The use of PVC (Poly Vinyl Chloride) in the context of a non-toxic environment

Final report

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LIST OF ABBREVIATIONS

Abbreviation	Full term
°C	Degree Celsius
ABS	Acrylonitrile butadiene styrene
AEL	associated limit values
AgPR	Association for PVC floor covering recycling
AoA	Analysis of alternatives
APC	Air pollution residues
BAT	Best available technologies
BAU	Business as usual
BBP	Benzyl butyl phthalate
BBzP	Butylbenzyl phthalate
BMU	German Federal Ministry for the Environment, Nature Conservation and Nuclear Safety
BOPA	Biaxially-oriented nylon
BPD	Bypass dust
BPF	British Plastics Federation
BREF	Best Available Techniques Reference document
C&D	Construction and demolition
$CaCO_3$	Calcium carbonate
CAGR	Compound annual growth rate
CAS	Chemical Abstracts Service
CC/PO	Cyclic olefin copolymer
CCTV	Closed-circuit television
CDC	United States Centre for Disease Control
CE	Conformité Européenne
CEWEP	Confederation of European Waste-to-Energy Plants
CKD	Cement kiln dust
CLP Regulation	Regulation (EC) No 1272/2008 on classification, labelling and packaging of substances and mixture
CMR	Carcinogenic, mutagenic or toxic to reproduction substances
CO_2	Carbon dioxide
CP	Chloroprene
CPE	Chlorinated polyethylene
CPUs	Central processing units
CRT	Cathode ray tube
CSPE	Chlorosulfonated polyethylene
CSS	Chemicals Strategy for Sustainability for a Toxic-free Environment
D4	Octamethylcyclotetrasiloxane

Abbreviation	Full term
D5	Decamethylcyclopentasiloxane
D6	Dodecamethylcyclohexasiloxane
DBP	Dibutyl phthalate
DEHP	Diethylhexyl phthalate
DEP	Diethyl phthalate
DIBP	Diisobutyl phthalate
DiDP	Diisodecyl phthalate
DINCH	1,2-Cyclohexane dicarboxylic acid diisononyl ester
DINP	Diisononyl phthalate
DMP	Dimethyl phthalate
DS	Dry Solis
DWV	Drainage water vents
EAP	Environment Action Programme
ECHA	European Chemicals Agency
ECVM	The European Council of Vinyl Manufacturers
EDC	1,2-dichloroethane
EEE	Electrical and electronic equipment
EFSA	European Food Safety Authority
ELV	End-of-life vehicle
ELV-Directive	Directive 2000/53/EC on end-of life vehicles
EPDM	Ethylene propylene diene monomer
EPPA	European Trade Association of PVC Window System Suppliers
EPR schemes	Extended producer responsibility scheme
EPS	Expanded Polystyrene
E-PVC	Emulsion polyvinyl chloride
eq	Equivalent
ESA	Environmental Service Association
ESWA	European Single ply Waterproofing Association
ETFE	Ethylene tetrafluoroethylene
EU	European Union
EuCertPlast	European Certification of Plastics Recyclers
EuPC	European Plastics Converters
EURiTS	European Union for Responsible Incineration and Treatment of Special Waste
EVA	Ethylene-vinyl acetate
EVOH	Ethylene vinyl alcohol
FAO	United Nations Food and Agriculture Organization

Abbreviation	Full term
FEP	Fluorinated ethylene propylene
Fraunhofer IVV	Fraunhofer Institute for Process Engineering and Packaging IVV
FU	Functional Unit
GDP	Gross domestic product
HBCDD	hexabromocyclododecane
HCl	Hydrogen chloride
HDPE	High-density polyethylene
HIPS	High Impact Polystyrene
HP	Hazardous properties
ICT	Information communication technology
IED	Directive 2010/75/EU on industrial emissions
IFC	International Finance Corporation
IMPVAL	Imported value
INTERPOL	The International Criminal Police Organization
IPEN	International pollutants elimination network
ISO	International Organization for Standards
IUPAC	International Union of Pure and Applied Chemistry
IV	Intravenous
IVK	Association of Coated Fabrics and Films
kg	Kilogram
kt	Kiloton
kW	Kilowatt
Landfill-Directive	Directive 1999/31/EC on the landfill of waste
LCA	Life Cycle Assessment
LDPE	Low-Density Polyethylene
LOI	Limiting oxygen index
LoW	List of Waste
LPCL	Low Pop Content Level
MBzP	Metabolite monobenzyl phthalate
MDF	Medium-density fibreboard
MDPE	Medium-density polyethylene
MEK	Methylethylketon
MJ	Megajoule
mm	Millimetre
MPa	Mega pascale
mPPE	Modified polyphenylene ethere
M-PVC	Bulk mass-PVC

Abbreviation	Full term
MSDS	Material safety data sheet
MSW	Municipal solid waste
MSWI	Municipal Solid Waste Incineration plant
MW	Megawatt
NaCl	Sodium chloride
$NaHCO_3$	Sodium bicarbonate
NBR	Nitrile butadiene rubber
NGO	Non-Governmental Organisation
NIAS	Non-intentionally added substance
NIR	Near-infra-red
OECD	Organisation for Economic Co-operation and Development
OTNOC	Other than normal operating conditions
PA	Polyamide
PAEs	Phthalate diesters
PAH	Polycyclic aromatic hydrocarbon
PB	Polybutylene
PBAT	Polybutylene adipate terephthalate
PBDD/F	Polybrominated dibenzo-p-dioxins and furans
PBDE	Polybrominated diphenyl ether
PBS	Polybutylene succinate
PBT	Persistent, bioaccumulative and toxic
PC	Polycarbonate
PCB	Polychlorinated biphenyl
PCDD/Fs	Polychlorinated dibenzo-p-dioxins and polychlorinated dibenzo-p-furans
PCDDs	Polychlorinated dibenzodioxins
PCDFs	Polychlorinated dibenzofurans
PCS	Potentially Contaminated Sites
PE	Polyethylene
PET	Polyethylene Terephthalate
PE-X	Cross-linked polyethylene
PFA	Perfluoroalkoxy alkanes
PFAS	Perfluoroalkyl and polyfluoroalkyl substances
PLA	Polylactic acid
PLASI	Plastic additives initiative
PMMA	Polymethyl methacrylate
PPWD	Directive 94/62/EC on packaging and packaging waste
POP	Persistent organic pollutant

Abbreviation	Full term
POP Regulation	Regulation (EU) 2019/1021 on persistent organic pollutants
PP	Polypropylene
PS	Polystyrene
PST	Post-shredding technologies
PTFE	Polytetrafluoroethylene
PUR	Polyurethane
PVB	Polyvinyl butyral
PVC	Polyvinyl chloride
PVC-E	Emulsion-PVC
PVC-HI	High impact polyvinyl chloride
PVC-O	Molecularly oriented polyvinyl chloride
PVC-P	PVC plasticised
PVCS	PVC Separation technology
PVC-S	PVC Suspension
PVC-U	PVC-unplasticised
PVDC	Polyvinylidene chloride
R&D	Research and development
RAC	Committee for Risk Assessment
RDF	Refuse Derived Fuel
REACH	Regulation (EC) No 1907/2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH)
RoHS	Restriction of Hazardous Substances in Electrical and Electronic Equipment
rPVC	Recycled PVC
RVCM	Residual vinyl chloride monomer
SCIP Database	Database on substances of concern in products
SBR	Styrene-butadiene rubber
SEA	Socio-economic analysis
SHF	Shredder heavy fraction
SLF	Shredder light fraction
SME	Small and medium-sized enterprise
SML	Specific migration limit
S-PVC	Suspension polyvinyl chloride
STOT	specific target organ toxicity
SVHC	Substance of very high concern
t	Tonnes
TEPPFA	European Plastic Pipe and Fittings Association
TEQ	Toxic Equivalent
the UK	The United Kingdom

Abbreviation	Full term
TPE	Thermoplastic elastomers
TPE-O	Thermoplastic olefin
TPU	Thermoplastic polyurethane
TRADA	Timber Research and Development Association
UNEP	United Nations Environment Programme
uPVC	Unplasticized PVC
US	United States
US EPA	United States of America Environment Protection Agency
UTC	unintentional trace contaminant
UV	Ultraviolet
VCM	Vinyl chloride monomer
VOC	Volatile organic compound
vPvB	Very persistent, very bioaccumulative
W/mK	Watt per meter and Kelvin
WEEE	Waste of electrical and electronic equipment
WEEE-Directive	Directive 2012/19/EU on waste electrical and electronic equipment
WFD	Framework Directive 2008/98/EC on waste
WSR	Regulation (EC) No 1013/2006 on shipments of waste
wt	Weight
Ω	Ohm

Executive summary

Polyvinylchloride (PVC) is a synthetic plastic polymer, available in rigid and flexible form. Various characteristics of rigid and flexible PVC, as well as its relatively low cost, make it an attractive material for various applications within the EU´s economy. However, various additives used in PVC to ensure the desired characteristics may lead to environmental and human health challenges, in the manufacturing, use and end-of-life phases of the product life cycle. As such, an assessment of PVC´s characteristics and the implications of its applications and end-of-life-fate can be considered highly relevant within the context of various EU Green New Deal policies, most notably the Circular Economy Action Plan (CEAP), the Sustainable Chemicals Strategy and the Zero Pollution Action Plan. Within this context, this report provides an overview of the current status of the production, trade consumption and end-of-life fate of PVC. In addition, the report assesses the feasibility of alternatives to PVC in selected applications, as well as a number of potential phase-out scenarios for PVC applications and their socio-economic implications.

<u>The chemistry of PVC</u>

PVC is a thermoplastic polar polymer, which is produced mainly via ethylene and chlorine. In its basic form, PVC is a hard and brittle compound, however, due to its polar nature it can incorporate a wide range of additives, to achieve various mechanical and chemical properties. The main types of additive in PVC are plasticisers, stabilisers and pigments.

It can be concluded from the literature that the vast majority of PVC additives are not covalently bound to the polymer and as such can migrate out of the polymer matrix. This migration depends on several factors such as the solubility of the additive in the polymer matrix, the size of the additive molecules, the temperature, the concentration of the additive and the type of polymer matrix. It is therefore difficult to make general assumptions about the migration potential and rate of an additive. Concrete data about the migration potential and rates for many additives in PVC is scarce. The type of PVC (flexible vs. rigid) is an important factor. Additives tend to have a higher migration rate in flexible than in rigid PVC. However, the specific migration potentials and rates of the additives in a certain polymer matrix need to be experimentally confirmed and general conclusions cannot be drawn.

PVC mixtures seem to be commonly classified as non-hazardous based on the argument that the additives are bound in the polymer matrix and do not migrate and hence are not bioavailable. In this regard, it is important to note that various literature sources indicate that migration of additives from plastics takes place, since such additives are not covalently bound in the plastic matrix. However, data on migration potential and bioavailability of PVC additives is currently limited. Nevertheless, from a precautionary perspective, a more critical assessment of the classification of PVC mixtures as hazardous or non-hazardous may be necessary.

<u>The market for PVC</u>

According to Eurostat, approximately 6 million tonnes of primary PVC were manufactured in 2019 in the EU27. Of this amount:

- 4.9 million tonnes were unmixed non-plasticised PVC[1];
- 0.4 million tonnes were non-plasticised PVC[2]; and
- 0.8 million tonnes were plasticised PVC mixed with other substances[3].

European production volumes of primary unmixed PVC declined in the period between 2008 and 2019, with a decrease of 26% between 2008-2012, and more gradual decline thereafter. Production volumes of primary plasticised and non-plasticised PVC have been more stable since 2008. Data on EU PVC use in different sectors is varied in quality and granularity. Overall, the largest volumes of use are in:

[1] As classified by the Eurostat database. This includes PVC not mixed with any substances, plasiticisers included.

[2] As classified by the Eurostat database. This includes PVC mixed with other substances, excluding plasticisers.

[3] As classified by the Eurostat database. This includes PVC mixed with plasticisers and/or other substances.

- Construction (pipes and fittings, window frames);
- Medical and healthcare applications;
- Packaging; and
- Cables for electronics.

Less detailed information has been found on global PVC production, which was estimated at 55 million tonnes in 2016. Capacity data indicates that China accounted for almost half of this volume. PVC demand appeared to be somewhat lower than this, in the order of 45 million tonnes in 2018 but precise figures should be treated with caution. Historical data reveal that total global production has steadily increased in recent years, driven by increasing production capacity in China. More recent market analysis (2020) states that China is expected to remain the largest global producer and consumer of PVC for the foreseeable future. Pre pandemic, the global PVC market was expected to continue to grow at the rate of 3.5% 2016- 2026. China was expected to account for the majority of this growth.

Simple market projections have been derived in this study, based on extrapolating historical data. Such forecasts are associated with assumptions and uncertainties and the economic disruption associated with the COVID-19 pandemic amplifies these. The projections suggest that PVC production in the EU27 could either stabilise – which would be a continuation of trend between c.2013-2019 – or continue to decline (following the overall trend between 2008-2019). Offsetting this projected trend, volumes of primary (i.e. pellets/powder) and processed (i.e. products such as plastic pipes) PVC could continue to increase.

<u>PVC waste management</u>

It is assumed that currently 2.9 million tonnes of pre- and postconsumer PVC waste are generated per year in the EU (2.4 million tonnes post-consumer PVC waste). While sector-specific take-back systems and separate collection are successfully applied to post-consumer waste from the construction and building sector (e.g. windows, pipes) in most of the EU Member States, the (limited) available information suggests that separate collection remains a challenge in practice for most post-consumer PVC waste streams. PVC waste from packaging, automotive and medical waste is rarely separately collected (and thus recycled less).

In terms of waste treatment, mechanical recycling is currently the only relevant recycling process for pre- and postconsumer PVC waste in the EU. No significant chemical recycling of PVC is done at industrial scale in the EU; while chemical recycling generally can be considered a technology in development, chemical recycling specifically of PVC is not practiced in the EU (except in pilot plants or in research projects) and in most initiatives of chemical recycling, PVC is not the targeted material. In 2020, 730,000 tonnes of (pre- and post-consumer) PVC waste have been recycled within the VinylPlus program (regional scope is EU-27 plus Norway, Switzerland and the UK). About 60% of the recycled materials by VinylPlus consist of rigid PVC (i.e. profiles, pipes and fittings). A 35% majority of PVC recyclate is redirected into new windows and profiles, whereas 15% is directed into traffic management products and 13% into pipes. Window frames, which is the biggest PVC waste stream, seems to be the only PVC waste stream that can be considered recycled in closed loop or at least partly.

Separation and sorting technologies were optimised in the last years, however contaminants, composites, laminates or other complex product composition still pose a problem for conventional mechanical recycling. Moreover, restricted or unwanted additives (e.g. cadmium and lead stabilisers or phthalate plasticisers such as DEHP) are an issue for PVC recycling. In the future, non-conventional mechanical recycling processes like selective dissolution could be an option to remove contaminants and/or legacy additives from PVC waste, however currently no feasible recycling technology exists to remove such substances from PVC waste.

In the absence of feasible decontamination processes, the presence of restricted or unwanted additives in recyclate requires finding a balance between the aim for a toxic-free environment and calls for an increased recycling in the Circular Economy. In pipe and window production, but also in some flooring companies, PVC recyclate is inserted between or under layers of virgin PVC (co-/tri-extrusion). Apart from the case of pipes, for which this requirement applies due to REACH, it is not clear to what extent this is only done for aesthetical reasons or also to protect the environment from substances (e.g legacy additives) in the recyclate or to dilute hazardous substances with virgin material.

With regard to PVC waste treatment routes other than recycling, the constructed mass flow analysis revealed that several important data gaps exist concerning the fate of PVC, a quantitative scoping of five major PVC-relevant waste streams (ELVs, C&D waste, WEEE, packaging and household waste and medical waste) demonstrates that incineration and landfilling represent the main non-recycling treatment routes for PVC and that the treatment of PVC waste strongly depends on the specific waste stream and its properties. For both treatment methods, environmental concerns exist with respect to the treatment of PVC. Moreover, given doubts about the correct classification of PVC as hazardous, since data on migration and bioavailability of PVC additives is limited, from a precautionary perspective, further assessment of the classification of PVC as hazardous or non-hazardous may be necessary for the determination of the most suitable disposal route. In any case, major data gaps exist concerning the fate of PVC present in the analysed waste streams and the quantities directed towards different incineration facilities or landfill types.

Statistical data suggest that 20,000 tonnes of PVC post-consumer waste have been exported from EU Member States to third countries in 2020. Due to the existing differences concerning the recycling infrastructure and cost structure for waste treatment, significant quantities of PVC waste are shipped legally within the EU but also imported from and exported to non-EU countries. Recent developments in the legal framework on transboundary shipments of plastic waste (including PVC waste) from the EU to third non-OECD countries (see Annex 1.1) may lead to considerable changes in shipment volumes and practices. Available information suggests that hazardous plastic waste is often mixed with non-hazardous waste to disguise illegal export. As a result, the mixed plastic waste is not suitable for recycling anymore due to hazardous characteristics.

It should be noted that illegal transboundary movements may occur in various ways, such as the misclassification of waste as non-waste or a misclassification into a wrong entry of the Annexes of the EU Waste Shipment Regulation (WSR). Illegal transboundary movements of PVC waste may lead to PVC waste inappropriately treated in substandard facilities, or ultimately to wild dumping and burning of PVC waste outside the EU, with effects for health and the environment. Illegal disposal options of mixed or hazardous waste containing PVC include uncontrolled burning, illegal recycling, illegal trade and illegal dumping. The uncontrolled burning of waste containing PVC leads to emissions of various pollutants which are harmful to the environment or human health, such as dioxins. Further impacts on human health and the environment can occur due to illegal landfilling of waste containing PVC. However, it should be noted that, on the basis of currently available data, the described impacts cannot be ascribed specifically to PVC but rather to illegal and uncontrolled disposal of mixed waste in general.

<u>Alternatives to PVC</u>

The analysis indicates that there are economically viable and technically feasible alternatives in the vast majority, if not all, applications assessed in detail where PVC is currently used. These are not without technical drawbacks. The extent of these drawbacks differs between applications.

Additional costs are associated with their use in several cases. The alternatives identified fall into three broad groups:

- Alternative non-plastic materials (such as wood, leather, cloth etc)
- Alternative plastics (here a wider variety of plastics were identified; some of these contain additives which pose similar risks to those used in PVC and others which are the subject of potential regulatory action)
- More novel alternatives (such as bioplastics), about which limited application-specific information has been obtained. Some alternative plasticisers (i.e. phthalate free) were identified.

The vast majority of the alternative materials and plastics identified are commercially available, often placed on the EU market in significant volumes at present. The balance of evidence suggests that the human health and environmental risks associated with a transition would decrease, although identifying net effects are very challenging given the absence of full lifecycle data and differences between applications.

Phase out scenarios for PVC applications

The report considers potential options to accelerate or mandate an EU-wide phase-out of PVC. Several options were outlined, which could be undertaken alongside an industry data collection exercise to fill current data gaps and uncertainties. This could generate more detailed application-specific data on additives use, potential speed of substitution, application-specific exposure risks, as well as costs.

The options include:

- Taking no further action;
- Agreeing a voluntary phase out programme with industry; and
- Taking regulatory action.

These options include – but are not limited to – an ordinary legislative procedure; further REACH restrictions on specific additives used in PVC or on PVC itself, based on specific use(s) or all uses. This would require proof that the risk was not "adequately controlled". Several options depends on current reforms being considered to the operation of REACH, for example, the potential application of the "essential use concept". The analysis suggests that further data would be useful for a targeted risk-based approach.

In terms of economic impacts, potential alternatives to PVC comprise alternative materials, often those used before PVC or that have retained some market share as PVC use has expanded. Other alternatives are a variety of alternative plastics; more novel products such as phthalate free plasticisers or bioplastics. Furthermore, in some specific cases it may be possible to re-design the article to accommodate alternative materials.

Overall, there are two main options in the event of a phase out. It is expected, however, that a restriction in any one application would, in practice, result in a combination of the two:

a. Replace PVC with alternative materials or substances if a feasible alternative (or mix of alternatives) is available or identified after further development.
b. If the above is not technically or economically feasible, or the companies are unable to make investment decisions based on their current knowledge of alternatives, then the affected product lines, revenues and employment in the EU will cease. Current exports of virgin PVC and PVC-containing products would likely be lost, depending on the scope of action. The scale of required reformulation efforts would likely preclude other R&D activities.

A key issue concerning the above relates to distributional effects. In many cases, completely different upstream supply chains will be affected by a potential phase out. Decreases in demand, likely associated with significant effects to that sector, offset by increases in the manufacture and sale of alternatives. Manufacturers as well as wholesalers and distributors, who manufacture or supply several different materials, would be less affected under this scenario but would likely incur increases in prices and costs.

In terms of social impacts, loss of markets that cannot be replaced by alternatives would have implications for employment. Net effects in downstream sectors are not clear, but significant effects are expected in the PVC manufacturing sector depending on their portfolios and the scope and speed of action.

1. INTRODUCTION

1.1 Background and objectives

Polyvinylchloride (PVC) is a synthetic plastic polymer, which is available in rigid and flexible form. PVC is used in various sectors of the economy, most notably the construction, automotive, electronics and medical sectors. To improve performance, functionality, and ageing properties of polymers, various additives are used in the PVC production process. The resulting characteristics of rigid and flexible PVC, as well as its relatively low cost, make it an attractive material for various applications within the abovementioned and various other sectors.

However, from an environment perspective, PVC can be associated with a number of challenges in various phases of its lifecycle, most notably the waste phase. The considerable volumes of PVC waste generated annually are challenging in terms of sounds waste management and disposal. Such challenges relate, inter alia, to potential emissions of PVC additives (e.g. heavy metals) into air (in case of incineration) and soil (in case of landfilling), but also to illegal practices of dumping and open burning. Various PVC additives possess hazardous characteristic and thus, upon emission, may pose risks for the environment and human health.

It should also be noted that considerable amounts of PVC waste are currently recycled in the EU and thus redirected into the PVC product cycle as secondary raw material. While this share of PVC waste is diverted from the abovementioned legal and illegal disposal practices, another challenge arises within the context of the EU´s transition to a circular economy. On the one hand, the EU has committed itself to increased resource efficiency through its circular economy action plan, thus envisaging increased recycling of economically valuable materials such as PVC.[4] On the other hand, the EU has also expressed the aim to ensure toxic free product cycles for its citizens, thus envisaging the phase out of hazardous substances from product cycles.[5]

The objective of this study is to identify and describe uncertainties, particularly under a chemicals angle, about PVC production, recovery and end of life treatment, in order to help defining which role this material should play in the context of the European Green Deal and the Circular Economy Action Plan. The report takes into account environmental, human health, economic, legislative, and technical aspects, in order to enable the consideration of policy measures which ensure maximum recycling rates of PVC, but at the same time minimize environmental impacts and human health risks associated with management of PVC waste. In particular, the report includes:

1. A general market assessment concerning PVC production, trade consumption and end of life fate, including a projection of PVC development until 2050;
2. An overview of the chemistry of PVC, describing relevant additives and reporting information on migration into humans and releases to the environment.
3. An overview of main PVC waste collection, sorting and recycling activities and technologies in the EU with special attention for their technical, environmental and economic aspects;
4. An overview of legal and illegal PVC disposal practices in the EU with special attention for their technical, environmental and economic and in particular human health aspects;
5. An analysis of alternatives to PVC applications, considering relevant life cycle assessments aspects and market availability; and
6. A preliminary assessment of potential phase-out scenarios for PVC, taking into account environment, human health, and socio-economic implications.

[4] European Commission, A new Circular Economy Action Plan: For a cleaner and more competitive Europe, COM(2020) 98 final, Brussels, 11.3.2020; https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1583933814386&uri=COM:2020:98:FIN

[5] European Commission, Communication: The European Green Deal, COM(2019) 640 final, Brussels, 11.12.2019; https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1596443911913&uri=CELEX:52019DC0640#document2

1.2 Applied methodology

The information presented in the various sections of this report is based on a combination of desktop research and stakeholder consultation in accordance with the following steps and sources.

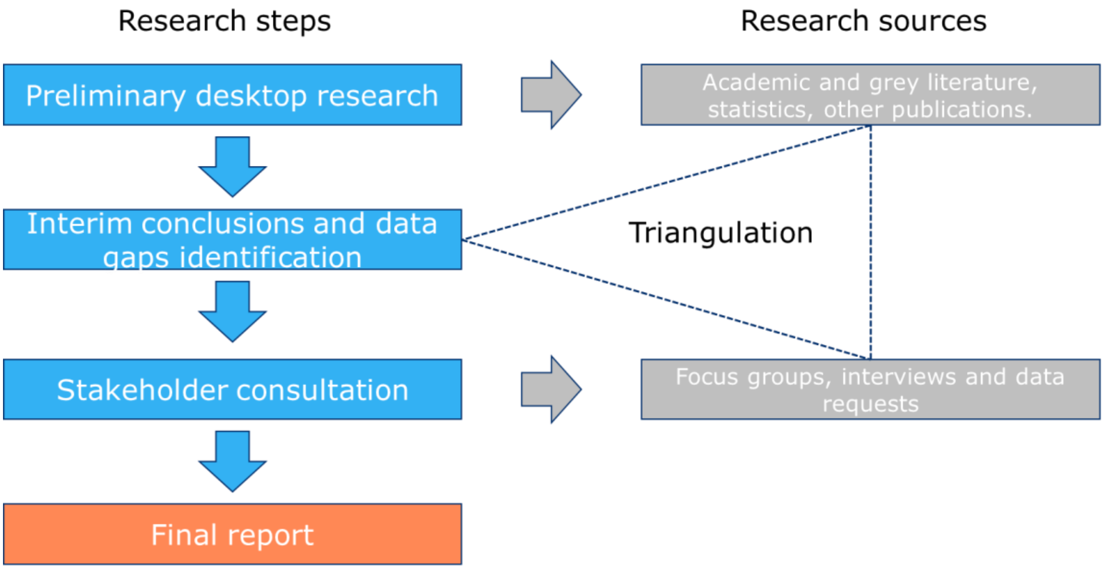


Figure 1. Applied methodology – steps and sources

Preliminary research on the basis of academic literature, statistical databases and other grey literature publications provided a first basis to assess the current state of knowledge on the PVC product lifecycle and its associated environmental, economic and social impacts. Preliminary findings were compiled to identify relevant data gaps. This step was followed by a round of consultation of relevant stakeholders and experts. A number of interactive focus groups formed the basis of the consultation, covering the following subjects:

- PVC recycling and decontamination practices;
- PVC legal and illegal disposal routes and export; and
- PVC alternatives for specified applications.

Participants of the focus groups were invited on the basis of their expertise and relevance for the identified data gaps. In addition to the focus groups, contact was sought with relevant experts and stakeholders via brief interviews or requests for data. It should be noted that various stakeholders have provided data and input in a more proactive manner. Such data or input was assessed critically by the research team and, were relevant, taken into account for the research. In general, data from desktop research and stakeholder consultation was triangulated with the expertise of the involved experts to ensure that findings are based on reliable and balanced data. Analysis of data has also taken into account the applicable EU legal framework for the placing on the market of PVC and PVC-based products, as well as the management of PVC waste in the end-of-life phase. An overview of applicable EU legal framework is provided in Annex 1.1.

Based on the abovementioned steps, the research team has drafted the present report. It should be noted in this regard that, in addition to presenting main findings (see chapter 9 for main conclusions), the research team has reflected on main data gaps which persisted throughout the study. An overview of these gaps is provided in Annex 7.1 and includes reflections on the implications of these gaps for the report, as well as suggestion on how the gaps could be addressed through future action.

2. THE CHEMISTRY OF PVC

Key Messages – Chemistry of PVC

Scope and Approach

The purpose of this chapter is to explore the chemistry of PVC and compare it with other plastic materials (for example PE, PP) used for similar purposes. This includes highlighting of PVC-specific additives (e.g., plasticisers) and their potential risk, based on the available data.

Properties of PVC

PVC is a thermoplastic polar polymer which is produced mainly via ethylene and chlorine. In its basic form, PVC is a hard and brittle compound, however, due to its polar nature it can incorporate a wide range of additives, to achieve various mechanical and chemical properties.

Additives used in PVC

The most important type of additive in PVC are plasticisers. These are added to provide the PVC with a degree of flexibility, enabling use in many different applications (e.g., flooring or medical gloves). PVC releases hydrogen chloride at temperatures above 60 °C, so heat stabilisers made from metal compounds are also often added. Other stabilisers used include co-stabilisers, lubricants, acid scavengers, flame retardants, pigments, fillers and impact modifiers.

Mapping of PVC-additives

The specific additives used in PVC have been mapped. This was based on the results of the PLASI initiative alongside several literature sources. In total around 370 additives were identified, most of which are pigments. Around 60 heat stabilisers, light stabilisers and plasticisers were also found. Many of the additives can fulfil multiple functions (e.g., cadmium compounds can be heat stabilisers and pigments). Some of the additives, especially plasticisers are substances of concern and have hazardous properties. The full list of additives is enclosed to this report as Annex 2.

Detailed description of PVC-additives concerning their migration and bioavailability

The vast majority of additives is not covalently bound to the polymer. As such, additives can migrate out of the polymer matrix and can cause negative effects to humans and the environment. This migration depends on several factors such as the solubility of the additive in the polymer matrix, the size of the additive molecules, the temperature, the concentration of the additive and the type of polymer matrix etc. It is therefore difficult to make general assumptions about the migration of a substance and concrete data about the migration for many additives is scarce. In the case of PVC, the type of PVC (flexible vs. rigid) is an important factor. Additives tend to have a higher migration rate in flexible than in rigid PVC. For certain additives, such as phthalates and metal compounds (e.g., lead and cadmium) migration has been confirmed from both flexible and rigid PVC, but the migration in flexible PVC is much higher than in rigid PVC. Pigments on the other hand tend to have a very low tendency to migrate, as they have an extremely low solubility in water, organic solvents and various other media. However, the specific migration potentials and rates of the additives in a certain polymer matrix need to be confirmed by experiments; general conclusions cannot be drawn.

Classifications of different mixtures of PVC including its additives

According to EU CLP Regulation a mixture needs to be classified as hazardous, if one of the substances it contains is has hazardous properties, and is added to the mixture in a concentration above the relevant limit values set out in Annex I of the Regulation; an exception to this rule applies if it can be proven through adequate and robust data that the substance is not bioavailable.

While many of the additives in PVC are hazardous, as a starting point a PVC mixture should be classified as hazardous if the relevant thresholds are exceeded; however, the argument is frequently used that additives are bound in the polymer matrix and do not migrate and hence are not bioavailable in the case of certain PVC mixtures, leading to a classification as non-hazardous. Chemically this is however inaccurate, as the vast majority of additives are not covalently bound to the polymer matrix and as such have the potential to migrate.

Against this background, publicly available material safety data sheets (MSDS) of PVC mixtures were analysed for data on listed additives and the statement that the additives are encapsulated in the polymer matrix. 24 MSDS were identified, of which five mentioned this statement. However, for the additives mentioned in those five MSDS no migration data could be found. As such, for these five MSDS it cannot be confirmed, whether the statement of the supplier is true or false.

2.1 Introduction

This chapter compares PVC to other plastic materials (for example PE, PP) used for similar purposes. This includes highlighting PVC-specific additive usage (e.g., plasticisers) required based on the PVC-type (for example plasticisers are not used for rigid PVC, but are used in high concentrations in flexible PVC) and the purpose (e.g., flame retardancy is needed for some PVC-pipes used in air ventilation). First, this will be done in a general step, explaining the use and function of each additive. Second, a mapping of the applied PVC additives has been undertaken and the results discussed. In section 2.5 the migration potential of the additives are explained. This includes the circumstances in which migration can occur as well as the special circumstances in the case of PVC. The migration plays an important role in the classification of PVC under CLP, as many of the additives are hazardous. If the hazardous additives migrate out of the polymer these may pose a risk to humans and the environment. As such in the last chapter the classifications of different PVC mixtures are analysed and conclusions drawn.

In general, plastics can be divided into three groups according to their thermal-mechanical-behaviour: thermoplastics, elastomers, and thermosets.

- Thermosets are plastics that cannot be deformed in shape by heating or other means after they have solidified, as they contain highly crosslinked polymer chains.
- Elastomers also contain cross links, but they are longer and further apart giving the polymer a certain elasticity. They can also not be moulded by heat.
- To shape a plastic, thermoplastics are used, as these contain non connected polymer chains, which gives this group of polymers the ability to be reshaped by applying heat. Examples of thermoplastics include polyethylene (PE), polypropylene (PP), polystyrene (PS), and polyvinyl chloride (PVC). The weight of common thermoplastics ranges from 20,000 to 500,000 Dalton (Rutledge 2018). Thermosets have a much higher molecular weight due to their crosslinking.

PVC is a synthetic thermoplastic made through the polymerization of vinyl chloride. It was discovered over one hundred years ago. PVC has the widest range of processing and applications of all thermoplastics, mainly because many functional additives have been developed enabling compounds with very high processing and application performance (Elsner, Eyerer, and Hirth 2012).

2.2 Production of PVC

Vinyl chloride is mainly produced by one- or two-step addition of chlorine to ethylene. Polymerisation takes place by a free-radical route facilitated via a suspension polymerisation. The degree of polymerization or the molar mass is influenced by the amount of initiator added, surfactant concentration and the stirring speed.

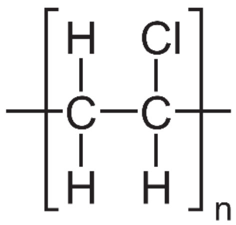


Figure 2. Repeating unit of PVC

The raw materials of PVC are obtained from crude oil (43 %) and salt (NaCl, 57 %) (Elsner, Eyerer, and Hirth 2012). Hence for the production of one kg PVC much less oil is needed in comparison to PE or PP. PVC offers the advantage to the manufacturer that the production is relatively cheap. The production process consists of 3 stages: Monomer synthesis, polymerisation, and preparation of a mixture needed to obtain other products.

PVC is manufactured on an industrial scale using three polymerisation processes: suspension (PVC-S), emulsion (PVC-E), and bulk (mass) methods (M-PVC) (Saeki and Emura 2002). These manufacturing processes influence the external appearance, such as grain size, grain shape and the absorption of plasticisers. The vast majority (ca. 90 %) of the PVC produced is a polymer obtained via the suspension method. Only 6-8 % of the plastic is obtained via emulsion polymerisation and is most often processed as a paste (Oxoplast 2015).
An alternative to production from ethylene is the production from acetylene, discussed further below.

Production from Ethylene

PVC is manufactured from two primary raw materials, ethylene and chlorine. Ethylene is a petrochemical product, attained via cracking of naptha, using crude oil as the raw material. The cracking process gives a number of by-products, used in a variety of other applications.

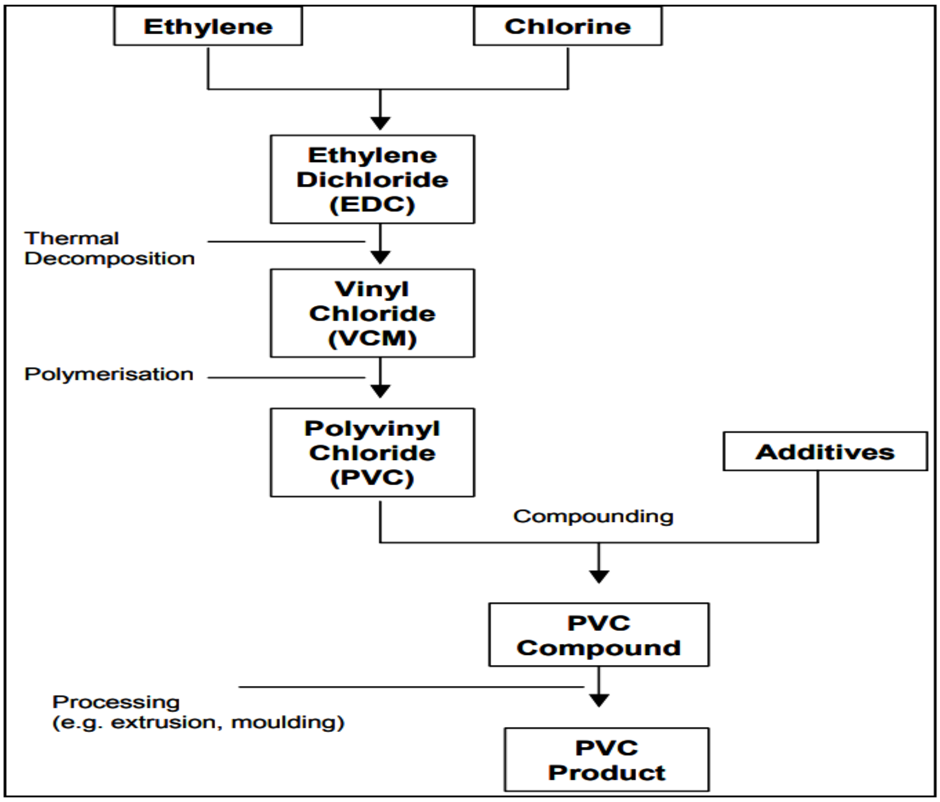
Chlorine is provided by the chlor-alkali industry, either via the electrolysis of industrial grade salt or as a by-product of other integrated processes (J. P. Pascault 2012). No sources have been identified which indicate the extent to which processes producing chlorine as a by-product can be integrated into the PVC production process for increased manufacturing efficiency.

Ethylene and chlorine contribute 43% and 57% to the polymer weight respectively. Ethylene and chlorine react to form 1,2-dichloroethane (EDC), which at high temperatures decompose to form the vinyl chloride monomer (VCM) and hydrochloric acid. The hydrochloric acid is usually fed back into the process to react again with ethylene and form further EDC.

This process gives side products of chloroethane, trichloromethane, tetrachloromethane, 1,1,2-trichloroethane and tetrachloroethane, as well as trace amounts of chlorinated aromatics, including traces of dioxins. These by-products constitute less than 2.5% of the reactants by mass. Various other solid waste chemicals form during the process in the form of carbon from process filters, calcium chloride from the dying process and sewage sludge.

VCM is polymerised via two techniques; suspension polymerization, which forms S-PVC, and emulsion polymerization, which forms E-PVC. The former is more commonly used and the difference between the two have little impact on the PVC performance over its life cycle (European Commission 2004).

Figure 3: Overview of PVC product manufacture



Production from Acetylene

An alternative method of PVC manufacture involves acetylene and anhydrous hydrogen chloride gas as precursors. The reaction takes place in a single step to form VCM with a high yield and selectivity. As acetylene is a flammable gas, transport of the precursor is comparatively costly and hazardous. Hence, only China uses this method extensively, due to proximity to large coal reserves, from which acetylene is produced (IHS Markit 2021). A similar flow chart to that provided above (see Figure 3) is not available, but the reaction takes place in a single step.

2.3 Thermoplastic properties and processing

As the crystallinity is only about 5 %, PVC is essentially of amorphous structure. This stems from the mostly atactic structure with small amounts of syndiotactic segments, which can crystallise. Due to steric hinderance, the monomers are mainly arranged head-to-tail, meaning that there are chlorides on alternating carbon centres.

The chlorine atom in the chain has a very large atomic radius which shields the carbon-carbon bonds in the main chain. This makes the PVC macromolecule very resistant to chemicals and UV-radiation but also comparatively brittle. The presence of chloride groups gives the polymer very different properties compared to structurally related plastics like polypropylene. One example is the polar nature of the PVC based on the presence of chlorine atoms repeated within its backbone (as shown above in Figure 2), which is the reason, that PVC can accept a much wider range of additives than other non-polar plastics like PE and PP (Leadbitter 2002) (for more information on the additives of PVC see chapter 2.3). Based on the molecular weight of chlorine, the density is also higher than structurally related plastics. The most important processes for thermoplastic processing of PVC are extrusion and calendering. The relevance of injection moulding, blow moulding, stretch blow moulding, pressing and sintering varies from country to country. The largest quantities of PVC are processed by extrusion. During extrusion, solid to viscous curable masses are continuously pressed trough a shaping die of the desired cross-section. The material premix, which is available as a dry blend, or the already compounded PVC granulate is melted in an extruder specially equipped for processing PVC and formed into the product. Calendering on the other hand, a process in which hot mass of a thermoplastic is fashioned into a continuous sheet by passage through a system of heated rolls (the calender), is also very important for PVC. Typical products are heavy foils, sheets, flooring and artificial leather.

The polymerisation processes has an influence on some of the properties of PVC. However, the chemical nature of additives has a greater effect on the properties of PVC than the production process itself. The most common additives types will be explained in the following. In chapter 2.4, a mapping of known PVC-additives by substance is given.

2.4 Additives used in PVC

Additives are used in almost every polymer and enhance its mechanical and chemical properties. This chapter discusses the general functionalities and advantages of the most common additives and goes into detail about the specific additives used in PVC.

2.4.1 Plastic additives

The following table gives a general overview of the most common additives used PVC and other polymers.

Table 1: Types of additives used in PVC and other polymers

Type of additive	Description
Plasticisers	• Incorporated into a substance or material (usually thermoplastic or elastomer) to increase its flexibility, workability, or distensibility • Plasticisers are added to PVC to soften up the matrix. Hence there are two main types of PVC, rigid PVC (unplasticized, uPVC) and flexible PVC (plasticized)[6] • Via the addition of plasticisers, completely new products can be developed for example floor cover, sheets, flexible tubes, synthetic leather, cable- and wire- coating and shoe soles • The plasticiser content for flexible PVC can be up to 50 wt%
Stabilisers	• The processing of the thermoplastic PVC e.g. by extrusion or injection moulding, requires temperatures of up to 160 to 220 °C • Unstabilised PVC, however, releases hydrogen chloride (HCl) gas at temperatures well below the processing temperature (60° C) • Insufficient stabilisation of PVC is noticeable in a discolouration of the material, the splitting off of hydrogen chloride and a deterioration of the mechanical and chemical properties • They are mostly heavy metal based compounds such as Organotin and lead stabilisers
Co-stabilisers	• Co-stabilisers often act synergistically with other stabilisers • They interrupt certain chains of degradation reactions, inert defect sites or capture intermediate products of the stabilisation or degradation reactions that have a catalytic effect on the decomposition • Co-stabilisers are mostly organic substances such as organophosphites, polyols, 1,3-diketones, epoxy compounds, antioxidants or UV stabilisers
Acid scavangers	• Acid scavangers used in PVC bind the hydrogen chloride which is formed during decomposition processes • They are often composed of mineral substances such as hydrotalcite and hydrocalumite, zeolites and other inorganic basic compounds
Lubricants	• Lubricants are added to adjust the rheological behaviour of the polymer during processing

[6] In North America also chlorinated polyvinyl chloride (CPVC) is a common sub type of PVC. It has been used extensively in fire sprinkle er systems and many industrial and process piping application and it´s usage is required in many standards. More information is available here (last accessed 03.02.2021)

Type of additive	Description
	• A distinction is made between internal (internal), chemically predominantly polar lubricants and external (external), predominantly non-polar lubricants
Flame retardants	• Flame retardants are applied to materials to prevent the start or slow the growth of fire • They work by reacting with and capturing free hydroxide radicals, which are formed during combustion processes • As PVC has a high limiting oxygen limit (see Table -2) and as such contains a natural flame retardancy, flame retardants are rarely used in PVC
Fillers for rigid and flexible PVC	• Fillers are added to a polymer to lower material costs, provide coloring, UV protection and lubrication • For both types of PVC calcium carbonate is the most important filler, as it servs as an ideal scavenger for acid cleavage and thus improves the thermostability of PVC (possibly saving on other stabilisers)
Impact modifiers	• Polymers like PP and PVC are sensitive to impact in the cold and as such impact modifiers are added to compounded materials to improve durability and toughness • Additionally, a number of other characteristics of the material such as optical and tensile properties, weatherability, processability, flammability, heat distortion and cost can also be improved by impact modifiers

2.4.1 Properties of plasticised and unplasticized PVC

All of the above additive types change the mechanical and electrical properties of PVC and have enabled its use in a wider set of applications. In Table 2-2 the mechanical and electrical properties of rigid and flexible PVC are listed. For comparison, the same properties of polyethylene and polypropylene are shown.

Table -2: Table containing the data collected on PVC and its subtypes and other plastics

Properties	Unit	Rigid PVC (PVC-U)	Flexible PVC (PVC-P)	Polyethylene (HD)	Polypropylene	Reference
Chemical Formula		$(C_2H_3Cl)_n$	$(C_2H_3Cl)_n$	$(C_2H_4)_n$	$(C_3H_6)_n$	
Density	g/cm^3	1.36	1.1 – 1.35	0.93 - 0.97	0.895 - 0.92	(Titov 1984) (PlasticsEurope 2021b)
Elastic modulus / Yield strength	MPa	1,450 – 3,600	10.0 – 24.8	600 - 1500	1,325 (average)	(Elsner, Eyerer, and Hirth 2012)
Max. operating temperature	°C	60 (without additives) 100 °C - 260	60 (without additives) 100 °C - 260	100 - 120	100 - 130	(Omnexus 2021b)
Volume Resistivity	Ω/cm	10^{16}	10^{16}	10^{15} - 10^{18}	10^{16} - 10^{18}	(Professional Pastics 2021)
Chemical Resistance		Acids, salts, bases, fats, alcohols, sewage	As PVC-U but also resistant to chemical solvents	Acids, bases, fats, alcohols, No oxidizing agents	Acids, bases, fats, alcohols, No oxidizing agents	(Engineering Toolbox 2021b)
Linear expansion coefficient	10^{-6} m/ m °C	50.4	n/a	120.0	72 - 90	(Engineering Toolbox 2021c, 2021a)
Limiting oxygen index (LOI)		0.42	0.42	0.18	0.18	(Van Krevelen and Te Nijenhuis 2009)
Thermal conductivity	W/mK	0.12 – 0.25	0.12 – 0.25	0.45 - 0.52	0.1 – 0.22	(C-Therm 2019)

This data indicates:

- PVC without heat stabilisers can only be used up to 60 °C. When heat stabilisers are added rigid PVC can be used in a max. operating temperature of 100-260 °C. In comparison, PE and PP, without heat stabilisers, can be used in temperatures of 110-160 °C.
- According to Altstädt (2021) the most important characteristic of a material is its price. A rigid component (for example a waste water tube) made of PE is 3 to 4 times more expensive in comparison to PVC when comparing their material price. PP is also more expensive than PVC.
- Pure PVC is a white, brittle solid powder. With the help of plasticisers, the material gets softer and more flexible and can then be used as in many articles ranging from window frames (unplasticized) to vinyl gloves (up to 50 wt%). Other plastics like PE or PP typically do not include plasticiser at all.

- PVC shows good heat insulation properties, but because of its polar nature caused by the chlorine, its electrical insulating property is inferior to PE and PP. For example, PE has been widely applied for manufacturing cables at the medium and high voltage range due to its high electrical strength. In general, thermal conductivities are significant at low filler loadings due to the polymer matrix's thermally conductive fillers' disjointing (Haque et al. 2021).
- Unplasticized PVC has a relatively high hardness and better mechanical properties compared to PP or PE. The mechanical properties increase with the molecular weight but decrease with increasing temperature. The mechanical properties of rigid PVC (uPVC) are very good; the elastic modulus can reach 1,500–3,000 MPa. The elastic limit of flexible PVC is 1.5–15 MPa. The good mechanical properties make PVC suitable for mechanical recycling (see also chapter 5.5)
- PVC is chemically resistant to acids, salts, bases, fats, and alcohols, making it resistant to the corrosive effects of sewage, which is why it is so extensively utilized in sewer piping systems. PVC is a very cheap material for outdoor applications like windows, etc.
- The higher expansion coefficients for plastic materials in comparison to metals or ceramics makes them extremely sensitive to temperature changes. The linear expansion coefficient of rigid PVC is relatively small, so that it is rather insensitive to temperature changes compared to PP and PE.
- The limiting oxygen index (LOI) is defined as the minimum fraction of oxygen in a mixture of oxygen and nitrogen that will just support combustion (after ignition) (air has 20% content of oxygen). PVC has good flame retardancy with the LOI being up to 42% or more. In PE and PP flame retardants such as e.g. tetrabromobisphenol A have to be used to achieve the same flame retardancy as in PVC (Polcher et al. 2020). However in some cases flame retardants are also used in PVC (see above) (Polcher et al. 2020).

2.4.2 Mapping of PVC-additives

As highlighted before, based on its final usage in products PVC needs a variety of different additives. This includes both former and currently used additives have been mapped, where data are available. The mapping of the PVC-additives has been done according to their main usage / type of additive (e.g. plasticiser).
The mapped additives have been mainly taken from the "Mapping Exercise – Plastic additives initiative" (PLASI) from ECHA in cooperation with the industry. However, it is important to point out the limitations of PLASI-initiative "*[…] the overview should not be considered complete and final.*" (p. 9 in the Supplementary Information 15.02.2019 (ECHA 2019)). In addition, the PLASI-initiative is based on specific further preconditions[7], for example, that the lists cover only substances which have been registered under REACH above 100 t/a.

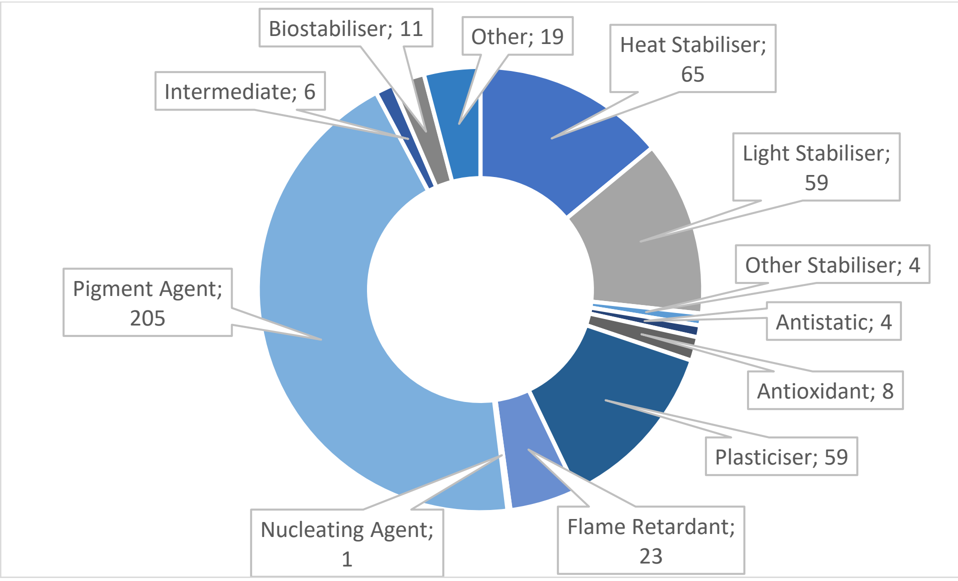
Additional information sources include a recent study from the German Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (BMU) (Polcher et al. 2020), which itself covers wider sources, for example a report from the Danish EPA on "Hazardous substances in plastic" (Danish Environmental Protection Agency 2014) and a report from the Nordic Council of Ministers on "Hazardous substances in plastics – ways to increase recycling" (Nordic Council 2017). A technical book devoted to plastic additives has been screened for any relevant information (Maier and Schiller 2016)[8]. This source mainly included pigments, which were added to the excel sheet if applicable. It is noteworthy, that in the table no distinction between former uses and current uses is done.

[7] Substances registered > 100 t/a under REACH; ertain functional additives & pigments; Substances identified based on non-confidential data & info from industry groups. Source: Communication by ECHA (ECHA presentation_PLASI overview.pptx).
[8] The most current version of the book from 2016 is only available in German.

The excel sheet with all found additives can be found in in Annex 2. In total around 370 additives with the following functions were identified (see also Figure 4):

- Heat Stabiliser (65)
- Light Stabiliser (59)
- Other Stabiliser (4)
- Antistatic (4)
- Antioxidant (8)
- Plasticiser (59)
- Flame Retardant (23)
- Nucleating Agents, substances that induce the formation of polymer crystals (1)
- Pigment Agent (205)
- Intermediate (6)
- Biostabiliser, Antimicrobials (11)
- Other (19), this includes for example viscosity modifiers, lubricants, slip promoter or generally "other stabiliser" for which the stabilisation process is unclear

Figure 4. Number of found additive types. Source: Ramboll



A substance can have more than one function, such as cadmium (and its compounds) which can be a heat and light stabiliser as well as a pigment. In addition, in this part of the study no market data, meaning portfolio of individual companies have been screened. Based on this, the here stated number of PVC additives should be seen as an indicative assumption.

It should be noted that plastics in general can contain non-intentionally added substances (NIAS), which are not listed as additives. Such substances include residual monomers or reaction chemicals such as solvent and surfactants. Additionally, they can be formed during the break-down of other additive and the polymer chain itself as well as through the reaction of break-down products. Contaminants which enter the polymer during production of the service life of the products are also NIAS. Examples of NIAS in PVC are nonylphenol, which is be formed during the degradation of tris(nonylphenol) phosphite, an antioxidant used in PVC. A different NIAS in PVC is semicarbazide, which is a break down product of azodicarbonamide, which is used in PVC as a blowing and pigment agent. This use is however prohibited in the EU since 2005 (Geueke 2018; ILSI 2016). These substances are not further elaborated on in this report, due to lack of data regarding their migration.

Additionally, a recent publication by Wiesinger, Wang, and Hellweg (2021) systematically investigated plastic monomers, additives, and processing aids on the global market based on a review of 63 industrial, scientific, and regulatory data sources. This includes substances used in PVC.
In total, the authors identified more than 10,000 relevant substances and categorised them based on substance types, use patterns, and hazard classifications wherever possible. Over 2,400 substances were identified as substances of potential concern as they meet one or more of the persistence, bioaccumulation, and toxicity criteria in the European Union. The authors of this study have been interviewed during this project to obtain PVC-specific data. In the following the data found by Wiesinger et al will be compared with the data found in the present study.

In total around 860 individual substances have been found by Wiesinger, Wang, and Hellweg that are used either in PVC specifically or denoted as being used in all types of plastics. The vast majority (>770 substances) are only used in PVC. The higher number of relevant PVC additives found in their study compared to present one is most likely explained by the fact that Wiesinger, Wang, and Hellweg analysed more information sources such as industry data. Additionally, the authors also included those additives that are used in all plastics, whereas in this project only additives specifically used in PVC have been considered. Lastly also more types of additives have been included in the study of Wiesinger, Wang, and Hellweg compared to this one (such as e.g., lubricants, fillers, blowing agents etc., compare also with Figure 3).

However, based on the mapping as undertaken in this project and also in general, trends based on the specific needs of PVC, are observable. The most important and commonly used types of additives for PVC will be discussed in the following. Whereby the results of this study and the results of Wang and Wiesinger et al. will be compared, and comments based on the cited publication regarding hazardousness will be included.

Pigments

The most common type of additive found in this study are the pigments with 205 individual substances. As plastics are often marketed in many different colours the use of pigments becomes essential. In the case of PVC however pigments are not the most important type of additive.

Many of the identified pigments are inorganic salts, minerals, and oxides, however, there are also organic based pigments. The advantage of organic based pigments is that the colour can be individually adjusted based on the molecular structure of the molecule, whereas inorganic materials cannot be further modified. A commonly used pigment is titanium dioxide which exhibits a white colour and is also used in other products as a colorant such as wall paint and sunscreen (Skocaj et al. 2011). It is classified as biologically inert (Skocaj et al. 2011) and has found a wide range of usages including PVC. Recently titanium dioxide has however been classified as a carcinogen category 2 via inhalation. There it serves a dual purpose: it gives the PVC a bright white colour and also protects it from harmful UV radiation. In the study by Wang and Wiesinger et al. in total 473 colorants used in PVC have been included.

Plasticisers

Plasticisers are the most popular type of plastic additive, which make materials more flexible and easier to process, and about 90% of them are used for PVC applications. They can be "internal" (chemically modifying the polymer or monomer) or "external" (i.e. low molecular substances). The mechanism of plasticisers action is explained in literature by several theories (for an overview see Marcilla and Beltrán (2012)). Based on modelling data, plasticisation of polyvinylchloride is explained by solvation of the polymer chains as well as an increase in the free volume caused by the introduction of plasticiser molecules (Chandola and Marathe 2008).

Most commonly PVC plasticisers are based on organic esters with long carbohydrate chains attached. In many cases these are based on aromatic dicarboxylic acids, the so-called phthalates. In recent years many of these have been found to have adverse effects on humans and the environment resulting in a shift towards other phthalate-based plasticisers (that are less hazardous) and non-phthalate plasticisers. However, phthalate plasticisers are still the most frequently used plasticisers around the world.

Many alternative plasticisers are based purely on aliphatic carbohydrate esters or long non-ester carbohydrate chains such as chlorinated paraffin waxes and medium chained chlorinated paraffins (MCCPs). Recently MCCPs that are UVCB (Unknown or Variable composition, Complex reaction products or Biological materials) substances consisting of more than or equal to 80% linear chloroalkanes with carbon chain lengths within the range from C14 to C17 were assessed as SVHC (substance of very high concern) and included into the candidate list for authorization in accordance to REACH due to their PBT (persistent, bioaccumulative and toxic) and vPvB (very persistent, very bioaccumulative) properties. Additionally, they are proposed to be restricted as a persistent organic pollutant under the Stockholm Convention.

In the present study 59 different plasticisers were identified which have been or are used in PVC. Of these substances 12 are listed as SVHCs. In the study by Wiesinger et al. 227 substances (including 31 SVHCs) were identified, which have been used as plasticisers in different kinds of plastics including PVC.

<u>Light stabilisers</u>
PVC can degrade through long term UV exposure, which is why many manufacturers add light stabilisers to their PVC mixture. UV light can induce the cleavage of hydrogen chloride from the molecule but can also break any other exposed chemical bond in the PVC leading to fragmentation and ultimately to the degradation of the molecule. Light stabilisers work by absorbing the high energy UV radiation via the excitation of an electron. The absorbed energy can then be dissipated via vibration, rotation or by emitting light of a lower wavelength (fluorescence). By doing so, the PVC matrix can be protected from the radiation and the lifetime of the products can be extended. Many light stabilisers are carbohydrate-based molecules, however, inorganic substances such as titanium dioxide and carbon black may also fulfil such a role. In total 59 different light stabilisers (including 2 SVHCs) have been found in the present study. In contrast to these relatively low numbers, Wang and Wiesinger et al. identified 269 light stabilisers (including 39 SVHCs) that are used in different kinds of plastics including PVC.

<u>Heat stabilisers</u>
As described in chapter 2.4.1, PVC releases hydrogen chloride when heated resulting in the loss of chemical and mechanical properties. This process is catalysed by the formed hydrogen chloride and is as such auto catalytic, meaning that once it has started it will continue without outside influence (Folarin and Sadiku 2011). The cleavage of hydrogen chloride leads to the formation of polyene sequences of conjugated double bonds and is accompanied by increasing discoloration (yellow). Further, strong thermal stress or the attempt to shape the material under high pressure to achieve a lower melt viscosity would not achieve the desired result. Autocatalytic processes, with further elimination of labile chlorine atoms - preferably at structural defects - drive a destructive chain reaction in the melt. In addition to the increasing discoloration above a certain chain length of the conjugated polyene sequences, cross-linking of the polymer units also sets in. This leads to a deterioration of the physical and mechanical properties.
As PVC is a thermoplastic polymer (meaning it can be reformed when heated) it can be shaped into any desirable form. Since high temperatures are needed to achieve this, heat stabilisers are added

to the PVC to prevent it from degrading. Their mechanism is based on the reaction of the stabiliser with the intermediately formed allylic chloride atoms preventing the auto catalytic reaction or based on the absorption of the formed hydrogen chloride reducing the propagation of the chain reaction (Folarin and Sadiku 2011). Most of PVC stabilisers identified are based on inorganic salts such as cadmium and lead, however, there are also tin and organophosphorus based heat stabilisers. In total 65 individual stabilisers could be identified. Wang and Wiesinger et al. identified 119 Heat stabilisers that are used in different kinds of plastics including PVC.

2.5 Detailed description of PVC-additives concerning their migration and bioavailability

This chapter focuses on the potential migration of the additives used in PVC as well as their bioavailability after leaving the polymer matrix. For this literature was gathered, analysed and a conclusion drawn.

2.5.1 Migration

Additives are essential for the various functions of PVC. Therefore, it is required that PVC additives remain in the plastics for a certain period of time. However, additives can migrate from any plastic products during their use and end of life phase, leading to potential risks to human health and the environment (ECHA 2019).

Poly(vinyl chloride) is a complex plastic system. Individual components of the PVC system, including residual vinyl chloride monomer (RVCM) and certain additives, may pose risks of harm to human health as most of these components are not covalently bound to the polymer matrix (Polcher et al. 2020; Wiesinger, Wang, and Hellweg 2021). For example, phthalate plasticisers can in certain conditions leave the polymer through migration, evaporation or extraction by liquids, as they are not covalently bound to the polymer.

During service, this loss may be problematic; it leads to (1) unwanted changes in the material properties (e.g., poorer mechanical properties) and (2) eventual contamination of the surrounding medium. After the service life the loss of additives can also lead to the contamination of the environment and subsequent health issues.

For the additive to leave the polymer matrix it first needs to travel to the surface of the polymer matrix. According to the information to the PLASI initiative, the diffusion of the substance in a plastic matrix indicates how fast the additive moves from the matrix itself towards the surface. The mechanism is described by the diffusion coefficient. The higher the coefficient, the faster the additive diffuses. The second mechanism is the partition between the surface of polymer matrix and another layer (the outside layer of the plastic e.g., air, water, sweat etc.), which is described by a partition coefficient which represents the ratio between the concentration in the plastic matrix at the surface and the concentration in the contact medium. For some routes (e.g., partition from plastic matrix to air), the transport speed of the substance from the polymer matrix to the contact medium also plays a role. The above mechanism is (largely) independent of the substance concentration in the matrix. However, the substance concentration plays a role in the estimation of the release-rate from plastics. A proportional relationship between release rate and concentration (i.e. half concentration resulting in half release rate) is valid for all release routes (ECHA 2019).

The real life extent of the migration, however, varies greatly depending on the additive, (effect of branching, molecular weight, end-group functionality, solubility in the polymer matrix and polydispersity) its initial concentration, and the type of PVC (rigid vs. flexible) (Augustsson and Henningsson 2011; Danish Environmental Protection Agency 2016; ECHA 2019). This is supported by the fact, that in landfills, research shows that both the nature and the content of the plasticiser influences the migration of the used PVC additives. Additionally, factors such as the shape of the object, the solubility of the additive in the outside medium (e.g., water, sweat etc.) and the temperature also play a factor in the migration. These factors are valid for the migration of additives in all types of thermoplastics and unbound additives (ECHA 2019).

Normally, manufacturers of high-value plastic materials and articles will seek to optimise the combination of these parameters to limit the release of the additives. Whether the amount of an additive released is of concern to human health and the environment depends on its toxicological and ecotoxicological properties and how it behaves in water, on the skin, and in the air after its release (ECHA 2019).

After the substance migrates out of the polymer matrix it may enter into organisms, the ability of which is called bioavailability. The more bioavailable a substance is, the more can be taken up by a living organism.

2.5.2 Bioavailability

According to the Guidance on the Application of the CLP Criteria[9] "Bioavailability is the rate and extent to which a substance can be taken up by an organism and is available for metabolism or interaction with biologically significant receptors. Bioavailability (biological availability) involves both release from a medium (if present) and absorption by an organism" as defined by the International Programme on Chemical Safety.

In general, a low molecular weight can be an indicator for a higher mobility and the ability to cross a biological membrane increasing the bioavailability (OECD 2019). In recent literature, however, it has been shown that plastics with a molecular weight above 1,000 Da can also release particles which may interact with animal cells. This shows, that high molecular weight substances may also be bioavailable (Lohmann et al. 2020).

2.5.3 Potential hazards arising from PVC

In order to assess the potential hazard arising from PVC additives a desktop research was performed for the substances identified in chapter 2.4.2. For the migration of the substances EU-regulations such as the Commission Regulation (EU) No 10/2011 of 14 January 2011 on plastic materials and articles intended to come into contact with food[10] and Directive 2009/48/EC of the European Parliament and of the Council of 18 June 2009 on the safety of toys[11] were screened for available specific migration limits (SML). The regulation on food contact material lists SML for individual substances as well as for substance groups. The SML are set out based on a worst-case scenario risk assessment performed by the European Food Safety Authority (EFSA) and as such do not represent the actual migration ability of the substance; it can however give a strong indication.

The directive on the safety of toys on the other hand only provides migration limits for metals as well as bisphenol A and phenol. For metal salts, the migration limits of the containing metals are listed, however, the actual migration of the specific salt may vary based on composition.
For bioavailability, the REACH registered substance factsheets were evaluated, as every additive above one tonne, is registered under REACH. Applicable information could be found for most compounds.

[9] https://echa.europa.eu/documents/10162/23036412/clp_en.pdf/58b5dc6d-ac2a-4910-9702-e9e1f5051cc5
[10] https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX%3A32011R0010
[11] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:02009L0048-20191118

If no information on the migration or bioavailability could be found in the above-named sources, a wider search based on PubMed, EuropePMC and Google Scholar as well as health and environmental ministries like the EFSA, US EPA and European Environmental ministries were screened for information. This included data on the chemical properties of the substances such as the sorption partition coefficient and the molecular weight.

The soil sorption coefficient measures the amount of the regarded chemical substances which can be adsorbed onto soil. If more substance adsorbs onto soil it may suggest a lower migration and bioavailability of the substance and a higher exposure to soil organisms (OECD 2019). If this data was found, it was also included into the excel sheet.

For some additives the release potential indicator was calculated in the PLASI initiative. It was derived from the chemical-physical properties of the substance, the concentration, and the polymer matrix. In the PLASI initiative the values are expressed on a logarithmic scale between 0 and -10 (where 0 indicates the highest release potential and -10 the lowest among all the additives). According to the disclaimer by ECHA, the release potential as such does not indicate any concern for uncontrolled risks. The indicator is meant to help prioritisation where more depth assessment may be needed. This data was not included in this project as it can be accessed online [12].

However, during this project, the not publicly available data of the PLASI-initiative have been provided by ECHA. According to this data several substances that are used in PVC have comparably high dermal and/or indoor air release factors. It has to be noted, that only substances used in flexible PVC are considered, in the PLASI data, as a worst case scenario of a high diffusivity matrix (flexible PVC) is taken as a basis for the calculation. In Annex 2.1, named "Dermal and indoor air release factors for substances used in flexible PVC (sorted smallest to largest by the sum of dermal and indoor air release)", the individual values for substances used in flexible PVC are shown.

In some cases, especially for more complex and less used compounds, no specific information on the migration and bioavailability could be found. All identified information was summarised in the excel sheet on additives accompanying this report (see Annex 2 for information on the additives. Due to the limited space, the migration and bioavailability data was excluded from this table).

Based on the data collected within the excel table and other sources (see following citations), some general aspects are observable. In the following, the found data is summarised for the relevant additive types separately. In many cases no real-life data on the migration of certain additives from PVC could be found and as such these additives are not discussed here.

Table 3: Found data on the migration of certain additives used in PVC

Additive	Data on migration
Pigments	• Organic pigments show low migration tendency by default due to their low solubility in water, organic solvents, and various kinds of other media [13] • As such a qualitative method is sufficient to characterise the release potential of organic pigments • The assignment to low release rates should however be confirmed by representative data

[12] https://echa.europa.eu/de/comparing-relative-release-potential

[13] https://www.chemicalbook.com/ProductCatalog_EN/161211.htm#:~:text=Organic%20pigment%20is%20a%20class,and%20various%20kinds%20of%20media.

Additive	Data on migration
	• For inorganic pigments this pre assumption is not applicable, as inorganic molecules consist of charged molecules/atoms and thus behave differently than organic molecules • Many inorganic pigments however show a very low solubility in their registration dossier and thus also have a relatively low potential for release (ECHA 2019). This data should however also be experimentally confirmed
Plasticisers[14]	• Migration of plasticisers in PVC has been thoroughly researched • The migration is dependent on the molecular size of the regarded molecule as well as the properties of the plastic it is in. The low molecular weight phthalates are relatively small molecules which allows them to travel through the amorphous phase of the plastic matrix, as they need less space to do so. Additionally, the phthalates also soften the PVC matrix making it easier for the additives to migrate out of the polymer compared to rigid PVC. The higher molecular weight phthalates can still migrate, however, due to their higher molecular weight they require more space and thus have a lower migration rate compared to the low molecular phthalates (Danish Environmental Protection Agency 2016) • The migration of phthalates from flexible PVC has been confirmed by various studies. In these studies, landfill leachates were analysed and multiple phthalates such as DEHP, DMP, DEP, DBP and DIBP could be found. DEHP was the most commonly found plasticiser and was also detected in a landfill which was decommissioned in 1976 indicating the slow degradation of phthalates and the potential for phthalates to be released over a long period of time (Ciacci, Passarini, and Vassura 2017; Eggen, Moeder, and Arukwe 2010; Kalmykova et al. 2013; Scheirs 2003; Wowkonowicz and Kijeńska 2017) • The migration of plasticisers used in PVC into food has also been confirmed (Hahladakis et al. 2018) and some phthalates are already restricted for the use as plasticisers in food packaging material in accordance with Regulation (EU) 10/2011[15] • A danish report analysed the migration of phthalates, which concluded that the realistic migration rates cannot be determined due to the different methods used in migration studies. The migration rates for five phthalates (DEHP, DINP, DIBP, DBP, BBP) were estimated by "head over heels" method (HOH)[16] which was recommended for the estimation of migration rates until further evidence is available. The determined migration rates for all five phthalates are in the same range (Danish Environmental Protection Agency 2016) • The migration of plasticisers from PVC (flooring) has also been confirmed by the Norwegian Environment Agency (2020), whereby DEHP was found to have the highest migration rate compared to DINP and DINCH, likely due to its small molecule size and higher volatility • The PLASI initiative of the ECHA developed a screening method based on the comparison of relative release potential, to identify plastic additives that should be prioritised for assessment in the light

[14] The reason for phthalates popularity is the combination of good properties and low price (Augustsson and Henningsson 2011). When evaluating phthalates toxicologically, a distinction must be made between low molecular weight (LMW) phthalates (DEHP, DBP, etc.) and higher molecular weight phthalates (DINP, DIDP, DPHP, etc.). LMW-phthalates are believed to be problematic for human health because they are suspected of acting like hormones. Under REACH LMW-phthalates like DEHP, BBP and DBP have been included in Annex XIV to REACH in February 2011 (entries 4, 5 and 6, respectively), and DIBP in February 2012 (entry 7) following their listing on the Candidate List due to their toxic for reproduction properties (category 1B). Since then, DEHP has been identified as a SVHC in accordance with Article 57(f) of REACH due to its endocrine disrupting properties for the environment and all four phthalates have been identified as SVHCs in accordance with Article 57(f) of REACH due to their endocrine disrupting properties for human health. The Candidate List has been amended in order to reflect these additional intrinsic properties of the four phthalates

[15] COMMISSION REGULATION (EU) No 10/2011 on plastic materials and articles intended to come into contact with food (consolidated version of 23/09/2020) https://eur-lex.europa.eu/eli/reg/2011/10/2020-09-23

[16] In the "head over heels" method a roughly pacifier nipple sized part of the sample is placed into a solution of saliva stimulant and tumbled at a rate of 60 rounds per minute for 20 minutes at 37 °C. This process is then repeated again with fresh stimulant, the extracts combined and subsequently analysed.

Additive	Data on migration
	of available hazard data. Multiple non-phthalate plasticisers have been identified to have a high release potential and should be investigated further
Heat and light stabilisers	• Metal based stabilisers (e.g., cadmium and lead compounds) oftentimes serve a dual function of being a heat and light stabiliser • The usage of some of these compounds is restricted under REACH (cadmium)[17] or a restriction process is currently on-going (lead). Since 2016, the PVC industry in the EU has voluntarily abstained from the use of lead stabilisers[18] • The migration of lead, cadmium and zinc stabilisers from recycled plasticised and unplasticized PVC has been investigated and the results show, as expected, higher migration rates from plasticised than unplasticized PVC, as the diffusion coefficients of the metal compounds are higher. Additionally, metal based organic stabilisers (e.g., lead stearate) diffuse faster than inorganic ones (e.g., oxides). The migration of lead into sweat was found to be significantly higher than into water or saliva. The authors attribute this to the fact that the present lactic acid and/or ammonium hydroxide components in the sweat can coordinate the Pb ions and thus increase the solubility of the lead containing stabilisers. A higher solubility in the outside medium leads to a fast leaching of the substance (Mercea et al. 2018)

2.5.4 Conclusion on the migration of additives in PVC

The migration of additives in PVC and other polymers depends on many factors and cannot be generally stated. An important factor is the molecular weight of the additive, as molecules with a low molecular weight require less space and as such travel more freely through the amorphous phase of the plastic matrix (Danish Environmental Protection Agency 2016). The larger the molecule the harder this process becomes and as such the migration is lower for larger molecules. Additionally, the solubility of the additive in the polymer matrix and the composition of the polymer matrix (flexible vs. rigid) play an important role. If the additive is more soluble and the polymer matrix is soft, the additive can migrate more freely, which has been proven for certain metal compounds (see the chapter on Heat and light stabilisers). Lastly, if the additive has a low solubility in the medium outside the polymer matrix e.g., water, saliva, sweat etc. the migration is hindered (ECHA 2019).

The available data indicate that scientific data on migration levels of additives is sparse and that "real-life" migration datasets are missing. This issue is also noted in a recent report by the Norwegian Environment Agency (2020) and has been confirmed by stakeholders (Wiesinger and Wang Interview) that have been contacted during this project.

The migration has however been confirmed for certain substances such as low weight organic additives (for example phthalates) and some metals (like lead). In the case of migration of substances from PVC, a differentiation needs to be made between plasticized and unplasticised PVC, whereby in plasticized PVC for some substances higher diffusion coefficients have been measured (Mercea et al., 2018). Based on this fact, and also considering the high concentration of plasticiser, it can be concluded that the additives in plasticised PVC have a higher potential to migrate out of the polymer matrix compared to unplasticized PVC. Additionally it has been shown, that metal based compounds such as heat and light stabilisers migrate faster, if the outside medium is sweat instead of water (Mercea et al. 2018). Pigments on the other hand have been shown to have a very low

[17] See here the Conditions of restriction for cadmium: https://echa.europa.eu/documents/10162/3bfef8a3-8c97-4d85-ae0b-ac6827de49a9 . Last accessed 14.04.2021

[18] https://www.kft.de/en/eu-prohibits-lead-from-pvc-products/ Last accessed 30.11.2021

tendency to migrate, as they are mostly insoluble in organic as well as inorganic solvents. Concrete data is however missing.

2.5.5 Actions to address data gaps

The Norwegian Environment Agency report noted above, notes that a study on migration data for additives in various different plastics is envisioned. These data could then be then combined with the release rate model of the PLASI initiate and in the end used for further regulatory work, e.g. under REACH and CLP (Norwegian Environment Agency 2020).

The data gathered by the PLASI initiative indicatethat some PVC additives show comparably high release potential, whereby this potential as such does not indicate any concern for uncontrolled risks, but can help prioritisation where more in depth assessment may be needed. Such data can be used to set out priority substances when commissioning a new study.

2.6 Classifications of different mixtures of PVC including its additives

This chapter will explore the classification of PVC mixtures. For this material safety data sheets of PVC will be analysed as to their classification and assessed whether their statements can be confirmed with the found scientific data.

According to the Classification, Labelling and Packaging Regulation (CLP Regulation)[19] every manufacturer, importer and downstream user shall classify substances or mixtures in accordance with the CLP regulation before placing them on the market. For mixtures, concentration limits have been set according to article 10 for substances based on their hazard class above which the presence of that substance in another substance or in a mixture as an identified impurity, additive or individual constituent leads to the classification of the substance or mixture as hazardous. These are listed in Annex I of the CLP regulation and range from <0.1-10 % depending on the hazard classification of the substance.

In the case of PVC, many additives are added to the polymer, which results in it being classified as a mixture. Many of these additives are used in concentrations above the limits set out in Annex I of the CLP-Regulation, which means that the PVC would need to be classified as hazardous if the additives are hazardous as well. A good indicator to assess whether a substance may be hazardous is to check the bioavailability of the substance, as migration is not mentioned in the CLP regulation (see chapter 2.5).

To have an effect on a biological or environmental system the substance must have some degree of bioavailability. It follows from this that a substance or mixture does not need to be classified as hazardous if it can be shown through experimental data that the substance or mixture is not biologically available. This is reflected in Article 12(b) of the CLP Regulation where it is stated, that if "*conclusive scientific experimental data show that the substance or mixture is not biologically available and those data have been ascertained to be adequate and reliable*" they should be taken into account for the purposes of classification. This argument is often used to not classify a mixture as hazardous, the rationale is that certain additives are not bioavailable because they are trapped in the polymer matrix and their migration out of it is negligible. However, for this argument the supplier needs to provide conclusive scientific experimental data showing that the substance does not migrate out of the polymer matrix.

As stated in chapter 2.5 the migration has been confirmed for certain plasticisers and metals such as lead, for which the argument of being trapped in the polymer matrix is invalid (Augustsson and Henningsson 2011; Danish Environmental Protection Agency 2016; Mercea et al. 2018; Norwegian

[19] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A32008R1272

Environment Agency 2020). The vast majority of additives are not covalently bound to the polymer chains and can as such potentially migrate out of the mixture (Polcher et al. 2020; Wiesinger, Wang, and Hellweg 2021).

It was assessed whether the suppliers of PVC mixtures take into consideration the concentrations and migration potentials of hazardous additives for the classification, in accordance with the CLP regulation.

ECHA does not have a database on mixtures as the classification and labelling information on notified and registered substances received from manufacturers and importers (C&L inventory) only applies to substances, but not mixtures. Therefore, material safety data sheets (MSDS) of PVC mixtures were screened in order to obtain the classification information of mixtures.

According to Article 31 of the REACH regulation the supplier of a substance or mixture shall provide the recipient of the substance or mixture with a safety data sheet in accordance with the CLP Regulation. As only the recipient is entitled to such a data sheet, only a limited number of relevant MSDS could be accessed. Most of the identified MSDS were from suppliers in the USA which did not always mention REACH. Additionally, several stakeholders were contacted for information on CLP-classifications, however, no answers on this have been received.
The found MSDS were screened for relevant information including mentioning the REACH regulation, listing of additives with concentrations, the listing of the additive's classification and the statement that the additives are bound to the polymer matrix.

In total 24 MSDS were found from various companies, most of which operate in the USA. In general, it was often unclear whether the MSDS referred to the pure PVC obtained after synthesis or a commercial PVC mixture including additives. Out of the 24 MSDS only 10 listed additives. From these 10 MSDS, 5 mentioned, that the additives are encapsulated in the polymer matrix and migration is negligible. The following table shows the additives from the MSDS which stated that the additives are encapsulated by the polymer matrix. All shown data stems from MSDS published for products sold in the United States. No European MSDS with this statement could be found over the course of this study. For most of the mentioned additives no data on the migration is available and as such no assessment was made. However, in some cases assumptions based on the findings in this report are included.

Table 4: Found MSDS of PVC mixtures which state, that the additives are encapsulated in the polymer matrix and their named additives

Country	Date	Listed additives (concentration)	Classification according to CLP	Assessment whether the additive is encapsulated in the PVC matrix or not
USA	04.01.2015	Titanium dioxide (5-10%)	Carc. 2	
		Calcium carbonate (1-5%)	None	
USA	-	Tin-maleate (either 2.5%)	N.a	
		Tin-mercaptide (either 2.5%)	N.a	
USA	10.02.2005	Organotin (<5%)	N.a	

Country	Date	Listed additives (concentration)	Classification according to CLP	Assessment whether the additive is encapsulated in the PVC matrix or not
		Calcium-zinc (<5%)	N.a	
		Calcium Carbonate	None	
		Carbon black	None	
		Titanium dioxide	Carc. 2	
		Antimony trioxide	Carc. 2	
		Arsenic compounds	N.a	
		Chromium compounds	N.a	
		Barium compounds (0-10%)	N.a	
		Zinc compounds (0-10%)	N.a	
USA	19.06.2015	Dibutyltin bis (2-ethylhexyl mercaptoacetate) (<2%)	Acute Tox. 4 Skin Sens. 1 Eye Irrit. 2 Repr. 1B STOT RE 1	
USA	15.03.2015	Plasticiser (0-60%)	N.a	The type of plasticiser is not stated, however phthalates have been proven to migrate and many have been identified as SVHC. The migration of additives from plasticized PVC is higher than from rigid PVC
		Inert filters (0-50%)	N.a	
		Impact modifiers (0-50%)	N.a	
		Flame retardants (0-30%)	N.a	
		Process aids (0-25%)	N.a	
		Lubricants (0-20%)	N.a	
		Colorant (0-15%)	N.a	The type of colorant is not stated, however, pigments in general have a low potential for migration.
		Heat stabiliser (0-10%)	N.a	The type of heat stabiliser is not stated, however, some metal and

Country	Date	Listed additives (concentration)	Classification according to CLP	Assessment whether the additive is encapsulated in the PVC matrix or not
				organic based stabilisers have been proven to migrate. Additionally, the migration from plasticized PVC has been proven to be higher than from rigid PVC
		Vinyl chloride (<0.0005%)	Press. Gas Flam. Gas 1 Carc. 1A	Has been proven to migrate

Many of the identified MSDS do not state what compounds was used and instead only name their function (e.g., flame retardant). In many cases they also do not list the hazard classification of the additives. An example of a statement that additives are bound to the polymer matrix can be found below.

Section 2. Hazards identification

This mixture has not been evaluated as a whole for health effects. All ingredients are bound in a PVC polymer matrix and potential for hazardous exposure as shipped is minimal. PVC resin is manufactured from Vinyl Chloride Monomer (VCM). PVC resin manufacturers take special efforts to strip residual VCM from their resins. Residual VCM in the resin is typically below 8.5 ppm. However, VCM is a known carcinogen. The end-user (fabricator) should take necessary precautions (mechanical ventilation, local exhaust, respiratory protection, etc.) to protect employees from exposure to any vapors or dusts that may be released during heating or fabrication. See Sections 8 and 11 for special precautions.After handling, always wash hands thoroughly with soap and water.

OSHA/HCS status	:	While this material is not considered hazardous by the OSHA Hazard Communication Standard (29 CFR 1910.1200), this SDS contains valuable information critical to the safe handling and proper use of the product. This SDS should be retained and available for employees and other users of this product.
Classification of the substance or mixture	:	Not classified.

Figure 5. Example of the statement, that the additives are bound to the polymer matrix. Taken from (PolyOne 2018)

According to the CLP Regulation every manufacturer, importer and downstream user shall classify substances or mixtures before placing them on the market.
Based on the data gathered within this working package it can be concluded that very often no information has been given on the specific substance or its hazard classification. However many of the found MSDS are for products sold in America and as such do not need to comply with CLP and REACH Regulations.

In several cases, it was stated that "all ingredients are bound in a PVC polymer matrix" and no hazard classification of the PVC mixture was given. Chemically, this is inaccurate, as none of the identified additives is chemically bound to the PVC (meaning that a covalent bound is formed). More importantly, data gathered in section 2.5 of this report shows that certain additives migrate and become bioavailable. However often concrete data for a specific additive in a specific polymer is missing and as such no assessment whether or not the statement is true can be made. It is the duty of the supplier to provide the data that proves, that the additive is not bioavailable, however, this information does not need to be presented in the MSDS.

It should be noted that the assessed data is limited to publicly available datasheets (or other types of information).

3. GENERAL MARKET ANALYSIS

Key Messages

European Production

- According to Eurostat, approximately 6 million tonnes of primary PVC were manufactured in 2019 in the EU27. Of this 4.9 million tonnes were unmixed non-plasticised PVC, 0.4 million tonnes were non-plasticised PVC and 0.8 million tonnes were plasticised PVC mixed with other substances.
- Germany and France are the largest manufacturers by volume of these types of PVC within the EU27, together accounting for approximately 2.7 million tonnes, mostly unmixed non plasticised PVC.
- The total volumes of primary placed on the EU27 market, based on EU27 production, plus imports, less exports, was around 5 million tonnes per year. The largest markets are Germany, France Italy, Spain and Poland.

Trends in European Production (2008-2019)

- Production volumes of primary unmixed PVC declined in this period, with a decrease of 26% between 2008-2012, and more gradual decline thereafter. Production volumes of primary plasticised and non-plasticised PVC have been more stable since 2008.

European Market Value

- The market value for the above volumes of primary unmixed, plasticised and non-plasticised PVC produced in the EU27 was just under €5 billion in 2019.
- Market values for processed PVC have been estimated also with Eurostat data. This suggests EU27 production values of just under €10 billion in 2019.

European demand by sector

- Data on EU PVC use in different sectors is varied in quality and granularity. Different sector classification are used depending on the source. Overall the largest volumes of use are in construction (pipes and fittings, window frames, medical and healthcare applications, packaging, and cables for electronics were the largest users based on 2020 data from ECVM.

Global Production

- Less detailed information has been found on global PVC production, which was estimated at 55 million tonnes in 2016. Capacity data indicates that China accounted for almost half of this volume. PVC demand appeared to be somewhat lower than this, in the order of 45 million tonnes in 2018 but precise figures should be treated with caution.

- Historical data reveal that total global production has steadily increased in recent years, driven by increasing production capacity in China. More recent market analysis (2020) state that China is expected to remain the largest global producer and consumer of PVC for the foreseeable future.

Global demand by sector

- Data on global PVC use in different sectors is varied in quality and granularity, with different sector classification used, but these reveal that the construction sector is the dominant consumer of PVC. Automotive components, electrical cabling, packaging and flooring are each estimated to be using smaller but still sizable volumes.

Waste and Recycling Data

- Data on PVC waste in the EU are limited, much information is dated and data occasionally conflict. But it is estimated that between 2.5 million (2018) and up to 4.1 million tonnes (1999) of PVC waste is generated in the EU annually.
- Data on global PVC waste is even more uncertain, estimated in the order of 16 million tonnes (2015).
- There is a significant lack of reliable data on volumes of PVC waste disposed of via landfill, recycling and incineration, but disposal routes are known to vary for different PVC applications. Recent estimates from a report commissioned by VinylPlus (Conversio Market & Strategy GmbH 2021) on PVC waste in the EU27 plus the UK, Switzerland and Norway suggest that approximately 35% of PVC waste is recycled, 46% is incinerated with energy recovery, and 19% is landfilled or incinerated without energy recovery.
- VinylPlus data indicates that EU PVC recycling rates are increasing in volume, with the majority of recycled volumes comprising window frames and related products.

Indicative projections to 2050

- Pre pandemic, the global PVC market was expected to continue to grow at the rate of 3.5% 2016-2026. China was expected to account for the majority of this growth.
- Simple market projections have been derived, based on extrapolating historical data. All such forecasts are associated with assumptions and uncertainties. The economic disruption associated with the COVID-19 pandemic amplifies these.
- These projections suggests that PVC production in the EU27 could either stabilise – which would be a continuation of trend between c.2013-2019 – or continue to decline (following the overall trend between 2008-2019. Offsetting this, volumes of primary and processed PVC could continue to increase.
- Volumes of annual PVC waste arising are subject to great uncertainty given the number of factors affecting this, which include policy packages aimed to decrease waste from all sources. But a plausible scenario would be that per capita volumes of PVC waste could peak around the mid 2020's and decrease thereafter in line with forecast demographic trends.

3.1 Introduction

This section provides an overview of the PVC market. We focus on the EU27, but include data for the UK where available. Where this detail isn't available or is only available prior the United Kingdom's departure from the EU, the section presents data for the EU28. The European PVC market is also compared to the market in rest of the world (RoW).

- The section provides an analysis of manufacturing volumes and values. The PVC supply chain, both upstream and downstream are analysed and significant uses (applications) of PVC are summarised, focussing on those that account for significant volumes and/or economic values.
- A summary of data on end-of-life treatment of PVC in the EU and other world regions, with a special focus on PVC recycling is also included.
- Economic issues associated with global PVC production are examined. This includes an overview of price differences for raw materials and energy, considering different regulatory standards, and production costs. The section explores available data on primary PVC (i.e., pellets/powder) and secondary PVC (i.e., plastic pipes) as far as data are available. Data on volumes of substances of concern (lead, cadmium, phthalates) within PVC products are drawn out where data are available.
- With reference to historical data on PVC manufacturing, a simple forecast is provided of key trends in PVC production volumes for the EU27, imports to it from the rest of the world and of PVC waste generated to 2050. The basis for the forecasts, the assumptions made, and the uncertainties associated with any such exercise are made clear.

3.1.1 Approach

PVC has been the subject of extensive research for many years. This assessment has not focussed on generating "new" data and it has not included a survey with industry, for example. Instead, the section collates statistical sources in the public domain, supplemented with a small number of interviews with key stakeholders for verification and further depth. A literature review has been undertaken and a review of data from key organisations, such as VinylPlus and Plastics Europe, alongside statistical databases such as Eurostat.

3.2 Manufacture of virgin PVC

The most widely used method for virgin PVC manufacture is based on ethylene feedstock, reacting with chlorine to form ethylene dichloride, which then undergoes thermal decomposition to form the vinyl chloride monomer (VCM), followed by polymerization to form polyvinyl chloride (PVC) (5). The exception to this is in China, where acetylene feedstock dominates production. These processes involve VCM as an intermediate. This chemical is used primarily as a precursor for PVC. In 2018, 7% of globally manufactured VCM was exported to other countries; the remainder was used within the country of manufacture. A 2020 market assessment noted that VCM trade was expected to fall further, as VCM manufacture is integrated into more PVC production plants (IHS Markit 2021). More information on the production of PVC is provided in section 2.2 of this report.

3.3 PVC Production Volumes

Eurostat provides time-series data on sold production volumes, imports and exports for PVC for several different "primary" PVC forms. These include PVC unmixed with other substances; non-plasticised PVC mixed with other substances; and plasticised PVC mixed with other substances (European Commission 2021a)[20]. Data are presented on both total production and volumes sold from 2008 to 2019. These data are subject to a degree of uncertainty, as confidential values are replaced with rounded figures or are not shown. Production tonnage data in the Eurostat dataset is confidential or unavailable for several EU Member States. Missing data includes, for example, the Netherlands, which hosts one of Europe largest capacity plants (ICIS 2013). Further sources have therefore been used to corroborate the data and fill gaps.

It should be noted that production data definitions are not consistent in the various sources reviewed. Therefore, for the purposes of this chapter, the following definitions have been used:

Table 3-1: Definitions used in PVC volume data

Defintion	Explanation
Total production	These data relate to total products sold outside of the manufacturing company as well as tonnage retained for internal use.
Sold production	Tonnage data regarding total products sold outside of the manufacturing company.
Production Capacity	The maximum production capacity of PVC production plants. This is typically greater than sold production.
Demand/Volumes placed on the EU 27 market.	The total volume of a product placed on the EU27 market. This is production, plus imports, less exports.

