global activities to phase hazardous chemicals out of products. For instance, triggered by Greenpeace's DETOX campaign, the Zero Discharge of Hazardous Chemicals (ZDHC) group has developed a Manufacturing Restricted Substances List (MRSL) of chemical substances that are banned from intentional use in facilities processing textile materials, leather, rubber, foam, adhesives, and trim parts for textiles, apparel, and footwear (ZDHC 2015; ZDHC 2021). The auto manufacturing industry and the electronics industry have similar lists of restricted substances, including restricted plastic additives.

In the food packaging sector, partnerships between food service companies and civil society organizations have led to the development of tools such as the Understanding Packaging (UP) Scorecard (SUM'D 2022), which helps procurement professionals avoid plastic food contact materials that contain hazardous chemicals.

5.3 Raising awareness about chemicals in plastics

In addition to reducing the production and use of non-essential plastics, educating businesses and consumers about the risks of exposure to chemicals in certain plastic products that are in use or essential can help reduce exposure. It can further nudge the general public towards an overall reduction in consumption, which is needed particularly in developed countries where the ecological footprint is three to ten times too high (Potocnik *et al*. 2018; Spangenberg and Loyek 2019; Global Footprint Network 2021).

A critical aspect is the disclosure of information on chemicals in plastics intended for use in consumer products, for instanceas requested by NGOs for UNEP's Chemical in Product activities (Speranskaya *et al*. 2017). Under the EU REACH Regulation, for example, consumers have a "right to ask" about SVHCs in the products they buy. This means that if a consumer wants to know whether or not a product (e.g., plastic item) contains an SVHC at concentrations above 0.1%, they can ask the supplier for this information. If the product indeed contains an SVHC, the supplier is legally obligated to respond to the consumer, as a minimum with the name of the SVHC. A smartphone app called Scan4Chem[9] developed by the EU LIFE project AskREACH is now available to consumers in some EU Member States. This free-of-charge app allows consumers to scan the barcode on a product and easily request information from suppliers about the presence of SVHCs.

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[9] The application may be accessed here: https://www.askreach.eu/app/. Accessed 26 April 2023.

Some governments have developed materials to educate different stakeholders (e.g., German Environment Agency 2010; German Environment Agency 2011). A new SCIP database in the EU, for example, compiles information for the waste management sector regarding SVHCs in products in the EU (see section 5.1.7). The Stockholm Convention regional centers have also developed educational materials on hazardous chemicals in plastics (Weber *et al*. 2018a), including films (Basel, Rotterdam and Stockholm Conventions Secretariat n.d.).

Science NGOs such as the Plastic Soup Foundation are running science-based educational campaigns to reduce and control plastics and related chemical exposures. Their Plastic Health Summits from 2019 and 2021 are available as video playlists and cover a wide range of information about plastics, microplastics, chemicals in plastic (including additives and NIAS), and related exposure and health risks (Plastic Soup Foundation 2019a; Plastic Soup Foundation 2021)). The Foundation also compiled additional information on various plastic-related topics (Plastic Soup Foundation 2019b), including information geared towards children and youth (Plastic Soup Foundation 2019c). Some NGO campaigns address specific plastics with a high content of additives, such as soft PVC. Examples include the Center for Health, Environment & Justice and their publications on PVC for consumers, parents and schools (Center for Health, Environment and Justice n.d.).

Several other awareness-raising tools and educational materials have been developed and are readily available in different formats to target the specific needs of different stakeholder groups. One example is the information compiled by the Food Packaging Forum for policymakers and consumers, such as their dossiers (Box 8), as well as the scientific presentations from their Food Packaging Forum Workshops, webinars and YouTube channel (Food Packaging Forum (e) n.d.). The organization also developed and maintains a set of interactive databases tracking chemicals found to have migrated from plastic FCMs into food (Food Packaging Forum (b) n.d.) as well as initiatives from major international food brands and retailers to improve the safety of chemicals in their packaging (Food Packaging Forum (c) n.d.). A set of fact sheets on different food contact materials, including plastics and bioplastics, are also available (Food Packaging Forum (d) n.d.). Grob (2019) illustrated the challenges in assessing and regulating chemicals in plastic and other FCMs and suggested potential ways forward for consideration by policymakers, industry and the scientific community.

Several civil society organizations are also raising awareness about the impacts of the plastics value chain on human health and the environment and the need for systemic change, including changes in consumption habits such as reducing, or even eliminating, plastic use in daily life (e.g., Break Free From Plastic, the Story of Stuff Project on Plastic or the Plastic Pollution Coalition[10]).

Organisations such as the International Pollutants Elimination Network (IPEN), a global network with an emphasis on low- and middle-income countries, have highlighted the need for regulatory controls to keep hazardous chemicals out of plastics (International Pollutants Elimination Network n.d.). As part of its work, IPEN conducts research in cooperation with scientists and scientific organizations (Adu Kumi *et al*. 2019; Budin *et al*. 2020; Endocrine Society and IPEN 2020; Petrlik *et al*. 2018; Petrlik *et al*. 2019a; Petrlik *et al*. 2019c)), generating data from countries where they are mostly lacking,

UNEP and its partner, GRID-Arendal, are currently collecting indigenous knowledge and nature-positive solutions for tackling marine litter and plastic pollution.[11] Also, UNEP produced a short film on chemicals in plastics (UNEP 2022d).

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[10] More information on these organizations may be found on their respective websites: https://www.breakfreefromplastic.org; https://www.storyofstuff.org/plastic/, https://www.plasticpollutioncoalition.org/ Accessed 26 April 2023.

[11] The survey may be accessed at: https://www.grida.no/activities/743 Accessed 26 April 2023.

Box 8. The Food Packaging Forum's dossiers on chemicals in FCMs

The Food Packaging Forum provides dossiers with in-depth background information about different topics on food contact materials (FCMs) and health, mainly related to plastics. The dossiers compile scientific knowledge, market data, and regulatory information in an easily accessible format. The dossiers include available data on chemical migration into food and human health effects.

- The dossiers on non-intentionally added substances (NIAS) (Geueke 2013a, Geueke 2018) give an overview of the sources and challenges of NIAS in plastic, which include side products, breakdown products, and contaminants. Although more and more NIAS have been identified over time, most remain unknown. Therefore, the risk assessment and management of NIAS are especially challenging.

- The dossier on plastic recycling (Geueke 2014b) addresses plastic recycling processes and legal requirements that have to be fulfilled for recycled plastic used in FCMs. Market and recycling data, safety issues and sources of pollution are discussed. A special focus is placed on the recycling of polyethylene terephthalate (PET).

- The dossier on bioplastics (Geueke 2014c) summarizes the types, applications, toxicity, and regulation of bio-based polymers and biodegradable plastic used in FCMs.

- The dossiers on BPA and BPS (Geueke 2014e) provide background information about the broad application of BPA in epoxy coatings and polycarbonates, its toxicity, and its regulation. Furthermore, it includes a short summary of the most debated issues regarding the research on BPA. The dossier provides information on the applications, toxicity, exposure, and regulation of BPS and discusses its role as a possible substitute for BPA in FCMs.

- The dossier on melamine resins (Geueke 2013b) summarizes their applications mainly in FCMs, toxicity, risks and regulations of the resins and substance itself.

- The dossier on silicones and silicone-based FCMs (Geueke 2015) provides an introduction to the versatile use of silicones as fluids, rubbers and resins in FCMs and additives in plastic. The dossier covers areas of application and current regulations and reviews the migration potential, exposure and toxicology of silicones.

- The dossier on Threshold of Toxicological Concern (TTC) (Geueke 2013c) gives an introduction to this tool for estimating toxic potential. Both TTC and the Threshold of Regulation assign human exposure thresholds to substances with known chemical structures, but unknown toxicity. The dossier reviews the scientific and historical background of threshold concepts and their advantages and challenges.

6

SUBSTITUTING CHEMICALS AND PLASTICS WITH MORE SUSTAINABLE ALTERNATIVES

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6 / SUBSTITUTING CHEMICALS AND PLASTICS WITH MORE SUSTAINABLE ALTERNATIVES

Alternatives assessment (used for replacing hazardous chemicals in a specific product or process application) and chemical substitution (used more broadly for replacing hazardous chemicals in one, many, or across all applications and sectors) are rapid-screening level tools suitable for evaluating the best options among various alternatives at early design stages (Hansson, Molander, and Rudén 2011; Geiser *et al*. 2015). These tools compare alternatives to hazardous chemicals in products and processes based on a common chemical or product function or a societal service (Tickner *et al*. 2015). This aligns with the initiative on safe and sustainable-by-design (SSbD) chemicals and materials under the European Chemicals Strategy for Sustainability (CSS). This initiative aims to develop a set of hierarchical criteria for safety and sustainability to benchmark newly developed chemicals and materials, while fostering innovation and market competitiveness (Caldeira *et al*. 2022).

Substitution is a tool that informs decisions affecting the entire life cycle of a chemical and the related product or process application(s), including the use of resources for synthesizing chemicals in plastics; emissions during synthesis and manufacturing of chemicals and plastic materials; exposure and emissions during product use; and waste and pollution streams at the end-of-life of plastic products.

To continue recycling “safe” molecules for a long time, industry needs to create materials that are benign by design and build systems, infrastructure, and technology that facilitate the design, production, and recycling of chemicals in plastics in a way that is both feasible and environmentally sustainable. This requires the use of quantitative chemical hazard and exposure evaluation tools that also consider relevant trade-offs across different chemical and plastic life cycle stages and across impacts. Such tools can be used to assess and then optimize material chemistry for human and environmental health, to support better decisions in the design phase. Chemicals must be assessed across a comprehensive set of human and environmental health criteria to enable selecting suitable chemistries for essential plastic applications (Werner *et al*. 2018).

However, as described in scientific reports, obtaining high-quality data on chemical toxicity and environmental impacts is difficult. Reasons for this include the high complexity of supply chains and manufacturers’ reluctance to openly share the chemical composition of their products due to commercially confidentiality claims (UNEP 2019i). Transparency regarding chemicals present in plastics could be increased by having an appropriate authority review confidentiality claims and by creating incentives that promote improved information-sharing for chemicals in products and along plastic supply chains (Wang and Praetorius 2022).

Existing substitution approaches and tools take as starting point information on chemicals of concern rather than on a portfolio of safe and sustainable candidates, since the latter is in many cases unavailable. There are, however, concerns about whether an approach based solely on a list of restricted substances can adequately identify and promote innovative and sound substitions of harmful chemicals in plastics without identifying what is safe or preferred for use (Werner *et al*. 2018). Circular economy strategies need to focus on proactively assessing and screening material chemistries to avoid regrettable substitutions and reduce the toxicity of the materials circulating in commerce. Broader application and further development of effect-based testing approaches can also guide the substitution efforts and ensure the toxicological safety of plastics in a circular economy (Groh *et al*. 2019).

Currently, hazardous chemicals in plastics are mostly substituted on a chemical-by-chemical basis, since using drop-in chemicals (i.e., replacement chemicals that can be used without altering existing production processes) is easiest and often perceived as most technically feasible (Jacobs *et al*. 2016). However, a much broader perspective is often needed to consider various other alternatives besides individual drop-in chemicals. Drop-in chemicals often show physicochemical properties very similar to those of the chemicals they replace, hence they may have similar or even worse impacts on human and environmental health (Wang *et al*. 2013; Stieger *et al*. 2014).

Additionally, existing substitution approaches are mainly focused on qualitatively assessing hazards and usually lack a streamlined consideration of human and ecosystem exposure (that is needed to differentiate between different chemical application contexts) and of life cycle impacts (that are needed to ensure the sustainability of chemicals across the products they are used in). Including such exposure- and life cycle-related aspects should of course comply with the requirements of a rapid-screening level substitution process (Geiser *et al*. 2015). An extensive review of existing substitution tools revealed gaps associated with data on exposure and life cycle impacts, a lack of rapid screening methods to quantitatively address different exposure patterns and largely missing data for chemical supply chains in support of a truly circular economy model (Jacobs *et al*. 2016). From this review and similar efforts, it becomes clear that a more global perspective, a comprehensive set of data and indicators, and a consideration of all relevant aspects and life cycle stages are necessary to facilitate a successful phase-out and substitution process for hazardous chemical additives in their various plastics applications. Furthermore, it is important to build the necessary capacity at the company and country levels to facilitate such phase-out processes, particularly in developing countries (Fantke *et al*. 2015). The time- and resource-intensive compilation of exposure data might also pose challenges to risk assessment. For instance, the EU's Chemicals Strategy for Sustainability mentions the ambition to move away from the current risk-based approach, which considers human exposure levels in addition to a chemical's hazard, and instead emphasizes an improved and harmonized hazard-based approach. Yet, it is important to include exposure considerations to develop national and global risk reduction targets.

6.1 Some major challenges with the sustainable substitution of plastics and related chemicals

6.1.1 Identifying safer and more sustainable alternatives

The biggest issue related to the phase-out and substitution of existing chemicals in plastics with non-toxic alternatives is that the potential decision contexts (e.g., replacing a chemical in a single plastic product from a company perspective vs. phasing out a chemical across all applications from a regulatory perspective) and focus points (e.g., worker vs. consumer vs. environmental health) in a substitution assessment are too often considered separately (European Commission 2006; European Commission 2008a; Giubilato *et al*. 2016; Fantke *et al*. 2020a). Several of these decision contexts may include various combinations of occupational, consumer, population, and ecosystem exposures and toxicity impacts along the plastic product life cyles. Thus, the potential trade-offs (e.g., decreased toxicity potential of an alternative that is more complex to produce, hence leading to higher energy consumption and related greenhouse gas and toxic pollutant emissions) and environmental sustainability impacts along the entire product life cycle often remain unaddressed (Fantke and Illner 2019). To avoid shifting the burden from one aspect to another, both specific risks and sustainability impacts should be addressed at the same time for each product life cycle when identifying possible solutions in chemical substitution and alternatives assessment (Sexton and Linder 2014; Fantke and Illner 2019).

Several substitution frameworks and supporting resources exist, such as those listed in the OECD SAAToolbox (OECD n.d.), or ECHA's information on substitution to safer chemicals (ECHA n.d.). Nowadays, the available alternatives assessment frameworks focus mostly on hazard as well as technical and economic feasibility, without considering environmental sustainability aspects or taking an absolute perspective for environmental sustainability (i.e., benchmarking chemical pollution pressure against actual ecological carrying capacities to stay within pollution limits that do not disrupt local-to-global ecosystems; see e.g., Kosnik *et al*. 2022). To support chemical and product design in relation to any existing or future environmental sustainability targets, such targets must be incorporated into current assessment tools. Furthermore, the current alternatives assessment practice relies mostly on qualitative or categorical metrics. Such metrics are not entirely suitable for sustainably managing chemicals in plastics in a circular economy, which also requires information on all possible applications, related exposure contexts and life cycle sustainability aspects to facilitate sustainability along entire chemical supply chains and all uses across product life cycles (Fantke and Illner 2019).

More importantly, the relevant methods for substituting hazardous chemicals in specific plastic applications are currently not included in design workflows at relevant companies. There are several drivers for the lack of consideration of exposure, life cycle impacts, and other data and metrics that could help identify suitable alternatives to hazardous chemicals in plastics. These include a lack of substitution frameworks that consistently consider all relevant aspects, a lack of understanding of design workflow requirements across different sectors and companies, and a lack of methods for extrapolation and filling data gaps to assess a wide range of chemicals as possible alternatives to hazardous chemicals. These gaps have been recently acknowledged in a European effort to identify research and implementation requirements for a more effective substitution of chemicals across various sectors and products (van der Waals *et al*. 2019).