3.3.1.1 Production in the EU

Primary PVC production volumes for the EU27 and Member States are presented in Table 3-2. The presented data relates to 2019, which are the latest data available. Where data are missing or confidential in the raw data, this is noted. Overall a total of 6,076,000 tonnes of PVC were manufactured in 2019 in the EU27. Of this, some 4.9 million tonnes were non-plasticised PVC (i.e. PVC with no plasticiser added) and not mixed with any other substance. "Non-plasticised PVC mixed with other substances" and "plasticised PVC mixed with other substances" accounted for a further 374,000 tonnes and 792,000 tonnes respectively.

Germany and France account for the largest production, together accounting for around 50% of primary PVC. Spain, Hungary and Portugal are also sizable markets, but data on production in several countries are not available, including the Netherlands, Belgium, Italy and Poland. Despite this, production capacity data from other sources suggests sizeable production volumes originate here. Production tonnage in 2019 has been estimated for the UK at around 200,000 tonnes;

[20] PRODCOM datasets: *Polyvinyl chloride, not mixed with any other substances, in primary forms*; *Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms*; and *Plasticised polyvinyl chloride mixed with any other substance, in primary forms*.

based on the difference between total production figures for EU28 and EU27. Stakeholder consultees indicated this data on the overall market size is accurate.

Table 3-2: Primary PVC Total Production Volumes (tonnage) [1] Source: Eurostat (2019 data)

Country/ Territory	Unmixed, Non-plasticised		Mixed, Non plasticised		Mixed, plasticised		Total
	Production (tonnes)	%age of EU27 Total	Production (tonnes)	%age of EU27 Total	Production (tonnes)	% age of EU27 Total	
EU							
EU27	4,910,000	-	374,000	-	792,000	-	6,076,000
EU28	5,110,000	-	414,000	-	876,000	-	6,400,000
Countries							
Germany	1,454,000	29.6	11,000	3.1	97,000	12.2	1,562,000
France	1,144,000	23.3	-	-	-		n/a
Spain	467,000	9.5	42,000	11.3	67,000	8.5	576,000
Hungary	248,000	5.1	13,000	3.1	-	-	n/a
Portugal	167,000	3.4	11,000	3.0	65,000	8.2	243,000
Unnaccounted	1,198,000	24.4	222,000	59.2	235,000	29.7	1,655,000
United Kingdom [2]	200,000	(4.1)	40,000	5.0	84,000	10.7	324,000

Source: Eurostat, *Total Production by PRODCOM list (NACE Rev. 2) – annual data* (European Commission 2021b). Datasets: *Polyvinyl chloride, not mixed with any other substances, in primary forms*; *Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms*; and *Plasticised polyvinyl chloride mixed with any other substance, in primary forms.* Accessed January 2021.

[1] Countries with productions of less than 100,000 tonnes in 2019 are not included in the table (but are included in the total).

[2] UK unmixed, non-plasticised production figures have been calculated by subtracting the total EU27 total production from EU28 total production. This production value has also been compared to EU27 total to find an approximate percentage contribution.

3.3.1.2 Production trends

Total European primary PVC production appears to be on a slight downward trend. Figure 6 and Figure 7 show EU27 production trends for primary PVC over the last decade. Unmixed PVC makes up the largest proportion by volume. Volumes decreased between 2008 to 2012 by 26% (from 6,930,000 to 5,010,000 tonnes). This pattern coincides with the recession of 2008-09. A partial and temporary recovery was observed in 2010, with a marginal decline thereafter to around 5.0 million tonnes. Plasticised PVC volumes also declined somewhat overall, here volumes stood at 740,000 tonnes in 2019. Non plasticised PVC volumes are more stable, fluctuating between 300,000 and 420,000 tonnes/year. These stood at 320,000 tonnes in 2019.

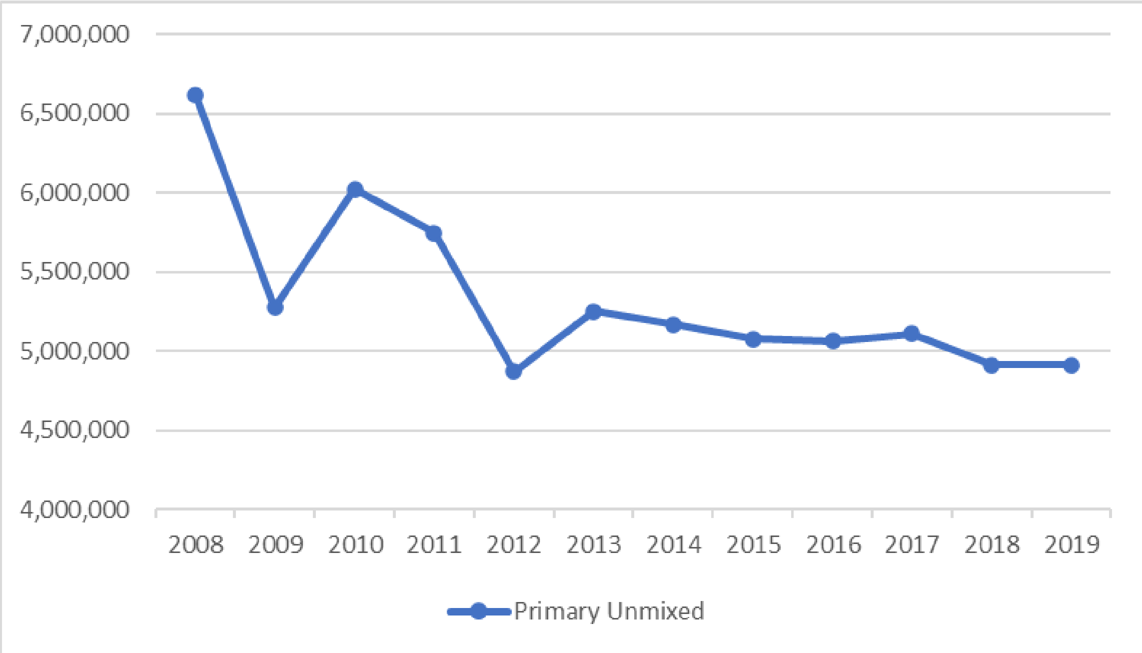


Figure 7. EU27 Primary Unmixed PVC production (tonnes) 2008 -2019

Source: [Eurostat, *Total Production by PRODCOM list (NACE Rev. 2) – annual data*] (European Commission 2021b)
Dataset: *Polyvinyl chloride, not mixed with any other substances, in primary forms.* Accessed January 2021

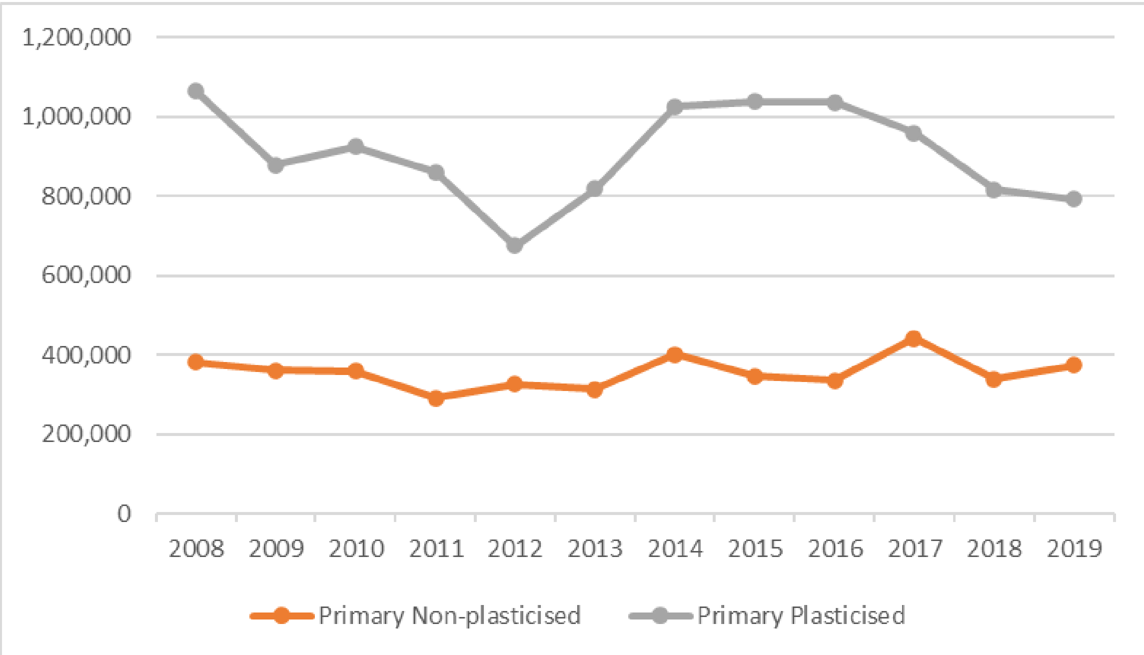


Figure 8. EU27 Primary Plasticised and Non-plasticised PVC production (tonnes) 2008 -2019

Source: [Eurostat, *Total Production by PRODCOM list (NACE Rev. 2) – annual data*] (European Commission 2021b)
Datasets: *Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms;* and *Plasticised polyvinyl chloride mixed with any other substance, in primary forms.* Accessed January 2021

3.3.1.3 Imports and Exports of Primary PVC in the EU

Imports and exports of primary PVC[21] are in Table 3-3. EU27 data in the table below relates to trade between the EU27 and the rest of the world. The remaining data for individual countries reflect trade with all countries (in the EU or not). Overall, it shows in 2019 the EU was a net exporter of primary PVC, exporting just over 1 million tonnes more than it imports. Nationally, the largest exporters of primary PVC are Germany, France, Belgium and the Netherlands. The largest importers are Italy, Germany, Belgium and Poland. Whilst data on production are not available for the Netherlands, Italy, Belgium and Poland (as shown in Table 3-2), they account for large import and export values. In terms of trade balance, the largest net exporter was France; the largest net importer was Italy.

Table 3-3: Total Import and Export tonnage of primary PVC within EU27 in 2019

Country/Territory	Exports (tonnes)	Imports (tonnes)	Trade Balance
EU27	1,636,364	580,363	1,056,001
Germany	1,102,433	646,098	456,335
France	910,741	265,034	645,707
Belgium	706,553	515,866	190,687
Netherlands	703,353	216,508	486,845
Spain	319,159	148,694	170,465
Hungary	272,825	48,449	224,376
Sweden	237,025	63,270	173,755
Portugal	155,638	37,277	118,361
Italy	154,314	685,734	-531,420
Poland	108,819	387,745	-278,926
United Kingdom	103,471	268,768	-165,297
Czechia	100,637	130,136	-29,499
Slovakia	13,966	33,529	-19,563
Austria	12,407	41,537	-29,130
Romania	11,587	90,168	-78,581

Source: Eurostat, *Sold Production, exports and imports by PRODCOM list (NACE Rev. 2) – annual data* (European Commission 2021a)
Datasets: *Polyvinyl chloride, not mixed with any other substances, in primary forms*; *Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms*; and *Plasticised polyvinyl chloride mixed with any other substance, in primary forms.* Accessed January 2021

Figure 88 shows trends in trade of primary PVC to/from the EU27 between 2003 and 2019 (excluding 2005, for which there are no data). In the context of a slight decline in overall production shown above, the overall trend was for broadly stable export volumes after an

[21] Within the Eurostat PRODCOM database, these datasets are respectively referred to as: *Polyvinyl chloride, not mixed with any other substances, in primary forms; Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms;* and *Plasticised polyvinyl chloride mixed with any other substance, in primary forms.*

increase between 2009-2011. After 2012, volumes of imports slowly increased. The EU27 has therefore remained a net exporter since 2003 by a significant margin. According to the Kunstoffe International Plastics World Market report (2019), the majority of this PVC came from North America from 2016 to 2018:

"*The European PVC Market recorded an average growth in demand of only 0.8% from 2015 to 2018. The capacity of the European producers declined between 2013 and 2018. That led to increased utilization of the remaining plants. A comparatively high price level in Western Europe made imports more attractive. From 2016 to 2018, therefore, over 1 million t (1 Mt) PVC was imported, mostly from North America. The average increase of imports to Western Europe in the period, at 1.3% per year, was slightly above the average increase of exports, which was 1.2% per year*." (Kunststoffe International 2019)[22]

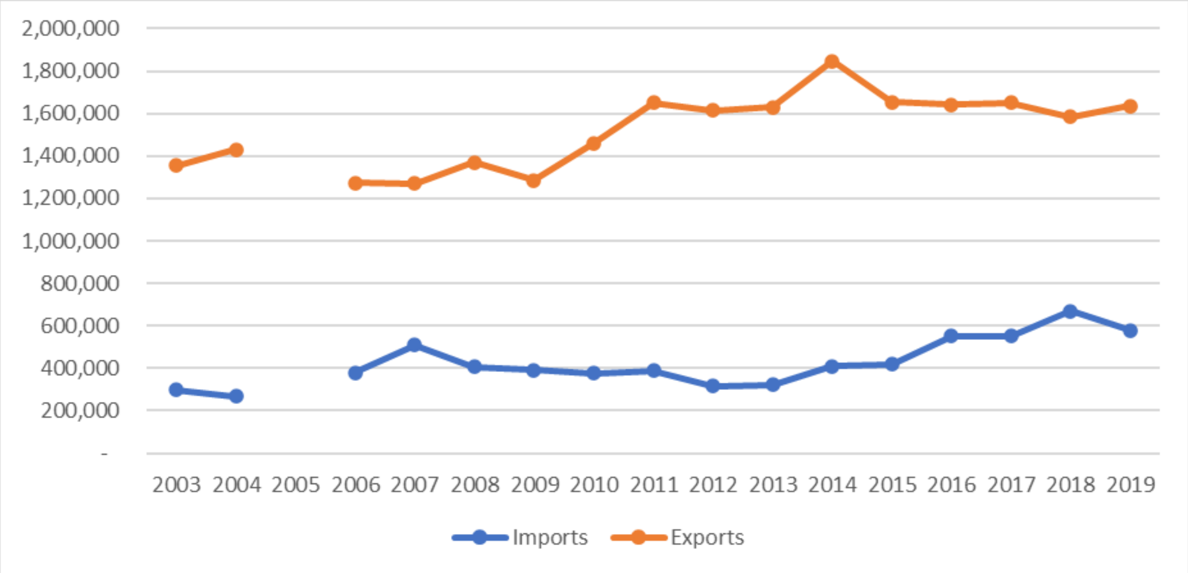


Figure 9. Total Primary PVC Trade with EU27 since 2003 (tonnes)

Source: [Eurostat, *Sold Production, exports and imports by PRODCOM list (NACE Rev. 2) – annual data*](#) (European Commission 2021a)
Datasets: *Polyvinyl chloride, not mixed with any other substances, in primary forms*; *Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms*; and *Plasticised polyvinyl chloride mixed with any other substance, in primary forms.* Accessed January 2021

Figure 99 shows trade data of primary PVC between the UK and the rest of the world since 1995 for context. The overall trend was for a narrowing of net imports to 2008-2009. Whilst there was a slight recovery in volume of both imports and exports after 2009, overall UK trade volumes appear to be declining.

[22] This source includes other forms of PVC within its total tonnage, as opposed to just primary, which is considered in Figure 9.

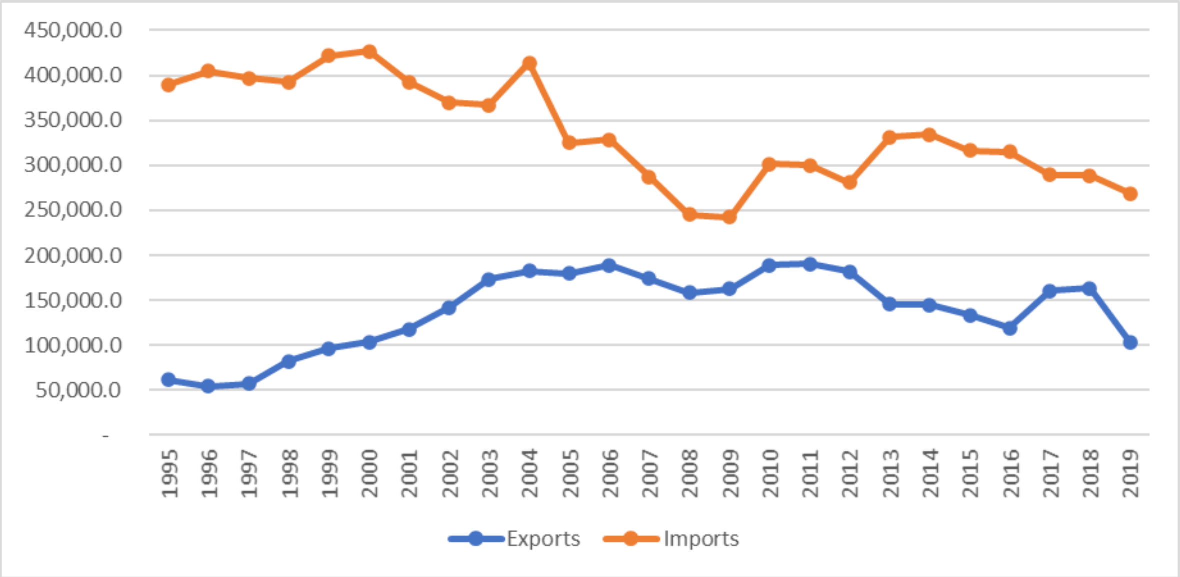


Figure 10. Total Primary PVC Trade with the UK since 1995 (tonnes)

Source: Eurostat, *Sold Production, exports and imports by PRODCOM list (NACE Rev. 2) – annual data* (European Commission 2021a)
Datasets: *Polyvinyl chloride, not mixed with any other substances, in primary forms*; *Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms*; and *Plasticised polyvinyl chloride mixed with any other substance, in primary forms.* Accessed January 2021.

3.3.1.4 Production, imports and exports

Figure 1010 summarises total EU27 sold production volumes, imports and exports of primary PVC for 2019. Market values are also shown. As above, primary unmixed PVC accounts for the largest volumes, whilst mixed PVC has a higher average value reflecting the further processing associated with these products. Overall, a total of 5,974,000 tonnes were manufactured and sold, with a market value of €4.736 billion (bn€). A total of 1,636,000 tonnes of primary PVC were exported from the EU, equating to a value of 1.502 bn€. A total of 580,000 tonnes of primary PVC were imported, with a market value of 0.517 bn€.

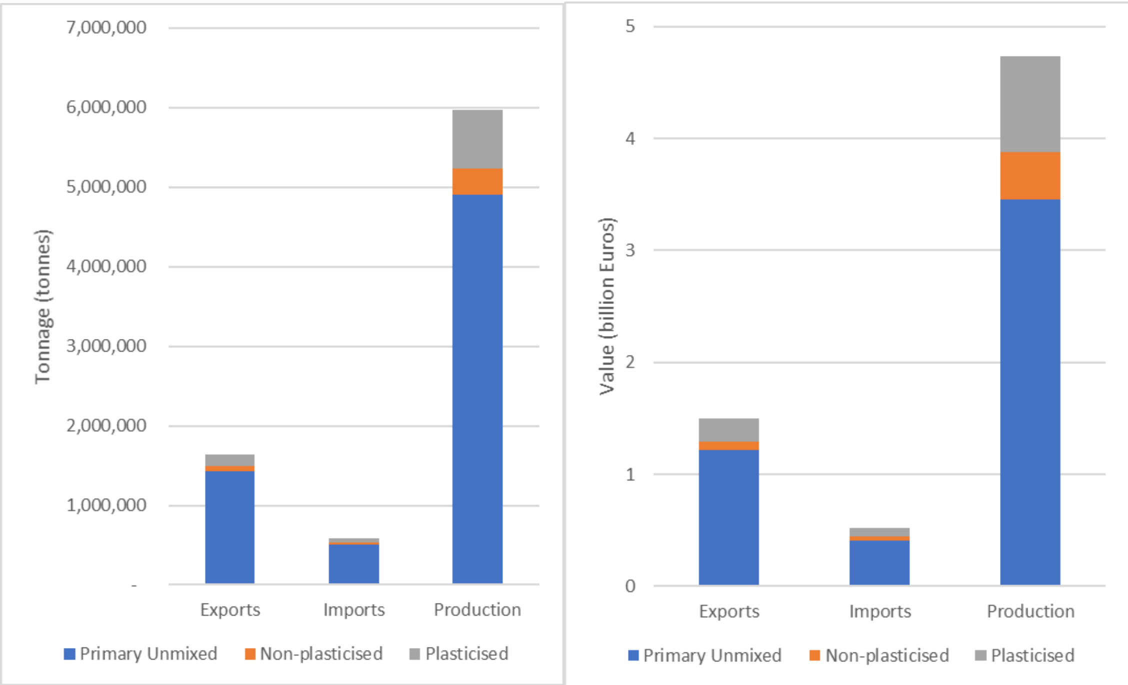


Figure 11. Sold Production, Import and Export Tonnage and Value for Primary PVC in EU27 (2019)

Source: [Eurostat, *Sold Production, exports and imports by PRODCOM list (NACE Rev. 2) – annual data*](#) (European Commission 2021a)
Datasets: *Polyvinyl chloride, not mixed with any other substances, in primary forms*; *Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms*; and *Plasticised polyvinyl chloride mixed with any other substance, in primary forms.* Accessed January 2021

3.3.2 Data on PVC product types and uses

Figure 11 summarises the same Eurostat data but provides greater detail on the physical form and type of products. These give an indication of the applications in which they are used as well as differences in average values. The following categories are those provided within the Eurostat database, which describe a number of different forms of PVC products. No further information is provided by Eurostat on the details of these products, and the categories are broad. Assumptions have been made on each product's major applications in the table below to provide context.

Table 3-4: Definitions used in PVC volume data

Category	Eurostat Description	Typical Applications [1]
Rigid Tubes	Rigid tubes, pipes and hoses of PVC	Construction, agriculture
Plates 1	Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness <= 1 mm	Construction, automotive, packaging, consumer, electronics, healthcare
Plates 2	Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness > 1 mm	Construction, automotive, packaging, consumer, electronics, healthcare

Plates 3	Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness <= 1 mm	Construction, automotive, packaging, consumer, electronics, healthcare
Plates 4	Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness > 1 mm	Construction, automotive, packaging, consumer, electronics, healthcare
Cellular Polymer Sheets	Plates, sheets, film, foil and strip of cellular PVC[23]	Construction, automotive, packaging, consumer, electronics, healthcare
Floor Coverings [2]	Floor coverings in rolls or in tiles and wall or ceiling coverings consisting of a support impregnated, coated or covered with PVC	Construction
Other coverings of PVC [2]	Other floor, wall, ceiling... coverings of PVC	(N.B precise products are not further described) Construction
Monofilament Profiles	Monofilament with any cross-sectional dimension > 1 mm, rods, sticks, profile shapes, of polymers of vinyl chloride (including surface worked but not otherwise worked)	*(Assumed to relate to window frames, door frames etc).* Construction, automotive, consumer.

[1] The applications provided are not an exhaustive list, but are included to provide a degree of context to the descriptions provided by the Eurostat database.
[2] Note that floor coverings and cellular polymer sheets are not included within the tonnage figures, as these are measured in metres squared[24]. They are however, included in the value data.

The tonnage totals are 3,380,900 tonnes of PVC products manufactured within the EU and 249,800 tonnes imported to the EU in 2019. A total 1,086,400 tonnes of PVC products were exported in 2019. Regarding exports, the largest product category by value were floor coverings, at 0.422 billion Euro (bn€), followed by monofilament profiles and “plates 3”, both at 0.361 bn€. The highest value imported product was ‘Other Floor Coverings’ at 0.517 bn€. By tonnage, the majority of PVC (not including coverings) is used within rigid tubes and monofilament profiles. These products are used primarily in construction.

[23] Cellular PVC is a form of PVC with bubbles within its physical structure, making it softer and more flexible than standard PVC. It is molded into cellular form while in a different state to standard PVC.
[24] The data for reference are: Floor Coverings: Production = 220,000 m2; Imports = 63,000 m2 and Exports = 93,000 m2. Other coverings: Production = 119,000 m2; Imports = 85,000 m2; Exports = 33,000 m2 (European Commission 2021a).

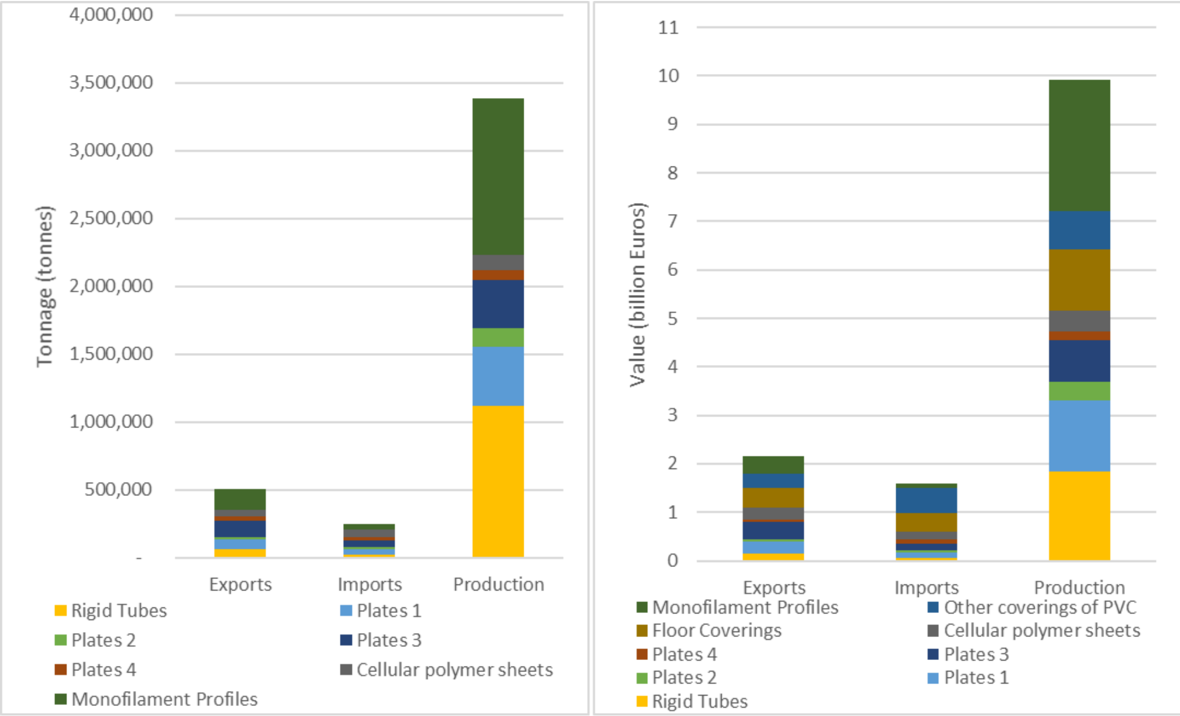


Figure 12. Sold Production, Import and Export Tonnage and Value for PVC Products (EU27, 2019)

Source: [Eurostat, *Sold Production, exports and imports by PRODCOM list (NACE Rev. 2) – annual data*] (European Commission 2021a)

Datasets: *Rigid tubes, pipes and hoses of PVC; Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness <= 1 mm; Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness > 1 mm; Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness <= 1 mm; Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness > 1 mm; Plates, sheets, film, foil and strip of cellular PVC; Floor coverings in rolls or in tiles and wall or ceiling coverings consisting of a support impregnated, coated or covered with PVC; Other floor, wall, ceiling... coverings of PVC;* and *Monofilament with any cross-sectional dimension > 1 mm, rods, sticks, profile shapes, of polymers of vinyl chloride (including surface worked but not otherwise worked).* Accessed January 2021.

Figure 12 shows the total value of sold PVC in the EU27 Member States (and the UK) by type of product. The largest market was in Germany was over €3.5 billion; in France some €1.3 billion and in Italy around €1.1 billion. Other sizable markets include Spain (€0.7 billion); Poland (€0.6 billion) and the Netherlands (€0.5 billion). Data for the UK are also shown with a market value of around €1.2 billion.

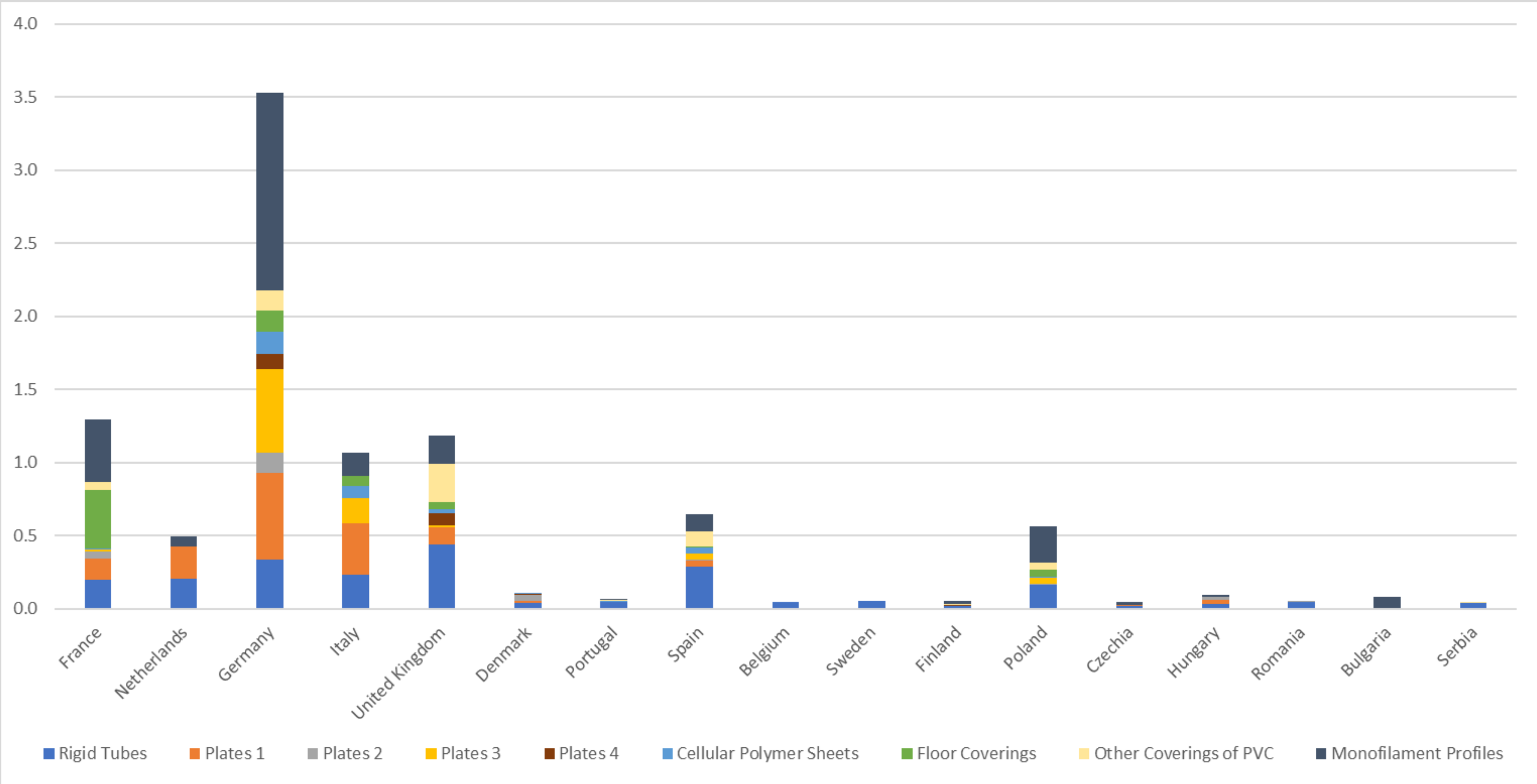


Figure 13. Distribution of Total PVC Product Value within Europe 2019 (bn€)

Source: Eurostat, *Sold Production, exports and imports by PRODCOM list (NACE Rev. 2) – annual data* (European Commission 2021a)
Datasets used are as above. Accessed January 2021

3.3.3 Volume placed on the market in the EU

Table 3-5 summarises available data on the total primary PVC volumes placed on the market within the EU. This has been calculated based on EU27 production, plus imports, less exports, for all primary PVC[25]. The total volume of primary PVC placed on the EU27 market was 4,919,000 tonnes (2019 data)[26]. This is based on total sold PVC production volume for EU27 of 5,975,000 tonnes in the same year. Total exported volume in 2019 was 1,636,000 tonnes and total imported volume was 580,000 tonnes. It has not been possible to produce a breakdown of demand by country within the EU, given data gaps. Data for the "EU27, plus the UK"[27] is also shown. This yields a total of 5,440,000 tonnes.

To cross reference these data, demand has been calculated based on research from Plastics Europe. Note that this data includes the UK, Norway and Switzerland and has been calculated based on applying an estimate that 10% of total plastics demand is accounted for by PVC. Overall they are in good agreement. EU27 PVC demand stood at around 4.9 million tonnes in 2019. Stakeholder consultation indicates this is an accurate reflection.

Table 3-5: Demand within the EU (tonnage)

Source	Source Year	Country/ Territory [1]	Demand (tonnes)	Year
Eurostat [1]	**2020**	EU27	4,919,000	2019
		EU27, plus the UK.	5,440,000	
Plastics Europe [2] [1]	**2019**	EU, plus UK, Norway and Switzerland.	5,120,000	2018
Plastics Europe [3] [1]	**2020**	EU, plus UK, Norway and Switzerland.	5,070,000	2019

[1] Demand tonnage from Plastics Europe has been calculated by applying the proportion of demand by resin type in each year, to the respective total plastic demand tonnage. These data are approximate. "Demand" is not defined in the source data.

Sources: [1] Eurostat, Sold Production, exports and imports by PRODCOM list (NACE Rev. 2), accessed January 2021 (European Commission 2021a)
[2] Plastics Europe, *Plastics – the Facts 2019* (PlasticsEurope 2019)
[3] Plastics Europe, *Plastics – the Facts 2020* (PlasticsEurope 2020)

Figure 13 shows the total volume of PVC placed on the market in the EU27 Member States and UK by type of product. The calculation of these values has been conducted using the same approach as above (Figure 12). Germany has the largest product tonnage, at 694,000 tonnes, followed by France at 493,000 tonnes. Italy, Spain and Poland also account for significant volumes.

[25] Within the Eurostat PRODCOM database, the primary PVC datasets are described as: *Polyvinyl chloride, not mixed with any other substances, in primary forms; Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms;* and *Plasticised polyvinyl chloride mixed with any other substance, in primary forms*

[26] Based on total sold PVC production volume for EU27 of 5,975,000 tonnes in 2019. Total exported volume in 2019 was 1,636,000 tonnes and total imported volume was 580,000 tonnes.

[27] This is based on the following figures, calculated from the Eurostat database: total production quantity – 6,307,000 tonnes; total imported quantity – 556,000 tonnes; and total exported quantity – 1,423,000 tonnes.

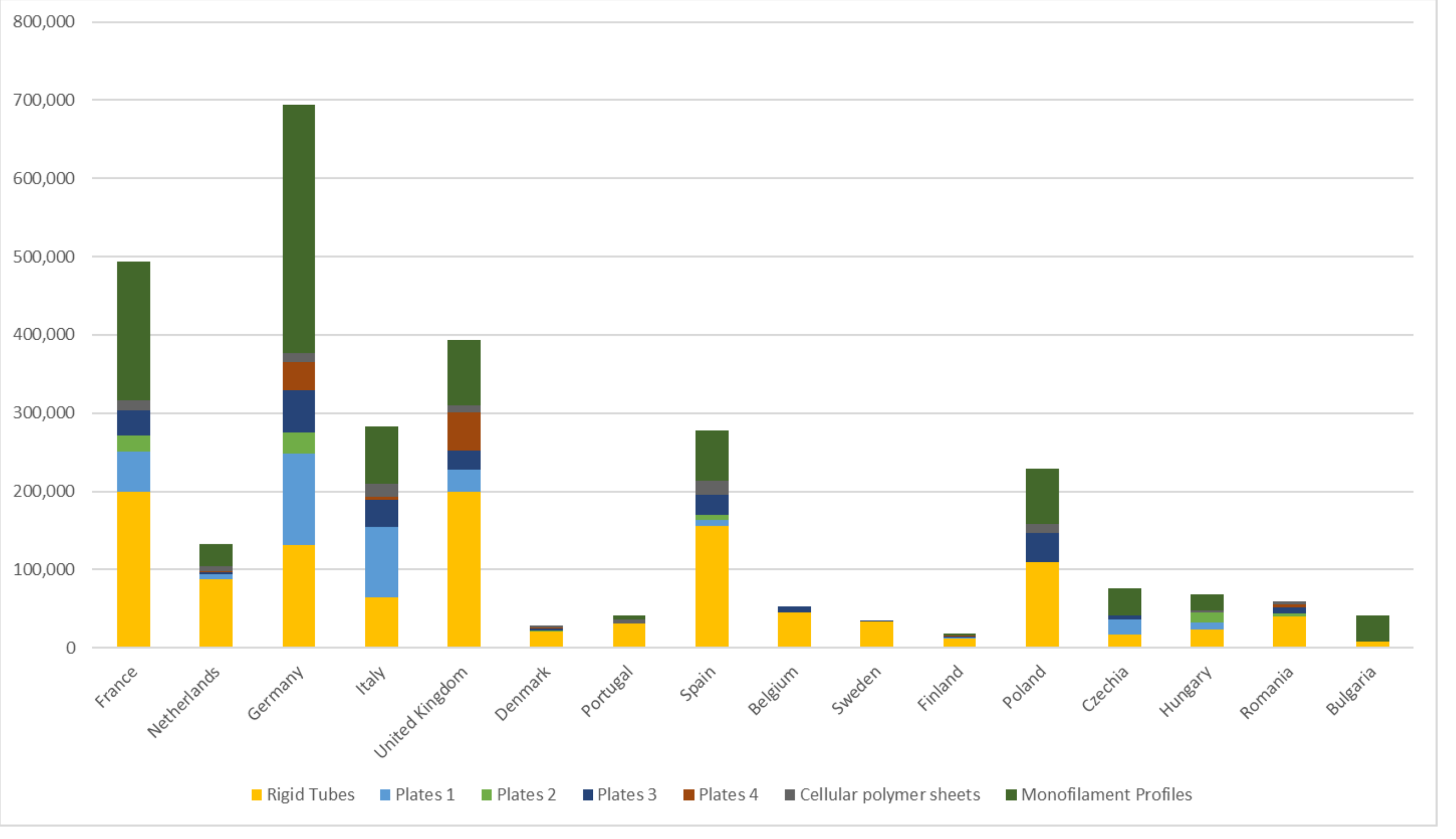


Figure 14. Distribution of PVC Product Placed on the Market within Europe 2019 (tonnes)

Source: [Eurostat, *Sold Production, exports and imports by PRODCOM list (NACE Rev. 2) – annual data*] (European Commission 2021a)

Datasets: *Rigid tubes, pipes and hoses of PVC; Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness <= 1 mm; Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness > 1 mm; Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness <= 1 mm; Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness > 1 mm; Plates, sheets, film, foil and strip of cellular PVC;* and *Monofilament with any cross-sectional dimension > 1 mm, rods, sticks, profile shapes, of polymers of vinyl chloride (including surface worked but not otherwise worked).* Accessed January 2021.

3.3.4 Production in the Rest of the World

Available data on global production are less detailed than for the EU27, but estimated at some 55 million tonnes in 2016 (latest available). At that time almost half was accounted for by China (Table 3-6:). More recent public data was unavailable.

Table 3-6: Production Volume (tonnage)

Source	Country/ Territory	Production (tonnes)	Year
Kunstoffe International	Global	55,000,000 [1]	2016

[1] Stated as "*approximate*" within the source text.

Source: Kunstoffe International, Commodity Plastics Trend Report 2017 - PVC (Kunststoffe International 2017)

Table 3-7: shows data on global *production capacity* compiled from several sources and relating to several years. The intention is to illustrate overall trends as well as the distribution across the world in more detail than has been possible with production data. Overall, global production capacity from 2013 to 2016 steadily increased (Kunststoffe International 2019). These forecast was made before the economic contracts associated with the COVID-19 pandemic. This is discussed further below in section 3.5. According to Kunststoffe International, 90% of global capacity growth over this time has been driven by China, which has averaged 2% growth per annum between 2011 to 2017 (Kunststoffe International 2016) (Kunststoffe International 2017).

Production capacity data by world region is also summarised below. Northeast Asia has the largest production capacity by a substantial margin, at 29,800,000 tonnes. Of this around 24,000,000 tonnes (43% of global capacity) was in China. Sizeable capacity exists in North America (9,400,000 tonnes), with Western Europe at 6,100,000. The data below indicate that capacity in North America has increased compared to that in Europe since 2012, at which time capacities were similar, with some plants in Western Europe closing (Kunststoffe International 2013) (Kunststoffe International 2019).

Table 3-7: Production Capacity (tonnage)

Source	Source Year	Country/ Region	Capacity (tonnes)	Year
Kunstoffe International [2]	**2019**	Global	55,100,000	2018
		Northeast Asia (of which, China)	29,800,000 (24,000,000)	
		North America	9,400,000	
		Western Europe	6,100,000	
		Southeast Asia	2,200,000	
		South America	1,700,000	
		Middles East/Africa	1,700,000	
		Indian Subcontinent	1,700,000	
		Central Europe	1,100,000	
		Russia and Baltic States	1,100,000	
Plasticsinsight.com [1]	**2017**	Global	53,000,000	2013

Source	Source Year	Country/ Region	Capacity (tonnes)	Year
			55,000,000	2014
			57,000,000	2015
			61,000,000	2016
Kunstoffe International [3]	**2016**	Global	62,000,000 [1]	2016
		China	32,000,000 [1]	
Kunstoffe International [4]	**2013**	Global	54,000,000	2012
		China	24,000,000	2013
		North America	8,000,000 [2]	2012
		Europe	8,000,000 [2]	2012

[1] Global production capacity stated as "approximate", and China's capacity stated to be "over 51%" of the global capacity.

[2] Note this has been estimated from the following noted in the text: "North America and Europe both holding "15% of worldwide PVC capacity".

Sources: [1] PlasticInsights.com, *PVC Production, Trading Price and Market Demand*, accessed January 2021 (Plastics Insight n.d.)
[2] Kunstoffe International, Special K – PVC, 2019 (Kunststoffe International 2019)
[3] Kunstoffe International, Special K – PVC, 2016 (Kunststoffe International 2016)
[4] Kunstoffe International, Special K 2013 - Plastics World Market - PVC
(Kunststoffe International 2013)

3.3.4.1 Demand in the Rest of the World

Table 3-8 provides available data on global demand[28]. Data specific to product type is unavailable. Global demand distribution shows North East Asia leading by a substantial margin, although Western/Central Europe and North America make up large shares. Recent market analysis indicates that structural economic and demographic trends were driving the global distribution of demand for PVC:

"In recent years, the stronger PVC consumption has been concentrated in the developing economies in Asia, such as China, India, Vietnam, and Indonesia. For high-demand locations, the common drivers of consumption are a large population base with a stable political climate that still needs considerable spending on infrastructure. Northeast Asia is expected to remain the largest regional market. China will remain the largest consumer and producer of PVC in the world for the foreseeable future." (IHS Markit 2020)

Table 3-8: Demand (tonnage)

Source	Source Year	Region	Demand (tonnes)	% of Global Consumption	Year
PlasticInsights.com [1]	**n.d.**	Global	38,300,000	-	2013
		Global	41,300,000	-	2016
Kunstoffe International [2]	**2019**	Global	45,600,000	-	2018
		North East Asia	21,400,000	47	

[28] Note that precise definitions are not clear with different sources quoting different terms most stating 'demand', but one referring to consumption. These have been assumed to refer to the same statistic, and the definitions in section 3.3.

		South East Asia	2,700,000	6	
		North America	5,900,000	13	
		South America	1,800,000	4	
		Western Europe	4,100,000	9	
		Central Europe	900,000	2	
		Russia (CIS) and Baltic States	1,400,000	3	
		Middle East/Africa	3,600,000	8	
		Indian Subcontinent	3,600,000	8	
Kunstoffe International [3]	**2017**	Global	41,500,000	-	2016
Kunstoffe International [4]	**2013**	Global	37,400,000 [1]	-	2012

[1] Stated as consumption in the source text

Sources: [1] PlasticInsights.com, *PVC Production, Trading Price and Market Demand*, accessed January 2021 (Plastics Insight n.d.)
[2] Kunstoffe International, Special K – PVC, 2019 (Kunststoffe International 2019)
[3] Kunstoffe International, Commodity Plastics Trend Report 2017 – PVC (Kunststoffe International 2017)
[4] Kunstoffe International, Special K 2013 - Plastics World Market - PVC (Kunststoffe International 2013)

3.3.5 Additives

PVC is supplemented with a variety of additives to tailor material properties for the various applications. These include: plasticisers; pigments; heat and light stabilisers; lubricants; fillers; flame retardants; impact modifiers; and fibres as reinforcing materials (European Commission 2004). Table 3-9 lists available data on the approximate shares by weight of each additive within different PVC applications.

Rigid PVC applications, such as in pipes, profiles and rigid films, contain a lesser proportion of additives by weight. Additives typically make up between 2 to 15% of rigid PVC products by weight. These products also make up the majority of PVC tonnage globally according to the data in Table 3-12. Data gathered by ECHA indicate that rigid PVC does not contain plasticisers, including phthalates (European Chemicals Agency 2021). In contrast, flexible PVC tends to include higher proportions of additives by weight, typically ranging between 35% and 65%. Flexible PVC comprise a lower proportion of tonnage globally.

Table 3-9: Typical composition of PVC compounds

Application	Component Share (weight - %)				
	PVC polymer	Plasticiser	Stabiliser	Filler	Others
Rigid PVC applications					
Pipes	98	-	1-2	-	-
Window profiles	85	-	3	4	8
Other profiles	90	-	3	6	1

Rigid films	95	-	-	-	5
Flexible PVC applications					
Cable insulation	42	23	2	33	-
Flooring (Calender)	42	15	2	41	0
Flooring (paste, upper later)	65	32	1	-	2
Flooring (paste, inside material)	35	25	1	40	-
Synthetic leather	53	40	1	5	1

Sources: European Commission, *Life Cycle Assessment of PVC and of principal competing materials*, 2003 (European Commission 2004)

Table 3-10 sets out the different additives associated with specific properties along with typical examples. The examples do not represent an exhaustive list, and focuses on selecting the most commonly used under each category and those listed as SVHCs.

Table 3-10: Properties Associated with Different Additives

Property	Additive	Examples [29]	
		Source: (European Commission 2004)	**Source:** (European Chemicals Agency 2021)
Chemical/weather resistance	Heat stabiliser	Lead Stabiliser*	
		Tin organic stabiliser*	
		Ba/Zn or Ca/Zn liquid systems in soft PVC and solid systems in rigid PVC.	
	Light stabiliser	No data identfied.	Aromatic organics
			Titanium Dioxide*
Thermal insulation, electrical insulation, rigidity	No data identfied		
Biocompatibility	Biostabilisers	No data identfied.	Oxybisphenoxarsine
Durability	Impact modifiers	No data identfied.	Amides, C16-C18 (even), N,N'-ethylenebis
Transparency	Native to PVC but requires lack of 'filler' additive	-	
Flame resistance	Flame retardants	Diantimony trioxide*	
		Chloroparaffins*	
		Aluminium hydrate	Azodicarbonamide*

[29] Substances listed under the ECHA Candidate List as a Substance of Very High Concern (see Section 2.2) are labelled with an asterisk (*), according to which; Lead Stabilisers are 'Toxic for reproduction', lead chromates are carcinogens, Tin Organic Stabilisers are 'Toxic for reproduction', azodicarbonamide has 'Respiratory sensitising properties', chloroparaffins are 'persistent bioaccumulative and toxic' and 'very persistent and very bioaccumulative'. Phthalates are also SVHCs and represent a number of different chemicals, many of which are classified as being 'Toxic for reproduction' and having 'Endocrine disrupting properties'. Titanium Dioxide and Diantimony Trioxide, also labelled with asterisks, are stated as 'Suspected Carcinogens', under Category 2 of the Classification of the European Union (CLP), but is not an SVHC.

		Aluminium hydroxide	
		Magnesium oxide	
Flexibility	Plasticiser	Phthalic acid esters and phthalates*	
Coloured	Pigments	Titanium white, lead chromates*	Tin dioxide

Sources: European Commission, *Life Cycle Assessment of PVC and of principal competing materials*, 2003 (European Commission 2004)
ECHA, *Candidate List of Substances of Very High Concern for Authorisation*, (European Chemicals Agency 2021)

Quantitative information on the approximate content in PVC of only a small number of substances of very high concern (SVHC) has been identified. Where available, the volume placed on the market and the volume in imports are noted below.

Table 3-11: Available data on substances in PVC

Substance	Additive Type	Tonnage placed on market	Imported Tonnage	Data Year
Compounds of Lead [1]	Stabiliser	1,057 – 4,579	634 - 4,579 ("at least 60% of total on market")	2016
Compounds of Cadmium[30] [2]	Pigment	Between 5,000 and 90,000 tonnnes of coloured plastics produced in the EU contain cadmium. PVC data not identified.	Data unavailable	n/a
Brominated Flame Retardant (DecaBDE) [3]	Flame Retardant	~[31]	~1,300 – 1,800 (in plastic articles)	2012
Phthalates (DEHP, DBP, DIBP, BBP) [4]	Plasticiser	171,135 (all plastics)[32]	124,245 (all imported articles)	2014
Bisphenol-A (BPA) [5]	Plasticiser/ Stabiliser	1,800	-	2008
Various [6]	Light stabilisers	N/A. The susbstances are noted as used in PVC applications with typical concentrations between <0.01 to 5.0%	N/A	Accessed 2021
	Heat Stabilizers	Used in PVC applications with typical concentrations of up to 4%		
1,3-diphenylpropane-1,3-dione; Calcium oxide	Other stabilisers	Limited uses in PVC with concentrations up to 1%		
Various [6]	Antioxidants	Used in PVC applications with typical concentrations of up to 3%		
	Pigments agents	Used in PVC applications with typical concentrations of up to 5%, with some (e.g carbon black, barium sulphate, titanium dioxide)		

30 Figures for Cadmium available only cover its usage in plastic pigments, which are not the primary application in PVC.
31 DecaBDE (or other Brominated Flame retardants, of which DecaBDE is the most prominent) is not manufactured in the EU.
32 DEHP is the most prevalent of the phthalate plasticisers, 95% of which is used in PVC manufacturing. The values quoted provide the sum of all four phthalates, regardless of their use, so it's tonnage within PVC will be significantly less than that quoted.

Substance	Additive Type	Tonnage placed on market	Imported Tonnage	Data Year
		up to 50%		
	Flame retardants	Uses in PVC with concentrations up to 35%		
	Lubricant	Uses in PVC with concentrations up to 50%, with most <5%.		
	Plasticisers	Various uses in PVC with concentrations up to 35%.		

Sources: [1] ECHA – Opinion on an Annex XV dossier proposing restrictions on Lead stabilisers in PVC (European Chemicals Agency - RAC, SEAC 2018)
[2] ECHA – Annex XV – Assessment whether the use of cadmium and it's compounds in plastic materials not covered by entry 23 of reach annex XVII should be restricted (European Chemicals Agency 2015)
[3] ECHA – Annex XV – Proposal for a Restriction of Bis(pentabromophenyl)ether (European Chemicals Agency 2014b)
[4] ECHA – Annex XV – Proposal for a restriction – Four Phthalates (DEHP, BBP, DBP, DIBP) (European Chemicals Agency 2016)
[5] ECHA – Annex XV – Proposal for a restriction - 4.4'-isopropylidenediphenol (bisphenol A; BPA) (European Chemicals Agency 2014a)
[6] A Full list of all substances, by category is published by ECHA as part of the "plastics additive initiative": (European Chemicals Agency n.d.)

3.3.5.1 Global demand by product and sector

This section explores global PVC volumes by product type and the downstream sector in which they are used. These data differentiate PVC demand[33] by product, however there is some inconsistency between the level of disaggregation. The sources in Table 3-12 provide information on the specific products (i.e. profiles, or films), but not the sector in which these volumes are used. So information on the typical sector in which these products are used has been added in the footnotes. These sector classifications are indicative estimates, based on desktop research. In contrast, the data from Table 3-13 contains sector of use, but only relates to 2003, so the more recent volume data has been applied to the relevant percentages used in each sector. The volumes are approximate. Data on global demand from Table 3-12 and Table 3-13 indicate:

- Pipes and fittings comprise the largest single share of global consumption, at 45% in 2018. The share has remained relatively stable, increasing by 3% since 2013. This equates to over 20 million tonnes of PVC in pipes and fittings. Based on desktop research, it is that assumed this is largely used in construction.

- Profiles comprise a further 17% global PVC consumption, equating to just under 8 million tonnes. Based on desktop research, it is assumed that the majority of these uses fall within the construction sector, although a proportion are likely to be used in the automotive industry. The data in Table 3-13, suggest the latter use accounts for around 2.5 million tonnes.

- Films and sheets have a similar share at 18% (over 8 million tonnes) of PVC. These have a wide variety of applications, primarily within construction, healthcare, packaging and furniture, so it is not possible to attribute this to a specific sector. But taking these three categories together, overall, this suggests that well over half of global consumption is attributed to construction, potentially as much 75%. This higher estimate is consistent

[33] The data headings are written as either 'consumption' or 'demand' within the source texts. These have been assumed to be the same.

with the views of consultees[34]. The data in Table 3-13, suggests somewhat lower, but still over 50%, in the order of 25 million tonnes used in construction overall.

- Wires and cables comprise 8% of global consumption – assumed to be applications within electronics. This equates to a little over 3.5 million tonnes. This is similar to the volumes in Table 3-13, of a little under 4.5 million tonnes.

- Uses in packaging (bottles), comprise around 1% in 2018, which appears to be declining somewhat; this equates to just under 0.5 million tonnes of PVC. The data in Table 3-13, suggests a higher amount, around 5 million tonnes used in packaging.

- The 'special field of pastes' is referred to in two sources. It appears to make up 6% of global consumption, with uses in construction (as floor coverings, body seals), furniture and automotive (as artificial leathers) and tarpaulins.

- The data in Table 3-13 suggests over 5.5 million tonnes are used in the consumer sector[35] with a further 2 million tonnes in other uses, globally.

[34] Pers. Comm.

[35] The 'consumer sector' is a categorization used in the underlying data, and is not defined in the source. It is likely to overlap with different classifications used elsewhere in this report.

Table 3-12: Global PVC Consumption by Product and Industry (2013-2018 estimates)

Source	Year	Estimate of sector in which prodcuts are likely to be used							
		Construction, & agriculture (Plastic Pipe Shop n.d.)	**Construction** (European Council of Vinyl Manufacturers n.d.a) **& automotive** (British Plastics Federation n.d.)	**Multiple Sectors** (European Council of Vinyl Manufacturers n.d.a) (European Council of Vinyl Manufacturers n.d.b) (British Plastics Federation n.d.)	**Electro-nics**	**Packaging** (European Council of Vinyl Manufacturers n.d.b)	**Construction, furniture & consumer** (Grand View Research 2017) (Kunststoffe International 2017)	**Multiple Sectors**	**Total**
		Type of PVC products							
		Pipes & fittings	**Profiles**	**Films and sheets**	**Cables**	**Bottles**	**Pastes**	**Other**	
Global Percentage Consumption									
Plastics Insights.com [1]	**2016**	42	19	17	9[2]	-	-	13	100
Kunstoffe International - Commodity Plastics Trend Report [2]	**2017**	44	17[1]	18	8	-	6	7	100
Kunstoffe International – Special K, Plastics World	**2018**	45	17	18[3]	8[2]	1	-	11[5]	100
	2016	43	17[1]	17	8	2	-	13	100

Market [3, 4, 5]	**2013**	42	18[1]	17[4]	8	2	-	13	100
Global Consumption Tonnage [6]									
Kunstoffe International – Special K, Plastics World Market [3]	**2018**	20,520,000	7,752,000	8,208,000	3,648,000	456,000	-	5,013,000	45,600,000

Sources: [1] Plastics Insights.com, accessed January 2021 (Plastics Insight n.d.)
[2] Kunststoffe International, Commodity Plastics Trend Report 2017 – PVC (Kunststoffe International 2017)
[3] Kunstoffe International, Special K 2019 - Plastics World Market - PVC (Kunststoffe International 2019)
[4] Kunstoffe International, Special K 2016 - Plastics World Market - PVC (Kunststoffe International 2016)
[5] Kunstoffe International, Special K 2013 - Plastics World Market - PVC (Kunststoffe International 2013)
[1] Listed as ‘Profiles and Tubes’ within the source text
[2] Listed as ‘Wires and Cables’ within the source text
[3] Listed as ‘Rigid film and plates’ within the source text
[4] Listed as ‘Rigid film/sheets’ within the source text
[5] The article text states that ‘*the special field of paste applications makes up 6% of total demand*’, it is assumed this 6% is included in “other”
[6] Consumption tonnage by application have been calculated based on the most up to date information (source [3]) and by using the total global consumption provided within the same source – 45,600,000 tonnes.

Table 3-13: Global PVC market volume share by application (%, 2013 applied to 2018 tonnages)

Source	Construction	Consumer	Packaging	Electronics	Transportation	Others	Total
Grand View Research, *Global Polyvinyl Chloride Market,* 2013	55.7	12.6	12.2	9.5	5.4	4.6	100
Estimated tonnage based on 2018 global total	25,400,000	5,745,000	5,565,000	4,330,000	2,460,000	2,100,000	45,600,000

Source: Grand View Research, *Global Polyvinyl Chloride Market,* 2017 (Grand View Research 2017). Tonnage based on Kunstoffe International, Special K 2019 - Plastics World Market - PVC (Kunststoffe International 2017). Numbers have been rounded to the nearest 5,000 tonnes.

3.3.5.2 European demand by product and sector

Table 3-14 provides a breakdown of PVC consumption (in the EU15) from 2003; this is dated, but used for comparison with more recent data from 2019 given the same inconsistencies between categorisation as in the global data. Table 3-15 provides information on the specific products (i.e. films), but not the sector in which these volumes are used. So, information on the typical sector in which these products are used have been added, based on desktop research, shown in the footnotes. The volumes are approximate. Overall:

- **Construction** accounts for the large proportion of consumption, however 'pipes & fittings' appear to account for a smaller share of European use than the global average. Overall, based on an EU PVC consumption of approximately 5 million tonnes, the data suggests PVC use in construction of not less than 450,000 tonnes, potentially as much as 3.55 million tonnes. The most common products are profiles, pipes and fittings. This is within the range of Table 3-14 which suggests just under 3 million tonnes. Stakeholder consultation estimated PVC usage in construction are toward the higher end of these estimates. Those that provided quantitative estimates suggested around 66 to 70% of total consumption. This would equate to around 4 million tonnes.

- **Automotive** uses include profiles and coated fabrics. These data are combined with construction, but overall volumes of use are not more than 1.5 million tonnes. If the 7% share from 2003 is applied, the figure is closer to 350,000 tonnes. Stakeholder consultation suggests usage closer to 5% (250,000 tonnes).

- Uses in **packaging** and **consumer uses**, includes rigid and flexible films and sheets, volumes are somewhat less than 750,000 tonnes. Using the share from 2003 for packing and household appliances, the figure is around 1.3 million tonnes. Stakeholder consultation suggests that the uses for packaging are around 450,000 tonnes.

- In cables, which correlates to the **electronics sector**. PVC consumption was around 350,000 tonnes, although some "other" and "misc. rigid" may be used in this sector, it is consistent with that in Table 3-14. Stakeholder consultation responses differ on volumes in this use. Some that this use may be somewhat higher, at around 10% (some 500,000 tonnes), whilst others somewhat below 350,000 tonnes.

- **Agriculture** and **healthcare** products include pipes and fittings, flexible tubes and profiles. Coated fabrics are also used in **furniture**. The volumes are more uncertain but appear to account for a small share. Stakeholders consultees estimated approximately 1% of PVC consumption is within healthcare, which equates to roughly 60,000 tonnes.

Data provided by the European Council of Vinyl Manufacturers (ECVM) on PVC demand in the EU27 plus the UK, Norway, Iceland and Switzerland was provided in Summer 2021. These are in Table 3-16. ECVM estimates for different applications are broadly in agreement with the calculated values displayed in Table 3-15. Correspondence with the ECVM revealed that their data features a sizeable portion of PVC consumption not assigned to any one specific application; when these are accounted for, their total estimate for demand in 2020 amounts to 4,770,000 tonnes, which is closely aligned to the estimated total of 5,000,000 tonnes presented in Table 3-15.

Table 3-14: PVC Consumption by Product and Industry n EU15 (%, 2003, applied to 2019 EU27 tonnages)

Source	Construction	Electronics	Packaging	Furniture	Other household appliances	Automotive	Other	Total
European Commission, Life Cycle Impacts of PVC, 2003	57	7	9	1	18	7	1	100
Estimated tonnage based on 2019 EU27 total	2,850,000	350,000	450,000	50,000	900,000	350,000	50,000	5,000,000

Source: European Commission, *Life Cycle Impacts of PVC*, 2003 (European Commission 2004). Tonnage data based on ECVM, *PVC Applications*, Accessed January 2021 (European Council of Vinyl Manufacturers 2017)

Table 3-15: PVC Consumption by product and Industry within the EU27 2019

Source	**Estimate of sector in which prodcuts are likely to be used**											**Total**
	Construction (European Council of Vinyl Manufacturers n.d.a)		**Construction & agriculture** (Plastic Pipe Shop n.d.)	**Construction** (European Council of Vinyl Manufacturers n.d.a) **& automotive** (British Plastics Federation n.d.)	**Construction, packaging & consumer** (European Council of Vinyl Manufacturers n.d.a) (European Council of Vinyl Manufacturers n.d.b) (British Plastics Federation n.d.) (Piedmont Plastics n.d.)	**Construction, automotive & furniture** (Kunststoffe International 2017)	**Construction, Agriculture & healthcare** (European Council of Vinyl Manufacturers n.d.a) (European Council of Vinyl Manufacturers n.d.b)	**Electronics**	**Packaging & healthcare** (European Council of Vinyl Manufacturers n.d.b) (Polymer Properties Database 2015)	**Multiple Sectors** (British Plastics Federation n.d.)	**Multiple Sectors**	
	Type of PVC products											
	Flooring	**Rigid plates**	**Pipes & fittings**	**Profiles**	**Rigid film**	**Coated Fabrics**	**Flexible tubes & profiles**	**Cables**	**Flexible film & sheets**	**Misc. Rigid**	**Other**	
Percentage Consumption												
ECVM, PVC Applications, 2017 [1]	7	2	22	27	8	3	2	7	7	6	9	100%
Tonnage Consumption												
Approx Volume [1]	350,000	100,000	1,100,000	1,350,000	400,000	150,000	100,000	350,000	350,000	300,000	450,000	5,000,000

Note: The tonnages have been calculated by applying the above proportions to total consumption derived from 2019 Eurostat data. This amounted to 5,974,000 tonnes; less exports of 1,636,000 tonnes, plus imports of 580,000, totally 4,918,500. (See data in Figure 10). The total tonnage value represents the quantity of **primary PVC** placed on the European market. If the same calculation is used with the PVC **product** data (Figure 11) the volumes are lower. However, the higher data is in good agreement the total consumption value quote by ECVM for PVC in Europe in 2017 – 5,000,000 tonnes. Given the uncertainties the slightly higher rounded figure has been used.

Sources: [1] ECVM, *PVC Applications*, Accessed January 2021 (European Council of Vinyl Manufacturers 2017). Tonnage data from the same source, cross referenced with: Eurostat, *Sold Production, exports and imports by PRODCOM list (NACE Rev. 2) – annual data* (European Commission 2021a)

Table 3-16: PVC Consumption by Product in EU27 plus UK, Norway, Iceland and Switzerland

Application	ECVM estimated tonnage, 2020 (t)	ECVM estimated tonnage, 2019 (t)
Pipes and fittings for wastewater and rainwater	679,000	681,000
Cables for electronics	284,000	320,000
Rigid profiles and sheets for window frames	610,000	613,000
Flexible profiles and foils for flooring	367,000	366,000
Packaging	356,000	427,000
Pipes and fittings for drinking water	279,000	281,000
Toys and sporting goods	64,000	63,000
Clothing (including footwear)	134,000	134,000
Automotive dashboards		
Medical and healthcare applications	366,000	329,000
Household/leisure/sports	189,000	205,000
Combined demand for the above applications	3,328,000	3,419,000
Total demand	4,770,000	-

Source: Data obtained through correspondence with ECVM dated 5th September 2021.

Note: ECVM's demand estimates are based on primary sales data collected by ARCO Association Management AG and further corrected for imports using publicly available data from Eurostat. Where data from ARCO were not available, ECVM have obtained data from a study by VinylPlus and extrapolated beyond 2018. ECVM's estimates include a significant volume of PVC demand not assigned to any one application, hence the discrepancy between the combined demand for listed applications and total demand.

3.4 EU sales and market values

The following section estimates market values and trends for PVC in the EU27, based on Eurostat data (European Commission 2021a). Primary PVC and processed PVC are assessed separately.

3.4.1.1 Product pricing within the EU27

Average PVC prices have been calculated by dividing the total sold production quantity in tonnes, by the total sold production value within EU27, for all different primary and processed PVC forms of PVC. This indicates primary unmixed PVC has an average price at €700 per tonne in 2019. Non-plasticised and plasticised mixed PVC are €1,310 and €1,160 per tonne respectively. Cellular polymer sheets are the highest value at an average of €3,650 per tonne.

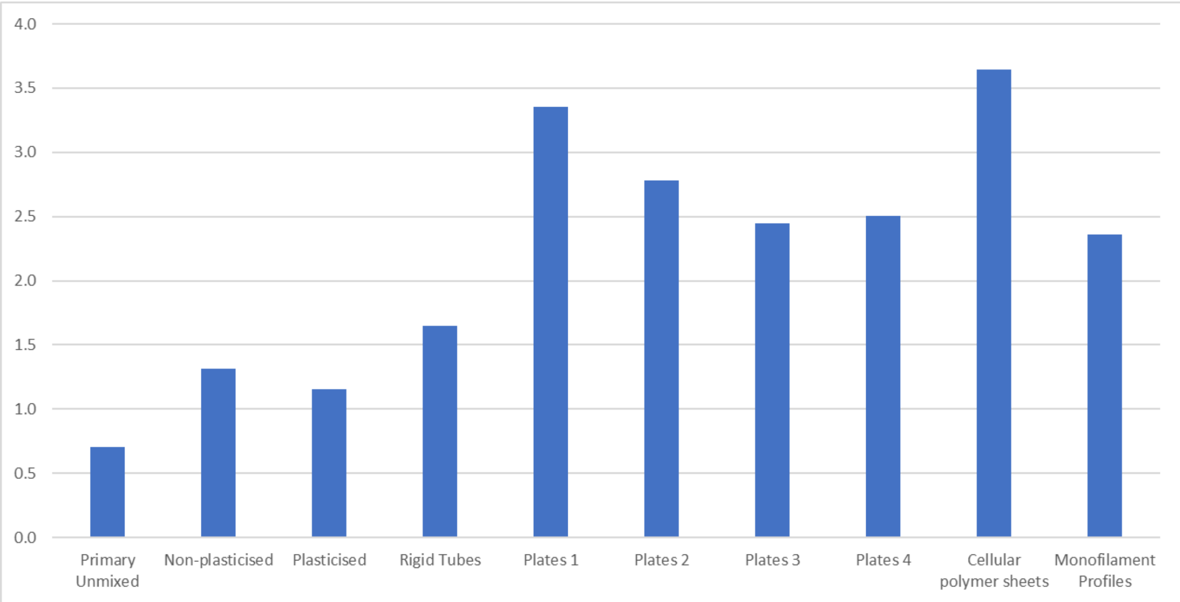


Figure 15. PVC product pricing in EU27 (1,000€/tonne) (2019) (European Commission)[36]

3.4.1.2 Primary PVC price trends

Since 2006, price per tonne for primary PVC has been relatively stable (Figure 15). As above, the data per tonne is calculated by dividing total sold production quantity by total sold production value from the Eurostat database (see Figure 10 and Figure 11). Primary unmixed PVC price per tonne has varied between 600 and 830 €/tonne between 2006 and 2019, and averaged 750 €/tonne. Plasticised and non-plasticised mixed PVC have varied slightly more but represent just over a quarter of the total market value of primary PVC. Non-plasticised PVC prices stood at around 1,300 €/tonne in 2019, with plasticised PVC at just under 1,200 €/tonne.

[36] Eurostat datasets:

Note that Floor Coverings and Other coverings of PVC have not been included as these are priced by meters squared.
Primary Unmixed - Polyvinyl chloride, not mixed with any other substances, in primary forms
Non-plasticised - Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms
Plasticised - Plasticised polyvinyl chloride mixed with any other substance, in primary forms
Rigid Tubes - Rigid tubes, pipes and hoses of PVC.
Plates 1 - Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness <= 1 mm
Plates 2 - Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness > 1 mm
Plates 3 - Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness <= 1 mm
Plates 4 - Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness > 1 mm
Cellular Polymer Sheets - Plates, sheets, film, foil and strip of cellular PVC
Floor Coverings - Floor coverings in rolls or in tiles and wall or ceiling coverings consisting of a support impregnated, coated or covered with PVC
Other coverings of PVC - Other floor, wall, ceiling... coverings of PVC

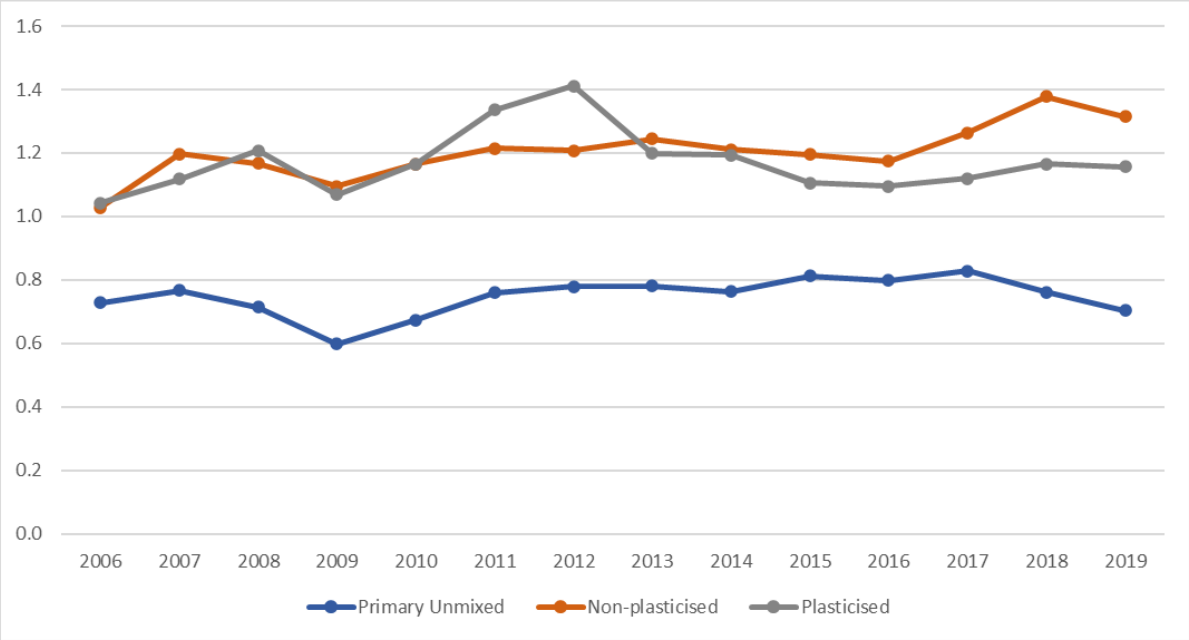


Figure 16. Primary PVC pricing in EU27 (1000€/tonne) since 2006 (European Commission 2021a)[37]

Total primary PVC production and import value, in the EU27, has been calculated by totalling the PRODVAL (sold production value) and IMPVAL (imported value) figures from the Eurostat database[38]. Figure 15 presents these data between 2006 to 2019. Following a dip in 2012, total production value remained relatively consistent at 5.6-5.8 bn€ until 2017, after which total value fell to 4.7 bn€ in 2019. Total imported value is much lower than production, but has slowly increased since 2009, from 0.33 bn€ to 0.60 bn€ in 2018. The trend in total production value for primary PVC shown in Figure 16 broadly reflects the change in primary PVC production over time (see Figure 6), and reflects the more recent fall in price shown in Figure 15.

The 2019 total market value based on production was 4.7 bn€. Including imports, the total market value of primary PVC placed on the European market was 5.2 bn€.

[37] Eurostat datasets:
Primary Unmixed - Polyvinyl chloride, not mixed with any other substances, in primary forms
Non-plasticised - Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms
Plasticised - Plasticised polyvinyl chloride mixed with any other substance, in primary forms

[38] As above, for the following datasets: Polyvinyl chloride, not mixed with any other substances, in primary forms; Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms; and Plasticised polyvinyl chloride mixed with any other substance, in primary forms

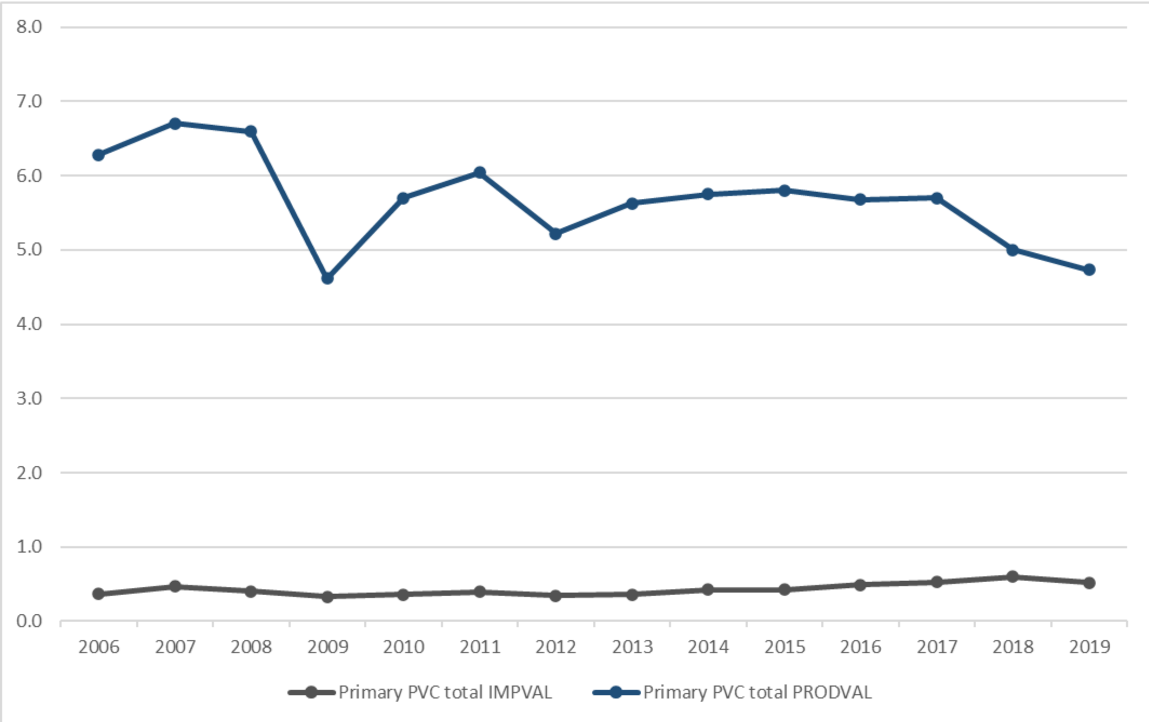


Figure 17. Primary PVC Production, Imported and Exported Value in EU27 (billion euros) since 2006 (European Commission 2021a)[39]

3.4.1.3 Processed PVC values

For processed PVC, total primary PVC production and imports in the EU27, has been calculated by totalling the PRODVAL and IMPVAL figures from the Eurostat database[40]. These data for 2006 to 2019 are shown below. Production value increased after 2009, to 9.92 bn€ in 2019. Somewhat lower than the market value leading up to 2008. The figure has not varied by more than 6% per year since 2011. At the same time imported value has more than doubled, from 0.59 bn€ in 2009 to 1.59 bn€ in 2019, increasing slowly year on year. The total value of goods placed on the EU27 market for processed PVC was around 11.5 bn€ in 2019.

[39] Eurostat datasets:

Primary Unmixed - Polyvinyl chloride, not mixed with any other substances, in primary forms

Non-plasticised - Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms

Plasticised - Plasticised polyvinyl chloride mixed with any other substance, in primary forms

[40] This is based on the following datasets: Rigid tubes, pipes and hoses of PVC; Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness <= 1 mm; Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness > 1 mm; Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness <= 1 mm; Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness > 1 mm; Plates, sheets, film, foil and strip of cellular PVC; Floor coverings in rolls or in tiles and wall or ceiling coverings consisting of a support impregnated, coated or covered with PVC; Other floor, wall, ceiling... coverings of PVC; and Monofilament with any cross-sectional dimension > 1 mm, rods, sticks, profile shapes, of polymers of vinyl chloride (including surface worked but not otherwise worked).

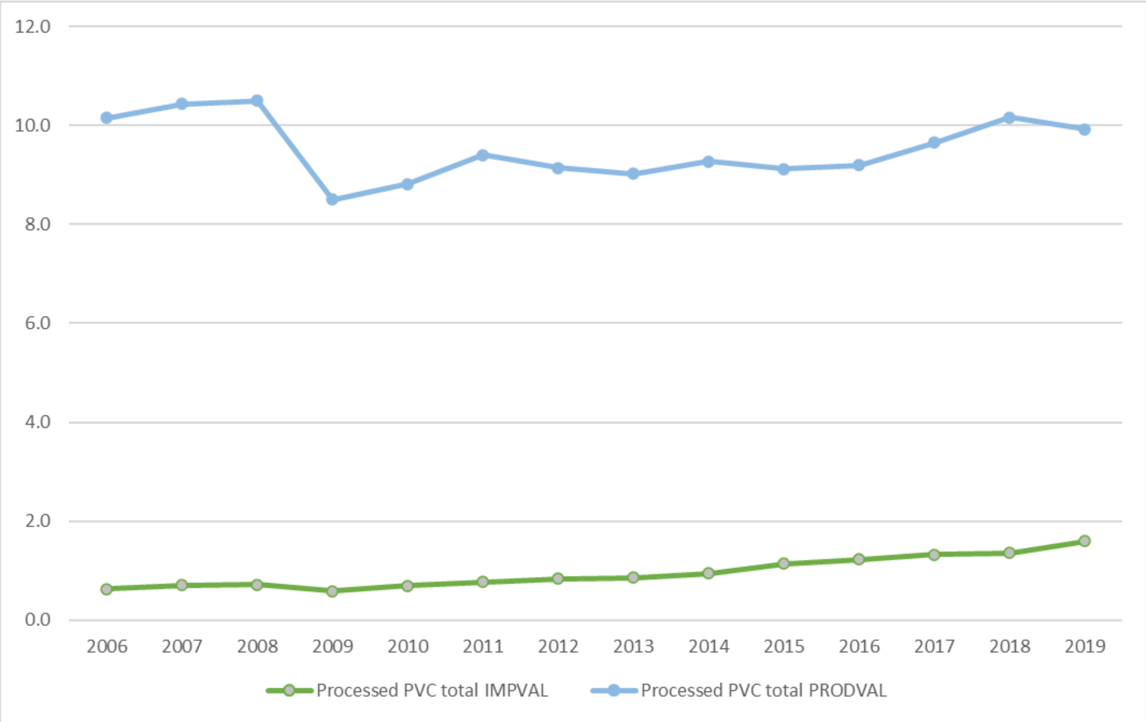


Figure 18. Processed PVC Production and Imported Value in EU27 (billion euros) since 2006 (European Commission 2021a)[41]

3.4.1.4 EU27 Total Market Value

Table 3-17 summarises data on EU27 PVC production value and import values.

- The primary PVC production value in 2019 is 4.74 bn€, 73% of which is within the manufacture of 'primary unmixed PVC'. The production value of processed PVC in 2019 is 9.92 bn€, with 'monofilament profiles' the largest contributor at 27%. Primary PVC imports represent a total 0.52 bn€, exports represent 1.50 bn€, with the total value of primary PVC placed on the EU27 market at 3.75 bn€. 'Primary unmixed PVC' is the largest contributor to primary PVC import and export values at 79% and 81% respectively.

- Processed PVC imports represent 1.57 bn€ and exports 2.17 bn€ total processed PVC placed on the EU27 market valued at 9.34 bn€. 'Floor coverings', 'monofilament profiles' and 'plates 3' account for 19%, 17% and 17% of the export value respectively, while 'other coverings of PVC' and 'floor coverings' contribute 32% and 25% to the total import value.

[41] Eurostat datasets:
Rigid Tubes - Rigid tubes, pipes and hoses of PVC.
Plates 1 - Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness <= 1 mm
Plates 2 - Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness > 1 mm
Plates 3 - Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness <= 1 mm
Plates 4 - Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness > 1 mm
Cellular Polymer Sheets - Plates, sheets, film, foil and strip of cellular PVC
Floor Coverings - Floor coverings in rolls or in tiles and wall or ceiling coverings consisting of a support impregnated, coated or covered with PVC
Other coverings of PVC - Other floor, wall, ceiling... coverings of PVC
Monofilament Profiles - Monofilament with any cross-sectional dimension > 1 mm, rods, sticks, profile shapes, of polymers of vinyl chloride (including surface worked but not otherwise worked)

Table 3-17: EU27 Total PVC value manufactured and placed on the market for EU27 in 2019 (bn€)

Product Type	Total Production Value	Total Exported Value	Total Imported Value	Total Value Placed on Market
Primary PVC				
Primary Unmixed	3.453	1.218	0.409	2.644
Primary Non-plasticised	0.423	0.075	0.032	0.380
Primary Plasticised	0.859	0.209	0.076	0.727
Total Primary PVC	4.736	1.502	0.517	3.751
Processed PVC				
Rigid Tubes	1.848	0.152	0.064	1.760
Plates 1	1.458	0.244	0.106	1.320
Plates 2	0.375	0.037	0.046	0.383
Plates 3	0.865	0.361	0.137	0.642
Plates 4	0.184	0.064	0.094	0.214
Cellular polymer sheets	0.416	0.232	0.152	0.336
Floor coverings	1.265	0.422	0.396	1.238
Other coverings of PVC	0.798	0.294	0.517	1.022
Monofilament Profiles	2.710	0.361	0.080	2.429
Total Processed PVC	9.919	2.167	1.592	9.343

Source: Eurostat, PRODCOM, NACE rev. 2, accessed January 2021 (European Commission 2021a)[42]

3.4.2 International price comparisons

The production cost of PVC resins is highly influenced by raw material prices, which include chlorine, ethylene & acetylene (Plastics Insight n.d.). Consultation feedback indicated that prices remained variable[43] but that PVC produced in China is extremely economically competitive. Here a different production process is typically used (see section 3.2). Similarly, production in the United States over recent years has benefited from decreases in raw materials costs associated with the development of shale gas (ICIS 2012).

Figure 18 provides available average price per tonne data of PVC for several non-EU countries. These prices are substantially larger than those provided for primary PVC in section 3.4.1.2, and

[42] Primary Unmixed - Polyvinyl chloride, not mixed with any other substances, in primary forms
Non-plasticised - Non-plasticised polyvinyl chloride mixed with any other substance, in primary forms
Plasticised - Plasticised polyvinyl chloride mixed with any other substance, in primary forms Rigid Tubes - Rigid tubes, pipes and hoses of PVC.
Plates 1 - Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness <= 1 mm
Plates 2 - Other plates, sheets, film, foil and strip, of PVC, containing >= 6 % of plasticisers, thickness > 1 mm
Plates 3 - Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness <= 1 mm
Plates 4 - Other plates, sheets, film, foil and strip, of PVC, containing < 6 % of plasticisers, thickness > 1 mm
Cellular Polymer Sheets - Plates, sheets, film, foil and strip of cellular PVC
Floor Coverings - Floor coverings in rolls or in tiles and wall or ceiling coverings consisting of a support impregnated, coated or covered with PVC
Other coverings of PVC - Other floor, wall, ceiling... coverings of PVC
Monofilament Profiles - Monofilament with any cross-sectional dimension > 1 mm, rods, sticks, profile shapes, of polymers of vinyl chloride (including surface worked but not otherwise worked)

[43] Pers. comm

the source does not state what form of PVC these figures refer to, nor to what extent variables such as currency fluctuations are reflected. China had the lowest trading price in 2017 at 2,031 €/tonne. Germany's price per tonne was middling compared to other international competitors, at 2,360 €/tonne. India and Japan sold on average at 2,617 and 2,814 €/tonne respectively. It is possible that these prices reflect differences in product types. Germany, for example is noted as competitive within the speciality PVC product market (Kunststoffe International 2016), but it is unclear why the stated price for India are amongst the higher range.

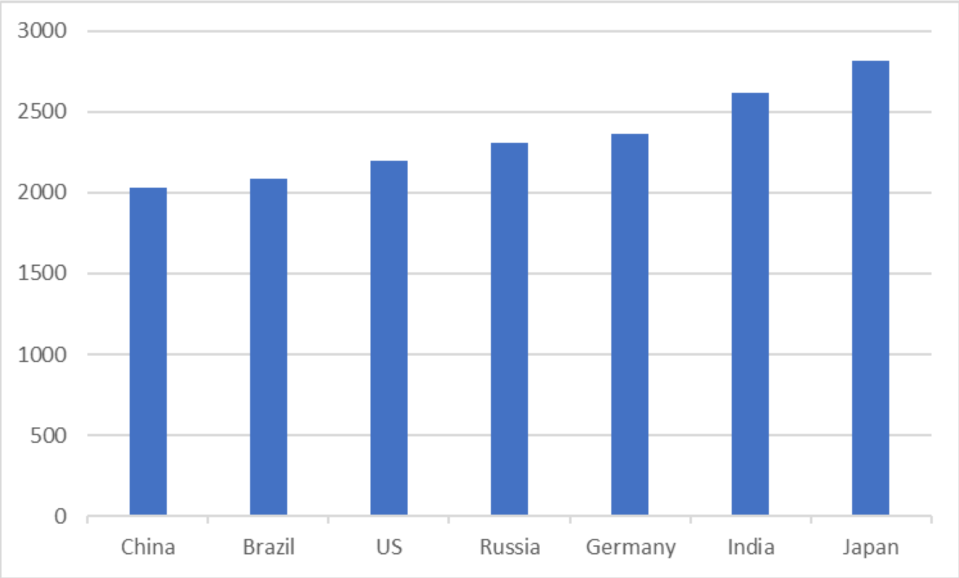


Figure 19. PVC price per tonne by country in 2017 (€/tonne) (Plastics Insight n.d.)

3.5 Projections to 2050

3.5.1 Short term forecasts from market intelligence publications

Figure 19 provides a near term forecast of the global PVC market value from 2018. It uses 2016 as the base year and forecasts to 2020. A further study in 2020 predicted a CAGR (compound annual growth rate) of 3.5% for the period 2016 to 2026 (Mordor Intelligence 2020). This forecast took into account the economic effects of COVID-19, but year by year forecasts for post 2020 were not publicly available. The largest and the fastest growing market in the forecast period was expected to be the Asia Pacific region, specifically China. One of the major drivers of this growth was anticipated to be increased use of PVC in the automotive industry. This reflected the expected contribution of PVC for weight savings/fuel efficiency as well as expected demand for use in electric vehicles. Demand in construction and in a growing number of healthcare applications was also expected to contribute.

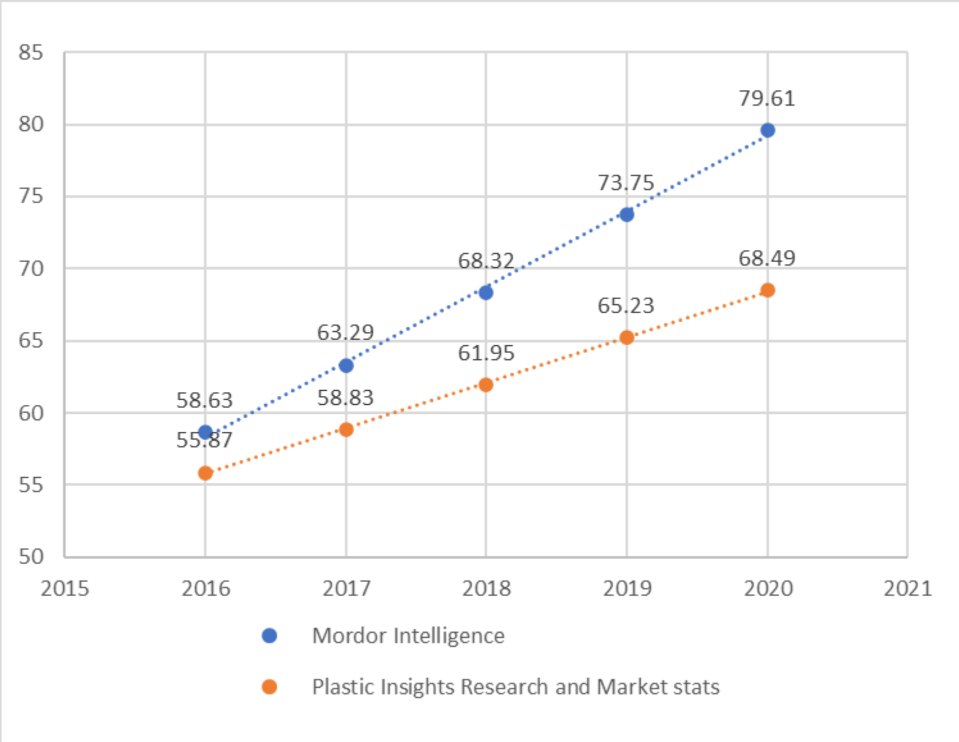


Figure 20. Forecasted global PVC market value (billion US$) from base year 2016 up to 2020 (Plastics Insight n.d.)

3.5.2 Potential trends to 2050

The information and data collated in the preceding sections have been used to project primary PVC production, primary and processed PVC imports, and waste arisings in the EU27 to 2050. It should be noted that production, trade and waste generation are influenced and determined by a complex array of economic, political, industrial and regulatory factors which cannot be directly accounted for in these projections. As such any forecasting exercise is subject to significant uncertainty and the forecasts presented here should be interpreted as potential trends rather than accurate predictors of tonnages. The approach used is explained below.

3.5.3 PVC Production Projections

Primary PVC production data for the EU27 area for 2008 to 2019 were obtained from the Eurostat PRODCOM database (European Commission 2021a). Tonnages of primary PVC production were projected to 2050 by extrapolating the linear trend, based on historic data. This forecast has been based on two separate periods.

- First, 2008-2019; this was used as a longest consistent time series data available. However, the data indicate two distinct periods in the overall trend; quite a fast decrease from 2008-2012, with relative stability thereafter[44]. So, a shorter period was also used, between 2012-2019. The orange and yellow lines in Figure 20 display the high and low forecasts for primary PVC production to 2050. These represent a reasonable high and low bound for future overall trends. Either the EU market will continue to lose market share to Asia, and the recession associated with COVID-19 may accelerate that trend; or the market adjustment that took place some 10 years ago will mean that EU production can remain competitive and relatively stable.
- An additional forecast of primary PVC production has been produced based on gross domestic product (GDP) projections to 2050. This provides an overall assessment of the

[44] See for example: https://www.icis.com/explore/resources/news/2013/05/17/9669605/chemical-profile-europe-pvc/ (ICIS 2013)

excepted demand strength. GDP data for the EU27 area for 2008 to 2019 were obtained from the Eurostat NAMA_10_GDP database (European Commission 2021c).Macroeconomic projections for the euro area developed by the European Central Bank (European Central Bank 2020) were used to identify EU27 GDP forecasts for 2020, 2021, 2022 and 2023. Beyond 2023, annual GDP was predicted by extrapolating the linear growth trend observed between 2008 and 2019. This projection therefore takes into account the economic damage from COVID-19 but assumes that economic impacts associated with the COVID pandemic will have subsided by 2023[45], and pre-COVID growth trends will resume thereafter.

A least squares regression was conducted to determine the linear trend between primary PVC production (dependent variable) and GDP (independent variable) during the period 2008 to 2019. The resulting regression equation was subsequently applied to the GDP projection to develop a primary PVC production projection informed by anticipated economic conditions. The grey line in Figure 20 displays projected primary PVC production based on this GDP forecast. This falls somewhere between the high and low figures.

Overall, the linear extrapolations of PVC production trends indicate that EU27 production may continue to decline, perhaps to 2.5 Mt by 2050, but it is also possible that production stabilises at a little over 6 Mt. The economic-based forecast projects a decrease to just under 4 Mt. Discussions held with industry representatives indicate that production within the EU27 is likely to decline in future years as China, Turkey and Asia-Pacific continue to expand as major global producers. The balance of likelihood is therefore weighted toward further declines of PVC production in Europe.

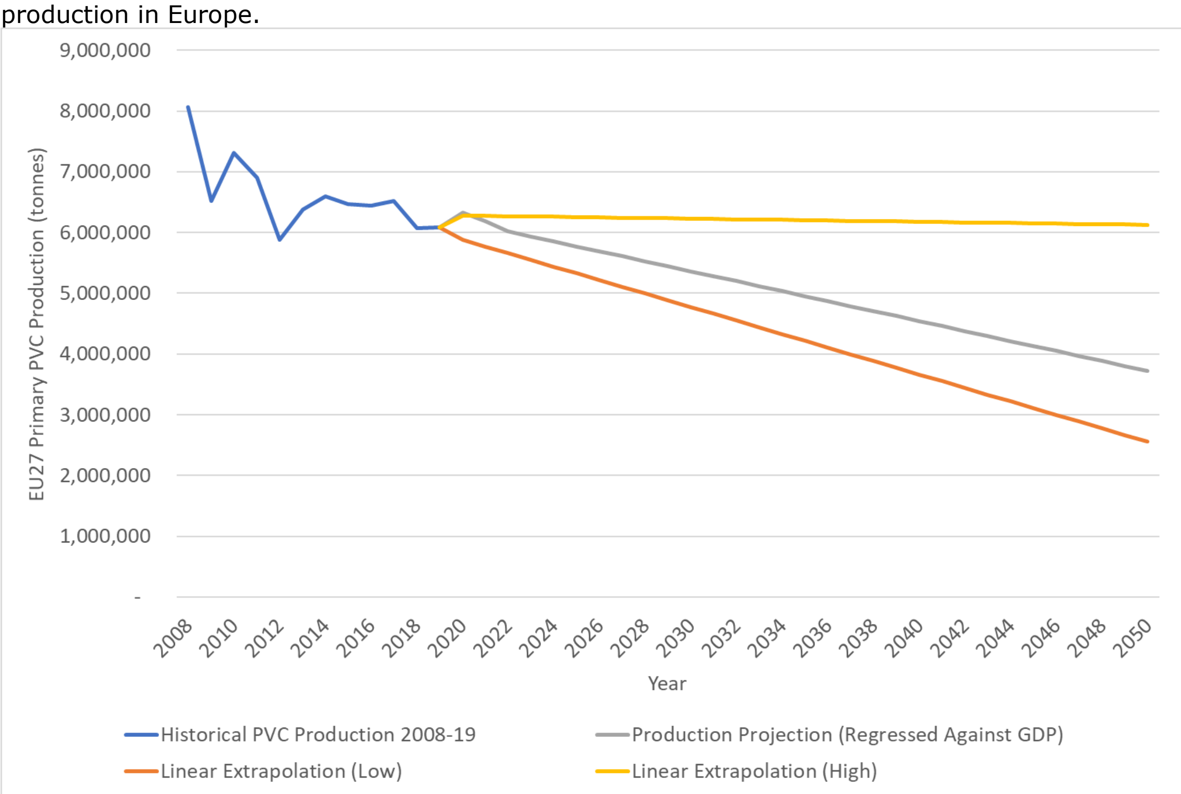


Figure 21. Primary PVC production projections for the EU27 to 2050

[45] This is in line with the short term ECB forecast which indicates that GDP will fall sharply from 2019 to 2020, but starts to increase again thereafter, roughly following pre-COVID growth to 2023.

3.5.4 PVC Import Projections

A similar exercise has been undertaken for PVC imports. Primary and processed PVC imports data for the EU27 area for 2003 to 2019 were obtained from the Eurostat PRODCOM database (European Commission 2021a). Tonnages of primary PVC imports and processed PVC imports were projected to 2050 by simply extrapolating the linear trends for the periods 2003-2019 and 2003-2014 into the future. The rationale for selection of the two time periods are the same as that for production, the two trends mirror one another. The orange and yellow lines in Figure 21 and Figure 22 show the high and low forecasts for primary PVC imports and processed PVC imports to 2050, respectively.

As above, an additional forecast of PVC imports has been produced using the same least squares regression approach linked to GDP forecasts as detailed above for PVC production. The grey lines in Figure 21 and Figure 22 show projected primary and processed PVC imports, respectively, based on the GDP forecast.

The projections all indicate that primary PVC imports will increase, perhaps to around 0.7 Mt and 1 Mt in 2050, while processed PVC imports, whilst rising, are expected to be somewhat lower (c. under 0.5 Mt). This is consistent with the declining primary PVC production in the EU27.

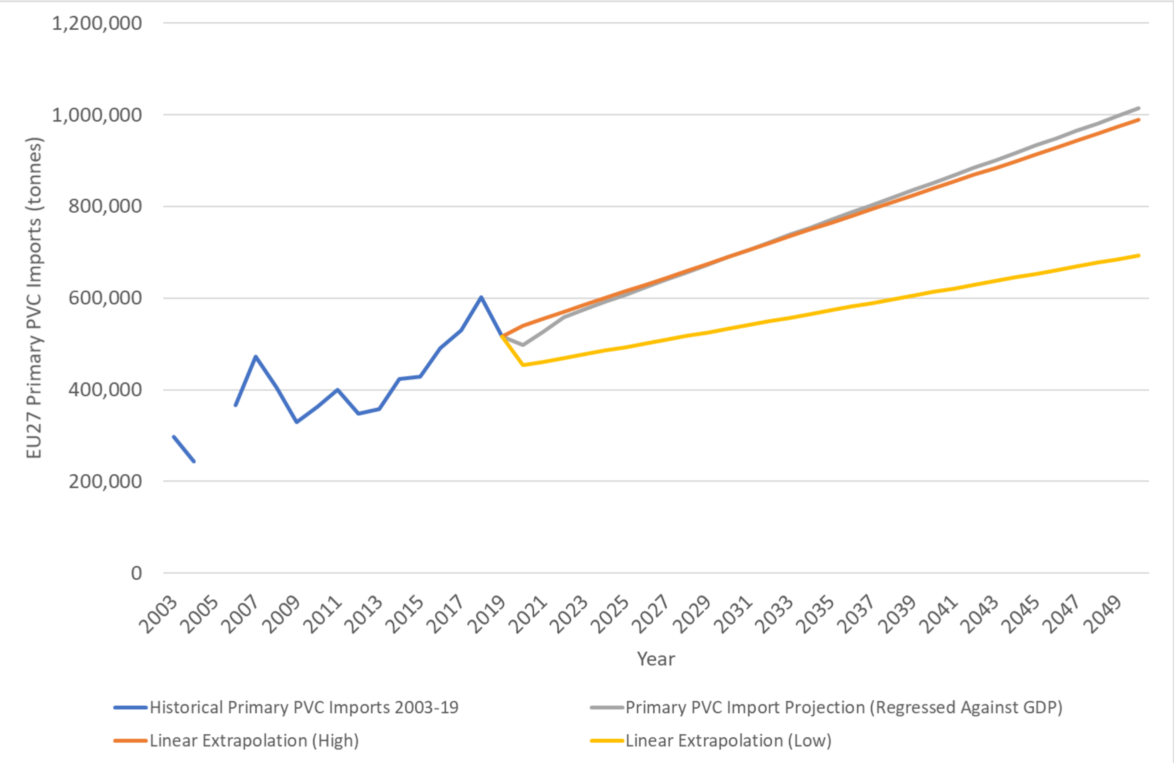


Figure 22. Primary PVC imports projections for the EU27 to 2050

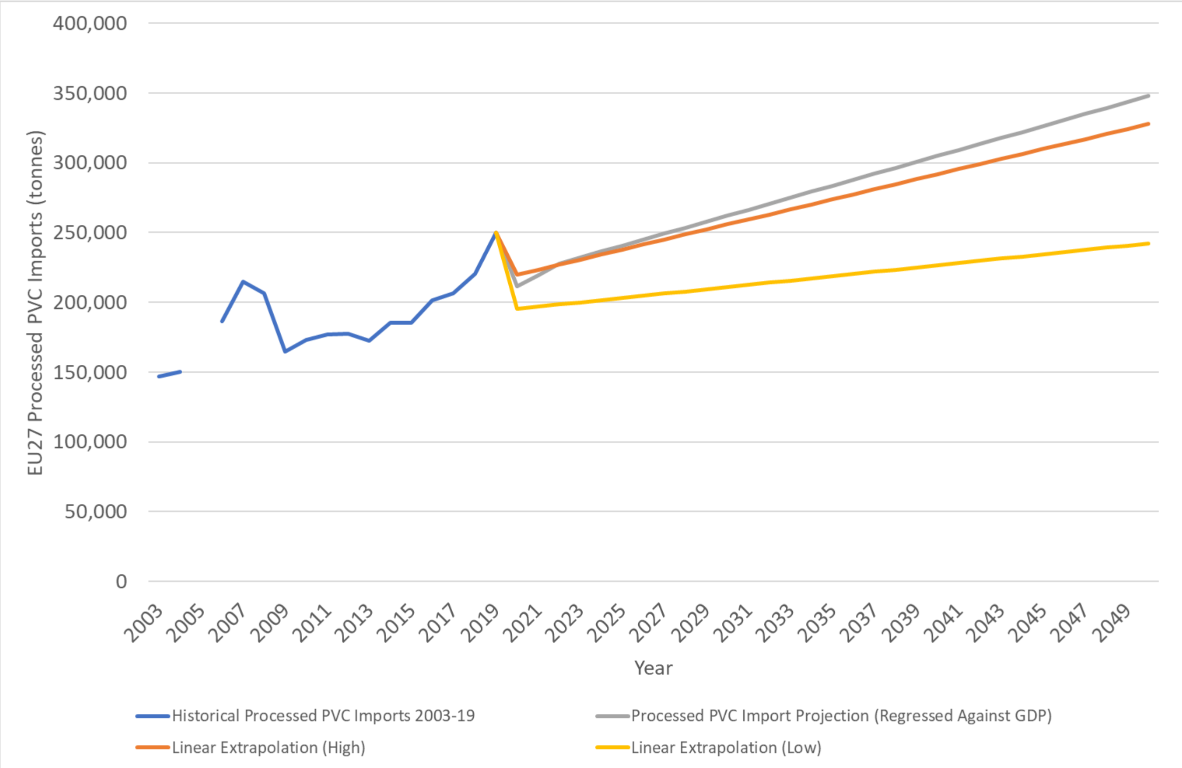


Figure 23. Processed PVC imports projections for the EU27 to 2050

3.5.5 PVC Waste Arisings Projections

The data on waste arisings is more limited hence any forecasts are subject to even higher uncertainty and in this case are subject to several compounding factors. A study conducted by the Nordic Council of Ministers (Fråne et al. 2018) examining PVC waste treatment concluded that PVC waste generation per capita was approximately 5 kg in 2018. Future waste generation - in the absence of more precise information - could be affected by overall demographic changes. Future population projections for the EU27 were obtained from Eurostat (European Commission 2021d). Applying the per capita waste arising figure to the Eurostat population projections, the order of magnitude of possible future waste arisings would stay largely constant, in the order of 2 million tonnes. Figure 23 displays a simple estimate based on a constant per capita waste arising for EU27 to 2050.

This approach assumes that per capita PVC waste arising will remain static after 2018. In reality, future waste policy and circular economy initiatives intend to achieve in a decrease in per capita waste generation across all waste streams[46], but it is not possible to accurately account for this in the projections. Nor it is clear how imports and exports of PVC (and waste) may change over time. Whilst the projection indicates that PVC waste arising may peak at over 2 Mt around the middle of the 2020s and decline thereafter to 2050, this suggests rather modest reductions, which largely reflects projected population trends in the EU27. This should be treated as a speculative finding; further data on PVC waste and actual outturn volumes should be monitored.

[46] See for example: European Commission (2020) Circular Economy Action Plan: For a Cleaner and more Competitive Europe. Available: https://op.europa.eu/en/publication-detail/-/publication/45cc30f6-cd57-11ea-adf7-01aa75ed71a1/language-en/format-PDF/source-170854112

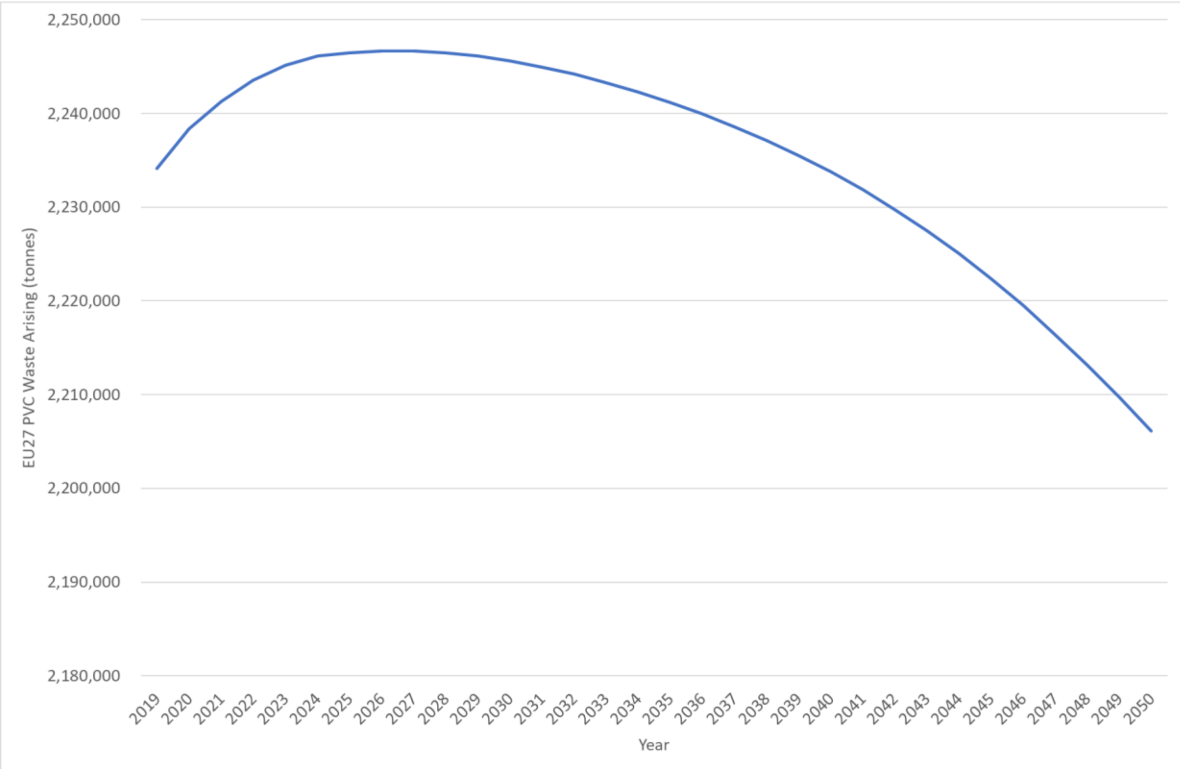


Figure 24. PVC waste arising projections for the EU27 to 2050

4. GENERAL FINDINGS ON MANAGEMENT OF PVC WASTE

Key messages – waste management of PVC
PVC waste generation • The available literature seems to indicate a decline in the volumes of generated PVC waste in the EU during the past few decades. • However, it is difficult to determine whether the assumed decline in generation volumes can be expected to continue in the future. Higher volumes could be generated in the future, if subsequent generations of PVC products reach their end-of-life phase. • Rigid PVC accounts for 63% of the generated PVC post-consumer waste whereas flexible PVC post-consumer waste accounts for 37% (Conversio Market & Strategy GmbH 2021; European Commission 2000). • 44.1% of PVC post-consumer waste is generated by the building and construction industry, 19.6% through packaging, 7.6% in the electro and electronic industry, 3.8% by the automotive industry and 24.9% by other segments (Conversio Market & Strategy GmbH 2021). PVC waste treatment • Data in the literature indicates that only a very limited amount of PVC waste has been directed toward material recycling during the second half of the last century. The literature also suggests a considerable volume of PVC waste being present in landfills in the EU. • Conversely, the available literature indicates a relative rise in recycled volumes and energy recovery during more recent years, as well as a relative decline in landfilling. In terms of recycling volumes of PVC waste by different industries, a considerable variation can be concluded. • The majority of PVC recycling takes place as part of the voluntary initiative by VinylPlus. The initiative entails a voluntary commitment of the European PVC industry. In 2020, 730,000 tonnes of PVC waste have been recycled via open-loop and closed-loop recycling within the VinylPlus initiative. • However, it is unclear how much post-consumer waste actually was recycled by VinylPlus as compared to pre-consumer waste. • Section 4.2.3 provides summary mass flows of PVC waste generation and treatment routes in the EU. Classification of PVC waste • The question whether PVC waste (or specific types thereof) should be classified as hazardous or non-hazardous is a highly relevant issue, as this has considerable influence on the manner in which such waste is to be managed, including its recycling and disposal. • Relevant actors and authorities seem to classify PVC waste as non-hazardous based on the reasoning that the additives are bound in the PVC matrix and, as such, do not migrate or become bioavailable. • In this regard, it is important to note that various literature sources indicate that migration of additives from plastics takes place, since such additives are not covalently bound in the plastic matrix (see chapter 2). • However, it should also be noted, that currently, data on migration and bioavailability of PVC additives is limited (see chapter 2). • From a precautionary perspective, further assessment of the classification of PVC as hazardous or non-hazardous may be necessary for the determination of the most suitable waste management options for PVC.

4.1 Introduction

Due to the large amounts of PVC used in various products, the long service life of most of these products and the use of various additives, it is important to elaborate on current waste management practices for PVC waste. As such, this chapter gives a general overview of the PVC waste generation volumes, as well as current waste treatment practices in the EU. A more detailed analysis concerning PVC waste management, most notably its recycling and disposal are subsequently discussed under chapters 5 and 6. When reading this chapter, as well as the subsequent chapters, it is important to bear in mind that waste management of PVC is influenced by several factors, namely:

- Legal and organisational factors
- Economic factors
- Health and environmental factors
- Technical factors

Figure 24 below provides a basic overview of these factors, while Annex 3.1 to this report provides a more detailed overview. In addition, factors which influence specific aspects of the PVC waste management practice are assessed in more detail under the relevant chapters and their sections.

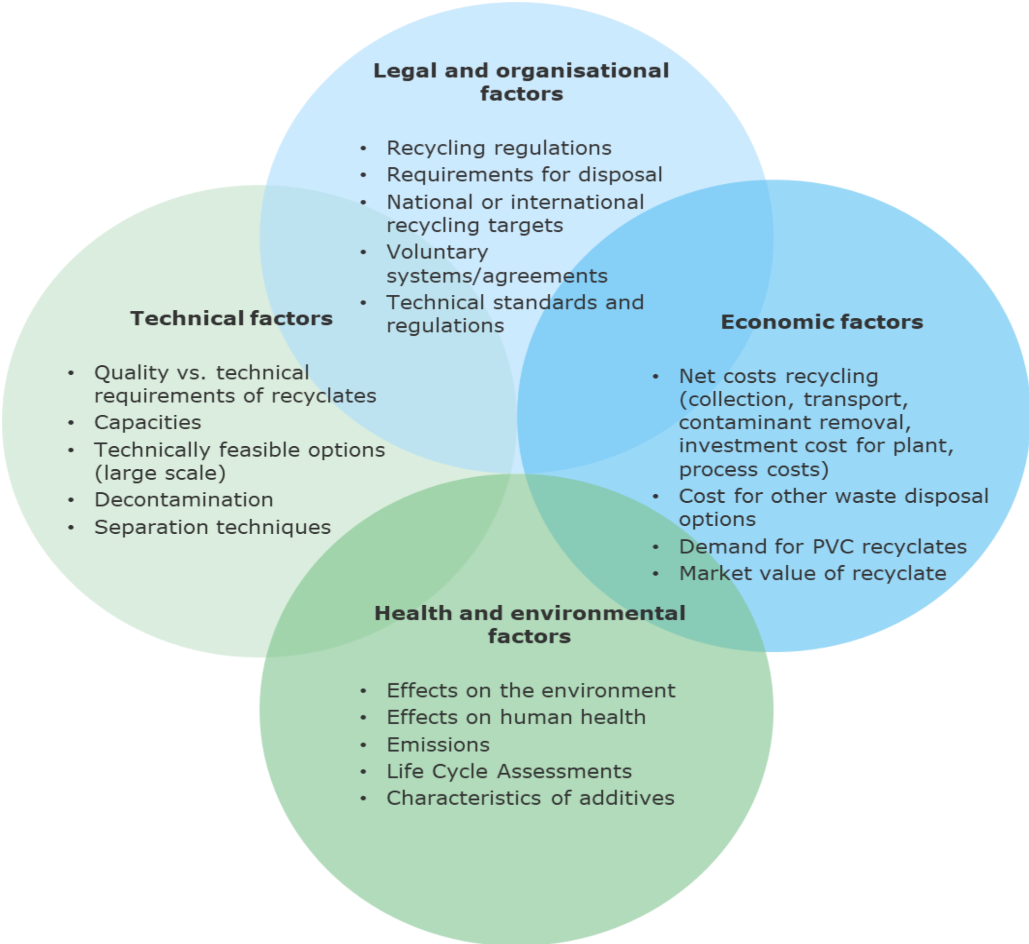


Figure 24. Influencing factors of PVC waste management (based on (Plinke et al. 2000)(Kelly et al. 2005)**)**

4.2 General data on PVC waste generation and treatment

The following sections provide a general overview of PVC waste amounts and how their treatment in the past and today in the EU.

4.2.1 PVC waste generation

The available quantitative data indicate the following:

- According to Conversio Market & Strategy GmbH (2021), in 2020 2.9 million tonnes of pre- and post-consumer PVC waste were generated per year in the EU[47] . From this total, pre-consumer waste accounted for 0.48 (16.5%) million tonnes and post-consumer waste for 2.4 million tonnes (83.5%).
- Rigid PVC accounts for 63% of the PVC post-consumer waste whereas flexible PVC post-consumer waste accounts for 37% (Conversio Market & Strategy GmbH 2021; European Commission 2000).

[47] EU 27+ UK, CH, Norway

- By sector, 44.1% of PVC post-consumer waste is generated by the building and construction industry, 19.6% through packaging, 7.6% in the electro and electronic industry, 3.8% by the automotive industry and 24.9% by other segments (Conversio Market & Strategy GmbH 2021).
- Fråne et al. stated that 2.5 million tonnes of PVC waste (not specified if only post- or post- and pre-consumer waste) is generated every year in the EU (2018), which is roughly in line with the order of magnitude provided by Conversio Market & Strategy GmbH.

Available historical data seems to indicate a decline in the volumes of generated PVC waste in the EU during the past few decades:

- For 1998, Argus (2000) indicates a PVC waste volume in the EU of 3.5 million tonnes (not specified if post- or post- and pre-consumer waste).
- For 1999 Plinke et al. (2000) estimated the annual (post- and pre-consumer) PVC waste generation in the EU to 4.1 million tonnes.
- Brown et al. (2000) estimate the amount PVC post-consumer waste to 3.6 million tonnes.

These forecasted volumes are significant higher than the 2.4-2.9 million tonnes indicated in the literature as current generation volumes. However, it should be considered that large volumes of PVC are used for long-life products (e.g. window frames and pipes). The average lifetime is 30-40 years but could also reach 100 years (Sadat-Shojai and Bakhshandeh 2011)(expert input Focus Group Recycling 2021). Therefore, it is difficult to determine whether the assumed decline in generation volumes can be considered a stable trend. Higher volumes could be generated in the future, if subsequent generations of PVC products reach their end-of-life phase.
Focus Group

Definition pre- and post-consumer waste (see also Annex 3.1)

Pre-consumer waste:
In this report, pre-consumer PVC waste includes waste generated during the production of final and intermediate products. However, it should be noted that such PVC will only be considered waste when it cannot be considered a by-product in accordance with Article 5 of the WFD.

However, this report will consider pre-consumer waste as all PVC residues from production processes as it is not possible to ascertain in a general manner whether the status of by-product applies. When pre-consumer material is not considered waste (i.e. by-product), it should not be included in waste statistics. However, some companies and organisations include production waste in waste statistics (VinylPlus, 2019).

Post-consumer waste:
Post-consumer waste means waste produced by end consumers or commerce (e.g. cut offs produced by end-consumers or, in some cases, commerce or installation waste from the handling or installation of products).

4.2.2 PVC waste treatment options

4.2.2.1 Historical data (1960-2012)

Relevant historical data on PVC waste treatment can be found in an aggregated model for the European PVC cycle which covers the period between 1960 to 2012 (Ciacci, Passarini, and Vassura 2017).

- According to this model 73 million tonnes of PVC post-consumer waste have been treated during this period.
- Of this total, 22 million tonnes were estimated to have been directed towards material recycling. However, it was estimated that, only 9 million tonnes of this input have been

actually materially recycled between 1960 and 2012, while 10 million tonnes were landfilled, and 3 million tonnes incinerated/energetically recovered.

- In total this means that from 1960 to 2012 12 million tonnes (16.4%) have been incinerated/ energetically recovered and the biggest share with 52 million tonnes has been landfilled (71.2%)
- From the 9 million tonnes of PVC waste that have been recycled between 1960 and 2012 only 3 million tonnes effectively supplemented the total amount of 220 million tonnes of primary PVC production (accounting for 1.3% of input material) (Ciacci, Passarini, and Vassura 2017).

4.2.2.2 More recent trends

More recent literature indicates a relative rise in recycled volumes and energy recovery, as well as a relative decline in landfilling. In 2020 in total 35.4% pre- and post-consumer PVC waste in the EU have been recycled (1 million tonnes), 45.6% have been used for energy recovery (1.3 million tonnes) and 19% have been disposed (0.55 million tonnes). For PVC post-consumer waste the shares differ. 24.6% of the PVC post-consumer waste has been recycled, 53.1% has been used for energy recovery and 22.3% has been disposed (Conversio Market & Strategy GmbH 2021).

PVC is almost completely recycled via mechanical recycling[48] processes. According to Conversio Market & Strategy GmbH (2021) less than 1,000 tonnes (<0.5%) are recycled via feedstock recycling (i.e. chemical recycling[49]). However, it is important to note that Conversio Market & Strategy GmbH (2021) counts the use of plastic (including 1% PVC) in the blast furnace process of the plant in Linz as chemical recycling, whereas this study does not[50]. Numbers from Germany show that in 2017 27% of the PVC post-consumer waste (37% of pre- and post-consumer waste) was recycled (99.6% mechanically; 0.4% chemically) (Conversio Market & Strategy GmbH 2018). Based on expert input at the conducted focus group on recycling these proportions also apply for the EU (Focus Group Recycling 2021). PVC recyclates (generated from pre- and post-consumer waste) account for 13.5% of the total processing amount of PVC (Conversio Market & Strategy GmbH 2018). Germany, the UK and the Netherlands (having one of the biggest recycling company in Europe for PVC (Van Werven)) are seen as the major recycling countries with regard to PVC (Sadat-Shojai and Bakhshandeh 2011) (Fråne et al. 2018).[51]

4.2.2.3 PVC and energy recovery

Conversio Market & Strategy GmbH (2021) states that energy recovery of PVC post-consumer waste mainly takes place in municipal waste incineration plants (92.5%; 1.2 million tonnes), while only 7.5% (0.097 million tonnes) of the energetically recovered waste has been used in power plants or as waste derived fuel. The relatively minor importance of incineration without energy recovery in EU Member States was confirmed by CEWEP (input Focus Group Disposal 2021).

[48] Mechanical recycling includes recycling processes where the material is treated mechanically through grinding, sieving, regranulating and compounding. Thereby the polymer structure does not significantly change (see Annex 3.1)

[49] Chemical or feedstock recycling refers to the conversion of plastic polymers into their monomers, or basic chemicals. Processes that use the energy content of PVC but not the carbon does technically not belong to chemical recycling (see Annex 3.1).

[50] This report does not count the treatment of PVC in blast furnaces chemical recycling, since PVC is not pre-treated and is used as a reducing agent together with mixed waste.

[51] It is interesting to note that in Denmark, the Danish waste ordinance obliges the municipalities to establish a collection scheme for PVC waste from households as well as to ensure recycling of collected recyclable PVC waste, and that collected non-recyclable PVC waste is disposed of (§ 31 Affaldsbekendtgørelsen).

4.2.2.4 PVC and landfilling

EU Member States Germany, Austria, the Netherlands, Sweden, Denmark, Luxembourg, Belgium and Finland have adopted landfill restrictions for plastic waste.[52,53] However, PVC waste is still directed to landfills in some EU Member States (input Focus Group Disposal 2021). This is supported by the data of Conversio Market & Strategy GmbH (2021) which indicates that 19% of all pre- and post-consumer PVC waste generated in the EU27+3 was directed towards landfills

4.2.2.5 Trends by sector

Management routes for post-consumer PVC waste differ significantly per relevant industry. An overview as presented by Conversio Market & Strategy GmbH (2021) is included in the table below (discussions on the data can be found in chapter 6.3.1):

Table 4-1: PVC post-consumer waste management by major industries (Conversio Market & Strategy GmbH 2021)

Industries	Waste generation [kt]	Mechanical recycling [%]	Energy recovery [%]	Disposal [%]
Packaging	478	15.3	66.1	18.6
Building and construction	1,075	33.0	44.0	23.0
Automotive	92	8.7	60.9	30.4
Electro and electronic	185	47.6	38.9	13.5
Others	605	12.6	61.8	25.6
Total	**2,435**	**24.6**	**53.1**	**22.3**

In terms of recycling of PVC waste by different industries, a considerable variation can be concluded. Recycling rates are relatively low for packaging waste (15.3%) and the automotive industry (8.7%). The building and construction industry (with a recycling rate of 33%) includes various products with very different product-specific recycling rates (Conversio Market & Strategy GmbH 2021). While PVC window profiles are recycled relatively often (in 2013/2014 44.9%), flooring (1.1%) or roofing and waterproofing applications (22%) have low recycling rates (Potrykus and Milankov 2015). Within the category electro and electronic mainly the recycling of PVC cables lead to the relatively high recycling rates in this industry segment (Conversio Market & Strategy GmbH 2021). Not specifically listed are other consumer goods, healthcare/medical devices or agricultural waste (e.g. agricultural films, pipes). These product PVC waste streams are seldomly directed towards recycling. PVC waste from healthcare and the agricultural products is often not generated in sufficient volumes to allow for a feasible operation of a PVC recycling system (Plinke et al. 2000)[54].
Despite this, for PVC waste from healthcare some initiatives/pilot projects exist, for example the recently launched project VinylPlus Med from VinylPlus (Vinylplus 2020d). This builds on the success of the VinylPlus funded project RecoMed in the UK (Hansen and Tobias 2017)[55].

According to Potrykus and Milankov (2015) the losses from recycling input to output of PVC window profiles, pipes and fittings, cables, floorings, roofing and waterproofing vary between 0% and 4%. As such, the recycling processes applied for PVC waste seem to be relatively material-efficient.

[52] Norway and Switzerland have also adopted restrictions on landfilling

[53] In Finland only three landfills are licensed to accept (limited) amounts of PVC waste. In Norway on the other hand landfilling PVC waste is allowed, but since the gate fees are high this treatment option is uncommon (Fråne et al. 2018)..

[54] These waste streams are partly separately collected but PVC is not the target material

[55] Since several years, the initiative RecoMed collects and recycles medical waste (see Annex: 4.3). In 2019, 9,153 kg and in 2020, 1,949 kg were collected (input BPF 2021a).

4.2.2.6 PVC waste recycling by VinylPlus

Analysis of the current recycling rates and practices for PVC waste in the EU must take into account the ongoing initiative of VinylPlus. This entails a voluntary commitment of the European PVC industry (including resin and additives producers, converters and recyclers) with the aim of supporting the transition to "an integrated, cross-border, sustainable and circular PVC value chain(VinylPlus 2021a)". It is important to note that VinylPlus initiative covers data from 60-70% of PVC recyclers and not the whole PVC recycling industry (interview with VFSE 2021)[56].

In 2020, 730,000 tonnes of PVC waste have been recycled within the VinylPlus program (regional scope is EU-27 plus Norway, Switzerland and the UK). Thereby, VinylPlus fell short of its self-imposed goal of recycling 800,000 tonnes of PVC waste by 2020 (Vinylplus 2020a). Depending on the analysed initiative, the applied definition 'recycling' includes post-consumer waste recycling and pre-consumer waste recycling (VinylPlus 2019). Data onhow much post-consumer waste actually was recycled by VinylPlus as compared to pre-consumer waste is not available, although this would be a highly relevant information.

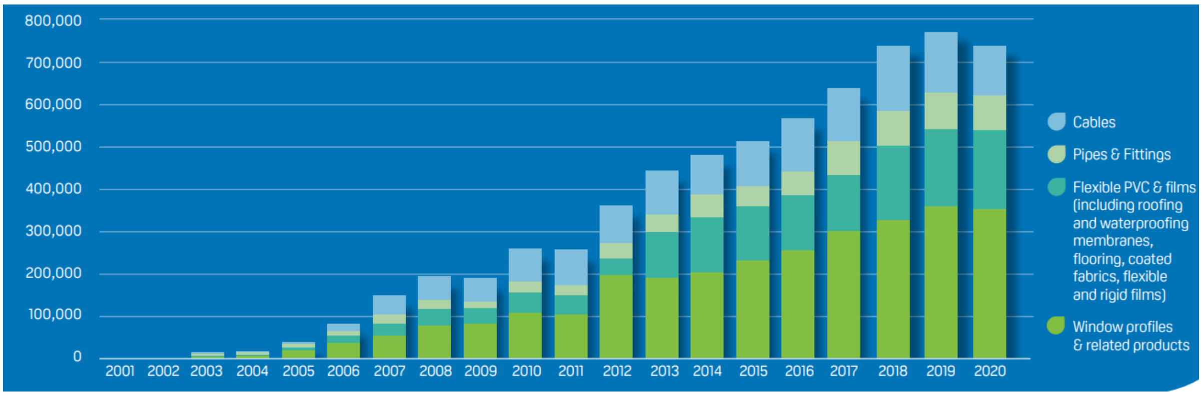


Figure 25. Timeline PVC recycled within the VinylPlus Framework (VinylPlus 2021c)

The total amount of recycled PVC under the VinylPlus program has steadily increased since 2001, with a slight decrease from 2019 to 2020 (VinylPlus 2021c) (see Figure 25). The registered recycled quantities of PVC within VinylPlus can be allocated to the following applications:

- 48% Window profiles and related profiles
- 25% Flexible PVC and films (incl. flooring, roofing and waterproofing membranes, flooring, coated fabrics, flexible and rigid films)[57]
- 11% Pipes and fittings
- 16% Cables.

About 60% of the recycled materials consist of rigid PVC (i.e. profiles, pipes and fittings), while 40% consists of flexible PVC waste (i.e. cables, films, flooring, coated fabrics). VinylPlus further aims to recycle 900,000 tonnes of PVC per year into new products by 2025. By 2030 they committed to recycle at least 1 million tonnes (VinylPlus 2021b).

4.2.3 Summary overview of PVC waste generation and treatment

The key findings from this section is summarised in a mass flow diagram for PVC post-consumer waste and one for PVC pre- and post-consumer waste in the EU, below. The mass flow on pre- and post-consumer waste also includes the recycled amounts reported by VinylPlus and the shares of different product groups that have been recycled via VinylPlus members. Please note

[56] Volumes provided by VinylPlus coming from recyclers of the network (audited)

[57] 2019 Flooring accounted to 0.4% and coated fabrics to 0.9% of the PVC recycled within the VInylPlus framework. The data for 2020 does not list these products separately (Vinylplus 2020a).

that the data from Conversio Market & Strategy GmbH (2021) assumes that 300 tonnes of PVC waste was chemically recycled in 2020, whereas we do not count the use of PVC as reducing agent as a chemical recycling process.

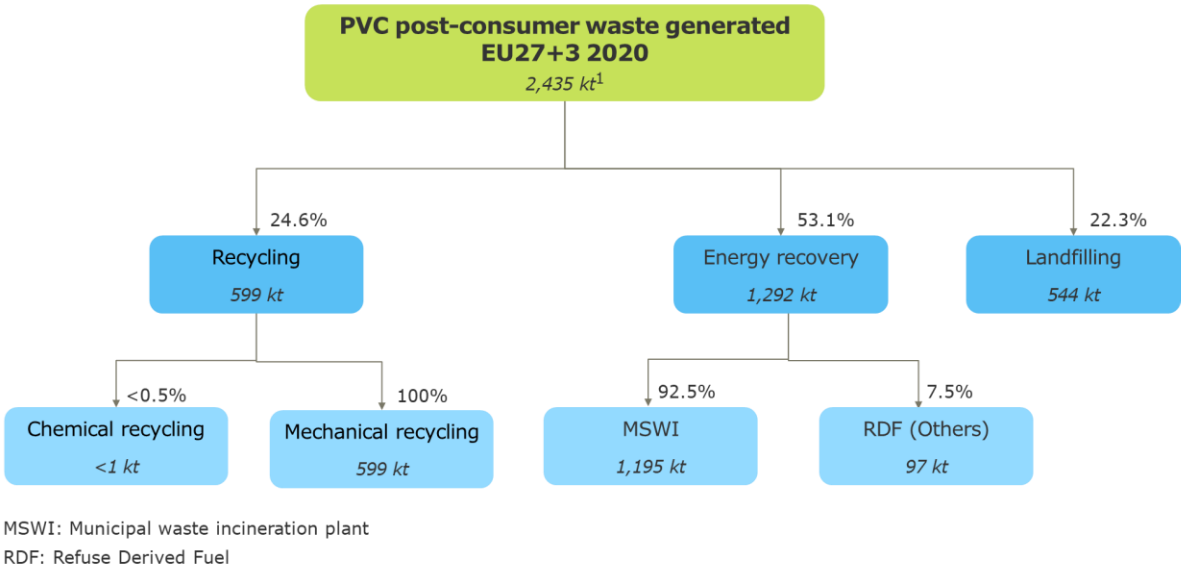


Figure 26. Overview waste management of PVC post-consumer waste in the EU27+3 (Conversio Market & Strategy GmbH 2021)

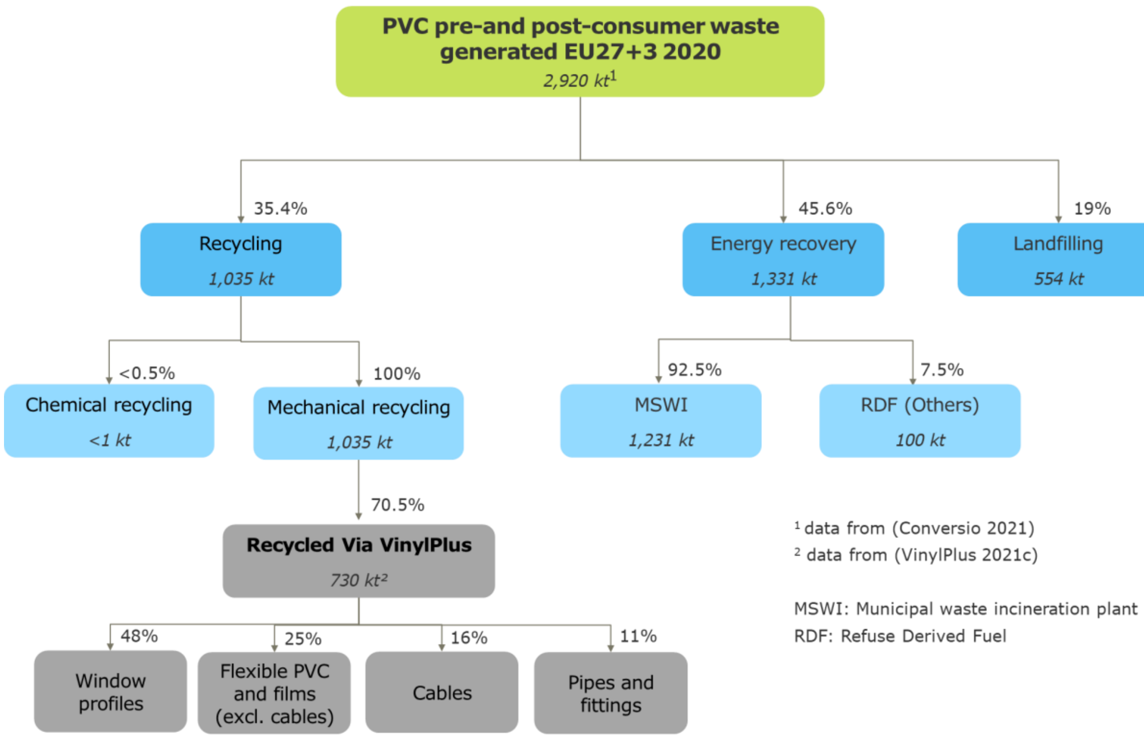


Figure 27. Overview waste management of PVC pre- and post-consumer waste in the EU27+3 (Conversio Market & Strategy GmbH 2021; VinylPlus 2021c).

4.2.4 PVC waste: hazardous or non-hazardous?

As will become apparent in various section of chapters 5 and 6, the question whether PVC waste (or specific types thereof) should be classified as hazardous or non-hazardous is a highly relevant issue. The status of PVC waste as hazardous or not has considerable influence on the manner in which such waste is to be managed, including its recycling and disposal. As such, this section will provide a number of considerations concerning the classification of PVC waste which should be taken into consideration when reading chapters 5 and 6 of this report.

As described in detail in Annex 1, the classification of PVC waste is to be done in accordance with Article 3(2) of EU Waste Framework Directive (WFD) and Commission Decision 2000/532/EC (List of Waste, LOW). The LOW features entries for hazardous and non-hazardous types of wastes which are defined by six-digit codes, entries for hazardous waste are marked with an asterisk (*).

- Certain entries are considered to be 'absolute hazardous', with the consequence that legal assumption is made that these wastes are hazardous wastes in the sense of Article 3(2) of the WFD.
- Other so called mirror entries can be defined as two or more related entries where one is hazardous and the other is not. In this case, it must be concretely assessed whether the waste at hand exhibits properties which render it hazardous in line with Annex III of Waste Framework Directive. Correct entries for PVC waste often are mirror entries.

Whether a concrete waste displays hazardous properties can be assessed

- Calculation if threshold limits based on hazard statement codes (individually depending on the properties HP4 to HP14 of Annex III of the Waste Framework Directive) are equalled or exceeded by the substances that are present in the waste under consideration; taking into account substances present in the waste and their classification under Regulation (EC) No 1272/2008 (CLP Regulation).
- Testing if the waste displays hazardous properties or not.

As indicated above, the question whether waste allocated to a mirror entry is hazardous or not can only be answered by taking into account a concrete waste with concrete composition and properties. An important aspect in this regard concerns the presence of substances in the waste which should be considered hazardous according to the CLP Regulation. Chapter 2 of the report indicates that PVC in general contains numerous additives which are classified as hazardous under CLP. For instance, according to the thresholds for reprotoxic and carcinogenic substances in Annex III of WFD, it is possible that flexible and rigid PVC waste (other than new, clean pre-consumer waste) would need to be classified as hazardous waste. To our knowledge, in Member States, PVC waste is not treated as hazardous waste. Evidence from Germany (Potrykus/Zotz 2020) suggests that PVC waste is generally assumed not be hazardous and information on specific tests on substances such as Lead in PVC is not available. This seems to be a general policy in various Member States given the indication as of section 6.2.2 that most PVC directed to landfilling is sent to landfills for non-hazardous waste. One explanation for this could be that relevant actors and authorities follow the reasoning that the additives are bound in the PVC matrix and, as such, do not migrate or become bioavailable. In this regard, it is important to note that various literature sources indicate that migration of additives from plastics takes place, since such additives are not covalently bound in the plastic matrix (see chapter 2). However, it should also be noted that, currently, data on migration and bioavailability of PVC additives is limited (see chapter 2). From a precautionary perspective, further assessment of the classification of PVC as hazardous or non-hazardous may be necessary for the determination of the most suitable waste management options for PVC. This seems especially relevant, considering the currently common practices of PVC recycling and the disposal of PVC waste in landfills for non-hazardous waste in the EU.

5. RECYCLING OF PVC WASTE

Key Messages – Recycling of PVC waste

Scope and Approach

- The purpose of this task is to give an overview and transparent picture about PVC recycling today and the near future, its market conditions, and its capacity of being a sustainable material with regard to its recyclability.
- To this end, different recycling options are described and analysed with a view on economic and environmental/health aspects, including mechanical recycling and chemical recycling.
- A specific focus is laid on decontamination technologies.

Collection:

- As for any waste stream, collecting PVC waste separately and cleanly is a precondition for high-quality recycling.
- While sector-specific take-back systems and separate collection are successfully applied to post-consumer waste from the construction and building sector (e.g. windows, pipes) in most of the EU Member States, the (limited) available information suggests that separate collection remains a challenge in practice for most post-consumer PVC waste streams. PVC waste from packaging, automotive and medical waste is rarely separately collected (and thus recycled less).

Mechanical recycling:

- Mechanical recycling is currently the standard recycling option for pre- and postconsumer PVC waste. This includes recycling processes where the material is treated mechanically through grinding, sieving, regranulating and compounding.
- While separation and sorting technologies have been optimised in recent years with a view to increased recycling, a significant problem for conventional mechanical recycling is contamination of the PVC waste streams with impurities (composites or laminates including a PVC fraction, but also mixed waste). These impurities hamper the recycling process.
- Another problem are “legacy additives”, understood as chemicals that are no longer used in new products, mostly due to their prohibitions or restrictions on account of their toxicity characteristic. Such legacy additives may lead to the recycled PVC not meeting material standards for products when placed on the market, besides being of environmental concern (see also specific conclusions below).
- While in the future, innovative recycling processes may be an option to remove contaminants and/or legacy additives from PVC waste, currently these technologies clearly are in their development phase and for the moment, no technology exists to effectively deal with the removal of impurities and legacy additives, likely leading to amounts of PVC waste delivered to recycling facilities but ultimately treated by incineration or landfilling.

Chemical recycling:

- Chemical recycling refers to the conversion of plastic polymers into their monomers, or basic chemicals, with a big advantage seen in the potential to remove problematic or hazardous substances and contamination from the material cycle.
- Available evidence suggests that currently chemical recycling is a technology which is still in its development phase. Furthermore, chemical recycling specifically of PVC is not practiced in the EU (except in pilot plants or in research projects); and in most initiatives of chemical recycling, PVC is not the targeted material.
- From an environmental perspective, comprehensive LCA results for chemical recycling are limited and should be interpreted with caution. Compared to mechanical recycling, chemical recycling is a highly energy-intensive process. Studies show that through gasification or pyrolysis of PVC waste, dioxins and dioxin-like PCBs can be produced. Even output products (e.g. pyrolysis oil) can be contaminated, requiring purification processes.
- Since chemical recycling processes for PVC do currently not operate on industrial scale, data is limited and economic profitability is impossible to assess. Experts point out that chemical recycling processes are generally more cost-intensive than mechanical recycling processes because chemical recycling is more complex and energy-intensive.

Decontamination techniques for legacy additives:

- Restricted or unwanted additives (such as cadmium and lead stabilisers or phthalate plasticisers such as DEHP) are an issue for PVC recycling. No feasible recycling technology exists at industrial scale to remove such substances from PVC waste. Conventional mechanical recycling processes are not able to remove legacy additives from the PVC material.
- However, non-conventional mechanical recycling technologies like selective dissolution processes have the potential to remove legacy additives.
- It is not clear whether mixed and contaminated waste can be easily chemically recycled at all. Rather there are indications that for most chemical recycling processes pre-sorting or energy-intense purification processes are necessary.
- As section 2 of this report demonstrates, legacy additives are frequently classified under CLP or subject to additional regulatory requirements under REACH (such as the need for authorisation), leading to strict requirements from chemicals and waste legislation. Available information suggests that, in practice, the presence of hazardous substances (e.g. lead or DEHP) and their potential for migration from the plastic matrix is not sufficiently taken into consideration for the correct classification of PVC waste in accordance with EU law.

This chapter provides a more detailed overview of the current status of PVC recycling in the EU. Technical, environmental and economic aspects are discussed for mechanical recycling and

chemical recycling. This includes assessment of current technologies but also collection processes or decontamination initiatives.
The potential for PVC as a sustainable material with reference to its recyclability and use of recyclate is assessed. Figure 28 gives an overview of existing recycling options for PVC waste and possible output materials (e.g. recyclate). Collection, sorting and pre-treatment are essential preliminary steps for mechanical recycling and other recycling processes like selective dissolution. For chemical recycling these pre-steps are less relevant. Based on this information, future trends concerning PVC recycling are derived. Note that definitions of specific terms linked to the recycling of PVC are included in Annex 3.1.

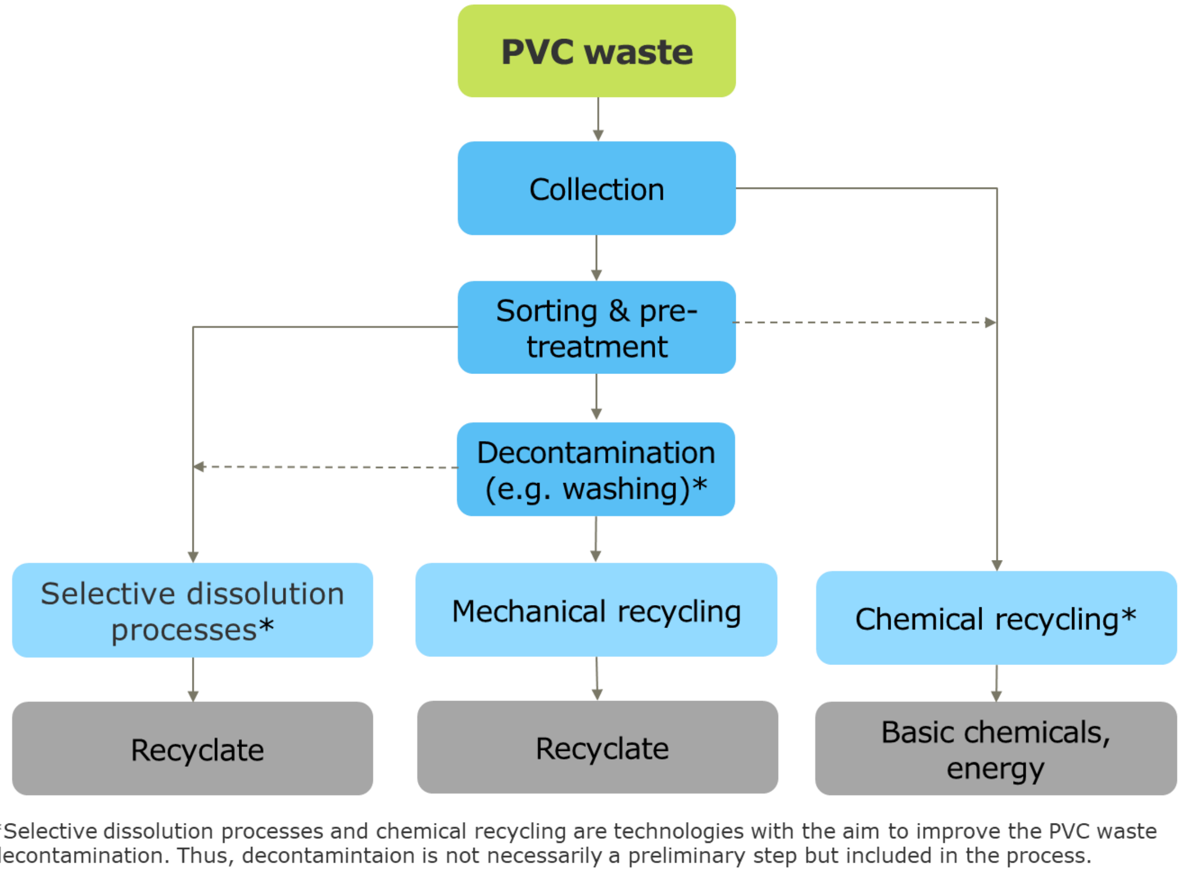


Figure 28. Overview on different recycling options for PVC waste

Separate collection as a challenge for PVC recycling
While sector-specific take-back systems and separate collection are successfully applied to post-consumer waste from the construction and building sector (e.g. windows, pipes) in most of the EU Member States, (limited) available information suggests that separate collection remains a challenge in practice for most post-consumer PVC waste streams. PVC waste from packaging, automotive and medical waste is rarely separately collected (and thus rarely recycled).

5.1 Collection of PVC waste

Waste collection is the precondition for (mechanical) recycling and significantly influences the applicable recycling technology and the recyclate quality (Ciacci, Passarini, and Vassura 2017). In

this study PVC waste collection is understood as any action to regain PVC waste. Thus, any form of collection like kerbside collection, self-delivery or take-back systems is included.
A major factor that determines the amount of collected PVC is the life span of each product group. Flooring, for example, has a lifetime of approximately 25-30 years, windows approximately 30-40 years and plastic pipes more than 80 years (see chapter 3). In contrast to construction and building products, plastic packaging typically has a very short lifetime.

Several separate collection initiatives for specific PVC product groups already exist in Europe. Sector-specific take-back systems and separate collections are already successfully applied to pre- and post-consumer waste from the construction and building sector (e.g. windows, pipes) in most of the European Member States. The collection systems either are integrated with recycling operators, or the collected PVC waste is transported to recycling partners (Fråne et al. 2018). Table 5-1 gives an overview of existing collection schemes in Europe that specialise in PVC post-consumer waste. More detailed information on AgPR, ROOFCOLLECT, Rewindo GmbH and EPCoat-IVK-Europe can be found in Annex 4.1.

Table 5-1: Overview on existing collection schemes in Europe (* = focus on pre-consumer waste)

Product groups	Type of PVC (Rigid/flexible)	Examples of European collection initiatives
Pipes (e.g. drainpipes, water pipes, sewage pipes)	rigid PVC	ÖAKR (A, 1991), KRV (DE), The Nordic Pipe Association (SE), WUPPI (DK), Norge AS (NO), Buizen Inzamel Systeem (NL), KURIO Recycling (BE), Pipelife* etc.
Window and door profiles	rigid PVC	Rewindo (DE), ÖKVA/ÖAKF (A), WUPPI (DK), VKG (NL), VEKA take back system etc.
Panels and gutters	rigid PVC	WUPPI (DK)
Flooring	flexible PVC	AgPR, The Swedish Flooring Trade Association, ARP Switzerland, Tarkett AS* etc.
Roofing membranes	flexible PVC	ESWA-ROOFCOLLECT (13 European countries), danosa, Protan AS
Tarpaulins	flexible PVC	EPCoat-IVK-Europe (EU)

Source separation and separate collection is generally advantageous for most of PVC product groups. An example from Denmark indicates that 97,5% of separately collected PVC was sent to recycling (Fråne et al. 2018). However, several PVC products end up in co-mingled collection (e.g. packaging waste). The possible impacts of this are discussed later (chapter 5.2.2). Separate collection and co-mingled collection can be undertaken for both mono-fractions (e.g. pipes, some films, window profiles) and for composite products (e.g. car components, floorings, cables) (see Figure 29). Composite products must be subjected to disassembly or mechanical treatment processes to extract PVC (Baitz et al. 2004).

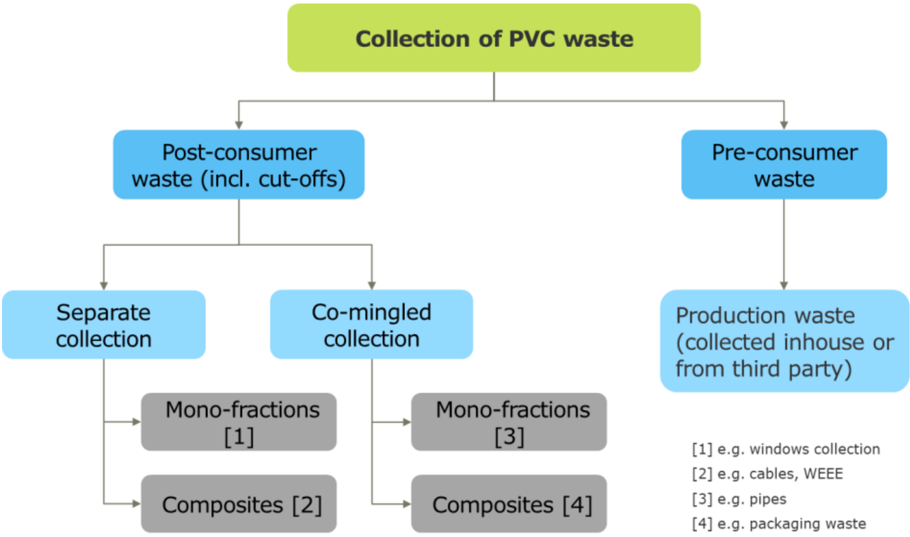


Figure 29. Collection of PVC waste

Post-consumer waste (including cut offs/installation waste) are mostly collected through separate collection schemes. Pre-consumer waste does not necessarily require such a scheme since it is generated at the manufacturer. Most of pre-consumer waste can directly be reused internally at manufacturing. However, there are also examples of companies that send their pre-consumer PVC to recycling plants abroad (e.g. material from Pipelife or Tarkett is recycled outside Norway in Sweden or the Baltic States) (Fråne et al. 2018). In Sweden, Norway and Finland several separate collection systems exists for pre-consumer PVC waste (mainly business-to-business), initiated by businesses producing/importing flooring and piping products. The amounts of PVC waste collected within these collection systems are uncertain, but are most likely minor (Fråne et al. 2018). It is important to note that pre-consumer waste should only be defined as "waste" when the PVC material is **not** considered a by-product in accordance with Article 5 WFD. It is not possible to assess under this study whether all pre-consumer PVC can be considered by-product.

Data from Sweden, Norway, Denmark and Finland show that a large amount of PVC enters mixed waste flows or is collected together with other separately collected waste streams, where it is not the target material for a recycling path (Fråne et al. 2018). The collection success of the different product groups of PVC post-consumer waste varies significantly:

1) The collection of **pipes** is comparatively well-established in Europe. Pipes can be separately collected from construction sites. PVC pipes are mostly collected together with other plastic pipes (PE and PP). However, these can be easily separated at a later stage (Fråne et al. 2018).
2) The collection of **windows** is also well-established. Window profiles comprise the largest share (nearly 50%) of recycled quantities of PVC waste in Europe. This suggests that shares of collections are also relatively high.
3) Collection systems for **flooring and roofing** exist as they can easily be collected when kept separately at construction sites. However, in practice, this does not often occur.
4) PVC included in **cables** (as insulation material) or in electrical and electronic equipment (**EEE**) is collected within existing separate collection schemes for cable waste or for waste of electrical and electronic equipment (WEEE). The WEEE collection systems in the Member States regulated by the WEEE Directive were initiated in 2003. This may influence the amount of collected PVC (Ciacci, Passarini, and Vassura 2017).
5) Further considerable amounts of PVC waste are created with the treatment of **end-of-life vehicles**. Larger plastic parts can be dismantled and reused or recycled. Nevertheless, PVC

waste from end-of-life vehicles mostly consist of cables (input from VinylPlus 2021). However, roughly one third of all vehicles are departing from the European vehicle stock annually and are not treated as end-of-life vehicle in one of the 13,000 authorised treatment facilities throughout Europe (Kitazume, Kohlmeyer, and Oehme 2020).

6) PVC waste is also included in **packaging waste** which is mostly collected within collection schemes from Producer Responsibility Organisations (PRO) on the basis of the EU Packaging and Packaging Waste Directive. However, data from the Conversio market study show that only 2-3% of this waste stream is PVC (Conversio Market & Strategy GmbH 2021).
7) In some countries, PVC waste is also collected as **rigid plastic** (e.g. buckets, trays, garden furniture, toys and pipes) and can be delivered to municipal waste recycling stations (e.g. in Norway, Sweden).
8) PVC from health care is collected either separately as hazardous waste or together with other waste streams (Ciacci, Passarini, and Vassura 2017). To date, only a few tonnes of PVC from **healthcare/medical devices** have been recycled as part of the initiative RecoMed (Blaeij et al. 2019). In 2019, 9,153 kg and in 2020, 1,949 kg were collected.
9) Composite products like **coating applications** (e.g. artificial leather) or **furniture** components as well as **agricultural waste** are not collected separately (Potrykus, Zotz, et al. 2020). The market share, as well as the mass stream of PVC, in these applications is negligible (Baitz et al. 2004).

Several initiatives are noted in the VinylPlus Progress Report 2017 (Vinylplus 2017). These are focussed on improved collection to facilitate and enable recycling (see Table 5-2).

Table 5-2: Several initiatives exist to improve the current collection and recycling of different PVC product groups

Initiative	Description of aims and current status
Recovinyl® organisation	Established in 2003 by the European PVC value chain, Recovinyl monitors and verifies the recycling of PVC waste and the uptake of PVC recyclate. Recovinyl is used to record how much PVC is being recycled in Europe and is the biggest contributor to the VinylPlus® recycling targets. Recyclers within Recovinyl are certified according to EuCertPlast.
Resysta	The Resysta® recycling consortium produces a recyclable wood-like material based on rice husks and PVC. In 2019, Resysta continued its communications and promotional activities for its applications, and improved its controlled-loop recycling system, which now has 20 collection points in Europe (Vinylplus 2020b).
RecoMed	RecoMed is a partnership project between the British Plastics Federation (BPF) and Axion aimed at collecting and recycling non-contaminated used PVC medical devices from UK hospitals, including face masks and tubing. The project currently involves 36 hospitals, and another 100 are ready to enrol. Over 9,000 kg of medical devices were collected in 2019, equivalent to more than 746,000 mask-and-tube sets (Vinylplus 2020b).
WREP	WREP (Waste Recycling Project) was launched in 2016 by PVC Forum Italia to assess the improvement potential for PVC recycling in Italy and to promote the development of pilot PVC waste collection and recycling schemes (Vinylplus 2020b).

5.2 Mechanical recycling of PVC

The following assessment is based on existing collection schemes, separation technologies and mechanical recycling processes. In order to evaluate gaps and potentials of current mechanical recycling of PVC, the technical, economic and ecological aspects that influence PVC recycling in the EU are assessed.

5.2.1 Overview mechanical recycling of PVC waste

The dominant recycling process for PVC pre- and post-consumer waste is mechanical recycling. This includes recycling of rigid and flexible PVC waste. Mechanical recycling includes processes where the material is treated mechanically through grinding, sieving, regranulating and compounding. Thereby the polymer structure does not significantly change (i.e. the polymer chains do not break into smaller components) (Plinke et al. 2000; Vinylplus 2017; Vogel, Krueger, and Fabian 2020). Mechanical recycling of PVC aims to (automatically) separate pure fractions of PVC and other materials and to produce recyclates with a defined particle size (Baitz et al. 2004).

Selective dissolution processes (non-conventional mechanical recycling)
Besides the conventional mechanical recycling methods, research is ongoing on selective dissolution processes. These are considered a mechanical recycling processes (Vinylplus 2017) or physical recycling (input Fraunhofer IVV 2021) as the polymer structure is not significantly changed. No plants applying selective dissolution processes are currently operating at industrial scale. The only industrial scale selective dissolution plant in the EU was closed in 2018.However, selective dissolution processes allow to remove contaminants and legacy additives without destroying the polymer structure. Several projects are ongoing and processes like the CreaSolv® Process have the potential to be applied at industrial scale.

Conventional mechanical recycling is well-established for certain PVC product groups (Deilmann et al. 2017; Fråne et al. 2018). The following table illustrates recycling practices for different PVC waste streams:

Origin	Product	Comment
Construction & demolition waste	**Window profiles**	**Window profiles** comprise the highest share (nearly 50%) of recycled quantities in Europe. In 2019, 363,137 tonnes of PVC window profiles were recycled mechanically within the VinylPlus framework. Around 40% were recycled in Germany, 30% in the UK and 30% in the rest of the EU-28. In 2019, the German initiative Rewindo recycled 35.000 tonnes post-consumer PVC waste (old windows, shutters and doors) and 65.000 tonnes pre-consumer PVC waste (waste cuttings during window production)(Rewindo 2019). Since 2018, the VEKA Recycling Group operates the most technologically advanced PVC window recycling plant in Europe (Wellingborough (UK)) (BVSE 2018a).
	Pipes	**Pipe** recycling is practiced in nearly all European countries. 11% of the total amount of recycled PVC quantities in Europe stem from pipes (Vinylplus 2019).
	Flooring	**Flooring** which is collected by the network "Association for PVC floor covering recycling" (AgPR) is transferred to the AgPR-plant in Troisdorf, Germany (AGPU e.V. 2016). They apply the rather atypical cryogenic grinding process to the flooring waste. Only 0.4% of the amounts published by VinylPlus are recycled flooring (Vinylplus 2020a) whereas the market share of PVC flooring amounts to 8%. In 2019, the carpet and flooring manufacturer Tarkett[58] collected and recycled 3,000 tonnes of PVC waste from construction sites worldwide and 50 tonnes post-consumer floorings from hospitals and retail.
	Roofing	Collected post-consumer PVC **roofing** is recycled to some extent. The collection system ROOFCOLLECT (covering 13 EU countries) has a partnership with the German recycling

[58] Tarkett is a French company in the flooring and sports surfaces sector

		company Fair ground Kunststoffmatten (formely PVR Pro Vinyl GmbH and before Jutta Hoser). Fair ground recycles post-consumer roofing membranes into various plastic mats.
Other waste streams	**Coated fabrics**	In 2019, 7,114 tonnes of **coated fabrics** were recycled within the VinylPlus network. The amount corresponds to 0.9% of VinylPlus volumes (Vinylplus 2020a).

In the case of cables and WEEE containing PVC, the PVC is sorted as it might contaminate other fractions (Martens and Goldmann 2016). When performing mechanical recycling of **cables,** PVC is not the target material, but metals. However, the separated PVC is likely to be recycled as secondary waste (Martens and Goldmann 2016).

Limited recycling of PVC from healthcare and automotive sectors
The majority of PVC waste in the **healthcare or medical devices** is incinerated (Fråne et al. 2018). These treatment options will be analysed in more detail in chapter 6. The same applies to PVC in the **automotive and transport sector** where PVC is mainly incinerated or landfilled (Ciacci, Passarini, and Vassura 2017).

5.2.2 Technological aspects of mechanical recycling

5.2.2.1 Process steps

The material properties of the different types of PVC products strongly vary. Different mechanical recycling techniques are necessary to adequately treat rigid and flexible PVC respectively (Fråne et al. 2018).

<u>Rigid PVC</u>

Typical process steps applied for mechanical recycling of rigid PVC consist of pre-sorting, shredding/grinding, separating (e.g. removing of metals), washing, drying and/or granulating and compounding in an extruder (Ciacci, Passarini, and Vassura 2017). Pre-consumer waste can also be reused as flakes or powder. Clean PVC waste is shredded and then melted to pellets which are then used as input material for PVC production (Sadat-Shojai and Bakhshandeh 2011; Wagner and Schlummer 2020) (Wagner and Schlummer 2020). It is also common practice to mix various PVC compounds with different material compositions. Depending on the year of production and the manufacturer, the recipe of the composition may differ (input TEPPFA 2021). A simplified schematic recycling process for rigid PVC and further description of recycling processes is illustrated in Annex 4.2.

<u>Flexible PVC</u>

In the case of flexible PVC (e.g. flooring, cables) and mixed fractions, cleaned PVC particles are not extruded, but directly recycled via compression or injection moulding (see Annex 3.1) or in the case or pre-consumer PVC rolled out on a calendaring machine (e.g. for high-quality flooring waste). Flexible material cannot be shredded and extruded into pellets like rigid PVC. The requirements to perform moulding are less challenging than for extrusion but at the same time uses of the secondary products created are limited. Additives are not removed in any of these processes. In Germany, a certain amount of flooring is recycled via cryogenic grinding (see Annex 3.1). Theoretically, PVC roofing could be treated with cryogenic as well (Martens and Goldmann 2016). However, no existing recycling plant performing cryogenic grinding of PVC roofing was identified. A simplified schematic recycling process for flexible PVC and further description of recycling processes is illustrated in Annex 4.2.

Typical conversion processes that are used within mechanical recycling of flexible and rigid PVC are summarised in Table 5-3.

Table 5-3: Converting processes

Mechanical recycling process	Input PVC waste	Recyclate type	Description
Shredding	Pre-consumer waste	Flakes, Powder	Some PVC products (e.g. pre-consumer waste) are directly reused after shredding.
Extrusion	Rigid PVC (e.g. Windows, pipes)	Pellets	Extrusion is the conventional process to achieve granulates/pellets and is used in the recycling industry since many years. Extrusion allows that the material or the macromolecule remains intact conserving polymerisation energy (Wagner and Schlummer 2020). Most extruders includes a filter segment that removes remaining impurities from the melted and homogenised PVC (so called melt filtration). The operation temperature is between 200–300 ◦C.
Compres-sion Moulding	Flexible PVC (e.g. cables, roofing, flooring)	Thick-walled objects	After removing large impurities and shredding steps the plastic particles are converted into a dough-like consistency and brought into the desired shape by pressing.
Injection Moulding	Rigid and flexible PVC	Pre-defined forms	Sheets or fittings
Cryogenic grinding	Flooring	Recyclate Powder	• Not very common (only one plant in Germany) • Grinding of flexible PVC in a cold stage • Coolant is liquid nitrogen • Flexible PVC needs to be cooled down to allow grinding
Selective dissolution processes	Composite products (one project for flooring)	Virgin PVC without the removed additive(s)/ contaminant.	Selective dissolution processes allow to dissolve specific plastics and/or additives (e.g. DEHP) in order to separate them from each other. Today the Remadyl project and the Circular Flooring project examine selective dissolution processes for PVC.

Theoretically, mechanical recycling technologies can be identical for pre- and post-consumer waste, but in practice PVC post-consumer waste is more complicated to recycle. As pre-consumer waste accumulates directly during production it can be easily reintegrated into the production process. As mentioned above, in some cases pre-consumer wastes could be considered by-products rather than waste. Post-consumer waste often contains more impurities, and some objects are collected separately and some co-mingled. These aspects make recycling of post-consumer waste more difficult to realise (Plinke et al. 2000). In general, pre-consumer PVC waste and clean post-consumer PVC waste materials are the preferred input for mechanical recycling (Sadat-Shojai and Bakhshandeh 2011).

5.2.2.2 Process challenges

Mechanical recycling as established option for PVC
Mechanical recycling is currently the well-established standard recycling option for pre- and postconsumer PVC waste. This includes recycling processes where the material is treated mechanically through grinding, sieving, regranulating and compounding. Relevant process challenges exist which are described in the following sections. Analysis shows there are technical, economics or regulatory limitations for the recycling of various PVC waste streams at present. These limitations in turn may explain the current focus of the PVC recycling

industry on PVC waste from the building and electronics sector, but also indicate that expansion of recycling activities to PVC waste from other sectors in the future is subject to considerable uncertainties. It should be noted that not all of these challenges apply to the same extent to post- and pre-consumer waste or to flexible/rigid PVC. Where possible this distinction is made in the subsequent sections.

The following technical challenges were identified:

1. **Effects of co-mingled collection** (e.g. mixed packaging collection);
2. **Contamination with impurities** such as glass, rubber, glue in PVC waste streams;
3. **Contamination with additives** such as legacy additives[59] in PVC products;
4. **Complex products and composites**;
5. **Degradation** mechanisms and **deterioration** of mechanical and physical properties during the mechanical recycling processes.

Effects of co-mingled collection

Co-mingled collection means that PVC is mixed with other non-PVC waste (e.g. other PE pipes, other packaging waste) when it is collected. Although sorting of PVC is feasible in many cases, co-mingled collection can still have a negative influence on the downstream mechanical recycling processes and may eventually impact the recyclate quality. The challenge of co-mingled collection mostly applies to post-consumer waste as pre-consumer can be collected at the manufacturers site and does not need to be mixed with other materials. In Table 5-4 typical examples of co-mingled collection of PVC waste and effects between the materials collected together are summarised. The table shows that the challenges of co-mingling mainly arise in the case of PVC waste arising from households and certain commercial sources (i.e. in separate packaging collection, mixed municipal waste or municipal drop-off points). In these cases, PVC is rarely transferred to recycling. This suggests barriers for increases of recycling of PVC from these sources.

[59] See Definition in Annex

Table 5-4: Co-mingled collection

Type of co-mingled collection	Description of negative consequences of co-mingled collection
PVC in mixed packaging waste	Packaging is collected in different EU countries via EPR schemes. Under such collection arrangements PVC packaging gets co-mingled with other packaging and materials. As PVC has a melting point with a relatively low range compared to other plastics (e.g. PP/PE) (Sadat-Shojai and Bakhshandeh 2011), PVC should not be melted together with other polymers. A co-processing of PVC with other polymers could lead to a lower quality, poor mechanical properties and few possibilities of application in new products (Sadat-Shojai and Bakhshandeh 2011). As a consequence of these technical impacts, PVC packaging is separated from other packaging to avoid contamination of these waste streams. This, however, leads to further challenges: ➢ To date, NIR (near-infra-red) sorting technology is state-of-the-art to separate different plastics types. As PVC waste arises only in small quantities in packaging streams the sorting effort in relation to this small fraction is high and costly. ➢ PVC is not separated as a single fraction but together with other less-valuable plastics, i.e. it ends up in mixed fractions which are directed to incineration or transformed into low-quality recyclate (e.g. thick-walled products for road traffic safety) (Vinylplus 2017). ➢ However, the association Plastics Recyclers Europe points out that it is questionable to separate PVC from mixed fraction. The amount of PVC in packaging streams is very low and from niche packaging. It would be free of legacy additives but due to specific mechanical and aging properties it could not serve as recyclate for building and construction waste (input PlasticsRecyclers 2021).
PVC in mixed municipal waste	Due to the wide use of PVC in several products (e.g. toys, packaging), PVC might occur in mixed municipal waste as well. However, post-separation is not common for mixed municipal waste as the current technical processes cannot adequately remove odours that PVC recyclate would gain from mixed municipal waste. In particular, the presence of organic substances (food leftovers) in the co-mingled collection system and related rotting processes can lead to intensive odour (input PlasticsRecyclers 2021). Instead, this waste stream is largely incinerated.
PVC pipes and PP/PE pipes	PVC pipes and PP/PE pipes are generally collected together (Fråne et al. 2018). However, this does not pose a problem for the recycling of one of the polymers as they can easily be separated from each other with standard NIR sorting technology. As most of the waste pipes are large parts, the sorting with NIR sorting technology is more reliable than for small particles.
PVC in containers at municipal drop off points	In the case of collection in "rigid plastics" or "bulky waste" containers at municipal drop off points, PVC is not negatively affected by the other material in the container. The different plastic parts could be easily separated with NIR sorting technology. However, in practice, PVC is rarely sorted out from the material mix (input PlasticsRecyclers 2021).
PET and PVC (bottles)	For several decades, PVC has been used for the production of bottles and containers. Due to an increase in legislative pressure to ban the use of PVC in bottles (Schyns and Shaver 2020), PVC is no longer used in bottles and has been replaced by PET (input VinylPlus 2021). Although PET bottles and PVC bottles are seldom in contact, the co-mingled collection of these two materials has been a considerable challenge in the past. PVC has different melting points and thermal stabilities than PET. These differing characteristics may have several negative impacts on recycled material: ➢ If PET is processed within its usual melting temperature (typically 275-290°C) PVC will degrade (Sadat-Shojai and Bakhshandeh 2011) (Schyns and Shaver 2020). Degradation of PVC leads to discoloration of the recycled PET. It obtains a yellow or dark brown colour whereas the dechlorinated PVC becomes brittle and creates black specs within the recycled PET (WRAP 2013). ➢ If PVC and PET are melted together within the temperature range of PVC, PET does not melt at these low processing temperatures (Sadat-Shojai and Bakhshandeh 2011) (Schyns and Shaver 2020). ➢ Similar densities of PVC and PET make it difficult to separate PVC from PET (especially PET bottles), as it is challenging to apply density separation (Sadat-Shojai and Bakhshandeh 2011). Today this problem is limited as PET took over in bottle production and PVC bottles are rarer (input from Vinylplus 2021).

Contamination with impurities

For this report, decontamination is defined as the removal of all undesired contaminants, that prevent the plastic recycling process or adversely affect the use of recyclate. Contaminants in PVC do not have to be toxic or added intentionally to pose a problem for the creation of high quality recyclate but can be soils, food residues, fibres or glue residues from the use phase of the PVC. Such contaminants can adversely impact the quality of the PVC recyclate (input Focus Group Recycling 2021).

Contamination with impurities vs. contamination with additives
It is important to note that this report distinguishes between contamination **with impurities** such as glass, rubber, glue in PVC waste streams and **contamination with additives** such as legacy additives (see next section).

The contamination of PVC is seen as a huge challenge for recycling of PVC (Ciacci, Passarini, and Vassura 2017). During the conducted Focus Group on recycling it was pointed out that there is still a lack of (economically) feasible technology to separate items and decontaminate specific waste streams (Focus Group Recycling 2021). Different PVC products are affected by different contaminants and its related negative consequences. For example:

- All material that is not rigid PVC can disturb recycling of rigid PVC. Especially, sealing lips made of rubber (part of window frames) can disturb the extrusion process as rubber does not mix with PVC. Coloured flexible PVC impacts recycling of rigid PVC as it can cause unwanted colour changes (Martens and Goldmann 2016).
- Recycling of cables (flexible PVC) is affected by impurities like cross-linked PE, PUR, rubber, textile, aluminium/copper, and aluminium composite films (input PlasticsRecyclers 2021).
- In the case of pipes the contamination concerns arises from dust particles which could not be removed with current cleaning technology (input TEPPFA 2021).
- Recycling of PVC flooring is more challenging due to the high degree of contamination from the glue (Yarahmadi, Jakubowicz, and Martinsson 2003) (input Tarkett, 2021). Flooring in general seems to be a problematic product group for mechanical recycling (Focus Group Recycling 2021). Moreover, mechanical recycling of post-consumer flooring is negatively affected by concrete particles that remain when removing flooring. Concrete has negative abrasive effects on the recycling tools and glue is unwanted in recyclate as its composition is unknown (input Tarkett, 2021).

It is important to note that the contamination of waste streams with impurities is a problem that is not limited to these examples or the recycling of PVC as such but is a general challenge for all mechanical recycling. Thus, most mechanical recycling processes include pre-treatment steps, e.g. pre-sorting or pre-washing to remove foreign materials (e.g. other polymers) and impurities from the target material (Wagner and Schlummer 2020). These steps aim to achieve a clean and single-origin recyclate. Washing processes with water containing detergents are used to remove contaminants and impurities originating from the use phase of PVC (oils, fats, food, cosmetics, paints, fuels, dirt particles) (Martens and Goldmann 2016). Moreover, large contaminants, such as glass, metal, plaster, wood, and other solid polymers, are removed through a series of filters with decreasing mesh size. This melt filtration can only remove contaminants with a minimum particle size of the smallest mesh.

Sorting and separation technologies that are used to remove impurities are presented in more detail in Annex 4.3. As shown in the Annex, several technologies exist and are applied in practice to remove impurities during mechanical recycling of PVC. However, impurities still seem a big challenge for PVC recycling.

A PVC window recycler confirmed that much of the recycling technology is well established, but each process can be improved with new updates or small improvements. The recycler claims that

not every improvement needs to be revolutionary, but that a steady evolution can allow to increase the percentage of reclaimed material as well (input BPF member 2021).
Finally, it is relevant to note that pre-consumer waste is less affected by the issue of impurities as in-house recycling or similar activities reduce situations that can cause a contamination (e.g. transport, use phase etc.).

Contaminations as an important challenge for mechanical recycling
While separation and sorting technologies have been optimised in recent years with a view to increased recycling, a significant problem for conventional mechanical recycling are contaminations of the PVC waste streams with impurities (composites or laminates including a PVC fraction, but also mixed waste). These impurities hamper the recycling process.

Contamination with additives

Another challenge for mechanical recycling technologies are additives. Flexible PVC particularly contains a high number of additives. As such, recyclate of flexible PVC cannot be used in rigid PVC applications and recyclers of rigid PVC avoid flexible PVC (input TEPPFA 2021). Another aspect are legacy additives. These are additives that are no longer used in new products, mostly due to prohibitions or restrictions on account of their hazardous characteristics. However, such legacy additives can be found in older PVC applications that now reach their end-of-life phase[60].

[60] As described in Annex 1 to this report, Annex XVII to REACH contains restrictions for various substances which may be or have been used as additives for the production of PVC or PVC-based products, such as cadmium.

Recycling derogations concerning legacy additives in PVC
To enable the recycling of specific PVC products, exemptions from the restriction under Annex XVII of REACH for cadmium in PVC exist for recyclates in specific[61] PVC building applications. The concentration limit for cadmium in these specific cases is 0.1%. A similar derogation allowing higher lead content in recycled PVC products is currently under discussion in the context of a proposed REACH restriction of lead compounds in PVC. The European Parliament opposed the initial Commission's proposal for a Regulation on 12 February 2020 and called on the Commission to draw up a new proposal[62]. A report commissioned by the Swedish Chemicals Agency (2020) outlined several assumptions from the industry with regard to the effects of the derogation for cadmium (used as heat and light stabilisers and pigment agent in PVC) in PVC applicable to specific construction applications. According to this report, Recovinyl concluded that the derogation leads to a large increase in recycling. In EPPA's and TEPPFA's opinion the derogation simply enabled the industry to continue with business as usual, but the increase of recycling volumes is rather a consequence of the general market conditions and renovation rates. As such, the beneficial effect of the derogation on recycling volumes does not seem completely clear, as industry associations for specific PVC products seem to indicate a limited effect. In the case of DEHP, BBP and DBP which were used as additive in flexible PVC, the use is prohibited under REACH since 2013 unless an authorisation is granted. This applies to the use in new products and in recyclate. Only three companies were given a REACH authorisation to have DEHP and DBP present in flexible PVC recyclate till 2019. With regard to the recycling of flexible PVC containing DEHP, Stena Recycling was one of the three companies with a REACH authorisation. Due to economic reasons (high costs and time for authorisation, lack of market) Stena Recycling did not re-apply for authorisation (Ramboll Environment and Health 2020). The company Plastic Planet applied for another authorisation to produce recyclate from flexible PVC containing DEHP. A decision by the European Commission on this application is still pending (date 3.12.2021)[63].

Unlike contaminating impurities, contaminating legacy additives do not mechanically impact the recycling process or used machines as such, but influence the composition of the resulting recyclate (see chapter 5.5). To date, plastic parts containing high levels of legacy additives (plasticizers and stabilizers) can only be removed via imprecise pre-sorting methods causing a high rejection rate (Wagner and Schlummer 2020). However, this method involves removal of plastic parts as a whole and does not remove additives from the PVC compound as such (Kelly et al., 2005). Chapter 5.4 considers the current status of decontamination techniques for legacy additives.

Pre-consumer PVC waste and legacy additives
The issue of legacy (i.e. prohibited or restricted) additives is not relevant for pre-consumer PVC waste. The composition of pre-consumer PVC waste is identical to newly produced PVC products (Sadat-Shojai and Bakhshandeh 2011) i.e. no additives that are restricted should be included. However, it is important to note that currently various additives with hazardous characteristics are still used in PVC and are not subject to prohibitions or restrictions. As indicated in chapter 2 of this report, such additives could lead to risks for human health and the environment if they are emitted from the PVC via migration and could be subject to future restrictions.

[61] (a) profiles and rigid sheets for building applications; (b) doors, windows, shutters, walls, blinds, fences, and roof gutters; (c) decks and terraces; (d) cable ducts; (e) pipes for non-drinking water

[62] https://www.europarl.europa.eu/doceo/document/B-9-2020-0089_DE.html

[63] REACH Authorisation Decisions, last update 2.12.2021: https://ec.europa.eu/docsroom/documents/47675

Moreover, it is relevant to note that a stakeholder mentioned that the decontamination of PVC is often not the primary aim of technologies which are currently in research. Most technologies focus on creating a recycling process for certain plastic waste streams, which are currently not recycled (expert input Focus Group Recycling 2021).

Complex products and composites

As PVC products have become more complex, their components are more difficult to separate in the end-of-life phase. Mechanical recycling is still difficult to apply to materials and plastics that are strongly connected to each other (e.g. laminates or composites) (Sadat-Shojai and Bakhshandeh 2011). This is true for post- and pre-consumer PVC waste.[64]

Examples of PVC composite products are paste-based products like flooring, wall coverings, car underfloor protection, artificial leather, tarpaulins or coatings, but also blister packs, cables and wood fibre-reinforced PVC composites. Laminates are found in laminated films or in blister packs (laminate of aluminium and PVC) (ECVM et al. 2006; Plinke et al. 2000; Sadat-Shojai and Bakhshandeh 2011). Flexible PVC materials, such as those used in flooring, are often bonded with other plastic films or fabrics, the separation of which presents an additional challenge for the mechanical recycling process (Krauß and Werner 2014). It is difficult to separate PVC from other materials contained in a composite product (Fråne et al. 2018). A separation of the different materials is key for recycling as a joint recycling is rarely possible. Several initiatives to improve the treatment of composite products are presented in Table 5-6. To date most of these processes are under development and not available at industrial scale. The VinylPlus initiative indicated that the recycling of composite PVC material is most likely only feasible through selective dissolution processes (Vinylplus 2017) (see Annex 4.4).

Table 5-6: Treatment of composite products

Electrostatic storing	-Used to separate PET/PVC, PVC/rubber, PVC/PE and to separate flexible PVC and aluminium (Martens and Goldmann 2016) (Ragaert, Delva, and Van Geem 2017). -As an example, the Neidhardt Rohstoff GmbH (Germany) uses electrostatic sorting to separate aluminium from PVC (Vinylplus 2017).
Separate fibres from PVC	The Japanese company R-Inversatech Ltd. uses a high-speed beating technique to separate fibres from PVC waste (Vinylplus 2017).
Separate fabric and tissue etc. from flexible PVC foils	The recycling company Hemawe and Caretta, a producer of flexible PVC products, have developed a technique for separating fabric and tissue, etc. from flexible PVC foils (Vinylplus 2017). In a company's report from 2012 it is stated that 100% of the obtained recyclate is used in Caretta's foil production (Caretta 2012). In 2021, no information was found on the companies Caretta and Hemawe, so it is likely that the venture has been stopped.
Separate PVC from laminates	VinylPlus reports from the EuPolySep project which objective is setting up a small pilot plant in Belgium to separate PVC from complex laminated products. To delaminate polymers and to separate them from polymer-composite structures for subsequent recycling the Australian PVC Separation (PVCS) technology[65] has been identified as the most promising. It was tested at a pilot scale (Vinylplus 2021).
Selective dissolution processes	The CreaSolv® Process could theoretically treat composite materials. To date this is not done at industrial scale. The VinyLoop process was realised in a commercial plant (able to treat up to 10,000 tonnes of PVC waste per year). However, it had to stop the operations as it was not feasible to separate the phthalate contaminants in an economical

[64] Especially when pre-consumer waste is scrap from the final product

[65] PVCS: PVC Separation Pty Ltd is a proprietary and patented process for separating laminated polymer and other materials (www.pvcseparation.com)

	way (PlastEurope 2018). The process was able to treat plaques (PVC/polyester) and cable waste (PVC/copper).

Considerations on status of decontamination technologies
While in the future, innovative recycling processes may be an option to remove contaminants and/or legacy additives from PVC waste, currently these technologies are not available at large scale. Selective dissolution processes have a high technical maturity (e.g. CreaSolv® Process), but a large-scale implementation is not yet there. The lacking removal of impurities and legacy additives is likely leading to amounts of PVC waste delivered to recycling facilities but ultimately not recycled but sorted and treated by incineration or landfilling.

Degradation mechanisms and deterioration of mechanical and physical properties

During the recycling process of PVC several effects can occur that impact the mechanical and physical properties of PVC. Degradation mechanisms like thermo-oxidation, crosslinking, functionalization of chains and hydrodechlorination are typical effects on PVC (Schyns and Shaver 2020). Additives such as vegetable fillers (e.g. wood fibres in wood fibre-reinforced PVC composites) can change or accelerate the hydrodedrochlorination process (Sadat-Shojai and Bakhshandeh 2011).

As PVC is unstable to thermal and photoinduced stress, oxidative decomposition can happen. In every life cycle phase (incl. recycling) oxidative decomposition of PVC can occur by the effect of atmospheric oxygen in combination with heat (e.g. during melting and extrusion), by UV radiation and by mechanical shear stress during plastic deformation (Martens and Goldmann 2016). Metallic and other impurities favour this decomposition process. In consequence physical and mechanical properties of PVC are affected (Sadat-Shojai and Bakhshandeh 2011). Schyns & Shaver state that PVC waste streams should be pure and highly stabilized to prevent degradation (Schyns and Shaver 2020).

Although no longer in use (as of 2015), the presence of lead-based and sulphur-based stabilizing systems react to form lead-sulfide which appears as dark sports in the material and have potential human and environmental toxicity concerns, especially as much of this legacy material remains in use. CaCO3 based stabilisers, that can replace lead-based and sulphur-based stabilizers, are at the same time fillers. They are improving mechanical properties of the material and are retained over multiple life cycles. However, high amounts of these fillers leads to embrittlement, necessitating increased addition of plasticizers (in flexible PVC) that further complicate the chemical composition of PVC and necessitating filler inclusion at low loadings (Schyns and Shaver 2020).