6.1.2 Challenges related to resource consumption and climate change

Since plastic production is increasing globally and will continue to do so in the foreseeable future (e.g., Geyer 2020), the extraction of resources for producing the necessary chemical constituents for the different plastic applications will also increase. Around 98% of plastic additives are still synthesized from extracted fossil resources, with the remaining 2% from bio-based chemicals (Ögmundarson *et al*. 2020a). Both fossil resources and bio-feedstocks come with their own challenges concerning human health and the environment. While chemically similar in structure after chemical synthesis, biochemical and fossil chemical supply chains differ widely in terms of feedstock (e.g., crude oil, corn, macroalgae) and related refinery processing. This implies that it is necessary to consider the entire supply chain to capture relevant trade-offs between impacts, such as climate change and ecotoxicity, to minimize the environmental impacts of chemical additives in plastics produced from both fossil and bio-feedstock resources.

While in many cases the greenhouse gas emissions from bio-based chemicals in plastics seem lower than those from fossil-based chemicals in plastics at a first glance, this is not typically the case when taking broader environmental impacts into consideration for assessing the production system (for example, the land-use pressure and related climate change impact from bio-based plastics and bio-based chemicals). The life cycle impacts of bio-based polymers on human health and ecosystem quality can be up to two orders of magnitude higher than those of fossil-based polymers, primarily due to pesticide use and pre-harvesting burning practices (Tsiropoulos *et al*. 2015). In addition, biochemicals may have a larger “environmental footprint” with respect to other impacts, such as ecotoxicological or eutrophication-related impacts from the use of pesticides and fertilizers, respectively, when using crops as biofeedstocks (Ögmundarson *et al*. 2020b).

Bio-plastic and bio-based chemicals may also have a larger “environmental footprint” with respect to other impacts. For example, due to increased land use change and other impacts, bioplastics are not yet a cost-effective strategy for climate change mitigation, nor for reducing pressures on scarce land and water resources compared to conventional feedstock, due to market-mediated greenhouse gas emissions from land use change (Escobar *et al*. 2018). Also littering on land and degradation in home-composting is a challenge (Greene 2012; Escobar *et al*. 2018). This is especially concerning given that the global bioplastics production capacity is set to increase from around 2.1 million tonnes in 2018 to 2.6 million tonnes in 2023 (European Bioplastic 2019). Moreover, biodegradable plastics are less well suited for mechanical recycling, and if not sufficiently separated from conventional plastics in sorting facilities, there is a risk of reduced quality of the recycled material. In addition, there are challenges with the biodegradation of “biodegradable” plastic in the marine environment, with polylactic acid (PLA) showing almost no biodegradation (Greene 2012; UNEP 2015). Toxicity testing of bio-based and biodegradable alternative materials has also shown that these materials may be as toxic as fossil fuel-based plastics (Zimmermann *et al*. 2020). Thus, while bio-based plastic can help address the fossil resource challenge, it will not solve the pollution problem caused by chemicals in plastic. A rapidly expanding area of research concerns the biological and environmental impacts of biodegradable and bio-sourced plastics, as well as industry labelling and certification. The results of field studies show that when PLA and some other bioplastics are outside industrial or controlled composting conditions, such as in the marine environment, they can persist for years (Greene 2012). Therefore, these types of plastics are likely to pose similar risks as conventional plastics in the environment (Narancic *et al*. 2018; Napper and Thompson 2019; Zimmermann *et al*. 2020).

6.1.3 Challenges related to the life cycle management of chemicals in plastics

The main challenges concerning life cycle management include a lack of information regarding the chemical content of products that can become secondary raw materials, as well as regarding the global flows of chemicals in recycled products (UNEP 2019i). Properly managing all chemicals across the plastics life cycle is a very ambitious objective and is considered one of the hardest tasks in the context of a global circular economy (Bodar *et al*. 2018).

The presence of possibly harmful substances in products, whether intentional or unintentional, that originate from the use of both virgin and recycled plastics, poses major challenges (Hahladakis *et al*. 2018; UNEP 2019i). Hence, a life cycle perspective is necessary in order to identify and minimize the exposure of humans and ecosystems to harmful chemicals along plastic-based consumer product life cycles (see Figure 8). It is important not only to understand the potential for cross-contamination with chemical additives that end up as unintentional

Figure 8. **The assessment flow necessary for evaluating chemicals in a given plastic material that enters different product application life cycles through recycling** (Reproduced from Fantke and Illner (2019), Copyright Elsevier). ▼

residues in recycled materials, but also to understand and manage the input of chemical additives in each product application, be it from virgin or recycled plastic.

In addition to the challenge of fully understanding the global supply chains, there is the lack of data and information exchange along the entire product supply chain systems, which include cross-interacting networks around the globe. Accurately managing chemicals in these complex networks is fundamental to achieving responsible consumption and production (Bodar *et al*. 2018; UNEP 2019i). Therefore, after efforts to avoid and reduce non-essential plastics and their associated chemicals such as through reusing, the next challenge for all supply chain actors is how to effectively minimize the potential trade-offs of non-avoidable and non-reusable products – between increasing recycling rates on the one hand, and avoiding consumer and environmental exposure associated with cross-contaminated products on the other. Other gaps and issues regarding managing recycling and circular economy approaches for plastics at the global level include the following (Grundmann *et al*. 2013):

- lack of data about chemicals in plastics products (e.g., chemical weight fractions);
- lack of data about releases and emissions of chemicals from materials and products;
- missing uncertainty factors for various data (including exposure and toxicity information) derived from animal tests (*in vivo*), cell culture tests (*in vitro*), and predictive models (*in silico*) based for example on quantitative relationships between chemical structure and activity, such as toxicity that aims to include all possible effect endpoints from acute and chronic test data (e.g., QSAR modelling);
- limited specific data about informal recycling processes (e.g., efficiency, emissions and releases);
- lack of detailed emission/release models for specific product life cycles and supply chains;
- lack of a clear link between emissions/releases and impacts of plastics-related products and chemicals;
- failure to consider the effects of chemicals leaching from plastic applications during product use; and
- difficulty to generalize findings across LCA studies due to differences such as in the underlying assumptions and modeling choices.

6.2 Improving the substitution of chemicals with safer and more sustainable alternatives

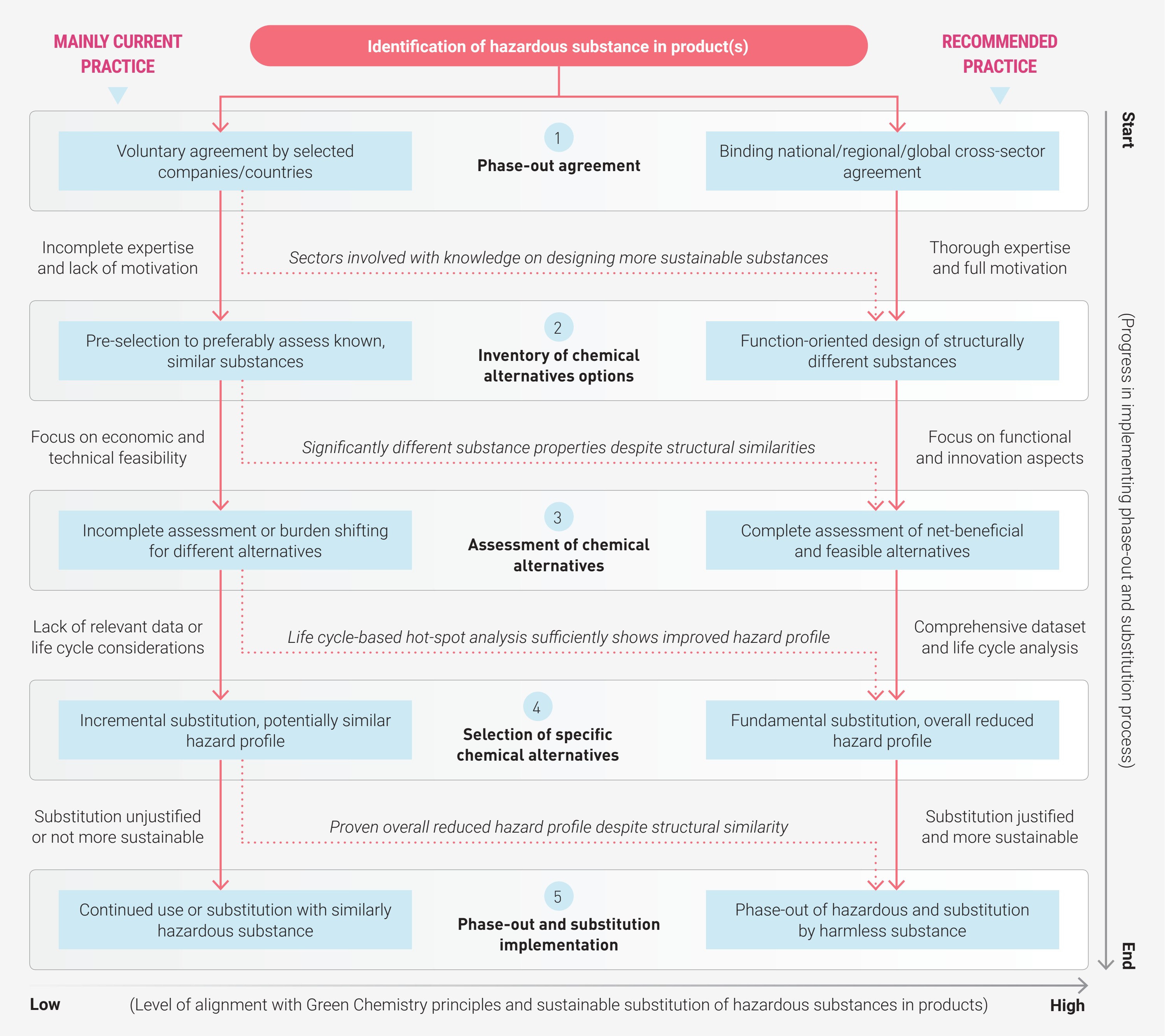
Avoiding regrettable substitutions and managing the wide array of chemicals throughout their different applications in plastics over the entire life cycle requires a reliable assessment of hazard properties as well as potential and measurable adverse impacts, to both human health and the environment, in addition to assessing the performance and costs of these chemicals in their given applications (Fantke *et al*. 2015).

As elaborated above, current substitution approaches mainly focus on technical feasibility, economic viability, and some chemical hazard elements for identifying suitable alternatives to hazardous chemicals in plastics and elsewhere (Geiser *et al*. 2015). However, this can lead to focusing only on similar chemicals that can serve as substitutes to harmful chemicals and leaves application-specific considerations of exposure and life cycle impacts unassessed, which may lead to burden shifting and unaddressed trade-offs (Fantke *et al*. 2015). Instead, in order to truly reduce risks and sustainability impacts, current substitution practices should be adapted to go beyond looking at specific hazards to also: (i) include exposure and life cycle impacts; (ii) require a global agreement on which chemicals to address across sectors and applications; (iii) consider alternatives more broadly, going beyond individual replacement chemicals; (iv) focus on very different chemistries; and (v) consider chemicals in groups, to avoid replacing harmful chemicals with similarly harmful chemicals based on similar molecular structures. This is only possible when substitution goes beyond drop-in chemicals as possible replacements and includes also alternatives at the level of material, product, technology and behaviour. An example is the substitution of chemical flame retardants in textiles and furniture, which might be replaced by fire-resistant fibres or fire barriers between the surface fabric and the interior foam core (Weber *et al*. 2018a). These requirements are summarized in Figure 9 and can facilitate a successful transition from the current limited substitution practice towards the recommended practice where hazardous chemicals in plastics are phased out and replaced with suitable alternatives (Fantke *et al*. 2015).

To include exposure and life cycle considerations, emerging approaches allow for a screening-level comparison of different exposure pathways and are consistent with available life cycle impact assessment methods. Such an approach is the PiF framework, which integrates consumer near-field exposure with environmental emission-based far-field exposure into a consistent, mass balance-based framework (Fantke *et al*. 2016). This framework, included in the California Department of Toxic Substances Control (DTSC) Alternatives Analysis Guide (DTSC 2017), could be considered a suitable starting point for closing gaps in current substitution approaches related to exposure and life cycle impacts.

Finally, it is important to focus on groups of chemicals rather than individual substances, to reduce the use of entire classes of hazardous chemicals rather than phasing out individual problematic chemicals one at a time. This approach helps develop coordinated

Figure 9. **Required modifications to the current substitution approach to achieve a successful phase-out and replacement of hazardous chemicals in plastics** (Reproduced from Fantke *et al*. (2015), Copyright Elsevier). ▼

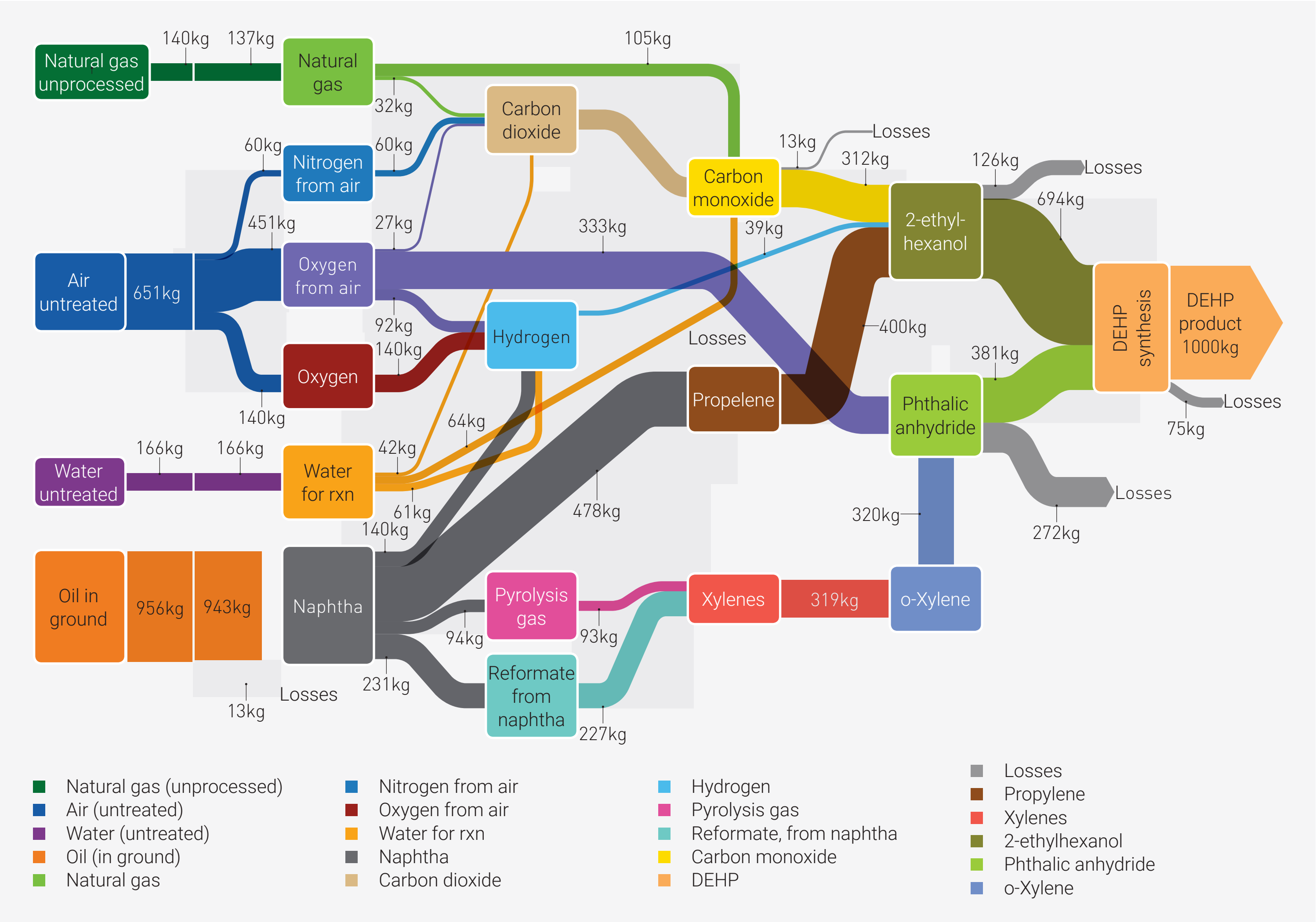


strategies for reducing the production and use of chemicals of concern and prevent regrettable substitutions. Tackling entire groups of chemicals can prove more effective given the great number of chemicals in use, most of which have not been well studied and their impacts on human and environmental health are not sufficiently understood. Moreover, when a harmful chemical is phased out, often only after years of research and negotiations, the replacement is likely to be a "chemical cousin" with a similar structure and potential for harm (Fantke *et al*. 2015; Weber *et al*. 2018a). Grouping strategies have been proposed by public institutions such as the European Commission, as well as by environmental organizations such as the NGOs Greenpeace (Camboni 2017) and the Green Science Policy Institute.[12] One such example is the assessment of bisphenols (Pelch *et al*. 2019).