5.2.3 Economic aspects of mechanical recycling

Although mechanical recycling well established technology, economic constraints are still a bottleneck and recycling processes are limited by cost (Sadat-Shojai and Bakhshandeh 2011)(Schyns and Shaver 2020). Economic assessments focusing on recycling routes for source-sorted plastic waste are scarce in the literature. Despite this , the identified economic aspects indicate, in general, PVC recyclate currently produced has a good market value and is able to compete with virgin PVC. However, economically feasible expansion of recycling operations to other PVC waste sources (e.g. municipal waste fractions or pipes) may be limited due to technical difficulties (e.g. collection and sorting) or competition from cheaper waste recovery and disposal options.

Although most of the economic challenges apply to the majority or PVC waste, different conclusions can be drawn for different PVC waste streams:

- It is reported that **not enough quantities of waste pipes** enter the recycling market. At the same time, the market for new pipes is increasing which makes it difficult to reach a high percentage of secondary raw materials in new products (Fråne et al. 2018).
- Compared to other construction waste, PVC waste **window** profiles are raw materials with a high economic value. Window recycling is an economically viable activity; however, it is still a challenge to collect window waste that occurs in small quantities (e.g. at small-scale construction sites).
- According to Potrykus et al. (2020) PVC recycling and the use of PVC recyclate within closed and controlled loops within the framework of a voluntary commitment (i.e. industry solution; VinylPlus commitment) is an economically viable and successful system.
- From a technical perspective, PVC **composite** products are difficult to treat, i.e. the economic burden of the separation techniques is high.
- In the last years, the recycling of **cables** became less attractive as the prices were negative (i.e. recyclers had to pay to treat them).

Table 5-7 gives a more detailed overview on identified economic aspects for PVC recycling.

Table 5-7: Overview on economic challenges for PVC recycling

Economic challenges	Description
Available volumes	The recycling company Van Werven claims that the biggest challenge for the rigid PVC recycling sector is the scarcity of PVC waste. They could sell two to three times more than the 12,000 tonnes per year but export of plastic waste (including PVC waste) out of Europe (e.g. to Pakistan) and obsolete PVC pipes that are left underground are limiting the availability of PVC waste volumes. TEPPFA confirms this statement. Rewindo states that especially the smaller quantities will be important for the continuous increase of the recycling volumes (Recycling 2020).
PVC content	Mixed waste streams with a low PVC content (below 10%) do economically not justify separation. For example, the amount of PVC occurring in packaging waste is too low for economically feasible separation (Ciacci, Passarini, and Vassura 2017). Similarly, in other product groups like health care products or non-packaging consumer and commercial goods (e.g. furniture components, toys, etc.) the amount of PVC is as well too low to justify a separation (Ciacci, Passarini, and Vassura 2017).
Collection area	Another economic challenge for separate collection is the area-wide collection of locally small amounts of waste and the possible high space requirements of collected material (Potrykus, Zotz, et al. 2020). The Dutch collection scheme WUPPI (mainly collecting pipes) confirms that the biggest challenge of collection is the costs. Thus, they are co-financed by VinylPlus (Fråne et al. 2018).
Collection, sorting and separation costs	A major part of the recycling cost for rigid PVC are fixed costs that arise from the collection, sorting and separation (e.g. container renting, logistics, recycling process costs) (input TEPPFA 2021). Detailed cost burdens of the market actors involved in the mechanical recycling of PVC are not publicly available.
Treatment costs	1) Automated sorting, as an example, saves labour costs but is a huge investment. 2) Co-mingled collection increases the separation cost, as PVC needs to be removed from other plastics. However, in most cases (e.g. separate packaging collection) PVC is not the target material. The separation of PVC pipes from PE/PP pipes do not seem to be a challenge or a huge cost driver. 3) With regard to selective dissolution, the VinyLoop process had to stop the operations as it was not feasible to separate the phthalate contaminants in an economical way (PlastEurope 2018). 4) High amounts of coolant and energy needed for cryogenic grinding are economic limits to the practical application (Martens and Goldmann 2016).

Competition with energy recovery and landfilling	Under current market conditions, mechanical recycling represents an economic alternative to energy recovery and landfilling, as PVC recycling granulates achieve a stable and positive market value (Potrykus, Zotz, et al. 2020). Especially in relation to PVC window profiles, waste recycling appears to be the least costly waste management option (Ramboll Environment and Health 2020). On the other hand, the AgPR (Working Group PVC Flooring Recycling, Germany) indicates that financial aspects of recycled flooring waste with cryogenic grinding are a barrier to the recycling of flooring as other disposal options, such as waste incineration, are often cheaper (input AgPR 2021). AgPR mentions that price changes in the disposal market and market demands for a share of recycled materials in products will promote the recycling of PVC (input AgPR mail contact 2021).
Recyclate price	Usually, market prices for recycled PVC are lower than virgin PVC (Correa, de Santi, and Leclerc 2019). Prices of rigid PVC recyclate are less cyclical than prices of virgin PVC (input TEPPFA 2021) but are related to it. Recycled materials have to compete especially against PVC materials produced from primary raw materials. In Finland, it could be observed that increasing costs for virgin raw materials could motivate local industries to consider PVC recycling as an economically feasible option (Fråne et al. 2018). In the last years, the price of recycled PVC was around 65–70% the price of virgin PVC (Fråne et al. 2018). At the beginning of 2021 virgin prices have increased sharply. At the same time, the recyclate price of pre-consumer regrind was increasing to over 500 €/t (in 2018: 470€/t) (plasticker 2021). In 2019, prices for virgin PVC were 800-900 €/tonne (expert input Focus Group Recycling 2021). In the same year, there was a huge demand on recycled material whereas in 2020 the demand decreased due to lower virgin material prices (input TEPPFA 2021) (this might also be in correlation with the Covid-19 pandemic). TEPPFA also indicates that their members often mix post-consumer and pre-consumer recyclate to improve the quality by lowering the costs at the same time. Pre-consumer waste (incl. off-cuts) is more expensive as it is nearly virgin quality (input TEPPFA 2021). Annex 4.3 gives an overview on the virgin and the recyclate prices of certain PVC products and of certain countries.
Recyclate demand	The European Plastic Pipe and Fittings Association (TEPPFA) mentions that nowadays the demand for recycled PVC is not only driven by price gains but also by sustainability factors. Since several years, customers are asking for recycled content in the products they are buying (input TEPPFA 2021).
Recyclate supply	It is often difficult to increase the supply side (i.e. the amount of recyclate). One reason is that offered volumes are dependent on the generated waste which is for example related to the renovation and demolition rate (especially for construction material) (input TEPPFA 2021). The European Plastic Pipe and Fittings Association (TEPPFA) mentions that today's collected amounts of pipes cannot satisfy the recyclate demand for new pipes. One reason is, that pipes have such a long lifetime (>100 years) that the amount of pipes that occur as waste as of today are not sufficient. Another reason is that a lot of PVC pipes are not coming out of the ground because it is more economically to leave them under the earth (Fråne et al. 2018). It depends on regional legislation if pipes are taken out of the ground. As an example, TEPPFA describes that Dutch municipalities prefer taking out the pipes whereas Danish municipalities are likely to fill them with concrete (input TEPPFA 2021). The missing amount of used PVC is not only limiting the recyclate content in new products but makes it difficult for recyclers to reach economies of scale (Fråne et al. 2018).
Recyclate quality	The European Plastic Pipe and Fittings Association (TEPPFA) mentions that the quality of recyclate is especially important to converters, as recyclate often leads to higher production scrap rate (e.g. in extruders) due to an elevated amount of impurities. It is stated that the general uptake of recycled material has increased over the past few years due to increased quality of the available products for extruders (input BPF member, 2021).
International developments	Due to restrictions of the Chinese government on the import of certain types of plastic in 2019, there is currently an abundance of cable waste on the market. Mainly cable sheathing, which is a side product from cable recycling and made from PVC, occurs in large quantities. Due to the abundance, in 2019, the prices for cable sheathing became negative (-200 to -50 €/t) (input from PlasticsRecyclers 2021). It is possible that the amendments of the Basel Convention and the EU Waste Shipment Regulation concerning exports of plastics in 2019/2020 have similar effects (see section 6).

5.2.3.1 Recycling capacities

Currently, there are more than 100 operations in Europe which recycle PVC pipes, window profiles, flooring, coated fabrics and membranes (Vinylplus 2017). Big players on the European recycling market for rigid PVC waste are Van Werven in the Netherlands, Rehau and VeKa in Germany and Paprec in France (Fråne et al. 2018). Moreover, plants in Romania (Terraplast), Czech Republic (Recycklo) and Poland (Rehau) with annual throughputs from 8,000 to 12,000 tonnes were identified (see Table 5-8). The capacity displayed in Table 5-8 (around 150,000 tonnes) is only a part of PVC recycling companies in Europe. As mentioned above, over 100 recyclers exist. The EuPC claims that one major barrier to increased PVC recycling is lacking investment in recycling plants. It is said that lacking regulatory stability causes reluctance on the side of investors (Focus Group Recycling 2021).

Table 5-8: Examples of capacity of European recycling plants (focus on post-consumer PVC)

Type	Type of PVC	Capacity of plants
Van Werven (Netherlands)	Pipes, profiles, roof plates	12.000 tonnes/year (material from UK, the Netherlands, Belgium and Ireland)
Paprec (France)	Wastewater pipes, telecommunications ducts, windows, garden furniture, others	not specified
Recyklo (Czech republic)	PVC cables	8,000 tonnes per year, largest processor of cable PVC in the world
REHAU Sp. z o.o. (Poland)	PVC	10.000 tonnes/year PVC
Terraplast SA (Romania)	Rigid PVC	12.000 tonnes/year
Sum of three VEKA plants (UK, Germany, France)	PVC windows	100.000 tonnes/year (BVSE 2018b)
Neidhardt Rohstoff GmbH	PVC aluminium composite used for blister packaging (Pre-consumer)	in 2015 it recycled 727 tonnes of PVC-aluminium blisters, producing 485 tonnes of R-PVC (Vinylplus 2016)
AgPR/ Association for PVC floor covering recycling (Germany)	Flooring	Recycled flooring: 2,500 – 3,000 tonnes/year Theoretical capacity: 4,000 t/yr
ESWA-Roofcollect/fair ground (Germany)	Roofing	2018: 3,531 tonnes/year

In terms of total installed capacity of PVC recycling, the UK and Germany are leading in Europe (see Figure 30). In comparison with other polymers, such as PET or LDPE, the total installed capacity of PVC is low (see Figure 30).

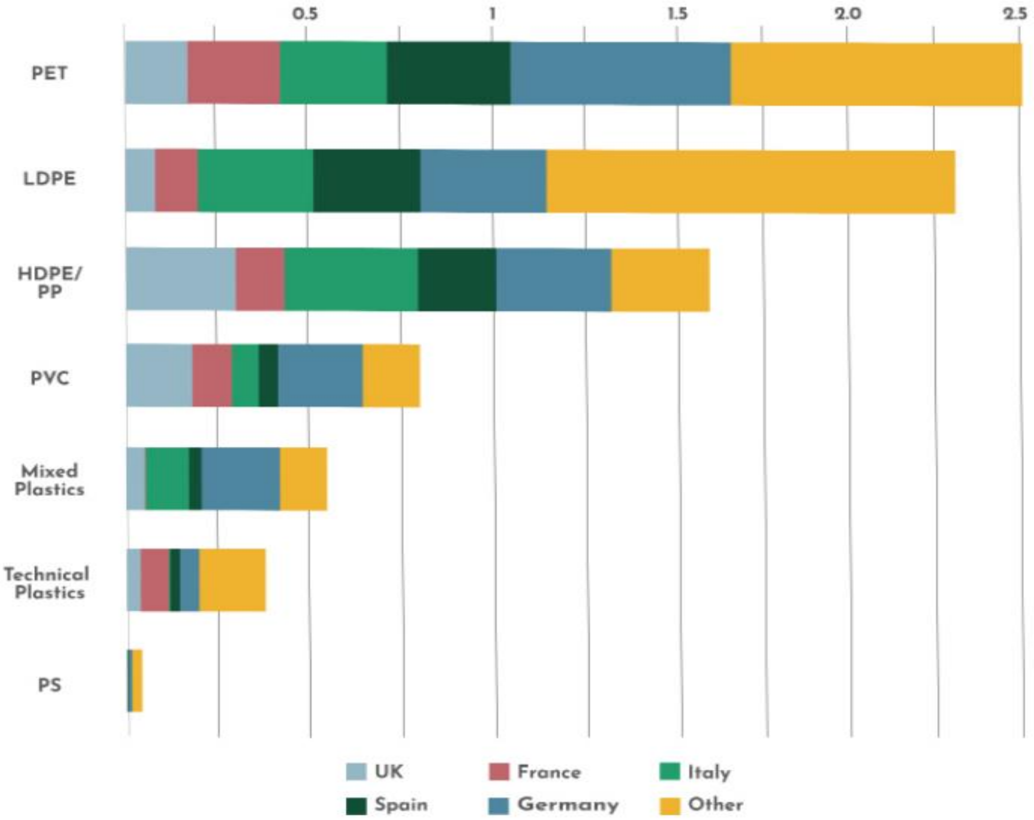


Figure 30. Total installed capacity per polymer by European country (Mt) (Plastic Recyclers Europe 2020)

5.2.4 Environmental and health aspects of mechanical recycling

Mechanical recycling needs to deliver environmental benefits to be an advantageous recovery option. Minimum factors that need to be taken into account are energy demand, potential emissions, used additives, as well as quality and use of the output recyclate. Recycling of PVC generally leads to reduced energy consumption and reduced CO_2 emissions compared to virgin production (Wagner and Schlummer 2020). Several studies prove that the use of recyclate can save CO_2 emissions.

Mechanical recycling and CO2 emissions
• The recycling of used **window** frames can save around 2 kg CO_2eq per tonne of window frames per year (200 kt of CO_2eq./year could be saved when recycling 100 kt) (Stichnothe and Azapagic 2013). • With regard to the primary energy demand and the strongly correlated greenhouse gas emissions, reduction potentials along the life cycle of 30% were identified for PVC **windows** with recycled content compared to windows without recycled content (Potrykus, Zotz, et al. 2020). • Compared to saving that can be gained by recycling of **plastic in general** (500–700 kg CO_2eq per tonne of plastic waste including PVC; data from 2008) the saving potentials of PVC seem low (Faraca, Martinez-Sanchez, and Astrup 2019). • An example of the **carpet and flooring** manufacturer Tarkett shows that in 2018, Tarkett could save 253,000 tons of CO_2 emissions by recycling 127,000 tons of pre-and post-consumer PVC flooring instead of using virgin raw materials and sending waste to incineration (input Tarkett, 2021). • In a comparative LCA commissioned by the European Plastic Pipe and Fittings Association (TEPPFA), PVC sewer **pipes** and concrete sewer pipes (DN 250 mm) were assessed. The PVC system performed better in the categories acidification, eutrophication, global warming and photochemical oxidation (Mihaela, Thuring; De Jaegher 2020).

In general, the recycling of PVC waste seems to save CO_2 compared to virgin production. However, the current results are not true for every PVC product/waste stream. It should be noticed that indications on the comparative CO_2 footprint of mechanical recycling mainly focus is on window recycling. The cited study on pipes (box above) does not compare recycling with virgin production but different types of pipes with each other. More detailed information on LCA results can be found in Annex 4.6. Table 5-9 gives an overview on other environmental and health impacts that arise for specific mechanical recycling processes.

Table 5-9: Mechanical recycling processes and its environmental impacts

<table>
<tr><th>Mechanical recycling process</th><th>Product examples</th><th>Environmental impacts</th></tr>
<tr><td>Extrusion</td><td>Windows, Pipes</td><td rowspan="2">Moulding and extrusion which are key stages in plastic recycling are normally operated at 200–300 ◦C which can cause a release of certain legacy substances (e.g. cadmium, lead) and several hazardous substances (e.g. toxic metals, volatile organic compounds (VOCs), phthalates, polycyclic aromatic hydrocarbons (PAHs), PBDEs, polybrominated dibenzo-p-dioxins and furans (PBDD/F) (Hahladakis et al. 2018). It can be a risk to employees of a recycling plant to be exposed to these substances. As such, care should be taken during the recycling process as the mechanical induced stress and heat may lead to the release of dioxins and of hydrogen chloride, which is a toxic gas (Hahladakis et al. 2018).

However, it is stated that leaching rates of cadmium as well as lead from rigid PVC are very low (EPPA profiles 2018; Potrykus and Milankov 2015). Literature indicates that this statement is true for additives in general. Although these risks are real, in practice they may only be relevant to a certain extent due to precautions taken. For example, lead is regulated by OSH legislation and lead levels at workplaces are controlled. Further influence of legacy additives on the use of recycled material will be discussed in chapter 5.5 which focus on PVC recyclate.

Every conventional mechanical recycling of PVC articles can give rise to emissions to air (from grinding and milling) and possibly surface waters (e.g. due to waste water arising from washing processes) (Committee for Risk Assessment (RAC); Committee for Socio-economic Analysis (SEAC) 2018).</td></tr>
<tr><td>Moulding</td><td>Cables, Roofing</td></tr>
<tr><td>Cryogenic grinding</td><td>Flooring</td><td>A stakeholder from the AgPR-plant running a cryogen process for flooring waste estimates that the energy demand they have is much lower than that of the virgin production (input AgPR mail contact 2021). However, the process of cryogenic grinding has a high demand for coolant (liquid nitrogen) (Martens and Goldmann 2016). According to AgPR, 0.7 kg nitrogen are used per kg PVC floor covering. The nitrogen is not reused as it evaporates completely (input AgPR 2021).</td></tr>
<tr><td>Selective dissolution processes</td><td>Composite products</td><td>According to PlasticsRecyclers selective dissolution processes need more energy than conventional mechanical recycling methods (input PlasticsRecyclers 2021). Moreover, they make solvents necessary (Schyns and Shaver 2020). However, selective dissolution processes like the CreaSolv® Process or the VinyLoop process include recovery and reuse of the solvent (Vinylplus 2017)(input Fraunhofer IVV 2021). In general, selective dissolution processes are not considered to replace conventional mechanical recycling but to be a complementary recovery solution for difficult to treat PVC products (input Focus Group 2021).</td></tr>
</table>

CO_2 emissions from mechanical recycling

In terms of energy demand and CO_2 emission, mechanical recycling is the most environmentally sound solution to management of waste PVC and economically viable, at least for specific sources of PVC waste. However, mechanical recycling strongly depends on the input material, which currently favours specific product groups for recycling.

5.2.5 Summary on mechanical recycling

To summarize, mechanical recycling is the most advanced and widely applied recycling technology for PVC waste in the EU. In terms of energy demand and CO2 emission it is the most environmentally sound solution to management of waste PVC.

However, mechanical recycling strongly depends on the input material, which currently favours specific product groups for recycling. Mainly clean, pre-consumer (rigid and flexible) PVC and post-consumer rigid PVC is favoured for recycling. As post-consumer waste is usually more contaminated with impurities and other materials, adequate sorting and washing process steps are necessary (Ragaert, Delva, and Van Geem 2017). Pre-consumer waste streams which are most likely to be less contaminated, collected separately and with a known composition are treated with less effort.

There are two main reasons that generally limit the recycling of post-consumer waste and apply also for PVC waste: the more challenging collection and the higher content of impurities. Collection represents the major bottleneck regarding the availability of waste and costs. Within post-consumer waste, mainly PVC from old PVC windows and pipes is regained as recyclate. Flooring and cable waste from post-consumer collection is recycled as well, but mainly end up in products like traffic products for road traffic safety. This kind of recycling can be seen as downcycling as it is very likely that such will be transferred to energy recovery after their use. Although these are less demanding applications, the applied treatment still qualifies as recycling according to the WFD. Moreover, the issue of legacy additives is still relevant for PVC recycling as no mechanical recycling allows a proper removal of legacy additives from the PVC matrix. Decontamination options for legacy additives (e.g. selective dissolution processes) are assessed in chapter 5.4.

Based on the above, a main uncertainty concerns the future potential for scaling recycling of PVC waste from less favourable sources. Without technical solutions and increasing economic feasibility of treatment operations, PVC recycling may remain a partial solution for a limited number of specific streams and may reach its limits at a certain stage.

5.3 Chemical recycling of PVC

5.3.1 Overview chemical recycling

Chemical (or feedstock) recycling of (mixed) plastic waste use thermochemical or chemical processes to depolymerise the materials. Its aim is to recycle the carbon in the form of low molecular weight compounds and as such differs from conventional incineration where the carbon is transformed into CO_2. Chemical recycling uses the obtained products as feedstock for the petrochemical industry to produce either the same or a related polymer or other substances.

Status of the recovery of plastics as liquid/gaseous energy sources/fuels
In some cases, the recovery of plastics as liquid/gaseous energy sources/fuels (e.g. use as liquid fuel for vehicles) is also defined as chemical recycling. This practice, however, contradicts the idea of material recycling since it is basically energy recovery (Sadat-Shojai and Bakhshandeh 2011; Vogel, Krueger, and Fabian 2020) and falls therefore, according to the WFD under "other recovery" and not "recycling" (Article 4 WFD). However, it should be noted that PVC can be chemically recycled through the processes of gasification and pyrolysis. As such, a strict separation is not always possible (more detailed description in Annex 4.5).

Depending on the applied recycling option, the obtained products can differ considerably.

Although chemical recycling aims to recover the carbon of the polymer, in the case of PVC, besides the recovery of the carbon content, either hydrogen chloride or a neutralised salt may also be obtained. Benzene is also mentioned as by-product (Ragaert, Delva, and Van Geem 2017; Vinylplus 2017; Vogel, Krueger, and Fabian 2020). In a best-case scenario HCl can be reused for the vinyl chloride production or in other chemical processes. In a steam atmosphere the hydrocarbons are obtained in the form of carbon monoxide, carbon dioxide and hydrogen.
In general, the better the dehydrochlorination process, the higher the efficiency of the recycling process. Initiatives therefore strive to increase the HCl yield via variations of the processes (Sadat-Shojai and Bakhshandeh 2011).

In general, the biggest advantages of chemical recycling over conventional mechanical recycling is considered to be the possibility to remove problematic or hazardous substances and contaminations from the cycle, to recycle mixed plastic waste and the possibility to reuse the resulting products as feedstock in the chemical industry. This view is countered by evidence that, currently, mechanical recycling is still more ecologically and economic advantageous (as discussed in the following sections) (Vallette et al. 2015; Vogel, Krueger, and Fabian 2020).
During the last years, chemical recycling initiatives in general have been increasing rapidly. Still, most initiatives are in a pilot stage and do not operate on an industrial scale.
According to Conversio Market & Strategy GmbH 2021)(2021) less than 1,000 tonnes of PVC are chemically recycled in the EU in 2020.[66] According to Simon and Martin (2019), chemical recycling on industrial scale could be expected from 2025 to 2030. Other experts do not expect much growth in chemical recycling of plastic or, more precisely, PVC waste (input Focus Group Recycling 2021). Some specific examples including process descriptions of more recent initiatives, as well as their advantages and challenges are provided in Annex 4.5. It should be noted that VinylPlus and PlasticsEurope see the biggest potential for chemical recycling of PVC waste in the Sumitomo metals and the Ebara-Ube technology (input Focus Group Recycling 2021) (see Annex 4.5).

5.3.2 Technological aspects of chemical recycling

Chemical recycling entails a depolymerisation process through thermochemical or chemical processes. The current technologies which can be considered forms of chemical recycling for PVC are:

- Gasification and
- Pyrolysis.

Annex 4.6 provides more detailed information on these processes. Two important considerations concerning the technological status of chemical recycling in relation to PVC should be mentioned. Firstly, a general advantage of chemical recycling is seen in the treatment of mixed or contaminated waste; in other words, input that cannot be recycled with high quality via conventional mechanical recycling processes (Plinke et al. 2000) (Focus Group Recycling 2021). Further the fact that a sorting process is not needed is often mentioned as a benefit of chemical recycling. However, practice shows that only few processes exist (e.g. ecoloop), where little or no pre-treatment is necessary. Other processes describe the necessity of the separation of the input material in order to gain pure waste fractions (e.g. Redop, Alzchem). Otherwise, energy-intensive purification processes would be necessary resulting in higher costs. Further, a lower process temperature needs better waste separation (Solis and Silveira 2020). If pre-treatment is a prerequisite, the advantages over mechanical recycling are diminished.

[66] This report does not count the treatment of PVC in blast furnaces chemical recycling. On this basis it can be assumed that chemical recycling of PVC is not practiced in the EU, except in pilot plants or within research projects. If chemically recycled at all, experts consider gasification as the process for the chemical recycling of PVC with the highest potential (input FocusGroup Recycling 2021).

Secondly, many chemical recycling initiatives do not seem to focus on PVC. Chemical recycling (especially pyrolysis) of PVC generates HCl. This acid can create several problems. One problem is the potential to cause corrosion of the process equipment. Another problem is linked to the fact that even small amounts of halogens in the output products can impede the use of the recyclate as petrochemical feedstock (Ragaert, Delva, and Van Geem 2017; Sadat-Shojai and Bakhshandeh 2011). Consequently, most plants limit the chloride content and therefore PVC waste as input material (e.g. Ebara Ube: 3-5%; Ecoloop and Alzchem: 10%). Nevertheless, it should be mentioned that some initiatives are based on established recycling processes and test the processes with the co-treatment of PVC (possibly under specific conditions - maximum chlorine content). The experience of co-treatment usually shows positive results, at least with regards to the fact that PVC does not interfere with the recycling process. In addition, some processes exist that can treat pure PVC waste (e.g. Sumitomo Metals).
Nevertheless, in most initiatives, PVC is not the targeted material and could only be processed in limited quantities. VinylPlus (2019) sees it as an advantage that existing infrastructure can be used to treat PVC, since it would decrease the investment risks and increase economic security. Experts at the Focus Group Recycling 2021 emphasised that processes for chemical recycling of PVC are rather new initiatives and therefore it will take time and more research to eliminate shortcomings and to become competitive with other large-scale (established) waste treatment processes.

The state of technological development of chemical recycling
Available evidence seems to suggest that currently chemical recycling is an immature technology. Even further, chemical recycling specifically of PVC is not practiced in the EU (except in pilot plants or in research projects); and in most initiatives of chemical recycling, PVC is not the targeted material.

5.3.3 Economic aspects of chemical recycling

It is difficult to assess the economic profitability for chemical recycling of PVC, since the processes do not operate on large industrial scale yet and data is limited. In addition, a large variety of processes exists and therefore costs differ. In general, it can be said that the profitability of chemical recycling depends on the investment costs for the plant, collection and pre-treatment costs, gate fee charges, transport costs, the process costs, the costs of the virgin products (especially petrochemical feedstock), the revenue of output products and the availability of substantial quantities of plastic waste for chemical recycling. The available literature does indicate that chemical recycling processes are only cost-effective if run in large facility capacities (Hopewell, Dvorak, and Kosior 2009; Solis and Silveira 2020; Vogel, Krueger, and Fabian 2020; Wagner and Schlummer 2020) (stakeholder input Focus Group Recycling 2021). EuPC stated that process costs for mechanical recycling are much lower than for chemical recycling due to the simpler technology. They assumed that the process costs for the latter will stay high but might decrease a bit due to scale effects (stakeholder input Focus Group Recycling 2021).

Since the 1970s technical but also economic problems led to closure of chemical recycling plants or the end of pilot projects. High costs for the treatment lead to inadequate revenue (Gleis 2012). In general, a study of Solis and Silveira (2020) concludes that the higher the temperatures the higher the expected purity of the final output products but also the costs. A high-quality output product can consequently achieve higher revenues. To lower the temperatures, save energy and costs, a catalyst could be added. The problem is that chloride and nitrogen components from the waste streams tend to deactivate the catalyst. This again could be counteracted with sorting and pre-treatment.

According to a US industry journal (2014) the capital costs for chemical recycling technologies are higher than for incineration calculated per kW. Calculating with an output of ~15 MW the cost differences are as follows (Renewable Energy World 2014):

Table 5-5: Capital costs for chemical recycling technologies compared to incineration (Renewable Energy World 2014)

Chemical recycling technology	Capital cost difference
Direct combustion	\$105-150 million (\$7,000 to 10,000 per kW)
Pyrolysis	\$120-172.5 million (\$8,000 to 11,500 per kW)
Gasification	\$112.5-165 million (\$7,500 to 11,500 per kW)
Plasma Arc Gasification	\$120-172.5 million (\$8,000 to 11,500 per kW)

Costs for the recovery of one tonne of mixed cable waste are included in the study from PE Europe on behalf of Vinyl2010 (Kreiβig et al. 2003). However, it should be noted that these figures are from 2003 and might have changed in the past two decades:

- Chemical recycling process – pyrolysis (Stigsnæs): 166 €/tonne
- Chemical recycling process – pyrolysis (Watech): 159 - 183 €/tonne

Comparison:

- Non-conventional mechanical recycling (Vinyloop): 50 €/tonne

5.3.4 Environmental and health aspects of chemical recycling

The analysed chemical recycling initiatives confirm the general technical feasibility of the chemical recycling of PVC waste. From a health and environmental point of view, chemical recycling must be assessed with regard to the protection of resources (including energy demand) as well as impacts on the environment or human health. In this regard, it is important to note that LCA results for chemical recycling are limited and should be interpreted with caution.

LCAs concerning chemical recycling

Some LCAs compare different plastic waste treatment/disposal options, including chemical recycling processes, but

- Most of them do not focus on PVC (e.g. in some studies PVC is even sorted out before treatment (Faraca, Martinez-Sanchez, and Astrup 2019)(Krüger 2020));
- Some show only parts of the LCAs (e.g. only selected results of impact categories and not all datasets which were used);
- Have different functional units and system boundaries, which make a comparison of the results difficult;
- Some studies compare chemical recycling and mechanical recycling, whereby the input material is not ideal for mechanical recycling (e.g. mixed plastic waste), leading to big shares of emissions attributed to incineration;
- The chemical recycling processes presented are initiatives on experimental scale or pilot projects consequently the results might differ (in a positive or negative way) if the processes are carried out on industrial scale;
- Some studies assume that the output from the chemical recycling process can create products with a quality compared to virgin plastics. This assumption cannot be confirmed as described above; and
- The environmental impact of plastic waste depends on the product it replaces.

(Jeswani et al. 2021; Zero Waste Europe 2020)

5.3.4.1 Emissions of hazardous substances from chemical recycling

An advantage of chemical recycling is the potential to remove additives and contaminants (Vallette et al. 2015; Wagner and Schlummer 2020). On the other hand, similar to incineration processes of PVC, the process of chemical recycling can produce dioxins (more details included in chapter 6).

Comparing the dioxin generated after pyrolysis and other combustion processes of PVC powder, results show a lower emissions for the pyrolysis process: 4.72 ng or 0.215 ng TEQ for pyrolysis and 122 ng or 4.6 ng TEQ for combustion (M. Zhang et al. 2015). Tests of thermal treatment of automobile shredder residues (3.9% PVC) by pyrolysis or gasification show that dioxin and dioxin-like PCBs are produced (at any operating condition). It seems that oxygen and catacytic metals facilitate the formation of dioxin and dioxin-like PCBs. Several pollution abatement technologies for removing dioxins and furans from flue gas exist (G. Zhang et al. 2019; M. Zhang et al. 2015) (see Annex 5.3 for further information). Nevertheless, sources indicate that thermal treatment of PVC-containing automobile shredder residues must be carried out with caution (M. Zhang et al. 2015).
It seems additives can increase or decrease the generation of dioxins. Although, to identify which additive contributes to which effect is not easy and still unclear (M. Zhang et al. 2015).
The other problematic products are hydrogen chloride (a sensory and pulmonary irritant) and carbon monoxide (an asphyxiant). According to Rollinson and Oladejo (2020) new carbon bonds can be formed when chlorine is removed. Aromatics such as indene[67], and naphthalene[68] could be created which are hazardous to human health.

Output products (e.g. pyrolysis oil) can be contaminated, resulting in the necessity of purification processes or in the use of only a very low proportion of the output (Rollinson and Oladejo 2020; Zero Waste Europe 2020).
Extensive purification activities of the process equipment are another consequence of the produced dioxins and furans. HCl could be neutralized with calcium carbonate (lime) and/or sodium hydroxide (caustic soda) to convert the released HCl to the salts (see e.g. Ecoloop); but, since these salts are often too contaminated they must be disposed of in special landfills (Hann and Connock 2020; Sadat-Shojai and Bakhshandeh 2011). Other measures might include catalytic cracking. This might inhibit the formation of chlorinated hydrocarbons. Catalytic dechlorination might lower the toxicity and generating reusable raw materials (Ragaert, Delva, and Van Geem 2017).

5.3.4.2 Energy demand and global warming potential

Chemical recycling is a highly energy-intensive process since reactor temperatures are high and internal temperature stability must be guaranteed. And compared to mechanical recycling, more process steps are necessary (Sadat-Shojai and Bakhshandeh 2011; Wagner and Schlummer 2020; Zero Waste Europe 2020). In some processes the generated gas can be re-used for the energy demand of the recycling process and therefore conserve input of fossil fuels (e.g. ecoloop, Alzchem). Some LCAs from industry show less energy demand for chemical recycling processes than mechanical recycling but this result is contested by environmental NGOs. According to Zero Waste Europe (2020), a disproportionate allocation of by-products might be one reason for this result. In the corresponding studies[69] by-products in the chemical recycling scenario were allocated to the treatment in cement kilns whereas in the mechanical recycling scenario

[67] https://echa.europa.eu/substance-information/-/substanceinfo/100.002.176
[68] https://echa.europa.eu/substance-information/-/substanceinfo/100.001.863
[69] BASF, CE Delft, Keller, Plastic Energy (Zero Waste Europe 2020)

incineration was assumed[70] . Generally speaking, the hydrocarbon fraction of PVC waste can positively influence the energy balance by generating heat and electricity, but PVC has a relatively low calorific value compared to other plastic types or materials.

Table 5-4: Calorific values of some plastics and natural substances (Gesamtverband der Deutschen Versicherungswirtschaft e.V. 2000)

Material	Calorific value [MJ/kg]
PE	46,1
PP	44,0
PS	40,2
PA	31,0
PC	30,6
PVC	18,0
PTFE	4,2
Oil (fuel)	42,8
Cole	30,4
Wood	18,5
Paper	16,8

The energy advantage created specifically by PVC is therefore estimated to be low. Independent studies came to the conclusion that, so far, mechanical recycling in general requires less energy than chemical recycling and in general chemical recycling has no net-positive energy balance (Hann and Connock 2020; Oladejo 2020; Sadat-Shojai and Bakhshandeh 2011; Zero Waste Europe 2020). According to Wagner and Schlummer (2020) in such cases, where energy consumption for the recycling process exceeds the caloric value of the polymer, incineration could a better option. However, other environmental impacts need to be considered as well.

It should be noted that a study carried out by **PE Europe** commissioned by VINYL 2010[71] (Kreiβig et al. 2003) compared the treatment of mixed cable waste (with 68% PVC incl. filler, plasticizer and other additives) through the treatment option of energy recovery (in MVR), chemical recycling via pyrolysis (Watech), chemical recycling via hydrolysis and pyrolysis (Stigsnaes), non-conventional mechanical recycling via Vinyloop and landfilling. The energy recovery rate is calculated with 0% for landfill, 22% for MVR, 35% for Stigsnaes, 38% for Watech and 64% for Vinyloop.
The same study (Kreiβig et al. 2003) also draws conclusions concerning global warming potential. The Vinyloop process seems to be the only process with a negative global warming potential (-220 kg CO_2/eq/FU)[72], the two chemical recycling process have a net global warming potential (375 and 550 kg CO_2/eq/FU) comparable to landfill (400 kg CO_2/eq/FU). With regard to the net acidification potential, all options (except landfill) show a net benefit with the highest for the recycling options (Kreiβig et al. 2003). Advantages of the chemical recycling processes, as shown in this study, are the relatively low amount of waste generation and the recovery of up to 99% lead (concentrated in the heavy metal fraction or solid products). In addition, high recovery yields of chlorine are reported (for the recycling options all between 94 and 99%) (Kreiβig et al. 2003). But as stated in chapter 5.3.4 these results need to be interpreted with caution.
Further results of LCAs on chemical recycling (focus not on PVC) can be found in Annex 4.7..

[70] It must be noted that the LCAs assessed by Zero Waste Europe do not include specific LCAs on the treatment of PVC waste and can therefore only apply to general assumptions of chemical recycling

[71] 1st Voluntary Commitment of the European PVC Industry

[72] Due to the recovery as PVC compound.

5.3.4.3 Resources

The overall aim of chemical recycling is to gain products, materials or substances which can be reprocessed whether for the original or other purposes. The initiatives do not clearly state if the gained chemicals are used to produce new PVC. The Ebara Ube process, for example, uses the recovered chlorine as fertiliser agent. The report 'An investigation of the U.S. Chemical Recycling Industry' came to the conclusion that none of the three currently operating chemical recycling facilities are recovering plastic to produce new plastic (Patel et al. 2020). The fact that no closed-loop chemical recycling process exists for PVC was confirmed by stakeholders at the Focus Group Recycling (2021). The main reason behind this is that chemical recycling is currently not carried out on a large industrial scale (stakeholder input Focus Group Recycling 2021).

For example, PVC is used in the Alzchem process to replace coal and coke as a substitute fuel, with the main goal being the calcium carbide production (Ciacci, Passarini, and Vassura 2017). Substituting fossil fuels has of course environmental benefits but does not aim the (closed-loop) recycling of PVC, which according to the waste hierarchy (Article 4 of the WFD) would be preferred. For some processes it even appears that treating PVC requires additional resources, since PVC has a low calorific value. For example, within the Sumitomo Metals process 7-10% additional coke or wood as carbon source is necessary to guarantee a steady operation. For the ecoloop process, lime has to be added to bind harmful substances (PVC presence may not be the only reason here).

The **PE Europe** study calculates a material recovery rate between 50 and 70% for the chemical recycling processes (Kreiβig et al. 2003). Thereby, the Watech process recovers mainly $CaCl_2$ (calcium chloride), coke and organic condensate. During the Stingsnaes process the main recovered material are solid restudies (not further specified), oil and NaCl (sodium chloride). (Kreiβig et al. 2003). It is not mentioned for which products the recyclates are used.

Based on the above, it can be concluded that the resource efficiency gains from chemical recycling in terms of PVC recycling are currently questionable. This may partly be due to the fact that the recycling of PVC is currently not the main focus of the majority of chemical recycling initiatives.

Additional considerations on pyrolysis
It should be noted that the NGO Zero Waste Europe (2020) questions whether the generated output (in this case oil from pyrolysis) can be compared with the quality of virgin products. Some studies (listed in (Rollinson and Oladejo 2020)) found that pyrolysis oil contains toxic pollutants. This could be solved through a purification and upgrade process, which is energy and carbon intensive, or to dilute the oil with virgin feedstock. The best way to achieve high quality output is based on a clean and homogeneous plastic waste. But should this be a pre-condition, current PVC waste could be mechanically recycled rather than treated with pyrolysis.

Comparison between chemical and mechanical recycling
From an environmental perspective, comprehensive LCA results for chemical recycling are limited and should be interpreted with caution. Compared to mechanical recycling, chemical recycling is a highly energy-intensive process. Studies show that through gasification or pyrolysis of PVC waste, dioxins and dioxin-like PCBs can be produced. Even output products (e.g. pyrolysis oil) can be contaminated, requiring purification processes.

5.3.5 Summary on chemical recycling

In general, the greatest advantages of chemical recycling processes are the possibilities to recycle mixed and contaminated plastic waste, to remove impurities and hazardous substances and to reuse the resulting products as feedstock for the chemical industry. Therefore, it is often argued that chemical recycling processes could be used for waste streams, where conventional mechanical recycling has its limits in order to avoid energy recovery of the plastic waste (which is not prioritised by the waste hierarchy).

A first conclusion from the analysis in previous sections is that PVC is not the target material in any chemical recycling processes or initiatives and consequently closed-loop chemical recycling for PVC (monomers) does not exist. The main purpose of the input of PVC in existing chemical recycling initiatives is rather to investigate where PVC waste does not interfere with the recycling process of other materials or to generate syngas. Hence, it is assumed that currently chemical recycling of PVC is not practiced in the EU, except in pilot plants or in research projects.

A second conclusion is that, especially for PVC waste, chemical recycling currently has no ecological or economic advantage over mechanical recycling, since the latter process is less complex, less energy intense and well-developed. Since chemical recycling of PVC generates HCl which could cause corrosion of the process equipment most chemical recycling plants limit the chloride content and therefore PVC waste as input material. Further, studies show that through gasification or pyrolysis of PVC waste dioxins and dioxin-like PCBs can be produced. Even output products (e.g. pyrolysis oil) can be contaminated, resulting in the necessity of purification processes.

Nevertheless, it is important to note that different opinions exist on how chemical recycling of PVC waste will develop in the future. Consequently, time and further research is necessary to achieve technological progress in the field of chemical recycling in general[73] and more specifically for PVC waste and to obtain clear results comparing the environmental impacts of chemical recycling with (non) conventional mechanical recycling and other waste treatment options.

Important considerations regarding chemical recycling
Since chemical recycling process for PVC do currently not operate on industrial scale, data is limited and economic profitability is impossible to assess. Experts point out that chemical recycling processes are generally more cost-intensive than mechanical recycling processes because chemical recycling is more complex and energy-intensive.

5.4 Decontamination techniques for removing legacy additives from PVC waste

5.4.1 The relevance of decontamination

As described in chapter 5.2, contaminants in PVC can occur in various forms. The most prominent group of contaminants that impair PVC recycling are legacy substances, such as heavy metals (e.g. cadmium and lead) or phthalates (e.g. DEHP) (see chapter 2). Removing legacy substances from PVC before it re-enters the material cycle is crucial to balance the policy goals of a toxic-free environment and to keep materials in their life cycle (e.g. through recycling) (Everard and Blume 2020). In addition, the market acceptance of products made from PVC recyclate depends on the assurance that legacy additives either have been removed or that they have been deemed safe for reuse (Vinylplus 2020b). Within the context of these challenges, a number of decontamination practices can be considered. The main decontamination activities presented below aim to remove

[73] PlasticsEurope will invest from 2.6 billion € in 2025 to 7.2 billion € in 2030 in chemical recycling technology. However, this investments concern chemical recycling in general and PVC waste is not explicitly mentioned (PlasticsEurope 2021a).

legacy substances to generate PVC recyclate that is compliant with the limits of REACH and other legislation and which has the same quality of virgin PVC. For future decontamination, it is intended to implement the decontamination processes as a step in the recycling process. However, it should be noted that under the currently available recycling technologies applied on a commercial scale, legacy additives cannot be removed. Instead, selective dissolution processes or chemical recycling technologies would need to be applied. Currently none of the selective dissolution processes or chemical recycling technologies presented below are applied on a commercial scale (input Focus Group Recycling 2021). Therefore, the assessment conducted in this chapter is based on the experiences of former plants (Vinyploop) and on pilot plants and on ongoing projects.

Considerations regarding co/tri-extrusion
One option to reduce the overall content of legacy additives is the application of co/tri-extrusion which results in an encapsulation of recyclate between one or two layers of virgin PVC. This might result in diluting the contaminated recyclate with virgin material. Yet, this practice is contrary to the objective of a toxic-free environment as the legacy additives remain in the recyclate and the risk that they re-enter the environment is not fully excludable. Further legal considerations on dilution and encapsulation will be discussed in chapter 5.5.2.5. According to Wagner & Schlummer (2020) moving towards a fully circular economy and a toxic-free environment requires legacy additives to be removed from the PVC compound prior to the recycling process. Due to the wide range of additives present in PVC application, the complete removal of all potential additives poses a significant challenge to plastic-recyclers. Current projects targeting the decontamination of legacy additives from PVC are still in an early research stage (input Focus Group Recycling 2021).

5.4.2 Options for Decontamination of PVC

5.4.2.1 Sorting of input material

For mechanical recycling in particular where the PVC polymers remain intact, sorting technologies are critical to limit legacy additives re-entering the material cycle. With conventional mechanical recycling processes, the legacy additives cannot be removed or separated. Only intensive pre-sorting could prevent PVC compounds containing legacy additives and other unwanted decontaminates from re-entering the material cycle. However, today, the majority of PVC post-consumer waste still contains unwanted additives (e.g. lead). Pre-sorting would lead to a significant decrease in recyclable volumes and is therefore not practiced. To manually test specific batches of PVC waste, X-ray fluorescence (XRF) and near-infrared (NIR) spectroscopy can be used. Due to economic reasons no automation of XRF technology for PVC waste is in place (input VinylPlus 2021). It should be noted that the identification of organic plasticiser is even more difficult. It requires that samples of the input material are analysed in a laboratory (Focus Group Recycling 2021). Table 5-10 gives an overview on sorting technologies that could be used to remove plastics that contain legacy additives during conventional mechanical recycling.

Table 5-10: Overview on sorting methods that allow to detect additives or plastics containing certain additives.

Sorting technology	Legacy additives that can be removed	Use in PVC recycling to detect (legacy) additives or flame retardants
XRF	Metals, bromine additives, lead to a certain concentration.	Rarely used as majority of products still contain unwanted or legacy additives. I.e. there is no added value of sorting them out.
NIR spectroscopy	NIR spectroscopy is capable of detecting plastics with a high rate of legacy additives. NIR cannot detect additives such as bromine (4R Sustainability Inc. 2011).	NIR technology is used to separate different plastic types from each other. In practice, PVC containing legacy additives are not separated from other PVC parts.

5.4.2.2 Selective dissolution processes

Most additives (e.g. DEHP) used in PVC are not covalently bound to the matrix, i.e. physical recycling methods (like selective dissolution) should be sufficient to remove them from the plastic. Through selective dissolution, PVC can be filtered out in its decomposed state. At the same time, other components of the input mixture remain undissolved. Originally invented for the purpose of PVC recycling, selective dissolution technologies have the potential to remove additives and other contaminants from the input material. Since only the selected polymer is dissolved in the process, the undissolved fraction consisting of contaminants (e.g. undissolved plastic fillers, reinforcing materials, metals, impurities, and additives) can be filtered out. After separating the solution from the dissolving residue, the solvent is evaporated from the solution, and a pure plastic powder is produced (Martens and Goldmann 2016). The solvents are reused to a large extent.

A number of limitations are associated with selective dissolution technologies. Compared to conventional mechanical recycling, solvent based methods are often more complex and have been developed to tackle PVC products that are more difficult to recycle. These materials are often composite materials or too contaminated to be accessible using conventional recycling (Vinylplus 2020b). Examples of such waste streams include PVC cables where PVC could be contaminated with copper, or tarpaulins in which PVC is combined with polyester fibre. Another disadvantage of selective dissolution concerns the high associated costs of the technology, its high complexity and the partial toxicity of the used solvents. Thus, selective dissolution processes are limited to a certain throughput and therefore are particularly suitable for input streams that cannot be processed in sufficient quality via conventional process chains. Moreover, several projects on selective dissolution technologies focus on flexible PVC, which only contains small amounts of heavy metal stabilizers but larger amounts of DEHP.

Selective dissolution processes are discussed since several years as a solution to challenges that could not be resolved by conventional recycling. However, at this moment, the available technologies for selective dissolution that focus on removing legacy additives (especially DEHP) are still in aresearch and testing phase (input Focus Group Recycling 2021) and not applied on a commercial scale.

The two most advanced selective dissolution processes which are aimed at PVC recycling and legacy additives are the CreaSolv® Process and the (closed) Vinyloop plant.
The CreaSolv® Process was developed in 2002 and large-scale applications for EPS are in place. The CreaSolv® Process allows to separate legacy additives like DEHP and other hazardous embedded impurities (Input CreaCycle GmbH 2021). However, for PVC the CreaSolv® Process is currently only available on small scale. The Circular Flooring Project examines how PVC flooring can be recycled via the CreaSolv® Process. According to Fraunhofer IVV, the first results were

promising. However, it is still unclear if an industrial plant will be constructed after the end of the project (it ends in 2023) (Input Fraunhofer IVV, 2021).

The only industrial-scale plant (VinyLoop) for the decontamination of PVC in Europe closed in 2018, possibly as a result of the restriction of DEHP under REACH (PlastEurope 2018). The VinyLoop plant was able to remove foreign material contaminations from tarpaulins and cable sheathing, but legacy substances were not removed in this process (input PlasticsRecyclers 2021). Due to the chosen process design, it was not feasible to separate DEHP from the input material economically (expert input Focus Group Recycling 2021).

Table 5-11 gives a more comprehensive overview on existing selective dissolution processes, their capability to remove legacy additives and their technical and economical maturity. The identified processes are described in Annex 4.4 in more detail.

Table 5-11: Overview on selective dissolution processes and the possible removal of contaminants

Selective dissolution process	Contaminants that can be removed	Technical and economical maturity
VinyLoop	PVC composites are dissolved in a fully recyclable solvent and separated into its different materials. Legacy additives were not removed.To remove DEHP the process design would have needed a complete new set up.	The first commercial plant (able to treat up to 10,000 tonnes of PVC waste per year) had to stop the operations as it was not feasible to separate the phthalate contaminants with the Vinyloop process in an economical way (PlastEurope 2018). A plant in Futtsu, Japan (26,000 tonnes of waste PVC) stopped its operation as well, since there was not enough input material to run the plant economically (Kobelco 2006; VinylPlus-2010 2007).
Polyloop	1) Aim is to recycle complex plastic waste such as PVC composite materials (Polyloop 2021). 2) Viruses, fungi and other bacteria can be removed with the Polyloop process. 3) Legacy additives are not removed. This is however not needed, as the input material is pre-consumer waste.	Polyloop is a start-up company, that is developing a recycling solution dedicated to complex plastic waste such as PVC composite materials (Polyloop 2021). The recycling process of Polyloop is based on the VinyLoop process and based on selective dissolution as well. It is mainly focused on pre-consumer waste which can be recycled locally at the manufactures site.
CreaSolv® Process	1) The CreaSolv® Process can remove at least DEHP and brominated flame retardants (althougho the latter is less relevant for PVC). 2) Cadmium and lead cannot be removed as metals	The Circular Flooring project aims to use the CreaSolv® Process for PVC flooring (see next row). According to Fraunhofer IVV, the CreaSolv® Process is theoretically applicable to several PVC product types (input Fraunhofer IVV 2021). The technological maturity level of the CreaSolv® Process applied to EPS is at around 7 (input Fraunhofer IVV 2021).
Circular Flooring	The CIRCULAR FLOORING project is further developing the CreaSolv Process in cooperation with Fraunhofer IVV. The process aims to remove legacy additives such as phthalates (e.g. DEHP).	A prototype plant applying the CreaSolv® Process for PVCrecycling, with a capacity of 15-20 kg/h is currently under construction (Cirular Flooring 2021). The project is EU-funded and ends in 2023.

Selective dissolution process	Contaminants that can be removed	Technical and economical maturity
Newcycling	Similar to CreaSolv® Process. Newcycling is suitable for the extraction of colour pigments and additives.	No PVC is currently processed with the method.
Remadyl	The process aims to remove phthalate plasticisers (mainly DEHP) and heavy metal-based stabilisers and further should have the potential to remove halogenated flame retardants. The projects focus is on Pb stabilizers, plasticizers DEHP, DBP, DIBP, BBP.	-EU-funded research project -Duration: 2019 – 2023

5.4.2.3 Chemical recycling

It is argued that chemical recycling processes (see in chapter 5.3) can recycle mixed and contaminated plastic waste or to remove impurities and hazardous substances, which is why it is seen as a promising method to be used where mechanical recycling has its limits. The assessment in chapter 5.3 makes clear that chemical recycling of PVC waste is currently not carried out at industrial scale, due to technological, environmental and economic challenges. In addition, most ongoing chemical recycling initiatives do not seem to focus strongly on PVC.

5.4.3 Fate of removed legacy additives

There is no general rule for the fate of the removed additives. Depending on the applied technology and the chemical properties of the included additives, the outcomes are different.

Two specific examples can be provided from the available literature with regard to the fate of legacy additives in solvents from selective dissolutions processes. In addition to the CreaSolv® Process, which is applied in the Circular flooring project, a technology to remove DEHP from the used solvent is under development. This process aims to transfer organic plasticizers with catalytic hydrogenation into non-critical alternatives for plasticizers (Cirular Flooring 2021). For heavy metals, i.e. lead from stabilizers, there are technologies in development that bind the heavy metal molecules together until they reach a sufficient size to be filtered out of the dissolution liquid (Thomas Diefenhardt, Fraunhofer IVV, Focus Group Recycling 2021)..

5.4.4 Economic aspects of decontamination processes

Currently none of the selective dissolution processes or chemical recycling technologies are applied on a commercial scale. Besides technological challenges for the removal of legacy additives, economic aspects are a major constraint (expert input Focus Group Recycling 2021). One aspect identified as a challenge for the potential upscaling of the decontamination technologies is the limitation in the available input material. According, to VinylPlus at least 30.000 to 40.000 tonnes of throughput are necessary to economically operate a plant using selective dissolution.
Given the above, information on costs are difficult to obtain. The only information available concerns the Remadyl process: The cost for the recycling of one tonne of PVC via the Remadyl process is estimated to be 570 €, including investment and operating expenses (European Commission 2019; Remadyl 2021). With prices for virgin PVC of 800-900 €/tonne in 2019 this seems to be a competitive cost (expert input Focus Group Recycling 2021). Although it is difficult to estimate the cost of decontamination of PVC in large-scale plants, incineration and mechanical recycling are expected to be more cost-effective alternatives.

5.4.5 Summary on decontamination techniques

Decontamination techniques to remove legacy additives from PVC waste are still scarce.

- Mechanical recycling cannot afford to remove legacy additives from the PVC matrix, since pre-sorting would limit the volume of recyclable PVC. As such this possibility is rarely used in practice. Conventional mechanical recycling is not able to remove legacy additives from the PVC material;
- while chemical recycling is considered a promising method where mechanical recycling has its limits, currently the application of this technology in general and for PVC also remains limited due to technological, environmental and economic challenges.
- Selective dissolution might be a solution to remove legacy additives from PVC. However, at this moment, most of the available technologies for selective dissolution are not available at industrial scale. The only industrial-scale plant (VinyLoop) for the decontamination of PVC in Europe closed in 2018.
- With regard to the ongoing development initiatives, it is relevant to note that two European projects (Circular Flooring project and Remadyl, both ending in 2023) currently develop methods to remove DEHP, cadmium and lead from PVC. The Circular Flooring project can already successfully remove DEHP from flooring (flexible PVC) with the CreaSolv® Process. The project also includes research on the removal of DEHP from the solvent. The Remadyl process seems to be economically viable with costs of 570€/recycled PVC tonnes. The former VinyLoop plant, as well as its successor Polyloop, are not able to remove legacy additives in an economically feasible manner.

Lack of technology to remove restricted/unwanted additives
Selective dissolution allows to remove restricted/unwanted additives from PVC waste. However, today, this technology is not implemented at industrial scale.

5.5 PVC recyclate in a Circular Economy

In a circular economy resources are recirculated in product cycles. The targeted output of recycling processes are recyclates (e.g. pellets, powder, flakes) that can be recirculated in new products. Recyclates play a major role in a circular economy, as their quality and quantity might influence the possibility to close cycles. This chapter assesses the potential and existing limitations of PVC recyclates that can hinder or boost a circular economy. However, it needs to be acknowledged that the potential for PVC recycling, including treatment and reuse options, is largely influenced by the chemical formulation and the choice of additives in the production stage (The Natural Step International 2018).

5.5.1 Application of PVC recyclate (R-PVC)

It is important to note that, to date, almost all recycled PVC has been produced through mechanical recycling processes. As regards chemical recycling of PVC, the analysis in previous sections shows that currently PVC is not the target material in these processes and therefore PVC, and PVC monomers, are not reused. If chemical recycling of PVC waste takes place at all, no new PVC products are produced with the output. Consequently, no closed loop-recycling exists for PVC through chemical recycling. Further, stakeholders claim (PlasticsEurope input Focus Group Recycling 2021) that monomers gained at chemical recycling processes cannot be reused in multiple circles or an unlimited manner, especially not vinyl chloride.

Figure 31 shows how much of the available PVC recyclate was processed into specific products.

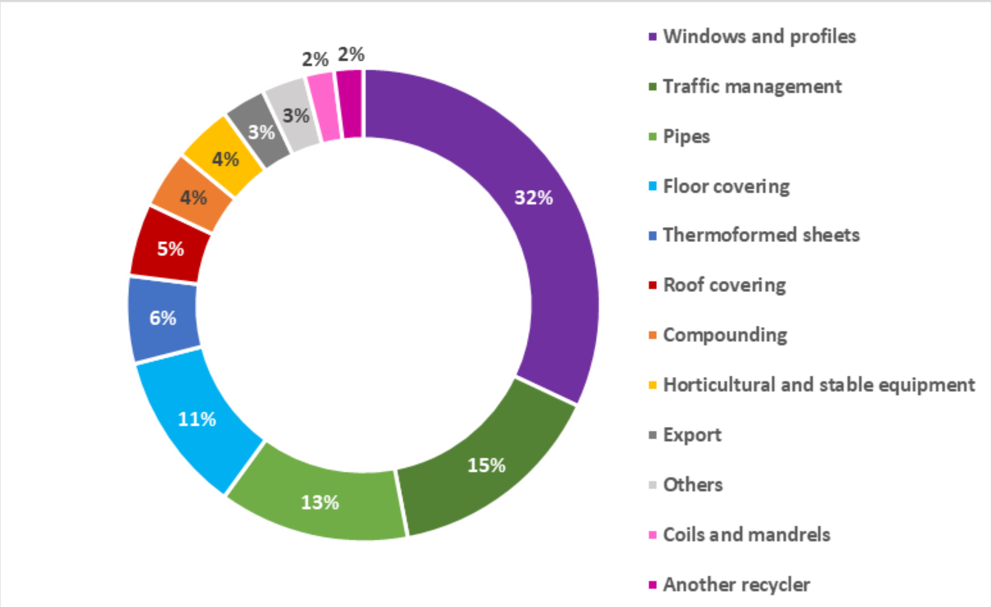


Figure 31. Use of recycled PVC (R-PVC) distributed on secondary products, 2018, adapted from (Vinylplus 2020b)

It is important to note that, due to lacking quantity or quality of PVC recyclate, many PVC end-of-life products are not redirected into their original product group but into another application of lower value. Table 5-12 shows which PVC end-of-life product is recycled into which new product. Moreover, the table gives an overview of the average recyclate content in new products.

Table 5-12: Applications for recyclate from rigid and flexible PVC products

End-of-life product	New product type	Average recyclate content in new product [%]
Pipes	-Pipes & fittings	20-100%
Window profiles	-Window profiles -Pipes & fittings	1) 5-10%: Average recyclate content in window market; 2) 18%: Average recyclate content in Germany; 3) 40-45%: Maximum average recyclate content (input EPPA 2021).
Cables	-Industrial Flooring (e.g. general purpose compounds) -Thick-walled products for road traffic safety (e.g. beacon bases or guard rail elements)	Not specified
Flooring	Flooring (when using pre-consumer waste)	1) Homogenous layers: 25%; 2) Heterogenous layers: 3-30% (input Tarkett 2021).
Roofing Membranes	-Membranes -Liner -Products from compression moulding (e.g. traffic products for road traffic safety) -Stall mats	Not specified

The table above provides three important indications concerning the current and potential contribution of PVC recyclate to the circular economy.

Firstly, there are indications that closed-loop recycling (i.e. recycling of products within their own product category) is currently limited to a number of applications. This is not a disadvantage per se, since PVC from specific products can be similar and can thus function effectively as input material (see section 5.5.2.5). However, from a circular economy perspective, closed-loop recycling could be significantly optimised. For example, a stronger focus on closed-closed loop may provide an incentive to find solutions for recycling limitations within a specific PVC product category, rather than seeking the most feasible application across a number of product categories (e.g. using PVC window profiles waste for production of pipes). In addition, closed-loop recycling may limit the cross contamination between PVC product groups with legacy additives.

Secondly, there are indications that PVC recyclate from a number of sources (e.g. cables or roofing membranes) may be currently downcycled. Such downcycling may limit the number recycling cycles and prevents the creation of markets for high quality secondary materials.

Thirdly, there are indications that, overall, the current share of recycled PVC in new products remains limited or unclear. While the aforementioned indications may be caused by various technical, economic and regulatory challenges, it is important to take them into account when assessing the current and potential contribution of PVC recyclate to the circular economy. This is discussed in more detail in the next sections of this chapter.

Considerations concerning alternative applications of PVC recyclate
As an alternative to the conventional uses of PVC recyclate as described above, PVC waste can be included into other (non-plastic) material to use the plastic's properties. These alternatives for the use of end-of-life PVC might be considered as downcycling (see Annex 3.1). Some main examples are assessed below. One option is to include plastic waste in concrete to decrease its density. Such 'light concrete' is currently manufactured using polystyrene and can be found in non-structural elements like roofs, insulation walls and slabs covering gutters. According to VinylPlus, one advantage of this option is that it can be applied in several small plants (Vinylplus 2017). Similarly, PVC pipe recyclate could, for example, replace river sand for the production of concrete. Sadat et al. summarize that the as-produced concrete in which sand is optimally replaced by recycled PVC (15% by volume) has lower density, higher ductility, lower drying shrinkage, and higher resistance to chloride ion penetration (Sadat-Shojai and Bakhshandeh 2011). However, the results showed that the included PVC reduces other mechanical properties of concrete such as lower workability, lower compressive strength and lower tensile splitting strength (Haghighatnejad et al. 2016) (Sadat-Shojai and Bakhshandeh 2011). In other cases, 3% to 5% of bitumen for road construction were replaced by PVC pipe recyclate. The tests were considered successful as strength and stability of the mix increased after incorporation of PVC pipe waste. Increased resistance to permanent deformation was also observed (Behl, Sharma, and Kumar 2014).However, these non-closed loop approaches for PVC pipe recyclate have to be carefully examined, as there is no surplus but a shortage for recyclates of PVC pipes (e.g. due to long life time of a pipe)) (input TEPPFA). Additional processes using pipe recyclate would even lower the amount of available recyclate for the pipe production. This is an even greater issue if one process (e.g. replacement of bitumen) leads to open-loop recycling whereas the redirection of pipe recyclate in the pipe industry is seen as closed-loop recycling. According to VinylPlus, PVC recyclate has also been tested along with other plastics such as PE and PP to be included in plastic-wood composites. Plastic-wood composites are gaining a share of the decking market and

some firms are investigating new applications, for example, structural wood lumber and cladding. In the US some companies claim that their decking is already manufactured from 95% recycled content (Vinylplus 2017).

It is important to consider that the use of PVC in non-plastic applications such as concrete, bitumen or wood may lead to a complex and composite material from which the PVC cannot be feasibly removed, thus blocking further recycling of PVC after the second life cycle. Such limitations on further recycling may pose challenges for the EU´s current circular economy objectives concerning increased recycling. Moreover, it is unclear what the implications of this kind of PVC applications are regarding legacy additives, both during the use and end-of-life phase of such concrete products. Overall, it is interesting to note that such applications are considered, despite the clear limitations which complex and composite PVC products pose for current recycling activities in the EU (see section 5.2.2.2).

5.5.2 Potential and limitations of PVC recyclate for a Circular Economy

Major aspects of PVC recyclate application are presented in the following chapter including its potential and limitations as a contributors to a Circular Economy.

5.5.2.1 Quantity and quality of recyclate

Quantity and quality of recyclate are interdependent in a Circular Economy as high volumes of high-quality recyclate are needed to limit the use of virgin material as much as possible. Lower quality grades of PVC recyclate cannot be used for the original products they are recycled from. Limited recyclate quantities lead to a continuous use of virgin material. Thus, resource quality is a driving force towards economically and environmentally sustainable recycling (Faraca, Martinez-Sanchez, and Astrup 2019).

To date, the amounts of post-consumer recyclate from rigid PVC currently available are not sufficient to include recyclate into all new window profiles (EPPA profiles 2018). In the case of pipes insufficient recyclate (originating from pipes) exists, so that pipe manufacturers include other PVC recyclate (input TEPPFA 2021). Only 3.8% of the recyclate used in PVC pipes originate from PVC pipes. The other 96.2% of the recyclate used in pipes are mixed recyclate of other products (mainly window profiles, cladding, roller shutters). On the other hand, 10-20% of recyclate gained from recycled windows are used for other products (e.g. pipes). As the largest PVC waste stream, 80–90% of PVC window recyclate are returned into new PVC windows. .

The association Plastics Recyclers Europe points out that sufficient quality in terms of physical mechanical properties can only be provided by material derived from the sectors in which it is mostly used, i.e. building and construction material. However, due to long product service lives in this sector (30-100 years), PVC recyclate with a quality (in terms of being devoid of lead and cadmium) will not be available for at least the coming decade (input PlasticsRecyclers 2021). Following this, improvements in collection and sorting will only result in an increase in the quantity of material with quality in terms of physical mechanical properties, but not an increase of material devoid of lead.

Faraca et al. point out that there is a systemic weakness of the recycling definition itself, because to date it is not focussed on recyclate qualitaty but purely on quantitaty. from a circular economy perspective "better recycling" may be preferable over "more recycling" (Faraca, Martinez-Sanchez, and Astrup 2019), although adequate recyclate quantities are essential to make recycling processes economically viable. As such, there seems to be a fundamental discussion on whether low-quality recycling is better than no recycling or if this approach leads to greater negative consequences. However, it is obvious that recycling is not a purpose in itself, but recycling needs to have its justification in concrete benefits, such as replacing virgin production and saving energy and CO_2 emissions.. Figure 32 gives an overview on factors that are either influencing the quantity or the quality of recyclates. These aspects were already described and assessed in chapter 5.2. The focus of this chapter lies on the use of recyclate and the impact of recyclate quantity and quality (see grey boxes in Figure 32).

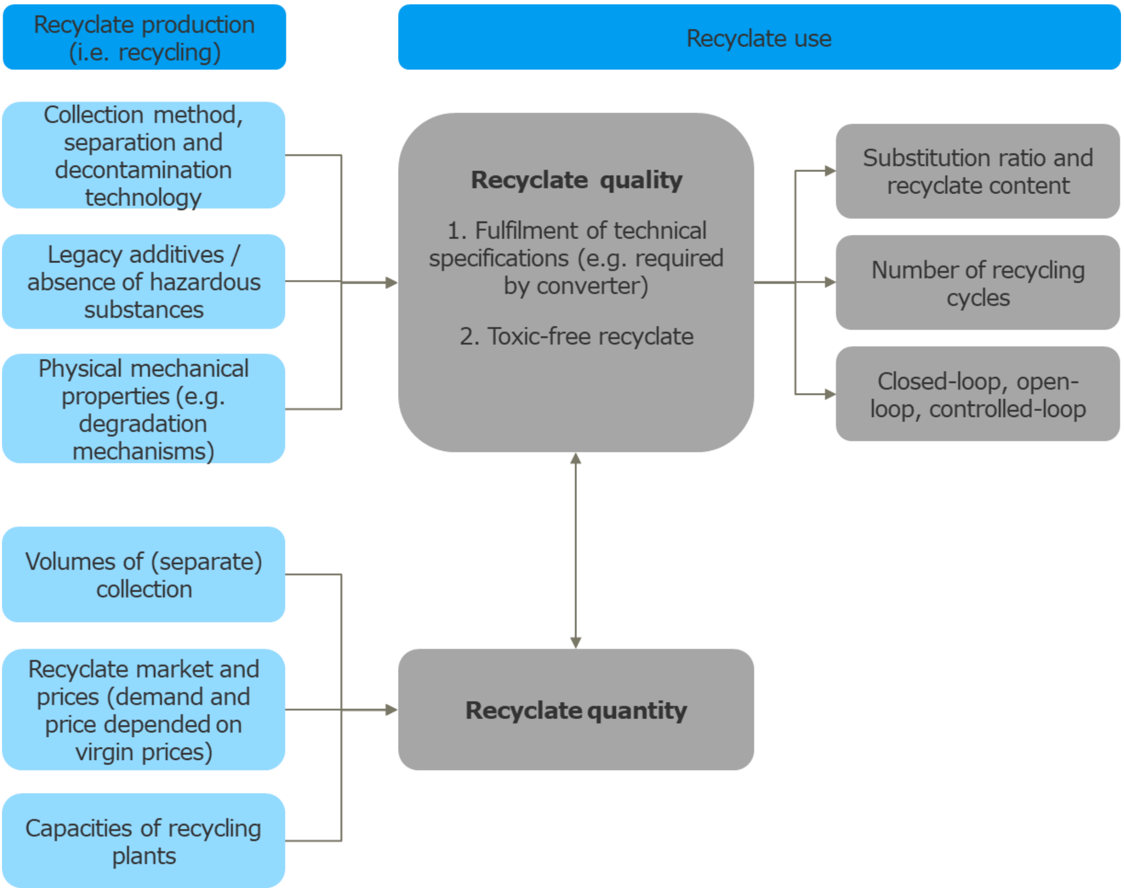


Figure 32. Influencing factors on recyclate quality and quantity (own image, inspired by (EuRIC Focus Group Recycling 2021)(Plinke et al. 2000)**)**

The recyclate quality influences the use ratio between virgin material and recyclate in a new product (substitution ratio), the number of recycling cycles and the transfer into closed- or open-loop recycling (Figure 32). Figure 32 also shows that next to the quality, adequate quantities of (high-quality) recyclate need to be available. The amount of recyclate depends on the waste volumes of (separate) collection which is directly related to the lifetime of products. Adequate recyclate quantities require sufficient capacities of recycling plants which is closely related to the recyclate market and the recyclate price (see chapter 5.2).

5.5.2.2 Share of substitution and recyclate content

To close the material loop in the circular economy, achieving recyclate uptake in new products near a 1:1 ratio is key[74]. The substitution ratio is closely connected to the recylate quality, as the quality is one factor determining the degree to which virgin material can be substituted by

[74] In the case of PVC the term "substitution ratio/share of substitution" was rarely used in identified sources. Instead average recyclate contents could be identified as applied term.

recyclates. Considerations concerning the substitution ration in a number of important PVC applications are described below. The findings imply that, currently, the full potential of recyclate uptake is not realised due to various technical and economic reasons.

PVC pipes

Depending on the legal standard, sewer pipes (grey colour) can include different percentages of recyclate (0%, 20% or 100%) (input TEPPFA 2021). Normally, 70% of recycled PVC can be used in such sewage pipes together with 30% virgin PVC (Fråne et al. 2018). Depending on the usage of a pipe, different amounts of recyclate are allowed. Due to hygiene and health protection, safety (e.g. withstand 10 bar pressure), and environmental protection concerns, only virgin material is utilized in the drinking water and gas supply sector. TEPPFA mentions that research is ongoing on how to maximise recycled content in pipes. TEPPFA members are prepared to use more recycled content without compromising on the quality of the pipe longevity (50-100+ years), performance, safety and hygienic requirements. Some of the members active on the Dutch market even have KIWA and SGS labels, certifying 40 or 50% recycled content for multilayer PVC sewage pipes (non-pressure). TEPPFA assumes that the demand for recyclate will strongly increase (input TEPPFA 2021).

PVC windows

As displayed in Figure 33 for the case of PVC windows, the maximum amount of recyclate that can be used in an individual product differs from the total recyclate content on the product market.

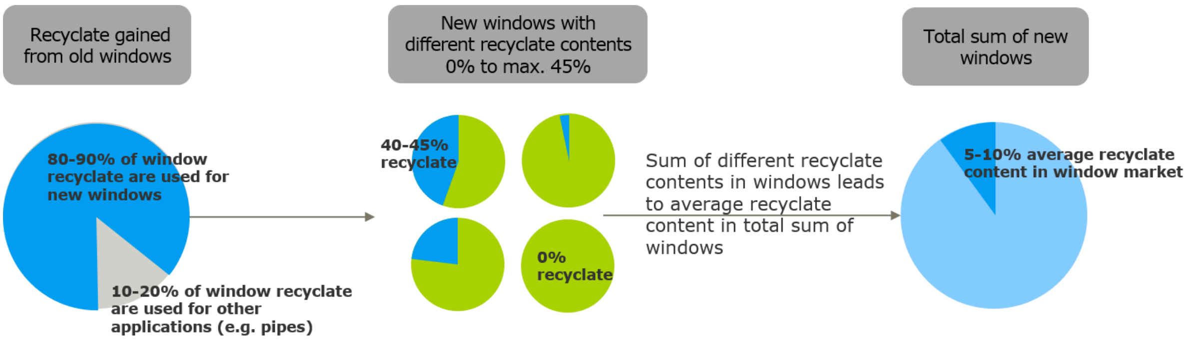


Figure 33. Overview on recyclate origin, its use in individual products and its total sum in a product market (the image is based on an example for window recycling).

For example, the average maximum recyclate content in a window profile (sash, casement) is usually 40-45% (input EPPA 2021). This is determined by the manufacturing process because recyclate is mainly used in the core of the profile (EPPA profiles 2018). Special constructions allow to achieve an average maximum recyclate content of 55% and in few cases up to 60%. Currently, there are products in the development phase to manufacture PVC-U window profiles out of 100% rPVC. These products, however, do not currently comply with requirements of EN 12608[75]. However, the window market includes not only windows with recyclate but also windows which have lower or no recyclate content. Due to this fact, across Europe the average recyclate content in the new window market is 5-10%, in Germany it amounts to 18% (EPPA profiles 2018) (Fråne et al. 2019). EPPA indicated that this range can be generalized for the entire sector and that 10% of European production is complemented with recycled material (Ramboll Environment and Health 2020). According to EPPA, the 5-10% amount derives from the fact, that not enough recycled PVC coming from used windows was available. For 2025, EPPA has set the target to achieve 25%

[75] DIN EN 12608-1: Unplasticized poly(vinyl chloride) (PVC-U) profiles for the fabrication of windows and doors - Classification, requirements and test methods

rPVC, on average in the European PVC window profiles production for the European market (input EPPA 2021).

PVC flooring

Tarkett noted that they can use recycled PVC (to date mainly pre-consumer waste) as a secondary raw material in homogenous or heterogenous flooring layers. Homogenous layers are mono-type layers composed of one or more layer of the same material composition. A typical composition accounts to 30% PVC, 30% mineral fillers and 30% additives, inks, plasticizers. 25% recyclate can be inserted in this layer type. However, a pre-recycling colour sorting is crucial, to achieve recyclate within specific colour ranges. Theoretically, it is feasible to include 55% of recyclate in a homogenous layer, but to date the quantity of recylate is lacking for this (input Tarkett, 2021). Heterogenous layers consist of a sublayer and a functional layer (e.g. dimpled; printed). 3-30% of recyclate can be inserted in the sublayer. The percentage of recyclate depends on the thickness of the sublayer in relation to the other layer(s). A 100% recyclate content would be unfeasible for heterogenous layers as aesthetic aspects like the colour are important. Especially for printing, a white coloured top-layer is needed (input Tarkett, 2021).

5.5.2.3 Number of recycling cycles

Another factor influencing a Circular Economy is the number of recycling cycles that a PVC recyclate can undergo before losing its necessary properties. Repeated life cycles can influence the concentration of a specific substance in a product (dilution or accumulation) depending on the recycling processes and resulting products. Based on this, the recycled product could be used in applications differing from the second life cycle (Blaeij et al. 2019).

Considerations concerning the relevance of recycling cycles
It should be considered that the recycling industry considers the approach of recycling cycles as an academic research field that does not necessarily fit with current practice. In practice PVC from different places and products is recycled together, i.e. they have already undergone a different number of recycling cycles. Hence, it is difficult to examine in which recycling cycle the mixed recyclate is directed. The Association of Plastics Recyclers Europe also indicates that most recyclate is directed towards applications with long lifetimes (e.g. construction material) making the question of how many times PVC can be recycled less meaningful than for example for packaging waste with a short life time.

Despite the different views on recycling cycles, specific polymer properties of PVC enable multiple material and raw material cycles before the technical performance of PVC deteriorates. The number of these cycles depends strongly on the former use and the properties of the material (rigid or flexible). After a certain number of recycling-cycles new additives can be introduced or the material can be blended with newer material to keep the product in use. It is not clear to what extent the age or the status of PVC is tested at recycling facilities to determine the need of adding new additives

Different data was identified concerning additional additive use in recyclate. According to Schyns and Shaver (2020) the added stabilisers to PVC are consumed during the processing and first usage which makes it necessary to add further stabilisers in the recyclate. The study of Yarahmadi et al. (2003) came to the same conclusion with regards to plasticizers. On the other hand, the same study has shown that post-consumer PVC floorings can be mechanically recycled, without adding new plasticizers (Yarahmadi, Jakubowicz, and Martinsson 2003). The study of Yarahmadi et al. (2003) suggested the same type of plasticizers should be used in the recyclate to avoid incompatibility. VinylPlus mentioned that the only company that is allowed to compound PVC with

DEHP (Plastic Planet) has no problems with adding other types of plasticizers. To avoid the re-adding of DEHP, other types of plasticizers are used.

Information on the following product groups have been identified:

- TEPPFA indicates that recyclate from **pipes** can be recirculated seven times. After this timeframe additives and stabilizer needs to be added to enlarge the lifetime (input TEPPFA 2021).
- It is estimated that PVC **window** profiles can be recycled up to seven (Umweltbundesamt 2012a, S. 88), eight (Vinylplus 2017) or 10 times (input BPF, 2021b) without adding additives and without loss of properties. EPPA points out that within the study from Leadbitter et al. four recycling cycles or an equivalent of 10 thermal histories were proven to be possible. The study experimentally showed that PVC-U formulations can be recycled four times without a significant loss in physical properties. As one recycling cycle comprises (i) injection molding, (ii) granulation, and (iii) pelletising, it means that PVC-U can pass 10 thermal histories without significantly losing it physical properties (Leadbitter 2002). In case of rigid PVC window profiles, usually no additives are added, as recycled PVC is used in the core of the profile where no requirements are placed on light fastness (input EPPA 2021).
- According to Tarkett, there is no limit to the numbers of cycles PVC **flooring** can be recycled. Recyclate destined for a homogenous flooring layer needs a reformulation i.e. addition of new additives or fillers whereas recyclate for a heterogenous flooring layer do not need this step as it is included in the sublayer. However, to date this is mainly performed for recyclate from pre-consumer wasteand installation waste.
- Flexible PVC needs a reformulation to fulfil specification of new products (e.g. higher softness). However, flexible PVC that undergo compression moulding do not need new additives. Independently from the number of lifecycles, new pigments are sometimes added, to adapt the colour to customer wishes (input VinylPlus 2021).

The examples show that pipes, windows and flooring can be recycled several times (7-10 times) without adding additives. Especially recyclate that is encapsulated in the core of a product (e.g. multilayer pipe or heterogenous flooring layer) is not enriched with new additives. Although adding of new additives can keep the product in use and to retain properties, at some point of time the recyclate will eventually lose its main characteristics. These points in time could not be identified. However, in the case of products with a long life expectancy, this will take a long time.

Reflection on closed loop recycling
Closed-loop recycling (i.e. recycling of products within their own product category) seems to be currently limited to a number of applications while other applications may be currently downcycled, thus limiting the number of recycling cycles and preventing the creation of markets for high quality secondary materials. There are indications that, overall, the current share of recycled PVC in new products remains limited or unclear.

5.5.2.4 Legacy additives in recyclate

The presence of legacy additives in PVC waste poses a considerable challenge for the uptake of PVC recyclate in products. Limit values on concentrations of restricted additives in products limits the use of recyclate containing such additives. Furthermore, current levels of additives in PVC waste may clearly exceed these limit values. For example, a use of only 10% of recycled PVC from flooring (due to its inferior quality) and 90% virgin PVC would already lead to a DEHP concentration of up to 2.0% in the (partly) recycled flooring (Blaeij et al. 2019). In addition, it is expected that various legacy additives will keep occurring in PVC waste in the future (e.g. lead) (see section 5.6.3). However, it should be noted that concentrations of legacy additives can decrease over time. For example, based on regular tests, industry states that the cadmium concentrations in recycled PVC are "far below the limit" or "substantially low". TEPPFA even suggests that the concentrations of cadmium (previously used as stabiliser) in multilayer pipes will decrease since the recyclate will be a mixture of pre-consumer and post-consumer PVC which does not contain cadmium (Ramboll Environment and Health 2020). However, it is important to note that cadmium was in many instances replaced as a stabiliser by lead, which was subsequently phased out voluntarily by the PVC industry and is subject to a proposed REACH restriction (see Annex 1). This example indicates that a sustainable solution to the issue of legacy substances may only be achieved through a combination of rigorous assessment of additives used in virgin PVC and carefuly balanced limit values for the concentration of legacy substances in PVC recyclate. In the absence of a careful assessment of characteristics of new additives, the regulatory solution of limit values for recyclate can be considered an imperfect treatment of problem symptoms. As indicated in chapter 5.4, technically and economically feasible options for the decontamination of PVC waste are currently limited. However, future developments in such technologies may provide further technical solutions for the challenge of legacy substances.

Encapsulation (co- and tri-extrusion)
With regard to legacy additives in a Circular Economy, the practice of co-/tri-extrusion (see Annex 3.1) requires explicit consideration. Tri-/co-extrusion increases the resource efficiency of recycled PVC but can be seen critically in the context of a toxic-free environment as the use of PVC recyclate containing legacy substances postpones the final disposal to a later stage. Co-/tri-extrusion is common practice in the industries of PVC pipes and PVC windows (used since over 30 years) as it allows to place recyclate between or under a virgin layer of PVC material, i.e. the recyclate is encapsulated between virgin material. Encapsulation is done for aesthetical reasons (e.g. gives window profiles a white surface) (Martens and Goldmann 2016), but also to prevent possible environmental and people's exposure to the additives in the recyclate (Polcher et al. 2020). The outer layer of virgin material is effectively preventing contact with the recyclate and leaching of legacy additives during product use.

According to EPPA (European PVC Profiles and related Building Products Association), in the case of window profiles, encapsulation of recyclate is the reason why the product-specific exposure and migration risks of legacy additives can be classified as rather low. Due to the encapsulation the inherent chemical properties and the hazardous substances concentration are in the core of the products (EPPA profiles, 2018). In the pipe industry, the encapsulation of recylate is done by using PVC recyclate as the middle layer in three-layered sewage pipes and other non-drinking water pipes. 70% of a pipe is the core layer whereby the inner and outer layer are each 15% of the total mass. The core layer (70%) may consist of up to 100% recyclate (input TEPPFA 2021).

According to several stakeholders, encapsulation is a relevant solution as it postpones the release of lead into an era where there may be a technology to safely remove lead from PVC. As most PVC products have a long service life, i.e many years will pass until co-/tri-extruded products will be disposed off again, a considerable amount of time should be available to develop removal technologies (Focus Group Recycling 2021 and input PlasticsRecyclers 2021). It should be noted that this argument is based on a measure of speculation and may only be relevant if PVC is traceable (e.g. via closed-loop systems). As indicated becomes clear from chapter 5.4, the development of decontamination techniques has been ongoing for a while now and development challenges are not solely of a technological nature.

Moreover, criticism has been raised in the literature concerning the point that PVC recyclates in new PVC window profiles or pipes will lead to increasing dilution and thus increasing emissions of pollutants into the environment (e.g. due to improper disposal, fires, etc.) (Hahladakis et al. 2018; Potrykus, Zotz, et al. 2020). It could indeed be argued that tri-/co-extrusion amounts to mixing of virgin PVC with contaminated PVC, meaning that the virgin fraction also becomes contaminated and more difficult to recycle and dispose of in the future (see Annex 1.1). In this regard, it may also be relevant to assess whether PVC recyclate containing high concentrations of legacy additives can be considered to have reached end-of-waste status in line with Article 6 WFD, or whether it should be considered hazardous waste (see Annex 1.1). As current recycling and decontamination processes do not remove legacy additives it may be difficult for PVC recyclate to reach end-of-waste status according to Article 6 of the Waste Framework Directive[76].

If PVC waste turns out not to have reached end-of-waste status and to be hazardous waste, encapsulation may amount to mixing hazardous waste with non-hazardous materials. This may be in contravention with the ban on the mixing of hazardous waste (Article 18 WFD). It is important to mention that currently various EU Member States do not seem to classify rigid PVC waste as hazardous waste, based on the assumed low bioavailability of included legacy additives. However, as indicated in chapter 2 the low bioavailability of legacy additives currently does not seem to be firmly supported by the available data. As such, an in-depth assessment of the classification of PVC waste and the circumstances under which it achieves end-of-waste status may be relevant.

Derogation from the mixing ban of Article 18 WFD
It is relevant to note that Article 18(2) WFD determines that, by way of derogation from the mixing ban, Member States may allow mixing provided where : a. the mixing operation is carried out by an establishment or undertaking which has obtained a permit in accordance with Article 23; b. the provisions of Article 13 are complied with and the adverse impact of the waste management on human health and the environment is not increased; and c. the mixing operation conforms to best available techniques.

It is also important to note that, within the context of the proposed REACH restriction on lead in PVC, ECHA (European Chemicals Agency)[77], has considered that recycling of PVC recyclate containing lead in applications such as three-layer sewer pipes or as middle-layer in window frames is preferable to incineration or landfilling. It is argued that a lower concentration limit of lead in recycled PVC articles is likely to increase overall releases of lead from PVC and therefore risk. The given reason for this is that a lower limit might result in a larger portion of the end-of-life PVC articles (waste arising) being directed to end-of-life disposal routes rather than being recycled. This would mean incineration which is associated with the greatest release to the environment. It is estimated that for each tonne of material that is not recycled there would be an average increase in lead releases of around 40 g (provided the current mix of disposal routes is maintained).

Finally, it should be noted that the above is mainly related to rigid PVC as recyclate from flexible PVC is not used for co-/tri-extrusion. Within the context of a potential restriction on lead in PVC,

[76] It might be especially difficult for PVC recyclate containing legacy additives to be in line with part (c) and (d) of Article 6:

- (c) the substance or object fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to products; and
- (d) the use of the substance or object will not lead to overall adverse environmental or human health impacts.

[77] Committee for Risk Assessment (RAC); Committee for Socio-economic Analysis (SEAC). (2018). *Opinion on an Annex XV dossier proposing restrictions on LEAD STABILISERS IN PVC*. https://echa.europa.eu/documents/10162/86b00b9e-2852-d8d4-5fd7-be1e747ad7fa

the RAC and SEAC committes under ECHA[78] mention that the concentration limit for lead does not need to be relaxed for applications involving flexible PVC (Committee for Risk Assessment (RAC); Committee for Socio-economic Analysis (SEAC) 2018). In current recyclates from rigid PVC the lead content is below 0.8% (Fråne et al. 2018).

Limits on further development of co-extrusion
VinylPlus states that further development of co-extrusion (e.g. for non-pressure sewage or stormwater pipes pipes) is limited by the acceptance of the customers which are -in the case of co-extruded pipes- mainly municipalities. For example, in the Nordic countries, Austria and Belgium, co-extruded pipe systems are not allowed (Vinylplus 2020b). In these countries the quality marks specify that only virgin material is allowed for the production of non-pressure sewage and stormwater pipes as the use of recycled content is considered as inferior and of lower quality. Secondly, a lack of material "at the right price" and of suitable quality hampers increasing the use of recyclates. TEPPFA is therefore working on a new approach aimed at increasing the R-PVC uptake in solid-wall pipes, such as sewer pipes approved according to the EN 1401 standard (Vinylplus 2020a).

Legacy additives as major challenge for the uptake of PVC recyclate in products
The presence of legacy additives in PVC waste poses a considerable challenge for the uptake of PVC recyclate in products. Limit values on concentrations of restricted additives in products limits the use of recyclate containing such additives. This is a problem that, against the findings of section 2 of this report, may increase in the future with further regulatory scrutiny towards problematic chemicals which are present in PVC.

5.5.2.5 Open-loop recycling

As open-loop and closed-loop recycling are discussed intensively, it cannot be concluded that only closed-loop recycling contributes to a Circular Economy. In open-loop recycling, the recycled material is used to produce a different type of product than the product the material was recovered from. This does not imply downcycling (see Annex 3.1) The new product can be of the same value (Huysman et al. 2015) (Ragaert, Delva, and Van Geem 2017). Several PVC waste streams are recycled in an open-loop:

- In the PVC recycling industry, it is common practice to use PVC **window** profile recyclate for PVC pipes, i.e. the recycled material is used to produce a different type of product than the product the material was recovered from. This is mainly due to the fact that PVC pipe waste is lacking in quantity. One stakeholder proposed to call the recycling of a building and construction application (e.g. window recyclate) into another building and construction application (e.g. pipe) "controlled loop" to demonstrate that it is close to closed-loop recycling but still not a complete closed loop (input PlasticsRecyclers 2021).
- One example for utilizing recyclate from PVC **pipes** in applications with lower requirements is the recycling system from the pipe manufacturer Pipelife. Here, PVC from gas pipes serves as the raw material for cable protection pipes. These can consist up to 80% of recycled material. At the end of their lifetime, the cable protection pipes are collected and recycled as material for sewer systems pipes. Pipelife calls this cascading concept "the pipe with three lives". As they assume a lifetime of PVC pipes up to 100 years their theoretical total service life can be around 300 years. With this concept plastics receive two further technically feasible life cycles (Pipelife 2019). Regarding this approach it has to be considered here as well that only a small amount of recyclate is incorporated in the pipes (input TEPPFA 2021).

[78] Committee for Risk Assessment (RAC); Committee for Socio-economic Analysis (SEAC). (2018). *Opinion on an Annex XV dossier proposing restrictions on LEAD STABILISERS IN PVC.* https://echa.europa.eu/documents/10162/86b00b9e-2852-d8d4-5fd7-be1e747ad7fa

- As the majority of **mixed PVC waste and flexible PVC** (e.g. flooring, cables) are not likely to be converted into pellets or flakes but directly pressed into thick-walled products such as traffic products for road traffic safety (Vinylplus 2017) (BiPRO 2002) these waste streams are considered to be recycled in an open-loop. According to the association Plastics Recyclers Europe this kind of open-loop recycling (see Annex 3.1) is not seen as generally "bad" as there is a need for material for traffic products and the use of recyclate can displace virgin material production in this sector (input PlasticsRecyclers Europe 2021). However, originally included additives are not removed during this kind of recycling either. Flooring treated with cryogenic grinding is converted into construction films (input AgPR 2021 email contact) which might be seen as a lower-level application as well. However, this concerns only a small amount of flexible flooring PVC that is treated in the German AgPR plant.
- Recycling of **cables** in closed loops is not possible due to the residual metal content of the recyclate (0.1-2%) and related safety issues. Instead, recyclate from cables is used for thick-walled products for road traffic safety (e.g. beacon bases or guard rail elements) (BiPRO 2002) or for floor coverings (Recyklo 2021).

5.5.2.6 Closed-loop recycling
Closed-loop recycling is considered a contributor to a Circular Economy. Closed-loop recycling means that the recycled material is used to produce the identical type of product it was originally recovered from. The new product can consist entirely of the recycled material or partially of the recycled material together with virgin material (Huysman et al. 2015) (Ragaert, Delva, and Van Geem 2017) (see Annex 3.1).