6.2.1 Function-based substitution of chemicals and plastics

The use of chemicals in plastics, as well as the use of various plastic materials, must be rethought. Alternative materials such as paper or natural fibers can substitute plastic in a wide range of uses, including bags (UNEP 2018b). Banana leaves and other plant leaves, for example, are commonly used in tropical countries as food packaging materials (Laistrooglai *et al*. 2009; Ng 2015). Other nature-based approaches such as biomimicry, which draws inspiration from nature to solve design challenges (Biomimicry Institute 2022), as well as the use of

Figure 10. **Chemical supply chain for di(2-ethylhexyl) phthalate (DEHP) with reactant mass flows (from left to right) required to produce 1,000 kg of DEHP and related releases into the environment. Nodes represent different chemical synthesis integration stages. Losses smaller than 10 kg are not shown** (Reproduced from Fantke *et al*. (2020b) with permission from the Royal Society of Chemistry). ▼



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[12] The Green Science Policy Institute has developed the Six Classes program (http://www. sixclasses.org/).

indigenous knowledge could be considered when substituting plastics and chemicals.

When replacing plastics in a use application, the life cycle and CO_2 footprint of the alternatives to harmful chemicals or the alternatives to plastic materials need to be assessed. The footprint of the plastic alternatives also needs to be critically evaluated, including in terms of consumer safety and eco-balances (Brandt and Pilz 2011, Denkstatt 2015; Trucost 2016). Thereby, it is important to consider the future trajectory of renewable energy implementation, where for example transport of plastics and other goods may in the future be more and more based on non-fossil fuels.

The assessment of the life cycle and CO_2 footprint of alternatives to harmful chemicals is important since different chemicals and materials (consisting of various chemicals) can have substantially different chemical synthesis and plastic manufacturing steps along their respective supply chains. Figure 10 illustrates this for DEHP as an example additive used as a plasticizer in various plastics applications, showing chemical emissions along the DEHP supply chain that are specific to the production of an individual chemical additive. These emissions can vary based on differences in molecular structure, virgin resources used (e.g., crude oil vs. biofeedstocks), and can get even more complex when looking at the composition of specific plastic materials.

To consider a broader scope of alternatives in the substitution process, it is necessary to move away from looking at merely the risks associated with a specific chemical. A more suitable and cost-effective place to start is by considering a broader societal service function, followed by the end-use function of a given product or material application context (this level could build on principles from life-cycle assessment), rather than looking at chemicals with a particular molecular structure (Tickner *et al*. 2015). For example, according to UNEP (2021d), "[r]eusable products have lower environmental impacts than single-use products. The more times a product can be reused, the lower the environmental impact of that product." Starting from the service and end-use function also allows to consider a broader set of alternatives beyond drop-in chemicals of either similar or very different molecular structure (with the former usually demonstrating similar hazards and risks as the chemicals that are to be replaced), and compare these alternatives based on factors beyond merely how they fulfil a given chemical function. Table 6 provides an overview of function-based substitution considerations at different functional levels. Starting at a higher level means increased cost-efficiency, better environmental benefits and fewer potential tradeoffs. This applies also to the substitution of plastics with other materials with the same end-use function (see Table 6 and case studies in Section 6.2.2).

Functional substitution refers to finding suitable alternatives for fulfilling a given function at the appropriate assessment level from chemicals (e.g., preventing/slowing down ignition by various types of chemical flame retardants) to substituting materials (e.g., different materials being inherently inflammable), to modifying the design (change to lower voltage) to societal services (e.g., technologies to provide fire-safe indoor environments). Functional substitution directly addresses the problem of drop-in chemicals with similar properties by evaluating very different chemistries based on fulfilling a specific function in a specific plastic application. However, since each application and alternative (chemicals,

Table 6. Different levels for a function-based substitution of chemicals and plastic materials in various product applications (adapted from Tickner *et al*. 2015).

Functional substitution level	Example of substituting a hazardous flame retardant in household plastic material
Function as service (System, business model, or technology change)	Are there alternatives that could achieve the same purpose at the service level (e.g., ensuring fire-safe indoor environments)? **Result: Alternative technologies, business models or systemic changes that eliminate and reduce plastic materials**
End-use function (Plastic material, product, or process change)	Is there another means to achieve the desired functional properties of a plastic material in a product (e.g., inherently non-flammable materials that would not require the use of chemical flame retardants)? **Result: Plastic material re-design or replacement with other functionally equivalent materials**
Chemical function (Chemical change)	Is there a functionally equivalent chemical substitute (e.g., alternative chemical flame retardants or even plastics)? **Result: Drop-in chemical replacement or alternative chemistries**

materials, technologies, behavioural changes) come with their own specific supply chains and related emissions and exposure settings for both workers, consumers and the environment, exposure and life cycle impacts need to be consistently considered in a function-based substitution approach (Greggs *et al*. 2019; Tickner *et al*. 2019).

6.2.2 Case studies on the substitution of plastics and chemicals in priority sectors

The review of toxic additives and groups of chemicals of concern highlighted a range of priority sectors relevant to human exposure and environmental impacts (Section 2.4). These priority sectors include toys and other children's products, FCMs, personal care products, medical devices, textiles, electronics, and a wide range of other plastic products containing problematic chemicals (e.g., flame retardants, bisphenols, phthalates, PFASs, biocides, heavy metals, PAHs) (see Section 2.3 and 2.4 for more details). In the following subsections, we present specific case studies on substitutions with chemical or non-chemical alternatives in some of these areas of concern (Box 9 to Box 12). Another way to reduce the use of hazardous chemicals is through reduced plastic consumption (e.g., through banning single-use plastics or introducing reuse and refill systems).

Box 9. Case study on substituting PC in baby bottles

PC baby bottles are produced from BPA (CAS 80-05-7), which is used as a building block (monomer) during the manufacturing of the PC polymer. Some BPA monomer remains as an impurity in the final product. BPA monomer impurities can migrate from PC baby bottles into baby food. In addition, BPA is generated and released through depolymerization due to washing with detergents, sterilization and product use. BPA is a well-recognised endocrine disrupting chemical that can cause various human health effects (Michałowicz 2014; EFSA 2015), which led the European Chemicals Agency to include BPA in their list of Substances of Very High Concern (ECHA 2017c). Thus, the effects of BPA and its presence in products used by infants are of concern. Safer alternatives for PC baby bottles are available and range from different plastics (e.g., PA, PES, PP, PE) to different materials such as glass and stainless steel (Simoneau *et al*. 2012; Weber *et al*. 2018a).

Box 10. Case study on substituting PVC-containing chemical additives in plastic toys

PVC toys and childcare products can contain hazardous additives such as phthalates, SCCPs and heavy metals (Chen and Zhang 2013; Ionas *et al*. 2014). The total mass of plasticizers in soft PVC can reach up to 50%. Phthalate esters used in PVC toys include DEHP, DBP, BBP, diisononyl phthalate (DINP), diisodecyl phthalate (DIDP) and di-n-octyl phthalate (DNOP). Children are directly exposed to the phthalates found in PVC toys mainly through mouthing, skin contact and possibly inhalation of volatile compounds. This is because phthalates are not chemically bound to the polymer chains and can therefore migrate and leach out (Ionas *et al*. 2014). The levels of hazardous chemicals ingested or inhaled by babies and children can exceed safe exposure margins (Weber *et al*. 2018a).

Phthalates in plastic toys are of major concern because young children are particularly vulnerable to their harmful effects. Weber *et al*. (2018a) highlighted various options and alternatives to overcome the issue of toxic additives in PVC toys, including the use of alternative plastics and other materials. First, there are other safer types of plastics that can be used instead of soft PVC for toys, such as thermoplastic elastomers (TPEs), ethylene vinyl acetate (EVA) and polyolefins (polyethylene/polypropylene). Second, several natural materials, such as organic textiles and wood, have traditionally been used to make toys, teethers and pacifiers. These materials are usually durable and repairable and are already available on the market (Weber *et al*. 2018a). Furthermore, such materials can be preferable to petrochemical-based plastics considering the global environmental impacts of non-renewable fossil fuels (individual assessment needed).

Box 11. Case study on substituting PVC and phthalates in medical devices

Single-use medical devices made of PVC, such as intravenous tubing and bags, IV catheters, nasogastric tubes, dialysis bags and tubing, blood bags, transfusion tubing, and air tubes, are still widely used in hospitals and health care centers (HCWH 2013). PVC is in fact the type of plastic most commonly used in hospitals (40% of all plastics-based medical devices), while DEHP (Bis(2-ethylhexyl)benzene1,2-dicarboxylate, CASRN: 117-81-7) is the most widely used plasticizer (PVC Med Alliance 2019). DEHP has been classified as toxic to reproduction (class 1B) since it can decrease fertility and may be particularly harmful to unborn babies (SCENIHR 2016). Like other phthalate plasticizers, it is also considered a threat to ecosystems since it can easily leach into the environment during plastic production, use and disposal. Furthermore, PVC is a source of PCDD/PCDFs releases from small-scale incinerators used in developing countries (Ikeguchi and Tanaka 2000; Zakaria and Labib 2003; Coutinho *et al*. 2006). However, a wide range of alternatives to DEHP-containing PVC in medical appliances are available and have been analyzed in terms of their technical and economic feasibility, hazards, risks, and life cycle considerations (Weber *et al*. 2018a).

First, there are many non-PVC materials available, suitable for a wide variety of medical applications, which do not require phthalates or other softeners. These materials include PP, PE, other polyolefins, EVA and silicone (Swedish Environmental Management Council 2012). The Safer Medical Devices Database[1] developed by Health Care Without Harm (HCWH) Europe lists around 200 alternative products to PVC medical devices (HCWH 2013). Examples of medical devices using alternatives to DEHP-containing PVC include:

- PVC-free bags made of EVA, multilayer polyethylene, or polypropylene, which are cost-effective and technically competitive with PVC bags;

- PVC-free intravenous and other bags, which are available for all applications;

- PVC-free tubing made of PUR or silicone, which are on the market for most medical applications;

- Disposable gloves made of nitrile, which are readily available.

Second, in cases where non-PVC materials are not technically or economically feasible, there is also the possibility of substituting DEHP with safer alternative plasticizers such as Hexamoll DINCH (1,2-cyclohexane dicarboxylic acid diisononyl ester), tris(2-ethylhexyl)benzene-1,2,4- tricarboxylate, citrates, adipates, or trimellitates (SCENIHR 2016; Weber *et al*. 2018a).

Box 12. **Case study on substituting halogenated flame retardants in electronics**

Organohalogen compounds have been widely used as flame retardants in plastic materials. DecaBDE was listed in 2017 as a POP under the Stockholm Convention and has for instance been substituted in electronics with halogenated alternatives such as decabromodiphenyl ethane (DBDPE) (CAS: 84852-53-9). Indoor exposure is a known major human exposure pathway for PBDEs (Imm *et al*. 2009), but also for alternative halogenated flame retardants (Araki *et al*. 2014; Niu *et al*. 2019). The EU Ecolabel restricts the use of halogenated flame retardants in the enclosures and stands of electronic displays (European Commission 2019a).

The substitution of PBDEs and other halogenated flame retardants in electronics can occur at three different levels: (i) flame retardant substitution, (ii) material substitution, and (iii) product redesign (Öko-Institut 2003; Pure Strategies Inc. 2005; Weber *et al*. 2018a). The most cost-effective non-halogenated substitutes can be obtained by varying the resin system and using phosphorous-based flame retardants. Of these alternatives, only HIPS/poly(p-phenylene ether) (PPE) is 100% halogen-free. The PC/ABS and PC systems typically contain a small amount of fluoropolymer (roughly 0.3%) for drip resistance. While other alternatives such as metal or wood are possible, they are not widely employed due to cost and performance issues (Öko-Institut 2003; Pure Strategies Inc. 2005; Weber *et al*. 2018a). Finally, it is possible to re-design the products in order to minimize or remove the need for flame retardants in plastic parts. For instance, electronics could be re-designed to separate the voltage supply or the heat supply from ignitable plastics. Alternatively, shielding the power supplies with metal components lowers flame retardancy requirements, making flame retardant-free HIPS a viable alternative (Öko-Institut 2003).

Hewlett Packard (HP) noted that most major information technology brands have largely eliminated PVC and halogenated flame retardants from their products. To avoid substituting with problematic chemicals, HP has assessed 160 substitutes to date using the Green Screen™ methodology, thus generating a positive list of recommended substances available to all its suppliers (PINFA 2017). HP has also been working with different ecolabel schemes. In 2015, the Swedish TCO Ecolabel adopted Green Screen™ and included a list of phosphorus and inorganic flame retardants assessed as Benchmark 2 or better, which are allowed in TCO-certified products (PINFA 2017).

6.3 Building capacity for chemical substitution and related research

6.3.1 Building capacity in support of successful substitution

The successful substitution of hazardous chemicals or plastics requires not only the use of adequate data and tools, but also the capacity to conduct the assessment and, more importantly, to implement the potential substitution (i.e., to phase out a given hazardous chemical or plastic material and replace it with a suitable alternative). Alternatives may include one or more suitable alternative chemicals, but also alternative materials, technologies, as well as behavioural, business model and system changes. Such transitions may be particularly challenging for developing countries and certain companies (in particular small and medium-sized enterprises (SMEs)), which might lack appropriate regulatory or institutional frameworks as well as technical and financial capacities (Fantke *et al*. 2015).

Therefore, awareness-raising, capacity-building and investments are urgently needed to boost the capacity in certain regions to allow companies to substitute hazardous chemicals in plastics around the world. This includes sharing knowledge and technologies with developing countries around safer alternatives and production processes. In many cases, global action is needed to avoid shifting the burden from one region (with sufficient capacity to replace harmful chemicals) to another (with insufficient capacity and hence continued use or import of harmful chemicals). International trade could result in the continued use or re-introduction of harmful chemicals by global supply chains, even where they have been removed from regulated production processes. Addressing this issue is key to controlling the international flow of plastics containing hazardous substances and underlines the need for global agreements for the phase-out and successful substitution of harmful chemicals across product applications (Wiedmann and Lenzen 2018). Capacity-building efforts may focus on building local or international networks wherever individual companies or specific sectors will not be able to develop enough capacity themselves. Through responsible supply chain management, companies with adequate capacity can facilitate improvements in other companies in their supply chain. In addition, capacity building can support the proper implementation and enforcement of existing legal instruments and standards, to promote successful substitution.

6.3.2 Advancing research for an informed, science-based substitution

Successfully assessing and managing chemicals in plastics requires an inclusive approach that involves all relevant stakeholders across the life cycle. Table 7 provides an overview of the stakeholders for each life cycle stage, the related drivers for human and environmental exposure and health impacts, and the available or emerging evaluation and management tools, which can serve as the foundation for a successful substitution.