Recyclate from rigid PVC such as pellets, flakes or powder from mechanical recycling processes contributes to closed-loop recycling. However, it has to be considered that the extrusion process leading to high-quality recyclate is only efficient when the waste stream is relatively clean and pre-sorted. Moreover, for successful closed-loop recycling, the secondary material must show the same physical and mechanical properties as the virgin material. It has also to be considered that the originally included legacy additives remain in the extruded recyclate (Martens and Goldmann 2016) i.e. it might happen that the recyclate can be reused from a technical perspective but not from a regulatory perspective if the PVC contains restricted legacy substances.

When strictly following the definition from literature, to date, only PVC recylate from windows is recycled in a closed-loop. 80-90% of PVC window recyclate (mostly white pellets) is used for new window profiles. The rest, mostly non-white regrind is included in other rigid products (e.g. pipes) (Fråne et al. 2018). For windows the maximum average recyclate content accounts to 40-45% (input EPPA 2021). The window recycling industry also benefits from the homogeneous market situation in the area of plastic window profiles. In the past, plastic windows were exclusively made of PVC (Potrykus, Zotz, et al. 2020). In terms of quantities, it has to be kept in mind that the largest fraction of recycled PVC waste is window profiles i.e. a large amount of existing PVC recyclate is circulated in a closed-loop (Fråne et al. 2018).
Pipe recyclate is also reused in pipes but in pipes with lower quality requirements (i.e. called "downcycling" in some literature) (see Annex 3.1). The recyclate used in pipes consist for 3.8% of recycled pipes and for 96.2% of mixed recyclate of other products (mainly window profiles, cladding, roller shutters).

One temporary remedy for safe closure of PVC material cycles is complete monitoring and documentation of the pollutant content (Potrykus, Zotz, et al. 2020). However, the control and documentation of the pollutant content of recycled PVC would involve considerable effort. A potential method for monitoring is closed-loop recycling including specific labelling. Certification systems like EuCertPlast to monitor recyclers might help to improve a Circular Economy. However, these certifications are granted to various recyclers not only to those that perform close-loop recycling (e.g. PVC windows). For example, cable recyclers are certified as well, although recycled cables are not transferred into new cables but into thick-walled products. Industry is aware of these challenges and a selection of initiatives is presented below (see Table 5-13)

Table 5-13: Overview selected initiatives engaging in PVC recyclate

Initiative	Description of aims and current status
„Controlled Loop PVC Recycling von Kunststofffenstern"	The initiative „Controlled Loop PVC Recycling von Kunststofffenstern"aims to create a European standard that, in conjunction with the EN 12608 standard for PVC-U window profiles and the EN 14351-1 standard for windows and doors, ensures safe production and closed cycles (Potrykus, Zotz, et al. 2020).
Initiative from TEPPFA (no name identified)	TEPPFA initiated a project to test increased levels of recyclates in pipe systems and to demonstrate the performance of the final products (input TEPPFA 2021).
PolyREC®.	Petcore Europe®, PlasticsEurope®, Plastics Recyclers Europe®, and VinylPlus® have mutually agreed to join forces to form the organisation PolyREC®. PolyREC will monitor, verify, and report their plastics recycling and uptake data in Europe. This will be achieved by using a common data collection system – RecoTrace®. PolyREC will ensure traceability, transparency, and trust in recycled materials along the entire plastics value chain (Plastic Recyclers Europe 2021)

5.5.3 Summary of PVC recyclate in a Circular Economy

Overall, the analysis in this chapter indicates that PVC recyclate is currently playing a certain role in the circular economy. Recyclate is applied in various PVC products, most notably in the construction sector. However, many PVC end-of-life products are not redirected in their original product group but into other applications, sometimes of lower value. In general, rigid PVC (e.g. windows) is more likely to be recycled in a closed loop whereas products of flexible PVC (e.g. flooring, roofing, cables) undergo downcycling when they are recovered. Alternative uses of PVC recyclate are partly performed (e.g. inclusion of plastic in concrete), however they are not considered to be closed-loop. In general, PVC recyclate from building and construction waste seems to be most promising in terms of a Circular Economy:

- 32% of total PVC recyclate was used in windows which is considered being closed-loop recycling as the biggest share of end-of-life PVC is retrieved from windows. The average recyclate share in the new window market account to 5-10%.
- New pipes can contain between 20-100 % recyclate (depending on pipe type). Due to lacking amount of pipe recyclate, window recyclate and recyclate from other rigid end-of-life PVC is used in the pipe production. Recycling of pipes is therefore considered open-loop or controlled-loop recycling.
- Recyclate from PVC flooring can be encapsulated into new floorings, as it is done with pipe or window recyclate. However, collection and recycling quantities are still lacking.
- In general, as regards the rate of uptake of PVC recyclate in new products the analysis in this chapter indicates that, for various applications, the full uptake potential is not realised reflecting several technical and economic barriers With regard to the number of recycling cycles, it is estimated that on a molecular level PVC from pipes and windows can theoretically be recycled at least 7 times. However, the approach of recycling cycles is not favoured by the recycling industry but seen as a misleading academic discussion. They point out that the focus should rather lie on the lifetime of a product.

The presence of legacy additives in PVC waste poses a considerable challenge for the uptake of PVC recyclate in products. Limit values on concentrations of restricted additives in products limits the use of recyclate containing such additives. Furthermore, current levels of additives in PVC waste may clearly exceed these limit values. As indicated in chapter 5.4, technically and economically feasible options for the decontamination of PVC waste are currently limited. However, future developments in such technologies may provide further technical solutions for the challenge of legacy substances. In the absence of feasible decontamination technologies, a sustainable solution to the issue of legacy substances may only be achieved through a combination of rigorous assessment of additives used in virgin PVC and carefully balanced limit values for the concentration of legacy substances in PVC recyclate. In the absence of a careful assessment of characteristics of new additives, the regulatory solution of limit values for recyclate can be considered an imperfect treatment of problem symptoms.

In the pipe and window industry it is common practice to encapsulate the recyclate content with layers of virgin PVC to prevent possible human and environmental exposure to the additives in the recyclate but also to be flexible in terms of colours. Within heterogenous flooring layers a recyclate layer can also be encapsulated. However, to date this is mainly performed for recyclate from pre-consumer waste and off-cuts. The encapsulation is seen as a solution to increase recycling and resource efficiency. At the same time it keeps legacy additives in use. Moreover, it is uncertain if this process leads to mixing non-waste (virgin material) with hazardous waste (PVC waste containing legacy additives).

Based on the analysis, it can be said, that PVC recycling in its current form can only partly contribute to a Circular Economy. Although window recyclate is regularly reused, recyclate from other PVC waste streams are lacking. This is not only due to lacking quantity of PVC waste of certain products (e.g. pipes) but of challenges related to recycling technologies, legacy additives and material properties of some PVC products that only allow downcycling (e.g. for cables) or make recycling wholly unfeasible. In total, the amount of recyclate of the total PVC production in Europe was very low (1,3%).

5.6 Future prognosis - PVC Recycling in the EU in 2050

Chapter 3.5 displays a general prognosis on PVC waste volumes until 2050. This chapter aims to make the prognosis more detailed by assessing the development of the volumes of the different product groups of PVC waste (e.g. pipes, windows etc.). Moreover, the prognosis of recycled amounts of PVC waste are part of the following section. The prognosis is made for the time period 2021 to 2050. Figure 34 gives an overview on the methodological approach of the future prognosis. More details can be found in Annex 4.7.

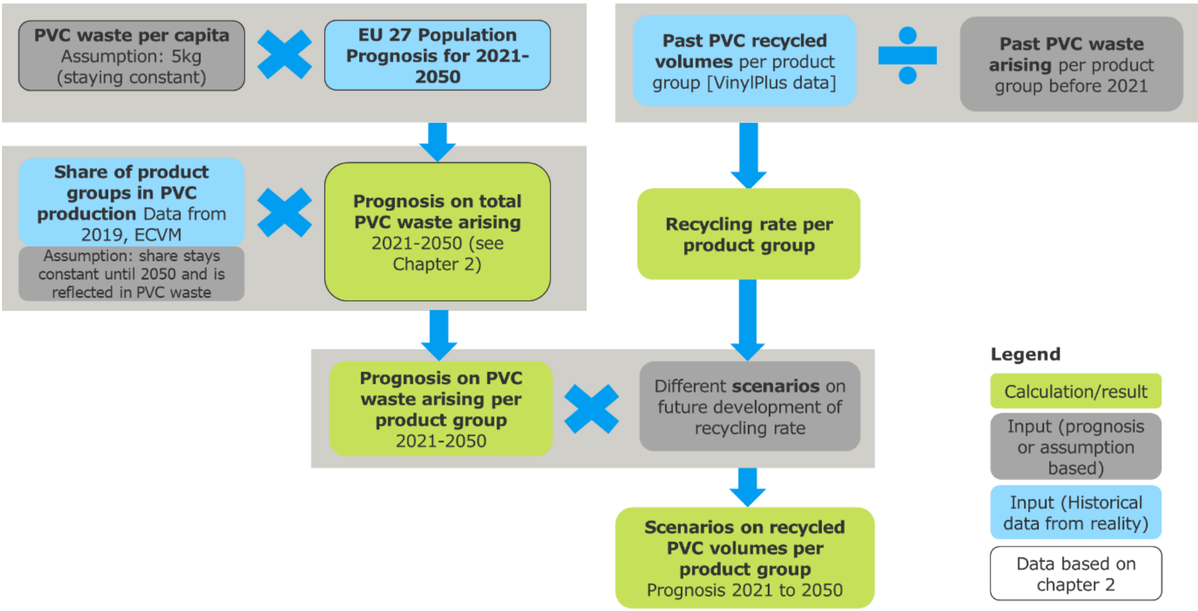


Figure 34. Methodical approach for the future prognosis on PVC waste arising and recycled PVC volumes per product group

It should be noted that any prognosis of PVC recycling is likely to be influenced by various additional regulatory, economic, technical and behavioural factors (see Figure 35). Such factors may influence, for example, the volume of PVC waste available for recycling, the quality of such waste volumes (e.g. in terms of contamination) and the competitiveness of recycled PVC in terms of price or final quality. As these factors are of a qualitative nature and in combination effects uncertain, it is difficult to include them in quantitative assessments. Annex 4.7 provides a more elaborate qualitative overview of potential factors. As such any forecasting exercise is subject to significant uncertainty and the forecasts presented here should be interpreted as potential trends rather than accurate predictions.

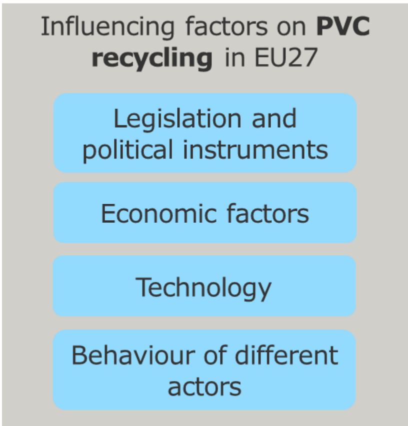


Figure 35. Influencing factors on PVC recycling in EU27. Detailed information can be found in Annex 4.7.

5.6.1 Prognosis on PVC waste arising per product group (2019-2050)

Data on past PVC waste arising per product group was not sufficiently available. As an approximation, the total PVC waste arising was distributed on different product groups by using the shares of each product group within PVC production (see Figure 36). As simplification, it is assumed that the distribution of product groups within PVC production is reflected in the waste arising. For example, with regard to rigid PVC, window profiles and pipes accounts to a share of 49% (27%+22%) in the PVC production.[79]

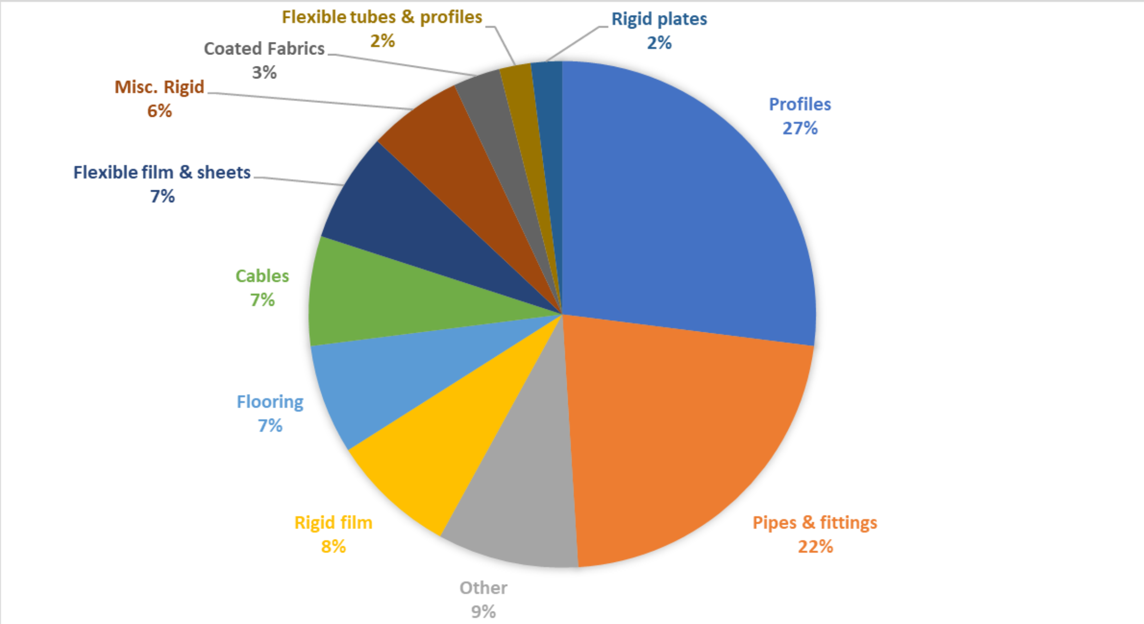


Figure 36. Shares of the product groups in PVC production resp. in waste generation (based on ECVM)

Figure 37 shows the future PVC arising per product group. As mentioned in Chapter 3.5 the PVC waste arising suggests rather modest reductions, which largely reflects projected population trends in the EU27. According to Figure 37 the PVC waste arising from the different product groups is not expected to decrease significantly. However, it has to be considered that external factors (e.g. significant production peaks in the last years which could lead to abrupt risings of waste) are not included in the calculation.

[79] The shares and the choice of the categorisation on product groups is retrieved from ECVM, PVC Applications, Accessed January 2021 (European Council of Vinyl Manufacturers 2017)

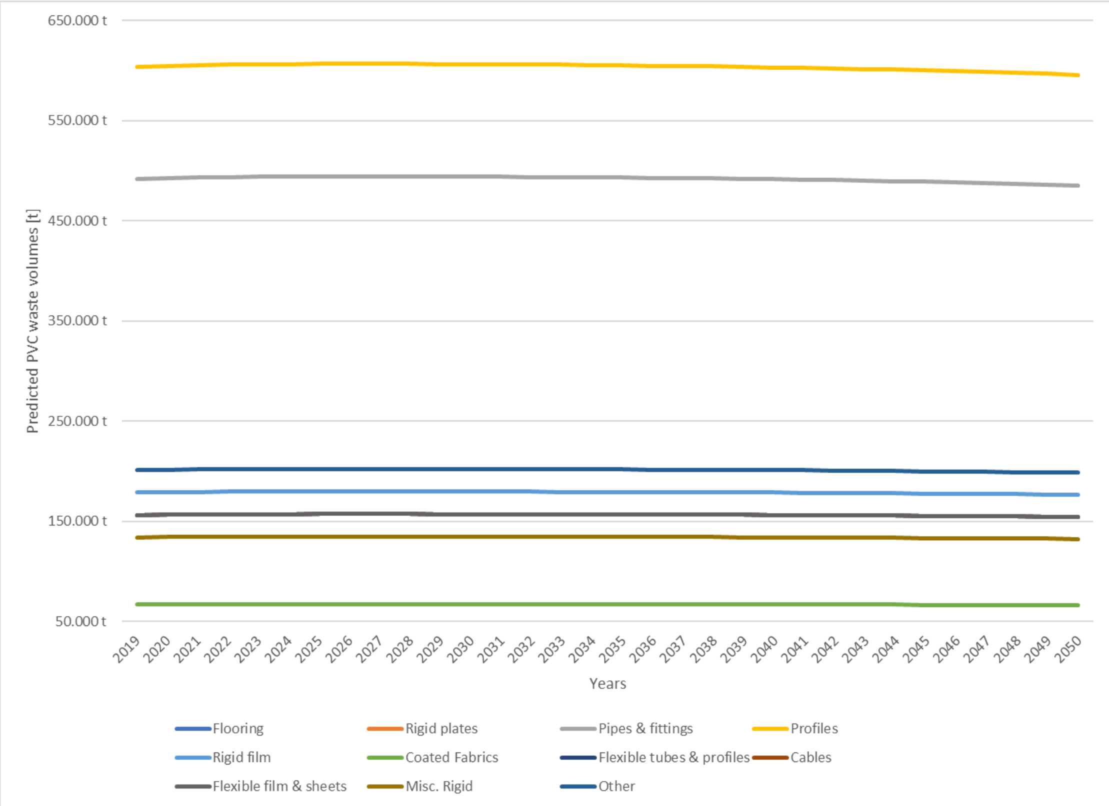


Figure 37. PVC arising per product group (2019-2050); data based on ECVM (shares per product group) and prediction on PVC waste arising (see chapter 3.5)

5.6.2 Future prognosis for recycling volumes per product group (2021-2050)

Based on data from VinylPlus on volumes of recycled PVC in the last years (see VinylPlus progress reports 2010-2021) recycling rates of different product groups could be approximated. The average PVC waste arising between 2010 and 2018 was estimated to be 2,425,540 tons (see Annex 4.7). Here as well, the share of the different product groups in past PVC waste arising was based on data from PVC (see Figure 36).

Based on the calculated recycling rates of the last years three scenarios were developed and displayed for the major product groups:

Scenario 1 - Constant recycling rates (median of 2017-2020)
Scenario 2 - Linear regression

In scenario 1 it is assumed that the beginning phase of PVC recycling with huge growth of recycling rates (up to 90% in 2012 for specific product groups) is over and that a certain stagnation is reached. Especially in the last years a certain constancy in the recycling rates could be observed. Due to this, scenario 1 displays a constant continuation of the respective current recycling rate which is based on the median of the recycling rates of the last four years. Scenario 2 builds on scenario 1. However, it is assumed that increasing/decreasing effects of the recycling rates of the last four years are continued in the future. Hence, the median of the growth rates (positive or negative values) of the recycling rates are included in the prognosis. The following should be noted:

- Each scenario includes data on total PVC arising. According to the modest decrease in total PVC arising (see Figure 37) the recycling volumes are likely to decrease in a similar manner. However, rising recycling rates can compensate the effect of decreasing waste volumes.

- The depiction of "Windows and profiles" includes additional information on the total amount of estimated waste volumes of PVC windows. This shows that the amounts of recycled volumes can only rise to a certain amount i.e. to the amount of available waste.
- None of the scenarios consider that past peaks in PVC production might lead to peaks in PVC waste arising in the coming years.
- It should be noted that every chart has a different scale (see y-axis) to ensure a better readability.

It is striking that for all product groups the recycling volumes are falling in 2019. This might be due to three aspects:

1) VinylPlus describes in its 2021 report that higher ambitions on recycling volumes (800.000 tons by 2020) were not reached due to the COVID-19 pandemic i.e. decreasing recycling volumes might also be a consequence from the pandemic (Vinylplus 2021).
2) The calculation of the recycling volumes is based on the total amount of PVC waste arising which was derived from chapter 3.5. Being based on the EU27 population it led to lower results (2,234,123 t in 2019) than other predictions for this year (e.g. 2,5-2,9 million tons t) i.e. this data set led to lower recycling volumes as well.
3) Another reason might be that the recycled volumes displayed from VinylPlus include data of EU28 and partly Norway and Switzerland. The scope of the data that was used to predict future waste arisings is based on the EU27.

Additionally, to the prognosis of this study, it has to be considered that VinylPlus claimed that they aim to recycle 900,000 tonnes of PVC per year into new products by 2025. By 2030 they committed themselves to recycle at least 1 million tonnes (VinylPlus 2021b). According to Figure 37 the PVC waste arising in 2030 is estimated at 2,245,608 tons. This would mean that recycling volumes of 1 million tonnes from VinylPlus would account to a recycling rate of 45%. With a current recycling rate of 30-32% the goal of VinylPlus would require a considerable increase in recycling rates. The images below present past (until 2020) and predicted (until 2050) volumes of recycled PVC per product group. The underlying data and calculations were presented in the previous sections.

Figure 38. Past (until 2020) and predicted (until 2050) volumes of recycled PVC per product group

5.6.3 Development of legacy additives and its amount in PVC waste

As mentioned in previous chapters, legacy additives are still a challenge for a Circular Economy as these additives remain in the recyclate when it is redirected into new products. However, several restrictions and voluntary phase-out commitments from industry lead to a decrease in additive use (e.g., lead, cadmium) in certain PVC product groups. Hence, it is of interest how long certain additives remain in the waste volumes and which waste streams will be free of certain substances soon. In general, it can be assumed that after a certain point in time waste will contain less hazardous substances when a phase-out or decrease of such substances is realised (e.g. via regulatory restrictions).

For windows, for example, EPPA indicates that 10-15 years from 2019 the peak of **cadmium** concentrations will be reached in recycled PVC (Ramboll Environment and Health 2020). The VITO report indicates that by 2020 the amount of cadmium containing waste from PVC window profiles will still be rising. However, after 2020 a decline in generated cadmium containing waste from PVC window profiles is expected. This is relevant, as waste from PVC window profiles seems to be the largest rigid PVC waste stream which may contain cadmium (Ramboll Environment and Health 2020). Table 5-14 demonstrates that the share of PVC waste from window profiles that contain cadmium will decrease to 2050 (from 40% to 2,7%).

Due to an industry commitment, since 2016 lead is not used anymore in PVC products. The share of lead containing products (e.g. pipes, windows) was decreasing from 2001. Lead stabilizer was mainly replaced by Calcium-Zinc stabilizers (Blaeij et al. 2019). With an average window lifetime of 30-40 years, it is assumed that concentrations of lead in PVC waste windows will decrease after 2035 (input VinylPlus 2021). Pipes, having a lifetime >100 years, will also generate waste containing lead at the end of this century. In 2000, 100% of newly produced pipes contained lead. However, it is unclear if the amount was as high before 2000. In flooring lead was already phased out before 2000 and in roofing products after 2011 (Tauw bv 2013). In 2007, no newly produced cables contained lead

Table 5-14. Share of post-consumer PVC waste containing cadmium/lead/DEHP in relation to waste amount without these legacy additives [calculations based on (Swedish Chemicals Agency 2020) and (Tauw bv 2013)**].**

Type	Share of cadmium containing PVC waste - 2020	Share of cadmium containing PVC waste - 2050	Share of lead containing PVC waste - 2021	Share of lead containing PVC waste - 2050	Share of DEHP containing PVC waste
Window profiles	40%	2,7 %	Estimated share in 2021: 12-42%[80]	Estimated share in 2050: 0-24%[81]	Do not include DEHP
Pipes	0%	0% (Assumption based on 2020 data)	In 2100 the share of pipes containing lead can still be 100%		Do not include DEHP
Flooring	0,3%	Not determined	Lead was only used in specific flooring. Lead in flooring disappeared before 2000	0%	Authorisation under REACH needed for recycling
Roofing	26%	Not determined	Lifetime 20 years[82]: 18% Lifetime 15 years: 8%	0%	Authorisation under REACH needed for recycling
Cables	58%	Not determined	Lifetime 20 years[83]: 85% Lifetime 15 years: 6%	0%	Authorisation under REACH needed for recycling

[80] This assumption is based on the Tauw report: With an avergage lifetime of 30-40 years windows from 1981-1991 arise as waste in 2021. In 1981 the share of lead containing PVC windows was 12% and in 1991 it accounted to 42% (Tauw bv 2013).

[81] This assumption is based on the Tauw report: With an avergage lifetime of 30-40 years windows from 2010-2020 arise as waste in 2050. In 2010 the share of lead containing PVC windows was 24% and in 2020 it should be 0% due to the industry phase out (Tauw bv 2013).

[82] As the lifetime of roofing could not be clearly identified, differnt shares for differenet lifetimes are displayed.

[83] As the lifetime of cables could not be clearly identified, differnt shares for differenet lifetimes are displayed.

5.6.4 Summary of future prognosis

According to the estimated future prognosis the waste volumes of PVC will not rise significantly in the next years. However, it has to be considered that this calculation is based on the future development of EU population and the average PVC waste amounts and is extremely uncertain. Several other further factors like the prognosed renovation wave and past production peaks will influence the PVC waste amount as well.

With regard to future volumes of recycled PVC, one scenario assumes that the recycling volumes will rise in the next years. This requires that the growth of the recycled volumes of the last years continues. In the other scenario the recycling volumes will stagnate due to the assumption that the recycling rate stagnates. It is assumed that factors like derogations of specific substances in recycled material (e.g. lead), technical developments (e.g. selective dissolution) or specific collection targets have significant influence on the volumes of recycled PVC.

Important considerations on the prognosis
Based on estimates of the current average PVC waste consumption and the declining EU population, PVC waste may decline in the future. However, it should be noted that external factors like the EU renovation wave and historical production peaks might lead to increasing PVC waste volumes. Improved collection and further technologies for the treatment of composite products and contaminated PVC are precondition for rising recycling volumes from post-consumer PVC. Currently, these technologies are immature.

6. OTHER RECOVERY, DISPOSAL AND SHIPMENTS OF PVC

Key Messages – Other recovery, disposal and shipments of PVC

This chapter provides a quantitative analysis of the main waste treatment options of PVC in the EU with a specific focus on forms of recovery other than recycling, as well as disposal options.. Moreover, it elaborates on the environmental, human health and economic impacts associated with different waste treatment options and discusses questions of transboundary movements of PVC waste.

Relevant PVC waste streams and general PVC waste management

- The most relevant streams of PVC waste are Construction and Demolition waste (C&D waste), packaging and household waste, Waste of Electrical and Electronic Equipment (WEEE) and End-of-life vehicles (ELVs). Further waste streams containing large quantities of PVC waste are medical products, houseware, furniture, gardening and agriculture as well as leisure products.
- Based on the results of the quantitative scoping and their comparison with existing studies, it can be concluded that
  - between 1,719,334 and 2,435,000 tonnes PVC waste are produced per year in the EU;
  - between 73.1 – 77.7 % are directed towards recovery (including preparation for reuse, recycling and energy recovery) and between 22.3 – 26.9 % are directed towards disposal (including landfilling and incineration without energy recovery);
  - the majority of PVC waste is incinerated (as energy recovery) (41.9 - 53.1 %); and
  - the recycling rate ranges between 24.6 and 31.2 %.
- The quantitative scoping of PVC revealed that several data gaps exist concerning the fate of PVC present in the analysed waste streams and that the quantitative scoping is associated with more or less large uncertainties. Therefore, specifying the quantities of PVC being treated in different incineration facilities or disposed of in different types of landfills is only possible to a limited extent.
- One general remark is that doubts remain about the correct classification of PVC waste as hazardous or not according to the EU List of Waste. Improper classification may lead to PVC waste being accepted and treated in facilities which do not have the necessary permit.

Information regarding environmental, economic and human health impacts associated with the incineration of PVC waste in different facilities

- Relevant incineration facilities for the treatment of PVC waste include i) municipal solid waste (MSW) incineration plants with and without energy recovery, ii) co-incineration plants (such as cement-kilns or blast furnaces), iii) hazardous waste incineration facilities and iv) medical waste incinerators including decentralized and centralized medical waste incinerators.
- In terms of the amount of PVC treated, MSW incinerators are the most relevant treatment option, while the co-incineration of PVC in cement kilns is considerably lower.
- While state-of-the-art MSW incinerators are able to meet current emission level standards, the input of PVC increases the risks of dioxin and furan emissions during transient phases and if optimal operation conditions are not met. In addition, environmental concerns remain concerning dioxin and furan emissions from the incineration of PVC in small on-site hospital incinerators.
- Regarding management of solid incineration residues, concerns exist in particular (but not limited to) for the leaching of soluble salts, heavy metals or dioxins from solid incineration residues which are landfilled or processed otherwise (e.g. used as raw material).
- From an economic point of view, the incineration of PVC is – depending on the type of facility – associated with hidden costs stemming from the production of neutralisation agents, the disposal of solid residues and the increased risks of corrosion.

Information regarding environmental, human health and economic impacts associated with the disposal of PVC waste by landfilling:

- If PVC is disposed of by landfilling, this is mainly done at landfills for non-hazardous waste while smaller quantities are directed towards landfills for hazardous waste and landfills for inert waste.
- Significant differences exist between EU Member States concerning the amounts of PVC being disposed of at landfills.
- Environmental and human health impacts of landfilled PVC mainly relate to the leaching of phthalates from flexible PVC, while only limited amounts of stabilisers are released from rigid PVC. It can however not be concluded whether other additives might or might not migrate from the PVC matrix.
- Further environmental concerns exist regarding the emissions of dioxins from accidental and intentionally set landfill fires.

Information regarding environmental, human health and economic impacts associated with illegal disposal and incineration

- While illegal waste treatment (especially illegal disposal and incineration under uncontrolled conditions) remains a problem in EU Member States and countries outside the EU, no specific quantitative data on illegal treatment of PVC waste could be obtained.

- An increase in illegal treatment since 2018 with regard to plastic waste has been noted. The reason behind illegal treatment is mostly financial, presumably one motivation being plastic import restrictions imposed by China.
- Illegal fires are often reported as unintentional, but there is reason to assume that fires were started to eliminate waste. Landfill fires have become a widely reported phenomenon, including in EU Member States or accession countries.
- The high chlorine content in PVC can lead to the formation of dioxin in uncontrolled fires. Since PVC lowers the incineration temperature the residence time is increased leading to a further increase of dioxin emissions. In addition, wet and compacted waste containing PVC promotes dioxin formation.
- This is especially relevant for landfill fires or backyard fires. However, the share of PVC within the incinerated waste is relevant, which can differ from landfill to landfill and is generally understood to be rather low in household waste.
- Measurements of house fires show also high dioxin emissions, and these can be assigned to PVC to a limited extent.
- Cable burning is not practiced in the EU anymore but is still a problem in third countries. PVC-based insulated wires lead to high dioxin emissions, in particular.
- Illegal landfilling can lead to serious problems for the environment and human health. The related health risk depends on the composition of disposed waste and conditions of the area. The extent to which PVC augments negative environmental and health effects cannot be clearly demonstrated due to a lack of data.

Information regarding environmental, human health and economic impacts associated with transboundary movements of PVC waste:

- Statistical data suggest that 20,000 tonnes of PVC post-consumer waste have been exported from EU Member States to countries outside the EU in 2020.
- Between EU Member States, PVC waste is subject to significant transboundary trade; for example smaller countries without recycling facilities for PVC waste have to export the waste to other EU Member States with appropriate recycling infrastructure
- In 2016 the biggest exporters of PVC waste were Germany (and the UK); in 2021 importing countries outside the EU for PVC waste are mainly Pakistan, United States, Turkey and Malaysia.
- With regards to illegal transboundary movements, this may occur via; the misclassification of waste as non-waste (such as cables) or a misclassification into a wrong entry of the Annexes of the EU WSR. Another illegal treatment action is the mixing of hazardous PVC waste with other materials or non-hazardous waste, in violation of EU law
- Illegal transboundary movements of PVC waste may lead to PVC waste inappropriately treated in substandard facilities, or ultimately to wild dumping and burning of PVC waste outside the EU, with effects for health and the environment as described above.
- In 2020 the EU tightened rules for shipments of plastic waste. In November 2021 the European Commission published a proposal for a comprehensive revision of the Waste Shipment Regulation. As such the regulatory and market situation is evolving. Concerns exist whether new restrictions may lead to more intensive illicit activities.

6.1 Aim and structure of this chapter

Besides the mechanical and chemical recycling operations described in chapters 5.1 and 5.2, the waste management options for PVC waste are incineration (either energy recovery or disposal) and landfilling (Ciacci, Passarini, and Vassura 2017; European Commission 2000). In addition, PVC waste can be exported out of the EU for recycling or treatment in third countries. Furthermore, it cannot be ruled out that amounts of PVC waste are treated in an unsound manner in contravention to EU and member State law. The aim of this chapter is to describe the state of these PVC treatment options in the EU and to give an overview of the associated environmental, human health and economic impacts. In addition, the chapter will provide an overview of available information on exports of PVC waste, as well as potential illegal treatment routes:

- In a first step, chapter 6.2 provides a general overview on the existing waste management options of PVC waste. In chapter 6.3, disposal routes are shortly described with regard to their relevance for treating PVC-containing waste;
- The second step includes the qualitative and quantitative scoping of major waste management options (chapter 6.3). Based on this analysis, in a third step, relevant disposal (including incineration without energy recovery and landfilling) and recovery options (other than recycling) are assessed with respect to relevant environmental,

economic and human health impacts (chapter 6.4 for incineration, chapter 6.5 for landfilling);
- Chapter 6.6 contains a description and scoping of illegal disposal activities (illegal dumping and illegal incineration); and
- Chapter 6.7 discusses transboundary movements of PVC waste.

6.2 General description of the main waste management routes (besides recycling) and their relevance for PVC waste

6.2.1 Incineration – introduction and types of installations

For describing options of PVC waste regarding incineration, a clear terminology of different types of incineration processes and incineration facilities is required. Relevant terms and their meaning in the context of this report are described in Annex 3.1. Focus GroupFocus GroupFocus Group

Note concerning the incineration of PVC
The mono-incineration of separately collected PVC waste does only play a subordinate role in the EU (Focus Group Disposal 2021). Incineration facilities do not tend to accept mono streams of PVC waste given the potential for corrosion in the flue gas cleaning system. Consequently, PVC waste is accepted at incineration facilities mainly as part of mixed waste streams – where the concentration of PVC is estimated to be below one percent (ECVM input Focus Group Disposal 2021). It is also possible that separately collected PVC waste is mixed at the facility with other waste types prior incineration (Fråne et al. 2018) (Focus Group Disposal 2021).

The following facility types are relevant for the treatment of PVC waste by incineration (Bernard, Hjelmar, and Jürgen 2000; European Commission 2000) (Focus Group Disposal 2021):

- Municipal Solid Waste (MSW) incineration plants with energy recovery;
- MSW incineration plants without energy recovery;
- Medical waste incineration plants;
- Hazardous waste incineration plants;
- Cement kilns; and
- Blast furnaces.

The above-mentioned incineration facilities are described in Table 6-1 with respect to their relevance for treating PVC-containing waste. Detailed descriptions of the facilities as well as facility-specific limitations for the treatment of PVC can be found in Annex 5.1.

Table 6-1: Incineration facilities and their relevance for treating PVC containing waste

Incineration facility	Relevance for treating PVC waste	Reference
MSW incineration plants with energy recovery	MSW incineration plants represent the most relevant incineration facilities for the treatment of PVC waste. Although the treatment of waste containing elevated levels of PVC is limited due to the potential for corrosion, a substantial share of the overall volume of PVC waste generated in the EU is treated in MSW incineration plants (almost 50 %; see section 6.3.1.7).	(Conversio Market & Strategy GmbH 2021; Fråne et al. 2018; Tukker et al. 1999)(EURiTS Focus Group Disposal 2021).

Incineration facility	Relevance for treating PVC waste	Reference
MSW incineration plants without energy recovery	Only of minor importance in Europe for treating MSW and PVC[84]	(CEWEP Focus Group Disposal 2021)
Co-incineration facilities – cement kilns and blast furnaces	Input of PVC as part of waste treated in co-incineration facilities is limited due to the quality requirements of the cement produced and the potential for corrosion.	(Bilitewski, Wagner, and Reichenbach 2018; Carpenter 2010; Ecofys 2016; IFC 2017).
Hazardous waste incinerators	Hazardous waste incineration plants are relevant for treating PVC waste classified as hazardous and for homogenous streams of PVC waste due to the high chlorine concentrations. In general, it should be noted that incineration is the main treatment method for hazardous PVC waste[85] (ECVM input Focus Group Disposal 2021)	(Fråne et al. 2018; Janssen et al. 2016; Trozzi 2019)
Medical waste incinerators (small, decentralized on-site incineration facilities located at the hospitals)	Medical waste containing PVC can incinerated in small, decentralized on-site incineration facilities located at the hospitals or in dedicated centralised medical waste incineration facilities. In general, a shift towards larger, centralised incineration facilities can be observed.	(Trozzi 2019) (EURiTS Focus Group Disposal 2021); (Bernard, Hjelmar, and Jürgen 2000; Trozzi 2019) (EURiTS Focus Group Disposal 2021)
Medical waste incinerators (dedicated centralised medical waste incineration facilities)		

The burning of municipal solid waste containing PVC outside facilities with a respective permit is an illegal disposal operation yet this is still a common practice in some European countries (Hoffer et al. 2020), see further details in chapter 6.5.

6.2.2 Landfilling – introduction and types of installations

In contrast to thermal degradation methods or recycling options, landfilling does not contribute to energy or material recovery or a reduction of the volume of solid PVC waste. Taking into consideration the priority order of the waste hierarchy, landfilling as disposal generally represents the least desirable option. Consequently, the EU has introduced targets for the reduction of landfilling. In general, Article 4 of the Landfill Directive 1999/31/EC defines three different categories of landfills:

- Landfills for hazardous waste;
- Landfills for non-hazardous waste; and
- Landfills for inert waste.

According to Article 5(3) of the Landfill Directive, waste that has been separately collected for preparing for re-use and recycling must not be accepted at landfills.[86] Concerning the disposal of PVC by landfilling, all three landfill categories can be relevant. PVC waste must only be accepted at landfills of the different classes when meeting the respective criteria as set out by the Council Decision 2003/33/EC[87]. In particular, the following parameters are relevant:

[84] CEWEP's conclusion does seem plausible when comparing the shares of the waste stream "Total waste" (hazardous and non-hazardous) directed towards incineration without energy recovery (Disposal – incineration (D10)) and towards incineration with energy recovery (Recovery – energy recovery (R1)) according to Eurostat data for the year 2018 (EU27). Following this comparison, that only around 0.7 % of the waste is treated in incineration facilities without energy recovery, whereas 6.0 % are treated in incineration facilities with energy recovery.

[85] Information on when PVC waste is considered hazardous is provided in chapter 6.4.4.

[86] With the exception of waste resulting from subsequent treatment operations of the separately collected waste for which landfilling delivers the best environmental outcome in accordance with the waste hierarchy as per Article 4 of the Waste Framework Directive

[87] https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32003D0033&from=en

- total organic content (TOC) leaching limit value of 3 % for waste acceptable at inert landfills
- TOC leaching limit value of 5 % for stabilised hazardous waste acceptable at landfills for non-hazardous waste: TOC of 5 %
- TOC leaching limit value of 6 % for waste acceptable at hazardous waste landfills.

Where PVC waste is landfilled, the majority is disposed of at landfills for non-hazardous waste, while smaller quantities are directed towards landfills for inert and hazardous waste (Focus Group Disposal 2021)[88]. Moreover, landfills for hazardous and non-hazardous waste are relevant for the disposal of solid incineration residues, namely bottom ash, fly ash and air pollution control (APC) residues (Blasenbauer et al. 2020; Vehlow 2012).

Considerations regarding the landfilling of PVC as non-hazardous waste
As indicated in section 4.2, a highly relevant question for the selection of legal and suitable waste treatment options for PVC waste concerns the classification of such waste as hazardous or non-hazardous. The analysis in section 4.2 indicates that, in the case of PVC, such classification will likely depend on the extent to which migration and bioavailability can or cannot be established for relevant additives contained in a relevant PVC waste stream. Chapter 2 notes migration of additives from plastics can take place, since such additives are not covalently bound in the plastic matrix. However, data on migration and bioavailability of PVC additives is limited. Further assessment of the classification of PVC as hazardous or non-hazardous may be necessary for the determination of the most suitable disposal route. This seems especially relevant, considering the currently common practice of disposal of PVC waste in landfills for non-hazardous waste in the EU.

Bans or restrictions for the landfilling of PVC exist for instance in Germany, Finland and Sweden (Miliute-Plepiene, Fråne, and Almasi 2021).[89]

6.3 Qualitative and quantitative scoping of waste treatment options – general overview

A primary objective of this section is the identification and quantitative scoping of treatment options of major PVC waste streams in the EU and the assessment of associated environmental, economic and human health impacts. Two aspects need to be taken into account:

- the application of PVC and its fate within relevant waste streams; and
- relevant legislation affecting the waste treatment options of PVC waste.

A main challenge is that only limited data is available concerning the quantities of PVC waste directed to the different waste treatment options. These data gaps were identified during the desktop research and were confirmed by different stakeholders during the Focus Group Disposal (2021). The same challenge is described within other studies focusing on the issue (Miliute-Plepiene, Fråne, and Almasi 2021).
Eurostat contains data on the treatment of specific waste streams which contain considerable amounts of PVC. On this basis, a qualitative and quantitative analysis of the fate of PVC present in these waste streams has been performed. Based on the assumptions made, for most of the waste streams analysed, the data allow the estimation of shares of PVC directed towards disposal (including landfilling and incineration without energy recovery) and towards recovery (including preparation for reuse, recycling and energy recovery).

[88] It should be noted that this information is based on expert input received during the FocusGroup disposal 2021 while information in scientific publications on the relevance of different types of landfills for the disposal of PVC-containing waste could not be identified.
[89] In Sweden and Finland some exceptions might exist (Miliute-Plepiene, Fråne, and Almasi 2021).

As outlined in chapter 3.3.2, PVC is primarily used in the construction, automotive and electronics sectors as well as in packaging and consumer uses. Relatively low volumes of PVC are used in agricultural and healthcare products. Medical waste reveals a relatively high share of PVC. Accordingly, PVC makes up approximately 5 to 15% of medical waste (Johnsen 2015b). Based on the consumptions per industry sector (see Table 3-15), it is therefore assumed that the most relevant waste streams for PVC are:

- PVC in End-of-life vehicles (ELVs);
- PVC in Construction and Demolition Waste;
- PVC in Waste of Electrical and Electronic Equipment (WEEE);
- PVC in Packaging and household waste; and
- PVC in Medical waste.[90]

As such, these waste streams were taken as a basis for the subsequent quantification of waste treatment options.

6.3.1 Qualitative and quantitative analysis of the fate of PVC present in the waste streams

The overall quantity of PVC waste generated should be equivalent to the total sum of the quantities of PVC recycled plus the amount of PVC that is treated by means of incineration (including incineration with and without energy recovery) or disposed of at landfills (Fråne et al. 2018). However, one needs to take into account at least:

- The quantities of PVC-waste being exported to and imported from non-EU Countries; and
- The quantities of PVC-waste that are subject to littering and illegal disposal (e.g. dumping, backyard burning,).

Where corresponding information or data was available, these aspects have been taken into account for the quantitative scoping of waste treatment options. It should be emphasised that, due to lack of data, assumptions have been made on various occasions. While assumptions are always explained in detail in the text, the approach results in uncertainties in the quantitative scoping. For instance, it was difficult to apply a correction factor for PVC littered, exported and illegally treated as relevant data was missing. Another limiting factor is the lack of information on the quantities or shares of PVC being directed towards disposal or recovery. In the absence of such information, the shares indicated for the overall waste stream were considered representative for PVC. Due to the assumptions made for the mass flows and the associated uncertainties, the information and data indicated in the mass flows within the following chapters need to be interpreted with caution. For their interpretation, it is recommended to take into account the methodology applied for the creation of the mass flows as outlined in **Annex 5.2.**
In general, the following steps were carried out for each of the waste streams for the quantitative scoping:

1) Identification of relevant waste streams, fractions of waste streams or applications containing PVC (in Eurostat);
2) Development of a mass flow indicating the shares directed towards disposal or recovery;
3) Identifying and/or calculating the shares of PVC present in the waste streams, fractions of waste streams or applications;
4) Performing the quantitative scoping using either the amount of PVC in the current waste stream (as indicated by Eurostat data[91]) or the consumption volume (as indicated in section 3.3 or in other literature sources); and
5) Qualitative description of relevant treatment facilities (different types of landfills and incineration facilities).

[90] For medical waste, no specific consumption volume is indicated in chapter 3.

[91] Latest data on Eurostat is available for the year 2018.

Furthermore, the results of the mass flow analysis were compared to data published in a recent study of Conversio Market & Strategy GmbH (2021). This study investigated the quantities of PVC post-consumer waste in different product categories which are mechanically recycled, energetically recovered and disposed based on the year 2020[92]. The product categories included i) packaging, ii) building and construction, iii) automotive, iv) electro and electronics and v) others (including furniture, houseware, medical products, gardening and agriculture and leisure products). Hence, their results were used for a critical evaluation and comparison with the results of the present study.

6.3.1.1 ELVs

<u>Qualitative considerations and quantitative scoping</u>
For assessing the share of PVC from ELVs directed to disposal or recovery, it is necessary to understand the recycling process of ELVs. This involves the following consecutive steps (see Figure 36):

Figure 37. Treatment route of ELVs (based on Potrykus et al. (2020))

For the quantitative scoping it was assumed that PVC present in the plastic fractions of ELVs is not subject to dismantling and end up primarily in the following fractions[93]:

- Shredder Light Fraction (SLF)[94];
- Shredder Heavy Fraction (SHF) representing the non-ferrous materials from shredding[95]; and
- Other materials arising from shredding.

According to information in Potrykus et al. (2020) and Miliute-Plepiene et al. (2021), typical plastic materials used in automotive largely remain in the SLF after shredding. Car parts containing PVC are made of flexible as well as rigid PVC (AGPU 2017; Omnexus 2021b). According to Nakamura et al. (2009), the waste stream of ELVs contains larger amount of flexible PVC compared to rigid PVC.

In a first step, Eurostat data for the total shredding material in the EEA (2018) was used to calculate the shares directed towards ferrous scrap (steel) from shredding and other fractions including the SLF, the SHF and other materials arising from shredding (see mass flow in Annex 5.2).
Based on the assumptions described in Annex 5.2, the quantitative scoping of waste treatment options for PVC in ELVs was performed (see Figure 37). Due to the relatively long average lifetime

[92] Their methodological approach included amongst others the analysis of general sources, secondary statistics and interviews with different types of stakeholders for cross-checking purposes.

[93] Residues originating from shredding processes present a very heterogenous mixture of different materials (Hyks et al. 2014). As regards shredder residues from ELVs, generally, the following categories apply for plastic residues: Shredder Light Fraction (SLF), Shredder Heavy Fraction (SHF) and other materials from shredding (Potrykus, Aigner, et al. 2020).

[94] The SLF is also referred to as "fluff" and typically comprises light metal, paper, cardboard, different types of plastic, foam textiles, wood and wires (Hyks et al. 2014).

[95] The SHF contains non-ferrous materials such as aluminium, copper, zinc and lead as well as high density plastic and glass (Hyks et al. 2014; Potrykus, Aigner, et al. 2020).

of ELVs, the quantitative scoping was carried out based on the current amount of waste in the ELV waste stream[96] (see mass flow in Annex 5.2).

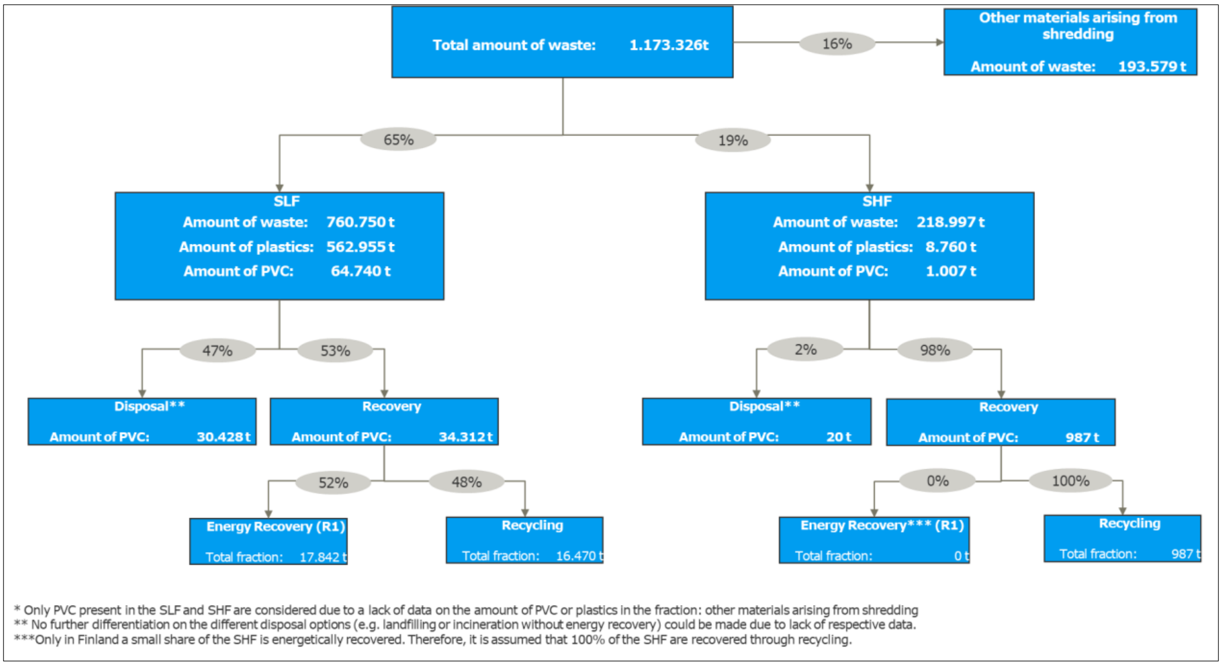


Figure 38. Quantitative scoping of waste treatment options for PVC present in the SLF and SHF of ELVs (based on data from Eurostat for the year 2018 (EU 27))

Of the 64,740 tonnes PVC found in the SLF, 47 % are sent to disposal. In this regard, data does not indicate whether this refers to disposal in landfills or incineration without energy recovery. As incineration without energy recovery in the EU is marginal (input Focus Group Disposal 2021), disposal is assumed to mean landfilling and other forms of disposal (D1 – D7; D12) (input Focus Group Disposal 2021). Furthermore, due to their heterogenous and complex composition, shredder residues from ELVs are frequently disposed of in landfills (Cossu and Lai 2015). According to the European List of Waste shredder residues are allocated to mirror entries (19 10 03* / 19 10 04 or 19 12 11* or 19 12 12). Thus, whether they are considered hazardous or as non-hazardous waste depends on the waste at hand. Traditionally, ELV shredder residues have been disposed of in landfills together with non-hazardous waste (Cossu and Lai 2015). However, as shredder residues from ELVs can also be considered hazardous they are also disposed of in landfills for hazardous waste (Fråne et al. 2018; Hyks et al. 2014). The remaining 53 % are directed to recovery. Thereof, 48 % are recycled, while 52 % are energetically recovered (R1). It can be assumed that this relatively high recycling rate is an overestimation which can be attributed to the recycling of metals present in the SLF (Evangelopoulos et al. 2018).
In summary, energy recovery and landfilling are the most relevant treatment options for PVC present in the SLF of ELVs. Relevant facilities for the treatment of PVC waste from ELVs are outlined in Annex 5.2.

The results of the quantitative scoping suggest that the treatment of PVC in ELVs depends on the treatment of the SLF[97]. This is consistent with the descriptions of PVC in ELVs by Fråne et al. (2018). Due to the different applications of PVC in cars, the waste stream of ELVs contains both

[96] The average lifetime of ELVs is 13 – 20 years depending on the respective Member State (Potrykus, Aigner, et al. 2020). For instance, in Germany an average lifetime of 17-18 years is assumed (BMU & UBA 2019). Therefore, performing the quantitative scoping based on the current consumption of the automotive sector was considered as not being reasonable. Data for the EU27 referring to the year 2018 was used.
[97] No information on the fate of PVC present in the fraction "other materials from shredding" is available.

types of PVC (flexible and rigid), while data of Nakamura et al. (2009) suggests, that the fraction of flexible PVC is higher compared to rigid PVC. This is also in line with the data provided by Conversio Market & Strategy GmbH (2021) for PVC post-consumer waste in automotive. Accordingly, flexible PVC accounts for 71% of PVC in the ELV waste stream.

Discussion

To verify the results, the outcome of the scoping was compared to the results of the study carried out by Conversio Market & Strategy GmbH (2021) for PVC post-consumer waste generation. Table 6-2 shows the annual quantity of PVC present in the waste stream of ELVs and the shares directed towards different treatment options[98].

Table 6-2: Comparing the results of the mass flow analysis for ELVs with the data from Conversio Market & Strategy GmbH

Treatment option	Data from Conversio Market & Strategy GmbH		Results of the mass flow analysis	
	Quantity of PVC [tonnes]	Share of PVC directed towards the treatment option	Quantity of PVC based on [tonnes]	Share of PVC directed towards the treatment option
Total PVC waste	92,000	-	65,747	-
Total recovery	64,000	69.6%	35,299	53.7%
Mechanical recycling/Recycling	8,000	8.7%	17,457	26.6%
Energy recovery	56,000	60.9%	17,842	27.1%
Disposal	28,000	30.4%	35,299	46.3%

As Table 6-2 shows, the overall amount of PVC in the waste stream of ELVs within the Conversio Market & Strategy GmbH data is about one third higher than in the mass flow analysis. As the mass flow analysis focused on PVC present in the SLF and SHF, this difference could be explained through the amount of dismantled PVC prior to shredding and PVC which might be present in the fraction "other materials arising from shredding". Thus, it can be assumed that the total amount of PVC in the waste stream of ELVs ranges between 65,000 and 92,000 tonnes per year. Considerable differences exist concerning the relevance of the different treatment options including (mechanical) recycling, energy recovery and disposal. The high recycling rate indicated for the mass flow analysis therefore probably presents an overestimation due to recycling of metals present in the SLF.

6.3.1.2 Construction and Demolition Waste

Qualitative considerations and quantitative scoping

In Eurostat, data is available for mineral waste from Construction and Demolition (C&D). As this category does not contain plastics or PVC, it cannot be used for the quantitative scoping. Plastics from the C&D sector (LOW code 17 02 03) are included in the stream "Plastic waste". However, no information on the share of plastics from C&D in the plastic waste streams is available. Moreover, PVC used in the C&D sector has a comparably long lifetime with average lifetimes for floorings, window frames and pipes ranging from approximately 25 years up to 100 years (Sadat-Shojai and Bakhshandeh 2011)(Focus Group Recycling 2021). This further hinders the development of a mass flow. Consequently, due to a lack of corresponding data and information, the quantitative scoping of relevant waste treatment options for PVC in C&D waste could not be

[98] Please note that the shares indicated in the table for the mass flow analysis are calculated in relation to the total waste and therefore differ from the shares indicated in the mass flows. The quantities and shares indicated for disposal include landfilling and incineration without energy recovery.

performed. A mass flow could only be compiled based on the data provided by Conversio Market & Strategy GmbH (2021) (see Figure 39).

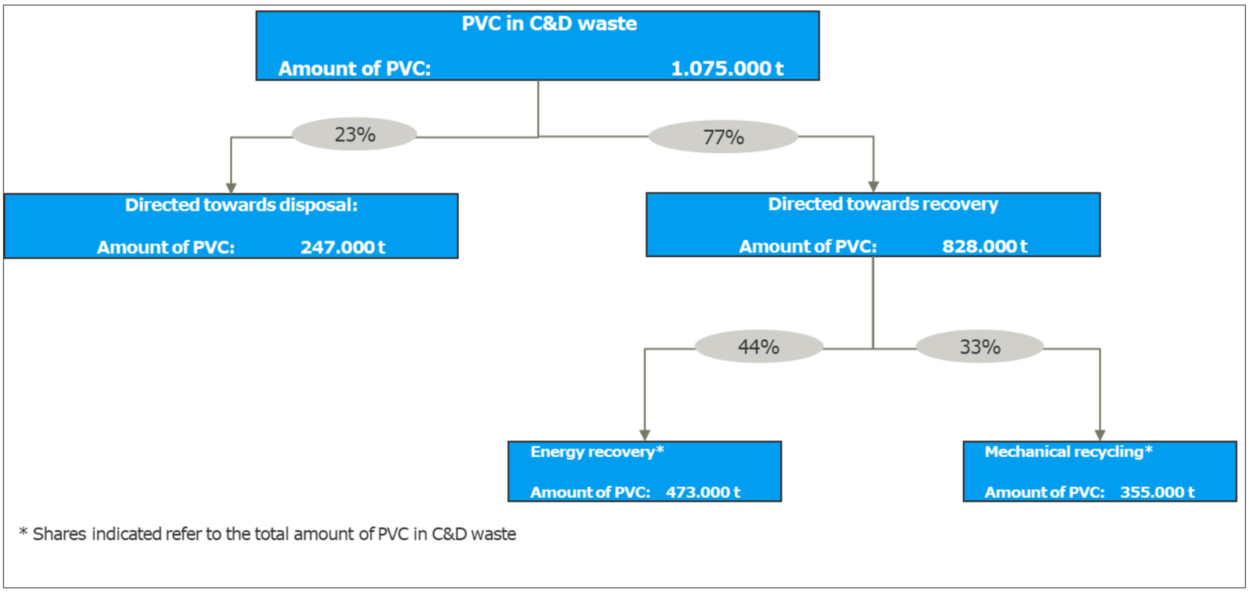


Figure 39: Quantitative scoping of waste treatment options for PVC present C&D waste (based on data from Conversio Market & Strategy GmbH (2021))

<u>Discussion</u>
In general, PVC-containing waste from the construction sector such as flooring, pipes or window frames are suitable for mechanical recycling and are therefore collected separately and recycled in many European countries, e.g. in Germany (Ciacci, Passarini, and Vassura 2017) (ECVM Focus Group Disposal 2021). As outlined in section 4.2.3, 47.1% of the registered recycled quantities of PVC within Vinylplus can be allocated to window profiles and related profiles and 11.1% to pipes and fittings and thus, rigid PVC (Vinylplus 2020c). According to Conversio Market & Strategy GmbH (2021) data, approximately 33% of the total amount of PVC present in C&D waste (1,075,000 tonnes) are mechanically recycled.

In some Southern European Countries (e.g. Spain) with less developed recycling systems and lower landfilling costs, C&D waste is disposed of in landfills (ECVM Focus Group Disposal 2021). C&D waste is also treated in different incineration facilities (Focus Group Disposal 2021). However, C&D waste like windows or doors made from PVC cannot be handled in cement kilns as these materials do not comply with the specifications of cement kilns due to the elevated chlorine concentrations (IFC 2017). Other types of C&D waste such as carpets might be part of the waste input of cement kilns (Material Research L3C Focus Group Disposal 2021). Around 23% of PVC in C&D waste are directed towards disposal and 44% are directed towards energy recovery.

In general, rigid PVC accounts for the largest share (70%) of PVC in post-consumer C&D waste compared to flexible PVC (30%) (Conversio Market & Strategy GmbH 2021). Relevant facilities for the treatment of PVC waste from C&D waste are outlined in Annex 5.2.

6.3.1.3 WEEE

<u>Qualitative considerations and quantitative scoping</u>
PVC is mainly used in EEE for cable and wire insulation as well as cable trunking (Pickard and Sharp 2020). Thus, PVC present in WEEE as part of electric cables and wires PVC is mostly flexible PVC (Nakamura et al. 2009). Based on the same Conversio Market & Strategy GmbH (2021) data, around 79% of PVC in post-consumer electro and electronics products are flexible PVC.

Other applications include plugs, casings and refrigerator door seals (Villanueva and Eder 2014). Furthermore, PVC can mainly be found in the following applications (Martinho et al. 2012; Villanueva and Eder 2014):

- Large cooling appliances/refrigerators;
- Toys, leisure and sports equipment[99];
- Dishwashers;
- CRT (cathode ray tube) monitors;
- Small WEEE[100]; and
- Central processing unites (CPUs).

In a first step, the EEE categories[101] comprising relevant PVC containing applications were identified[102]. In addition, the shares of PVC in the respective applications were analysed. In a second step, the quantitative scoping of the quantities directed towards recovery or disposal was performed for each of the EEE categories separately, since Eurostat data is available for each of the identified EEE categories. Eurostat data for the year 2018 was used. In the next step, the sum of all EEE categories was calculated and used for the development of a mass flow showing the total amount of waste from relevant EEE categories containing PVC and the associated fractions directed towards different recovery or disposal (see mass flow in Annex 5.2)[103]. Further assumptions were necessary to perform the quantitative scoping. They are described in more detail in Annex 5.2.

Based on these assumptions, the quantitative scoping of PVC in WEEE was performed (Figure 40).

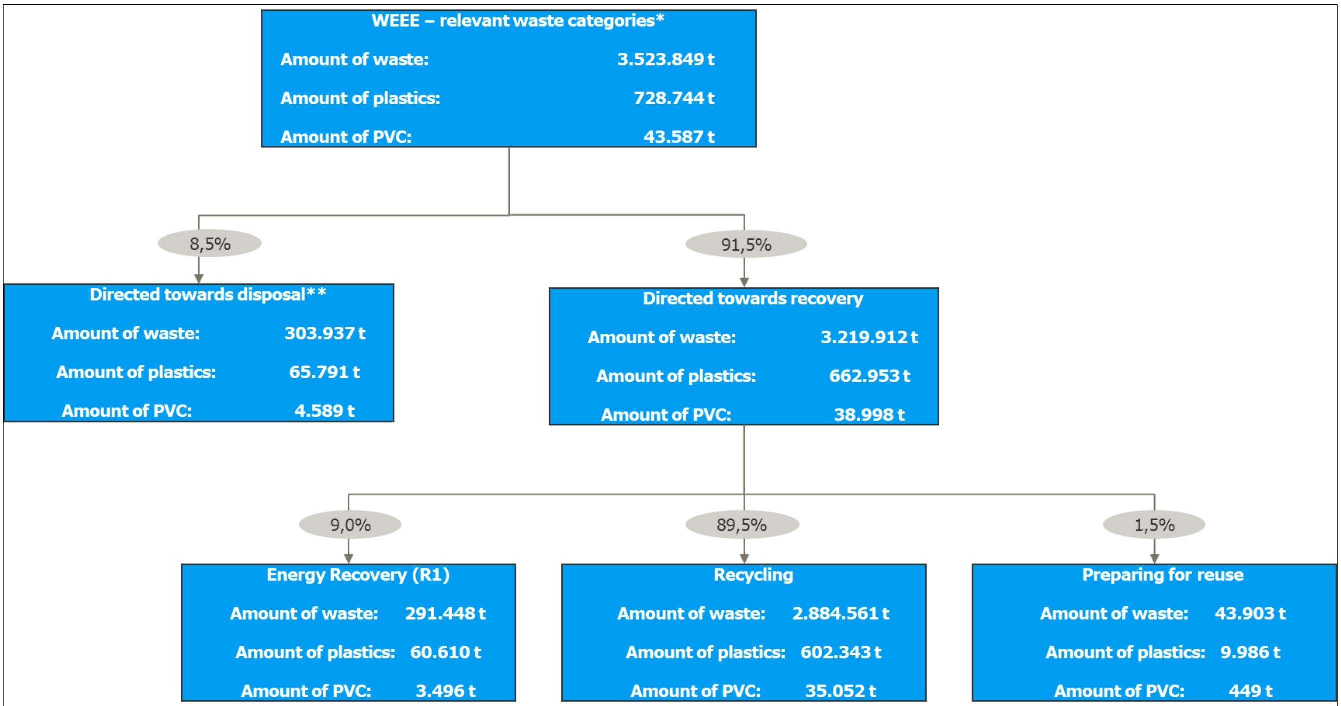
[99] This includes for instance electric trains or car racing sets or sports equipment with electric or electronic components (for more EEE which fall into this category, see ANNEX II of Directive 2012/19/EU (WEEE Directive (https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:02012L0019-20180704)

[100] This includes for instances beaters, balances, radios, toasters, printers or copying equipment (Martinho et al. 2012)

[101] This refers to EEE categories as outlined in ANNEX I of Directive 2012/19/EU (WEEE Directive (https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:02012L0019-20180704).

[102] On Eurostat, data is available for the following EEE categories: large household appliances; small household appliances; IT and telecommunications equipment; consumer electronics; photovoltaic panels; lighting equipment; gas discharge lamps; electrical and electronic tools; toys, leisure and sports equipment; medical devices; monitoring and control instruments; automatic dispensers.

[103] In order to verify whether the assumptions and calculations lead to a reasonable estimate of the total amount of PVC in the current waste stream of EEE, further data indicated by Baxter et al. (2014) and Maisel et al. (2020) was used (see Annex 5.2).



* This includes large household appliances, small household appliances, IT and telecommunications equipment, consumer equipment and photovoltaic panels as well as toys, leisure and sports equipment
** A further differentiation of waste treatment options was not possible.

Figure 40. Quantitative scoping of waste treatment options for PVC present in relevant categories of WEEE (based on data from Eurostat for the year 2018 (EU 27))

As Figure 41 indicates, PVC present in WEEE is mainly directed towards recovery, while disposal only plays a minor role. According to the assumptions made, 91.5 % of PVC in WEEE are directed towards recovery. Thereof, the majority (89.5 %) of PVC is recycled. Comparing this to the total amount of PVC in WEEE (43,587 t), this results in an overall recycling rate of 81.5 % for PVC in WEEE. When analysing these shares, it should be taken into account that the waste treatment options for PVC present in different EEE categories are calculated based on the waste treatment options of the whole EEE category. The EEE categories comprise further materials such as copper or aluminium which represent target materials in the recycling of cables (Díaz et al. 2018). Aluminium is estimated to make up between 35% and 55% of the total weight of cables (Grimaud, Laratte, and Perry 2019) and therefore it can be assumed that the high recycling rate of metals contributes to a high overall recycling rate across the EEE categories assessed. This could be taken as an indication, that the percentage of PVC from WEEE directed towards recycling according to Figure 41 might be an overestimation. This is also supported by the information in Fråne et al. (2018). They point out that PVC present in WEEE – mainly flexible PVC –is normally shredded and sent to energy recovery or landfilling in the Nordic countries suggesting that the high recycling rate calculated based on this report´s assumptions is rather unlikely.

Generally, shredder residues from WEEE and ELVs are considered to be one of the most relevant waste streams containing PVC which is treated in hazardous waste and MSW incinerators (Poll Focus Group Disposal 2021). Shredder residues of WEEE are also disposed of by landfilling (Miliute-Plepiene, Fråne, and Almasi 2021). However, no information or data could be identified on whether and in which quantities shredder residues from WEEE are disposed of in landfills for hazardous waste. High concentrations of DEHP can be found in PVC from post-consumer cable waste (Miliute-Plepiene, Fråne, and Almasi 2021). Relevant facilities for the treatment of PVC in WEEE are outlined in Annex 5.2.

Discussion

For verifying the results of the mass flow analysis, they are compared to the results of the study carried out by Conversio Market & Strategy GmbH (2021) for PVC in post-consumer electro and electronic products (see Table 6-3)[104].

Table 6-3: Comparing the results of the mass flow analysis for WEEE with the data from Conversio Market & Strategy GmbH

Treatment option	Data from Conversio Market & Strategy GmbH		Results of the mass flow analysis	
	Quantity of PVC [tonnes]	Share of PVC directed towards the treatment option	Quantity of PVC based on [tonnes]	Share of PVC directed towards the treatment option
Total PVC waste	185,000	-	43,587	-
Total recovery	160,000	86.5%	38,998	89.5%
Mechanical recycling/Recycling	88,000	47.6%	35,052	81.5%
Energy recovery	72,000	38.9%	3,496	8.0%
Disposal	25,000	13.5%	4,589	10.5%

Table 6-3 shows that significant differences exist regarding the total quantity of PVC waste in WEEE. The relatively low number of PVC waste present in the mass flow analysis could potentially be attributed to the assumptions made concerning the fraction of plastics in the analysed EEE categories and the share of PVC within those fractions (see Annex 5.2.). It can therefore be concluded that the total quantity of PVC in WEEE (mainly flexible PVC) ranges between around 43,000 and 185,000 tonnes. The shares of PVC directed towards disposal (13.5 % vs. 10.5 %) and towards recovery are in the same range (86.5 % vs. 89.5 %), however the shares directed towards recycling and energy recovery vary considerably between both analyses. As indicated above, the high recycling rate resulting from the mass flow analysis might be an overestimation due the higher recyclability of target materials such as copper or aluminium present in electro- and electronic products. Consequently, the recycling rate for PVC can be assumed to be significantly lower and the rate indicated by Conversio Market & Strategy GmbH (2021) of ca. 47.6 % might be a more realistic estimation. The same applies to the share of PVC which is energetically recovered according to the mass flow analysis. Here, it can be assumed that the suggested share of 8.0 % represents an underestimation.

6.3.1.4 Packaging and household waste

Qualitative considerations and quantitative scoping

As outlined in chapter 3.3, with a current consumption of 450,000 tonnes, the utilization of PVC for packaging purposes is an important application area. PVC is used in various household packaging applications such as cleaning products, toiletries, caps and closures of bottles, stretch film or trays for salads or desserts (Villanueva and Eder 2014). Therefore, household plastic waste containing PVC ends up in streams of MSW (Dahlbo et al. 2018; Schwarzböck et al. 2017). Furthermore, due to its frequent use in the packaging sector (PlasticsEurope 2019a), considerable amounts of PVC are expected to end up in plastic packaging waste.

[104] Please note that the shares indicated in the table for the mass flow analysis are calculated in relation to the total waste and therefore differ from the shares indicated in the mass flows. The quantities and shares indicated for the mass flow data for recycling include recycling and preparation for reuse.

Both types of PVC (flexible and rigid) are used for packaging purposes. For instance, rigid PVC is used in blister pack applications while flexible PVC is used for food packaging or shrink foils (see section 7.9). According to the data provided by Conversio Market & Strategy GmbH (2021), 65% of PVC post-consumer packaging waste is rigid PVC while flexible PVC accounts for only 35%.

For developing a mass flow, the main waste streams for PVC used in packaging applications are assumed to be plastic packaging waste and MSW. On Eurostat, relevant data is available for the following waste streams:

- Household and similar waste (Eurostat Code 10.1)
- Plastic wastes (Plastic packaging waste[105])

In a first step, a separate mass flow was prepared for both waste streams showing the shares of household and similar waste as well as plastic packaging waste being directed towards disposal and recovery. The approach for developing the mass flows is described below. In a second step, the quantitative scoping of PVC present in both waste streams was performed based on the current consumption of PVC in the packaging sector. The methodology as well as associated assumptions are described in detail in Annex 5.2. For performing the quantitative scoping of PVC packaging waste, the current consumption of PVC used for packaging purposes in the EU27 was taken as a baseline. The consumption is estimated to be 450,000 tonnes[106] (see section 3.3). In comparison to plastics used in the C&D sector or the automotive industry, packaging waste is characterized by a short lifetime and has an almost direct impact on the current waste generation (Schwarzböck et al. 2017).

According to PlasticsEurope (2019b), 52 % of post-consumer plastic waste included in household residual- and municipal waste is collected via mixed waste collection, while the share of post-consumer plastic waste collected via separate waste collection systems is 48 %[107]. In the absence of other data, the following assumptions were made:

- 52 % of PVC used as packaging material ends up in the MSW stream (applied to the quantities directed towards disposal and recovery as outlined in Annex 5.2.3waste stream household and similar waste).
- 48 % of PVC used as packaging material ends up in the plastic packaging waste stream (applied to the quantities directed towards disposal and recovery as outlined in Annex 5.2.3; waste stream plastic packaging waste).

Based on these assumptions, the quantitative scoping for PVC as packaging material was performed (see Figure 41).

[105] Data is collected based on the EU Parliament and Council Directive 94/62/EC of 20 December 1994 on packaging and packaging waste (https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:01994L0062-20180704&from=EN).

[106] According to ECVM expert input, PVC packaging waste is between 300.000 and 400.000 tonnes per year (ECVM input FocusGroup Disposal 2021).

[107] It should be noted that there is an increasing trend for separate collection of waste plastics in many countries (Schwarzböck et al. 2017).

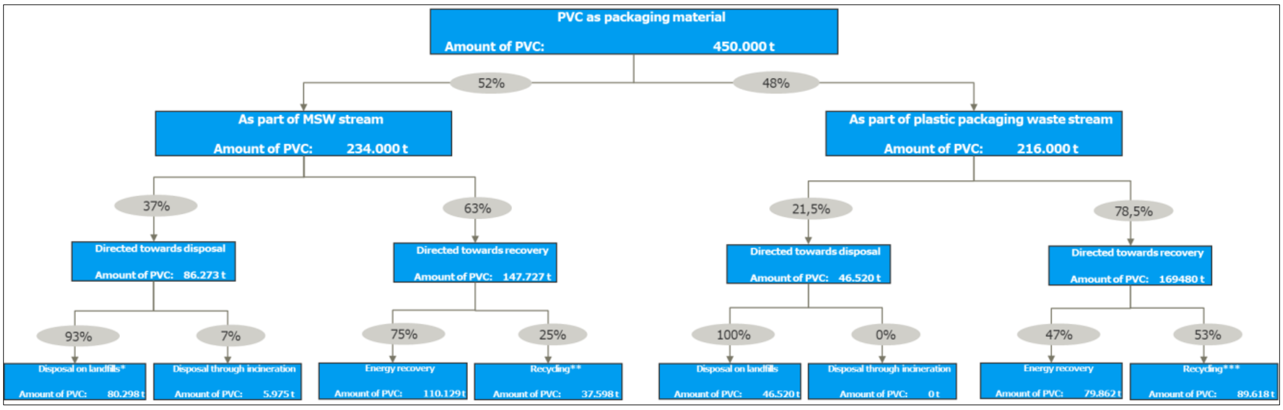


Figure 41: Quantitative scoping of waste treatment options for PVC present in the packaging and MSW stream (based on the current consumption volume and Eurostat for the year 2018 (EU 27))

According to the assumptions made, a relatively equal share of PVC is disposed of as MSW and as packaging waste (see Figure 41). If PVC in packaging waste or MSW is directed towards disposal (37% and 21,5%), the disposal on landfills represents the main waste treatment option, while incineration without energy recovery only plays a minor role. Based on our assumptions, PVC directed towards disposal as part of the plastic packaging waste stream entirely ends up in landfills. When PVC is directed towards disposal as part of the MSW stream, the majority is disposed of in landfills (93 %) and only a minor share ends up in incineration facilities without energy recovery (7 %).

Moreover, Figure 41 indicates that the largest fraction of PVC present in both waste streams is directed towards recovery (63% and 78,5%). Here, the treatment depends on the respective waste stream. The largest part of PVC present in MSW stream directed towards recovery is energetically recovered (75%) and a significantly lower share (25%) is recycled. If PVC present in the plastic packaging waste stream is directed towards recovery, it is recycled or energetically recovered in approximately equal proportions (53% and 47%) indicating a higher recycling rate as compared to PVC packaging waste present in MSW. However, when interpreting this data, it is important to bear in mind that no specific information or data on the fate of PVC present in the MSW and the plastic packaging waste stream could be identified. Hence, the indicated shares, i.e. the quantities directed towards disposal (including disposal on landfills and incineration without energy recovery) and towards recovery (including energy recovery and recycling) are based on Eurostat data for the overall waste streams and not specifically for PVC. In the absence of other data, these shares were applied to PVC present in the two waste streams analysed. Thus, the high recycling rate indicated in Figure 41 for PVC in plastic packaging waste might be an overestimation, since recycling technologies for other polymers of the plastic packaging waste stream such as PET, PP or PE are more advanced (Antonopoulos, Faraca, and Tonini 2021).

In conclusion, the main treatment options for flexible and rigid PVC present in packaging and household waste are landfilling and energy recovery, while it can be assumed that the high recycling rate of PVC present in the plastic packaging waste stream represents an overestimation. Relevant facilities for the treatment of PVC from packaging and household waste are outlined in Annex 5.2.

<u>Discussion</u>

A comparison of the results of the mass flow analysis with the data provided by Conversio Market & Strategy GmbH (2021) for post-consumer PVC packaging waste is given in Table 6-4[108]

[108] Please note that the shares indicated in the table for the mass flow analysis are calculated in relation to the total waste and therefore differ from the shares indicated in the mass flows. The quantities and shares indicated for the mass flow data for recycling include recycling and preparation for reuse.

Table 6-4: Comparing the results of the mass flow analysis for packaging and household waste with the data from Conversio Market & Strategy GmbH

Treatment option	Data from Conversio Market & Strategy GmbH		Results of the mass flow analysis	
	Quantity of PVC [tonnes]	Share of PVC directed towards the treatment option	Quantity of PVC based on [tonnes]	Share of PVC directed towards the treatment option
Total PVC waste	478,000	-	450,000	-
Total recovery	389,000	81.4%	317,207	70.5 %
Mechanical recycling/Recycling	73,000	15.3%	127,216	28.3 %
Energy recovery	316,000	66.1%	189,992	42.2%
Disposal	89,000	18.6%	132,793	29.5%

In terms of the total annual quantity of post-consumer PVC packaging waste, both analyses come to a similar result ranging between 450,000 and 478,000 tonnes. The total amount of PVC directed towards recovery (81.4 % vs. 70.5 %) and disposal (18.6 % vs. 29.5 %) shows a relatively small to moderate variation. As regards the mass flow analysis, this difference could be explained through the influence of PVC being part of the MSW stream, where – based on the assumptions made – relatively large quantities (37 %) are directed towards disposal.

More significant differences exist concerning the shares of PVC recycled and energetically recovered. Here, the mass flow analysis revealed a comparatively high recycling rate of 28.3 %, which can most likely be attributed to the higher recycling rate of other plastic types present in the plastic packaging waste stream (see right side of Figure 41). It can therefore be concluded that the recycling rate of PVC packaging waste ranges between 15.3 % and 28.3 % with a tendency to the lower value, whereas the share of PVC which is energetically recovered ranges between 42.2 % and 66.1 % with a tendency to the higher value.

6.3.1.5 Medical waste

<u>Qualitative considerations and quantitative scoping</u>
On Eurostat, data for the waste stream "healthcare and biological waste" (Eurostat Code 05) is available. This waste stream contains "infectious health care wastes" (Eurostat Code 05.1) as well as "non-infectious health care wastes" (Eurostat Code 05.2). PVC in medical waste is mainly present in tubes, hoses, head covers, perfusion products, blood- and urine bags, gloves, oxygen and anaesthetic masks, catheters, drip chambers, probes as well as transfusion sets (Ciacci, Passarini, and Vassura 2017; Fråne et al. 2018; Vinylplus 2020d). The reasons for the popular use of PVC in health care applications such as flexibility or heat resistance are further described in section 7.14. Thus, primarily flexible PVC is used in the health care sector (Omnexus 2021a).

Based on available information provided by different sources such as scientific publications or expert input (see Annex 5.2), it was assumed that PVC present in medical waste is disposed of as non-hazardous waste (LOW codes 18 01 02 and 18 01 04) and/or as hazardous waste (LOW code 18 01 03*). Both are included in the Eurostat data on non-infectious health care wastes and infectious health care wastes, respectively.[109] Since no specific information on the distribution of PVC between these two waste streams could be identified, the shares of PVC present in medical

[109] In this context, it should be noted that data taken from Eurostat does not only include non-infectious and infectious human health care waste but also animal infectious health care waste and non-infectious animal health care waste. However, for calculating the shares of PVC in medical waste directed towards different waste treatment options, no differentiation between animal and human health care waste was made.

waste directed towards disposal or recovery were calculated based on the shares as indicated for non-infectious and infectious health care waste on Eurostat (see mass flow in Annex 5.2).

In a next step, these shares were used to carry out the quantitative scoping for PVC present in medical waste. Many of the PVC applications or products used in hospitals are made for single use (PVC MED Alliance 2021b). Thus, it was assumed that medical devices containing PVC have a short lifetime and end up in the waste stream within a year of production. The current consumption of PVC in the health care sector in Europe was taken as baseline for the quantitative scoping. Fråne et al. (2018) indicate an annual PVC consumption of around 85,000 tonnes in the European health care sector.[110] This value was taken as a baseline for the quantitative scoping.

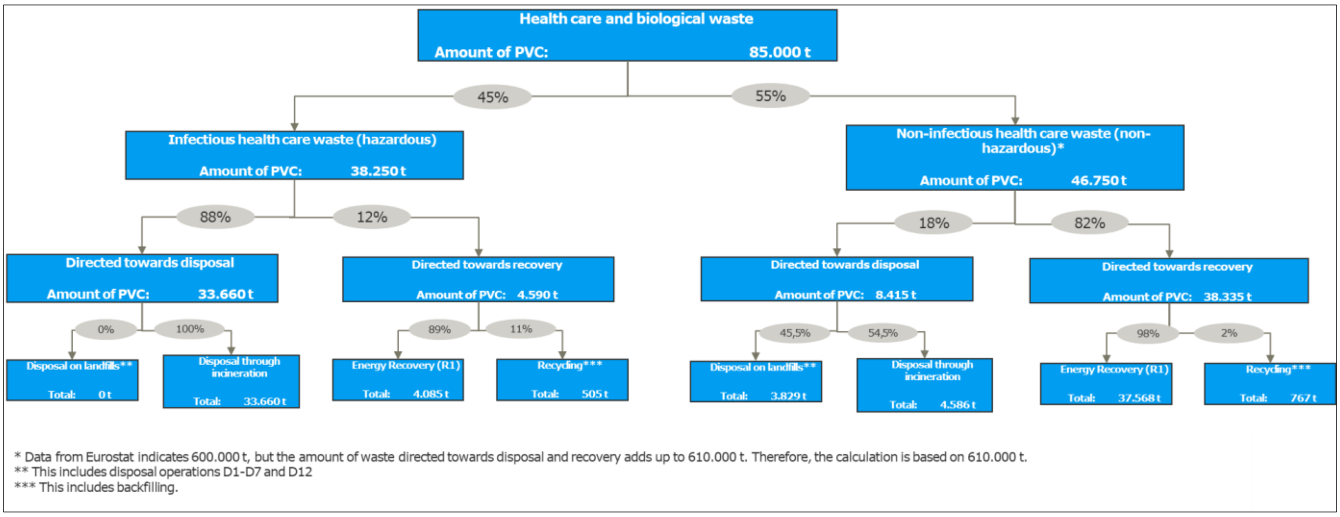


Figure 42. Quantitative scoping of waste treatment options for PVC present health care and biological waste (based on the current consumption volume as indicated by Fråne (2018) **and Eurostat Data for the year 2018 (EU27))**

Based on the assumptions, 45 % of PVC is treated as infectious health care waste, while around 55% are treated as non-infectious health care waste. Landfilling does not represent a relevant treatment option for infectious medical waste containing PVC. For non-infectious medical devices containing PVC, landfills play a minor role. According to the quantitative scoping, only 3,829 tonnes of medical waste containing PVC are disposed of in landfills. This corresponds to approximately 4,5 % of the current consumption volume. An even smaller proportion of health care waste (1,272 t of infectious and non-infectious waste) is recycled (ca. 1.5 %).
The majority of infectious health care waste (33,660 t) is disposed of by incineration, either without energy recovery or with (4,054 t). In contrast, for non-infectious health care waste, energy recovery represents the most relevant treatment option, while incineration without energy recovery is less relevant (4,596 t).
Thus, taking into account the total amount of infectious and non-infectious health care waste, flexible PVC present in waste streams of medical waste is mainly energetically recovered (ca. 49%) and or incinerated without energy recovery (ca. 45 %). Based on our assumptions, recycling of PVC medical waste takes place only to a very small extent (see Figure 40).

[110] Moreover, according to Johnsen (2015), PVC is estimated to make up 5% to 15% of medical waste. When applying this share to the total amount of health care and biological waste (Eurostat data; 1.350.000t), the share of PVC is within the range of ca. 68.000 t (5%) and 204.000 t (15%). Accordingly, the total amount of 85.000 tonnes lies within the range estimated by Johnsen (2015). In addition, according to PVC Med Alliance (2021a), the share of PVC among the 455.000 tonnes of polymers used in medical devices in Europe is 27%. This corresponds to approximately 122.850 tonnes of PVC.

Consideration regarding the recycling of PVC waste from the medical sector
Several programmes for the recycling of medical waste containing PVC exist. This concerns non-infectious medical devices.[111] [112] As Figure 40 shows, only a minor fraction of the PVC medical waste directed towards recovery is recycled (2 %). An example for PVC recycling in the healthcare sector from Australia demonstrates that there is a considerable potential for PVC recycling. In 2020, 129 tonnes of PVC waste were collected and recycled from 248 hospitals (VinylCouncil Australia 2021). In Europe, a pilot project was initiated in February 2021 in Belgium by VinylPlus Med (VinylCouncil Australia 2021). However, no data or information on the quantities or shares of PVC recycled during this pilot project could be identified.

Furthermore, no information or data could be identified on the disposal of medical waste containing PVC in landfills for hazardous waste. Generally, hazardous medical waste is assumed to be disposed of by incineration rather than being landfilled (ECVM Focus Group Disposal 2021). This is in line with the quantitative scoping based on Eurostat data. Different incineration facilities are used for the incineration of hazardous as well as non-hazardous shares of medical waste (see Annex 5.2). After pre-treatment, e.g. through autoclaving, medical waste can also be disposed of in landfills for non-hazardous waste (Windfeld and Brooks 2015). Therefore, it can be assumed that small fractions of PVC from medical waste also end up in non-hazardous waste landfills together with MSW. In a study carried out by Miliute-Plepiene et al. (2021), there is no indication that health care waste containing PVC is disposed of in landfills in any of the Nordic countries.

Discussion

The study carried out by Conversio Market & Strategy GmbH (2021) does not specifically address PVC in medical waste.[113] Since no further data on the treatment of PVC in medical waste could be identified, no overall comparison with other studies was possible.

6.3.1.6 Summary of the quantitative scoping and relevant treatment options

As the descriptions of the approaches used for the quantitative scoping of PVC in different waste streams show, data was taken from various sources and is associated with varying degrees of uncertainty. Data gaps exist especially concerning the amount of PVC in C&D waste directed towards disposal and recovery. The long average lifetime of C&D waste further hinders the development of a mass flow for PVC in the C&D waste stream based on Eurostat data. Consequently, the data provided in the study of Conversio Market & Strategy GmbH (2021) was used.

As regards the quantitative scoping of PVC present in the four other waste streams analysed (ELVs, WEEE, packaging and household waste as well as medical waste) further data gaps exist. Data on Eurostat is only available for the overall waste streams which often include different types of waste and materials, while specific information on the fate of PVC within these waste streams is lacking. For the quantitative scoping, several assumptions regarding the share and presence of PVC in these waste streams were necessary. Another limiting factor is the lack of information or data on specific quantities of waste or PVC directed towards different incineration facilities or types of landfills. Consequently, the quantitative scoping can only provide a rough estimation on the quantities of PVC waste per year directed towards recovery including energy recovery, recycling and reuse or disposal including the disposal on landfills and through incineration without energy recovery.

[111] https://pvcmed.org/healthcare/pvc-waste-management/

[112] http://recyclinginhospitals.com.au/

[113] Within the study of Conversio Market & Strategy GmbH (2021), PVC in medical products is covered within the category “others”. The quantities of PVC within this category directed towards disposal and recovery are described in the section 6.3.1.7.

A general mass flow showing the quantities of PVC waste directed towards recovery and disposal based on the four waste streams analysed was compiled. In the absence of other data, it was decided to also incorporate the data on PVC in C&D waste as provided within the study of Conversio Market & Strategy GmbH (2021). The result is presented in Figure 43. It should be noted that due to the different lifetimes of PVC containing products and applications, the quantitative scoping for PVC present in packaging and household waste and in medical waste was performed based on the current consumption volume, whereas the quantitative scoping for PVC in ELVs and WEEE was performed based on the current amount of PVC present in the waste streams[114]. The data for C&D waste refers to the year 2020.

Overall, based on the assumptions made, the total amount of PVC present in the five waste streams analysed is 1,719,334 tonnes per year. Thereof, around three quarters (1,256,054 t) are directed towards recovery and one quarter is directed towards disposal (463,280 t). If PVC is directed towards recovery, approximately 57.3 % are energetically recovered and 42.6 % are recycled, while only a very small fraction is prepared for reuse. If PVC is directed towards disposal, 28.0 % are disposed of in landfills and only 11.1 % are incinerated without energy recovery. For the remaining share of PVC (60.9 %) directed towards disposal, it is not further specified whether the PVC is directed towards landfills or incineration without energy recovery.

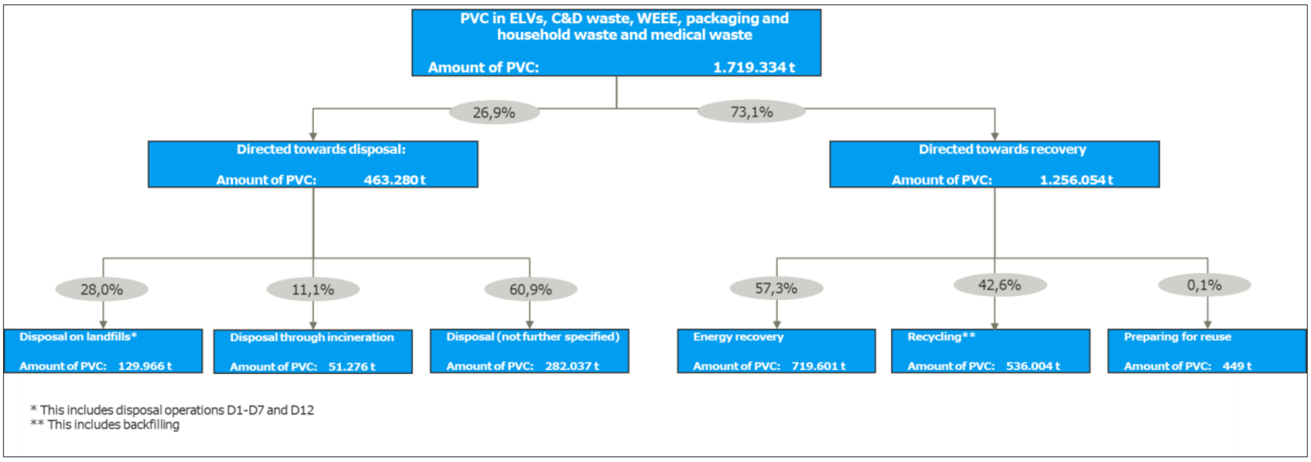


Figure 43: Mass flow showing the waste treatment options for PVC present in the waste streams analysed (ELVs, C&D waste, WEEE, packaging and household waste and medical waste)

[114] This refers to the current among of PVC present in the waste stream of ELVs and WEEE based on the assumptions made. The assumptions are described in the respective sections above.

6.3.1.7 Discussion and overall conclusion

For a critical evaluation of the results, the results of the mass flow analysis are compared to the results of the study by Conversio Market & Strategy GmbH (2021) (see Table 6-5).