The most critical research needs are related to improving and complementing existing data, especially data on the chemical content of plastics, the diversity of plastic product applications, related emissions and material-specific processes such as volatilization and partitioning, human and ecosystem exposure patterns, human exposure assessment, toxicity, and methods to evaluate toxicity. Binding rules should ensure that plastic manufacturers make the chemical composition of their products transparent. This would save the need for additional supply chain research later, for example when hazardous chemicals are identified.

Data on the chemical content of the different plastic applications is required as a starting point for identifying chemical exposure. Initial approaches derive a probability for the most likely function of a given chemical across all product applications and estimate the weight fraction of the chemical in different applications (Isaacs *et al*. 2016; Phillips *et al*. 2017). However, this approach requires further refinement to (i) identify multiple functions per chemical where applicable (e.g., a phthalate may act as a plasticizer in certain plastics, while it may be used as a solvent in certain cosmetics); (ii) consider application-specific functions of chemicals; (iii) include additional applications (currently, various data collection efforts for characterizing chemicals in consumer products have mainly focused on cosmetic products, while information for other consumer product types is only just emerging, such as for building materials and FCMs); and (iv) cover additional chemicals.

Data for supply chain emissions differ depending on the function or chemical structure (different synthesis steps required), location and company (different technical capacity and resources available). Initial approaches for deriving chemical-specific supply chain information were developed as part of the Environmental Genome of Industrial Products

Table 7. **Acting stakeholders, impact drivers, and available evaluation and management tools along the product life cycle of plastics containing chemicals** (Source: adapted from UNEP 2019a).

Life cycle stage	Acting stakeholders	Main chemical risk impact drivers and exposures	Evaluation tools	Management tools and entry points
Fossil or bio-based resource extraction and processing	Fossil resources/ bio-based resources companies; enforcement authorities	Worker exposures and industrial releases resulting in population-wide exposure to fossil-based resources; land use-related impacts and feedstock production impacts for bio-based feedstocks	Life cycle inventory of raw materials and feedstock production processes	Restrictions and prohibitions around certain resources and processes
Chemicals and plastics manufacturing	Chemicals and plastics manufacturers; enforcement authorities	Worker exposures and industrial releases resulting in population-wide exposure	Environmental Genome of Industrial Products (EGIP) (Overcash 2016)	Supply chain management
Plastic product design, manufacturing, assembling	Original equipment manufacturers; market surveillance enforcement authorities	Product design worker exposure and industrial releases	Design for environment worker exposure assessment (Kijko *et al*. 2016); Plastics Exposure Scenario Tool (PESTOOL) (Polymer Comply Europe 2021)	Supply chain management; ensure traceability of products and components
Trade, distribution, retail	Retailers, traders; enforcement authorities; testing organizations	Transportation	Disclosure of product composition	Retailer disclosure policy; labels; reuse and refill business model change
Purchase and use	Consumers; NGOs	Near-field exposures to chemicals in products; energy usage	Product Intake Fraction (PiF) modeling (Fantke *et al*. 2016)	
Recycling	Recyclers; enforcement authorities	Chemical residues and contamination of recycled material	Substance flow analysis; life cycle approach tools (e.g., LCA)	
Waste management	Municipalities; waste treatment facilities; enforcement authorities	Occupational exposure; waste treatment releases	End-of-life life cycle approach tools (e.g., LCA)	From waste management to resource management; recycling

(EGIP) initiative (Jiminez-Gonzalez and Overcash 2014; Overcash 2016). Yet, these efforts are currently limited to approximately 1,600 chemicals and are not yet fully operational for implementation in different substitution tools. Hence, further efforts are required to establish a comprehensive database of chemical supply chain information that can be adapted to varying capacities across regions and companies and can adequately handle proprietary or otherwise restricted data (UNEP 2019a).

Data and methods for estimating exposure to chemicals in the different plastic applications are currently emerging, such as the PiF approach (Jolliet *et al*. 2015; Fantke *et al*. 2016) for consumer exposure, which is also consistent with approaches for assessing occupational exposure (Kijko *et al*. 2016). However, while such methods are now able to address some specific plastic applications (e.g., FCMs and building materials), further research is needed to also cover the various other product categories relevant to plastics, such as plastic toys, electronics and medical products (Huang *et al*. 2017).

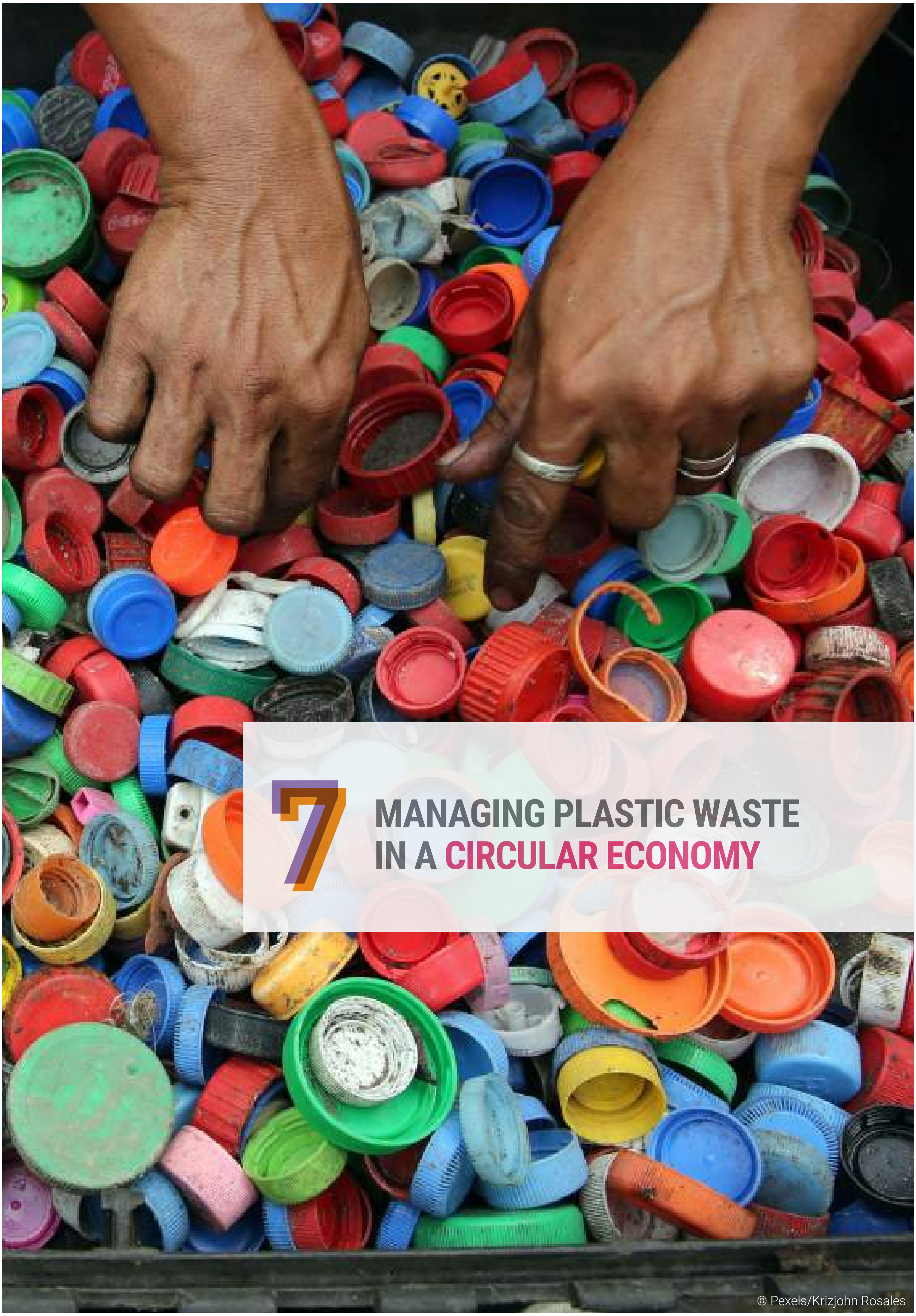
Finally, information related to human toxicity and ecotoxicity is currently available only for a relatively small fraction of chemicals in plastics and on a few toxicological endpoints. While some toxicological endpoints, such as genotoxicity, are more regularly assessed, others are rarely considered, such as endocrine-disrupting properties. Therefore, the toxicological assessment should be expanded to include additional endpoints relevant to human and environmental health. Approaches such as *in silico* (i.e., computational) could be further explored and advanced to derive reliable predictions whenever data gaps are apparent, using for example advanced statistical methods, such as machine learning (Fantke *et al*. 2021). In addition, methods for assessing exposure to the plastics themselves, including debris, microplastics and nanoplastics, are being developed and are urgently needed to complement data on exposure to plastic-associated chemicals and related risks. These future research advances will enable more complete and reliable assessments to inform science-based substitutions of chemicals in plastics and of plastic materials.

6.3.3 Responding to priority research needs

There are major knowledge gaps for occupational exposure along the life cycle of plastics (including their management as waste), particularly in developing countries and in countries with economies in transition. There is a need for research that considers and assesses the complexity of chemicals, using appropriate tools, including test methods for measuring toxic endpoints such as CMR and endocrine effects. Given the increase in plastic recycling and the move towards a (more) circular economy, it is important to assess and reduce exposures to harmful chemicals from recycled plastics, including for workers involved in waste management activities such as recycling, the general population and the environment at large.

Additives and NIAS in plastics are a complex field of research that requires, among others, cooperation between analytical chemists specialized in non-target analysis and toxicologists with expertise in mixture toxicity, as well as increased transparency along the chemicals and plastics value chains, in particular from companies. Understanding the effects of the large number of additives and NIAS in plastics, especially in recycled plastics, requires assessing the effects of chemical mixtures. The long-term effects of exposure through multiple pathways, including effects on the food chain, also need to be better assessed. This should include studies on low-dose effects and the exposure of vulnerable groups such as infants and children. Additionally, more research is needed on the releases of chemicals along the life cycle of the plastic, including during recycling.

While developed countries may have guidance on best technologies and practices, such as the EU BAT reference document for polymer production (European Commission 2007), the needs of developing countries and countries with economies in transition should also be assessed and taken into consideration. For instance, the guidance on BAT/BEP for the PBDEs listed under the Stockholm Convention stresses that technologies to separate BFR-POPs are available in developed countries, but that such approaches have not yet been described for developing countries (UNEP 2021c).



7

MANAGING PLASTIC WASTE IN A CIRCULAR ECONOMY

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7 / MANAGING PLASTIC WASTE IN A CIRCULAR ECONOMY

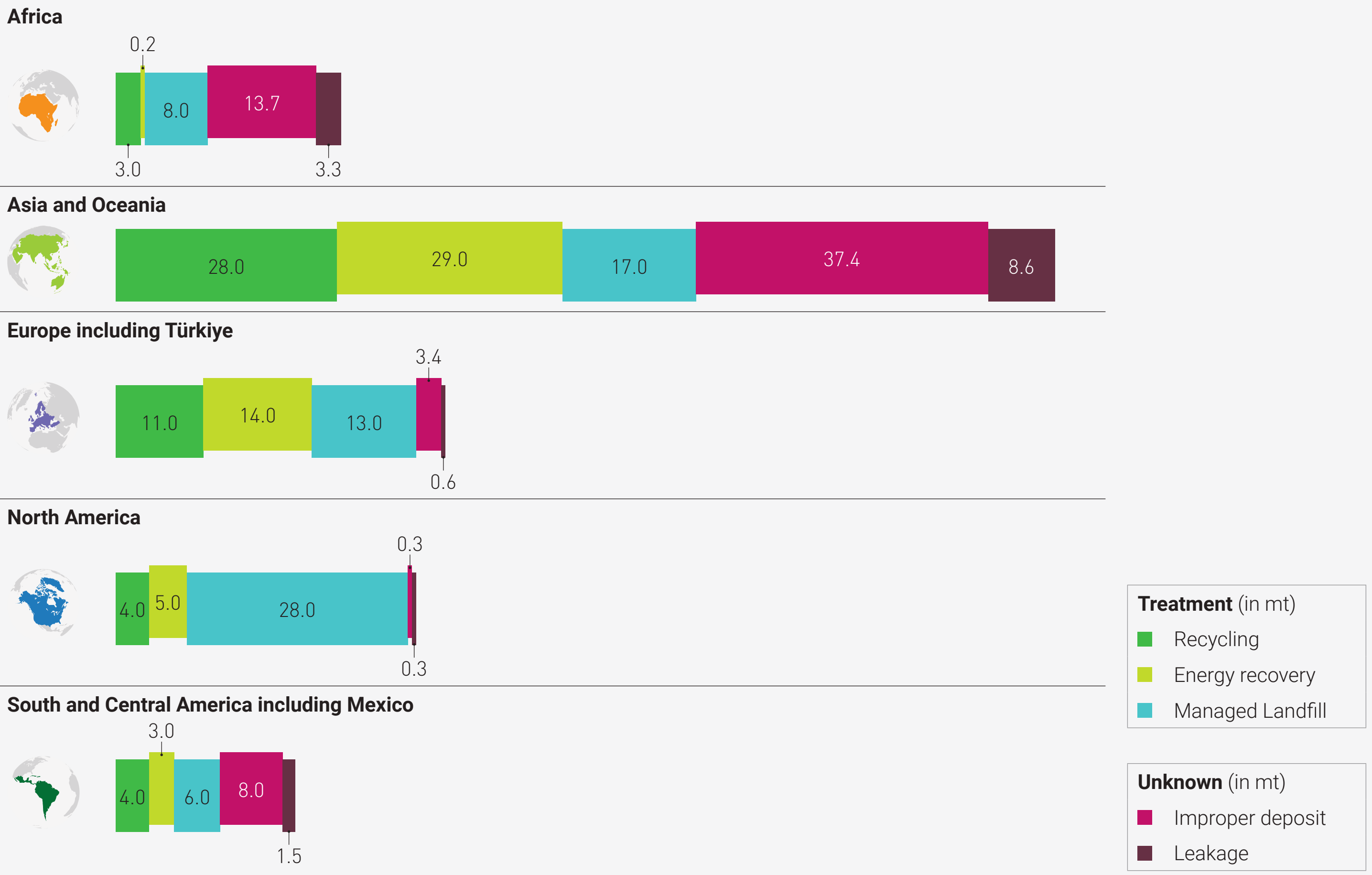
7.1 Global plastic flow and waste management

7.1.1 Plastic waste and related chemical flows

A study on global plastic waste flows documented that 56% of the generated plastic waste, along with its associated chemicals, is disposed to landfills or dumpsites or leaks directly into the environment, largely in developing countries (Figure 11; Market & Strategy GmbH Conversio 2019). This results in related releases of chemicals in leachates (Section 7.2.6).

Before the plastic waste amendment of the Basel Convention (Decision BC-14/12) became effective in January 2021 (UNEP 2021h), a large share of plastic waste was exported from OECD countries to developing countries (Brooks 2018; Greenpeace 2019; BIR 2021). The majority of such imported and domestically generated plastics in developing countries was improperly disposed of and leaked hazardous chemicals into the environment, due to a lack of waste management infrastructure (Figure 11). It was estimated that approx. 70% of all plastic waste exports were from OECD members, largely to low-income countries in East Asia and the Pacific (Brooks 2018; Geyer 2020). In these countries, only a fraction of the imported plastic waste is recycled. The majority is discarded into dumpsites or littered in the environment (Azoulay *et al*. 2019; Brooks 2018). Open burning of plastics in these countries is a major source of pollution (UNEP 2013b; Morales *et al*. 2018), releasing hazardous chemicals, including unintentional POPs (see below Sections 7.2.1, 7.2.5 and 7.2.6). Due to the Basel Convention's plastic waste amendment and restrictions on plastic waste import in many developing countries, plastic waste exports have been reduced, though they still take place to some extent, as documented for instance by the Basel Action Network (BAN 2022).

Figure 11. **Regional shares of globally generated plastic waste and their treatment** (Source: Conversio Market & Strategy GmbH 2019; the impact of the export of plastic from industrial countries to developing countries is not separately accounted for here). ▼



Of the 7 billion tonnes of plastic waste generated globally so far, less than 10% has been recycled (Geyer 2020), with a lower recycling quota in developing countries (Figure 11) and challenges to control hazardous chemicals in recycling (Section 7.2.2 and 7.2.3). An exception is China, with a recent recycling rate of approx. 30% (IPEN 2022; NDRC 2022).

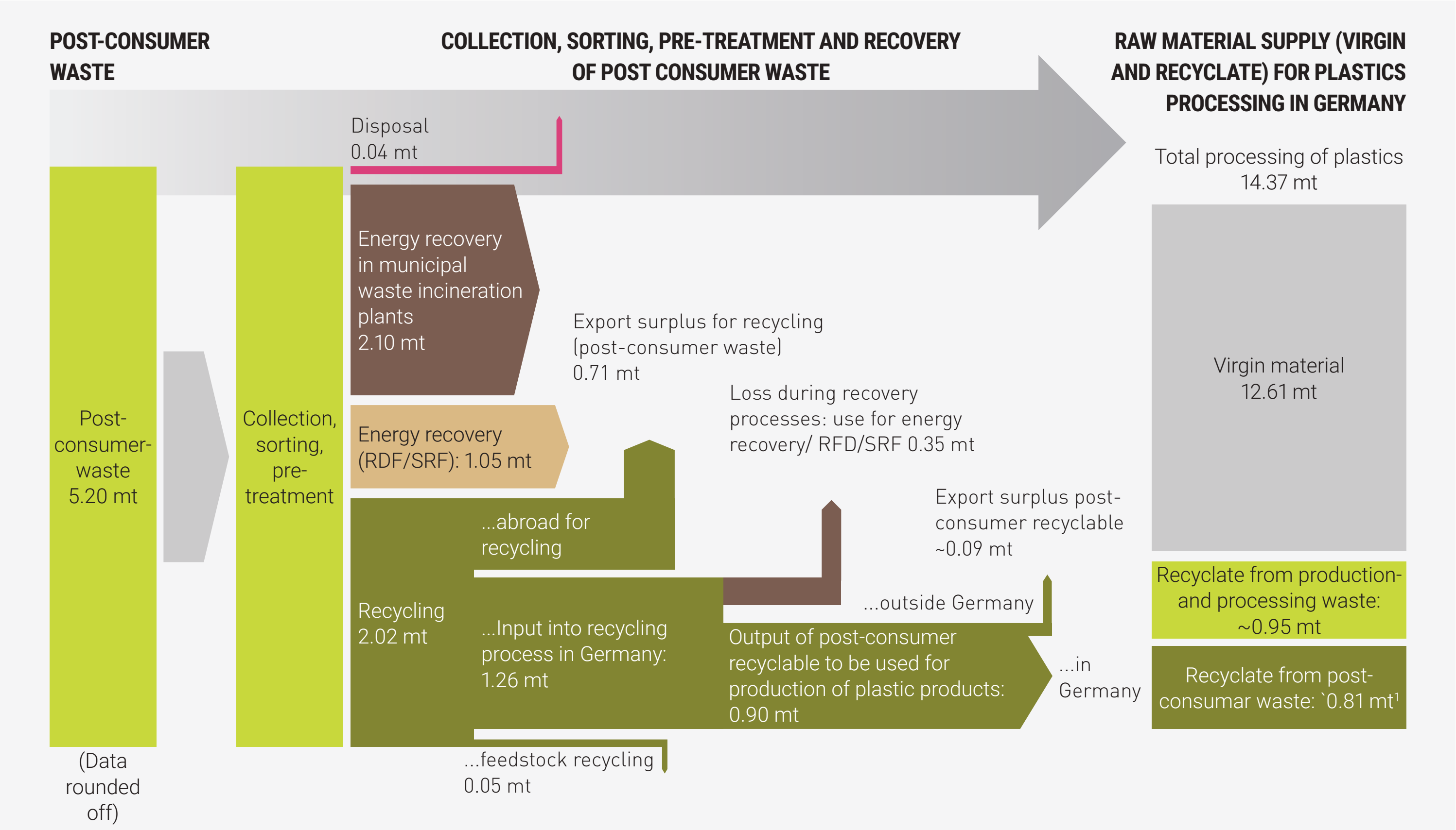
7.1.2 Plastic waste management in developed countries and drivers for exports

A plastic material flow analysis published for Germany revealed the most important issues with the current recycling practices in a developed country (Figure 12). The study found that 5.2 million tonnes/year of post-consumer plastics were collected, sorted and pre-treated in Germany. Of this, 2.1 million tonnes/year were used for energy recovery in municipal solid waste incinerators or cement kilns with associated destruction of hazardous chemicals, and another 1.05 million tonnes/year were transformed into refuse-derived fuel. Only 2.02 million tonnes (39%) were sent to material recycling, of which 0.71 million tonnes were exported for recycling. Only 1.26 million tonnes (24%) were treated in German plastic recycling plants producing 0.9 million tonnes of recycled plastics, while a fraction amounting to 0.35 million tonnes was again thermally recovered (Figure 12). The new plastic production (total 14.3 million tonnes/year) uses 5.6% post-consumer recycled polymers and 6.5% post-industrial recycled polymers. This study identified the following critical issues with current plastic recycling practices in a developed country (Conversio Market & Strategy GmbH 2018):

1. Not all collected plastics can be sorted into recyclable fractions, due to: (i) the presence of hazardous chemicals in plastics; (ii) insufficient sorting technologies; (iii) composite products that cannot be separated into single plastics streams; and (iv) more valuable waste fractions (e.g., metals) that are sorted out together with some plastics.

2. The low economic value of separated plastic fractions means a lower process cost for thermal treatment compared to material recycling. As a result, only easy-to-access plastics are used for mechanical recycling.

Figure 12. Flows of post-consumer plastic in Germany (Source: Conversio Market & Strategy GmbH 2018). ▼



1 Including ~0.135 mt recyclate at recyclers with own production

3. The quality of recycled plastics is comparably low, leading to: (i) unattractive low revenues; (ii) low shares of recycled plastics in new products (low impact on the CO_2 footprint of the products); and (iii) a bad reputation for plastic recycling and recycled plastics in general.

4. Additional additives are used in recyclates, for instance, to provide heat stabilization and resistance to weathering and aging (Glättli 2020; Rosato 2021).

5. The low attractiveness of plastic recycling and the high cost of thermal treatment favour export solutions, via both legal and illegal pathways depending on the concentration of hazardous compounds.

7.2 Challenges of recycling, energy recovery and managing plastics containing hazardous chemicals

7.2.1 Informal recycling in developing countries

It is important to recognize that in many developing countries the informal recycling economy is a fundamental part of plastic waste management (Babayemi *et al*. 2018b; 2019; Haarman 2016). However, little is known about the fate of plastic-associated chemicals and the resulting recyclates during these informal practices. This is particularly relevant for waste streams such as end-of-life vehicles, WEEE, and construction and demolition waste (Table 8).

Table 8. Overview of major mixed plastic waste streams, plastic types, and major waste management related issues, including hazardous additives.

Waste stream	Polymer type	Major issues related to waste management and hazardous additives
WEEE	PP	Often filled with talcum, which increases the density and hampers the separation from ABS and PS; blends with PE coexist in recycled PP fractions
	ABS	ABS from specific applications contains BFRs, some of which are listed as POPs (PBDEs; HBCDD, HBB) or may render the waste hazardous (TBBPA > 0.25%)
	PS/HIPS	PS/HIPS from specific applications contains BFRs, some of which are listed as POPs (PBDEs, HBCDD)
	PC/ABS	Often separated with a heavy fraction from bulk ABS, PS and PS fractions by density separation; needs to be separated from other polymers quite effectively, as contaminations higher than 1.5% may deteriorate the quality; the market segment for recycled PC/ABS has high quality standards
	PVC	May contain REACH-listed phthalates and POPs (SCCPs); traces of PVC in other target fractions deteriorate the quality, including thermal recovery
End-of-life vehicles	PP	Often filled with talcum or glass fibers, which increase the density and hamper the separation from ABS and PS; blends with PE coexist in recycled PP fractions
	ABS	Difficult to separate from PP, often used with metal plating
	PA	Often used together with glass or carbon fibers, not a standard collection stream
	PVC	May contain REACH-listed phthalates and Stockholm Convention-listed SCCPs; traces of PVC in other target fractions deteriorate the quality
	PUR	Not a standard collection stream, not recyclable by re-melting
	PET fibers	Not a standard collection stream; used as a composite with other fibers and coatings
	PC	Not a standard collection stream; used in multicomponent parts such as headlights, with or without metal plating
Construction	PE/PE-X	Not a standard collection stream; PE and cross-linked PE (PEX) are not separated; PEX cannot be re-melted
	PVC-soft	Used in flooring and cables; may contain REACH-listed phthalates and Stockholm Convention-listed SCCPs; traces of PVC in other fractions deteriorate the quality
	EPS/XPS	Contains HBCDD; low density hampers collection and incineration
	PVC-hard	Used in window frames and pipes; may contain lead and cadmium stabilizers
	PUR	Not a standard collection stream; may contain BFRs; not recyclable by re-melting

The elimination of hazardous compounds from plastic wastes is further complicated by the fact that the informal sector (even more so than the formal sector) does not have access to information on the hazardous chemicals that may be found in the plastic waste, nor is it equipped with tools such as elaborate density separators or XRF to identify plastics containing hazardous compoundse (Haarman 2016). Consequently, informal sector workers might become exposed to hazardous compounds when moulding or pelletizing the plastic, and hazardous chemicals such as PBDEs, HBCDD, and SCCPs become diluted in the recycled plastics (Schlummer 2014). The use of such plastics in the production of new products results in uncontrolled dissemination of and exposure to these contaminants (Chen *et al*. 2009; Samsonek and Puype 2013; Ionas *et al*. 2014; Puype *et al*. 2015; Kuang *et al*. 2018). The informal sector also frequently practices open burning of non-recyclable plastics, recovery of metals, or incineration in non-BAT facilities, which leads to severe pollution and human exposure, including food chain contamination (Wong *et al*. 2007; Tue *et al*. 2013; Labunska *et al*. 2014; Zeng *et al*. 2018; Petrlik *et al*. 2019a: Petrlik *et al*. 2019b; Petrlik *et al*. 2022). While this issue has improved in China due to the restriction on plastic waste and e-waste imports (IPEN 2022), it has worsened in other countries (Petrlik *et al*. 2019a; Petrlik et al. 2019b; Gündogdu and Walker 2021; Petrlik *et al*. 2022).

7.2.2 Challenges of mechanical plastic recycling

The challenges of mechanical recycling of post-consumer plastic waste are diverse and differ for specific waste streams (e.g., Buekens and Yang 2014; Buekens and Zhou 2014; Ragaert *et al* 2017). Importantly, one must distinguish between thermoplastics and thermosets. Whereas thermoplastics can be re-melted and lend themselves to recycling processes (Grigor 2017), thermosets are not suitable for such processes. Instead, they might be mixed in limited amounts as fillers in new thermosets or chemically recycled (see Section 7.2.4).

Thermoplastics, however, are produced by polycondensation (e.g., PET, PA), polyaddition (e.g., thermoplastic PUR) or polymerization (e.g., PE, PP, PS, ABS). Products of polyaddition are susceptible to hydrolysis but can re-condensate. Products of polymerization may suffer from thermo-oxidative processes that lower their average chain length. After their useful lifetime and recycling processing, only polycondensates can usually reach the quality of virgin plastics. The products of polymerization age with each thermo-oxidative attack and will degrade

Figure 13. **Different grades of recycled polyethylene (left and middle) compared to virgin polyethylene** (photo by Roland Weber). ▼

step by step. As a result, the quality of recycled plastics rarely reaches that of virgin plastics, especially in terms of physical and mechanical properties (Brouwer *et al*. 2018). High-quality recycled polymers can resemble virgin plastic in terms of their properties. However, post-consumer recycled polymers generally have color issues (Figure 13) due to the presence of colourful plastics in the recycling feed (Hahladakis and Iacovidou 2019).

WEEE, end-of-life vehicles and demolition waste, which use approximately 35% of all plastics, can contain a range of hazardous legacy additives and have issues regarding the presence of multiple different polymer resins and collection and sorting fractions. Some plastics in these waste streams contain regulated hazardous additives (e.g., POPs) and other potentially harmful compounds (Table 8). These hamper recycling significantly, resulting in only minor amounts of plastics from these sources being subject to primary or secondary material recycling (Buekens 2014a; Buekens 2014b). Table 8 lists the most abundant polymer types in these three waste streams and their most prominent waste management related issues, including problematic additives.

If plastics from such waste streams are considered for recycling, they require costly manual dismantling operations or automated bulk treatment. Such waste stream-specific strategies to separate plastics from other wastes consist mainly of shredding and so-called post-shredder-technologies (PST). Sorting lines sort only one plastic per step; hence, sorting plants operate with a series of such lines. However, due to economic reasons, they focus on a limited number of plastic types (Arends *et al*. 2015).

Another important challenge to recycling plastics from WEEE, end-of-life vehicles, and construction waste is the lack of quality standards that may qualify recycled polymers for specific secondary uses. In developing countries standards are often lacking even for sensitive uses such as toys or FCMs.

Currently, infrastructure is most commonly available for PET recycling, which can reach high collection and recycling rates using deposit schemes. PET has also been used in bottle-to-bottle recycling (reuse) approaches, which significantly reduces the chance of introducing hazardous chemicals as unwanted contaminants during the sorting and recycling process. Such schemes are an example of primary recycling of PET for food packaging, and food packaging is a use type with very high chemical exposure potential for consumers (Welle 2016). A recent study identified a range of chemicals in recycled PET not authorised for instance in the EU, but mostly below regulatory limits (Gerassimidou *et al*. 2022).

7.2.3 Chemicals of concern in recycled plastics

Most industry efforts have focused only on addressing waste or increasing the use of recycled content, with the intention of keeping materials in the value chain longer (Werner *et al*. 2018). However, the wide range of hazardous chemicals found in currently used products (see Chapter 2.3; Groh *et al*. 2019) can contaminate recycled plastics. Monitoring plastic consumer products has demonstrated that hazardous additives may be recycled into new products, including in sensitive sectors such as toys and FCMs (Chen *et al*. 2009; Samsonek and Puype 2013; Ionas *et al*. 2014; Puype *et al*. 2015; Guzzonato *et al*. 2017; Kuang *et al*. 2018). A study in Australia found that a high proportion (40%; n=1714) of plastic consumer products contained PBDEs and other BFRs originating from recycling (Gallen *et al*. 2014).