Table 6-5: Comparing the results of the mass flow analysis for all waste streams analysed with the data form Conversio Market & Strategy GmbH

Treatment option	data from Conversio Market & Strategy GmbH		Results of the mass flow analysis	
	Quantity of PVC [tonnes]	Share of PVC directed towards the treatment option	Quantity of PVC based on [tonnes]	Share of PVC directed towards the treatment option
Total PVC waste	2,435,000	-	1,719,334	-
Total recovery	1,891,000	77.7%	1,256,054	73.1%
Mechanical recycling/Recycling	599,000	24.6%	536,004	31.2%
Energy recovery	1,292,000	53.1%	719,601	41.9%
Disposal	544,000	22.3%	463,280	33.5%

When interpreting the quantities and shares presented in Table 6-5, the differences concerning the analysed waste streams need to be taken into account. The major difference refers to the waste stream medical waste, which has not been separately analysed in the analysis of Conversio Market & Strategy GmbH (2021). In the analysis of Conversio Market & Strategy GmbH (2021), medical waste containing PVC is part of the waste stream "others" which also includes PVC present in houseware, furniture, medical products, gardening and agriculture and leisure products. This stream accounts for around 605,000 tonnes of PVC waste per year[115] compared to the assumed 85,000 tonnes of PVC in medical waste in the mass flow analysis. Therefore, this waste stream can explain a large part of the difference between the mass flow analysis of this report and the data from Conversio Market & Strategy GmbH (2021) in terms of the total PVC waste[116].

In summary, the following conclusions can be drawn from the comparison of the data and the waste streams analysed (see also Table 6-5):

- Between 22.3 % and 33.5 % of total PVC waste are directed towards disposal[117].
- The share of PVC waste directed towards recovery ranges from 73.1 % to 77.7 %.
- The majority of PVC waste is energetically recovered (41.9 % to 53.1 %).
- Between approximately one quarter (24.6 %) and one third (31.2 %) of PVC waste is recycled, respectively mechanically recycled.

As regards the recycling rate indicated for the mass flow analysis (31.2 %), it should be noted that the amount of PVC in the packaging and household waste stream was assumed to be 450,000 tonnes making up more than one quarter of the PVC present in the five waste streams analysed. It can be assumed, that the recycling rate calculated for PVC in the packaging and

[115] PVC present in the waste stream "Others" is mainly directed towards recovery (74.4%) and only 25.6% are directed towards disposal. The majority of PVC in this waste stream is energetically recovered (61.6%) and only 12.6% are mechanically recycled (Conversio Market & Strategy GmbH 2021)

[116] It should also bet noted, that the mass flow analysis included the data from Conversio Market & Strategy GmbH (2021) for C&D waste, which accounts for the largest share of PVC waste in all waste streams analysed.

[117] According to Conversio Market & Strategy GmbH (2021) disposal means landfilling, while the mass flow analysis also referred to incineration without energy recovery.

household waste stream on the basis of the mass flow analysis represents an overestimation. This could explain the relatively high recycling rate indicated for the mass flow analysis. According to Conversio Market & Strategy GmbH (2021), the largest fraction of PVC directed towards recycling is mechanically recycled (see also Section 4.2). If PVC is energetically recovered, it is mainly treated in MSW incineration plants (ca. 1,195,000 t). A far smaller proportion (ca. 97,000 t) is used as refuse-derived fuel (RDF) (Conversio Market & Strategy GmbH 2021). In general, the results of both analyses should be interpreted with caution due to the uncertainties and data gaps which exist concerning the fate of PVC present in the waste streams assessed.

6.4 Incineration: Description and scoping of the major treatment options for PVC in the EU

6.4.1 MSW incineration with energy recovery

PVC from all identified waste streams is treated in MSW incineration plants (Focus Group Disposal 2021). Therefore, MSW incinerators are the most relevant treatment facilities for the incineration of PVC waste (see chapter 6.3.1.7). The shares of PVC accepted at MSW incineration plants depend on the type of plant and on the combustion temperature (EURiTS Focus Group Disposal 2021). As indicated in section 6.2.1, MSW incineration plants do not tend to accept PVC-rich waste streams due to the potential for corrosion. Thus, the majority of MSW incineration plants treats MSW with chlorine concentrations ranging from 0.1 – 1.0% DS (dry solids) (Neuwahl et al. 2019). The process of a typical MSW incineration plant including information on exhaust gas cleaning systems is described in **Annex 5.1**.

6.4.1.1 Associated environmental impacts – emissions to air

Chlorine content, release of HCl and neutralisation residues

In comparison to other energy carriers (e.g. fossil fuels), MSW has a complex composition with high contents of chlorine and sulphur and is therefore more complicated to treat (Brunner and Rechberger 2015). The share of PVC present in the MSW fraction to be incinerated mainly influences the concentration of acids in the raw gas (Bernard, Hjelmar, and Jürgen 2000; Brunner and Rechberger 2015). When MSW containing PVC is incinerated, the main acid compounds produced are HCl and SO_x (Bernard, Hjelmar, and Jürgen 2000). During combustion, the release of HCl begins between temperatures of 200 – 360°C. At temperatures of around 550°C, PVC is almost completely decomposed through a combination of thermal decomposition and de-polymerisation (Ma et al. 2010).

According to the Waste Incineration BREF (Best Available Techniques reference document), combustion gases in European waste incineration facilities are maintained at a minimum temperature of 850°C for two seconds to ensure a good burnout of combustion gases (Neuwahl et al. 2019).

In summary, concerns exist regarding the formation of hydrochloric acid (HCl) during MSW incineration processes:

- when released into the atmosphere, HCl has the potential to contribute to the formation of acid rain (Sadat-Shojai and Bakhshandeh 2011; Shemwell, Levendis, and Simons 2001; Wisniewski et al. 2020)
- the elevated concentration of HCl causes corrosion in the boiler tubes and other plant equipment (Lu et al. 2017; Ma et al. 2010; Sadat-Shojai and Bakhshandeh 2011; Shemwell, Levendis, and Simons 2001)
- HCl provides chlorine atoms for the formation of chlorinated dioxin-like compounds and furans (Q. Huang et al. 2015; Johnsen 2015b; Shemwell, Levendis, and Simons 2001; G. Zhang et al. 2019) (see section on dioxins and furans below)

- high concentrations of HCl might increase the vaporization of heavy metals (Pedersen et al. 2009) (see chapter on heavy metals below).

Different gas treatment systems (dry, semi-dry and wet) exist for the abatement of HCl (Astrup 2008; Bernard, Hjelmar, and Jürgen 2000). Since HCl is highly soluble in water, it can be easily scrubbed out from the flue gas using wet flue gas cleaning systems (Buekens and Cen 2011). In addition, in dry or semi-dry systems, neutralisation agents[118] can be applied for HCl adsorption (Buekens and Cen 2011; Lu et al. 2017). Due to the rapid reaction of the neutralisation agents with the HCl, the HCl concentration can be significantly decreased to less than 1 ppmv[119] (Dou et al. 2012).
Generally, scrubbers can remove HCl efficiently from the flue gas. The BAT-associated emission levels for channelled emissions of HCl to air are < 2-6 mg/Nm³ for a new plant and < 2-8 mg/Nm³ for an existing plant (daily average emissions).[120] Based on the requirements of the Waste Incineration BREF, HCl emissions need to be monitored continuously in European MSW incineration facilities.[121] According to Quina et al. (2011), HCl emissions from MSW incineration generally are in a range between 0.1 and 6 mg/Nm³ and therefore below the European regulatory limits. This is also confirmed by Fråne et al. (2018), who state that HCl emissions from MSW incineration are far below the legally permitted emission limits.

Dioxins and furans
Between 1993 and 1995, 40% of the dioxin emissions in the European Union were attributable to incinerators (European Commission 2000). Although PVC constitutes only 0.7% of the total waste treated in MSW incineration facilities, 40 – 70% of the chlorine input are attributable to PVC providing the basis for high emissions of chlorinated dioxins (in the following only called "dioxins") (Johnsen 2015b). Thus, the influence of PVC on the formation of dioxins and furans is significant and has attracted considerable attention within the scientific community (European Commission 2000; Johnsen 2015b). Screening of available literature shows that the correlation between the chloride content and the dioxin formation is widely recognised (Katami et al. 2002; M. Zhang et al. 2015). Accordingly, the presence of chloride in the waste fraction significantly correlates with the formation of dioxins in exhaust gases (Katami 2002). Therefore, in comparison to the combustion of chlorine-free waste, the combustion of PVC waste leads to an increased discharge of dioxins, especially PCDDs, PCDFs and coplanar PCBs (Katami et al. 2002; M. Zhang et al. 2015). The formation processes of dioxins and furans are described in **Annex 5.3.**
However, according to several studies, the combustion conditions during the incineration process are of decisive importance for the formation of dioxins and furans (Buekens and Cen 2011; Font et al. 2010; M. Zhang et al. 2015). By maintaining adequate operating conditions in terms of temperature, turbulence, reaction/residence time and oxygen supply, it can be ensured that precursors of dioxins and furans are eliminated through the combustion process. Therefore, ensuring adequate operating conditions is of utmost importance for preventing dioxin emissions (Buekens and Cen 2011; Kulkarni, Crespo, and Afonso 2008; Lu et al. 2017; M. Zhang et al. 2015). Furthermore, several emission abatement technologies for reducing emissions of dioxins and furans from incineration facilities exist (see Annex 5.3).
To conclude, the operating conditions are more important for the formation of dioxins and furans than the chlorine input from PVC waste (Ciacci, Passarini, and Vassura 2017; M. Zhang et al. 2015). As such, it is vital for the environmental performance of incineration of PVC waste,

[118] Commonly applied absorption sorbents are: dolomite, limestone, chalk, lime, carbonates and sodium hydrogen carbonate (Tsvetkov, Polianczyk, and Zaichenko 2018). Most often, calcined limestone and CaO-based sorbents are utilized (Cao et al. 2014).

[119] Parts per million by volume

[120] https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32019D2010&from=EN

[121] The requirements of the BAT conclusions adopted under the Industrial Emissions Directive (IED) including the emission levels associated with the use of the best available techniques (BAT-AELs) are legally binding for all European facilities under the IED and must be implemented at least four years after the publication.

including avoiding emission of dioxins and furans that the incineration process is done under optimal operating conditions. According to various sources, today, MSW incineration facilities do no longer represent a significant source of dioxin and furan emissions (Johnsen 2015b) (PVC Information Council DK Focus Group Disposal 2021). This is demonstrated by the significant reduction of emissions of dioxins and furans from MSW incineration plants in France, Germany and the United States, for example (see Annex 5.3 for further information).

Considerations concerning emissions during OTNOC conditions
According to IPEN, concerns exist with respect to emissions of MSW incineration plants during transient phases such as the start- and shutdown phase and when plants are operated in by-pass or cleaning mode (IPEN Focus Group Disposal 2021). These phases are referred to as OTNOC situations[122] (Neuwahl et al. 2019). During OTNOC situations, emissions of POPs, especially of dioxins and furans might be higher (IPEN Focus Group Disposal 2021). A report from ToxicoWatch (2018) indicates that dioxins emissions during these phases can exceed the standard emissions levels even in state-of-the-art incineration facilities. Therefore, IPEN emphasises the importance of continuous long-term sampling (IPEN Focus Group Disposal 2021). In this context, CEWEP points out that BAT emission level values are calculated on the basis of the effective operation of the plants (including OTNOC situations) (CEWEP Focus Group Disposal 2021). Monitoring data from France, Belgium and Italy indicates that the emissions monitored with continuous sampling are comparable to the measurements during normal operating conditions, which are collected periodically in accordance with the IED (CEWEP Focus Group Disposal 2021).

Heavy Metals

Heavy metals might be present in the flue gas and solid residues of MSW incineration plants (Yu et al. 2015). In PVC heavy metals are present in the form of stabilisers, such as cadmium, lead, tin or zinc (Brown et al. 2000; Lu et al. 2017). In the EU, cadmium- and lead-based stabilizers have been replaced with other compounds, however the current PVC waste stock still contains large quantities of both cadmium and lead (Brunner and Rechberger 2015; M. Zhang et al. 2015).

While emissions of heavy metals from MSW incineration plants have been of environmental concern for many years, new filter technologies are able to remove fine particulates from the flue gas and therefore significantly reduce their concentration in fly ash. Even though the fine particulates in fly ash represent potential carriers of heavy metals, the emissions of heavy metals from MSW incineration plants strongly decreased due to state-of-the-art air pollution control technologies (Brunner and Rechberger 2015).

When PVC is incinerated, heavy metals primarily end up in the solid residues, i.e. in the bottom, fly ash and APC residues (Astrup, Riber, and Pedersen 2011; Scheirs 2003). Pb, Cu and Zn are the most relevant heavy metals found in these solid residues (Van Gerven et al. 2005). Generally, higher concentrations of volatile heavy metals such as Cd, Pb or Zn can be found in the fly ash, especially in fine particulates, while less volatile heavy metals end up in the bottom ash. (Nzihou et al. 2019). In fly ash, the most abundant heavy metals are Pb and Zn (Zacco et al. 2014). Higher contents of chlorine in the waste input to MSW incineration plants due to the presence of PVC containing waste affect the partitioning of trace elements. Accordingly, high levels of chlorine in the waste input material induce an enhanced vaporization of heavy metals, especially of Pb, and increase the share of heavy metals partitioned to the fly ash and flue gases (Pedersen et al. 2009; Yu et al. 2016). Astrup, Riber, and Pedersen (2011) also found that additional inputs of PVC waste increased the concentration of Pb in fly ash. They further found that adding PVC (5.5%) to the base-load waste contributes to higher air emissions of heavy metals (Sb and Cd). In addition, heavy metals released from the grate contribute to corrosion processes in the boiler

[122] OTNOC is an acronym for "other than normal operating conditions".

equipment (e.g. on heat transfer surfaces) and thus, besides alkali chlorides, heavy metals represent the primary sources for corrosion in MSW incineration plants (Pedersen et al. 2009). Moreover, diffuse emissions of heavy metals might occur from improper handling of solid residues during landfilling or reuse (Scheirs 2003). As regards the effects of PVC on the concentration and behaviour of heavy metals in bottom ash, literature suggests that an elevated input of PVC leads to decreasing concentrations of heavy metals in bottom ash because the chlorine contributes to the transportation of heavy metals to fly ash (Johnsen 2015a). On the other hand, higher chlorine concentrations in the bottom ash might contribute to the mobilisation of heavy metals and promote their leaching (Dou et al. 2012).

In summary, while state-of-the-art air pollution control techniques are able to effectively reduce heavy metals emissions to air, heavy metals from PVC incineration mainly end-up in the solid residues. Nevertheless, emissions to air might be considerably influenced by the waste input, as experiments carried out by Astrup, Riber and Pedersen (2011) demonstrate. Accordingly, high concentrations of PVC in the feedstock can contribute to increased stack emissions of heavy metals (especially Sb) (Astrup, Riber, and Pedersen 2011). As such, it can be concluded that the incineration of PVC increases the concentration of heavy metals in the solid residues (fly ash and APC residues) and might contribute to an increased risk of emissions of certain heavy metals to air. In case of the study of Astrup, Riber and Pedersen (2011), the addition of 5.5 % PVC significantly increased air emissions of heavy metals. Thus, input of PVC into MSW incineration facilities should be limited.

Concerning the environmental impacts stemming from the incineration of PVC, the "best available techniques" (BAT) conclusions for Waste Incineration[123] need to be taken into account. These BAT conclusions, which include emission limit values based on best available emission abatement technologies, have been published on December 3rd, 2019. Member States are obliged to ensure implementation of these conclusions in relevant permits for incineration facilities within four years from the date of publication. Therefore, it cannot be assumed that currently all European facilities are state-of-the-art. For instance, a study by Jedelshauser et al. (2020) concludes that 50 out of 66 waste incineration plants in Germany will need to undergo modernisation by 2030. In addition, European waste incineration facilities vary as regards the year of commissioning. Hence, further research efforts should investigate the decree to which MSW incineration facilities are equipped with state-of-the-art technologies.
Furthermore, there are several EU Member States with no or only limited waste incineration capacities. These countries are mainly located in Central and Eastern Europe as well as in the Baltic region (M. Quina, Bordado, and Quinta-Ferreira 2011; Wilts et al. 2017). In many of these countries high shares of MSW are landfilled (Scarlat, Fahl, and Dallemand 2019).

6.4.1.2 Associated environmental impact – MSW incineration residues

Waste incineration produces mainly three types of solid residues (Amutha Rani et al. 2008; Bernard, Hjelmar, and Jürgen 2000; Brunner and Rechberger 2015; Defra 2013; Huber et al. 2016):

- Bottom Ash;
- Fly ash; and
- Air Pollution Control (APC) residues.

Bottom ash represents the main residual material from MSW incineration (Defra 2013; Seniunaite and Vasarevicius 2017). It shows a heterogenous composition containing slag, non-ferrous and

[123] See in this regard: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=uriserv%3AOJ.L_.2019.312.01.0055.01.ENG&toc=OJ%3AL%3A2019%3A312%3ATOC

ferrous metals, glass and ceramics as well as other non-combustible and residual organic matter (Seniunaite and Vasarevicius 2017). In terms of weight, bottom ash accounts for 20 – 30% of the original waste input while it represents only approximately 10% of its volume (Defra 2013). Bottom ash is landfilled and is also used in the construction sector, as raw material for cement production or as feedstock in the ceramics industry (Brunner and Rechberger 2015; Huber et al. 2016; Verbinnen et al. 2017). However, the disposal and use of bottom ash is associated with environmental risks such as leaching of soluble salts and heavy metals (Sorlini, Collivignarelli, and Abbà 2017) (see further information in below in this section).

Regarding further solid residues, it is necessary to distinguish between fly ash and APC residues. Fly ash can be understood as the particle matter transported in the gases from the combustion chamber prior to the addition of air pollution control reagents (Amutha Rani et al. 2008; Huber et al. 2016; Reijnders 2018). In contrast, the residues captured downstream of the exhaust gas treatment are referred to as APC residues (Amutha Rani et al. 2008; Reijnders 2018). Thus, APC residues include the residues of dry- and semi dry scrubbers as well as the solid residues from wet scrubber systems (Amutha Rani et al. 2008). Fly ash can be either captured separately prior to treatment of air pollution residues or incorporated into the APC residues (Amutha Rani et al. 2008; Defra 2013).

Therefore, potential constituents of APC residues are fly ash, lime as well as bicarbonate and carbon (Defra 2013). Furthermore, fly ash can contain high concentrations of chlorides and other soluble salts, heavy metals such as Cd, Cu, Hg, Pb or Zn as well as dioxins and furans (Fellner et al. 2015; Huber et al. 2016). Consequently, both fly ash and APC are considered as hazardous waste (LOW code 19 10 13*) which need to be disposed of in special landfills (Huber, Laner, and Fellner 2018; Vehlow 2012). Further information on the management and disposal of APC residues and fly ash is provided in chapter 6.3.1.7.

It is important to note that bottom ash, fly ash and APC residues represent the most relevant pathway for emissions from MSW incinerators to the environment (Astrup 2008; M. J. Quina, Bordado, and Quinta-Ferreira 2008).

<u>Bottom ash</u>

Bottom ash from MSW incineration is produced in large quantities in the EU. Estimations show that MSW incineration generates from 12 – 15 Mt (Verbinnen et al. 2017) up to 17.6 Mt (Blasenbauer et al. 2020) of bottom ash per year. The amount of bottom ash generated equals approximately 20% of the waste incinerated (Recupero, Robb, and Vahk 2019). Bottom ash is produced in larger quantities than fly ash and air pollution control residues. Nevertheless, the latter two show a higher potential for environmental pollution (Astrup 2008). In contrast to fly ash and APC residues, bottom ash reveals significantly lower concentrations of chlorines, heavy metals, persistent organic pollutants (POPs) and dioxins (Margallo, Aldaco, and Irabien 2014; Recupero, Robb, and Vahk 2019). Typically, the metal fractions of bottom ash are separated and recycled, while the mineral fractions are either disposed of in landfills or used as raw material in the construction sector (Blasenbauer et al. 2020). For instance, in the Netherlands, Belgium or Denmark, the use of bottom ash in the construction sector represents a common practice (Margallo, Aldaco, and Irabien 2014; Recupero, Robb, and Vahk 2019). Unlike fly ash and APC residues, bottom ash can be classified as non-hazardous waste, if it can be verified that it does not show hazardous properties[124] and that concentrations of POPs do not exceed the limit values as indicated in the EU Regulation No. 2019/1021[125] (Blasenbauer et al. 2020).

[124] As indicated in the Commission Regulation (EU) No 1357/2014 of 18 December 2014 (https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32014R1357&from=EN).

[125] Referring to Regulation (EU) 2019/1021 of the European Parliament and the Council of 20 June 2019 on persistent organic pollutants (https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32019R1021&from=en)

In order to improve the physical and mechanical characteristics and to reduce potential releases of hazardous contaminants, bottom ash usually undergoes a stabilization process which uses inorganic reagents such as lime or cement (Margallo, Aldaco, and Irabien 2014). Environmental concerns exist concerning the leaching of toxic compounds from landfilled bottom ash and from bottom ash utilized as road construction material or as aggregate in concrete (Allegrini et al. 2015). According to Toller et al. (2009), utilisation of bottom ash in the construction sector might lead to increased leaching of heavy metals. This is in line with the findings of the European Chemicals Agency (2017a), which highlighted potential releases of lead from bottom ash used in road construction. On the other hand, it should be noted that the results of Birgisdóttir et al. (2006) indicate that the environmental benefits from the reuse of bottom ash as road construction material might be offset by the reduced environmental damage associated with landfilling bottom ash. From an environmental point of view, the potential for groundwater pollution from reusing bottom ash in road construction is higher than for landfilling, given the extent of road distribution (Birgisdóttir et al. 2006). A study of Allegrini et al. (2015) shows, that the recycling of concrete containing MSWI bottom ash might result in higher leaching of heavy metals than landfilling and utilization as road construction material. Moreover, during transportation and offloading processes, bottom ash might be released into the environment and cause contamination of water bodies and the ground. (Recupero, Robb, and Vahk 2019).

Finally, concerns exist regarding the level of POPs in bottom ash. In November 2021, the European Commission proposed a revision of the limit values in Annex IV of (Regulation (EU) 2019/1021)[126] including a new Low Pop Content Level (LPCL) for dioxins/furans and dl-PCB of 5 µg TEQ/kg (replacing the current limit value of 15 µg TEQ/kg); exceeding these limit values would lead to bottom ash being classified as hazardous waste. In this regard, CEWEP pointed out that the use of bottom ash will also be possible in future as the current levels of dioxins and furans in bottom ash are three to four times lower than the current LPCL of 15 µg TEQ/kg (CEWEP Focus Group Disposal 2021). No specific data focusing on dioxin levels in bottom ash could be identified for verifying this statement. However, Potrykus et al (2019) reported that dioxin concentrations in bottom ash are lower than in fly ash. Moreover, they analysed dioxin concentrations in fly ash from 35 observations provided by CEWEP. The 35 observations showed an average concentration of 2.5 µg TEQ/kg and for 31 of the 35 observations, the level of dioxins was at least 3 times below the current LPCL as outlined in Annex IV of the POP Regulation.

Fly ash and APC residues

Fly ash and bottom ash are often regarded as a unique output from MSW incinerators (De Boom and Degrez 2012; Zacco et al. 2014). Their safe disposal is associated with high costs and remains a global challenge (Huber, Laner, and Fellner 2018; Zhou et al. 2015). However, in contrast to bottom ash, fly ash and APC residues may possess higher concentrations of contaminants such as heavy metals, dioxins and furans as well as soluble salts (Fellner et al. 2015; Huber, Laner, and Fellner 2018; Recupero, Robb, and Vahk 2019). Leaching of pollutants from landfilled residues represents the most relevant environmental risk (Astrup 2008; M. J. Quina, Bordado, and Quinta-Ferreira 2008). Therefore, both fly ash and APC residues usually require treatment prior to being disposed of in landfills as hazardous waste (De Boom and Degrez 2012; Huber et al. 2016). In this way, the leaching potential of heavy metals including lead can be successfully reduced (European Chemicals Agency 2017b; M. J. Quina, Bordado, and Quinta-Ferreira 2008). Relevant disposal options and initiatives for the treatment of APC residues and fly ash are described in **Annex 5.4** to this report.

[126] https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32019R1021&from=DE

In conclusion, environmental concerns associated with the disposal of fly ash and APC exist with regard to the leaching of (Astrup 2008; European Chemicals Agency 2017b; Fellner et al. 2015; Pedersen et al. 2009):

- soluble salts (Cl, Na) and their impacts on ecosystems and drinking water resources;
- heavy metal such as Cd, Cr, Cu, Ni, Pb and Zn and their harmful effects on ecosystems and humans; and
- dioxins.

For instance, the mobility of heavy metals in fly ash residues tends to be higher in comparison to the mobility of heavy metals in PVC which is directly landfilled (Brown et al. 2000). According to the European Chemicals Agency (2017b), fly ash and APC residues are considered a long-term reservoir of lead from PVC and concludes that releases to the environment might occur over a long time period.

6.4.1.3 Associated economic aspects

Data on gate fees associated with different types of waste containing PVC treated in MSW incineration facilities could only be identified to a limited extent and further research should be carried out for assessing the costs. Kreißig et al. (2003) indicate that the gate fees associated with the energy recovery of one tonne mixed cable waste in a MSW incineration plant are estimated to be 100€ per tonne. These costs refer to a MSW incineration plant in Hamburg, Germany and are from 2003. Moreover, a comparison of different thermal treatment methods shows, that the estimated capital costs calculated per kW are lowest for incineration (see chapter 5.2) (Renewable Energy World 2014)[127]. Since PVC reaches MSW incineration facilities mainly in mixed waste streams, it might be challenging to quantify the costs associated with the specific treatment of PVC. However, the influence from incinerating PVC on the operation of a plant and associated economic aspects are described qualitatively below.

<u>Higher costs due to neutralisation agents and APC residues</u>
High concentrations of PVC in the waste input result in a higher demand of neutralisation agents due to the increased need of neutralising hydrochloric acid (Brown et al. 2000). This raises costs for MSW incineration facilities for two reasons: First, the increased demand of neutralisation agents leads to higher costs due to the cost-intensive production of neutralisation agents. Second, the higher quantities used for the neutralisation of hydrochloric acid result in higher quantities of neutralisation residues which need to be disposed of as hazardous waste (Fråne et al. 2018). Since these costs are not directly attributed to PVC waste, but to the incineration of PVC waste in mixed MSW streams, Brown et al. (2000) introduced the term "incineration subsidies" for PVC waste incineration.

According to Defra (2013), the weight of the air pollution control residues produced is on average around 2-6% of the weight waste fed into the incinerator. Stena Recycling (n.d.) points out that approximately 200,000 tonnes of flue gas waste is transported to landfill sites every year. However, the costs arising from the management of flue gas cleaning residues depends on the respective air pollution control system (Dong et al. 2020; Tukker et al. 1999). For instance, wet-cleaning systems cause higher costs than dry and semi-dry systems due to the required chemicals and treatment of waste water (Astrup 2008).

Therefore, various factors influence the level of costs associated with the incineration of PVC in MSW incineration plants (Fellner et al. 2015; Huber et al. 2016):

- Waste management of APC residues in the respective EU Member State (e.g. landfilling or export);

[127] Costs are calculated based on an output of 15 MW (Renewable Energy World 2014).

- Gate fees of landfills;
- Production costs of neutralisation agents; and
- Air pollution control systems (type and new technologies).

For instance, the costs for disposal in hazardous landfills or the stabilisation of residues with cement range between 200 and 250 € per tonne of fly ash (Astrup 2008; Fellner et al. 2015). According to Quina et al. (2018), the landfilling costs for APC residues or fly ash might be even higher and range between 150€ and 500 € per tonne.

Energy efficiency
Compared to MSW, PVC waste reveals a higher calorific value and therefore the incineration of PVC produces more energy than the incineration of MSW (European Commission 2000). The average heating value of flexible PVC is ca. 20 MJ/kg compared to 16 MJ/kg for rigid PVC and 10 MJ/kg for MSW (Menke, Fiedler, and Zwahr 2003).
While elevated inputs of PVC waste in the incineration plants could theoretically increase the generation of energy, high acid concentrations in the raw gas and the associated risks in terms of corrosion limit the operating parameters (steam temperature and steam pressure) (Brunner and Rechberger 2015). In order to prevent corrosion, a relatively low steam pressure must be maintained (ca. 40 bar) and the temperature in the heat recovery boiler must be kept below 300°C, reducing the energy efficiency (Bernard, Hjelmar, and Jürgen 2000; Sadat-Shojai and Bakhshandeh 2011).

Consequently, energy efficiency of waste-to-energy plants is considerably lower than the energy efficiency of conventional thermal power plants as they cannot recover the total calorific value of the waste materials (Brunner and Rechberger 2015; Sadat-Shojai and Bakhshandeh 2011). Although chlorine from PVC limits the steam parameters, the reduced energy efficiency of MSW incineration plants compared to conventional thermal power plants can rather be attributed to the variety of waste inputs in MSW incineration plants than to the presence of chlorine (CEWEP Focus Group Disposal 2021). Thus, according to CEWEP, the energy efficiency of MSW incineration plants would not be considerably higher if PVC was absent in the waste stream (Focus Group Disposal 2021)).

Maintenance costs due to corrosion processes
Various studies have proven the potential of volatile chlorine compounds for corrosion of boiler equipment (Persson et al. 2007; Pronobis 2020). For instance, Persson et al. (2007) investigated the impacts of adding PVC to the fuel mix of a MSW incineration plant in Sweden and found that the corrosion rate significantly increased due to the enriched chlorine content. Accordingly, chlorine compounds (mainly HCl) might destroy protective oxide layers surrounding the metallic surfaces of the equipment and cause severe corrosion of the steel itself (Lu et al. 2017; Pronobis 2020). The corrosion can significantly reduce the lifetime of boiler equipment and thus increase maintenance costs (Buekens and Cen 2011; Lu et al. 2017). This needs to be considered for evaluating the incineration of PVC in MSW plants from an economic perspective. The costs associated with the corrosion problems might account for around 5% of the total cost of a plant (Lee, Themelis, and Castaldi 2007). As such, it can be concluded that considerable additional costs result from the increased risk of corrosion due to the presence of PVC in the feedstock of incineration facilities.

Associated impacts on human health
Exhaust gases consisting of gaseous and particle emissions from incineration facilities require treatment because they might contain harmful and hazardous substances (Bilitewski, Wagner, and Reichenbach 2018). Therefore, the treatment and cleaning of exhaust gases as well as an

optimal management and control of the incineration process are of major importance for preventing or reducing potentially harmful emissions (Bilitewski, Wagner, and Reichenbach 2018). Concerning emissions to air, MSW incineration facilities can be considered clean technologies for waste treatment since health risks from flue gases can be avoided by using proper and contemporary cleaning technologies (Bilitewski, Wagner, and Reichenbach 2018). Accordingly, as regards air pollution, PVC incineration in MSW incineration plants does not represent a threat to human health if the plants are equipped with state-of-the-art cleaning technologies (Bilitewski, Wagner, and Reichenbach 2018).[128]

An empirical study investigating the long-term environmental levels of dioxins and furans in the vicinity of a MSW incineration facility demonstrated that impacts of dioxins and furans emitted by the MSW incineration facility on human health are negligible compared to the health risks associated with the dietary intake of dioxins and furans (Schuhmacher and Domingo 2006). These findings are in line with another empirical study by Vilavert et al. (2015) at a MSW incineration facility in Spain.

On the other hand, the treatment and cleaning of exhaust gases produces gas cleaning residues containing hazardous and leachable substances (e.g. heavy metals, dioxins and furans). The gas treatment residues require proper aftercare and safe disposal in order to minimize the risks for releases of harmful substances into the environment through leaks in soil, water or air (Bilitewski, Wagner, and Reichenbach 2018; Fellner et al. 2015).

Environmental concerns as well as impacts on human health exist regarding the leaching of heavy metals, soluble salts and dioxins and furans from air pollution control residues and fly ash as described in chapter 6.3.1.7.

A further negative impact on human health is associated with potential diffuse emissions of heavy metals during pre-treatment and recycling processes of fly ash (Zhou et al. 2015).

6.4.1.4 Summary

The improvement of air pollution control technologies and the effective treatment and cleaning of exhaust gases has significantly decreased air pollution from MSW incineration plants. Thus, nowadays, state-of-the-art MSW incineration plants are able to meet current emission level standards (Bilitewski, Wagner, and Reichenbach 2018; Brunner and Rechberger 2015).[129] In terms of PVC incineration this applies particularly to emissions of dioxins and furans, heavy metals and hydrochloric acid. On the other hand, environmental concerns exist regarding elevated emissions of dioxins during transient phases even in modern incineration plants (ToxicoWatch 2018). The input of PVC increases the risks of dioxin and furan emissions if optimal operation conditions are not met. Consequently, further research should focus on the degree to which MSW incineration plants are able to ensure adequate operating conditions and on the emissions of dioxins and furans from MSW incineration plants during OTNOC situations.

In addition, the analysis of economic aspects reveals that the input of PVC in incineration facilities with and without energy recovery also involves hidden costs stemming for instance from the production of neutralisation agents and the disposal of solid residues. Furthermore, considerable additional costs result from the increased risk of corrosion due to the presence of PVC in the feedstock of incineration facilities. The impacts of PVC on the energy efficiency of incineration plants with energy recovery can be considered limited.

[128] It should be noted that the requirements of the BAT conclusions adopted under the Industrial Emissions Directive (IED) including the emission levels associated with the use of the best available techniques (BAT-AELs) are legally binding for all European facilities under the IED and must be implemented at least four years after the publication of the BAT conclusions. However, it cannot be assumed that all incineration facilities are state-of-the-art (see chapter 6.4.1.1).

[129] However, it cannot be assumed that all incineration facilities are state-of-the-art (see chapter 6.4.1.1). The decree to which incineration facilities are equipped with state-of-the-art technologies should therefore be subject to further investigation.

In conclusion, the development of air pollution control technologies has moved the environmental focus from air pollution to the solid incineration residues (Astrup 2008; M. J. Quina, Bordado, and Quinta-Ferreira 2008). This also applies to the concerns about human health impacts. Thus, management options for MSW incineration residues, especially for APC residues and fly ashes become more relevant (Zacco et al. 2014). These options will be discussed in section 6.4.1.5.

6.4.1.5 Management of APC residues and fly ash

In this section, we will give a brief overview of the existing technologies, initiatives, and processes for treating fly ash and APC residues since they are assumed to be more relevant in terms of environmental risks and impacts on human health.

In general, there are three different routes for the waste management of APC residues and fly ash, including (Astrup 2008):

- the utilization as aggregates (e.g. backfilling of mines, utilization in asphalt production, application for neutralization purposes)
- the recovery of materials (salts, acids gypsum, metals)
- the disposal through landfilling (in underground and surface level landfills)

All of the above options require treatment and/or stabilization to a certain extent (Astrup 2008)[130].

In comparison to bottom ash, attempts for resource recovery from APC residues and fly ash have been barely considered and exist only in a few countries (Astrup 2008; Fellner et al. 2015). However, due to the increasing costs of landfilling, research on recovery of MSW incineration residues has increased (Zacco et al. 2014). Astrup (2008) estimates that globally around 20-30 technologies for the management of APC residues and fly ash exist. Some of the most relevant are described in **Annex 5.4**.

Each of the management options is associated with certain advantages and disadvantages. According to Hjelmar et al. (2009), treatment options combining the extraction of soluble salts, chemical stabilisation and the destruction of dioxins and furans through the recirculation of the stabilized material into the incinerator are most promising due to their efficiency and the development and status of the applied technologies.

Thermal treatment processes require high amounts of energy and might therefore potentially perform worse from a lifecycle perspective (Fruergaard, Hyks, and Astrup 2010; Hjelmar et al. 2009). Based on a LCA on management options for APC residues, Fruergaard, Hyks, and Astrup (2010) highlight the need for developing treatment methods with low energy demands and minimum leaching. The authors conclude that, considering data uncertainties, backfilling in salt mines as done in Germany or the neutralization of acid waste (see Langøya, Annex 5.4) are among the most suitable management options for APC residues from PVC incineration. Disposal of APC residues in German salt mines or on the Norwegian island of Langøya are very competitive solutions and are therefore applied in many EU countries like Sweden, Denmark, the Netherlands and also non-EU countries such as Switzerland (Hjelmar et al. 2009; Recupero, Robb, and Vahk 2019; Schlumberger 2019).

On the other hand, concerns and data gaps exist regarding the environmental implications of these management options, especially in the long term (Hjelmar et al. 2009). Fruergaard, Hyks, and Astrup (2010) point out that the backfilling in salt mines and the neutralisation of acid waste in Langøya are considered to have no long-term leaching. They further highlight that this

[130] More detailed information on the different treatment techniques is for instance provided by Astrup (2008), Amutha Rani et al. (2008) or Huber, Laner, and Fellner (2018)

conclusion is probably drawn based on an assumption of geological stability in these areas which might be questionable in the very long-term future (Fruergaard, Hyks, and Astrup 2010). The following table (Table 6-2) gives a brief overview of selected technologies, the respective status of implementation and the costs associated with the treatment of APC residues. The information is mainly based on the studies of Astrup (2008); Hjelmar et al. (2009) and Vinylplus (2017b) as well as input from the Focus Group Disposal (2021).

Table 6-6: Overview of selected technologies for the management of APC residues and fly ash from MSW incineration

Technology	Commercial availability of the technology	Estimated treatment costs [€/tonne]
NEUTREC, Solvay S.A.	commercially available	n.a.
SUEZ	commercially available	n.a.
HALOSEP	not commercially available, pilot plant	n.a.
FLUWA	commercially available	150 – 250
NOAH, Langøya	commercially available	50
INERTEC	commercially available	200 – 220
DRH	commercially available	n.a.
Integrated LAB ash Treatment process	commercially available	n.a.
WesPhix	commercially available	15 – 25
INDAVER	n.a.[131]	n.a.
Backfilling of mines	commercially available	70 – 100

6.4.2 Municipal waste incineration plants (without energy recovery)

MSW incineration without energy recovery is of minor importance for the treatment of MSW and PVC in the EU (CEWEP in Focus Group Disposal 2021)[132]. Therefore, the environmental and economic aspects as well as impacts on human health are not further discussed individually within this chapter. However, most of the information provided in chapter 6.3.1 also apply to MSW incineration without energy-recovery. This mainly concerns the

- environmental impacts;
- impacts on human health;
- economic aspects; and
- management of solid residues from incineration.[133]

6.4.3 Co-incineration in cement plants (and blast furnaces)

Co-incineration in cement kilns and blast furnaces represents another option for the incineration of PVC-waste. Waste or waste-derived fuels containing PVC are co-incinerated in cement kilns in order to reduce the consumption of fossil fuels and other primary energy sources (Bilitewski,

[131] Indaver's Indachlor plant in Dunkirk will recycle production waste and chlorinated waste streams for recovery HCl. The plant received the first delivery of wastes, however no specific information on the further status of the project could be identified. (Indaver 2020) (EURiTS FocusGroup Disposal 2021)

[132] For assessing the relevance of MSW incineration with and without energy recovery, Eurostat data for the year 2018 (EU27) was analysed. To this end, the shares of the waste stream "Total waste" (hazardous and non-hazardous) directed towards incineration without energy recovery (Disposal – incineration (D10)) and towards incineration with energy recovery (Recovery – energy recovery (R1)) were calculated. The result showed that only around 0.7 % of the waste is treated in incineration facilities without energy recovery, whereas 6.0 % are treated in incineration facilities with energy recovery. Therefore, the conclusion of CEWEP can be considered plausible.

[133] Of course, certain aspects such as the energy efficiency of MSW incineration plants with energy recovery do not apply to MSW incineration plants without energy recovery

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Wagner, and Reichenbach 2018; Zeschmar-Lahl, Schönberger, and Waltisberg 2020). To a lesser extent, plastic waste is also used in blast furnaces as a reducing agent for the production of iron (Bilitewski, Wagner, and Reichenbach 2018). Consequently, the focus in the following chapters will be rather on cement kilns than on other co-incineration facilities such as blast furnaces or lime kilns.

Waste streams treated in cement kilns need to fulfil certain criteria due their potential impacts on environmental, operational as well as health and safety aspects (Hinkel et al. 2020). When co-incinerating waste-derived materials containing PVC in cement kilns, the most relevant properties concern (Hinkel et al. 2020):

- The calorific value;
- The chlorine contents; and
- The concentration of heavy metals.

As large parts of the waste material co-incinerated in cement kilns eventually end up in the clinker, the chlorine input into cement kilns is limited due to specific requirements of the cement produced. Accordingly, the concentration of chloride in cement is limited to 0.10% to prevent alteration of the product qualities (Bilitewski, Wagner, and Reichenbach 2018; Seo et al. 2019). Thus, certain product requirements need to be fulfilled and a low chlorine concentration is a pre-requirement for promoting the substitution of fossil fuels by alternative waste derived fuels (Gerassimidou et al. 2020; Hinkel et al. 2020).

The cement industry is experienced in treating a broad range of waste derived fuels (Gerassimidou et al. 2020). Relevant waste streams containing PVC used as alternative fuels in the cement industry are construction and demolition waste, shredder residues from ELVs and WEEE as well as packaging and household waste (Focus Group Disposal 2021). Furthermore, hazardous waste is co-incinerated in cement kilns, however, the share of PVC in hazardous waste is estimated to be relatively small (EURiTS in Focus Group Disposal 2021).

The incineration of untreated MSW in co-incineration facilities is rather unlikely. Typically, waste streams burned in cement kilns undergo some kind of pre-treatment and do not end up in the feedstock untreated (EURiTSin Focus Group Disposal 2021). Hence, PVC reaches co-incineration facilities mainly in the form of refuse-derived fuels (RDF) (EURiTS in Focus Group Disposal 2021). Refuse-derived fuels can be understood as waste material that was pre-treated in order to achieve a defined composition which meets certain properties (Bilitewski, Wagner, and Reichenbach 2018; R. Sarc and Lorber 2013).

For instance, in some Member States (e.g. Austria or Germany; also Switzerland) RDFs have to meet certain quality criteria (R. Sarc and Lorber 2013). In Austria, PVC waste represents an unwanted material for the production of RDF due to its high chlorine content and is separated from the waste stream (R. L. Sarc 2016). A common quality standard concerning the use of RDF in cement plants is, that the chlorine concentration must not exceed 10 g/kg of dry matter (Bilitewski, Wagner, and Reichenbach 2018). Moreover, chlorine concentrations in various refuse-derived fuels from different waste types are in a range between 0.1 and 1.0 weight percent[134] (Neuwahl et al. 2019). According to the BREF for the Production of Cement, Lime and Magnesium Oxide, the acceptable chlorine levels depend on the individual plant and are typically in a range between <0.5 and 2 % (Schorcht et al. 2013). For avoiding operational problems in the kiln system, the chlorine concentrations are kept as low as possible (Schorcht et al. 2013).

[134] These concentrations refer to RDF-fractions treated in fluidised beds.

To sum up, the share of PVC co-incinerated in cement-kilns in streams of RDF or hazardous waste is considered to be minor (EURiTS in Focus Group Disposal 2021). However, that waste inputs in cement kilns are also an issue of governance, i.e. there might be differences between industry best practice and the actual waste inputs in cement kilns. Thus, unsorted or mixed waste might be treated in cement kilns, as an example in Romania shows (Global Alliance for Incinerator Alternatives in Focus Group Disposal 2021).

The substitution rates of alternative fuels in cement-kilns can vary considerably. In some kilns, substitution rates of 100% have been reached, whereas others reveal significantly lower substitution rates due to the local waste market situation and permit conditions (Schneider et al. 2011).

Thermal processes in cement kilns are characterized by high temperatures and long residence times due to the length of the kiln (Hinkel et al. 2020; Tukker et al. 1999). In this way, an efficient and complete destruction of waste-derived fuels can be achieved (Ecofys 2016). The process for co-incineration of waste in cement kilns is described in **Annex 5.5.**

6.4.3.1 Associated environmental impacts

Emissions to air are the most relevant environmental concerns associated with cement kilns (Hinkel et al. 2020). In general, formation of toxic gases is avoided due to the high temperatures in the cement kilns (~2000°C) (Bilitewski, Wagner, and Reichenbach 2018). Furthermore, large parts of the waste input get incorporated and bound in the clinker during the reactions taking place at 1450°C (Bilitewski, Wagner, and Reichenbach 2018). Within this study, only the specific impact of PVC co-incineration on emissions from cement kilns has been investigated.

<u>Chlorine content</u>
In contrast to MSW incineration facilities, cement kilns do not require additional scrubber systems for the removal of HCl (Scheirs 2003). This is due to the presence of large quantities of lime in the clinker process which effectively neutralises hydrochloric or hydrofluoric acid (Scheirs 2003).

<u>Dioxins and furans</u>
Industrial sources such as cement-kilns might represent potential sources for emissions of dioxins and furans (Nzihou et al. 2019). The mechanisms for the formation of dioxins and furans during combustion in cement-kilns are similar to the formation mechanisms during MSW incineration (see chapter 6.3.1) (Karstensen 2008). Although there are still uncertainties regarding the formation of dioxins and furans in cement kilns, surface-catalysed reactions and de novo synthesis in the preheater and post-preheater seem to be the most relevant formation mechanisms (Karstensen 2008).

Therefore, the exhaust gases of the kilns need to be cooled down quickly to temperatures lower than 200°C in order to avoid dioxin and furan emissions. This rapid cooling process is state-of-the-art in most modern preheater and precalciner kilns (Hinkel et al. 2020). Consequently, cement kilns usually are able to meet emission levels of PCDDs and PCDFs (< 0.1 ng TEQ/Nm^3 at 10% O_2) (Hinkel et al. 2020).[135]

Karstensen (2008) investigated emissions of PCDDs and PCDFs from more than 2000 cement-kiln measurements from around the world including European countries using different technologies and waste feeding scenarios and concluded that most modern cement-kilns co-incinerating waste

[135] It should be noted that the requirements of the BAT conclusions adopted under the Industrial Emissions Directive (IED) including the emission levels associated with the use of the best available techniques (BAT-AELs) are legally binding for all facilities under the IED in the EU and must be implemented at least four years after the publication.

are able to meet current emission levels through proper management and operation of the facility. Furthermore, Karstensen (2008) pointed out that a proper and responsible co-incineration of waste does not seem to promote the formation of dioxins and furans. In this regard, controlling the temperature of the air pollution control device and preventing elevated concentrations of chlorine in the feedstock are suitable methods for reducing dioxin and furan emissions (Hinkel et al. 2020; Karstensen 2008). Moreover, according to a study of Rovira et al. (2010), the substitution of fossil fuels by refuse derived fuels (with an average substitution rate of 15%) did not have a significant impact on the environmental levels of PCDD/Fs in the vicinity of a cement plant in Alcanar, Spain. This is in line with the findings of Ames et al. (2012) who investigated emission profiles of dioxins and furans from a cement kiln using hazardous waste as an alternative fuel.

In conclusion, for preventing emissions of PCDD/Fs from co-incineration of waste in cement kilns, input materials, including raw materials and alternative fuels, should be selected carefully especially with regard to the presence of chlorine (Hinkel et al. 2020). The PCDD/F emissions of most European cement kilns are below the limit value of 0.1 ng I - TEQ/Nm^3 if primary techniques are applied. One of the primary techniques is to ensure a waste input with a low chlorine concentration. Typically, chlorine concentrations range from <0.5 – 2 % (Schorcht et al. 2013).[136]

<u>Heavy metals</u>

Heavy metals present in the fuels used in cement kilns are oxidised and incorporated into the clinker as silicates or ferrites through chemical binding during the clinker reactions at 1,450°C (Bilitewski, Wagner, and Reichenbach 2018; Martens and Goldmann 2016). Small quantities of volatile metal chlorides which might enter the flue gas can be absorbed by the air pollution control technologies (Martens and Goldmann 2016). However, the majority of heavy metals enters the kilns through the cement raw material rather than the waste derived fuels (Martens and Goldmann 2016).

Emphasis should be given to the mercury content of alternative fuels. Because of its volatile nature, mercury forms gaseous compounds and is not bound in the clinker and retained in the kiln system (Hinkel et al. 2020). PVC in the feedstock of cement kilns does not contribute to the input of mercury in the cement kilns (ECVM in Focus Group Disposal 2021). According to B. Zhang et al. (2018), chlorine in the feedstock might promote the volatilization of heavy metals including mercury during clinkerization and therefore could potentially increase mercury emissions.

<u>Incineration residues</u>

Cement kilns produce cement kiln dust (CKD) as a by-product of the cement production (Seo et al. 2019). CKD is a by-bass dust with a very heterogenous chemical and physical composition (Maslehuddin et al. 2008). It consists of fine powdery solids and highly alkaline particulate material collected in the control devices such as bag filters, cyclones, or electrostatic precipitators during the clinker production process (Kunal, Siddique, and Rajor 2012; Seo et al. 2019).
The use of alternative fuels including PVC has increased the chlorine input into cement kilns. As elevated inputs of chlorine might cause operational problems in the cement kilns, co-incineration of alternative fuels containing PVC increased the need for effective chlorine bypass systems (Hinkel et al. 2020; Seo et al. 2019). The volatile chlorine can primarily be found in the fine particles of the CKD, also known as Cl bypass dust (BPD) (Gerassimidou et al. 2020; Hinkel et al. 2020). While larger particles can be recirculated to the kiln, BPD is often treated as waste and is disposed of in landfills (Gerassimidou et al. 2020; Kunal, Siddique, and Rajor 2012; Stevulova et al. 2021).

[136] Of course, besides the chlorine content, other aspects and techniques need to be taken into consideration for minimising PCDD/F emissions from cement kilns (see (Schorcht et al. 2013))

High concentrations of chlorine in the CKD are problematic as they prevent its recirculation into the clinker process (Maslehuddin et al. 2008; Seo et al. 2019). Moreover, the use of alternative fuels containing chlorine potentially enhances the volatilisation of heavy metals and thereby increases heavy metal concentrations in the CKD (Seo et al. 2019; B. Zhang et al. 2018). Therefore, environmental concerns exist regarding the leaching of hazardous compounds from landfilled CKD (Kunal, Siddique, and Rajor 2012). Besides the reuse in the clinker process and landfilling, alternative applications of CKD include its use in agriculture, civil engineering or waste treatment (Kunal, Siddique, and Rajor 2012; Maslehuddin et al. 2008).

6.4.3.2 Associated economic aspects

Operating costs and resource efficiency

The use of waste derived fuels in cement kilns might involve financial benefits for the operators due to decreased costs associated with the use of fossil fuels (Gerassimidou et al. 2020). Thus, operating costs of cement kilns can be decreased significantly by increasing the substitution rate (Zeschmar-Lahl, Schönberger, and Waltisberg 2020). However, at the same time, the quality requirements – especially the chlorine content – of the alternative fuels must be respected to prevent deterioration of the cement quality (Gerassimidou et al. 2020).

Corrosion

Similar to the incineration of PVC in MSW incinerators, the chloride present in the waste input to cement kilns promotes corrosion in the boiler and the passages for flue gases. Higher corrosion rates might lead to increased maintenance costs and raise the risks for breakdowns and the implementation of repair measures (Bilitewski, Wagner, and Reichenbach 2018).

6.4.3.3 Associated impacts on human health

The assessment of environmental impacts from co-incinerating PVC in cement kilns has demonstrated that potential threats to human health from emissions of dioxins and furans can be avoided, if operation of the kiln and the concentration of chlorine in the waste input is well-controlled. Information provided by the Global Alliance for Incinerator Alternatives in Focus Group Disposal (2021) suggests, that this might not always be the case across all EU Member States.

Exhaust gases of cement kilns are partly of worse quality than those of MSW incineration facilities, especially regarding the emissions of mercury, and therefore additional air pollution control technologies for the removal of heavy metals will be required in future (Bilitewski, Wagner, and Reichenbach 2018). Although PVC does not contribute to the mercury input in cement kilns, the chlorine present in PVC might increase the volatilisation of mercury from other alternative fuels (Wang et al. 2017; B. Zhang et al. 2018).

Similar to MSW incineration, the residues from cement kilns (CDK) represent a further environmental concern and the leaching of hazardous compounds from CKD disposed of in landfills or used in other applications might pose a threat to human health, e.g. through the leaching of heavy metals from landfilled CKD (Kunal, Siddique, and Rajor 2012).

6.4.3.4 Summary on co-incineration

The substitution of fossil fuels by alternative fuels is a widely used method and substitution rates might increase in the future. PVC enters co-incineration facilities mainly in the form of refuse-derived fuels or in other pre-treated waste streams. However, certain product requirements related to the chlorine concentrations need to be considered representing a limiting factor for the use of PVC as alternative fuel in cement kilns. Furthermore, the treatment of PVC in cement kilns leads to increased concentrations of heavy metals in the solid residues (CKD) of cement kilns and partly prevents the recirculation of residues into the process leading to environmental concerns and potential human health impacts regarding the leaching of heavy metals from solid residues. In addition, elevated levels of PVC in the feedstock should be avoided to prevent emissions of dioxins and furans. Similar to MSW incineration, ensuring adequate operating conditions is crucial for the prevention of dioxin and furan emissions. As regards economic impacts, the incineration of PVC contributes to higher maintenance costs due to corrosion, while potential cost savings through the substitution of fossil fuels are limited due to the product requirements in terms of the chlorine content. For this reason, the total input of PVC in co-incineration facilities is limited (see chapter 6.3.1.7).

6.4.4 Incineration in hazardous waste incineration facilities / hospital waste incineration facilities

According to the EU Waste Framework Directive 2008/98/EC, hazardous waste is defined as waste that displays one or more of the hazardous properties listed in Annex III of the Directive. This includes inter alia, carcinogenic, flammable, oxidising, mutagenic, toxic, or infectious properties. PVC-containing waste can be classified as hazardous waste or is present in hazardous waste streams primarily in the following cases (see also Annex 5.6):

- Infectious properties of medical waste containing PVC
- Incorporation of hazardous substances into PVC
- Homogenous streams of PVC
- PVC present in the packaging of hazardous waste

While in general, the sources of PVC in hazardous waste streams can be considered to be similar to those of MSW incineration (EURiTS in Focus Group Disposal 2021), according to EURiTS, specialised MSW incineration plants are able to treat waste with high chlorine concentrations (up to 40%). Typically, these plants recover HCl and are therefore interested in treating chlorine-rich waste (EURiTS in Focus Group Disposal 2021).

A potential – but not primary – source for PVC in hazardous waste streams is medical waste (EURiTS in Focus Group Disposal 2021). Different treatment options are relevant for medical waste containing PVC and PVC classified as or present in hazardous waste streams. These include (Bernard, Hjelmar, and Jürgen 2000; Gaudillat et al. 2018; Trozzi 2019; UBA 2015):

- Incineration in hazardous waste incinerators;
- Incineration in MSW incinerators[137];
- Incineration in small, decentralized on-site incineration facilities located at the hospitals (hospital incinerators); and
- Incineration in dedicated medical waste incineration facilities.

For the disposal of medical waste, most countries apply a combination of these treatment options (EURiTS in Focus Group Disposal 2021). However, de-centralised hospital incinerators are becoming less relevant and there is a tendency towards larger and centralized incineration facilities for the incineration of hospital waste (Trozzi 2019). Accordingly, the incineration of clinical waste has been prohibited in some European Member States such as Germany or Denmark, however, the incineration in hospital incinerators is still practised in other countries (Bernard, Hjelmar, and Jürgen 2000; Trozzi 2019) (PVC Information Council DK in Focus Group Disposal 2021). As MSW incineration has already been described in chapter 6.3.1, the focus in this chapter will be on hazardous waste and dedicated medical waste incinerators as well as on hospital waste incinerators.

[137] Treatment in MSW incineration facilities applies primarily to the non-hazardous shares from health care activities, i.e. household-like waste (Bilitewski, Wagner, and Reichenbach 2018; Gaudillat et al. 2018)

6.4.4.1 Description of relevant processing steps and characteristics

The combustion temperature in hazardous waste incinerators is higher than in MSW incineration plants. Hazardous waste incineration facilities are characterized by combustion temperatures ranging from 1,000°C to over 1,200°C and can treat higher shares of PVC (Block et al. 2015; Fråne et al. 2018). PVC incinerated as hazardous waste needs to be treated at temperatures of at least 1,100°C, however it is unknown how much PVC is incinerated as hazardous waste (Fråne et al. 2018). Typically, dedicated medical waste incinerators are operated like hazardous waste incinerators, while small hospital incinerators might lack adequate air pollution control systems (EURiTS in Focus Group Disposal 2021).

6.4.4.2 Associated environmental impacts

Because of the higher plastic content in medical waste compared to MSW, major concerns associated with medical waste incineration are potential emissions of dioxins and furans (Jiang, Li, and Yan 2019; Windfeld and Brooks 2015). According to Kulkarni (2019) hospital waste incinerators might represent a major source of dioxin emissions due to the burning of waste with high chlorine content. According to Block et al. (2015), emissions from hazardous waste incinerators are below the European emission limit values if the plants are equipped with state-of-the-art technologies.[138] Based on the hazardous waste incineration facility in Indavar, Belgium, Block et al. (2015) investigated the environmental performance of a typical modern incinerator for hazardous waste. They conclude that the contribution of hazardous waste incinerators to total European (EU27) air pollution are negligible. This is in line with the results of Bujak (2015), who assessed the emissions from the incineration of medical waste at a large complex of hospital facilities. The results indicate that the emissions to atmosphere were below the EU emission standards.

Mininni et al. (2006) investigated concentrations of PCDD/Fs in the flue gas of a full-scale hospital incinerator in Italy. Their findings indicate that emission standards at the stack (0.1 ng/Nm³) were exceeded, although the removal efficiency of the air pollution control technologies was higher than 90%. They conclude that the operating conditions are of major importance for the emissions of dioxins and furans.

As demonstrated above, there are various factors influencing the level of emissions from incineration facilities for hazardous and medical waste. These include the size and type of the plant, its operation as well as the applied pollution abatement technologies (Trozzi 2019).

In conclusion, the incineration of medical waste or hazardous waste containing PVC in hazardous waste incineration facilities or in large, dedicated medical waste incinerators does not represent a major source for emissions of dioxins and furans. Here, special attention needs to be paid to the operating conditions. In contrast, emissions of dioxins and furans from hospital incinerators might be more relevant (PVC Information Council DK in Focus Group Disposal 2021). For evaluating the total emissions of dioxins and furans (and other pollutants such as heavy metals and acids) from incineration of medical waste, further data on the number and emission levels of small on-site incineration facilities would be required.

Hazardous waste incinerators also produce bottom ash, fly ash and APC residues. Therefore, the environmental concerns regarding the leaching of contaminants as described in section 6.4.1.2 also apply to hazardous waste incineration facilities.

6.4.4.3 Associated economic aspects

Similar to MSW incineration, the costs for disposal of solid residues from the incineration of hazardous waste must be considered (see section 6.3.1.2).

[138] However, it cannot be assumed that all incineration facilities are state-of-the-art (see chapter 6.4.1.1). The decree to which incineration facilities are equipped with state-of-the-art technologies should therefore be subject to further investigation.

Moreover, the treatment of infectious or hazardous medical waste from hospitals involves high costs (PVC Information Council DK in Focus Group Disposal 2021). At the same time, results from hospitals analysing the hazardousness of generated waste show, that only a small share of waste is infectious (PVC Information Council DK in Focus Group Disposal 2021).

Thus, when analysing the costs associated with the disposal or other recovery of medical waste, it is important to differentiate between infectious and non-infectious fractions. Since the majority of medical waste generated is non-infectious, it can be treated like MSW and entails lower costs than when treated as infectious waste (Windfeld and Brooks 2015). Furthermore, recycling opportunities for medical waste are becoming more important as the new project Vinylplus Med[139] in Belgium is indicating (PVC Information Council DK in Focus Group Disposal 2021) (see section 4.2.4).

6.4.4.4 Associated impacts on human health

Similar to the environmental impacts, the impacts on human health from hazardous waste incinerators are mainly related to the leaching of contaminants from incineration residues. Regarding the emissions to air, Block et al. (2015) conclude that hazardous waste incinerators using state-of-the art emission control technologies are typically characterized by low emissions. Hence, the associated health risks for workers of or people living close to such plants are low. Also Nadal et al. (2019) investigated the exposure for the people living in vicinity of a hazardous waste incinerator to dioxins and furans. In the study, no additional and significant risks to human health could be identified.

Nevertheless, further information on the emissions of dioxins and furans from small on-site incineration facilities is required, as the uncontrolled burning of medical waste in incinerators without adequate pollution abatement technologies results in high emissions of dioxins and furans (Windfeld and Brooks 2015).

6.4.4.5 Summary

Environmental impacts from incinerating medical waste in hazardous and dedicated medical waste incineration facilities mainly are related to the solid residues. Environmental and human health concerns exist regarding the air emissions of small on-site hospital incinerators. Therefore, further research and data on the number and emission levels from hospital incinerators are required for evaluating the environmental and human health impacts. With respect to the economic aspects, two factors are important: The share of medical waste considered infectious or non-infectious and the increasing efforts for recycling PVC of medical waste.

6.5 Landfilling: description and scoping of the major disposal routes by in the EU

6.5.1.1 Description of relevant landfill types

As outlined in chapter 6.1.3, there are three different categories of landfills according to Article 4 of the European Landfill Directive[140]:

- landfills for hazardous waste;
- landfills for non-hazardous waste and
- landfills for inert waste.

According to input from the Focus Group Disposal 2021, all types of landfills are relevant for the disposal of PVC). It is assumed that the largest share of PVC waste is landfilled in landfills for non-hazardous waste (input from Focus Group Disposal 2021). Smaller amounts of PVC are disposed

[139] https://vinylplus.eu/community/vinylplus-med

[140] https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:31999L0031&from=EN

of in landfills for inert waste. This mainly applies to PVC present in C&D waste classified as inert waste (ECVM in Focus Group Disposal 2021). For instance, in Denmark, parts of the shredder residues from ELVs are disposed of at landfills as hazardous waste (Fråne et al. 2018). In this context, Miliute-Plepiene et al. (2021) point out that Denmark is the only country among the Nordic countries, where landfilling of flexible PVC is legal. However, PVC-containing waste classified as hazardous waste is disposed of by incineration rather than landfilling (ECVM in Focus Group Disposal 2021)[141]. Therefore, the amounts of PVC disposed of in landfills for hazardous waste are expected to be low (ECVM in Focus Group Disposal 2021). In this regard it should be noted that in 2009, 3,286 non-hazardous landfills, 60 hazardous landfills and 666 inert landfills within the EU did not comply with the Landfill Directive (Watkins 2015).

Furthermore, landfills for hazardous waste are used for disposal of APC residues and fly ash from the incineration of PVC in MSW incineration facilities (Vehlow 2012). Also, bottom ash can be disposed of in non-hazardous waste landfills[142] (Huber et al. 2016).

6.5.1.2 Associated environmental impacts

In 2000, the main concerns regarding the environmental impacts of landfilling PVC waste at that time according to the EU Green Paper on PVC were (European Commission 2000):

- The leachate of plasticisers, especially phthalates and their degradation products from flexible PVC;
- The accumulation of long-chain phthalates (e.g. DEHPs) on suspended solids;
- The losses of phthalates and their potential contribution to gaseous landfill emissions
- The release of lead stabilizers from flexible PVC; and
- Potential formation of dioxins and furans during accidental landfill fires.

According to the European Commission (2000), the polymer matrix was regarded as being resistant under landfill conditions and migrations of stabilizers in rigid PVC waste were considered to be negligible. Therefore, the following chapters investigate the environmental impacts of PVC in landfills taking into consideration, where available, the current scientific literature. The screening of available literature revealed that several studies focusing on the environmental impacts from landfilling PVC were published at the beginning of the 2000s, e.g. Kreiβig et al. (2003); Mersiowsky (2002); Mersiowsky, Weller, and Ejlertsson (2001) as well as Scheirs (2003). Thus, there is a lack of recent publications focusing on the leaching of landfilled PVC waste.

<u>Degradation of the PVC polymer matrix</u>

According to Kreiβig et al. (2003), the behaviour of PVC polymer must be considered inert when taking into account surveyable time periods. Moreover, it does not contribute to gaseous emissions from the landfill (landfill gas). This is in line with the conclusions drawn from different studies focusing on landfill simulation experiments, which have concluded that the polymer matrix is stable and not subject to degradation under landfill conditions (Mersiowsky 2002; Scheirs 2003).

<u>Leachate of plasticisers</u>

Typical plasticisers used in PVC are butylbenzyl phthalate (BBzP), di(2-ethylhexyl) phthalate (DEHP), diisononyl phthalate (DiNP), and diisodecyl phthalate (DiDP), with DEHP constituting up to 80% of the plasticiser volume for PVC production (Hahladakis et al. 2018; Zota, Calafat, and Woodruff 2014). Since phthalates are not chemically bound in PVC products, they might be

[141] Please see section 2.4. for more information on when PVC is classified as hazardous.

[142] Typically, incineration facilities in Europe are equipped with metal recovery technologies (Šyc et al. 2020) and therefore it can be assumed that metallic components are recovered before bottom ash is disposed of in landfills.

released into the environment (Ciacci, Passarini, and Vassura 2017; Zota, Calafat, and Woodruff 2014).
For the environmental decomposition of phthalates, there are three major processes:
i) biodegradation, ii) photolysis and iii) hydrolysis (J. Huang et al. 2013). The losses of plasticisers depend on the external physical and chemical conditions and on the properties of the plasticizers themselves (J. Huang et al. 2013; Mersiowsky 2002).
Various studies have confirmed the potential leachate of plasticisers from flexible PVC (Kalmykova et al. 2013; Mersiowsky 2002; Potrykus and Milankov 2015; Scheirs 2003; Wowkonowicz and Kijeńska 2017). Phthalate diesters (PAEs) were detected in at least three leachate samples at three different landfills contributing between 5 and 20 % of the total pollutant mixture in leachate (Eggen, Moeder, and Arukwe 2010). In a more recent study, Wowkonowicz and Kijeńska (2017) analysed concentrations of plasticisers in samples of leachate from five municipal landfills and found that the following PAEs were present in at least one sample: Di(2-ethyl- hexyl) phthalate (DEHP), Diethyl phthalate (DEP), Dimethyl phthalate (DMP), Di-n-butyl phthalate (DBP), Di-isobutylphthalate (DIBP). Of all PAEs, DEHP was predominant being present in 65% in concentrations ranging up to 73.9 µg/L. Thus, Wowkonowicz and Kijeńska (2017) concluded, that particularly DEHP constitutes an ubiquitous component in landfill leachate.
A further study conducted by Kalmykova et al. (2013) detected high levels of DEHP in leachate of a Swedish landfill for household and industrial waste with a high organic content, which was decommissioned in 1976. They concluded that the DEHP emissions can mainly be traced back to the use of phthalates as plasticisers in PVC and that after more than 30 years of potential degradation, DEHP is still emitted in high quantities. This result underlines, that the degradation of phthalates occurs at low rates and that phthalates might be released over a long time period (Ciacci, Passarini, and Vassura 2017).

Leachate of lead stabilisers

As stated in chapter 2.5, the migration of metal stabilisers from PVC has been proven, however the extent of the leaching is in most cases unknown and should be determined via empirical data. According to Mersiowsky (2002) stabilisers are expected to be rather fixed within the PVC matrix, however the authors investigated PVC cables in a landfill simulation, in which a certain release of the lead stabiliser was detected. Relevant stabilisers are organotin stabilisers and heavy metals stabilisers (Mersiowsky 2002) (see also chapter 2.4.1 and Annex 2).
Mersiowsky (2002) concluded that there is no substantial leaching of heavy metals stabilisers or organotin compounds from PVC products and that concentrations of the latter substances in landfill leachate are in a similar range if PVC is absent. Potrykus and Milankov (2015) come to a similar conclusion. The results of their literature review[143] suggest that leaching of heavy metals (Cd, Pb) from rigid waste PVC (old pipes) is very low, since the heavy metals leach out during the first weeks of service life (new pipes). Therefore, in comparison to other wastes, PVC present in landfills only contributes marginally to emissions of heavy metals (Ciacci, Passarini, and Vassura 2017). It should be noted that the European Chemicals Agency (2017b) also indicates that landfilled PVC has a low potential for releases of lead from the PVC matrix. However it is also stated, that some release is expected over time and that the exposure of PVC pipes to UV radiation possibly promotes leaching of lead from PVC (European Chemicals Agency 2017b; Potrykus and Milankov 2015). In general, the leaching of metal stabilisers from plasticised PVC is much higher than from rigid PVC (Mercea et al. 2018).

Emissions of landfill gas

Kreiβig et al. (2003) studied the contribution of plasticisers (DINP) present in landfilled cable waste to landfill gas generation. They found the contribution to be insignificant. One tonne of PVC cable waste would result in landfill gas emissions of 31 m³, however the average expected share

[143] It should be noted that some of the studies reviewed by Potrykus and Milankov (2015) were funded by industry.

of PVC cable waste in landfilled waste is expected to be 0.15% and therefore its contribution to landfill gas formation is minor (Kreiβig et al. 2003).
This is in line with the findings of Mersiowsky (2002), who concluded that no significant emissions of landfill gases can be conclusively attributed to landfilled PVC or its additives. This also applies to vinyl chloride present in landfill gas (Mersiowsky 2002).
Therefore, emissions of carbon dioxide or methane from landfilling PVC can be considered minor. At the same time, it should also be noted that landfill fires can contribute to emissions of greenhouse gases from landfilled PVC.

Formation of dioxins and furans from landfill fires
Increasing numbers of landfill fires can be observed across several European countries such as Poland, Italy or Romania. According to Polish authorities, many landfill fires in Poland are set up deliberately which could be one of the reasons why landfill fires are an "unexpectedly common" (Vaverková 2019). Further information on the amount of landfill fires in the EU is provided in section 6.5.1. It should be noted that the uncontrolled burning of PVC during landfill fires represents a significant source of dioxin emissions (see section 6.5.2), which has up to now never been quantified and thus further efforts are required to prevent landfill fires (accidental and intentionally set).

6.5.1.3 Associated economic aspects

The costs associated with landfilling depend on landfill quality and landfill tax systems and differ significantly between EU Member States (Tukker et al. 1999) (Focus Group Disposal 2021). Besides existing bans on landfilling PVC, high landfilling costs lead to larger shares of PVC being directed to incineration or recycling (Fråne et al. 2018). In general, the amount of PVC disposed of in landfills is related to the associated landfilling costs. Thus, in Southern European countries, costs for landfilling are relatively low and therefore higher shares of PVC are landfilled[144] (ECVM and CEWEP in Focus Group Disposal 2021).

Only limited data and information on costs associated with landfilling specific streams of PVC waste could be identified. Date from EEA (2013) on the typical costs for legal landfilling of non-hazardous municipal waste reveals that significant differences exist between EU Member States. Accordingly, the overall charge including landfilling tax and gate fee varies ranges between 3 € and over 150 € per tonne. An overview of the costs in different EU Member States is provided in **Annex 5.7.**

6.5.1.4 Associated impacts on human health

Impacts on human health from landfilling PVC concern the leaching of plasticisers and stabilisers as well as the emissions of dioxins and furans from potential landfill fires (Eggen, Moeder, and Arukwe 2010; Jamarani et al. 2018; Lardjane, Belhaneche-Bensemra, and Massardier 2013). As the analysis of environmental impacts has shown, the most serious concern relates to the leaching of phthalates. These show persistent properties and might bioaccumulate in the environment resulting in human exposure through inhalation and ingestion as well as through ingestion, inhalation or dermal exposure (Lardjane, Belhaneche-Bensemra, and Massardier 2013; Ramakrishnan et al. 2015).

6.5.1.5 Summary

PVC from various waste streams is disposed of in all types of landfill types. Based on expert opinions, the disposal in landfills for non-hazardous waste is most common (Focus Group Disposal 2021). However, landfills for hazardous waste need to be taken into account for the disposal of PVC – for instance, when PVC contains high concentrations of phthalates or heavy metals due to

[144] No further information on concrete shares could be identified.

the use of certain additives – and incineration residues. Quantitative data on the volumes of PVC disposed of as hazardous nor non-hazardous waste in landfills could not be identified.

As indicated in section 4.2, a relevant question for the selection of the correct landfill option for PVC waste concerns the classification of such waste as hazardous or non-hazardous. The analysis in section 4.2 indicates that, in the case of PVC, such classification will likely depend on the extent to which migration and bioavailability can or cannot be established for relevant additives contained in a relevant PVC waste stream. In this regard, it is important to note that it was concluded in chapter 2 of this report that various literature sources indicate that migration of additives from plastics can take place, since such additives are not covalently bound in the plastic matrix. However, the analysis in chapter 2 also indicates that, currently, data on migration and bioavailability of PVC additives is limited. From a precautionary perspective, a more critical assessment of the classification of PVC as hazardous or non-hazardous may be necessary for the determination of the most suitable landfilling option.

With respect to the environmental aspects of already landfilled PVC, the major environmental and human health impact concerns the leaching of phthalates, which might be released over a long time period. Data suggests that only limited amounts of stabilisers might be released from rigid PVC, while leaching of small quantities of plasticisers from flexible PVC might occur over a long-term period. However, based on the available data, it cannot be concluded whether other additives might or might not migrate from the PVC matrix. Bearing in mind the above point on phthalates, flexible PVC should rather not be disposed of in landfills.
Furthermore, landfilled PVC may increase the chance of emissions of dioxins from accidental and intentionally set landfill fires (see also section 6.6.2.1).
In general, the amount of PVC disposed of in landfills is expected to decrease with increasing separate collection of PVC (Global Alliance for Incinerator Alternatives in Focus Group Disposal 2021). At the same time, it should be noted that despite existing landfilling bans in some EU Member States, considerable quantities of PVC in the EU are still disposed of by landfilling (CEWEP in Focus Group Disposal 2021).

6.6 Description and scoping of illegal disposal

Waste (and thus PVC waste as well) is not always treated or disposed of properly and in accordance with the law. The motivation behind illegal waste treatment (especially illegal disposal and incineration under uncontrolled conditions) is primarily financial (e.g. to avoid disposal fees, licensing fees, recycling costs, taxes) (INTERPOL 2020). Illegal disposal means the non-authorised deposit of waste onto land. This includes unauthorised disposal on a licensed site or the disposal on a non-authorised site which can be private or public. Related activities might be fly tipping, burying, landfilling and dumping onto land but can also refer to dumping at sea. In order to reduce the volume of illegal dumping sites, illegal waste fires can be set. These can also occur accidentally or in an uncontrolled manner (due to unsafe conditions) (INTERPOL 2020).
Incineration of waste on a non-authorised site and under uncontrolled conditions is defined as illegal incineration. Further illegal incineration includes the incineration on an authorised site of improper waste types or waste types that are improper for thermal treatment (INTERPOL 2020).

Illegal recycling (recycling by a non-licensed facility; recycling of unproper waste) is another type of illegal waste treatment that this chapter does not focus on.
Illegal waste treatment is highly linked to illegal waste trade (see section 6.6), meaning transboundary movement of waste not complying with export and/or import regulations (INTERPOL 2020).

6.6.1 Quantitative numbers and relevance of illegal disposal

It should be noted that since no specific quantitative data on illegal treatment of PVC waste exist, this chapter presents data on plastic waste in general (mixed plastic waste including about ~10% of PVC).

Uncontrolled burning of waste or dumping waste at non-authorised landfills is prohibited in the EU. Consequently, quantitative data on illegal activities or resulting emissions are rare.

According to INTERPOL (The International Criminal Police Organization) (2020) an increase of illegal trade and illegal treatment since 2018 with regard to plastic waste has been noted (2018 China implemented new imports restrictions of 24 types of solid waste including plastic waste). It concerns both, plastic waste export countries (e.g. UK, Germany, Italy, …) as well as (emerging) import countries (e.g. Brazil Malaysia, Cambodia).

Illegal landfilling is not a new phenomenon in European countries, but in export countries a significant increase in waste disposal in illegal landfills of plastic waste destinated for recycling has been noticed since 2018. Same applies for irregular waste fires (accidental or deliberate).

This increase of illegal waste treatment is a reaction to the large volumes of untreated waste that would have been exported to China. For example, in Ireland 95% of the domestic recyclable plastic waste used to be sent to China. In some cases, the cement industry is used for the co-incineration of this waste (including partly illegally imported waste).

In emerging (Asian) import countries the increase in plastic waste imports lead to an increase of illegal recycling activities and illegal landfills (INTERPOL 2020). Further, accidental but also deliberate waste fires have probably increased.

Fires (in open fields, factories, recycling plants, stockpiling houses, at disposal facilities) are usually reported as accidental. Nevertheless, the study by INTERPOL (2020), suggests that some of the fires in e.g. Italy, Netherlands, Poland, Romania, Malaysia and Thailand were started on purpose to eliminate waste that was illegally landfilled. In Southern Europe, organized crime business models of irregular and unsafe waste disposal are seen as one driver of the rising fires. Especially Italy reported an increase in waste fires and an increase of the waste amounts that have been burned. In Spain especially fires in landfills and waste treatment centers have increased; An increase of 100% is estimated between 2017 und 2018. Spain used to export ca. 60% of plastic waste for recycling to China.

Illegal landfills exist in all EU Member States. Analysing illegal landfills is problematic due to missing data which leads to data gaps with regard to characteristics, spatial distribution and number of illegal landfills (Jordá-Borrell, Ruiz-Rodríguez, and Lucendo-Monedero 2014). It is assumed that each year in the EU-28 2.8 million tonnes of waste are illegally dumped (Watkins 2015). Based on the results of the report from INTERPOL these numbers might have increased since 2018. A case study for an Italian Region Campania identified a high number of Potentially Contaminated Sites (PCSs), whereas 74% can be lead back to waste disposal (potentially (suspected but not verified) contaminated area especially based on the disposal and uncontrolled incineration of toxic waste since the 1980s) (D'Alisa et al. 2015).

In general, illegal plastic waste treatment is reported to have increased especially in the following countries[145] (INTERPOL 2020):

[145] "almost half (40%) of the countries that provided data to INTERPOL on the evolution of illegal waste treatment in their territories since 2018, reported an increase in such illegal activities" (INTERPOL 2020)

Table 6-7: Overview of increased illegal plastic waste treatment activities since 2018 (reported cases)

	Illegal recycling	Illegal landfills/ stockpiling	Waste fires	Illegal dumping
Ireland	x			
Sweden		x	x	x
France		x		
Spain			x	
Italy	x		x	x
Slovakia				x
Czech Republic		x		
Ireland	x			
Non-EU countries:				
Thailand	x			x
Malaysia	x			
Australia		x		
Chile		x	x	
Malawi				x

The same source reports that illegal waste treatment activities decreased in Hungary (only).

Highest rates for waste mismanagement have been reported for the following plastic waste import countries: India (87%), Indonesia (83%), Vietnam (88%) and Malaysia (57%) (INTERPOL 2020). Although England was not explicitly mentioned in the INTERPOL study, current studies show that illegal disposal is a big problem in England as well (ESA and Eunomia 2021; Purdy and Crocker 2021). Purdy and Crocker (2021) estimate that in 2019/2020 circa 1.82 million tonnes of waste have been fly-tipped on public land or put on illegal landfills. Thereby WEEE has a high share, so it can be assumed that this illegally disposed waste also includes PVC plastic waste. In the report of ESA and Eunomia (2021) the authors estimate the following cost of illegal disposal in 2018/2019 (costs for waste in general)[146]:

- Fly-tipping at £391 million,
- illegal waste sites at £236 million, and
- waste fires at £22 million

Experts at the Focus Group Disposal 2021 were not aware of any specific cases of illegal treatment of PVC, as illegal treatment mainly involves mixed or hazardous waste and is difficult to document in detail.

6.6.2 Environmental and health effects of illegal disposal

6.6.2.1 Uncontrolled combustion and open fires

While the incineration of PVC is rarely a problem in professional and authorised plants as regards air pollution (see section 6.3.1.2), it is problematic in uncontrolled fires like e.g. house fires, back-yard burning or landfill fires. Uncontrolled means without temperature control, consistent oxygen supply, turbulences and air pollution control (The Natural Step International 2018; M. Zhang et al. 2015).

[146] These listed costs together with costs of misclassification, exemption breaches, illegal exports and enforcement amount to total costs of waste crime of £924 million (ESA and Eunomia 2021)

Thermal decomposition of PVC generates hydrogen chloride, benzene as well as aromatics, for example. The forming of dioxins is linked to further transformation of secondary and tertiary products and depend on the formulation[147] of PVC and treatment conditions. Incomplete combustion promotes the formation of dioxins' precursors (halogens) (M. Zhang et al. 2015).

Major gaseous products are carbon monoxide, carbon dioxide and hydrogen chloride. Hydrogen chloride inhibits the conversion of carbon monoxide to carbon dioxide and can cause irritancy (see also section 5.2.3). According to Scheirs (2003) the mutagenicity of particulates from incinerating PVC is higher than from e.g. PS, PET and PE. A summary of generated compounds and their harmful effects during the incineration of PVC can be found in Nagy (2016).

Further it seems that PVC lowers the incineration temperature (due to its high chlorine content which is not flammable), which increases the residence time and results in an increase of dioxin emissions. In addition, wet and compacted waste containing PVC promotes dioxin formation (Scheirs 2003). Focus Group

The UNEP Toolkit "for Identification and Quantification of Dioxin and Furan Releases" includes the following emission factors of open burning and accidental fires activities (see table below) (UNEP 2005) (not PVC specific).

Table 6-8: Emission factors [-µgTEQ/t] for waste burning and accidental fires (UNEP 2005)

Classification	Air	Water	Land	Product	Residue
Landfill fires	1,000	ND	600	NA	[600]
Accidental fires in houses, factories	400	ND	400	NA	[400]
Uncontrolled domestic waste burning	300	ND	600	NA	[600]
Accidental fires in vehicles	94 (per vehicle)	ND	18 (per vehicle)	NA	[18 (per vehicle)]
Open burning of wood (construction/demolition)	60	ND	10	NA	[10]

It shows that emissions are not only occurring in air but also soil contamination might be a consequence.
With regard to uncontrolled combustion and open fires Zhang et al. (2015) differentiate between house fires, backyard burning and landfill fires and cable burning.

<u>Landfill fires</u>
Landfill fires can occur above ground as well as very often underground, sometimes as deep as 50 m. Due to their low temperatures of 80-230°C, which is far below the temperatures in MSW incinerators or professional combustion processes, landfill fires are more problematic: The risk of pyrolysis products or incomplete combustion and formation of dioxin is therefore much higher. Data from 1999 indicate that PVC pipes, rigid foils, flooring and cable wires contribute to 40% of the chlorine content, facilitating dioxin formation in landfills (M. Zhang et al. 2015).
Other aspects that facilitate the generation of fires and the generation of dioxins in landfills are the mixed composition of the waste, the poorly mixed compositions, lack of oxygen and moisture. Measurements from open burning at a municipal waste landfill shows dioxin emissions[148] that are five times higher than those from backyard burning and 2,000 times higher than those form

[147] E.g. chlorinated PVC and polyvinylidene-chloride show fire resistance as well as specific additives (M. Zhang et al. 2015); DEHP-plasticized PVC gives higher levels of PAHs than EDOS-plasticized PVC (Scheirs 2003).
[148] 202 to 1,700 ng TEQ kg-1 C burned, with an average of 823 ng TEQ kg-1 C burned,

modern MSW incinerators. Further Zhang et al. (2015) draw the attention to smouldering combustion which tends to generate more dioxins than flaming combustion.

House fires

Houses may contain many components or products made of PVC or containing PVC. House fires may not count as illegal incineration but data from house fires can still be applicable for the assessment of environmental and health effects of uncontrolled and open fires. And compared to dioxin limits for MSW incinerations the dioxin concentration of house fires is relatively high. On the other hand, house fires occur irregularly and rarely. Nevertheless, toxicity tests show that PVC decomposition products are not as toxic as other common building materials[149] (Ragaert, Delva, and Van Geem 2017). Based on samples of soot and ash from (simulated or accidental) house fires the annual generation of dioxins in the US as a result of PVC burning in house fires was estimated to be very low (0.47 to 23 g TEQ y^{-1}) compared to the annual deposition from the air (20–50 kg TEQ). But dioxins might not only be found in ashes or soot, so emissions might be higher in reality.

Other studies show that for example the potential to generate dioxins is similar for PVC and wood and the dioxin level is not always clearly related to PVC (M. Zhang et al. 2015).

Backyard burning

Burning household waste in an open fireplace or, for example, furnaces, wood stoves or open pits mainly occurs in rural areas. PVC and common salts contribute to high chlorine concentration. The consequence of burning household waste is the generation of toxic by-products (particle pollution, cancer-causing polycyclic aromatic hydrocarbons[150] and harmful volatile organic compounds) and dioxins which pose a great health risk especially since the emissions are released close to the ground (Johnsen 2015b; M. Zhang et al. 2015).

Zhang et al. (2015) present results of studies showing that burning waste with a higher PVC and copper mass fraction creates significantly higher emissions of HCl, chlorinated organics (dioxins and chlorobenzene) than burning mixed household waste with less PVC. Scheirs (2003) states that studies show that an increase from 0 to 1% of PVC in waste increases the dioxin emissions seven fold. Another result presented in the study of Zhang et al. is the effect that only a high chlorine content, which is atypical of household waste, is significant for the formation of dioxin emissions (M. Zhang et al. 2015).

Cable burning

Burning (PVC-insulated) cables is a common practice especially in poor parts of the world or developing countries to reclaim the copper. Although this is reported as not being practiced in the EU Member States currently[151], cable waste exported from the EU can be treated this way (input from ECVM and Global Alliance for Incinerator Alternatives at Focus Group Disposal 2021). Cable burning has a very high potential for dioxin formation: carbon (sheath), chlorine (PVC), and a catalyst (copper). Tests of burning PVC-based insulated wires (25% copper, 65% PVC) show dioxin emission values (11,900 ng TEQ kg^{-1}) 100 times higher than emission values of uncontrolled barrel burning of residential waste (M. Zhang et al. 2015) .

6.6.2.2 Illegal disposal

Potential environmental and health risks resulting from landfilling of PVC waste (see section 6.4.1.2) also applies for illegal disposal of PVC waste.

[149] which materials, is not mentioned in the report

[150] With a toxicity and carcinogenic potential that is many times higher than that of dioxins (Scheirs 2003)

[151] The industry realised that the value of the recovered copper/aluminium is lowered with this practice (ECVM FocusGroup Disposal 2021).

Illegal landfills as one type of illegal disposal occur often on forest margins, in ditches or on the peripheries of inhabited areas. Problems arising from uncontrolled waste disposal may include (Correa, de Santi, and Leclerc 2019; Vaverkova et al. 2019):

- Contaminated soil;
- Contaminated ground water with leaking chemicals;
- Change of vegetation (e.g. disturbance of native vegetation and enforcement of synanthropic species);
- Change of ecosystem functionality; and
- Land degradation.

Further, Triassi et al. (2015) and Macropoulos and Newman (2015) conclude in their studies that dumpsites present serious health effect However, both studies underline that there is a lack of systematic (long-term) epidemiological studies and a verified connection between health effects and unsafe disposal is not always possible. The related health risk depends on the composition of disposed waste and conditions of the area (Mavropoulos and Newman 2015; Triassi et al. 2015) (Triassi et al. 2015).

It should be noted that these effects cannot be ascribed specifically to PVC, but to illegal and uncontrolled waste disposal in general.

6.7 Transboundary movement of PVC waste

6.7.1 Legal transboundary movements[152]

"International trade has been an option for dealing with the growing amount of plastic waste in Europe, given the weakness of domestic economically viable post-collection treatment and recycling and the willingness of China and Asian countries, amongst others, to import waste." (D'Amato et al. 2019). Figures from Eurostat [env_wastrd] show that the EU exports large volumes of recyclable plastic waste. About half of the plastic waste collected in the EU is sent abroad for treatment (D'Amato et al. 2019).

Wagner and Schlummer (2020) mention that export of plastic waste occurs when a country does not have the necessary treatment infrastructure or when the treatment in the country of origin is not economically attractive. The latter might be the case in the EU as labour costs are higher than in other world regions. The authors criticise that although transboundary movement of plastics containing legacy additives (plastics like PVC) to developing regions is forbidden, still such waste is still imported to countries with less developed waste management systems.
The import-restrictions in China from 2018[153] changed the worldwide transboundary movement of plastic waste. Plastic waste exports from the EU to China and Hongkong decreased (drastically in the case of China), whereas Turkey and Indonesia recorded the highest increase (D'Amato et al. 2019).
In general, in the EU28+NO/CH there has been a decline of plastic waste exports: the share of plastic waste exported for recycling decreased by about 40% after beginning of 2018. This does not consequently mean that more plastic waste is being recycled within the EU. Illegal transboundary movements or higher incineration and landfilling rates might be another explanation (INTERPOL 2020).
With regards to pre-consumer PVC waste, clean waste (mainly from the construction sector) generated in Nordic countries is mainly exported for recycling, with some exemptions (small on-site recycling capacities) (Fråne et al. 2018). According to UN Comtrade Database[154] the EU-28

[152] Please see Annex 1.1 and the legal aspects of the Waste Shipment Regulation.
[153] import restrictions of 24 types of solid waste, including post-consumer plastics waste; 0.5% contamination limit for all other solid waste imports
[154] https://comtrade.un.org/data/

imported in 2020 5.647 tonnes of PVC waste, parings and scrap and exported 18.688 tonnes (worldwide). The import and export figures for the individual Member States are presented below.

Table 6-9: CE 391530 Vinyl chloride polymers: waste, parings and scrap (UN Comtrade Database[155])

Reporting country (2020)	Import [tonnes]	Export [tonnes]
Belgium	1,011	0
Bulgaria (2019)	*27*	*27*
Czech Republic	25,953	2,243
Denmark	0.17	1,098
Germany	26,339	7,632
Estonia (2019)	*No data*	*6*
Ireland	13	1,065
Greece	99	152
Spain	6,942	5,069
France (2019)	*27,319*	*0*
Croatia	29	335
Italy (2019)	877	1,179
Cyprus (2019)	*No data*	72
Latvia	116	87
Lithuania	227	214
Luxembourg	9,919	454
Hungary	312	581
Malta (2019)	*No data*	26
Netherlands	11,157	10,841
Austria (2019)	5,653	10,984
Poland (2019)	13,140	11,141
Portugal	1,068	2,017
Romania (2019)	4,456	733
Slovenia	562	985
Slovakia	2,532	3,576
Finland	0.157	1.6
Sweden (2019)	55	781
United Kingdom	0	31,062

Based on these data the biggest importers of PVC waste in absolute numbers are France, Germany, Czech Republic and the Netherlands; related to the population these are Luxembourg, Czech Republic and the Netherlands. The biggest exporters on the other hand are UK, Poland, Austria and the Netherlands. Related to the population the biggest exporters of PVC waste within the EU are Austria, Luxembourg and the Netherlands.
Which countries do import the exported PVC waste from the EU cannot be identified via the database of UN Comtrade (but see information from the study from Conversio Market & Strategy GmbH below).