Some of the hazardous chemicals in plastics or other polymers being recycled are already banned or restricted at the national level, or even at the global level by the Stockholm Convention (e.g., HBCDD, HBB, PBDEs, PFOS, PFOA, SCCPs). Others are in the process of assessment for potential listing and banning (e.g., dechlorane plus, UV-328, MCCP). Setting limits for these legacy pollutants in wastes under the Basel or Stockholm conventions or in product specifications might reduce the recycling of plastics, such as plastic fractions from e-waste, while reducing the spread of these POPs. Recycling becomes difficult, or even impossible, if the smaller provisional low POPs limit of 50 mg/kg is selected for PBDEs[13] (Straat and Nilsson 2018). This highlights the risks POPs pose to the circular economy and the need for the swift phase-out of all such chemicals. Cadmium and lead, which were used as stabilizers in rigid PVC, also hinder recycling due to contamination of recycled products (Friege *et al*. 2018). DEHP, which is listed as an SVHC under REACH, is a major legacy additive in PVC; hence, the recycling of DEHP-containing PVC reintroduces this hazardous additive into new products (Pivnenko *et al*. 2016). Similarly, PFOS and PFOA, which are listed under the Stockholm Convention, as well as other PFASs such

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[13] The provisional low POP content for PBDEs in the Basel Convention is 50 or 1,000 mg/kg (UNEP 2019d).

as side-chain fluorinated polymers in carpets and synthetic textiles, might also be transferred into new products through recycling (ECHA 2015a; Vallette *et al*. 2017). Hazardous chemicals from recycled rubber/tires such as PAHs, SCCPs, phthalates, adipates, vulcanization additives, and antioxidants have been detected in recycled tire crumbs used on children's playgrounds and recreational sports surfaces (Celeiro *et al*. 2014; ANSES 2018; Brandsma *et al*. 2019).

It is currently not feasible to obtain full information about the composition of mixed plastic waste, such as from EEE or end-of-life vehicles, and it would be impractical to attempt to reverse engineer the large number of plastic parts in vehicles or EEE to identify all the chemical constituents (Werner *et al*. 2018). It is difficult to assess the toxicological impacts of the output of recycling products with complex multiple plastic types and chemicals, such as EEE or vehicles (UNEP 2017a). Hence, in the absence of measures to address these issues, inadvertently, humans and the environment may increasingly be exposed to hazardous chemicals through recycled products and materials. To address this challenge, the EU has developed the SCIP database for information on ***S****ubstances of* ***C****oncern* ***I****n articles as such or in complex objects (****P****roducts)* (see Section 7.3.1.3).

Another challenge is that, in a circular economy, plastics undergo recycling multiple times and polymer chains become shorter with each cycle (Ragaert *et al*. 2017). Plastics and additives also undergo degradation and oxidation reactions over time, which lower their performance (e.g., Azeez 2019). Hence, high-quality recycled polymers require re-additivation and re-stabilization. However, degraded additives remain in the polymer matrix and may be released during secondary use. The toxicity and associated risks of such NIAS are in general not well-known (see Chapter 2.3.10) (Horodytska *et al*. 2019).

While plastic with BFRs can be separated (UNEP 2021c), there are other hazardous compounds such as SCCPs or PAHs that might occur in waste plastic streams and are not separated in recycling processes due to the lack of separating technologies or knowledge about their presence. Black plastics make up a large proportion of WEEE and end-of-life vehicles (Arends 2014; Turner 2018a; Turner 2018b), containing high amounts of carbon black. Consequently, PAHs, well-known contaminants in carbon black, may occur in such plastic waste and recycled plastics.

7.2.4 Challenges of chemical recycling of plastics

Mechanically recovered thermosets from waste streams cannot be recycled like thermoplastics and thus cannot replace virgin materials. However, for thermosets and duroplastics such as PUR or PEX chemical recycling might be a valuable option to recover high-quality plastic/polymers (Pickering 2006; Vollmer *et al*. 2020). Chemical recycling might also be a treatment option for other plastics that cannot be recycled for other reasons, such as multilayer plastic or complex or dirty plastic mixtures.

Chemical recycling aims for depolymerization and targets the recovery of monomers and other degradation products to be used in the production of new polymers, feedstock, or fuel[14] (Ragaert *et al*. 2017; Quicker *et al*. 2022). However, the chemical recycling of plastics results in the generation of toxic VOCs such as monomers (e.g., styrene, isocyanates, vinyl chloride), benzene, toluene, PAHs, formaldehyde, hydrogen cyanide and high-temperature tars (Williams and Williams 1999; Scheirs and Karminsky 2006; Rollinson and Oladejo 2020). The release of these compounds needs control and regulatory limits to minimize human exposure and environmental contamination. The fates of the additives, (hazardous) monomers (e.g., styrene) and degradation products entering chemical recycling routes need to be monitored and their releases and potential exposures understood. The pyrolysis of plastic from WEEE or end-of-life vehicles containing halogenated flame retardants and PVC can result in oils highly contaminated with PCDD/PCDFs (Rosemann *et al*. 1998; Weber and Sakurai 2001; Zhang *et al*. 2015; Mei *et al*. 2017). The degradation of PBDEs and other BFRs can generate PBDD/PBDFs (Weber and Kuch 2003; Ortuno *et al*. 2014). The co-generated acidic gases (HCl, HBr, Br_2) can also cause corrosion (Bockhorn *et al*. 1999; Rademakers *et al*. 2002; Fekhar *et al*. 2019) and impact the quality of the fuel product.[33] Furthermore, the pyrolysis and gasification of plastic fractions generate high PAH and BTEX levels (Zhou *et al*. 2015; Gaurh and Pramanik 2018). The pyrolysis of fluoropolymers or side-chain fluorinated polymers can result in the formation and release of fluorinated POPs (e.g., PFOA), other PFASs, ozone-depleting substances and greenhouse gases (Arito and Soda 1977; Ellis *et al*. 2001; Sinclair *et al*. 2007; Ochi *et al*. 2008).

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[14] Please note that plastic to fuel is not considered recycling by ISO Standard definition (ISO 15270:2008) or EU regulation (European Parliament 2018).

Other general challenges relate to the input streams, which require pre-sorting and controlling certain elements such as nitrogen or chlorine, which hamper or limit chemical recycling. A range of pyrolysis plants was discontinued due to a lack of suitable waste streams, difficult logistics, hazardous emissions, and poor commercial viability (Gleis 2011; Vollmer *et al*. 2020). In recent years, a wide range of chemical recycling plants have started operation at pilot or full-scale (Vollmer *et al*. 2020; Closed Loop Partners 2021; Maisels *et al*. 2021), and similar challenges are reported for several chemical recycling plants today (Patel *et al*. 2020; Brock *et al* 2021b). Also, hazardous waste can be generated in high amounts from chemical recycling plants (National Development and Reform Commission of China [NRDC] 2022).

7.2.5 Energy recovery

The recovery of energy by thermal waste treatment is a common approach to the end-of-life treatment of plastics in developed countries (Hopewell *et al*. 2009; PlasticsEurope 2019; UNEP 2021h, UNEP 2021i). However, developing countries have frequent difficulties with waste incinerators due to, for instance, poor waste quality, inadequate waste segregation, lack of appropriate policy and financing, as well as technological and monitoring challenges (ISWA 2013; Nixon *et al*. 2017). Halogens emitted from plastic waste containing halogenated additives or PVC can cause corrosion in incinerators and other thermal facilities (Rademakers *et al*. 2002). Incineration emits higher levels of GHGs compared to recycling (Schwarz *et al*. 2021; Tenhunen-Lunkka *et al*. 2022).

The energy recovery from plastic waste in facilities with limited air pollution control results in the emission of PCDD/PCDFs, other unintentional POPs, heavy metals, particulate matter and other air pollutants (US National Research Council 2000; Ikeguchi and Tanaka 2000; Weber and Kuch 2003, Zhang *et al*. 2015). POPs releases contaminate the surrounding soil over time. In Indonesia, for instance, releases from energy recovery of plastics in boilers were found to have contaminated chicken eggs with POPs (unintentional POPs, SCCPs, PBDEs, PCBs), including with PCDD/PCDF levels as high as 90 times above regulatory limits (Petrlik *et al*. 2019b; Petrlik *et al*. 2022). Improved filter technology can reduce PCDD/PCDFs and heavy metal emissions to air. But these technologies transfer the pollutants to residues such as fly ash. Ashes are often not adequately managed in developing countries but are dispersed or dumped into the environment (Petrlik and Bell 2020), contaminating the food chain with PCDD/PCDFs (Weber *et al*. 2018c; Petrlik *et al*. 2022).

Additionally, combustion at low temperatures in non-BAT incinerators or open burning of fluorinated polymers can result in the unintentional formation and release of fluorinated POPs (e.g., PFOA), other PFASs, other potentially toxic substances, ozone-depleting substances, and greenhouse gases (Arito and Soda 1977; Ellis *et al*. 2001; Sinclair *et al*. 2007; Ochi *et al*. 2008).

Plastic waste is also used as a substitute for primary fuel in cement kilns, which can destroy POPs. In many developing countries, cement kilns are currently the only facilities capable of destroying POPs. However, chlorine and bromine must be present at low levels in the waste (normally <0.5%). Otherwise, they may accumulate in cement kiln systems, causing operation problems (Buekens and Cen 2010; Waltisberg and Weber 2020) and potentially leading to blockages and unplanned maintenance stops.

7.2.6 Disposal to non-engineered landfills and dumpsites

In many developing countries, the largest proportion of plastic and other wastes ends up in dump sites due to a lack of sanitary landfills and other treatment methods (Figure 11; Hoornweg and Bhada-Tata 2012; ISWA 2016; Babayemi *et al*. 2019). Plastic is the major “fuel” of fires in dump sites and landfills, releasing particulates, PAHs, PCDD/PCDFs and other unintentional POPs (UNEP 2013b; Morales *et al*. 2018). Persistent plastic additives such as PBDEs or heavy metals are released during open burning, together with degradation products such as brominated PBDD/PBDFs (Weber and Kuch 2003; Gullett *et al*. 2010; Hahladakis *et al*. 2018). PVC as well as brominated and chlorinated flame retardant-containing plastics are major contributors to the release of PCDD/PCDFs and PBDD/PBDFs (Ikeguchi and Tanaka 1999; Shaw *et al*. 2010; Gullett *et al*. 2010) and the associated environmental contamination and human exposure (Tue *et al*. 2013; Weber *et al*. 2018c). Eggs and milk around dump sites and recycling sites where plastics were burnt in developing countries contain high levels of POPs such as PBDEs and SCCPs (Labunska *et al*. 2014; Zeng *et al*. 2018; Adu-Kumi *et al*. 2019; Oloruntoba *et al*. 2019, Oloruntoba *et al*. 2021).

Plastic additives such as BPA, DEHP, PBDEs, TBBPA and a wide range of PFASs, OPFRs and SCCPs are released via leachates from landfills and dumps into the hydrosphere, contaminating the groundwater, soil and surrounding environment (Yamamoto *et al*. 2001; Asakura *et al*. 2004; Osako *et al*. 2004; Weber *et al*. 2011, Kalmykova *et al*. 2013; Li *et al*. 2021; Gomi *et al*. 2022). In particular, BPA and PFASs (e.g., PFOS, PFHxS and PFOA) have been detected at high concentrations in closed landfills even 50 years after closure (Hepburn *et al*. 2019; Propp *et al*. 2021). US EPA scientists calculated a half-life of 33 to 112 years for the degradation of side-chain fluorinated polymers, with associated releases of non-polymeric PFASs such as PFOA (Washington *et al*. 2015). Additionally, the debromination of PBDEs was observed in landfills in tropical climates (Kwan *et al*. 2013), resulting in increased toxicity and mobility (Shaw *et al*. 2010; Weber *et al*. 2011).

High levels of POP plastic additives such as PBDEs, SCCPs and HBCDD have been detected in chicken eggs, milk and vegetables produced around dump sites in developing countries (Oloruntoba *et al*. 2019; Oloruntoba *et al*. 2021; Petrlik *et al*. 2019a; Petrlik *et al*. 2022). One study found that waste pickers on a dump site had high PBDE levels in their blood (Athanasiadou *et al*. 2008). Another study found that humans around dump sites in some developing countries had increased dioxin levels in their breast milk (Kunisue *et al*. 2004). However, this type of contamination and resulting human exposure is not unique to developing countries. For instance, PBDE levels in human blood around landfill sites in California were elevated up to 10 km away from landfills, indicating exposure (Liu *et al*. 2016). While releases of PBDEs, PFOA and PCDD/PCDFs from landfills and associated human exposures have been documented, such assessments are missing for most plastic additives. This highlights the need of engineered landfills that meet the Basel Convention technical guideline (UNEP 2021j).

7.3 Options for recycling and managing plastics containing hazardous chemicals

The different recycling processes need to ensure that hazardous chemicals are separated and destroyed instead of recycled and diluted into new products that can lead to exposure. A plastic waste management hierarchy framework can be developed based on the different approaches of physical recycling, chemical recycling and energy recovery (including plastic-to-fuel) (Panda *et al*. 2019; UNEP 2021h). Allocation of resources and prioritization of actions should be in alignment with the waste hierarchy priorities in a stepwise manner (Section 5.1), i.e., emphasis should be placed first on prevention and reuse, followed by physical/mechanical recycling, chemical recycling and energy recovery. The non-contaminated thermoplastics fractions can be selected for material recycling, while the contaminated plastics can be used to some extent for chemical recycling and for energy recovery, which can destroy hazardous chemicals but also generates unintentional POPs and hazardous residues in many cases. This requires, however, a robust waste management scheme, a regulatory framework and enforcement, suitable sorting and recycling processes, and destruction capacity for plastics with hazardous chemicals. Due to the lack of destruction capacity in many developing countries, this chapter also includes a section on landfilling – the least preferred option in the waste hierarchy – and its challenges related to chemicals in plastics.

7.3.1 Physical/mechanical recycling of plastics considering hazardous chemicals

7.3.1.1 Mechanical-based recycling to separate hazardous chemicals

Closed-loop recycling is defined as recycling into the same or similar products, whereas open-loop recycling produces products with less demanding specifications (UNEP 2021h). Mechanical recycling processes do not intentionally degrade the polymer chain. They recover a polymeric product through a series of mechanical/physical processes that include spectroscopic or density-based sorting, washing, drying, shredding, melting, and melt filtration (details are described in the Basel Convention technical guideline (UNEP 2021h)). Such process cascades can recycle LDPE, HDPE and PP from packaging (Ragaert *et al*. 2017), or PP and ABS from WEEE, while phasing out legacy additives (Schlummer 2014; UNEP 2021c).

Mechanical plastic recycling has sector-specific characteristics. As an example, we elaborate here on end-of-life plastics from WEEE and packaging. WEEE plastics are part of complex EEE items and need to be separated from other materials such as glass or metals. Due to economic restrictions in developed countries, this is usually done by post-shredder procedures, which create a shred of mixed plastics that requires subsequent sorting. Sorting technologies have to cope with small particle sizes >50 mm, lacquered plastics, and black plastics. A significant proportion of ABS and PS plastics in WEEE has been treated with BFRs, which are partly restricted (as POPs and by RoHS), or TBBPA, which is regulated by product legislation (Schlummer *et al*. 2007; European Commission 2008a; Tsydenova and Bengtsson 2011). Thus, these plastics must be sorted effectively. Legacy additives such as certain BFRs or phthalates can only be separated by specific technologies, such as density separation or spectroscopic sorting, as described for plastics from WEEE (UNEP 2021c). Spectroscopic sorting (laser spectroscopy) and density separation have proven to produce ABS, PP and PS recyclates with acceptable mechanical properties and BFR levels that comply with the RoHS directive and are below a low POP content of 1,000 or 500 mg/kg (Arends 2014; Arends *et al*. 2015, Dewulf *et al*. 2019). The separated fraction with high BFR content can either be thermally recovered, which destroys the POPs but requires that unintentional POP formation meet emission standards (see Section 7.3.3; Weber and Kuch 2003), or subjected to innovative solvent-based recycling technologies (Schlummer 2014; see Section 7.3.1.2).

The recycling of plastic packaging waste also exemplifies the challenges of sorting monoplastics such as PE, PP, PS and PET from the bulk of the packaging. Labels on the plastic hinder identification. Black plastics and composite packaging materials hamper sorting. These plastic wastes have resulted in large quantities of mixed plastic streams being used for energy recovery or refuse-derived fuels (see Section 7.3.3) rather than for materials recycling. Even the sorted fraction of monoplastics poses significant challenges, as near-infrared (NIR) sorting reaches purities of roughly 90% only. Color pigments and odours are contaminants that reduce the quality and the market acceptability of recycled plastics from packaging (Cimpan *et al*. 2016).

Overall, current mechanical recycling processes are limited by high costs, degradation of mechanical properties and inconsistent quality products (Schyns and Shaver 2021). Additionally, Geueke *et al*. (2018) reviewed the chemical safety of recycled packaging products and found that the products made from bulk plastic waste packaging sources failed to comply with strict regulations for the use of recycled plastics in new food packaging (Geueke *et al*. 2018). Thus far, EFSA has only approved PET from controlled bottle-to-bottle recycling for reuse in food packaging, to control the presence of hazardous chemicals (EFSA 2022). Recently, the European Commission prepared a fast-track approval of 140 recycling processes for use with food and drink packaging (Simon 2018). This was criticized by NGOs warning that post-consumer plastics may contain toxic chemicals and pose risks to human health (Simon 2018), as documented recently (Gerassimidou *et al*. 2022). The use of recycled plastics in sensitive products such as food packaging and toys needs to be strictly regulated by laws (Geueke *et al*. 2018) to avoid the presence of legacy additives (Chen *et al*. 2009; Samsonek and Puype 2013; Ionas *et al*. 2014; Kuang *et al*. 2018; Li *et al*. 2019).

7.3.1.2 Solvent-based recycling to separate hazardous chemicals

A specific type of physical recycling includes so-called solvent-based plastic recycling processes that apply polymer-specific solvents to dissolve a target polymer out of a plastic mixture or from composites. After dissolution, these processes separate the undissolved matter, clean additional contaminants from the polymer solution and finally produce granules of recycled polymers while recovering the solvent within the process. Examples of this technology are the CreaSolv® Process used for a multitude of polymers (Schlummer *et al*. 2006; Schlummer *et al*. 2012; Schlummer *et al*. 2017, Tange *et al*. 2016; Knappich *et al*. 2017,), the Vinyloop process for PVC,[15] the Newcycling® Process for PE and PA (APK AG n.d.), and the PureCycle process for PP (Purecycle n.d.). Also, the delamination of multilayer plastic can be achieved with solvent-based recycling (Ügdüler *et al*. 2021).

The CreaSolv® process reportedly can separate legacy additives such as BFRs (Schlummer *et al*. 2016; Schlummer *et al*. 2017). The first full-scale plant for the recovery of polystyrene and bromine by separation of HBCDD from EPS/XPS has started operation in July 2021 (Tange 2021). A current research project ("NONTOX") deals with plastics from WEEE and combines the separation of BFRs and recycling of WEEE plastics by CreaSolv® with pyrolysis-based degradation of BFRs to recover the bromine[16] (European Commission 2019b). Another current European project (CircularFlooring) addresses the recycling of plasticized PVC, including the separation and safe valorisation of phthalates such as DEHP, DiBP, DBP and BBP (Bavarian Research Alliance 2019). The separated restricted DEHP can be catalytically hydrogenated to DINCH and thus recycled into an alternative plasticizer that does not exert reproductive

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[15] Information on the process may be found at: https://en.wikipedia.org/wiki/VinyLoop Accessed 26 April 2023.

[16] Residues containing high levels of bromine might also be treated by non-combustion technologies such as ball milling (Cagnetta et al. 2016).

toxicity, genotoxicity or carcinogenicity, and is therefore not restricted (Windels *et al*. 2021).

7.3.1.3 Information about chemicals of concern for safeguarding ESM of plastic waste

Besides the objectives of protecting human health and the environment and supporting a toxic-free circular economy, information-sharing about chemicals of concern in plastic is needed all along the supply chain, until the waste management sector, for safeguarding physical recycling processes and overall ESM. The EU has for instance developed the SCIP database under the Waste Framework Directive to support the circular economy and to make available to waste recycling companies information about hazardous substances at the end of their life cycle.

7.3.1.4 Traceability of plastic waste and recyclates for safeguarding recycling

Good traceability of waste and recyclates is one of the most powerful tools to avoid trafficking and dilution of contaminated plastics with non-contaminated plastics. A traceability system specifically for the recyclates (from the recycling facilities to the end users) is necessary due to restrictions or bans for certain uses. Traceability schemes would help improve the quality of closed-loop and open-loop recycling.

7.3.1.5 Standards and certifications for recycled plastics to safeguard and encourage recycling

Regulations to ensure the purity of plastics used in sensitive products such as food packaging, cooking items, drinking water supply or toys are required globally to minimize human exposure to hazardous chemicals. Standards, and possibly certificates, for plastic recycling that include restrictions on hazardous substances are needed in developed and developing countries. The standards may allocate recycled material to either high or low-level downcycling opportunities, depending on the mechanical and chemical properties, including the levels of contaminants and other hazardous compounds. If the material quality is sufficiently high, this can facilitate close loop applications in EEE or the automotive sector, for instance. Such an approach could help prevent plastics originating from waste sectors containing hazardous chemicals from entering sensitive applications in toys or FCMs. Instead, recycled plastics are used in the same product categories (e.g., vehicles or EEE), either as 100% material or, most often, as 5-50% recycled plastics content in new compounds.

For instance, the EU establishes standards (CEN standards) for recycled plastics, which define methods of specifying delivery conditions for recyclates and give the most important characteristics and associated test methods for recyclates intended for use in semi-finished/finished products. EU standards exist for recycled PVC, PS, PE, PP and PET. There are also EU standards for the preparation of samples of recycled plastics prior to testing and for a system of sampling procedures for testing plastic waste and recyclates; these standards take into account the specifics of the plastic waste and recyclates (Stenmarck *et al*. 2017). General standards, such as for food-grade materials, apply to both virgin and recycled polymers. Recycled plastics might also be certified. The EuCertPlast is an EU-wide certification for post-consumer plastics recyclers, which aims at increasing plastics recycling rates, promoting innovative applications for plastics recyclate, enhancing plastics industry transparency, and meeting REACH requirements and food contact compliance for recyclers. The certification is based on the European Standard EN 15343:2007 (Stenmarck *et al*. 2017). The European Committee for Electrotechnical Standardization (CENELEC)[17] creates voluntary and harmonized standards intended to be technologically neutral and developed thanks to the contributions of sector-specific experts. CENELEC has developed a standard for the treatment of WEEE (EN 50625-1 Collection, Logistics & Treatment Requirements for WEEE – Part 1: General Treatment Requirements.). This CENELEC standard sets a requirement of a maximum of 2,000 ppm bromine in plastics sent for recycling, to control banned brominated additives such as PBDEs, PBBs and HBCD. This requirement also controls other known hazardous BFRs such as TBBPA and other BFRs that have only now started being assessed for toxicity (Hennebert 2021). Implementing this standard requires investments in sorting technology. Several screening techniques might have to be used to separate materials that contain high levels of bromine (Stenmarck *et al*. 2017; UNEP 2021c).

7.3.2 Chemical recycling of plastics

Chemical or feedstock recycling is a general term used to describe technologies where post-consumer plastic waste is converted into chemicals to be used

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[17] Information on CENELEC may be found at: https://www.cenelec.eu/ Accessed 26 April 2023.

as feedstock by the chemical industry or, in some countries, as fuel.[18] These technologies include pyrolysis, gasification, chemical depolymerization, catalytic cracking and reforming, and hydrogenation (Maisels *et al*. 2021; Rageaert *et al* 2017; Vollmer *et al*. 2020). While some stakeholders classify these processes as recycling, others argue they should be considered thermal recovery (Goldsberry 2021). The production of fuel is not considered chemical recycling according to ISO Standard definition (ISO 15270:2008) or EU regulation (European Parliament 2018).

Chemical recycling technologies currently contribute less than 0.5% of the waste plastic recycling, with projections of considerable growth (Maisels *et al*. 2021). Chemical recycling is indeed estimated to address 6% of plastic waste in 2040. Another study estimates that chemical recycling could grow to account for 4 to 8% of the total global polymer supply by 2030 (McKinsey & Company 2022). However, the technology has high energy requirements, with greenhouse gas emissions 110% higher than mechanical recycling (The Pew Charitable Trusts and SYSTEMIQ 2020). A holistic life cycle approach including the use of robust LCAs is needed for the selection of technologies, including robust footprints for the individual technologies.

As described in Section 7.2.4, halogenated plasticizers, flame retardants and halogenated polymers can negatively impact chemical recycling processes by corrosion and formation of unintentional POPs (Weber and Sakurai 2001; Weber and Kuch 2003; Mei *et al*. 2017). The levels of halogens in plastic production should be reduced to facilitate chemical recycling, combined with the use of process products in refineries (e.g., OMV 2021, OMV 2022). Chemical recycling can destroy to some extent hazardous organic additives and remove some of the halogens present in PVC and plastic containing chlorinated or brominated additives with BAT, as demonstrated at the pilot scale (Hornung and Seifert 2006).

Technologies for chemical recycling are already available. Several pilot and full-scale plants in developed countries (Europe, US, Japan) and emerging economies such as China and India have started operations or are researching new processes to recycle plastic waste (Damodharan *et al*. 2019; Kapur-Bakshi *et al*. 2021; Maisels *et al*. 2021). Several of these processes were already operated or researched in the 1990s but were stopped due to a lack of suitable plastic waste for input, insufficient logistics, and lack of commercial viability (Vollmer *et al*. 2020). Recent assessments have shown that many plants do not function properly (Rollinson and Oladejo 2020; Brock *et al*. 2021b) and that major industries responsible for the generation of plastic waste are cooperating with the cement industry for energy recovery (Brock *et al*. 2021a; see Section 7.3.3). A holistic life cycle approach with the application of robust, independent and transparent LCAs, including the environmental and climate impacts of chemical recycling based on primary data sources, is needed before developing further legislative frameworks that incentivize chemical recycling technologies (ZeroWaste Europe 2020), bearing in mind the intrinsic limitation of LCA methods.

All toxic releases from these processes, such as toxic monomers (e.g., styrene, isocyanates), BTEX, PAHs and high-temperature tars (Williams and Williams 1999; Scheirs and Karminsky 2006; Rollinson and Oladejo 2020), as well as greenhouse gases, need to be assessed and their emissions controlled and included in permits and operation licenses. Stringent emission limits for relevant pollutants and emission monitoring are needed to ensure that relevant emission parameters remain below regulatory limits (Earthjustice 2021; Good Company 2021). Also, the hazardous waste generated needs to be managed in an environmentally sound manner (NRDC 2022).

7.3.2.1 Thermochemical treatments

Thermochemical processes, in particular pyrolysis, that generate monomers from plastic streams are increasingly used as recycling routes for waste streams that cannot be economically valorised by other primary recycling routes. The main target polymers for chemical recycling to produce monomers are PS (Prathiba *et al*. 2018), PMMA (Maisels *et al*. 2021), PET (Sinha *et al*. 2010), PUR (Lucas *et al*. 2018b) and PA (Mihut *et al*. 2001). The general process involves a controlled transformation of polymers into monomers or oligomers of the same polymer that can be fed into the polymerization step of polymer production. The benefits and disadvantages have been reviewed elsewhere (Rahimi and Garcia 2017).

Information about chemical recycling routes and existing facilities and companies has been compiled in recent reviews (Vollmer *et al*. 2020; Maisels *et al*. 2021). PS and PMMA depolymerization plants

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[18] Please note that plastic to fuel is not considered recycling by ISO Standard definition (ISO 15270:2008) or EU regulation (European Parliament 2018).

(Toshiba Plant System, Toyo, JSW, Agilyx) perform a true chemical recycling of polymers into monomers, which allows for any further use of the monomers and aligns with the circular economy concept (Maisels *et al*. 2021). However, there is little to no information regarding industrial-scale depolymerization plants, and further critical assessment is needed. Since many of the monomers are hazardous chemicals, these processes pose at least a similar occupational health risk as the respective polymer production (see section 3.4.1). Furthermore, thermal degradation products such as carcinogenic PAHs or BTEX can be formed and need to be considered for emission and exposure assessment and control.

Plastic additives are likely degraded to a large extent in these processes, depending on the operation temperature. The effects of additives in these processes, including the formation of unintentional POPs, are discussed below in Section 7.3.2.3 and need further assessment.

7.3.2.2 Solvolysis

The decomposition of polymers into oligomers or monomers in solvents under the action of these solvents or of the chemical reagents dissolved therein (bases, acids, alcohols, etc.) is called solvolysis. The main target polymers are PET, PUR and PA.

Chemical recycling of PET addresses the PET that cannot be mechanically recycled, such as coloured, complex or multi-layered PET products. In general, two types of processes can be distinguished: (i) hydrolysis/saponification, which produces purified terephthalic acid (PTA) and mono ethylene glycol (MEG); and (ii) glycolysis by MEG that produces bishydroxyethyl terephthalate (BHET), which can be further hydrolysed to MEG and PTA. Several processes follow one of these routes (Carbios, Garbo, Gr3n, Ioniqa, Loop) and a large-scale European project aims to implement the technology (Demeto 2019). Recent innovations such as the solvolysis of PC and PET at room temperature with a zinc catalyst (Payne *et al*. 2022) indicate the future potential of the technology.

Chemical PUR recycling has been reviewed extensively (Lucas *et al*. 2018a; Lucas *et al*. 2018b). PUR is produced by the reaction of di- or polyisocyanate with a polyol. Solvolysis of PUR reverses these reactions and produces oligomers of the polyols and hazardous isocyanates. Different processes have been described in the literature including alcoholysis, hydrolysis, hydroglycolysis and aminolysis, but have not been applied at a large scale (Behrendt and Naber 2009).

The extent to which additives can be eliminated or degraded in solvolysis processes has not been documented. Depending on the chemical process and the chemical composition of the additives, the additives will be present in the same form or undergo chemical reactions during solvolysis. This will likely result in complex chemical mixtures. These process mixtures are subjected to refining steps such as distillation to isolate the monomers, which might eliminate additives and reaction products to a large extent, but result in distillation residues that require proper waste management.

7.3.2.3 Pyrolysis and gasification

Pyrolysis (pure or aided by catalysts or by the addition of hydrogen) and gasification processes aim to decompose the polymers into feedstock chemicals (Ragaert *et al*. 2017; Maisels *et al*. 2021; Quicker *et al*. 2022). Pyrolysis results in a more or less complex mixture of degradation products (pyrolysis oil) that can be used as a substitute for Naptha and, together with Naptha, is fed into cracking processes at refineries at the beginning of polymeric value chains. In contrast, gasification processes produce syngas, a mixture of mainly CO_2, CO, H_2, CH_4 and other light hydrocarbons. The global state of pyrolysis/gasification technologies has been reviewed elsewhere (Maisels *et al*. 2021; Quicker *et al*. 2015, Vollmer et. al. 2020) and BAT have been compiled (European Commission 2019c). Many pyrolysis and gasification projects for waste plastic and other wastes have failed or were not economically viable in developed countries (Gleis 2011; Schlotter 2019; Brock *et al*. 2021a; Brock *et al*. 2021b; Quicker *et al*. 2022). Developing countries lack experience with such projects, in particular with the assessment of releases. The long-term use of fuels from the chemical recycling of plastics can impact the engine and result in operation failure after some time (Kalargaris *et al*. 2017).