[155] https://comtrade.un.org/data/

Experts at the Focus Group confirmed an intense movement within the EU, since for example smaller countries without recycling facilities for PVC waste have to export the waste to other EU Member States with appropriate recycling infrastructure. ECVM and VinylPlus are of the opinion that PVC waste is not imported into the EU from outside, as waste treatment in the EU is fundamentally more expensive than outside it (ECVM in Focus Group Disposal 2021; interview VinylPlus 2021)[156]. A study carried out by Consultic Marketing & Industrieberatung GmbH in 2016 on behalf of Recovinyl gathered data on export of PVC waste to non-EU countries including C&D waste, packaging, ELV, WEEE and others from Belgium, France, Germany, Italy, Luxembourg, Netherlands, Poland, Spain and the UK (Consultic Marketing & Industrieberatung GmbH 2016b). The study identified Germany with 34.4% and the UK with 21.2% as the biggest exporters among the investigated countries. The study concluded that in 2016 the (pre- and post-consumer) PVC waste from the specific EU Member States was mainly exported to China and Hong-Kong (due to the import restriction, this is no longer valid today). Further PVC waste was exported to India, Pakistan, Malaysia, Vietnam, Korea, and Morocco. Cables accounted for the largest shares (40%), followed by pipes and films (Consultic Marketing & Industrieberatung GmbH 2016b). More current data from Conversio Market & Strategy GmbH (2021) show, that in 2010 a total of 20 kt of PVC waste (especially post-consumer waste) generated in the EU27+3 was exported mainly to Pakistan, United States, Turkey and Malaysia.
In general, the main volume of traded PVC cable waste remains within the EU for recycling, mainly in Germany or the UK. Looking at export to non-EU countries of (mainly post-consumer) PVC cable waste the following picture occurs: 2016 Germany was the biggest exporter among the investigated countries with 31.6%, followed by UK with 19% and France with 12.7%. Cable export is an interesting business as metals from cables can be recovered. Although the PVC cable insulation in not in focus for the importing countries, it is assumed that the cable waste will be recycled in the importing countries. The main importing countries 2016 were Hong Kong, China and Morocco (due to the import restriction, this is no longer valid today) (Consultic Marketing & Industrieberatung GmbH 2016a). VinylPlus stated that in 2019, more PVC waste was available from cables particularly in the Czech Republic, France, Germany, Poland and the UK. It is estimated that this increase was due to reduced exports to China which imposed a ban on imports (Vinylplus 2020b).

6.7.2 Illegal transboundary movements

Illegal waste trade[157] in general means the transboundary movement of waste not complying with the export and/or import regulations. It includes illegal shipment of waste despite export restrictions by the origin country, import restrictions by the destination country or in violation of waste treatment requirements (INTERPOL 2020).
Since 2018 re-routing of illegal waste shipments to emerging import countries have been observed[158]. Shipments to South and South-East Asia and to a lesser extent several African countries (Ghana, Malawi and South Africa) and Eastern Europe were reported.
Malaysia and Thailand have become the countries with the biggest increase in illegal plastic waste imports (INTERPOL 2020).

Another trend concerns the misclassification of waste as “green listed plastic waste” among some Central and Eastern European destinations (especially Czech Republic and Romania). Exports mentioned (input Focus Group Disposal and Recylcing 2021) cases where PVC waste is classified

[156] One exception might be the USA: the US shipped PVC roofing membrane to the Vinyloop program in the EU (ECVM and Material Research L3C input FocusGroup Disposal 2021)

[157] The Basel Convention defines illegal „illegal traffic“ in waste as: a transboundary movement of hazardous wastes (http://www.basel.int/Implementation/LegalMatters/IllegalTraffic/Overview/tabid/3421/Default.aspx)

[158] reported by 60% of the participating countries

as cables for which the import is still authorised in Asia. Further, experts explain that transparency and data collection is difficult in general for waste shipments and so is the monitoring (input Focus Group Disposal 2021).

Specific data on transboundary movement for PVC is difficult to obtain. Not only the specific (illegal) trade routes but also publicly available data on the waste treatment options of exported PVC waste is scarce. With regard to the illegal transboundary movement, it should be noted that hazardous waste is often mixed with non-hazardous waste to disguise illegal export. This mixed plastic waste is then not suitable for recycling (D'Amato et al. 2019).

7. ANALYSIS OF ALTERNATIVES

Key Messages – Analysis of Alternatives to PVC

Scope and Approach

This section explores alternatives to PVC in a total of ten specific applications. These applications were selected from a longer list where PVC is used, based on several criteria:

- First, applications using larger volumes of PVC were prioritised.
- Second, an overall indication of *potential* risk and exposure was made. Detailed information on the specific additives that are used within each and every PVC application is not available. Consequently, no quantitative estimates of exposure and risk associated with different downstream uses has been possible. To provide an indication of potential, the following indicators were used to select applications for detailed analysis:
    - Indoor or outdoor usage: Uses where the final or end consumer comes into contact with the PVC product in interior spaces (I.e. inside the home, workplace) were prioritised.
    - Percentage of non-PVC substances: This was used as an indicator of the potential amount of additives present in typical formulations. Applications where the evidence suggested higher typical concentrations were prioritised. It is recognised these may have changed in line with ongoing regulation. The specific additives vary considerably, both in type and risk posed, therefore this indicator is simply a proxy for potential risk.
    - Flexibility: Flexible PVC formulations typically contain higher proportions of additives by weight; plasticisers in particular. Flexible PVC application were therefore prioritised.
    - Food/medical contact: PVC uses where there is the potential for contact via food or with patients were prioritised.
    - We also considered recyclability to take into account end of life impacts associated with different applications, and selected applications from a broad range of downstream sectors (including construction, electronics and medical) where possible.

Based on these selection criteria, the following applications were identified for inclusion in the analysis of alternatives: pipes and pipe fittings (waste, rain water and drinking water); cables; window frames; flooring; packaging; inflatable toys; boots and shoe soles; automotive interiors (dashboard and artificial leather) and medical applications.

The assessment approach follows ECHA's format for the analysis of alternatives (AoA) in applications for authorisation under REACH. As such, we examine technical feasibility, economic feasibility, comparative hazard and risk, and availability of alternatives to PVC identified in our key applications. The section was based on a review of secondary information supplemented with a targeted focus group of industry stakeholders, including some downstream users and manufacturers of alternatives to PVC, alongside targeted follow-up interviews. A balanced representation was sought.

Overall Messages

The analysis indicates that there are economically viable and technically feasible alternatives in the vast majority, if not all, applications assessed in detail where PVC is currently used. These are not without technical drawbacks. The extent of these drawbacks differs between applications. Additional costs are associated with their use in several cases. The alternatives identified fall into three groups:

- Alternative non-plastic materials (such as wood, leather, cloth etc)
- Alternative plastics (here a wider variety of plastics were identified; some of these contain additives which pose similar risks to those used in PVC and others which are the subject of potential regulatory action)
- More novel alternatives (such as bioplastics), about which limited application-specific information has been obtained. Some alternative plasticisers (i.e. phthalate free) were identified.

Key Messages – Analysis of Alternatives to PVC

The vast majority of the alternative materials and plastics identified are commercially available, often placed on the EU market in significant volumes at present. The balance of evidence suggests that the human health and environmental risks associated with a transition would decrease, although identifying net effects are challenging and differ between applications.

Application specific messages:

- In construction, there are a number of technically feasible and economically viable alternatives to PVC for several specific uses in drinking, waste and rain water **pipes and fittings.** These include alternative plastics and metals. It is likely several alternatives would need to be adopted, given different requirements for different piping systems. Costs for some alternatives are greater, but with comparable durability. In **window frames**, alternative materials are commercially available and in use. These are typically more expensive at the point of sale, often offset by longer lifetime of use. There are a range of technically and economically feasible alternative materials and plastics for commercial and residential uses in **flooring**, this includes alternative flooring in healthcare settings.

- In electronics, alternative **cable coatings and jackets** are available that are technically feasible – although it is recognised that specific requirements differ. The environmental and health risk reduction is less clear - all alternatives are plastics, some of which pose similar concerns to PVC, but which are the subject of potential regulatory action. Costs of alternatives differ.

- There are several different non plastic, plastic and emerging bioplastic alternatives in **packaging.** Consultee information indicates there has been a strong decrease in PVC use within the last decade.

- Alternative plastics are technically feasible for use in **children's toys and in inflatable boats**. These are typically both more expensive, but more durable. Plasticiser content has historically been comparatively high in this use. A 2018 restriction on several phthalates, with particular controls on children's toys, could be expected to decrease impacts associated with a group of additives of particular concern.

- Alternative materials such as leather, fibres or plastics are technically feasible for use in **shoe soles and wellington boots** – although some are more expensive at the point of sale.

- Alternative materials, plastics and novel formulations have been developed and are in use for **automotive interiors (dashboards and seat coverings**). Several manufacturers have made public statements on the phase out of PVC in car interiors.

- Alternative materials have been used in several specific **medical applications**. These include alternative plastics as well as phthalate-free alternative plasticisers. Several health authorities/hospital have phased out specific uses in this sector. The alternatives have increased costs. Challenges have been encountered substituting PVC in blood bags.

7.1 Introduction

This section explores potential alternatives to PVC in a selection of applications. The scope of this study does not extend to assessing the potential alternatives additvies that may be used in PVC, but where such information has been identified for specific applications, these are highlighted. This is noted furter in chapter 8. The methodology follows ECHAs format for the analysis of alternatives (AoA) in applications for authorisation under REACH (European Chemicals Agency, n.d.). Note the latest guidance suggests combining the AoA with the socio-economic analysis (SEA), but these two assessments are separated in this study (See chapter 8). Potential alternatives to PVC in different applications were identified based on a critical desktop review of a variety of academic, technical and commercial literature, as well as a targeted review of data from online marketplaces. Additionally, a focus group with representatives of the PVC industry, downstream users, NGOs and manufactures of alternatives was carried out in July 2021.

It is not possible within the scope of this study to evaluate all potential alternatives for all PVC applications in any detail. A subset of applications are examined, based on a series of criteria explained from section 7.3. Nor was primary research – beyond the focus group noted above -

possible. The AoA guidance notes that where the number of products or processes associated with the use is very high (e.g. many hundreds of different products), meaningful categories of uses should be defined, underpinned by reasonably foreseeable combinations of processes, products, technical requirements and market sectors. This approach has been adopted. The categories used are based on those identified in earlier assessments of PVC published since the early 2000's. More recent data on PVC volumes (and used in other data sources such as Eurostat) have used slightly different categorisations; these are compared to draw reasonable estimates about current volumes of use in specification applications. This section contains several steps:

- First, we provide a brief overview of the various environmental and human health concerns posed by PVC at different stages of the life cycle. This is necessarily general, we recognise risks will differ depending on use, lifecycle stage and on users. The purpose is to focus the AoA on the key issues. This section provides a brief recap only, using the information contained in chapters 2, 4, 5 and 6.
- Second, we summarise the process used to shortlist specific PVC applications for more detailed analysis. These are grouped by uses (e.g. flooring, window frames etc). We consider several possible alternatives in each use under four headings, as per the guidance (we do not consider each alternative individually to avoid an overly long and repetitive chapter). These four headings are:
  - **Substance ID and properties:** For each substance or alternative technology, we provide the chemical name and CAS number, where relevant, alongside relevant information on its properties and chemical composition, including classification and labelling information.
  - **Technical Feasibility:** This evaluates whether the alternative can deliver the required technical functions to the same or acceptable standard. We note the changes required for possible substitution and how its adoption may affect technical performance, where possible. We note available information on the extent to which substitution is underway along with actions (including R&D, production trials, etc.) and timeframe within which technical feasibility may be achieved where possible.
  - **Economic feasibility:** This focuses on the direct and indirect costs and revenues associated with the placing on the market or using the alternative. This is considered in more detail as part of chapter 8.
  - **Reduction on overall risk due to transition to the alternative:** We evaluate whether transfer to the alternative can be expected to result in reduced overall risks to human health and the environment. This includes consideration of additives used in alternative plastics where this detail is available. We document the data used, its quality and reliability, the assumptions made, the uncertainties in the analysis and their impact on the conclusions of the assessment. Where it cannot be concluded the transition to the alternative will result in an overall risk reduction, we highlight possible actions to enable this and estimate timeframes where possible.
  - **Availability:** we describe whether alternatives are available in the required quantity without undue delay. If it is not available, we note the actions required.

7.2 Concerns from manufacture, use and disposal of PVC

Concerns from PVC relate to various stages of the product lifecycle. These are set out in detail in earlier work packages and in various detailed publications, so a brief overview is provided to help focus the later comparison with alternatives.

7.2.1 Manufacture

The production process for PVC is outlined in chapter 3. PVC production is the largest user of chlorine gas in the world (Thornton, 2002). Chlorine-rich hazardous wastes are generated in the synthesis of ethylene dichloride and vinyl chloridemonomer (EDC and VCM, the feedstocks for PVC). These substances are carcinogens and are also associated with Dioxin bi-products (Thornton, 2002). Chlorine production is also associated with carbon tetrachloride, an Ozone depleting substance, as well as emissions of mercury (prior to December 2017, when EU regulation on this process came into effect) (Health Care Without Harm, 2021).

7.2.2 Use

Various PVC additives possess hazardous characteristics and environment and human health risks during the PVC use phase. The release potential of the additives differ, depending on: the properties of the substance itself (for instance a low molecular weight can be an indicator for a higher mobility); concentrations in the plastic material; diffusivity of the polymer matrix; dimensions of the article (e.g., surface area) and conditions of use (e.g., temperature). (PLASI Initiative – ECHA 2021;(Augustsson and Henningsson 2011). Common additives include:

- **Plasticisers** soften the structure and enable a wider range of PVC applications such as insulation and flooring (Ying Xu, 2010). These are often based on aromatic dicarboxylic acids (phthalates, notably DEHP). Plasticisers are predominantly used in PVC, with about 80% of all plasticisers used in PVC (Hahladakis, et al. 2017). Non-PVC polymers that use plasticisers in significant volumes include acrylics, polypropylene (PP), polyethylene (PE), polyethylene terephthalate (PET) and polyurethanes (PUR) (ProductPedia 2021). No information was found indicating that different plasticisers are used preferentially with different types of plastic.
- It is important to note that the European Commission restricted use of DEHP (and several other phthalates) in consumer products, based on reproductive and neurodevelopmental effects, designating it an endocrine disrupter for humans and the environment under the REACH regulation (Thornton, 2002). This restriction was adopted in 2018 (European Commission, 2018) and applies to articles placed on the market from 7th July 2020, it covers the use of DEHP (and DBP, BBP, DIBP), as follows:
    - Shall not be used as substances or mixtures, nor placed on the market at or greater than 0.1% w/w, in toys and childcare articles.
    - Shall not be placed on the market after 7th July 2020 in articles individually or in combination at or greater than 0.1% w/w in the plasticised material[159] in the article. This does not apply to:
        - articles exclusively for industrial or agricultural use, or for use exclusively in the open air, provided that no plasticised material comes into contact with human mucous membranes or into prolonged contact[160] with human skin;
        - aircraft, placed on the market before 7 January 2024, or articles, whenever placed on the market, for use exclusively in the maintenance or repair of those aircraft, where those articles are essential for the safety and airworthiness of the aircraft;
        - motor vehicles within the scope of Directive 2007/46/EC, placed on the market before 7 January 2024, or articles, whenever placed on the market, for use exclusively in the maintenance or repair of those vehicles, where the vehicles cannot function as intended without those articles;
        - measuring devices for laboratory use, or parts thereof.
- **Light and heat stabilisers** are used to extend the useful life of articles because PVC catalyses its own composition. Many are based on inorganic salts such as cadmium and lead, however there are also tin and organophosphorus based heat stabilizers (European Environmental Bureau, 2020). These are more commonly used in PVC, compared to other

[159] This includes PVC, PVDC, PVA and polyurethane and any other polymer except silicone rubber and natural latex coatings.

[160] prolonged contact with human skin' means continuous contact of more than 10 minutes duration or intermittent contact over a period of 30 minutes, per day.

plastic types (Hahladakis, et al. 2017). Heat stabilisers are based on tin, barium, and zinc compounds. Those based on lead and cadmium have largely been phased out, but only relatively recently (Health Care Without Harm, 2021). Various **pigments** are also used.

More detail on the types of additives used in PVC can be found in section 2.2, and section 2.3 expands on the release potential of different additives from PVC.

7.2.3 Disposal and end of life

At end-of-life harmful emissions to air can arise where PVC articles are incinerated, illegally dumped / from open burning, or in building fires. Additives can leach into soil when they are landfilled. Whist industry data indicates increasing volumes of PVC are recycled - in some applications - additives can be reintroduced as secondary raw material. While this share of PVC waste is initially diverted from landfilling or incineration, it poses concerns in the context of a transition to a circular economy, the EU Green New Deal as well as the ambitions of the Chemicals Strategy for Sustainability. Further information on disposal routes for PVC waste and their associated environmental impacts can be found in chapter 6. According to an interview with VinylPlus (2021), in the EU approximately one third of the PVC waste is being landfilled, one third is being incinerated and one third is sent for recycling[161]. Recycling rates are not constant and differ between Member States, but recent data on registered recycled quantities of PVC amongst members of VinylPlus specific applications provide an indication of trends (Vinylplus 2020b):

- Coated fabrics (0.9%);
- Flooring (0.4%);
- Pipes and fittings (11.1%);
- Cables (18.5%, note these typically contain high concentrations of DEHP (6-7% in PVC from copper cables; 5-7% in PVC from aluminum cables));
- Flexible PVC and films more generally (22%);
- Window and related profiles (47.1%).
- It is understood the majority of PVC waste in the healthcare or medical devices is incinerated, whilst that PVC in the automotive and transport sector is mainly incinerated or landfilled (Ciacci, Passarini, and Vassura 2017).

Volumes of recycled flexible PVC waste are currently lower than those of rigid PVC. This reflects several factors including the higher number of additives and the technical challenges of shredding flexible material. Similarly, post-consumer PVC waste is more complicated to recycle, given it tends to have higher impurities and other materials (Ragaert, Delva, and Van Geem 2017). Pre-consumer waste streams are likely to be less contaminated, are collected separately with more information about their content (Sadat-Shojai and Bakhshandeh 2011). The factors influencing PVC recycling rates are detailed in Section 4.1.

7.3 Short-list of Alternatives for detailed assessment

PVC is found in a wide range of applications and sectors. This chapter has focussed on a shortlist of PVC uses to enable more detailed evaluation of alternatives. The shortlist was based on the criteria discussed in the following sections.

7.3.1 Volume of use

Applications using larger volumes of PVC were prioritised. To approximate the volume of PVC used in each application we identified the share of total PVC mass (in %) per application. This was based on a detailed but dated, assessment from 2004 by the European Commission. This remains the most detailed information on PVC products by applications (clothes, cables etc) rather than

[161] Note: this does not necessarily mean that all of the one third sent for recycling is ultimately recycled.

product types (rigid films etc) (European Commission 2004). To validate the 2004 data, this was aggregated and compared with less detailed, but more recent European Council of Vinyl Manufacturers data from 2017. As shown in Table 7-1, PVC shares for different applications have remained relatively similar. The share comprised by Profiles and Rigid plates has increased by 9% whilst Flexible films & sheets has decreased by around 5%. This suggests that the PVC market for packaging has declined, for example, since the European Commission report was published, but that the data remain reasonably accurate. These estimates were tested with stakeholder consultees as part of chapter 3. This indicated that uses of PVC in construction uses have likely increased as a total share of European PVC use. To estimate volumes of PVC per application, the detailed sector information from 2004 was applied to the product information from 2017 (Table 3-15 in chapter 3). This provides reasonable confidence that the overall volumes of PVC product by application type are accurate but are approximate. Where specific PVC applications are considered with a greater degree of granularity, the less certain the precise volume data becomes.

Table 7-1: Market share comparison for different PVC applications in 2004 and 2017

Application	European Commission, 2004 (1)	ECVM 2017 (2)	Difference
Profiles	20%	27%	+9%
Rigid plates		2%	
Pipes & fittings	19%	22%	+3%
Other	10%	9%	-1%
Rigid film	8%	8%	=
Flooring	8%	7%	-1%
Flexible film & sheets	12%	7%	-5%
Cables	11%	7%	-4%
Misc. Rigid	8%	6%	-2%
Coated Fabrics	1%	3%	+2%
Flexible tubes & profiles	2%	2%	=

Sources:

(1) (European Commission, 2004)

(2) (European Council of Vinyl Manufacturers, 2017)

7.3.2 Potential Risk

The *potential* risk of different PVC products was assessed considering the criteria presented below and in the absence of detailed information enabling a differentiation of risk between uses. The majority of the information to classify the PVC components was taken from the report published by the European Commission in 2004 (European Commission 2004). When data on a specific application was not available in this report, a desk-based research was carried out to close the gaps. The specific sources – and full data - are listed in Annex 6.1.

- **Indoor or outdoor usage of PVC application:** Each PVC application was classified as mainly used indoors or outdoors to provide an indicator of the potential exposure during usage. Indoor uses (e.g flooring) was prioritised.
- **Percentage of non-PVC substances:** The proportion of non-PVC substances present in the compound was used as an indicator of the potential amount of harmful additives. Those with typically higher proportions were prioritised. The potential for different additive types to migrate from PVC formulations and their bioavailability are discussed in Section 2.3.

- **Flexibility:** Flexible PVC typically has a higher proportion of additives (specifically, plasticisers) by weight, typically ranging between 35% and 65%. As such flexible PVC applications were prioritised[162].
- **Food/medical contact:** PVC used for food packaging or drinking water has potential to leach harmful substances into food and drink ingested by the population (Guo, et al., 2010; Dong, et al., 2013; Mandrile, et al., 2020). PVC components used in the medical sector were also considered to be of higher potential risk due to their application in close physical contact with vulnerable patients and doctors. As such, examples for both uses were prioritised.

7.3.3 Recyclability

PVC components with low recyclability rates were also prioritised. The information to classify PVC applications according to their recyclability was taken from the VinylPlus framework (VinylPlus 2017) and a report on PVC Waste Treatment published by the Nordic Council of Ministers (Fråne, 2019). It is important to note that the classification provided considered the current recyclability potential of the PVC components, however, the recycling capacities vary in different European Countries and will evolve over time as new technologies are developed and mature and as regulation evolves. The current status of PVC recycling in Europe, including recycling rates across different applications, are discussed in Section 4.2.

7.3.4 Other considerations

In selecting applications for inclusion in the analysis, at least one application was chosen from each of the main sectors in which PVC is used so as to ensure a broader scope of analysis, and to aim at better representation of all PVC uses.

7.4 Analysis of alternatives – PVC in Pipes and Pipe Fittings (Waste, rain water and drinking water)

7.4.1 Key performance criteria

PVC is used extensively as a material for pipes and pipe fittings. The majority of PVC use is rigid PVC, but note that industry and literature sources differentiate between PVC-C (chlorinated PVC – used in DWV and water piping systems), PVC-O (molecularly oriented PVC – used in pressurised water mains and sewage systems) and PVC-HI (High Impact PVC – used in natural gas conveyance):

- **Drinking water**:
  - Water mains – underground transportation of pressurised drinking water, using pipes of wide diameter (PVC 4 Pipes, n.d.).
  - Water service lines – underground transportation of drinking water from mains to individual buildings (PlumbingSupply.com, n.d.).
- **Waste and rainwater:**
  - Drainage waste vents (DWV) – release of wastewater from buildings to sewage.
  - Rain water pipes – conveyance of rain water to sewage system.
  - Sewage – buried pipes for transferring sewage water (PVC 4 Pipes, n.d.)
- **Other:**
  - Water piping systems – distribution of water within buildings, handling water of varying temperatures (PlumbingSupply.com, n.d.).
  - Irrigation – buried pipes for transferring water for irrigation (PVC 4 Pipes, n.d.)
  - Natural gas conveyance – note this usage appears to be limited to/focussed in the Netherlands (PVC 4 Pipes, n.d.)
  - Industrial Processes (PVC 4 Pipes, n.d.)

[162] These two categories overlap somewhat but are both judged to be useful indicators.

The key properties shared by all of the above applications are **durability** and **water and chemical resistance**. Requirements for each depend on the specific application.

- In terms of water mains, irrigation and sewage, these typically require higher durability, due to their placement underground and need to withstand high water pressures, with diameters of 160 to 630 mm. Other desirable properties include ductility and crack propagation. Concerns have been raised as to the longevity of PVC sewage pipes, suggesting that lifetime in practice is not as long as suggested by some literature, citing a critical failure in Holland. The study identified need for further work inspecting the performance of pipes in practice(Konstantinos F. Markis, 2019) . Similarly a 2013 study using CCTV footage in two Dutch Municipalities noted a number of defects in PVC piping, showing: 10-12 joint displacements; 4-6 infiltration defects; and 0-3 bending defects per kilometre (Dirksen, et al. 2013).
- Water piping systems in buildings also require heat resistance, allowing the pipes to withstand a wide range of temperatures.
- PVC can lose durability at high temperatures (Atkore Heritage Plastics 2021a) and becomes brittle at low temperatures (Engineering Systems Inc., n.d.; The Madison Group, n.d.) . PVC-C, which is has a broader resistance to different temperatures, is therefore used in its place (PVC 4 Pipes, n.d.).
- PVC water service line installations, while generally the most cost-effective option of available materials, requires a back fill of gravel for structural stability, as the plastic would not withstand the weight of being buried directly in soil PlumbingSupply.com n.d.).
- Potable water pipes such as water mains, service lines and internal piping systems require resistance to bacterial formations. This is provided both by PVC and other plastic pipes (Durapipe UK, 2012; European Council of Vinyl Manufacturers, n.d.). Some studies indicate however that PVC performs better than certain other plastics (HDPE, PE-X) in preventing migration of chemicals from the plastic into water supplies (Ingun Skjevrak 2003).
- Rainwater pipes, along with other external uses above ground, require resistance to UV exposure. While this property is not inherent in rigid PVC, this issue is avoided by coating the material in paint. Where pipes are un-coated, a loss of impact resistance is observed (Atkore Heritage Plastics 2021b).
- Natural gas conveyance by PVC pipe tends to be largely used in the Netherlands (PVC 4 Pipes n.d Hermkens, et al 2008). PVC pipes for this application are required to meet specific criteria relating to durability, set out in international standard ISO 6993 (*"Buried, high-impact poly(vinyl chloride) (PVC-HI) piping systems for the supply of gaseous fuels")* (PVC 4 Pipes, n.d.; International Organization for Standardization, 2020). Durability and corrosion resistance are particularly important here, corrosion being a major cause of pipe leaks in 2017 (S&P Global Marketing Intelligence 2018).

Several other ISO standards are relevant for PVC use in piping. These standards do not state that only PVC may be used for said applications, but they do state the specific performance criteria of PVC pipes:

- PVC-U is required to meet the ISO 1452 standard for applications in water mains and service lines buried underground, piping systems above ground inside and outside buildings, drainage and pressurised sewage (International Organization for Standardization 2019).
- PVC-O is required to meet the ISO 16422 standard for applications underground or above-ground where not exposed to direct sunlight, for water mains and services, pressurised sewer systems and irrigation systems (International Organization for Standardization 2014).
- PVC-C is required to meet the ISO 15877 standard for applications in hot and cold water system installations.

The following ISO standards are relevant for plastics in piping (including PVC) more generally:

- Plastics (including PVC) in the design of hot and cold systems are required to meet the ISO 10508 standard (International Organization for Standardization, 2019).
- Plastic piping systems for the supply of pressurised gaseous fuel or water are required to meet the ISO 17885 standard (International Organization for Standardization 2021).

7.4.2 Substance ID and properties (or Description of alternative technique)

Several alternatives for PVC pipes have been identified in several specific pipe applications (Table 7-2). These have been identified via desktop research which indicates the materials listed are currently used in piping applications.

Table 7-2: Potential alternatives to PVC in Pipes

Alternatvie substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classifiction and Labelling Information	Pipe Application(s)								
			Sewage	Mains	Service Line	Piping Systems	DWV	Rain and Drainage	Irrigation	Natural Gas	Industrial
High-density polyehtylene (HDPE)	Manufactured from ethylene (ethene) monomer, which is extracted from petrochemical sources or via dehydration of ethanol (Dusan Jeremic, 2014).	The precursor is ethene (CAS No. 200-815-3). PE water pipes should adhere to ISO 4427 (International Organization for Standardization, 2019). Contains less substances toxic to health than PVC but is not toxin free (Ackerman, 2006; Ahmadi, 2013), although stakeholder engagement suggests that the rate of diffusion of chemicals from PE pipes into water is greater. Contains low proportions of stabilisers, antioxidants (0.1-2.0 %w), UV stabiliser (0.2-5.0 %w) (John N. Hahladakis, 2018), fillers, plasticisers, colourants (1-5 %w), flame retardants (3-10 %w) (assumed for natural gas piping), blowing agents, crosslinking agents and UV degradable additives (Eltayef, 2015) (additive percentage by mass taken from PLASI March Annex 1). These include Irgafos 168(R) anti-oxidant additive, which can decomposed to detectable levels of 2,4-di-tert-butyl-phenol (CAS No. 96-76-4 – has endocrine disrupting properties) in water samples. Irganox 1010 (CAS No. 6683-19-8), Irganox 1330 (CAS No. 1709-70-2) and Naugard xL-1 (CAS No. 70331-94-1) were also detected in water pipelines. Other trace esters, aldehydes ketones, aromatic hydrocarbons and terpenoids were also detectable (Ingun Skjevrak, 2003; Bonnie Ransom Stern, 2008). In gaseous fuel applications, pipes should adhere to ISO 4437 and ISO 10839 standards (International Organization for Standardization, 2014; International Organization for Standardization, 2014).	✓	✓	✓			✓	✓	✓	✓
Medium-density polyethylene (MDPE)	A preferred material for long distance piping for drinking water, although often not suitable in colder climates (WHO, 2006). See HDPE for precursors.	As above regarding precursors and additives.		✓					✓	✓	
Low-density polyethylene (LDPE)	Low density PE is suitable for low pressure irrigation (WHO, 2006). See HDPE for precursors.	Eicosane (CAS No. 112-95-8), tetracosane (CAS No. 64742-51-4) and nonadecane are impurities present in the polymer which can potentially leach out, but papers show that these can be extracted from the plastic prior to use (Cinthya Soreli Castro Issasi, 2019). Leaching is slightly less relevant here as the material is not used in potable water piping.							✓		

Polypropylene (PP)	Manufactured from propylene (CAS No. 115-07-1) extracted from petrochemical sources (Saeed Doudiani, Chul b. Park, Mark T. Kortschot, 2004). Hot and cold water installations and industrial applications (Borouge, 2012) typically in acid & alkali, chemical processing, industrial waste treatment, pharmaceuticals and effluents (Durapipe, n.d.).	Pipes should adhere ISO 10508 (International Organization for Standardization, 2019). Numerous studies on PP packaging show release of antioxidants into food (J. A. Garde, 2001; J. Alin, 2010; J. Alin, 2011) but there is limited information on water pipe leaching, with available literature on water pipes focusing on PE, PVC and PE-X (Bonnie Ransom Stern 2008).				✓					✓
Glass fiber reinforced polypropylene composite pipes	Used in hot water and drainage installations (Wavin, n.d.). Manufactured from a polypropylene (CAS No. 115-07-1) and glass fiber composite. Fiberglass is manufactured from silica sand and a blend of alumina, magnesia, boron oxide and various other precursors in varying degrees (Composites World, n.d.).	Glass fiber (CAS No. 65997-17-3) is believed to have carcinogenic properties (REACH, n.d.). Limited data on glass fibre leaching into water mains, however no applications in potable water, so less relevant.				✓		✓			
Cross-linked polyethylene (PE-X)	Hot and cold water installations (Grey, n.d.). See HDPE for precursors.	Pipes should adhere to ISO 15875 (International Organization for Standardization, 2003). Study shows no significant health impacts from PE-X pipe drinking water, but can produce detectable odours and tastes in the water. 2.4di-tert-butyl-phenol (CAS No. 96-76-4 – has endocrine disrupting properties)and methyl-ter-butyl ether were found in PE-X pipe water samples (CAS No. 1634-04-4 – has endocrine disrupting properties, causes skin irritation), the latter of which in concentrations above the recommended US EPA taste and odour value for drinking water (Vidar Lund, 2011). Applications in natural gas should adhere to ISO 14531 (International Organization for Standardization, 2019).				✓				✓	
Polybutylene (PB)	Manufactured from butylene monomer precursor extracted from petrochemical sources (CAS no. 9003-29-6)	Pipes should adhere to ISO 15876 (International Organization for Standardization 2017a)				✓					

Acrylonitrile-Butadiene-Styrene (ABS)	Widely used in DWV systems in place of PVC (PlumbingSupply.com, n.d.) . Manufactured from three monomers: Acrylonitrile (CAS No. - 107-13-1, carcinogenic, skin sensitising) a synthetic monomer produced from propylene and ammonia, butadiene (CAS No. - 106-99-0, carcinogenic, mutagenic) a by-product of ethylene production from steam crackers, and styrene (CAS No. - 100-42-5, toxic to reproduction) manufactured from dehydrogenation of ethyl benzene. Many of these precursors arise from the petrochemical industry (J. R Peeters 2014). Not suitable for pressurised systems (Deziel, n.d.).	Contains flame retardants (J. R Peeters, 2014)					✓				
Polyamide/Nylon (PA-U, PA11, PA12)	Appropriate for "most" potable cold water fitting applications and generally more durable than PVC (Deziel, n.d.), Applications in cold water have not been assessed further due to lack of literature data. Used in gaseous supply applications (Plastics pipe Institute, 2017)	Applications in natural gas should adhere to ISO 16486 (International Organization for Standardization, 2020). Polyamides can retain their stability under high temperatures without the need for heat stabilisers (Hahladakis, et al. 2017).								✓	
Galvanised Steel	Used for water distribution in buildings and DWV systems.	High durability, lower water/chemical resistance (PlumbingSupply.com, n.d.). Rarely used in recent years due to problems associated with zinc (CAS No. 7440-66-6) layer corrosion and costs (PlumbingSupply.com, n.d.) as well health issues for applications in water distribution due to lead (CAS no. 7439-92-1, toxic to reproduction) impurities in the zinc coating with corrosive water supplies (Brandi N. Clark, 2015; PlumbingSupply.com, n.d.)				✓	✓				
Stainless Steel	Hot and cold water systems in buildings (Opus Piping	A number of different stainless steel alloy standards exist which are resistant to different environments (Nicks Boots , 2020). If correct				✓				✓	✓

	n.d.) . Natural gas pipes (Opus Piping n.d.) and industrial water piping systems (SPLASH, n.d.)	standard is used, it shouldn't corrode or rust (Electronic Fastneres Incorporated, 2019) . As with most metals, if installed incorrectly and welded/attached to a dissimilar metal, it can result in galvanic/bimetallic corrosion (Marlin Steel, 2021) . Studies show that chemical leaching occurs to a degree, but it is well below the threshold of concern set out in the European Drinking Water Directive. Trace amounts of Nickel (CAS No. 7440-02-0 – suspected carcinogenic, skin sensitising (European Chemicals Agency, n.d.) and Chromium (CAS No. 7440-47-3) were detectable in water samples, but its use in drinking water would be "unobjectionable on public health grounds" (Cutler 2003). Applications in natural gas should adhere to ISO 3183:2019, 13616:2007 and 13623:2017 (International Organization for Standardization 2017b, International Organization for Standardization 2019).									
Copper	Noted for service lines and DWV systems. In water service line use, a roll of continuous tubing is used insead of rigid pipe. Used less in water applications in recent years due to higher cost than PVC (PlumbingSupply.com, n.d.). Also used gas pipes (The Engineering Toolbox, 2008).	CAS No. 7440-50-8 – under assessment as endocrine disrupting as of July 2021 (European Chemical Health Agency, n.d.). Copper corrosion can lead to copper levels in water which exceed health guidelines and cause metallic taste (Dietrich, et al., 2004), with sources stating that older pipes are safer due to a build up of minerals coating the pipe (Apollo drain & rooter service, n.d.). Applications in natural gas should adhere to ISO 13616:2007 and 13623:2017 (International Organization for Standardization, 2017b; International Organization for Standardization, 2019)			✓		✓			✓	
Yellow Brass	Frequently used for fittings in potable indoor water systems (Sage Water, n.d.) and gas pipe use. For gas applications; limited to indoor use (The Engineering Toolbox, 2008) and usage by country depends on local building regulations (On demand supplies 2018).	CAS No. - 12597-71-6. Low grades of brass contain zinc, which can leach into the water supply forming zinc oxide (CAS No. 1314-13-2 – very toxic to aquatic life, may damage fertility or the unborn child, is harmful if swallowed, harmful if inhaled, may cause damage to organs through prologued or repeated exposure). Applications in natural gas should adhere to ISO 13623:2017 and 13623:2017 (International Organization for Standardization, 2017b; International Organization for Standardization, 2019).									✓
Aluminium	Gas pipe use is limited to over-ground applications and not approved by all jurisdictions (The Engineering Toolbox, 2008) .	CAS no. - 7429-90-5. Stated as a flammable solid and releases flammable gases in contact with water (European Chemicals Agency, n.d.). Piping therefore requires coating for use (McHone Industries 2019). Applications in natural gas should adhere to ISO 13623:2017 and 13623:2017 (International Organization for Standardization, 2017b; International Organization for Standardization, 2019).									✓

Cast Iron	Manufactured from iron and flakes of graphite to form an iron, carbon, silica alloy – a traditional method of treating iron (Pamline, n.d.). Used in DWV systems. Main pipe used in older systems, but less in recent years due to cost. Benefits include high sound insulation for sound sensitive systems (PlumbingSupply.com, n.d.). Can degrade in underground environments (The Law Office of Daniel D. Horowitz, III, 2018; Shankar, et al., 2020).	ISO 2531 (International Organization for Standardization 2020). Use in gas pipelines is declining following increasing distribution mains failures (S&P Global Marketing Intelligence, 2018). Applications in natural gas should adhere to ISO 13623:2017 and 13623:2017 (International Organization for Standardization, 2017b; International Organization for Standardization, 2019).				✓	✓	✓		✓	
Ductile Iron	Manufactured from iron and nodules of graphite – the modern method of treating iron, providing higher durability and malleability than cast iron (General Kinematics n.d.). Different grades of coating to protect the pipe from corrosion in different soils, particularly for drainage applications (Pam Line, n.d.) . Internal coatings are commonplace to limit corrosion, usually in the form or cement mortar (most frequently), polyurethane and polyethylene. Pipes frequently feature coatings of other materials, such as, PE (National Research Council, 2009), acrylic, zinalium (zinc-aluminium blend plus epoxy) and thermoplastics (Pam Line, 2012). Gas pipe use is limited to under-ground applications and is not	Applications in natural gas should adhere to ISO 13623:2017 and 13623:2017 (International Organization for Standardization, 2017b; International Organization for Standardization, 2019). Studies have shown mortar cement lining in ductile pipes to leach chemicals into water supplies, including arsenic, barium, cadmium and chromium at concentrations of 10-20% of their maximum contaminant levels (Environmental Protection Agency, n.d.; Guo, et al., 1998). An incident whereby a cement-lined ductile iron pipe was installed adjacent to industrially treated seawater was recorded in 1999, in which aluminium leached through the pipe and raised concentrations to 690 µg/L. This resulted in a number of illnesses. This concentration exceeded the EU limit for dialysis machine-use of 30 µg/L and raised the mortality rate to 32% at a downstream dialysis centre (Berend & Trouwborst, 1999; Environmental Protection Agency, 2002)	✓	✓			✓	✓		✓	✓

	approved by all jurisdictions. (The Engineering Toolbox, 2008)										
Concrete	Limited to non-pressurised and moderately pressurised applications. Can contain steel reinforcements (Condron Concrete Works 2015) (Perkins and Will, 2015). Performs well against social and environmental criteria but tend to be expensive and difficult to install (Ahmadi, 2013).	Highly durable, copes with high pressure water jetting (Condron Concrete Works 2015). Evidence of chemical leaching of barium (CAS No. 7440-39-3 – toxic if swallowed), cadmium (CAS No. 7440-43-9 – carcinogenic, suspected mutagenic, suspected as toxic to reproduction), vanadium (CAS no. 7440-62-2) and chromium (CAS no. 7440-47-3)	✓					✓			
Vitrified Clay	Manufactured from a clay and shale blend. Lifecycle performs well against social and environmental criteria and is ISO 14001 compliant but is more expensive and onerous to install than plastic alternatives (National Clay Pipe Institute 2015, Ahmadi 2013).	"Longest lifespan in market" and resistance to all relevant domestic and industrial waste, except concentrated caustic waste hydrofluoric acid and caustic soda (National Clay Pipe Institute 2015). Offers more flexible cleaning options than alternatives (National Clay Pipe Institute 2020).	✓					✓			

7.4.3 Technical feasibility

There are several technically feasible alternatives to PVC pipes. All of the alternatives described above are commercially available and are used at present in at least some piping applications, several have been used for many years, including before the use of PVC.

A wide range of pipe products are available, and even within each material, variations are available with different characteristics, reinforcement, linings and coatings. The impacts of this has been considered as far as possible. Each material has specific benefits and drawbacks making it more or less suitable for specific uses. The main advantages and drawbacks are summarised in the table below, with reference to the technical requirements for PVC. Much of this information is based on commercial product literature.

Table 7-3: Potential alternatives to PVC in Pipes

Alternative	Durability	Water, Chemical and UV Resistance (if applicable)	Safety (Chemical leaching, bacterial growth susceptibility)	Other (ease of installation, cost, application limitations)
Pressurised Sewage				
High-density polyehtylene (HDPE)	Durable for pressurised applications (Durapipe UK, 2012). More flexible and less rigid than PVC, requires thicker walls to construct (PVC Pipe Association, 2016). Lifetime is upward of 50 years (International Organization for Standardization, 2019; PE100+ Association, 2018)	Corrosion resistance to hydrogen sulfide gas present in sewers, does not tuberculate and resistant to bacterial growth (Durapipe UK, 2012; Harts Services, n.d.; Plastics Pipe Institute, 2020).	No available information on chemical leaching from sewage pipes. Available sources focus on chemical leaching into potable water. The extent to which leachates impact on waste-water after treatment at a sewage treatment works and returning to mains for potable water distribution is unclear. There is also minimal available data on impacts of chemical leaching from pipes into soils.	More flexible and less rigid than PVC, requires thicker walls to construct so more material and greater mass. This may be significant for pressurised applications. Flexibility and light weight may improve ease of installation (PVC Pipe Association, 2016). Lightweight material (Plastics Insight, n.d.).
Ductile Iron	Highly durable (Pamline, n.d.; Ductile iron Pipe Research Association, 2016; McWane Ductile, 2019).	Corrosion resistant if installed with correct grade of coating and lining. These allow for resistance to different sewage applications, soil corrosion and levels of depth (Pam Line, n.d.; Pam Line, 2012). Not always possible to predict the soil type, making it hard to determine an appropriate pipe with risk of premature failures (PVC Pipe Association, 2016; Health & Safety Executive, 2005).	Lacking data, as with HDPE.	Requires correct coating for each location. In the past, soil corrosion induced failures in gas line pipes resulted in the decommissioning of all 'Medium Pressure Ductile Iron' pipes for underground gas conveyance (Health & Safety Executive, 2005). Incorrectly using cement linings/coatings has also resulted in leaching of heavy metals into water supplies on occasion, as well as raising aluminium levels on others, raising fatality rate at a downstream hospital (see Table 7-2 for details) (Guo, et al., 1998; Berend & Trouwborst, 1999). This highlights the importance of selecting the correct coating lining and coatings, although it has been highlighted that this is not always possible (PVC Pipe Association, 2016) (Health & Safety Executive, 2005). Stronger than many metals, so thinner walls required (PVC Pipe Association, 2016).
Concrete	Appropriate for moderately pressurised applications (Mulhbauer, 2004).	Failures have been recorded whereby steel reinforcements beams have corroded away (Mulhbauer, 2004) . Low annual break rate compared to other applications (Utah State University, 2018)	There were no recorded concrete leaching incidents recorded in a study conducted in 1991 (Environmental Protection Agency, 2002). Otherwise lacking data, as with HDPE.	Costly to install due to weight (Ahmadi, 2013).
Pressurised Mains Water				

High-density polyehtylene (HDPE)	As above.	Corrosion resistance. Resistant to bacterial growth enough for sewer uses (Durapipe UK, 2012; Harts Services, n.d.)	Contains Irgafos 168(R) anti-oxidant additive, which can decomposed to detectable levels of 2,4-di-tert-butyl-phenol (CAS No. 96-76-4 – has endocrine disrupting properties) in water samples. Other anti oxidants and trace esters, aldehydes ketones, aromatic hydrocarbons and terpenoids are also detectable as leachates (Ingun Skjevrak, 2003; Bonnie Ransom Stern, 2008). HDPE is regarded as less toxic than PVC however (Ackerman, 2006; Ahmadi, 2013).	As above. Study stated that there were no significant health implications of drinking water from these pipes, but that it did contain 2,4-di-tert-butyl-phenol which has endocrine disrupting properties (Ingun Skjevrak, 2003; Bonnie Ransom Stern, 2008).
Ductile Iron	As above	As above	Studies show degree of harmful chemical leaching from mortar cement linings (Environmental Protection Agency, n.d.; Guo, et al., 1998; Berend & Trouwborst, 1999).	As above. Not possible to assess effects of all internal linings – this should be treated as an unknown.
Concrete	As above.	As above.	No recorded concrete leaching incidents recorded in a study conducted in 1991 (Environmental Protection Agency, 2002), although studies show a degree of harmful leaching from mortar cement linings into stagnant water, such as barium, cadmium, and chromium (Environmental Protection Agency, n.d.; Guo, et al., 1998; Berend & Trouwborst, 1999; Hillier, et al., 1999)	As above. The identfied sources are all dated.
Pressurised Service Lines				
High-density polyehtylene (HDPE)	As above.	Corrosion resistance and resistant to bacterial growth (Durapipe UK, 2012; Harts Services, n.d.).	As above.	As above.
Medium-density polyehtylene (MDPE)	Durable enough for pressurised water. Not suitable for colder climates (WHO, 2006). Resistant to stress cracking (PipeStock.com, n.d.).	Water and resistant to some chemicals. Certain pipes are UV resistant from additives so can be installed above ground (Chemical Support Systems, n.d.).	Suitable for drinking water (BP Plastics, n.d.; WHO, 2006) . Contains Irgafos 168 (Nadejzda Haider, 2002) so likely poses same potential health problems as HDPE.	Lightweight and flexible, so easy to install (Chemical Support Systems, n.d.).
Copper	Durable enough for pressurised applications and heat resistant (WaterWorld 2019, Copper Development Assocation Inc. 2020). Can be installed as a roll of continuous tubing to minimise joints	Resistant to UV (Copper Development Assocation Inc., 2020). Susceptible to corrosion (Below Ground Solutions Ltd, 2016; The Engineering Toolbox, 2008)	Allegedly resistant to bacterial growth, but data is inconsistent (Abraham C Cullom, 2020). Copper corrosion can lead to copper levels in water which exceed health guidelines and cause metallic taste (Dietrich, et al., 2004),	Expensive (Total Pipe Systems, n.d.) but light and easily installed. Is sensitive to low soil pH and high sulphate and chloride levels, so used selectively when installed underground (The Engineering Toolbox, 2008). Expensive (Total Pipe Systems, n.d.).

	and risk of leak and breakage (PlumbingSupply.com, n.d.).		with sources stating that older, more corroded pipes are safer due to a build up of minerals coating the pipe (Apollo drain & rooter service, n.d.). Copper is under assessment as endocrine disrupting as of July 2021 (European Chemical Health Agency, n.d.).	
Ductile Iron	As above	As above	As above	As above
Hot and Cold Piping Systems				
Polypropylene (PP)	Durable and wide range of temperature resistance (Borouge, 2012; IPS Flow Systems, n.d.)	Contradicting data from competing industries. Little resistance to chlorinated water, deactivating antioxidants and causing the pipe to become brittle and resulting in a short service life (Plastic Expert Group, 2020) . Other sources claim the opposite with resistance to concentrated hydrochloric acid (Borouge 2012, IPS Flow Systems n.d.) and service life of up to 100 years (albeit in unpressurised sewage pipes) (The European Plastic Pipes and Fittings Association, 2015)	Studies show antioxidants from PP leach into food from packaging but little data available regarding water pipes (J. A. Garde, 2001; J. Alin, 2010; J. Alin, 2011).	Lightweight (IPS Flow Systems, n.d.) but weaker than CPVC so requires thicker walls (DRTS, 2019)
Glass fiber reinforced polypropylene composite pipes	More durable than PVC and polypropylene and retains strength at lower temperatures at a comparable price (Cibse Journal, 2018)	No data identified.	Little data available regarding leaching, potential for similar risk as polypropylene (see above). Potential concern regarding carcinogenic glass fibers present in pipe.	Performs well acoustically, minimising drainage noise (Cibse Journal, 2018).
Cross-linked polyethylene (PE-X)	Short term temperature resistance up to 250 °C and long term up to 120 °C. Commonly used in heating systems.	Wide range of chemical resistances (Plastics Pipe Institute, 2020)	Study shows no significant health impacts from PE-X pipe drinking water, but can produce detectable odours and tastes from water. 2.4di-tert-butyl-phenol (CAS No. 96-76-4 – has endocrine disrupting properties) and methyl-ter-butyl ether were found in PE-X pipe water samples (CAS No. 1634-04-4 – has endocrine disrupting properties, causes skin irritation) (Vidar Lund, 2011).	Flexible, can stretch around bends, connects via threaded compression fittings (PVC connects with glue) leading to ease of use and installation (Grey, n.d.).
Polybutylene (PB)	No longer manufactured in the U.S for plumbing purposes due to ruptures causing water damage (Gromicko, n.d.)	Ruptures supposedly caused in part by disinfectants weakening pipe (Gromicko, n.d.).	Study shows that polybutylene makes up a high proportion of total water contamination incidents from permeation (Environmental Protection Agency, 2002).	Highly flexible, easily installed, low cost, suitable for high and low temperatures. May still be appropriate for in-building non-pressurised, non-potable applications (Polypipe, 2020). It is noted that

				sources are relatively old and technology may have improved.
Glavanised Steel	Highly durable and impact resistant (Total Pipe Systems, n.d.).	Susceptible to corrosion in corrosive water supplies (e.g. private wells) (PlumbingSupply.com, n.d.), with a lifespan of up to 40-50 years in ideal conditions (Square One, n.d.)	Lead impurities in the zinc galvanised layer pose a health risk in potable water (Brandi N. Clark, 2015; PlumbingSupply.com, n.d.).	Very heavy, so harder to install and more costly (Total Pipe Systems, n.d.) (PlumbingSupply.com, n.d.). Incompatible with brass or copper due to increased corrosion rate. Expertise required in installation (Square One, n.d.).
Stainless Steel	Highly durable (DS Steel Pipe, 2018; Jacob, 2020) and highly resistant to turbulence corrosion (SPLASH, n.d.) Industry sources state a typical life of up to 50 years (Jacob, 2020), with some claiming up to 100 years (SPLASH, n.d.).	Highly corrosion resistant if installed correctly and using appropriate grade (Nicks Boots , 2020; Total Pipe Systems, n.d.).	Some chemcial leaching potential as noted in substance ID section above, but aligns with health standards (Cutler, 2003).	Expertise required in installation regarding metallic links to avoid galvanic corrosion and selecting the correct grade (Marlin Steel, 2021; Nicks Boots , 2020). Very heavy, and labour intensive installation process (Total Pipe Systems, n.d.).
Copper	Durable enough for pressurised applications and resistant to high and low temperatures at little drop in physical strength (WaterWorld, 2019; Copper Development Assocation Inc., 2020; European Copper Institute, n.d.)	Resistant to UV (Copper Development Assocation Inc., 2020). Good corrosion resistance if type L pipes are installed (Copper Development Assocation Inc., 2020; PlumbingSupply.com, n.d.).	As above.	Expensive (Total Pipe Systems, n.d.) but light and easily installed. High heat conductivity means temperature can be lost to the pipe and surrounding air (beneficial for heat (Siegenthaler, 2002; Pneumosys Advance Energy Solutions, n.d.)
Polyamide/Nylon (PA-U, PA11, PA12)	Limited data available. Not appropriate for high temperature applications as degrades in presence of hot water (El-Sherik, 2017). “Generally” more durable than PVC (Keith Specialty, n.d.) .	Limited data available. Low resistance to acids and bases (MatMatch, n.d.).	Limited data available. Degrades at high temperatures in presence of water (El-Sherik, 2017)	Very light and flexible, easily installed (Freelin Wade, n.d.).
Yellow Brass	Durable (PVC Fittings Online, 2019; On demand supplies, 2018; Fluid Controls Ltd, n.d.)	High temperature and rust resistance (PVC Fittings Online, 2019; On demand supplies, 2018; Fluid Controls Ltd, n.d.) but susceptible to dezincification, which results in zinc oxide residue and blockages, sometimes resulting in failure/leakage (Sage Water n.d., Lorber, Greenfield & Polito LLP n.d.) .	Dezincification results in the formation of zinc oxide, which is harmful if swallowed (Pasupuleti, et al., 2011; European Chemicals Agency, n.d.) . There has been no research identified regarding the effects of zinc oxide from brass fittings in drinking water on human health.	Malleable, so can facilitate bends without the requirement for specialised tools (On demand supplies, 2018).
Cast Iron	Durable, but significantly lower tensile strength than ductile iron counterpart (Pamline, n.d.). Has greater rupture rate per kilometre in water supply networks of mainstream uses (Rezaei, et al., 2015).	Susceptible to rust (Shankar, et al., 2020) and graphitisation (The Law Office of Daniel D. Horowitz, III, 2018) resulting in brown water and metallic taste (Mulheron 2014, Washington State Department of Health 2011).	Rust in water is not a health concern (Washington State Department of Health, 2011). Can discolour water.	High sound insulation for sound sensitive systems. High cost and thicker wall requirements than ductile iron (PVC Pipe Association, 2016) (PlumbingSupply.com, n.d.)

Ductile Iron	As above.	As above. High temperature resistance linings available (McWane Ductile, 2020)	As above.	As above.
Drainage Waste Vents (DWV)				
Acrylonitrile-Butadiene-Styrene (ABS)	Durable and rigid but restricted to indoor use due to low UV resistance (IPS Flow Systems n.d.).	Good corrosion resistance (IPS Flow Systems, n.d.).	As with sewage pipes, there is no available information on chemical leaching from DWV. Available sources focus on chemical leaching into potable water. The extent to which leachates impact on waste-water after treatment at a sewage treatment works and returning to mains for potable water distribution is unclear. There is also minimal available data on impacts of chemicals leaching from pipes into soils.	Lightweight and simple one-step installation method (PlumbingSupply.com, n.d.). Not suitable for pressurised systems (Deziel, n.d.).
Glavanised Steel	As above.	As above.	Lacking data, as with ABS	As above.
Cast Iron	As above.	Susceptible to rust (Shankar, et al., 2020) and graphitisation (The Law Office of Daniel D. Horowitz, III, 2018).	Lacking data, as with ABS	As above.
Ductile Iron	As above.	As above.	Lacking data, as with ABS	As above.
Rain, Drainage, Irrigation and Non-pressurised Sewage				
High-density polyehtylene (HDPE)	Durable (Durapipe UK, 2012). Lifetime is upward of 50 years (PE100+ Association, 2018; International Organization for Standardization, 2019).	Corrosion resistance and resistant to bacterial growth enough for sewer uses (Durapipe UK, 2012; Harts Services, n.d.).	See Pressurised Sewage and DWV.	Lightweight (Plastics Insight, n.d.) and more flexible and less rigid than PVC (PVC Pipe Association, 2016), making installation easy. Requires thicker walls to construct meaning more material required (PVC Pipe Association, 2016).
Medium-density polyethylene (MDPE)	Less durable but more flexible than HDPE, not suitable for cold environments (WHO, 2006). Resistant to stress cracking (PipeStock.com, n.d.)	As above.	See Pressurised Sewage and DWV.	Too flexible for rain and sewage but used in garden irrigation (Easy Garden Irrigation, n.d.).
Low-density polyethylene (LDPE)	Durable enough for low pressure irrigation (WHO, 2006) and drainage systems.	Water resistance with limited resistance to chemicals (CP Lab Safety, n.d.).	See Pressurised Sewage and DWV.	Highly flexible, ease of use.
Cast Iron	As above.	As above.	See Pressurised Sewage and DWV.	High cost and thicker wall requirements than ductile iron (PVC Pipe Association, 2016) (PlumbingSupply.com, n.d.)
Ductile Iron	As above.	As above.	See Pressurised Sewage and DWV.	As above.

Concrete	Appropriate for moderately pressurised applications (Mulhbauer, 2004).	As above.	See Pressurised Sewage and DWV.	As above.
Clay	Good abrasion resistance but lower strength than alternatives. Appropriate for non-pressurised drainage applications (Saudi Vitrified Clay Pipe Co., n.d.; Continental Steel & Tube Company, n.d.; National Clay Pipe Institute, 2015; Mulhbauer, 2004)	High corrosion resistance (Mulhbauer, 2004; National Clay Pipe Institute, 2015).	See Pressurised Sewage and DWV.	Heavy and costly to install (National Clay Pipe Institute, 2015; Ahmadi, 2013).
Natural Gas				
High-density polyethylene (HDPE)	As above.	Corrosion resistance (Durapipe UK 2012, Harts Services n.d.). However, "Recently installed plastic" was present in a number of gas leak incidents reported in 2018, although which plastic is not stated (S&P Global Marketing Intelligence, 2018). As the primary plastic pipe used in natural gas transportation, it is possible these were HDPE (DOE Hydrogen Programme, 2005)	No available information on chemical leaching from natural gas pipes into soils. Available sources focus on pipes in water conveyance applications.	As above.
Medium-density polyethylene (MDPE)	As above.	UV resistant so can be installed above ground (Chemical Support Systems, n.d.). Specific 'yellow' MDPE appropriate for gas supply (BP Plastics, n.d.). However, the hazards associated with HDPE in natural gas lines are assumed to also apply to MDPE.	As with HDPE, no information available.	As above.
Stainless Steel	As above.	Highly corrosion resistant if installed correctly and using appropriate grade (Nicks Boots , 2020; Total Pipe Systems, n.d.). However, corrosion from 'wet gasses' can occur under conditions where high temperatures, pressures, and stress are present resulting in failures (Weimin Zhao 2018). This can be avoided by using cathodic protection to reduce corrosion (S&P Global Marketing Intelligence, 2018).	As with HDPE, no information available.	Expertise required in installation regarding metallic links to avoid galvanic corrosion and selecting the correct grade (Marlin Steel, 2021; Nicks Boots , 2020). Very heavy, and labour intensive installation process (Total Pipe Systems, n.d.).

Copper	Durable enough for pressurised applications and resistant to high and low temperatures at little drop in physical strength (WaterWorld, 2019; Copper Development Assocation Inc., 2020; European Copper Institute, n.d.).	Resistant to UV (Copper Development Assocation Inc., 2020). Appropriate for gas pipes in homes and will be resistant to gas related corrosion if hydrogen sulfide content is below guidance amount (International Association of Plumbing and Mechanical Officials, 2004) .	As with HDPE, no information regarding chemical leaching. A study conducted in 1996 in the UK revealed no copper gas pipe failures recorded (International Association of Plumbing and Mechanical Officials, 2004).	Expensive (Total Pipe Systems, n.d.) but light and easily installed. Copper pipes should not be used where natural gas contains 7 milligrams of hydrogen sulfide per metre cubed of gas (International Association of Plumbing and Mechanical Officials, 2004) . EUROMOT stated that the limit on sulfide content in Europe should be 10 milligrams per metre cubed (EUROMOT 2017) , meaning copper pipes would need to be installed very selectively, where sulfides are below this level, to remain safe.
Yellow Brass	Less durable than steel and copper (On demand supplies, 2018) .	Susceptible to galvanic corrosion if not installed properly (Pipeotech 2020, RMMCIA 2015) .	As with HDPE, no information regarding chemical leaching. No breakage rates available. Lower durability and susceptibility to corrosion would suggest a higher risk of rupture.	Requires expertise in installation in order to avoid galvanic corrosion (Pipeotech, 2020; RMMCIA, 2015) . Limited to indoor applications and only in areas where installation meets building regulations (The Engineering Toolbox, 2008).
Aluminium	Susceptible to cracking from impact if under stress (BuyersAsk, 2021).	Galvanic corrosion of aluminium is significant only in highly conductive media such as sea water (Products Finishing 2019).	As with HDPE, no information regarding chemical leaching. No information on breakage rates. Lack of permitting in various jurisdictions as well as cracking susceptibility (The Engineering Toolbox, 2008) would suggest associated risk.	Limited to over-ground applications and only in areas where installation meets building regulations (The Engineering Toolbox, 2008). Requires coating in order to attain a good lifespan (McHone Industries, 2019)
Cast Iron	Durable, but significantly lower tensile strength than than ductile iron counterpart (Pamline, n.d.)	As above.	Recorded gas line failures (S&P Global Marketing Intelligence, 2018; The Law Office of Daniel D. Horowitz, III, 2018)	Limited data available on installation and modern use in natural gas as no longer used.
Ductile Iron	Past instances of breakages have resulted in banning of 'Medium Pressure Ductile Iron' pipes (Health & Safety Executive, 2005)	As above.	As with HDPE, no information regarding chemical leaching.	As above.
Polyamide/Nylon (PA-U, PA11, PA12)	Stress cracking resistance and highly flexible, but soft and can be cut easily (MatMatch, n.d.).	Degrades at high temperatures in presence of water (El-Sherik, 2017). Low resistance to acids and bases (MatMatch, n.d.)	As with HDPE, no information regarding chemical leaching. ISO approved material for gaseous fuel conveyance (International Organization for Standardization, 2020). Degrades at high temperatures in presence of water (El-Sherik, 2017) though so risk of leak.	Very light and flexible, easily installed (Freelin Wade n.d.).
Industrial				
Polypropylene (PP)	As above.	Wide range of chemical resistances (IPS Flow Systems, n.d.).	No information on chemical leaching from industrial pipes. Sources focus on chemical leaching into potable water. The extent to which leachates impact on industrial processes is not well	Lightweight (IPS Flow Systems n.d.) but weaker than PVC requiring thicker walls (DRTS, 2019)

			documented, nor are the impacts of chemical leachates into soil. PP is durable, although it is not within the scope of this assessment to address resistances to different chemical mixtures over different temperatures ranges, which may cause failures, so this should be treated as an unknown.	
Stainless Steel	As above.	As above.	As with PP evidence of chemical leaching of quantities insignificant to human health (Cutler, 2003)- see Table 7-2 for details	As above.
Ductile Iron	As above. Used extensively in pressurised sewage treatment centres (Pam Line, n.d.)	As above.	As with PP.	As above.

7.4.4 Economic feasibility and economic impacts

There are alternatives to PVC pipes that have been commercially available and have been installed by a range of downstream users and consumers, in some cases for many years.

Life cycle assessments of pipes have shown that the main costs of pressurised water distribution systems are pumping, repairs and installation, while the initial cost of the pipe is minimal in comparison (Thomas, et al., 2016). Repair depends primarily on the severity and situation of the damage (Water UK, 2018) but also on ease of repair, including in situ repair considering labour costs for excavating and replacing the pipe as well as associated delays to the public. Therefore, this section compares the expected service life of each material, along with the expected repair frequency based upon available mains failure survey data.

For smaller scale applications, such as those within personal homes, and those which do not depend on pumping, such as irrigation and gravity applications, the cost of pipes is likely to be more significant. Note that the costs of individual fittings are not covered within this section and are not practicable to estimate with any accuracy.

The alternatives mentioned in the literature and via consultation are a mixture of alternative materials and alternative plastics. Whilst some manufacturers are involved in the production of several different types of pipes (including PVC), the key economic effects may well result in decreased demand in one supply chain, offset by an increase in another. Such changes in demand may well affect prices for alternatives which are in turn influenced by speed and scale of demand increases and any related constraints in supply.

Table 7-4: Unit cost and typical lifetimes of PVC and alternatives in flooring

Alternative	Cost per metre	Approx lifetime	Installation	Repair
PVC	PVC-U (used in DWV, Drainage, Irrigation, Industrial) costs vary substantially based on diameter, wall thickness, pressure rating and a variety of other factors. As such the pricing ranges between £1/metre and £150/metre for diameters of up to 7", and up to £2,000 for 12" diameters (Durapipe, 2018; GF Piping Systems, 2019). CPVC (used in hot and cold water systems) also vary substantially for the same reasons, from £50 to £1100/metre for diameters up to 2" and up to £2,200 at 4" diameter (Apollo Pipes n.d.) . There is no publically availale data regarding PVC-O (mains, sewage, industrial) cost per metre. Given it's application however, the primary costs will likely be installation, repair and replacement.	Wide variety of lifetime claims: **In building plumbing:** • 20-25 years (Thumbtack, 2020) • "Old" PVC: 25-40 years • Recent PVC: 70 years+ (Smiths Plumbing Service, 2019) **Sewage/Mains** 50-100 years (Whittle & Tennakoon, 2013; Meerman, 2008; Folkman, 2014; Environmental Protection Agency, 2002; Thomas, et al., 2016; Makris, et al., 2021)	For underground applications, requires a back-fill of gravel to protect the pipe from soil damage (PlumbingSupply.com, n.d.). Low weight makes installation straighforward (Vahidi, et al., 2015).	Represents a weighted average of 7% of mains pipe repairs and 2.3 breaks per 100 miles of piping (Rezaei, et al., 2015; Utah State University, 2018).
HDPE, MDPE and LDPE	£1-3,000 up to diameters of 47" for potable water pipe usage. £2-1,200 for gas pipe usage for diameters diameters between ½" and 30" diameter (GPS UK, 2016)	50-100 years (Environmental Protection Agency, 2002; Nguyen, et al., 2021; Frank, et al., 2019; Bredacs, et al., 2014)	Light and flexible, so relatively easy to install (Plastics Insight, n.d.).	Represents a weighted average of approximately 3% of mains pipe repairs (Rezaei, et al., 2015).
ABS	£25-1,100 for diameters of ½" to 8" in DWV applications (Durapipe 2018).	50-100 years (How to Look at a House, n.d.; Water Quality Products, 2013; Aronpr, 2017; Durapipe, 2018)	Lightweight and easy to install (Crescent Plastics Incorporated, n.d.)	-
PE-X	No reliable information identified. Approximately £2-40 for ¼" to 2" diameters in plumbing applications (Uponor, 2021).	40-100 years (The European Pipes and Fittings Association, n.d.; PEX Universe, n.d.)	Flexible, can stretch around bends, connects via threaded compression fittings (PVC connects with glue) leading to ease of use and installation (Grey n.d.).	-
Ductile Iron	£25-700 for 2"-23" diameters of drainage and soil pipe usage (Pam Line, 2021).	60-100 years (Thomas, et al., 2016; Environmental Protection Agency, 2002; Folkman, 2012)	Heavier than plastic alternatives, so harder to install (Plastics Insight, n.d.)	Represents a weighted average of approximately 5% of mains pipe repairs and 5.5 breaks per 100 miles of piping per year (Utah State University, 2018; Rezaei, et al., 2015)
Concrete	No reliable information identified.	70-100 years (Canadian Concrete Pipe & Precast Association, n.d.; Trenchlesspedia, 2020)	Costly to install due to weight (Ahmadi, 2013; Vahidi, et al., 2015)	Represents 3.1 breaks per 100 miles of piping per year (Utah

				State University, 2018; Rezaei, et al., 2015)
Clay	No reliable information identified.	50-100 years (Reline Solutions, 2021; Aronpr, 2017; Option One Pluming, 2021).	Heavy and costly to install (National Clay Pipe Institute, 2015; Ahmadi, 2013).	-
Copper	£10-2,200 for ¼” to 8” diameters in potable water applications (Mueller Streamline Co., 2021)	50-80 years (Smiths Plumbing Service, 2019; Today's Homeowner, n.d.; Houselogic, n.d.).	-	Installed as a roll of continuous tubing in service lines to minimise joints and risk of leak and breakage (PlumbingSupply.com n.d.).
Stainless Steel	No reliable information identified. Approximately £15-100 for 2” to 8” diameters in drainage applications (Drainage Superstore, n.d.)	50+ years (Jacob, 2020; Blucher Marine, n.d.; BSSA, 2003)	Heavier than plastic alternatives, so harder to install.	Has a break rate of approximately 7.6 per 100 miles of piping per year (Utah State University 2018).
Brass	No reliable information identified.	40-100 years (Smiths Plumbing Service, 2019) (Today's Homeowner, n.d.)	-	-

7.4.5 Reduction of overall risk due to transition to the alternative

The range of materials, products available and the global supply chains for these make an overall assessment of the net risk to human health and the environment subject to uncertainty. PVC itself (CAS: 9002-86-2) is not listed in the CLP although vinyl chloride (CAS: 75-01-4) is listed as a Carcinogen (1A: H350: may cause cancer) (European Chemicals Agency, n.d.). The concerns related to PVC relate to exposure to harmful substances, during manufacture, leaching of harmful additives during the lifetime use into water supplies, as well at end of life, via leaching from landfill to soil, reintroduction via recycling, or emissions to air via incineration (see Polcher et al. 2020 in section 5.4). Leachate from PVC into water supplies includes organotin stabilisers, which was present in 29-40% of water samples collected in Canada 1996 (Adams, et al., 2011; Sadiki & T. Williams, 1999). Other harmful substances detected in water supplies from PVC include phthalates, titanium dioxide and the vinyl chloride monomer (Tomboulian, et al., 2004. There is very little data available on impacts of additives or any other leachates from PVC and its alternatives in soil surrounding installed pipes.

Both PVC and polyethylene products are manufactured from petrochemical sources, thus the same sustainability challenges associated with PVC production are applicable to production of polyethylene. However, polyethylene also has a possible sustainable material source in the form of dehydration from ethanol, which is commercially available (European Biomass Industry Association, 2014).

During use, polyethylene (HDPE, MDPE, LDPE) pipes may lead to an overall decrease in leachates, as polyethylene typically contains less toxic additives than PVC (Ackerman, 2006; Ahmadi, 2013). However, stakeholder engagement suggests that polyethylene allows for greater rates of diffusion of chemicals from the pipe material into water supplies (note it is not clear if this reflects the structure of the polymer matrix or the tpye of addtvies used). Leachate of 2,4-di-tert-butyl-phenol (used in preparation of UV stabilisers) in pipe water for example, while endocrine disrupting, is believed to be present in minor concentrations with no significant impact to health. Other leachates such as Irganox 1010, Irganox 1330 and Naugard xL-1 (all antioxidants), are present but were not known to have any health impacts (Ingun Skjevrak, 2003; Bonnie Ransom Stern, 2008). Toxic substances are still present (Irgafos, phenols) in water pipeline samples (Ingun Skjevrak, 2003; Bonnie Ransom Stern, 2008), and present similar potential risks as PVC in relation to exposure to additives (see chapter 5) through drinking water.

Similarly, concrete pipes exhibit a degree of harmful chemical leaching, with vanadium, barium, cadmium and chromium being detectable, although the levels of these substances in non-stagnant consumer water are believed to be undetectable (Environmental Protection Agency, n.d.; Guo, et al., 1998; Berend & Trouwborst, n.d.; Hillier, et al., 1999). Concrete pipes perform relatively well with respect to their environmental impact in the production and use phase in comparison to ductile iron and plastic pipes (Vahidi, et al., 2015). Recycled concrete and cement is frequently downcycled to less demanding uses such as filler and road surfacing (SteelConstruction.info, n.d.), although recycled concrete can exhibit leaching of harmful alkaline and heavy metal substances into soils (Gupta, et al., 2018; Chen, et al., 2013). The extent to which steel reinforced concrete pipes are recycled is not known.

In terms of disposal, the presence of certain additives, including flame retardants, in polyethylene and other plastic alternatives can present a technical obstacle to their widespread recycling; an issue currently faced by many PVC waste streams. In addition, the potential for presence of toxic substances in polyethylene piping poses risks in the form of reintroduction to future products through recycling, leaching into soil from landfill, and emissions to air from incineration (see section 5.4). These are similar to risks presented by the end-of-life phase of PVC products.

However, polyethylene is noted as typically containing fewer toxic additives than PVC (Ackerman, 2006; Ahmadi, 2013). Consequently, the barriers to recycling, risks of leaching and toxic emissions to air through incineration associated with polyethylene may be lower than for PVC.

The metals present in ductile iron can be recycled effectively but requires stripping of any lining or coating to extract the metal. The process used for recycling and percentage of coated/lined ductile iron recycled is not known. A life cycle assessment study shows that the production phase of ductile pipe is particularly environmentally demanding when compared with concrete and plastic pipe manufacture (Vahidi, et al., 2015). There is also uncertainty regarding leachates from internal linings of ductile iron pipes and their toxicity, as while the properties of the ductile iron body material have been assessed, risks associated with the variety of different linings and coatings have not been investigated.

Metallic construction materials are widely recycled (SteelConstruction.info, n.d.) although the recycled products vary in quality (Söderholm and Ekvall 2020). "New" stainless steel generally has a make-up of approximately 60% recycled material (British Stainless Steel Association, n.d.). Copper pipes leach copper into water supplies, sometimes exceeding health guidelines, and brass water pipes can form zinc oxide, both of which are harmful to human health. Some metallic pipes used for natural gas applications, such as brass, aluminium and cast iron, present potential risks due to lower durability and susceptibility to breaks.

7.4.6 Availability

All the alternatives identified are commercially available. For several materials, detailed information on EU production, import and export volumes are available via PRODCOM illustrating the current scale of existing supply. Overall, this information indicates that several, if not all, alternative materials are available in significant quantities. PRODCOM data for the EU (28) for 2019 show the following production tonnage values (European Commission, 2021a):

- Rigid tubes pipes and hoses of polymers of ethylene - 1,202,800 tonnes[163]. Other significant uses of ethylene polymers include packaging (both film and rigid) and cable insulation (The Essential Chemical Industry, 2017);
- Clay/ceramic pipes, conduits, guttering and pipe fittings – 159,900 tonnes[164];
- Pipes of cement, concrete or artificial stone – 6,580,000 tonnes[165]. Competing uses of cement and concrete are construction and furniture, where it is used across a wide range of applications (Cembureau, n.d.);
- Line pipe, of a kind used for oil or gas pipelines, seamless of stainless steel – 30,400 tonnes/[166];
- Tubes and pipes, of circular cross-section, seamless, of stainless steel (excluding pipe, of a kind used for oil or gas pipelines) – 228,700 tonnes/[167];
- Precision tubes and pipes, of circular cross-section, cold drawn or cold-rolled, seamless of steel, other than stainless steel – 544,200 tonnes/[168]

[163] Source:– annual data (DS-066341) – 22212153 - Rigid tubes pipes and hoses of polymers of ethylene
[164] Source:– annual data (DS-066341) – 23321300 - Ceramic pipes, conduits, guttering and pipe fittings
[165] Source: annual data (DS-066341) – 23691930 - Pipes of cement, concrete or artificial stone
[166] Source: annual data (DS-066341) – 24201110 - Line pipe, of a kind used for oil or gas pipelines, seamless of stainless steel
[167] Source: annual data (DS-066341) – 24201310 - Tubes and pipes, of circular cross-section, seamless, of stainless steel (excluding pipe, of a kind used for oil or gas pipelines
[168] Source: annual data (DS-066341) – 24201330 - Precision tubes and pipes, of circular cross-section, cold drawn or cold-rolled, seamless of steel, other than stainless steel

- Aluminium tubes and pipes (excluding hollow profiles, tube or pipe fittings flexible tubing, tubes and pipes prepared for use in structures, machinery or vehicle part, or the like) – 30,500 tonnes/[169];
- Copper tubes and pipes – 405,800 tonnes[170];
- Tube or pipe fittings of malleable cast iron - 58,900 tonnes*[171]

Precise changes in demand will depend on the timescales and scope of any potential PVC phase out as well as the timescales for development of new alternative formulations and the extent to which they are required. This is discussed in chapter 8.

7.4.7 Conclusion on suitability and availability of alternatives

There are technically and economically feasible alternatives to PVC for the applications in piping assessed. These are commercially available and have been used extensively for many years, in several cases. An overall reduction in human health and environmental risk is judged to be likely, primarily given the presence of organotin additives in water pipelines, as well as titanium oxide and phthalates and their exposure throughout their lifecycles (Adams, et al., 2011; Sadiki & T. Williams, 1999; Tomboulian, et al., 2004). An overall indicative assessment, based on the information summarised above, is below (Table 7-5).

Table 7-5: Overall Suitability

Substance	Technical Feasibility	Economic Viability	Potential indicative risk	Overall Performance
PVC	Good	Good	Fair	Good
HDPE/MDPE/LDPE	Good	Good	Fair	Good
Glass Fiber Reinforced PP	Insufficient data	Not considered further	Not considered further	Not considered further
PB	Likely poor	Not considered further	Not considered further	Not considered further
PE-X	Good	Good	Fair	Good
ABS	Good	Good	Fair	Good
Stainless Steel	Good	Fair	Low	Good
Galvanised Steel	Fair	Fair	Fair	Fair
Copper	Fair	Fair	Fair	Fair
Brass	Fair	Good	Low	Good
Aluminium	Fair	Good	Low	Good
Cast Iron	Likely poor	Not considered further	Not considered further	Not considered further
Ductile Iron	Good	Good/Excellent	Fair	Good
Concrete	Good	Fair	Low	Good
Vitrified Clay	Fair	Fair	Good	Good

7.5 Analysis of alternatives – PVC in Window Frames

7.5.1 Key performance criteria

Window frames need to be:

- Made of **strong, durable, waterproof** materials (Modernize Home Services, n.d.)
- **Resistant to pressure and warping** from temperature changes (Eco Home, 2021)

[169] Source: annual data (DS-066341) – 24422630 - Aluminium tubes and pipes (excluding hollow profiles, tube or pipe fittings flexible tubing, tubes and pipes prepared for use in structures, machinery or vehicle part, or the like)

[170] Source: annual data (DS-066341) – 24442630 - Copper tubes and pipes

[171] Source: annual data (DS-066341) – 24513050 - Tube or pipe fittings of malleable cast iron

- **Fire resistant** to limit the risk of fire spread between adjacent properties (Government of the United Kingdom, 2010)
- Able to support **energy efficiency** by preventing of heat loss from buildings (AEA, Harwell, 2010).

PVC is extruded into the required frame before fitting with glass (Modernize Home Services, n.d.). Whilst these frames offer good heat insulation performance and versatility (Everest, 2017), UV rays can cause the vinyl to break down and release dioxins into the air. Additionally, at temperatures lower than -10ºC, PVC window frames can start to contract, and the seals can detach. At high temperatures, PVC frames are prone to bending and warping (Eco Home, 2021). They may become discoloured because of exposure to sun (Barbour, 2017).

PVC window frame manufacturers state that high quality PVC window frames that are installed correctly can last over 35 years (The London Economic, 2018). Further evidence shared by the participants of the focus group from a study on PVC windows demonstrated that no detectable erosion of PVC was identified after more than 40 years of natural weathering (Elias, 2019). The study also presented the results of a simulated weathering test showing an expected live time of 'several decades'. In comparison, engineered timber regularly maintained can last up to 40, aluminium frames over 45 years and fiberglass windows more than 50 years (Elias, 2019). However, in many cases window frames are replaced in advance of the maximum lifetime use for reasons of fashion/consumer preference, irrespective of material.

7.5.2 Substance ID and properties (or Description of alternative technique)

Desktop research indicates that the main alternatives to PVC window frames are timber, aluminium, "clad-wood" and fiberglass. These are discussed below.

Table 7-6: Potential alternatives to PVC in flooring

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
Timber	The most common timber types used in window frames are oak, sapele, redwood and Acoya© (CAD Joinery, n.d.).	Timber windows frames are usually treated with wood preservative, including resins and waxes so that paint can be applied more easily and to make the frame more resistant to water. Fungicides consisting of inorganic salts of boron such as disodium octaborate, propiconazole or tebuconazole are also applied (WWF, 2005). In some cases, timber may also be impregnated with flame and fire retardant chemicals (Think Smart Think Green, n.d.). Timber frames need to be painted or varnished as protection against rotting and swelling. Paints and varnishes commonly contain solvents and emit VOCs, however, there are some 'Free from VOCs & Solvents' alternatives on the market (Lakeland Paints, n.d.) (The Wood Window Alliance, 2017) .
Aluminium	CAS no.: 91728-14-2. Aluminium is one of the strongest alternative materials for window frames, which means that they can be thinner and lighter compared to other alternatives, including PVC.	There are no notified hazards to users for aluminium window frames (European Chemicals Agency, n.d.). However, aluminium has been recognized as a toxic agent to aquatic freshwater organisms, making its disposal potentially harmful to the environment if it is not reused/recycled (Eco Home, 2021). Unlike PVC, aluminium windows do not represent a health hazard

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
		while in use, and not emit toxic fumes when exposed to fire (Extrual n.d.). Nevertheless, recycling rates of aluminium are high, and aluminium can be recycled as many times as required without any loss of its inherent properties, since its atomic structure is not altered during melting.
Fiberglass	Fiberglass is made with resin and glass fibres to add strength.	Unlike PVC, fiberglass windows don't emit toxic fumes in case of fire (Kenny Leung, Chun Gordon, Yeh Daniel Liu, 2010). Fiberglass windows can be painted. Common paints contain solvents and emit VOCs, however, there are some 'Free from VOCs & Solvents' alternatives in the market (Think Smart Think Green, n.d.). Fiberglass windows are inert after disposal (European Chemicals Agency, n.d.).
Clad-wood	Wooden windows with an aluminium-clad, PVC-clad or fiberglass-clad facing the exterior.	Clad windows combine timber with a protective layer of exterior cladding using aluminium, PVC or fiberglass. The properties are therefore a combination of the materials employed.

7.5.3 Technical feasibility

The alternatives to PVC window frames presented above are currently commercially available and commonly used for this type of application. In some cases, these materials have been used for many years before the use of PVC window frames were introduced. Historically, most window frames were made of wood. Steel window frames came into use in the early part of the 20th century, followed by aluminium frames in the post-war period. PVC and fiberglass window frames were introduced in the early 1980s (Everest, 2017) (Kess-Kosa, 2017).

Some evidence suggests that whilst PVC window frames are durable, they can be less so than alternatives, including fiberglass, aluminium and engineered timber[172] (Everest, n.d.) frames (Fixr n.d., Windows on Washington n.d.). Overall, the alternative materials currently available come with specific advantages and disadvantages making them more or less suitable for specific designs or weather conditions. These are summarised in the table below.

[172] Timber frames require periodic maintenance to increase their durability

Table 7-7: Main characteristics and ratings of alternative window frame materials

Alternative	Durability and strength	Resistant to weather conditions and humidity	Fire resistance	Thermal diffusivity	Other (ease of installation, cleaning etc)
Timber	Engineered timber frames will not warp or bow and are resistant to rot and fungus. With proper maintenance, some manufacturers claim that wood windows can last up to 60 years (House Beautiful, 2016; Windows Guide, n.d.). Wood also has the smallest coefficient of expansion than any other material used in window production (DenGarden, 2021).	Timber windows need to be painted or varnished to withstand humidity and insect invasions. This has to be done on installation and then periodically to maintain the protective coating of the frame and extened its durability (Barbour, 2017).	Timber window frame can be impregnated with flame and fire retardant chemicals. Treated timber frame can withstand up to 120 minutes of exposure to fire (WWF, 2005).	Lowest thermal conductivity of all window frame materials. It is a natural insulator that absorbs and retains heat (Eco Home, 2021). (Fixr, n.d.)	This material can be painted in any colour and is often the preferred option in period properties and conservation areas. However, unlike other materials, timber frames need regular maintenance (Barbour, 2017)..
Aluminium	Aluminium is a light material, but still strong. It doesn't require much maintenance and is resistant to corrosion and decay (Barbour, 2017). Aluminium frames will not be affected by changes in temperature and will not flex, twist, expand or contract (Origin, n.d.)	Aluminium windows may be prone to condensation due to their high thermal conductivity. Moisture can lead to the growth of fungi, (Barbour, 2017).. In coastal areas, aluminium frames can eventually corrode (Excellent Windows, n.d.). However, it is possible to apply a special coating to protect the frames from the more corrosive coastal environments (Hedgehog Aluminium Systems, n.d.).	Non-flammable material with a melting temperature of over 600°C (Reynaers Aluminium, n.d.). When aluminium is exposed to a prolonged fire environment it will begin to melt (not burn). Its high thermal conductivity allows it to quickly dissipate large amounts of heat from the flame and absorb more thermal energy from the centre of the fire, 'cooling' the environment and restricting 'very hot spots'.	Aluminium has the highest thermal conductivity of all window frame materials. They can be fitted with a thermal break and spacers to improve their thermal efficiency (Origin, n.d.). Thermal breaks are usually made of polyamide plastic (Teenou, 2012).	Aluminium window frames require very low maintenance over time. When cleaning the windows no specialist cleaning substances, treatments or techniques are required (Hedgehog Aluminium Systems, n.d.).
Fiberglass or composite	Fiberglass can be up to eight times stronger than PVC and has a similar strength to aluminium (CAD Joinery, n.d.). Its lifespan can be up to 50+ years. Fiberglass maintains its integrity at extreme temperatures.	Fiberglass is not susceptible to decay, mildew, UV degradation, or corrosion, which makes it suitable to different climates and environments (Cascadia, n.d.).	Fiberglass is naturally fire resistant and is rated to withstand temperatures up to 540°C before it will melt (Firefighter Insider, n.d.).	Fiberglass can be up to 15% better at insulating than vinyl (Fixr, n.d.).	Fiberglass is more rigid than PVC, which means that it can be more difficult to fit into the window opening, taking more time to install. Additionally, fiberglass windows should be installed by a professional (Fixr n.d.). They may fade or peel and need to be repainted periodically.
Clad-wood	Clad windows combine timber with a protective layer of exterior cladding using aluminium, PVC or fiberglass. The overall properties are therefore a combination of the properties of timber and the particular cladding material in use.				

7.5.4 Economic feasibility and economic impacts

All the alternative materials for window frames have been commercially available prior to or at around the same time as PVC. However, the alternatives are more expensive on a unit cost basis. Whilst PVC window frames require low maintenance, they are difficult to repair for the purpose of extending their lifetime. It has been described as a similar or slightly less durable material compared to most alternatives[173], typically requiring replacement after 40+ years (if installed correctly) after installation, compared with longer lifespans for other materials (Table 7-7)

- Timber windows are the most expensive alternative, but they also have higher thermal insulation properties, which can help improve energy efficiency of the property, lowering energy costs. Nevertheless, periodic maintenance to the frames, including repainting or revarnishing, is essential to ensure their durability.
- Aluminium is a durable alternative with low maintenance costs but higher initial investment. However, these frames are thermally inefficient and therefore not appropriate in extreme weather conditions. New technologies such as built-in thermal breaks are attempting to address this issue but installing these would likely further increase the price of the frames.
- Fiberglass window frames are a comparatively costly alternative to PVC, and this material is not currently as widely used as the other alternatives. Once more companies start to sell these window frames, they will likely lower in price (Love to Know, n.d.), alongside new technologies and improved pultrusion[174] processes (Composites World, 2012). In addition, these windows will lower energy costs and may last almost double the typical lifetime of PVC.
- The price of both timber and aluminium higher than any other alternative, however, they have lifetimes of 40 years or more. Aluminium windows require low maintenance and are easy to repair (Aluminium Trade Supply n.d.). The combination of both materials usually increases the durability of the frame, however, it may result in increased difficulties during the recycling process at the end of their lifetime (Weather Shield 2015).

The prices, durability and maintenance characteristics for different window frame materials were obtained through a web research of data published by window manufacturers and specialists and presented in the table below.

[173] Apart from timber frames, which can have a similar durability, depending on mantainence activities. However, if adequately maintained, wooden window frames can be longer than PVC frames.

[174] Pultrusion is a process where raw materials (in this case glass fibre reinforcements and resins) are drawn into a profile die by a mechanical pulling force to form a window frame.

Table 7-8: Comparison of cost, durability and maintenance of alternative window frame materials

Material	Price range		Durability (years)	Maintenance and repairability
	Average Low	Average High		
PVC (Windows Guide, n.d.; Doubleglazing Pro, 2021; UPVC Windows Fitted, 2021; Muhammad Asif, n.d.)[175]	€ 290	€ 530	25-40	Low maintenance, difficult to repair
Timber (Windows Guide, n.d.; Doubleglazing Pro, 2021; UPVC Windows Fitted, 2021; Muhammad Asif, n.d.)	€ 900	€ 1,090	40 (with regular maintenance)	High maintenance, easy to repair
Aluminium (Windows Guide, n.d.; Doubleglazing Pro, 2021; UPVC Windows Fitted, 2021; Muhammad Asif, n.d.; The Eco Experts, n.d.; Marselli, n.d.)	€ 630	€ 840	45	Low maintenance, easy to repair
Fibreglass (Get a Window, n.d.; McCarter Construction, 2017)	€ 700	€ 820	50	Low maintenance, easy to repair
Clad-Wood	Dependant on clad material (comprable price data not identified)			
Notes: Prices for a for a 600 x 900 mm casement window averaged from listed sources including installation and VAT. Note the original price data was obtained in £ Sterling, these have been converted using the following exchange rate: **GBP 1.17 per EUR**.				

7.5.5 Reduction of overall risk due to transition to the alternative

The alternative materials for window frames generally present lower health and environmental risks compared to PVC but are not free of concerns and greater use may involve trade-offs. The Timber Research and Development Association (TRADA) estimates that it takes more energy to manufacture a PVC window than an equivalent timber frame, although the source presents limited supporting data (Student Conservation Association, n.d.). Ensuring / supporting sustainable sourcing of timber are a further consideration, discussed below in section 7.5.6 and in section 7.7. on flooring. Timber frames also have a reuse potential currently greater than PVC waste (European Aluminium, 2019). For instance, in 2018 the global consumption of post-consumer recovered wood exceeded 27 million tonnes (Food and Agriculture Organization of the United Nations, n.d.). A further risk associated with timber window frames use is related to additives that are applied as protection against rotting, swelling and damage by insects. Paints and varnishes may contain solvents and emit VOCs, however, there are some 'Free from VOCs & Solvents' alternatives in the market (Lakeland Paints, n.d.; Think Smart Think Green, n.d.). Timber window frames may also incorporate flame retardant chemicals, as do PVC window frames.

The extraction of aluminium has significant environmental impacts, such as the removal of native vegetation in the mining region, resulting in a loss of habitat and food sources for local wildlife as well as significant soil erosion. The caustic red sludge and toxic residuals that remain are commonly deposited into excavated mine pits where they can leach into aquifers, polluting water sources (Recycle Nation 2010). However, these could be mitigated by post-mining rehabilitation and efficient recycling. It has a high impact in terms of the energy required during production. It has been recognised as a toxic agent to aquatic freshwater organisms, making its disposal potentially harmful to the environment (Adams, et al., 2017; Poléo, et al., 1997). However, it can be recycled as many times as required without any loss of its inherent properties, since its atomic structure is not altered during melting. Around 75% of the aluminium that has ever produced is still in use (Student Conservation Association, n.d.; European Aluminium, 2019), with between 92% and 98% of the aluminium used in building construction recycled (European Aluminium,

[175] Bernhard Elias, RAL-quality certification for PVC window profile systems (2019), Joint research report influence of natural weathering on the durability of the outer surface of a PVC-window-profile, Bonn, Germany

2019). Recycling aluminium requires only 5% of the energy needed to produce aluminium from bauxite (Student Conservation Association, n.d.; Defra, 2010). It is important to consider, however, that recycled aluminium has the potential to contain lead (ThermoFisher Scientific, 2017).