Heavy metal additives are largely transferred to the pyrolysis char (Day *et al*. 1999). Organic additives are degraded to some extent, depending on the temperatures used, which normally range between 400 and 800°C. Brominated and chlorinated additives can be degraded to a large extent, releasing HBr and HCl. Modern plants can deal with a certain amount of chlorine (Maisels *et al*. 2021) introduced by PVC and additives such as SCCPs. Systems for removing the chlorine and bromine resulting from additives and polymers have been developed or are being researched (Bockhorn *et al*. 1999; Fekhar *et al*. 2019; Hornung and Seifert 2006). PAHs are formed in large amounts during plastic pyrolysis processes (Zhou *et al*. 2015) and need control and regulation. Pyrolysis processes treating plastic with BFRs, chlorinated additives or PVC can release unintentional POPs such as PCDD/PCDFs and PBDD/PBDFs (Weber and Kuch 2003; Mei *et al*. 2017). Therefore, these processes need to be monitored and regulated similar to incinerators.

7.3.3 Energy recovery and destruction of POPs and other hazardous chemicals

Since plastics on average have an energy value higher than coal (Themelis *et al*. 2011), the direct recovery of energy from plastic waste is an option considered in the waste hierarchy, in particular for plastic that cannot be recycled or contains POPs (UNEP 2002a; UNEP 2021a). In developed countries, a large proportion of plastic waste is utilized for energy recovery (e.g., over 40% in Europe; PlasticsEurope 2019). Thermal recovery is also an increasing trend in developing countries; however, the facilities are usually not equipped with the necessary pollution control mechanisms, resulting in the contamination of the environment (air, soil) and food (Haarman 2016; Petrlik *et al*. 2019b; Petrlik *et al*. 2022; see section 7.2.5).

Energy recovery technologies can be used to destroy the POPs present in plastic wastes, as required by the Stockholm and Basel Conventions. The destruction of POPs-containing plastics requires appropriate monitoring schemes, ideally with continuous monitoring of any releases of PCDD/PCDFs and other POPs (Weber 2007; Reinmann *et al*. 2010). When destroying plastic wastes containing halogenated POPs or other halogenated additives, or in general polymers containing chlorine (e.g., PVC, PVDC, chloroprene) or fluorine (e.g., PTFE, polyvinyl fluoride, perfluoroalkoxy polymer), the emissions of acid gases (HCl, HF or HBr) need to be controlled using appropriate air pollution control measures (European Commission 2019c). Since aromatic BFRs are dioxin precursors and chlorine and bromine generally increase the formation of unintentional POPs (Weber and Kuch 2003; UNEP 2017a), only BAT/BEP facilities with appropriate combustion efficiency, air pollution control devices and residue management can be utilized for the treatment, to avoid environmental contamination and human exposure (see Section 7.2.5 and 7.2.6). Particularly demanding is the destruction of PFASs such as side-chain fluorinated polymers due to the high stability of the carbon-fluorine bond (Huber *et al*. 2009; UNEP 2021k). Heavy metals contained in plastic waste are not destroyed but are transferred to ashes and other waste residues.

7.3.3.1 Energy recovery and destruction of hazardous chemicals by co-processing in cement kilns

Cement kilns are increasingly used in waste management schemes in both industrial and developing countries (GTZ and Holcim 2006; Waltisberg and Weber 2020). Plastic waste can be used as a substitute for primary fuel in cement kilns. The Stockholm Convention BAT/BEP guidance and the Basel Convention technical guidelines suggest BAT cement kilns as BAT/BEP for the destruction and energy recovery of PBDE-containing plastic and can normally meet regulatory limits for unintentional POPs (UNEP 2011; UNEP 2019d; UNEP 2021c). International projects for energy recovery from plastic waste, such as the Norwegian regional project “Ocean Plastic Turned into an Opportunity in Circular Economy” (OPTOCE) for the thermal recovery of non-recyclable plastic waste in five Asian countries, establish the needed management and supervision scheme in pilot developing countries (Norad and SINTEF 2019).

Chlorine and bromine in waste plastic feed need to be controlled (normally kept at <0.5%), otherwise they may accumulate in the cement kiln system and cause operation problems (Buekens and Cen 2010; Waltisberg and Weber 2020). The introduction of a “chlorine bypass” in a cement kiln can overcome this challenge (Zhan *et al*. 2016) but generates cement kiln bypass dust (CKBD) with high chlorine content, which needs to be managed in an environmentally sound manner (European Commission 2013b; Czapik *et al*. 2020).

Cement kilns can also destroy organofluorine-containing polymers, such as PUF containing hydrochlorofluorocarbons (HCFCs) or hydrofluorocarbons (HFCs) (UNEP 2002b; Vehlow and Gloël 2020) and side-chain fluoropolymers containing PFOS, PFOA, or treat non-polymeric PFASs such as those adsorbed on resins (Patterson and Dastgheib 2020). One advantage of the cement kiln system is that the HF generated reacts with CaO and remains in the clinker. The addition of fluorine is even beneficial as a flux in clinker production and can reduce the sintering temperature (Yamashita and Tanaka 2011).

7.3.3.2 Energy recovery and destruction of hazardous chemicals in waste incinerators

Waste incinerators that use BAT/BEP (European Commission 2019c) can co-incinerate plastic waste with high content of POPs (e.g., PBDEs or HBCDD), and can normally meet regulatory limits for unintentional POPs (Sakai *et al*. 2001; Vehlow *et al*. 2002; Weber and Kuch 2003; Mark *et al*. 2015). The Stockholm Convention BAT/BEP guidance and the Basel Convention technical guidelines suggest BAT waste incinerators as a BAT/BEP for the destruction and energy recovery of PBDE-containing plastic and other POPs-containing waste (UNEP 2019d; UNEP 2021c). To destroy the extraordinary stable C-F bonds in PFASs, these substances and related plastic wastes must be exposed to very high temperatures (1,100 °C) for a sufficiently long time (UNEP 2021k).

For developing countries, BAT waste incinerators are costly and challenging to operate (ISWA 2013; Nixon *et al*. 2017). Therefore, the International Solid Waste Association (ISWA) developed guidelines that set prerequisites for operating waste incineration, including a “mature” waste management system with proper waste collection, a minimum average calorific value of at least 7 MJ/kg, and affordable required tipping fees (ISWA 2013). In addition, a robust regulatory framework and strict management of the toxic ashes to prevent contamination of the environment and food are needed (Petrlik and Bell 2020; see Section 7.2.5).

7.3.3.3 Energy recovery and destruction of hazardous chemicals in metal industries

The Stockholm Convention BAT/BEP guidance documents suggest a range of other thermal facilities as BAT/BEP for the destruction and energy recovery of PBDE-containing plastic, such as metal industries (e.g., primary steel plants, copper smelters, aluminium smelters) (UNEP 2017a; UNEP 2021c), which need BAT operation and flue gas cleaning to control UPOP release to meet regulatory limits. In particular, plastics are used as feedstock in the primary steel industry, either directly in blast furnaces as coke substitutes, or as substitutes for coal in the production of coke (Japan National Institute for Environmental Studies 2010; European Commission 2013c; Hesham 2018). According to the European BAT reference document for iron and steel, light shredder residues are feedstocks in primary steel production (European Commission 2013c).

An LCA for plastic containing PBDEs concluded that feedstock recycling of PBDE-containing material in the primary steel industry was the best choice (Hirai and Sakai 2007). The study stressed that the Japanese primary steel industry only accepts a halogen content of up to 0.5% (bromine or chlorine) (Hirai and Sakai 2007). In Europe, the allowable halogen content is higher, at 1.5% chlorine (Tukker 2002; European Commission 2013c). However, the BAT reference document emphasizes that the heavy metal content of the plastic waste fractions used as feedstock (e.g., from WEEE or automobile shredder residues) also determines the suitability and limitation (European Commission 2013c).

7.3.4 Improving landfilling of plastic waste containing hazardous chemicals in developing countries

The disposal of plastic waste in engineered landfills should be avoided, in line with the waste hierarchy (UNEP 2021j). Disposal of plastic waste containing POPs and other hazardous chemicals in engineered landfills should only be considered if no BAT/BEP destruction options for such plastic waste are available (UNEP 2017a). The drawbacks of landfilling plastic waste are described in Section 7.2.6.

7.3.4.1 Engineered landfills for disposal of plastic wastes containing hazardous chemicals

A framework for landfilling PBDE-containing plastic wastes, primarily aimed at developing countries, was included in the Stockholm Convention BAT/BEP guidance to improve the landfilling of these plastic fractions and allow potential future recovery by landfill mining (UNEP 2017a). Only engineered landfills with at least the following engineered measures should be considered for the disposal of plastic and polymers containing hazardous chemicals such as PBDE-containing waste (UNEP 2017a):

- Appropriate landfill siting that takes into account proximity to residents and other sensitive receptors, local groundwater, surface water and flood risks;
- Strict site security and control of wastes entering the landfill site for disposal;
- Management of the landfill to minimize fire risks by appropriate compaction, daily cover, etc. (see UNEP 2017a);

- State-of-the-art bottom liners to minimize leakage;
- Profiling and capping to minimize infiltration;
- Collection, control and treatment of leachates, including plastic additives (e.g., BPA, phthalates, PBDEs, PFAS, OPFRs, SCCPs);
- Collection, control and treatment of landfill gas;
- Comprehensive monitoring regime with long-term aftercare.

7.3.4.2 Long-term aftercare considerations, closure of engineered landfills and restriction of disposal

Engineered landfills also need aftercare, especially for releases of leachates that frequently contain plastic-associated chemicals, including POPs (e.g., PFOS, PFOA, PBDEs; SCCPs; Weber *et al*. 2011). In particular, BPA and PFASs (including PFOS, PFOA, and PFHxS) can be released at a high concentrations from closed landfills even 50 years after closure (Hepburn *et al*. 2019; Propp *et al*. 2021). A regulatory framework for the aftercare of landfills is needed, including monitoring of plastic-associated chemicals (e.g., PFAS, BPA, OPFRs) and other pollutants in leachates. To end the aftercare, the landfill operator needs to prove to the competent authority that the landfill no longer poses a threat to the environment (Scharff *et al*. 2010).

The long-term leaching and degradation of plastics and the related releases of persistent chemicals and degradation products need to be compared to the durability of containment and engineering measures (Buss *et al*. 1995; Allen 2001; Simon and Mueller 2004; Weber *et al*. 2011). Plastics containing hazardous chemicals with long-term release potential should be restricted from disposal to landfills. This is particularly important for highly persistent plastic additives, such as PFASs, and related waste products (Washington and Jenkins 2015; Section 7.2.6), such as waste containing side-chain fluorinated polymers (e.g., treated furniture, synthetic carpets, car shredder residues). Plastics that depolymerize in landfills releasing hazardous monomers such as BPA (Gomi *et al*. 2022) also need to be controlled. Further research is needed into the long-term fate of polymer degradation and related long-term releases from landfills.

7.4 Developing capacity for waste management, recycling, recovery and destruction

Without a solid waste management system, plastics (and other wastes) containing hazardous chemicals cannot be appropriately recycled, recovered or destroyed**.** The International Environmental Technology Centre of UNEP commissioned a needs assessment study on capacity building for waste management in the context of the Global Partnership on Waste Management (GPWM) (de Luna Era 2016). Specific need areas included organizational solid waste management capacity (e.g., legislation) and technology, human resource capacity, advocacy, knowledge management, and partnership and alliance-building capacity. A holistic capacity building for waste management in developing countries is needed to set the framework for robust management of plastic waste and address the hazardous chemicals in plastics and their challenges (see Section 7.2.3), while utilizing the best options for material recycling, chemical recycling or energy recovery for the contaminated plastic fractions. This is particularly important to address the complex issues of hazardous chemicals in plastic, phase them out in recycling and destroy them in thermal processes while controlling the releases.

Robust collection infrastructures are key to avoiding the leakage of plastics and related chemicals into the environment and reducing the release of persistent chemicals and unintentional POPs from open burning. To generate added value in the plastic waste management sector, investment into waste type-specific sorting plants is required, as well as subsequent plastic recycling facilities for the dominating polymers in each waste stream. Within such a type-specific sorting, hazardous chemicals should be managed and eliminated (UNEP 2021c). For the waste management sector, in particular the informal sector, a fair compensation and risk protection is needed, as well as a legal framework for the just transition of waste pickers into newer roles and their involvement in any just transition to newer systems, materials, collection, recycling and distribution options with respect to plastics (ILO 2002; ILO 2014; Global Alliance of Waste Pickers 2022). Furthermore capacity building in the sector is needed to separate plastic containing hazardous chemicals (Haarman 2016). Capacity building in the education of end-of-life managers is crucial for implementing modern end-of-life structures in developing countries that can also manage and eliminate hazardous chemicals. The first important step is cooperation between the formal and informal sectors and practical guidance for creating successful formal-informal partnerships (GIZ 2018a; GIZ 2018b), including education about hazardous chemicals and how to eliminate them in an environmentally sound manner. Furthermore, a stepwise transformation of informal sectors into formal ones is a great opportunity for implementing new structures (Godfrey 2017).

Destroying the large number of plastics containing POPs and other hazardous chemicals in an environmentally sound manner (Babayemi *et al*. 2015; Babayemi *et al*. 2018; Guida *et al*. 2022) requires building capacity for management, destruction and emission monitoring. This includes establishing facilities appropriately designed for the destruction of POPs and other chemicals of concern with BAT control of releases. The Stockholm and Basel Convention developed BAT/BEP guidance documents that can be used for capacity building (e.g., UNEP 2021c; UNEP 2021h). Furthermore, the Stockholm Convention has the financial mechanism to establish projects for the management of plastics containing POPs. Currently, several projects are already in the implementation phase. For instance, a UNIDO/UNITAR GEF project (GEF 2019) is establishing the separation and recycling of POPs-containing plastics from WEEE and end-of-life vehicles in Côte d'Ivoire. Other UNIDO GEF projects (GEF 2021a; GEF 2021b) are focusing on the management and destruction of HBCDD-containing EPS/XPS in China and Turkey. The lessons learned from these projects can be utilized for further capacity building in developing countries. Global legally-binding instruments can also support the development of regulatory frameworks and the necessary management and implementation capacity.

ANNEX 1 COMPILATION OF CHEMICALS IN PLASTICS

A comprehensive overview of additives and other chemicals found in plastics is impeded by scattered information in the public domain, such as in industry databases, scientific studies, or regulatory lists. Two recent studies (Aurisano *et al*. 2021b; Wiesinger *et al*. 2021;) attempted to compile this information. Annex 1 combines the results of these two studies and provides references for further information on the individual substances identified. It contains more than 13,000 unique chemical substances, including plastic additives, processing aids and other plastic chemical constituents identified across various sectors. The list may be accessed here:

https://wedocs.unep.org/bitstream/handle/20.500.11822/40858/chemicals_plastics.xlsx?sequence=3&isAllowed=y

The overview list in Aurisano *et al*. (2021b) is based on the mapping of plastic additives conducted by the European Chemicals Agency (ECHA) (ECHA 2019a) and expanded with data from a wide range of other scientific sources, including the Database of Chemicals associated with Plastic Packaging (CPPdb) compiled by Groh *et al*. (2019) and the Chemicals and Product Categories database (CPCat) from the United States Environmental Protection Agency (US EPA) (Dionisio *et al*. 2015). As a result, Ausario *et al*. (2021b) compiled data on more than 6,000 chemical substances in plastic. Annex 1 includes information on the functions, typical material application, and typical concentration ranges of these chemicals where they were available.

The list compiled by Wiesinger *et al*. (2021) is based on information from various sources, including the scientific literature, industry databases such as the SpecialChem Selector, industrial handbooks such as the Plastics Additives Database by Ernest Flick (2004), and regulatory lists such as the Union List of monomers and additives authorized for use in plastic food contact materials (European Commission 2011a). As a result, Wiesinger *et al*. (2021) compiled data on more than 10,000 chemical substances related to plastic.


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For more information, contact:

Chemicals and Health Branch
Industry and Economy Division
United Nations Environment Programme

Palais des Nations
8-14 avenue de la Paix
CH-1211 Geneva 10, Switzerland

www.unep.org