For fiberglass window production, the embodied energy used to extract the glass from the sand and convert the raw materials through pultrusion is comparatively low. Sand mining is also considered to be more sustainable compared to other materials, however, large sand extractions can accelerate the erosion of riverbanks and coastal areas, destabilise bridges or constructions, change the flow of rivers and increase the risk of flooding by eliminating buffers against storm surges (Chatham House, 2019).

On the other hand, fiberglass has a long lifetime, is very durable and does not degrade (Green Home Guide, n.d.). This material does not require additional substances to reinforce them. As a thermally set inert material, it will not out-gas or emit any VOCs over its lifespan, which makes it a safe alternative during usage. However, some fiberglass windows are painted and this may expose users to VOCs, although 'Free from VOCs & Solvents' alternatives are on the market (Lakeland Paints, n.d.; WWF, 2005). If placed in landfill, it will not leech chemicals into the ground or waterways (Fenestration & Glazing Insustry Alliance, n.d.). Information on recycling rates for fiberglass window frames is limited - they were introduced in the 1980s and their life expectancy is over 50 years. It can, however, be recycled using a two-stage grinding process. Downcycled materials, such as a fine bi-product that can be utilised as a filler building components such as concrete and asphalt, appear common (Fenestration & Glazing Industry Alliance, n.d.).

The table below provides an overall – approximate - comparison of the embodied energy, recyclability and disposal hazards of different window frame materials.

Table 7-9: Health and environmental impact of window frame materials

Material	Hazardous additives	Embodied energy	Recyclability	Disposal hazards
PVC	Higher potential	Higher potential	Lower potential	Higher potential
Timber	Medium potential*	Lower potential	Medium potential	Lower potential
Aluminium	Lower potential	Higher potential	Higher potential	Higher potential – where it is disposed of**
Fibreglass	Lower potential	Medium potential	Lower potential	Lower potential
Clad-Wood	Dependant on clad material			

Source: https://www.wwf.org.uk/sites/default/files/2017-06/windows_0305.pdf
* Can be reduced to low if a free from VOCs and Solvent paint is used
**92% and 98% of the aluminium used in building construction is recycled (McCarter Construction 2017)

7.5.6 Availability

Timber from well-managed forests is considered to be a renewable resource. However, many of the world's forests are not felled sustainably or legally. According to the latest figures from the UN's Food and Agriculture Organization (FAO) (Food and Agriculture Organization of the United Nations, n.d.), global forest production hit record levels in 2018. This increased demand is not only due to a rise in wood-based products, but because of the replacement of fossil fuels with biofuels. Globally, sawn wood production grew by 2% in 2018, reaching a record high of 188 million tonnes. It is crucial that timber used in window manufacture or any other application has been certified as coming from a well-managed source by an independent body, to prevent deforestation and habitat deterioration (Food and Agriculture Organization of the United Nations, n.d.). Sustainably managed forests also play an important role in storing carbon emissions in leaves, branches and soil (Teenou, 2012).It is crucial that timber used in window manufacture or any other application has been certified as coming from a well-managed source by an independent body, to prevent deforestation and habitat deterioration (Food and Agriculture Organization of the United Nations, n.d.). Sustainably managed forests also play an important role in storing carbon emissions in leaves, branches and soil (Teenou, 2012). It is important to consider that there are other applications competing for the availability of wood in the market, such as wooden furniture, wooden flooring, wood charcoal and wood fuel (Eurostat, 2020).

Aluminium is one of the most abundant elements and the most common metal found on Earth (comprising 8% of the planet's crust) (Muhammad Asif, n.d.). Bauxite is the world's main source of aluminium (The Aluminium Association, n.d.). Recycled aluminium volumes have increased steadily in recent years. Similarly, the stock of aluminium in usage is increasing rapidly, but the secondary aluminium available from products at end-of-life is growing at a slower rate, due to the long lifespan of aluminium applications. As this limited recycled volume cannot meet the increasing demand for aluminium, the shortage must be met by the primary aluminium industry (European Aluminium, 2019). Therefore, a rise in the demand for aluminium window frames may well require increased primary aluminium production, which has a high environmental impact, as explained in section 7.5.5. Other competing aluminium-based applications include power lines, structures for high rise buildings, consumer electronics, cans, foils and aeroplane, ship and train parts (Flagel, 2020).

The main ingredient in fiberglass is glass. Glass is made from sand, which is an abundant natural and non-depleting resource. Largescale extraction of sand can have negative environmental impacts as explained in section 7.5.5.

The available data indicates that all the alternative materials are available in significant quantities. In terms of the size of the market for wood and aluminium window frames, detailed information on EU production, import and export volumes is available via PRODCOM (European Commission, 2021a), showing the following production volumes for 2019:

- Wooden window frames - approximately 16.9 million units
- Aluminium window frames - approximately 44 million units
- No data was reported for fibreglass window frames in PRODCOM but, according to the European Glass Fibre Producers Association (APFE), EU producers supply about 60% of the European market. Non-EU countries, excluding China, supply just over 20% of the market, while China supplies the remaining 20%. EU producers are operating at about 80% of their capacity due to placement of highly subsidised imports from China in the EU. Therefore, capacity to increase production appears to be feasible.

7.5.7 Conclusions on suitability and availability for alternative

The alternative materials for window frames presented are currently technically and economically viable substitutions for PVC windows. They are typically more expensive at the point of sale, however most alternatives last longer than PVC window frames, which can offset this cost. They are commercially available in significant quantities. The available alternatives would likely result in some environmental trade-offs. In the case of aluminium window frames, it would be important that an increased extraction of the mineral is paired with post-mining rehabilitation strategies, responsible production practices and efficient recycling. For timber windows, it is crucial that the wood comes from a sustainably-managed source.

7.6 Analysis of alternatives – Cables

7.6.1 Key performance criteria

Wires and cables are widely used for power transmission and information and data transfer. Specific uses include household electricity cables for domestic appliances and mains electricity, high voltage underground electricity transmission lines for in long-distance distribution, to coaxial cables for relaying radio frequency signals, ethernet cables in internet networks, and various connector cables used with Information Communication Technology (ICT) devices (Electrical Technology 2020).

While many different forms of cable are used to meet particular requirements, they are constructed similarly, consisting of a flexible wire of conductive material used for data or power transfer, an insulating layer covering the conductor, and, in many cases, a 'jacket' forming a protective outer coating. Plastics, including PVC, are widely used as wire insulators and as cable jackets, meeting the following key technical requirements:

- **Electrical insulation** - the material must provide strong electrical insulation to enable safe handling and use of the wire/cable;
- **Fire resistance** – the insulation and protective layer should be fire resistant and/or self-extinguishing to so as to prevent electrical fires;
- **Chemical resistance** – insulation and outer casings must not be susceptible to chemical degradation from a variety of substances. The suite of substances that cables and wires must be resistant to depends on their particular use;
- **UV resistance** – where cables are deployed outdoors, the insulation and jacket must be resistant to solar UV radiation and other environmental stresses;
- **Durability** – insulation and coverings must provide a tough protective layer that protects the conductive wire from damage, moisture and exposure;
- **Flexibility** – insulating and coating materials must not be too rigid so as to inhibit flexible use of wires and cables; and
- **Temperature range** – depending on the application, electrical wires may be required to operate at a particular temperature range. Insulating and casing materials must be able to withstand this.

7.6.2 Substance ID and properties

Desktop research identified that the alternatives to PVC for use in wires and cables broadly fall into three categories: thermoplastics, thermosets and elastomers (Table 7-10).

Table 7-10: Potential alternatives to PVC in electrical cables

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
Polyethylene (PE)	9002-88-4	A semi-crystalline polymer with poor fire resistance. Fire resistance, as well as resistance to UV radiation, weathering and chemical degradation, is often augmented by the addition of fillers (including both halogenated and halogen-free additives such as plasticisers, fire retardants, stabilisers and pigments) (Eland Cables, 2021). Where greater flexibility is required, ethylene propylene rubber (EPR) can be added to the polymer (Tycab Australia, 2021).
Chlorinated polyethylene (CPE)	64754-90-1	A variation of PE (above) with hydrogen atoms substituted with chlorine atoms. It has improved chemical and fire resistance compared to PE, and is very durable (Anixter, n.d.). Emissions of carbon monoxide and hydrogen chloride can arise from combustion (GreenSpec, 2021). Plasticisers are often used to improve flexibility, primarily dioctyl adipate (DOA; CAS No: 123-79-5) and dioctyl sebacate (DOS; CAS No: 122-62-3) (VIA Chemical, 2021). Frequently used

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
		stabilisers include sodium stearate (CAS No: 822-16-2), barium stearate (CAS No: 6865-35-6), magnesium oxide (CAS No: 1309-48-4). DOA and sodium stearate are both known to cause serious eye and skin irritation. Barium stearate is harmful if swallowed or inhaled.
Polypropylene (PP)	9003-07-0	Resistant to stress cracking. Tough but semi-rigid thus impairing flexibility. Resistant to a wide variety of chemicals, including acids, bases, water and detergents. Highly flammable (Plastics Insight, n.d.).
Polyurethane (PUR)	9009-54-5	Excellent resistance to oxidation, ozone and chemicals. Some formulations have good fire resistance (Teenou, 2012), but PUR is generally highly flammable without additives (Galaxy Wire and Cable Inc, 2021).
Thermoplastic elastomers (TPE)	308079-71-2	Resistant to UV radiation, oxidation and atmospheric ozone. Generally good resistance to chemical damage, but susceptible to degradation when exposed to hydrocarbons. Good abrasion resistance (Teenou, 2012).
Modified polyphenylene ether (mPPE)	-	A thermoplastic material marketed as environmentally friendly and 100% recyclable (Galaxy Wire and Cable Inc, 2021) due to its lack of halogens.
Fluorinated ethylene propylene (FEP)	25067-11-2	Fluoropolymers are a group of fluorine-based copolymers including FEP, ETFE, PTFE and PFA. The polymerisation process is typcially completed in the presence of a fluorosurfactant from the group of per- and polyfluoroalkyl substances (PFAS). Long-chain PFAS, inclduing perfluorooctanesulfonic acid (PFOS; CAS No: 1763-23-1), perfluorooctanoic acid (PFOA; CAS No: 335-67-1) or perfluorononanoic acid (PFNA; CAS No: 375-95-1), are restricted under REACH for use as a processing aid to fluoropolymers and in several other uses. Short-chain PFAS are used in place of long-chain PFAS in the production of fluoropolymers. While they are assumed to have a lower bioaccumulation potential than long-chain PFAS, there is increasing evidence to suggest that short-chain PFAS are also highly persistent (Brendel, et al. 2018). Short-chain PFAS are also linked to cryotoxic and neurotoxic effects and endocrine disruption (Danish Environmental Protection Agency 2015). Fluoropolymer alternatives to PVC in wires and cables include: • FEP - a highly UV- and chemical-resistant copolymer of hexafluoropropylene and tetrafluoroethylene (Fluorocarbon, 2016); • ETFE – a highly corrosion-resistant partially fluorinated copolymer of ethylene and tetrafluoroethylene (TFE) (Omnexus , 2021); • PTFE – a durable and highly chemical-resistant polymer of tetrafluoroethylene, which is also resistant to UV damage (Omnexus, 2021); and • PFA – copolymers of tetrafluoroethylene and perfluoroethers with high resistance to chemical damage and UV-resistance (Fluorocarbon, 2016)
Ethylene tetrafluoroethylene (ETFE)	25038-71-5	
Polytetrafluoroethylene (PTFE)	9002-84-0	
Perfluoroalkoxy alkanes (PFA)	-	
Cross-linked polyethylene (PEX)	-	A thermoset variation of PE (above) with an altered, cross-linked chemical structure that increases the temperature range of the polymer. The cross-linked structure also improves fire resistant properties without the need for halogenated additives, but makes recycling more difficult at the end of the products lifecycle, as the cross-linked material cannot be easily returned to its non-cross-linked state for reuse (UK Research and Innovation n.d.).
2-Chlorobuta-1,3-diene (Chloroprene; CP)	126-99-8	A vulcanised synthetic rubber resistant to UV degradation, is flame resistant and self-extinguishing. Strong and abrasion

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
		resistant. According to the harmonised classification and labelling approved by the EU, the substance may cause cancer, is highly flammable in liquid and vapour form, is harmful if inhaled, may cause damage to organs through prolonged and repeated exposure, causes skin irritation and may cause respiratory irritation. Additionally, the notified classifications identifies that this substance is toxic if swallowed and is toxic to aquatic life with long-lasting effects (European Chemicals Agency, 2021).
Ethylene propylene diene monomer (EPDM)	25038-36-2	A halogen-free elastomeric comonomer of ethylene, propylene and diene, cross-linked via a vulcanisation process. EPDM is highly flexible and abrasion resistant, and its fire resistance can be enhanced through formulation (Anixter , n.d.)·
Ethylene-vinyl acetate (EVA)	24937-78-8	A halogen-free elastomer. Highly resistant to UV radiation and stress cracking. Poor resistance to aromatic hydrocarbons and halogenated hydrocarbons compared to other polymers (Plastics Insight, 2021).
Silicone rubber	63394-02-5	A silicone-based halogen-free elastomer offering excellent UV resistance, limited durability and poor scuff resistance (Eland Cables, 2021), but greater fire resistance compared to other elastomers (Eland Cables, 2021). Production of silicone rubber can involve use of silicone polymers which can contain cyclosiloxanes octamethylcyclotetrasiloxane (D4; CAS No: 556-67-2), decamethylcyclopentasiloxane (D5; CAS No: 541-102-6) and dodecamethylcyclohexasiloxane (D6; CAS No: 540-97-6). These substances are judged persistent in the environment, bioacccumulative and toxic (PBT) and have been placed on the REACH candidate list as substances of very high concern (SVHCs) by ECHA (Rubber and Plastic News, 2019). An Annex XV restriction report has been published proposing restrictions on professional and consumer uses of the monomers D4, D5 and D6(European Chemicals Agency , 2019).

7.6.3 Technical feasibility

Whilst each of the alternatives has specific advantages and drawbacks, the available evidence suggests there are technically feasible alternatives to PVC in cables. All of the alternatives to PVC outlined above are currently commercially available for use in wires and cables. They are all plastics or synthetic rubbers with the necessary insulating qualities required, but each perform differently. Table 7-11 summarises their performance against key technical criteria based on a review of technical literature.

PEX provides excellent resistance to fire on account of its thermoset chemical structure, while the fluorine in the molecular composition of fluoropolymers (ETFE, PTFE and PFA) contributes to a similarly high level of fire resistance. Thermoplastic polymers (PE, PP and PUR) perform more poorly in this regard, although performance can be augmented by the use of additives; chlorination of PE is noted as significantly improving its fire-resistant properties. Similarly, elastomers typically provide poorer fire resistance (in particular, EPDM and EVA), although silicone rubber performs better due its formation of a protective fused silica layer when exposed to fire (Omnexus , 2021).

Most of the assessed alternatives display a similarly high level of resistance to damage from acids and alkalis, and susceptibility to chemical damage from aromatic hydrocarbons (Plastics Insight, n.d.). EPDM is particularly susceptible to damage from hydrocarbon fuels, including oil and kerosene (Galaxy Wire and Cable Inc , 2021), limiting use in automotive, transport and some industrial cables. The alternatives are highly resistant to solar UV radiation making them suitable for use in outdoor environments. The alternatives are highly durable, although silicone rubber and polypropylene can be susceptible to abrasion damage. Elastomeric compounds are highly flexible, while PP requires additives to modify its otherwise rigid structure for use in flexible cables. Of the alternatives to PVC in wires and cables, CPE, TPE, mPPE, FEP, ETFE, PTFE, PFA, PEX and CPA are noted as performing to a good standard across all of the technical requirements. An overall indicative assessment is below.

Table 7-11: Technical specifications of PVC and alternative substances used in cables – Source, unless otherwise stated: (Galaxy Wire and Cable Inc, n.d.)

Substance	Nominal temperature ratings (°C)	Fire resistance	Chemical resistance	UV resistance	Durability	Flexibility
PVC	***-20 to 105***	***Good***	***Good***	***Good***	***Good***	***Good***
PE	-60 to 80	Lower	Fair	Good	Good	Good
CPE	-20 to 105	Good	Good	Good	Good	Fair
PP	-40 to 105	Lower	Good	Good	Fair	Lower
PUR	-55 to 80	Lower	Good	Good	Good	Good
TPE	-50 to 105	Good	Fair	Good	Good	Good
mPPE	-40 to 105	Good	Good	Good	Good	Fair
FEP	-80 to 200	Good	Good	Good	Good	Fair
ETFE	-100 to 150	Good	Good	Good	Good	Good
PTFE	-60 to 200	Good	Good	Good	Good	Good
PFA	-100 to 260	Good	Good	Good	Good	Good
PEX	-40 to 105	Good	Good	Good	Good	Good
CP	-20 to 90	Good	Good	Good	Good	Fair
EPDM	-55 to 125	Lower	Lower	Good	Good	Good
EVA (Maziyar Sabet, 2013)	-40 to 105	Lower	Fair	Good	Good	Good
Silicone rubber	-80 to 180	Good	Fair	Good	Fair	Good

7.6.4 Economic feasibility and economic impacts

All of the alternatives to PVC reviewed in this assessment are commercially available for use in cables, indicating they are economically viable in at least some cabling applications. A review of literature and online marketplaces was conducted in order to determine the main economic impacts associated with a transition to alternatives. Table 7-12 displays material costs for the various alternatives considered (per tonne). Prices for a specific product – in this case 1 m of 8AWG gauge hook-up wire[176] - was also obtained to illustrate how these prices may feed through the final consumer. For several alternatives, it was not possible to obtain reliable price information, but available data suggest wide ranges in material costs. In some cases, CPE and EVA for example, costs are comparable to PVC while other materials, notably PTFE, they are significantly higher, likely to prohibit viable use in some applications.

The market prices for cables indicate that PE, mPPE, ETFE, PEX and EPDM cable costs are similar to PVC cable costs. PTFE and silicone rubber cables are amongst the highest prices identified. A search for information on cable lifespans was conducted and, where information could be obtained, it is displayed in Table 7-12. The available data indicate that alternatives have the same, if not greater longevity, as PVC. For alternatives where lifespans could not be determined,

[176] This is an all-purpose type of electrical cable with a single insulated conductor used for connections in low voltage applications, such as domestic appliances.

Table 7-11 indicates that alternative materials generally offer the same or better levels of durability and resistance to UV and chemical damage as PVC, and as such their material lifespans are assumed to be similar to that of PVC, when used in similar operating conditions.

Table 7-12: Material and product costs, and cable lifespans associated with PVC and alternative materials

Substance	Material cost (EUR/t) **[177]	Cable cost (EUR/m) ***[178]	Cable lifespan
PVC	***€425 – €1,700***	***€0.40 - €1.70***	***Typically exceeds 25-30 year service life*** (Chaplin, n.d.)***.***
PE	No reliable information identified.	€0.84	No reliable information identified.
CPE	€850 - €1190	No reliable information identified.	No reliable information identified.
PP	No reliable information identified.	No reliable information identified.	Over 30 years (KIYOSHI KURAHASHI, 2006).
PUR	No reliable information identified.	No reliable information identified.	No reliable information identified.
TPE	€1,020 - €3,825	No reliable information identified.	No reliable information identified.
mPPE	No reliable information identified, although mPPE-insulation is confirmed costlier than PVC (AlphaWire , 2009).	€0.45 – €1.10	No reliable information identified.
FEP	No reliable information identified.	No reliable information identified.	No reliable information identified.
ETFE	No reliable information identified.	€0.45 - €1.45	In excess of 40 years (Architen, 2009).
PTFE	Approximately 8-10 times more than PVC (Anixter , n.d.).	€1.33 - €3.42	No reliable information identified.
PFA	No reliable information identified.	No reliable information identified.	No reliable information identified.
PEX	No reliable information identified.	€0.36 - €1.30	40-60 years at a 90 °C rated operating temperature (A. Alghamdi, 2020).
CP	No reliable information identified.	No reliable information identified.	No reliable information identified.
EPDM	No reliable information identified.	€1.25	No reliable information identified.
EVA	€850 - €1,390	No reliable information identified.	No reliable information identified.
Silicone rubber	No reliable information identified.	€1.09 - €3.02	No reliable information identified.

7.6.5 Reduction of overall risk due to transition to the alternatives

Four of the alternatives identified – FEP, ETFE, PTFE and PFA – are fluoropolymers, a subgroup within the broader chemical class of PFAS associated with several concerns. Environmental and health concerns relate to the production of fluoropolymers, which typically involves use of PFAS as emulsifiers in the polymerisation process. Since the introduction of REACH restrictions on the use of long-chain PFAS processing aids – including PFOS, PFOA and PFNA (European Chemicals Agency n.d.) – in the production of fluoropolymers, short-chain PFAS are increasingly used in the manufacture of fluoropolymers. There is increasing evidence to suggest that short-chain PFAS are also highly persistent in the environment (Brendel, et al. 2018), and they have been linked to a range of human health impacts including alterations to cell membrane properties, cryotoxicity,

[177] USD to EUR exchange rate of 0.85 used.

[178] Prices determined for one metre of 18AWG gauge hook up wire. Data obtained from https://uk.farnell.com/ on 7th July 2021. GBP to EUR exchange rate of 1.17 used.

neurodevelopmental impacts and endocrine disruption (Danish Environmental Protection Agency 2015). There is increasing evidence that these production processing aids are able to leach out of fluoropolymers during production and use (The Danish Environmental Protection Agency , 2018; Galaxy Wire and Cable Inc, 2021). PFAS are among the most persistent man-made substances once released in the environment and are highly bioaccumulative (European Chemicals Agency, 2021). As such, various regulatory actions has been taken, or are currently being considered to mitigate the risks from these substances, including current restriction activity on all PFAS (European Chemicals Agency, n.d.; Rudy Dams, 2016).

With the exception of CPE, CP and the fluoropolymers, all of the alternatives considered are halogen-free substances. Halogenated formulations, including PVC, have fire retardant properties, but can lead to emissions of toxic smoke and acid gases when exposed to fire (Axon Cable and Interconnect , 2010), which presents an issue where they are disposed of via incineration. This risk would remain where CPE, CP and fluoropolymers are used in place of PVC, but is not a concern for the other alternatives considered where halogen-based additives are avoided.

Like PVC, most of the alternatives considered will require additives when used in cables. These include pigments to alter cable colour, plasticisers such as phthalates to improve flexibility (although elastomers do not typically require plasticisers and can be used as an additive themselves to improve flexibility of thermoplastics including PE and PP (Tycab Australia , 2021)) and light and heat stabilisers as well as flame retardants (MICC Group, 2014). As with PVC, there are concerns that these additives can be released during the lifecycle of the cable, some of which possess hazardous characteristics; notably phthalates (Carl-Gustaf Bornehag, 2018; Kim G Harley, 2018). Table 7-10 notes that PP and PE often include plasticisers and stabilisers that are harmful to human health.

While industrial recycling of fluoropolymer waste and scraps generated during production is common practice, fluoropolymers in consumer articles are not widely recycled due to the technical difficulties associated with recycling polymers where fillers and additives are used (Pro-K Fluoropolymergroup , 2018). Therefore, at present, disposed fluoropolymers are generally landfilled. Landfilling of fluoropolymers is known to result in leaching of PFAS into the environment as well as release of microplastics (Lohmann, et al. 2020).

The effectiveness of waste incineration in destroying PFASs, and its potential to form other toxic fluorinated compounds, is not currently well understood (US EPA , 2020). A recent study indicated that only negligible amounts of PFAS can be found after the incineration of PTFE (depending on temperature and residence time). However, not all the fluorine and fluorinated gases were analysed (Aleksandrov, et al., 2019). Fluorinated gases such as CF_4, CHF_3 and C_2F_6 can be formed during the incineration of fluorinated compounds and are also classified as PFAS. Tetrafluoro methane CF_4 is the molecule which requires the highest temperature for its destruction with estimates being around 1.400 °C (US EPA , 2020). The waste incinerators in Europe operate at minimum at temperatures of 850 °C for household waste and at 1.100 °C for hazardous waste and as such may not be suitable for the adequate destruction of PFAS. Exact data for the emissions of fluorinated gases from waste incineration plants in Europe is lacking. As such a transition to these fluoropolymers from PVC , if it occurred, would bring new challenges in managing the hazards associated with their production, use and disposal, if indeed their fuure use was permitted; depending on the scope and timing of further regulatory action on these substances.

The use of additives in flexible plastic applications presents a similar obstacle to widespread recycling as those currently observed in PVC cables. Due to its thermoset chemical structure, recycling of PEX is not feasible using current recycling techniques (R Huuva, E. Ribartis, 2019). As such, alternative cable materials face the same technical end-of-life challenges as PVC, with incineration and landfill currently the most technically and economically feasible disposal methods. Consequently, the same issues of hazardous emissions during incineration and leaching during landfill are applicable to alternative materials. Of the assessed alternatives, only mPPE is widely marketed as an environmentally friendly cabling solution; it is halogen-free and free of heavy metal additives and phthalates (Alpha Wire , n.d.). No reliable information could be found on current recycling rates for mPPE.

7.6.6 Availability

All of the alternatives for PVC use in wires and cables assessed are commercially available. EU-27 material production and import data for 2019 for the alternatives to PVC have been obtained from Eurostat PRODCOM database (Eurostat, 2021) (European Commission, 2021a) (Table 7-13). As an illustration of scale, the data indicate the EU-27 production capacity and imports of PE and PP exceed current PVC production and imports. PUR production and imports are less than that of PVC, while fluoropolymers, EVA and silicones are produced and imported in much smaller quantities. Data are not available for TPE, mPPE, PEX, CP or EPDM. Our estimates indicate that the quantity of PVC used in cables is around 11% of the total mass; approximately 550,000 tonnes. Depending on the timescales of any potential phase-out – and taking account of the fact that several alternatives may be used in place of PVC – it is likely that increases in production and/or imports of alternatives could meet additional demand associated with a PVC phase-out without undue delay. The main competing applications for PE, PP and PUR are packaging, consumer goods, industrial applications and automotive uses (Omnexus, n.d.; Omnexus, n.d.).

Table 7-13: Material production and import quantities for alternatives to PVC in cables

Substance		**2019 production quantity (tonnes)** (European Commission 2021b)	**2019 import quantity (tonnes)** (European Commission 2021a)
PVC	***Primary unmixed PVC***	***4,960,064***	***504,285***
	Non-plasticised mixed PVC	***384,389***	***28,518***
	Plasticised mixed PVC	***796,204***	***47,560***
PE (including CPE)	**Primary linear PE (specific gravity <0.94)**	1,870,616	804,410
	Primary non-linear PE (specific gravity <0.94)	3,960,226	845,846
	Primary PE (specific gravity >=0.94)	5,987,404	1,947,229
Primary PP		10,435,915	1,370,022
Primary PUR		2,954,510	81,574
TPE		No data	
mPPE		No data	
Fluoropolymers (including FEP, ETFE, PTFE and PFA)		148,715	36,129
PEX		No data	
CP		No data	
EPDM		No data	
Primary EVA		400,000	99,922
Primary silicones (including silicone rubber)		712,320	208,583

7.6.7 Conclusion on suitability and availability for alternatives

The analysis of alternatives to PVC in wires and cables has found that a variety of technically feasible alternatives exist which possess acceptable, similar or better material properties to PVC. There is a lack of reliable economic and cost information on every alternative, although available data suggest at least some alternatives have similar costs to PVC, while others are significantly more. Alternatives are generally as durable, if not more so, than PVC suggesting shortened lifespan and more frequent replacement would not represent major issues, depending on the specific needs of the application.

The alternative substances assessed pose many of the same risks associated with hazardous additives and halogens, their potential release through use, incineration and landfilling, and the technical difficulties currently faced in recycling flexible plastics with a high additive content. In addition, the fluoropolymers considered in this evaluation present risks in the form of potential release of hazardous PFAS used during polymer production, specifically PFOS, PFOA and PFNA. These risks may be addressed in a "broad" REACH restriction proposal for PFAS by the Competent Authorities of Germany, Netherlands, Norway, Sweden and Denmark (European Chemicals Agency, 2020).

There is evidence to suggest that mPPE is an additive-free, halogen-free, 100% recyclable alternative, although current levels of mPPE recycling are not known. A review of the volumes of alternative materials currently placed on the EU-27 market suggests that alternative materials are used in sufficient quantities so that a phase-out of PVC could be met, potentially requiring increased material production and/or import of alternatives, depending on the timescales for any potential phase out.

7.7 Analysis of alternatives: PVC in Flooring

7.7.1 Key performance criteria

PVC flooring is available in planks, tiles, rolls and sheets (Flooring Stores, 2021). Key performance criteria include:

- **Durability** – although PVC can be damaged with sharp, heavy objects (Great Mats, 2021)
- **Flame retardancy** (Dugdale PVC, n.d.) - including ignitability, ease of extinction, flame spread, heat release) (The Vinyl Institute, 2017)
- **Water/moisture repellence** - where used in bathrooms and kitchens (DumaPlast, n.d.). However, some manufacturers do not recommend PVC use in extremely wet areas such as showers or pools, given the risk of moisture entering the bevels and in certain areas with very high temperatures, such as conservatories (Quick-Step, n.d.).
- Other benefits noted by manufacturers include **ease of installation, maintenance and cleaning** (Quick-Step, n.d.) **and sound and thermal insulation** (Great Mats, 2021).

There are general recommendations on fire protection in the EU via consistent technical standards for structural design; the Eurocodes (European Commission, n.d.). The Construction Products Regulation also set out seven basic requirements for construction works, which includes safety in case of fire. No evidence has been identified that specifies a legal requirement for the use of PVC for flame retardancy/repellence were used in flooring/construction products. Indeed, a European Commission smoke toxicity study for materials used in construction, cited a test in which flammable mattresses provided sufficient heat and radiation to ignite PVC flooring that then become the main source of fire and smoke (European Commission 2017)[179].

[180] See for example: https://www.sensoboden.de/anwendungen/krankenhaus

7.7.2 Substance ID and properties (or Description of alternative technique)

Desktop research identifies several alternatives for PVC in flooring. These fall into three broad categories, for each we provide a brief description along with notes on key properties:

- Alternative materials, such as Wood, linoleum, tiles, stone, rubber, carpet and cork.
- Plastic alternatives, or (including examples of phthalate free PVC).
- Biobased plastics.

Table 7-14: Potential alternatives to PVC in flooring

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
Wood/hardwood	Wooden floors based on solid lengths of hardwood. Materials include Oak (Flooring Supplies, n.d.; MyMove, 2021; Flooring 365, n.d.), Walnut, Cherry, Birch Maple, Ash, or Pine, including reclaimed Wood (Flooring 365, n.d.). Bamboo is also commercially available (the spruce, 2021).	Various finishes are typically applied, which include varnishes, polishes, wax oils. These typically contain high levels of VOCs, although" non-toxic" alternatives are available (The Organic & Natural Paint Co., n.d.).
Laminate	A multi-layer synthetic material which can be designed to look and have the same texture as wood. Primarily manufactured form melamine resin and fibre board (Floor Techie, 2019).	Some sources indicate the adhesives used in laminates can contain formaldehyde (Floor Techie, 2019) and VOCs. Although manufacturer data indicates this is not common amongst in North American or European producers (Coswick. Inspire, 2012).
Linoleum	Made primarily from linseed oil, pine rosin, sawdust/cork dust, limestone, and jute (for the backing) (The Flooring Group, n.d.).	Manufacturer's literature suggests titanium dioxide (CAS No: 13463-67-7) a suspected carcinogen (European Chemicals Agency, n.d.) can be used as the pigment, alongside small amounts of zinc based drying agents and a topcoat of acrylic are used. The manufacturing process also results in emissions of VOCs (The Flooring Group, n.d.).
(Ceramic, porcelain, glass or cement) Tiles (Simple Home Simple Life, n.d.)	Flooring tiles are available from a range of materials. Ceramic tiles are made from a mixture of clay, sand, quartz (Designing Buildings, 2021); Porcelain tiles have a higher density and are fired for longer but are otherwise very similar (Designing Buildings, 2021). Cement or concrete floor tiles have been available for many years (Concrete Flooring Solutions, n.d.).	Various finishes can be applied (e.g., polished) and some manufacturers include polyurethane varnish (Festfloor, n.d.).
Stone	Stone flooring tiles can be made from limestone, sandstone, slate, marble, travertine or "engineered" (Floors of Stone, n.d.).	Several materials require the addition of waterproof sealants, and some are set in a base of epoxy resin (Urban Customs, n.d.).
Rubber (European Chemicals Agency, n.d.)	Both natural (latex) and synthetic rubber (manufactured using styrene-butadiene rubber (SBR), nitrile rubber and butyl rubber as common monomers) compounds are used in flooring (The Rubber Floor Store, 2013; All the Floors, n.d.; the spruce, 2020; Coruba, 2016).	SBR is a copolymer of butadiene and styrene (CAS: 100-42-5), Styrene content is typically between 10% and 25% (Polymer Properties Database, n.d.). Styrene is suspected of being toxic to reproduction (European Chemicals Agency, n.d.).
Carpet (Health Care Without Harm, n.d.)	Typically manufactured by weaving fibres of polypropylene (CAS No: 9003-07-0), polyester (CAS No: 25037-45-0), acrylic, nylon (nylon-66 is the most common polymer type CAS No 32131-17-2), and	There are no harmonised classifications or notified hazards for Polypropylene, polyester or (Nylon 66). Polyester/PET carpet can contain recycled plastic bottles (the spruce, 2021).

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
	wool blends. An adhesive coating and secondary backing can then be applied to hold the fibres in place (https://www.jhscarpets.com/carpet-manufacturing, n.d.; the spruce, 2021).	
Cork	Natural cork is harvested from trees including in commercial plantations.	In flooring it is ground and bonded with "resins" (the spruce, 2021), some literature indicates these are urethane (CAS 51-79-6, a carcinogen) binders (Natural Cork, n.d.). Some cork flooring includes a layer of MDF.
Plastic alternatives	Several plastic alternatives are commercially available. These include polyurethane (PU) and polyethylene terephthalate (PET – CAS No: 25038-59-9) (Building Green, n.d.; European Chemicals Agency, n.d.).	PU involves reacting a polyol with a diisocyanate or a polymeric isocyanate in the presence of catalysts and additives (American Chemistry Council, n.d.) Isocyanates are potent sensitisers, and are associated with asthma (Yale School of Medicine n.d.) as well as rhinitis, dermatitits and hypersensitivity pneumonitis (Lockey, et al. 2015). There are no harmonised classifications or notified hazards for PET. It is manufactured from ethylene glycol and terephthalic acid (PETRA PET Resin Association, n.d.).
Recycled and bio-based plastic (European Chemicals Agency, n.d.)	Bio and plant-based plastics are commercially available for flooring applications, often advertised as a sustainable alternative to PVC (Tarkett n.d.) (Armstrong Flooring, n.d.). The raw materials appear to vary, but often cork, bamboo or linoleum (see above) products are marketed as bio-based alternatives (Wolfe Flooring, n.d.; Plano Commercial Flooring, n.d.) as well as those based on polylactic acid (PLA) (Renewable Carbon News, 2010).	These polymers are based on renewable resources like corn. Plasticizers are used, but at least in some products these too are renewable, based on citrate and derivatized vegetable oil, for example (Renewable Carbon News, 2010).

7.7.3 Technical feasibility

There are several technically feasible alternatives to PVC flooring and these are available for a range of commercial, residential and healthcare settings[180]. All of the alternatives described above are commercially available and are used at present, several have been used for many years, including before the use of PVC. Other examples, such as biobased plastics, appear to be more recent product innovations, marketed specifically as alternatives to PVC.

A wide range of flooring products are available and even within each material various products are available with different characteristics (colour, thickness, additional surface treatments for durability, scratch protection etc). Each material has specific benefits and drawbacks making it more or less suitable for specific uses. The main advantages and drawbacks are summarised below, with reference to the technical requirements for PVC. Much of this information is based on commercial product literature, supplemented with consultation with a small number of industry stakeholders.

[180] See for example: https://www.sensoboden.de/anwendungen/krankenhaus

Table 7-15: Technical feasibility of potential alternatives to PVC in flooring

Alternative	Durability	Waterproof	Flame retardancy	Other (ease of installation, cleaning etc)
Wood/hardwood	Durable, but can dent, scratch, and show wear from heavy use. Sanding and refinishing can extend life and restore presentation (MyMove, 2021). Some literature indicates solid wood flooring can last up to 100 years (Old House, 2021), although this depends on thickness, use and type of wood (Fin Wood, n.d.).	Can be damaged from water, use of waterproof treatments are recommended (Wood n' Beyond, n.d.).	No, wood is flammable, although extent depends on density of wood (Hudson Flooring, 2017). Wood based flooring typically belong in class C, D or E (moderate flammability) (Michalovic, 2014). Some research indicates this is one of the better materials from a fire safety (Michalovic, 2014).	Requires careful installation, and moisture isolation, given natural expansion and contraction with moisture. Less suitable for basements and/or wet rooms without additional treatments. Requires maintenance and occasional renovation (Coswick. Inspire, 2012). Detergents and other aggressive cleaning fluids can damage the finish.
Laminate	Durable and resistant to staining, scratch and fading, manufacturer warranties vary, but often between 10-25 years (Coswick. Inspire, 2012).	Resistant to moisture damage; manufacturer literature indicates they are suitable for use in wet areas.	Moderately flammable and often treated with flame retardant. The adhesives can produce harmful gasses when ignited. Some literature does not recommend use in high-risk areas, such as hospitals (Tilen Space, n.d.). Some research indicates flammability is lower than PVC, however (Michalovic, 2014).	Localised dents and damage may require replacement of the entire floor. But otherwise, easy to clean and maintain (Coswick. Inspire, 2012).
Linoleum	A durable material with lifespan of c. 30 and 40 years (Ben Mattison, n.d.; Flooring Inc, n.d.) and available with different thickness and compressive load strength. Slightly spongy texture means it can dent/mark and it can become less flexible over time (The Flooring Group, n.d.). Specific flooring products are available for use in hospitals (Flooring Inc, n.d.).	It is suspectable to water damage and manufacturers recommend periodic resealing, especially in bathroom or kitchens (Flooring Inc, n.d.).	Low flammability (Class C) (Flooring Inc, n.d.). Moderately combustible (Michalovic, 2014), with some research indicating this is lower than PVC (Michalovic, 2014).	Anti-static properties, useful for computer rooms, easy to clean, resistant to disinfectants. Some literature suggests it is particularly suitable for healthcare, given ongoing oxidation process help kill bacteria (The Flooring Group, n.d.).
(Ceramic, porcelain, glass or cement) Tiles	Tiles are hard and durable, whilst they can crack it is possible - although often difficult in practice - to replace individual tiles. Porcelain tiles are denser than ceramic, so can make it more suitable for commercial uses with higher footfall (Designing Buildings, 2021).	Porcelain tiles are impervious to water. Ceramic tiles are more porous (Designing Buildings, 2021).	Non-flammable and heat resistant (E-Boss, n.d.) (Institut de Promoió Cerámica, n.d.).	Commonly used in residential and commercial applications. Market data indicates this is a growing market (Designing Buildings, 2021).
Stone (Urban Customs, n.d.)	Hard wearing and durable, can tolerate heavy use although some stones can chip, crack and scratch. May be affected by acids.	Stone has varying porosity and some manufacturers recommend use of a sealing agent.	Non-flammable.	Heavy, hard to install and impractical for some uses.

Rubber (the spruce, 2020)	Durable, but can be damaged by sharp objects and cut with a knife. Some literature suggests they can last for up to 20 years. Synthetic rubber is more durable than latex, on average. Specific flooring products are available for use in hospitals[181].	The rubber itself is water resistant but water can enter via edges. Whilst treatments are available for moisture protection, it is not likely to be suitable for bathrooms.	No, rubber is flammable; flame retardants are sometimes added (Everlast Epoxy, n.d.).	Can be difficult to clean to hygienic standards and strong detergents can discolour; easy to install and replace. Gives off an odour (more so natural rubber than synthetic). Latex can cause allergies (The Rubber Floor Store, 2013). An application for playrooms gyms, sports facilities.
Carpet	Whilst hard wearing materials are available, it will show wear and tear as the pile flattens and may not be appropriate for heavy use. Whilst they can be cleaned using detergents, some carpets will show staining, although nylon is comparatively stain resistant (the spruce, 2021).	No. Whilst some materials have reasonable resistance to moisture, it will retain water (the spruce, 2021).	Some polyamide carpets are described as "easily combustible (Class F) with some research suggesting this is more combustible than PVC (Michalovic, 2014).	A wide variety of styles and materials are available. Can be practically difficult to lay. Cannot replace small damaged/stained areas.
Cork (the spruce, 2021)	Comparatively less durable, can break, snap or wear. Some manufacturers recommend sealing to protect from water damage. Can scratch.	May require additional treatment for waterproofing (the spruce, 2021). High humidity can cause warping.	Cork is flammable and cork dust is extremely flammable (Phelps, 2021) without treatments.	A variety of thicknesses are available, comparatively easy to install/replace for DIY household applications. Depending on thickness, they can be sanded and refinished extending lifetimes.
Plastic alternatives	PU is noted as more commonly being used for floor underlay to extend the life of the main flooring material (i.e. carpet), and/or as a top coating/ finish to help resistance to abrasion, solvents and add comfort and noise insulation (American Chemistry Council, n.d.). However novel alternatives that are PU based have been developed and are commercially available (Wineo, n.d.). Flooring from 100% PET (40% recycled) is commercially available as a PVC and phthalate free alternative (Floor trends, n.d.).	In liquid form (I.e. as a coating) PU is extremely water resistant (Alpine Trek, 2018). PET is also moisture resistant (Creative Mechanisms, 2016).	The foams are combustible and flame retardants are often added to PU foams and coatings (P. M. Visakh, 2019). Most grades of PET are flammable.	Limited further details available.
Bio-based plastic	Whilst marketing literature suggests it is durable with "long life" and can withstand heavy duty foot and rolling load traffic, this is not quantified (Armstrong Flooring, n.d.; Plano Commercial Flooring, 2021) ,whilst others indicate it may need "increased maintenance" (Wolfe Flooring, n.d.) and sealant to protect from scratches and stains (Old House, n.d.). Some alternatives marketed as bio-based include cark, bamboo, limestone, or	No reliable and detailed information found, likely to be similar to linoleum.	No reliable and detailed information found, likely to be similar to linoleum.	Some manufacturers note that research on new materials continues they expect an "increased selection of sustainable flooring that meets, cost, performance and environmental criteria" (Armstrong Flooring, n.d.).

[181] See: https://www.nora.com/united-states/en/market-segments/healthcare/area/emergency-room

	linoleum, so are likely to have broadly similar durability.			

7.7.4 Economic feasibility and economic impacts

There are a number of alternatives to PVC flooring which appear to be economically viable . These have been commercially available and have been installed by a range of downstream users and consumers, in some cases for many years. The evidence suggests users take into account a variety of factors when making their purchase and deciding if and when to replace flooring. The total cost of any flooring product will include the unit price of the flooring itself (typically described in cost per M_2 or equivalent), as well as typical, approximate, average lifetime of the flooring before it needs replacing. Users will also incur costs for professional installation (where this is needed), incur costs depending on the ease and frequency of maintenance, the need for underlay and other treatments. The costs shown below are based on cost of material and typical lifetimes only, as two key determinants of overall cost, that can be identified accurately and compared consistently. Overall, whilst the unit costs vary significantly – reflecting the range of products available - there are several of higher, comparable or lower cost and lifetime of use (Table 7-16).

The alternatives mentioned in the literature and via consultation are largely alternative materials, rather than alternative plastics. Whilst some manufacturers are involved in the production of several different types of flooring (including PVC), the key economic effects may well result in decreased demand in one supply chain, offset by an increase in another. Such changes in demand may well affect prices for alternatives which are in turn influenced by speed and scale of demand increases and any related constraints in supply.

Table 7-16: Unit cost and typical lifetimes of PVC and alternatives in flooring

Application(s)	Cost[182] per M_2	Approx lifetime
PVC	Ranges start from €7-16 (HomeHow, n.d.) a common range is between €23 and €70 (Smart Spender, 2021; CheckaTrade, n.d.; Price this Please, 2020).	Information differs but between 10 to 20 years, depending on product quality (Flooring Stores, 2021; America's Floor Source, 2017; Flooring Inc, n.d.).
Wood/hardwood	Typical prices between €23 (Which?, 2021) to €59 (Checkatrade, n.d.) and up to €117 (Price Your Job, 2021) to €152 for solid hardwood (Tradesmen Costs, 2021).	Depends on material quality but can last upwards of 25 years potentially up to 75 years (Floor Monster, 2020; Speedy Floor Removal, n.d.).
Laminate	A wide variety of finishes are available, typically between €5 and €35 (Checkatrade, n.d.; Price Your Job, 2020).	Approximately 15 to 25 years (Speedy Floor Removal, n.d.; SF Gate, n.d.).
Linoleum	Often marketed as vinyl flooring, prices range between €21 to €30 (HomeHow, n.d.; Uk Contract Flooring, 2021) although prices can be up to €53 (Bricofloor, 2021).	Warranties of 25 years are common, but can last up to 40 years (the spruce, 2020; Flooring Inc, n.d.; Bob Vila, n.d.)
(Ceramic, porcelain, glass or cement) Tiles	Between €23 and €29 (Household Quotes, 2021; My Builder, 2021) with some prices for e.g., porcelain or cement tiles at around £30.	Up to 75 – 100 years (Buildipedia, 2011). Porcelain tiles typically have longer durabilty than ceramic (Sarana Tile, 2019).
Stone	Marble between €47 to €59, limestone €35 to €47 (My Job Quote, n.d.) although prices can be significantly higher.	Depends on material, but over 50 years and up to to 100 years or more for marble and granite (Natural Stone Council, 2009; Buildipedia, 2011).
Rubber	A wide range of product types are available, but typically between €35 to €47 (Sodt Surfaces, n.d.; Vinyl Flooring Online, n.d.) although some are upwards	Around 20 years and up to 30 years (The Rubber Company, n.d.; the spruce, 2020).

[182] Exchange rate of 1.17 GBP per Euro used.

Application(s)	Cost[182] per M_2	Approx lifetime
	of €70 (The Colour Flooring Company, n.d.)	
Carpet	From €5 to €35 (HomeHow, n.d.) with a typical cost for mid-range carpets of around €17 to €23 (Flooring Megastore, n.d.).	Estimates vary, between around 5 to 15 years (Martel Carpets, n.d.; Palmetto Carpet, n.d.).
Cork	Prices typically start form around €23 for tiles (Which?, 2021), but can be up to €59 (Evening Standard, 2019)	Between 10 and 20 years, and up to 30 years with regular maintenance (Do it yourself, 2021; Aspen Wood Floors, 2014; Floor Factors, 2020).
Plastic alternatives	No reliable information has been identified. Phthalate free flooring is commercially available from €23, although it is not clear if this is non-PVC (Vinyl Flooring Online, n.d a).	Limited quantiatvie data identified, which suggests 15 years warrantly for commercial uses, longer for residential (Moto, n.d.). Qualitative information suggests that comparable or better durabilty including "in high traffic areas". (The Source, n.d.)
Recycled and bio-based plastic	As above for linoleum and cork. Bamboo flooring starts from €23 (Which?, 2021).	As above for linoleum and cork. Bamboo can last between 30 and up to 50 years (GreenFloors, n.d a).

7.7.5 Reduction of overall risk due to transition to the alternative

An increase in the use of wood flooring products will increase raw material demand, as well as for recycled/reclaimed wood. As noted above, varnishes and coatings applied to wood can themselves contain harmful chemicals including polyurethane and other VOCs, although more benign coatings – such as those from linseed oil - are commercially available (SF Gate, 2018; UL LLC, 2018). The use of these may increase in the event of a phase out of PVC.

The United States Centre for Disease Control (CDC) tested Chinese manufactured laminate flooring for indoor formaldehyde (CAS 50-00-0) levels. Formaldehyde has several harmonised classifications under CLP, including carcinogenicity (1B: H350) and mutagenicity (2: H341). The CDC assessment concluded some elevated risk for irritation and breathing difficulties, which may be compounded by other sources in the home. Longer term, some elevated cancer risks were identified for those with very high and extended exposure (Centers for Disease Control and Prevention, n.d.). The raw materials of linoleum are natural and are often marketed as a sustainable alternative to PVC, containing no chlorine or plasticizers, although literature refers to VOC emission during manufacturing as well as the presence of drying agents and pigments (Building Green, 1998). Several of the other alternatives are natural (stone, ceramic, cork, latex). Synthetic rubber can contain SBR which contains styrene (CAS 100-42-5) – toxic to reproduction - as a copolymer (European Chemicals Agency, n.d a).

Additives of concern in flooring include phthalate plasticisers, which can leach out of PVC and contribute to the presence of phthalates in indoor dust. Studies measuring urinary metabolites of a number of different phthalates found that children with PVC flooring in their bedrooms had significantly higher levels of the butylbenzyl phthalate (BBzP) metabolite monobenzyl phthalate (MBzP) (F Carlstedt, 2012; Huan Shu, 2018). Other studies have also found correlation between PVC flooring and the presence of DiBP and DEHP in dust (Shu, et al., 2019).

7.7.6 Availability

All the alternatives are commercially available. For several materials, detailed information on EU production, import and export volumes are available via PRODCOM. This information illustrates the scale of existing supply. Overall, this information indicates that several, if not all, alternative materials are available in significant quantities.

For example, PRODCOM data for the EU (28) for 2019 show the following production values (European Commission, 2021):

- Marble, granite and slate slabs - around 13.6 million tonnes[183]
- Carpets - some 458 million m^2 [184]
- Wood panels and oak blocks for flooring - just under 20 million m^2 [185] (a small subset of a large number of potentially relevant wooden products from a much larger global forestry products market)[186]
- Cork - just over 50,000 tonnes of cork (although several hundred thousand tonnes are imported and exported)[187]
- Rubber flooring products - just under 730,000 tonnes[188]
- Linoleum - 32.8 million m^2 [189]
- Ceramic tiles and flags - over 1 billion m^2 (part of a larger subset of potentially relevant products)[190].

Precise changes in demand will depend on the timescales and scope of any potential PVC phase out as well as the timescales for development of new alternative formulations. Information that has been provided as part of the stakeholder consultation indicated that non-PVC plastic-based flooring formulations have been developed and placed on the market. This is discussed further in chapter 8.

7.8 Conclusion on suitability and availability for Alternatives to PVC in flooring

Alternatives to PVC flooring are technically feasible and appear to be economically viable . They are commercially available and in large volumes. An overall reduction in human health and environmental risk is judged likely, primarily given the presence of phthalate plasticizers in PVC flooring. The precise net change is uncertain, given the potential for adverse effects from some of the alternatives.

7.9 Analysis of alternatives - Packaging

7.9.1 Key performance criteria

PVC is used for a number of different applications in packaging. These include:

- **Food packs**, either transparent or opaque rigid disposable boxes (Meridian Specialty Packaging, n.d.; The Bag'n Box Man, n.d.; Food Packaging Technology, 2019a);
- **Shrink foils** (note these are referred to interchangeably as shrink foils, shrink wraps and shrink films). These are transparent plastic wraps which shrink to fit around a product when exposed to heat (Industrial Packaging, 2019; Industrial Packaging, n.d.);

[183] Source: annual data 08111136 - Marble and travertine merely cut into rectangular or square blocks or slabs; 08111236 - Granite merely cut into rectangular (including square) blocks or slabs; 08114000 - Slate, crude, roughly trimmed or merely cut into rectangular or square blocks or slabs. Converted from KG. This excludes imports or exports.

[184] Source: annual data 13931100 - Knotted carpets and other knotted textile floor coverings; 13931200 - Woven carpets and other woven textile coverings (excluding tufted or flocked); 13931300 - Tufted carpets and other tufted textile floor coverings

[185] Source: annual data 16221030 - Assembled parquet panels of wood for mosaic floors; and 16101277 - Oak blocks, strips or friezes for parquet or wood block flooring, planed but not assembled (excluding continuously shaped)

[186] See for example: https://www.forestresearch.gov.uk/documents/7825/FRFS020_TZKZgnZ.pdf
http://www.fao.org/3/i7034en/i7034en.pdf

[187] Source: annual data 16292150 - Natural cork, debacked or roughly squared, in rectangular or square blocks, plates, sheets or strips

[188] Source: annual data 22192070 - Plates, sheets and strip of vulcanised rubber; 22192083 - Extruded rods and profile shapes of cellular vulcanised rubber; 22192085 - Plates, sheets, strips for floor covering of solid vulcanised rubber; 22197200 - Floor coverings and mats of vulcanised rubber, non-cellular

[189] Source: annual data 22231500 - Linoleum, floor coverings consisting of a coating or covering applied on a textile backing (excluding sheets and plates of linoleum compounds)

[190] Source: annual data 23311000 - Ceramic tiles and flags

- **Blister packs**, rigid plastic sheets thermally formed into a number of blisters which hold individual products or pharmaceutical doses. (European Commission, 2004; Air Sea Containers, 2020; Bizongo, n.d.). Stakeholder engagement has shown that PVC packaging demand has declined in recent years, however PVC remains the most popular choice for blister pack applications (Air Sea Containers, 2020);
- **Closures and can linings** - where enclosed containers are coated in PVC polymer-based coating (International Life Sciences Institute, 2003; Food Packaging Forum, 2017); and
- **Bottles** (European Commission, 2011), stakeholder engagement indicates that whilst PVC is still used in bottles, the volume of use has singficantly decreased in recent years. It is not clear if PVC use is focused on specific drinks/market segments.

Key performance properties for those applications include:

- **Durability**
- **Water proofing**
- **Transparency**

PVC in blister packaging offers these properties, alongside a competitive price. In uses where particular water and oxygen barrier properties are required, coatings of PVDC (see Table 7-17 and Table 7-18) are applied (Air Sea Containers, 2020). Flexibility is needed for shrink foils, and oil proofing and organoleptic properties are important for food packaging. Airtightness and adhesive properties are important for closures and linings. Further desirable properties include ease of moulding/processability and print clarity, allowing for clear branding and text on external packaging. Usage of PVC in shrink foil is limited to CD/DVD/software disk package boxes and other non-edible items, where it has been noted for low resistance to high and low temperatures, low puncture resistance and leaving by-product deposits on sealing machinery (Industrial Packaging, n.d.; SwiftPak, 2021).

Several of these products come into contact with food[191]. The specific monomers permitted for use in contact with food are listed and regulated by EC 10/2011, which includes the vinyl chloride monomer (European Parliament, 2011). Commission Regulation (EU) No 10/2011 of 14 January 2011 on plastic materials and articles intended to come into contact with food (and its various amendments) sets out specific requirements for plastic materials and articles that come into contact with food; this contains several concentration limits for substances used in PVC (European Parliament, 2011). There are a large number of ISO standards which relate to packaging more generally.

7.9.2 Substance ID and properties (or Description of alternative technique)

Desktop research identifies several alternatives for PVC packaging in a number of different applications which are currently used in the listed applications (Table 7-17). In addition to the alternatives identified below, there is potential for substitution of plasticisers (including phthalates) with DINCH, which was approved for use in a variety of food contact applications by the European Food Safety Authority in 2006 (see also section 7.14 on medical applications).

[191] Under EU Framework Regulation EC 1935/2004 (European Parliament, 2004) food packaging materials are required to, "under normal or foreseeable conditions of use, […] not transfer their constituents to food in quantities which could: (a) endanger human health; or (b) bring about an unacceptable change in the composition of the food; or (c) bring about a deterioration in the organoleptic characteristics thereof".

Table 7-17: Potential alternatives to PVC in Packaging

Alternative substance (or technology)	Applications	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
PET (polyethylene terephthalate)	Food packs, (rigid)	CAS No. 25038-59-9. Manufactured from ethylene glycol (CAS No. 3775-85-7) and terephthalic acid (CAS no.: 100-21-0). p-xylene (CAS no.: 106-42-3) is a precursor to terephthalic, manufactured from naptha. Can be manufactured from biomass (Siracusa & Blanco, 2020).	p-xylene is harmful in contact with skin, harmful if inhaled and fatal if swallowed (European Chemicals Agency, n.d.). There are no harmonised classifications for remaining substances. An ILIS study states that PET is biologically inert if ingested and is not hazardous if inhaled, with no evidence of toxicity shown in animal studies (Omnexus, n.d.).
PE (HDPE/LDPE/ LLDPE)	Food packs (rigid), shrink foils (flexible)	CAS No. 9002-88-4. Manufactured from ethylene (ethene) monomer, which is extracted from petrochemical sources or via dehydration of ethanol (Dusan Jeremic, 2014). Can be manufactured from biomass (Siracusa & Blanco, 2020).	Described as a 'cleaner' plastic than PVC, but is not toxin free (Ahmadi, 2013; Ackerman, 2006). Eicosane (CAS No. 112-95-8), tetracosane (CAS No. 64742-51-4) and nonadecane are impurities present in the polymer which can potentially leach out, but papers show that these can be extracted from the plastic prior to use (Cinthya Soreli Castro Issasi, 2019). Also contain low proportions of stabilisers, antioxidants, UV stabiliser (John N. Hahladakis, 2018), fillers, plasticisers, colourants, flame retardants, blowing agents, crosslinking agents and UV degradable additives (Eltayef, 2015). These include Irgafos 168(R) anti-oxidant additive, which can decomposed to detectable levels of 2,4-di-tert-butyl-phenol (CAS No. 96-76-4 – has endocrine disrupting properties) in water samples.
PP (polypropylene) inclduing BOPP (biaxially oriented PP)	Food packs (rigid), shrink foils (flexible)	CAS No. 9003-07-0. Can be manufcatured from biomass (Siracusa & Blanco, 2020). Stakeholder engagement identified PP as an effective alternative for PVC films for water barrier applications.	Numerous studies on PP packaging show release of antioxidants into food, such as Irgafos 168, Irganox 1010 and Irganox 1330, the former of which can break down into harmful chemicals (see PE) (J. A. Garde, 2001; J. Alin, 2010; J. Alin, 2011; Blázquez-Blázquez, et al., 2020).
PS (polystyrene)	Food packs	CAS No. 9003-53-6 Manufactured from styrene monomer (CAS No. 100-42-5) which is manufactured from petrochemical precursors (James & Castor, 2011).	Styrene is suspected to be toxic to be reproduction, may be fatal if swallowed and causes serious eye irritation (European Chemicals Agency, n.d.), and thus presents an occupational health risk to those involved in PS production. It is not clear if this is present as a residual monomer.
PA (BOPA - biaxially oriented nylon)	Food packs (flexible)	Nylon-66 is the most common polymer type CAS No 32131-17-2) followed by Nylon-6 (CAS No. 24993-04-2). Nylon can be cast as unoriented or biaxally oriented (BOPA). The latter is used as a specialty film with superior physical and chemical properties. Small volumes of Nylon is	There are no harmonised classifications for these substances.

Alternative substance (or technology)	Applications	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
		currently manufactured from 100% biomass (Kind, et al., 2014; Creative Mechanisms, 2016)	
COC/PO (cyclic olefin copolymer)	Blister packs (rigid)	Manufactured from cyclic monomers and olefin (ethylene – CAS No. 74-85-1, propylene - CAS No. 74-85-1) polymeristation (SHIN, et al., 2005; Plastics Today, 2021).	COCs are a category of polymer, thus vary substantially. Both ethylene and propylene are highly flammable and ethylene may cause drowsiness or dizziness.
EVOH (ethylene vinyl alcohol)	A secondary film or used with polymer in Blister packs and Shrink foils	Manufactured from ethylene (CAS No. hazards and source identical to PE, see above) and vinyl acetate (Maes, et al., 2017) (CAS No. 108-05-4). Vinyl acetate is primarily manufactured from ethylene, acetic acid (various production methods (Han, et al., 2004).	The vinyl acetate monomer is a suspected carcinogen, is harmful if inhaled, may cause respiratory problems and is highly flammable (European Chemicals Agency, n.d.).
PVDC (polyvinylidene chloride)	Secondary film in Food packs, Blister packs and Shrink foils	Manufactured from the vinylidene chloride monomer (CAS No. 75-35-4), arising from similar sources to PVC (petrochemical and industrial salt) (Encyclopedia Brittanica, n.d.; Environmental Protection Agency, 1985).	The VDC monomer is a suspected carcinogen, is harmful if swallowed and extremely flammable (European Chemicals Agency, n.d.).
Bio-plastics	Available as shrink foil, blister packs and rigid food packs	**Biodegradable films:** industry sources state products are biodegradable 3 years from manufacturing and are manufactured using up to 20% biomass (typically sugar cane). Remaining precursors are polyolefins of petrochemicals sources (Kempner, n.d.). **Blister packs:** fewer biodegradable options available, with the first ever bioplastic blisterpack developed in 2019, manufactured from a PVC/PVDC blend. The plastic degrades to methane, carbon dioxide, water and "biomass", although sources are unclear as to what this biomass constitutes and it's safety (Business & Innovation Magazine, 2019). See PVC and PVDC for CAS No. and sources.	PBAT (polybutylene adipate terephthalate; bio-degradable) is used in packaging as a film (Jian, et al., 2020). PBS (polybutylene succinate; biodegradable) is used in packaging as a film and can be used in pharmaceutical blister packs (Xu & Guo, 2009). PLA (polylactic acid) can be used as rigid or film food packaging (Law Print & Packaging Management Ltd, 2017).
Aluminium	Food packs	Aluminium (CAS no.: 7429-90-5) is commonly alloyed	Widely used in tins and cans.

Alternative substance (or technology)	Applications	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
		with iron (CAS no.: 7439-89-6) and silicon (CAS no.: 7440-21-3) for packaging.	
Paper	Food packs, blister packs	**Waterproofed** papers feature plastic film linings (e.g PE, CAS No. 9002-88-4) (GM Packaging (UK) Ltd, n.d.); **Greaseproof** paper is treated with alginates (e.g Sodium Alginate (Shandongjiejing Group n.d.), CAS no.: 9005-38-3), caboxymethyl cellulose (CAS no.: 9004-32-4) or starches to gain oil barrier properties. **Wax paper** is manufactured from wood pulp and coated by waxes derived from vegetable oils or crude oils (Eurowaxpack n.d.).	There are various types of paraffin waxes with different toxicities. No reliable sources on which are used in specific ifc paper coatings have been identified, although risk are present in some (European Chemicals Agency, n.d.; European Chemicals Agency, n.d.) (European Chemicals Agency, n.d.).
Ceramics	Food packs	Ceramics are typically manufactured from clay extracted from quaries (Cermer, n.d.).	Little reliable information available on manufacture for packaging and variants.
Glass	Food packs	Manufactured from silica (CAS no.: 99439-28-8), soda ash (sodium carbonate - CAS no.: 497-19-8) and limestone (1317-65-3) extracted from quaries, and cullet (recycled glass).	Widely used in drinks bottles and other food packaging. During production, silica can cause eye, skin and respiratory system irritation, soda ash (sodium carbonate) causes eye irritation and limestone causes skin irritation (European Chemicals Agency n.d.).

7.9.3 Technical feasibility

There are several technically feasible alternatives to PVC packaging, in blister packs, rigid food packs and shrink foils. All of the alternatives described above are commercially available are used at present. Several have been used for many years, including before the use of PVC, with others relatively recent.

The three packaging types considered (blister packs, rigid food packs and shrink foils) each require different properties. The table below sets out how each material meets a number of these criteria, but it is noted all are not necessarily required for each application. The overall performance of each material for use in its intended application is summarised in Section 7.9.7. According to stakeholder engagement, PVC in packaging products has been banned in a number of individual countries and cities (Center for Health, Environment and Justice, n.d.), including Norway and Sweden.

Table 7-18: Technical feasibility of alternatives to PVC in packaging

Alternative substance (or technology)	Durability	Water tight, oil resistance and non-leaching	Temparature/ UV resilience	Transparency	Other (moulding, processability, printability, sterility etc.)
PET	High durability (Omnexus, n.d.; AZO Materials, 2003; Creative Mechanisms, 2016)	Good gas (oxygen, carbon dioxide) and moisture barrier properties. High chemical resistance except for strong alkalis. (Omnexus, n.d.; AZO Materials, 2003; Creative Mechanisms, 2016)	Heat resistance between roughly -60 and 130 oC. (Omnexus, n.d.; Creative Mechanisms, 2016)	Transparent (Omnexus, n.d.; AZO Materials, 2003; Creative Mechanisms, 2016)	Ease of molding. "Transparent" to microwave radiation (i.e. assumed to mean compatible) (Omnexus n.d., AZO Materials 2003, Creative Mechanisms 2016)
HDPE	Rigid, durable, high tensile strength (MatMatch, n.d.)	Watertight, good chemical resistance (MatMatch, n.d.)	Heat resistance up to 100 ºC. Little/no UV resistance (MatMatch, n.d.)	Lower transparency than LDPE, translucent when unpigmented (MatMatch n.d., Essentra Components 2019)	-
LDPE	Flexible, moderate durability against impact (MatMatch, n.d.)	Watertight, good chemical resistance. Permeable to CO2 (MatMatch, n.d.)	Lower resistance to high or low temperatures. Little/no UV resistance (MatMatch, n.d.)	High transparency (MatMatch, n.d.)	Easy to process with heat sealing (MatMatch n.d., Essentra Components 2019)
PP	Durable and semi-rigid (Plastics Insight n.d.), high flexural strength (Creative Mechanisms, 2016).	Good resistance to many chemicals. Good water barrier properties (Omnexus, n.d.; British Plastics Federation, n.d.; Creative Mechanisms, 2016)	Poor UV resistance. Becomes brittle below -20oC and has an upper service temperature of between 90 and 120oC (Omnexus, n.d.; Creative Mechanisms, 2016).	Can be manufactured as transparent (Creative Mechanisms, 2016)	Susceptible to mold and bacteria. Highly flammable. Poor paint adhesion. Heat aging is worsened by metal contact (Omnexus, n.d.)
PA	Highly durable, "specialty" flexible film, resistant to scratches and punctures (PolymerDatabase.com, n.d.)	Used for food applications requiring strong oxygen barrier, but CO2 permeation (cheeses, meat, fish). Oil and acidic food resistant (PolymerDatabase.com, n.d.)	Microwavable (PolymerDatabase.com, n.d.)	Transparent (PolymerDatabase.com, n.d.)	"Relatively easy" to process. "Burns without releasing harmful chemicals to the atmosphere". More expensive than other plastic packaging (PolymerDatabase.com, n.d.)"
PS	Rigid low impact resistance (Custompac Ltd, n.d.; AZO Materials, 2001)	Low water absorption, but usceptible to UV degradation (Custompac Ltd, n.d.; AZO Materials, 2001). Low resistance to oils (Rochling Formaterm, n.d.)	High thermal insulation (Custompac Ltd, n.d.; AZO Materials, 2001)) but susceptible to cold temperatures and has a low softening temperature (Rochling Formaterm, n.d.)	Opaque	Shock absorbing properties, lightweight (Custompac Ltd, n.d.; AZO Materials, 2001)

Alternative substance (or technology)	Durability	Water tight, oil resistance and non-leaching	Temparature/ UV resilience	Transparency	Other (moulding, processability, printability, sterility etc.)
COC/(PE or PP)	Durable, widely used as pharmaceutical packaging (Plastics Today, 2021)	Strong moisture barrier. Low leachables, appropriate for drug contact (Plastics Today, 2021)	Grades with high heat resistance avialable (Plastics Today, 2021)	High clarity (Plastics Today, 2021)	Sterilizable by multiple methods. High detail molding (Plastics Today, 2021)
EVOH	N/A (usually applied as a film over other plastics)	Good oxygen and moisture barrier properties (Lagaron, et al., 2013). Usually applied as a second film to existing cheaper polymers to increase barrier properties. Loses gas barrier properties when exposed to water unless co-extruded (Impact Plastics, 2018).	No information identified.	Thicker layerings of EVOH film result in yellow tint/haze and gel and bubble formation. Thicker layers are avoided for recycling purposes though. (PlastEurope, 2019).	
PVDC	N/A (applied as a film over other plastics)	Excellent oxygen and moisture barrier properties allows for longer shelf life. Often applied as a second film to existing polymers to increase barrier properties, especially onto PVC in blister packs (Future Market Insights, 2021; Formulated Polymer Products Ltd, 2019)	Degrades at relatively low temperatures (Packaging Birmingham n.d.), losing chlorine atoms. Leaves polyene behind - brown film. Also turns yellow/brittle with age(Engineering 360, 2016)	Transparent	Difficult to process (Packaging Birmingham n.d.)
Bioplastics	If biodegradable, will generally be less durable over time (Reef Repair, 2019)	Bioplastic films have not been tested in many applications (Industrial Packaging, 2020) and properties vary substantially.	As left.	As left.	As left.
Aluminium	Durable. Properties vary with alloy. Can be laminated for higher durability requirements. (Aluminium Federation, n.d.; Plus Pack, n.d.)	Strong barrier properties. Coatings availalable for highly acidic/alkali foods (Aluminium Federation, n.d.; Lamberti & Escher, 2007; Plus Pack, n.d.).	High heat resistance for high and low temperatures. UV resilience better than plastic alternatives (Aluminium Federation, n.d.; Lamberti & Escher, 2007; Plus Pack, n.d.)	Opaque	Lightweight. Compatible with printing processes. Laminates and coatings reduce ease of recyclability. High thermal conductivity (Aluminium Federation, n.d.; Plus Pack, n.d.), presumably bad for insulating.
Paper	Less durable than alternatives (SwiftPak, 2021).	Susceptible to damage from damp (SwiftPak, 2021). Different coatings allow for greese proof, leak proof and watertight applications at the cost of either	Paper frays/burns at higher temperatures but grease proof paper offers improved resilience (beeco, 2021).	Opaque	Can be harder to stack than alternatives. Very good printability and design (SwiftPak, 2021).

Alternative substance (or technology)	**Durability**	**Water tight, oil resistance and non-leaching**	**Temparature/ UV resilience**	**Transparency**	**Other (moulding, processability, printability, sterility etc.)**
		biodegradaibility or recyclability (SwiftPak, 2021; Waste Away Group, 2020; GM Packaging (UK) Ltd, n.d.).			
Ceramic	High durability, but brittle (Cermer, n.d.).	Water tight and oil resistant, low leachability (Cermer, n.d.)	Withstands high and low temperatures and compatible with oven/fridge (Cermer, n.d.)	Opaque	Higher weight likely reflected higher transport cost
Glass	High durability, but brittle (Epicurious, 2018).	Inert, air tight, water tight, oil resistant, low leachability (Epicurious, 2018; Medium, 2019)	Resistance to range of temperatures (Medium, 2019)	Transparent	High weight, higher transport cost (Epicurious, 2018). Allow for washability and repeated use (Medium, 2019) .

7.9.4 Economic feasibility and economic impacts

There are a number of alternatives to PVC packaging with are economically viable . These have been commercially available and have been in use, in some cases for many years. The lifetime of packaging is typically relatively short, often single use. It is imprtant to note the materials have different strength to weight ratios; in some cases, less material can be used to manufacture a product of similar strength. However, as an indication, costs for the material per tonne are below (Table 7-19).

- With regard to wall thickness and transport costs due to weight, ceramics and glass perform poorly, with glass weighing over 20 times more than a typical plastic product for an identical liquid container application (Tapp Water, 2021; Plastic Packaging Facts, n.d.).
- Plastics are generally cheaper per tonne than non-plastic alternatives such as aluminium and paper.
- PVDC applied as thin layer films or coextruded with another polymer, making their total mass in the end prodcut lower. The additional cost associated with co-extruding and layering additional layers of film has however not been considered.
- COC/PO appears to be more expensive than alternatives, but appears to be limited to specialty usage, such as pharmaceutical plaster packs, and benefits from ease of molding, potentially reducing the price of processing.
- Costs of bioplastics vary substantially and tend to have a greater upfront cost than non-bio alternatives, although there are exceptions to this, with the price expected to fall in future (Oever, et al., 2017) .
- It has not been possible to assess the transport costs per product resulting from their respective weights and stacking compactness, as there is very little data available in this area.
- PVC appears to be more expensive than PET from the data provided. This is inconsistent with information surrounding blister packaging, whereby PVC is generally used in its place due to its low cost (Air Sea Containers, 2020). It is possible that these costs arise from processability.

It should be noted that costs for plastics refer to virgin plastic price per tonne, while non-plastic costs refer to costs per tonne of product.

Table 7-19: Material Cost per Tonne

Alternative substance (or technology)		Material cost (EUR/t)*
PVC		700-1,300 (See chapter 3)
PET		600-1,100 (Staista, 2021)
HDPE		1,450 – 1,900 (Plastic Portal, 2021a)
LDPE		1,800 – 1,900 (Plastic Portal, 2021b)
LLDPE		700 – 1,800 (Alibaba, 2021)
PP		1,650 – 2,000 (Plastic Portal, 2021a)
PS		No reliable information identified
PA (BOPA - biaxially oriented nylon)		700 -1,750 (Plasticker, 2021a)
COC/PO (cyclic olefin copolymer)		13,000 – 37,650 (Alibaba, 2021)
EVOH (ethylene vinyl alcohol)		No reliable information identified
PVDC (polyvinylidene chloride)		1,800 – 3,500 (Alibaba, 2021)
Bio-plastics		No reliable information identified for bioplastic as a raw material for packaging, however, these are noted as more expensive than non-bio alternatives (Industrial Packaging, 2020)
Aluminium		2,000 (Index Mundi, 2021a)
Paper	**Greaseproof**	1,300-3,000 (Alibaba, 2021; Paper Index, 2021)
	Wax coated	2,500 (Alibaba, 2021)
	PE coated	800-1,300 (Made-in-China, 2021)
Ceramics		Values highly dependent on the type of ceramics required.

Glass	Highly dependent on source (e.g virgin, recycled from windows/jars/bottles).
* USD to EUR exchange rate of 0.85 used.	

7.9.5 Reduction of overall risk due to transition to the alternative

PVC in shrink foils can release harmful by-products, such as hydrogen chloride (CAS no.: 7647-01-0) gas (SwiftPak, 2021; Industrial Packaging, n.d.). Hydrogen chloride is known to cause severe burns, serious eye damage, is toxic if inhaled, may damage fertility, may cause damage to organs through prolonged/repeated exposure, may be corrosive to metals and cause respiratory irritation (European Chemicals Agency n.d.). Additives used in PVC packaging can leach into packaged goods (Guo, et al., 2010), including phthalates in films (Dong, et al., 2013) and organo-tin compounds (Mandrile, et al., 2020). Further additives include calcium/zinc stearates, methylmethacrylate butadiene, acrylate, PE wax, glycerol mono oleate, stearic acid, white mineral oil, adipate and epoxidised soya bean oil (International Life Sciences Institute, 2003). No evidence has been identified as to whether these leach into food.

- In terms of the alternatives, a study on the extent of migration states that PP exhibits highly inert behaviour in the presence of oils, while PA and PE exhibit slight degrees of migration and PVC exhibited a relatively high degree of migration compared to these plastics (Galotto, 2004). Similarly, another study identified PVC and polyurethane as inducing higher toxicity through chemical release than HDPE and PET, while the toxicities of LDPE, PS and PP varied (Zimmermann, et al. 2019). It should be noted that stakeholder engagement has indicated the opposite; that PE presents a higher rate of diffusion of chemicals into samples than PVC.
- PET in particular has a low diffusion coefficient, meaning less migration potential of substances into packaged goods, making it appropriate as a recycled material (Omnexus, n.d.).
- Flexible and film plastics however are only recycled in select countries in Europe (Flexible Packaging Europe, n.d.), with recognition that improvements are needed in this field (The National Law Review, 2020a).
- EVOH offers the benefits of PVDC, coating a lower quality plastic with the better oxygen and chemical barrier layer in order to reduce total plastic usage, but is composed of non-toxic chemicals.
- Microplastic particlesare noted to posie a risk to marine environments due to their ability to absorb toxic contaminants, which are then ingested by marine wildlife when the plastic is inadvertently consumed.
- Other chemicals that have been observed to migrate from packaging include:
    - PA, PE, PS, PP and PET can leach directly into products in appropriate conditions;
    - Irganox 1086, Oleamide, Erucamide, Stearamide and Diphenyl butadiene from LDPE;
    - Toxic metals from PET; and
    - 2- and 3-t-butyl-4-hydroxyanisole, Dibutyl phthalate, Butylated hydroxytoluene, Irganox1010, 1076, Irgafos168 and Ethanox 330 from LDPE, PS and PP (Hahladakis, et al., 2017)

With regard to disposal options, PVDC shares many of the same issues as PVC due to its similar chemical make-up. It is not widely recycled, leaving incineration and landfill as the current likely disposal route with potential for chemical release into the atmosphere and soil (Packaging Birmingham, n.d.). EVOH is not widely recycled, with thicker coatings of over 5% film by weight not recyclable at all (Packaging Birmingham, n.d.; PlastEurope, 2019). Bioplastic films are not industrially recyclable and instead must be composted by the user or at an industrial compositing site (Industrial Packaging, 2020). Biodegradable bioplastics also benefit from being lightweight, as well as minimising damage to ecosystems at end-of-life if composted. At present, materials and articles made entirely or partly from recycled plastics that are used in contact with food must be sourced from processes that EFSA has assessed for safety, and that the EC has authorised (European Food Safety Authority, n.d.). PET, PE and PP in rigid forms are recyclable (The UK Plastics Pact, Wrap, 2020).

Waterproofed PE-lined paper can be recycled, but it is not widely done due to the specialist plant processes required (Glasdon, n.d.; Charlotte Packaging Limited, 2015). Greaseproof paper cannot be recycled, due to its coatings and oil deposits from use (beeco, 2021). Wax papers are biodegradable, with vegetable oil coatings offering better biodegradability than paraffin coatings, but are not recyclable (Eurowaxpack, n.d.; Waste Away Group, 2020). The main precursor for paper manufacture is timber. Manufacturing is also a water and energy intensive process. (FoodPrint, 2019). Paper can be recycled, but costs more to transport given its higher weight (SwiftPak, 2021).
Ceramics are generally not recycled (Recycle Nation, 2014) and, where possible, appear to be downcycled to less demanding uses such as paving or construction (Cermer, n.d.), although there is limited detail on the extent of this activity.
Aluminium can be recycled after use repeatedly with "*no loss in quality*" (Aluminium Federation, n.d.), with it holding on average 35% recycled content, but can result in elevated lead to increased lead content (ThermoFisher Scientific, 2017). Recycling aluminium presents a 95% saving in energy against extracting virgin materials (Plus Pack, n.d.), due to its energy intensive manufacturing process, which can also result in the emissions of heavy metals (FoodPrint, 2019). Emissions can result in increasing surrounding soil corrosivity if not managed correctly (FoodPrint, 2019). Aluminium containers benefit however from being relatively lightweight (Tapp Water, 2021) and result in saving transport emissions .

Where glass is repeatedly recycled, there is potential for heavy metals accumulation, although it is believed that these do not pose a threat to health, as the structure of glass allows insignificant quantities of heavy metals to pass through to container contents. Glass is particularly energy and emission intensive due to the high temperatures required during the melting process. (FoodPrint, 2019) .

7.9.6 Availability

All of the alternatives identified are commercially available. Table 7-20 sets out available production and import quantity data (European Commission, 2021) to give an illustration of scale. Note that current estimated market share of PVC in packaging to that of other plastics represents about 1% of the global plastic packaging market (Plastics News, 2019)). Similarly, plastics comprise only 37% of the global food packaging market share (Food Packaging Forum, 2012). This suggests sufficient quantities of alternative plastics could be commercially available if increases in demand arose.

There is little available data available with respect to EVOH production tonnage in Europe. Globally, the main manufacturers appear to have total capacity of 132,000 tonnes per year in 2012 (Market Research Reports, n.d.). The largest EVOH plant in Europe had an annual capacity of 35,000 tonnes as of 2017 (EVAL, n.d.). COC/PO production is estimated to be roughly the same as EVOH, the two having comparable market sizes (Market Watch, 2017; Transparency Market Research, 2021). The global production capacities of commercially available bioplastics (e.g. PLA, PBS, PBAT) are currently relatively low compared to the existing manufactured tonnage of PVC for packaging. Bio-degradable bio-based plastic film manufacturers are also currently limited in number (Industrial Packaging, 2020). Biomass based plastics are manufactured on a global scale in quantities of 221,550 tonnes for PE; 164,580 tonnes for PET; 251,109 tonnes for PA and 29,540 tonnes PP (European Bioplastics, 2020).

Table 7-20: Material production and import quantities

Substance		2019 production quantity (tonnes)	2019 import quantity (tonnes)
PVC	***Primary unmixed PVC***	***4,960,064***	***504,285***
	Non-plasticised mixed PVC	384,389	28,518
	Plasticised mixed PVC	796,204	47,560
PET	**Primary PET of viscosity number >78 ml/g (20164062)**	2,517,575	1,091,015
	Other primary PET (20164064)	398,927	306,356
PE (including CPE)	**Primary linear PE (specific gravity <0.94)**	1,870,616	804,410
	Primary non-linear PE (specific gravity <0.94)	3,960,226	845,846
	Primary PE (specific gravity >=0.94)	5,987,404	1,947,229
PS	**Expansible primary PS (20162035)**	1,646,119	100,587
	Non-expansible primary PS (20162039)	1,737,955	199,180
Primary PP		10,435,915	1,370,022
PA -6,11,12,6.6,6.9,6.10,5.12 (20165450)		1,859,082	242,952
COC/PO		No data available	
EVOH		No data available.	
PVDC		No data available	
Bioplastics (Global production in 2020)[192]	**PBAT**	284,850 (globally in 2020)	No data available
	PBS	86,510	
	PLA	394,570	
Glass Bottles <2.5 litre capacity (23131140, 23131150, 23131160)		72,012,234	4,053,216
Ceramic articles for use in agriculture and for conveyance or packaging of foodstuffs (23491100)		8,000	2,340
Paper	**Vegetable parchment, greaseproof, tracing, glassine or other glazed translucent papers (17126000)**	844,924	53,194
	Bleached paper and board impregnated/coated with plastics (17127755)	713,140	75,772
	Sacks/bags of paper/board or cellulose (17211250)	1,029,239	151,369
	Cartons, boxes and cases of corrugated paper/board (17211300)	22,156,823	240,468
	Folding cartons, boxes and cases of non-corrugated paper/board (17211300)	7,381,021	434,470
Aluminium plates, sheets and strips (24422430, 24422450, 24422500)		6,629,102	1,561,526

7.9.7 Conclusions on suitability and availability of alternatives

There are a number of technically and economically feasible alternatives to PVC for shrink foils, rigid food packs and flexible food packs in packaging. These are commercially available. In terms of reduction in human health and environmental risk there are alternatives which pose lower risk, but increases in their use would involve a number of trade-offs. These are summarised by theme below (Table 7-21), identifying alternatives that have advantages and disadvantages in each criterion considered in the analysis.

Table 7-21: Summary of material performance in packaging applications

Technical Feasibility	Economic Feasibility	Comparative Hazard Risk	Availability	Uncertainties and quality of information
Various plastic-based alternatives	Plastic alternatives, tend to be lower	Aluminium and paper have lower hazard risk	Well established alternatives	Significant uncertainties in

[192] Data represents global production in 2020, as opposed to European production in 2019. Source: (Bioplastics Europe, 2020)

of similar or better technical performance to PVC are available for all applications, such as PET, PE, PA, COC/PO, EVOH, PVDC.	cost alternatives to manufacture and transport, due to their light weight.	due to high recyclability and low leachables.	such as PET, PE, PP, PS, PA, Aluminium, paper, and glass are readily available.	relation to bioplastic performance due to breadth of field and relatively recent arrival to market.
Non-plastic (glass, paper, aluminium, ceramics) alternatives are available, but lack certain properties such as durability, transparency, flexibility or stackability.	Non-plastic alternatives either tend to be more expensive to manufacture or more expensive to transport.COC/PO is generally higher prcie, as are current rpcies for some bioplastics.	Highly recyclable rigid plastics such as PET, HDPE and PP perform well, although additives leaching into products, in landfill and through incineration emissions pose risks. Glass and ceramics are highly inert with little known leachates. Glass is energy intensive to recycle, while ceramics are not widely recycled.	~	Lack of data in relation to transport and product cost make economic feasibility hard to judge. Limited granular data exisits on impact of specific additives in plastic throughout plastic life-cycle.
Bio-plastics are available with varying properties.	-	Flexible and multi-layer film plastics such as flexible LDPE, PP, PVDC, EVOH, COC/PO and PA are problematic to recycle and contain varying degrees of leachable additives, although may reduce transportation emissions due to weight.	Bioplastics are relatively novel and not currently available in the quantities required to effectively replace PVC use in packaging immediatly.	Ecological impacts of non-plastics from manufacture and recycling are difficult to judge due to lack of life-cycle assessment availability.

7.10 Analysis of alternatives – PVC in inflatable toys

7.10.1 Key performance criteria

Various inflatable toys and boats are made of soft PVC, which contain up to 50% by weight of plasticizers, usually phthalate esters. The articles are used outdoors and therefore must have good resistance to environmental conditions (temperature and weather) as well as ultraviolet (UV) stability. Materials used for this application should be:

- **Resistant to water**.
- **Flexible** yet strong enough to withstand rough usage and prevent tears/breakage.
- **Resistant to repeated flexing** (fatigue resistance) and ability to retain shape are also important.
- Have a **high strength to weight ratio**, so that small amounts of material can result in product with good strength.

PVC is also used in larger inflatable boats with combustion engines, where the material must be stronger and more durable still, whilst also resistant to solvents, oils and chemicals (Tickner, 1999). Some boat retailers describe the strength and ability to resist wear and tear of PVC boats and rafts as poorer than other available materials (such as polyurethane and certain polyolefins), however (Tornado, n.d.). PVC has been used for inflatable toys and boats because of its comparatively low cost, flexibility and colourability. These articles tend to have relatively short lifecycles.

A key requirement for alternative materials is that they retain the appearance achieved by PVC inflatable toys. Substitutes should match or enhance the colour compatibility, transparency and textures of PVC products. Inflatable toys and boats must also meet government regulations or industry specifications concerning flammability and electrical insulation properties (Tickner, 1999). Alternative materials should preferably be easily processed, using equipment used to manufacture PVC inflatable toys.

7.10.2 Substance ID and properties (or Description of alternative technique)

The main alternatives to PVC inflatable toys and boats identified via a desktop review of literature are below.

Table 7-22: Potential alternatives to PVC in inflatable toys and leisure boats

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
Thermoplastic elastomers (TPE)	CAS Number: 308079-71-2. Most TPEs are copolymers or alloys of conventional plastics (such as polystyrene and polyethylene) which allow tailoring of product qualities through variation in the quantity of each polymer involved (Drobny, 2014). After R&D investments, the inflatable boat industry has recently started using TPE polyurethane (PUR) as it is a very durable material (Inflatables Guide, n.d.).	Resistant to UV radiation, oxidation and atmospheric ozone. Generally good resistance to chemical damage, but susceptible to degradation when exposed to hydrocarbons. Good abrasion resistance (Anixter, n.d a). Most of the TPEs can be produced in a wide hardness range without the use of additives (Tickner, 1999). Styrene and polyurethane production are hazardous and both off-gas toxic chemicals in fires (Tickner, 1999). Polyurethane is the result of the chemical reaction between a polyol and a diisocyanate. Once the chemical reaction has taken place, the resulting material is inert and harmless to humans. It does not give off any toxic fumes when burned or heated. However, PUR presents an occupational hazard to workers involved in its production. Health effects include irritation of skin and mucous membranes, chest tightness, and difficult breathing. Isocyanates include compounds classified as potential human carcinogens and known to cause cancer in animals (Synthesia Technology, n.d.).
Ethylene vinyl acetate (EVA)	CAS Number: 24937-78-8 The copolymer of ethylene and vinyl acetate. Vinyl acetate usually varies from 10 to 40% w/w, with the remainder being ethylene (Sabu Thomas, 2017).	This is a halogen-free elastomer. It is highly resistant to UV radiation and stress cracking. Chloride catalysts are used in some vinyl acetate production. There is also exposure potential to by-products from ethylene production. EVA does not require phthalate additives to achieve flexibility, however, other additives may be used to increase its functionality (Tickner, 1999).
Polyolefins	Polypropylene (PP) (CAS Number 9003-07-0), Chlorosulfonated polyethylene (CSPE) (CAS Number 9008-08-6). Polyolefins are extremely versatile. A common material for inflatable boats is Chlorosulfunated polyethylene (CSPE) (Anchor Travel, n.d.).	PE is a semi-crystalline polymer with poor fire resistance. Fire resistance, as well as resistance to UV radiation, weathering and chemical degradation, is often augmented by the addition of fillers (including both halogenated and halogen-free additives such as plasticisers, fire retardants, stabilisers and pigments) (Eland Cables, n.d.). Where greater flexibility is required, ethylene propylene rubber (EPR) can be added to the polymer (Tycab Australia, 2019).PP is resistant to stress cracking, it

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
		is tough but semi-rigid thus impairing flexibility. Resistant to a wide variety of chemicals, including acids, bases, water and detergents. Highly flammable (Plastics Insight, n.d.). CSPE's known emissions resulting from burning / incineration include hydrogen chloride (HCl), carbon monoxide (CO), sulphur dioxide (SO2), organic acids, aldehydes, and alcohols (Greenspec, n.d a). CSPE can also contain additives made from lead or lead compounds.

7.10.3 Technical feasibility

The performance of alternative materials against the required performance criteria is assessed in Table 7-23. All of the alternatives to PVC presented here are currently commercially available for use in this application. For example, many TPEs on the market were widely used for applications where PVC or rubber was used in the past (Tickner, 1999).The available evidence suggests that the assessed alternatives display similar – or improved - technical characteristics. The materials are also easily processed and for some of the alternatives, the same equipment used to process PVC can be used. In terms of specific alternatives:

- CSPE (for inflatable boats) needs to be cold glued by hand and therefore the process is more labour intensive and expensive. This material also has a limited colourability and a porous surface that has a tendency to dull and get dirty (PumpupBoats.com, n.d.). However, given CSPE is used to make boats for leisure, this tends to require less colour compatibility. CSPE boats are popular because of their enhanced durability and UV stability compared to PVC boats, as well as their good resistance to abrasion and chemicals such as fuel (Anchor Travel, n.d.).
- Inflatable toys such as swimming and paddling pools, beach balls, boats and floats, are frequently made by cutting and sealing plasticized PVC sheet. Materials such as EVA and metallocene catalyst-based Polyethylene require simple joining techniques, such as via heat and radio frequency welding, and are noted as "an excellent candidate" to replace PVC in inflatable toys (Tickner, 1999). EVA has a poor chemical resistance and therefore it is not suitable for the production of leisure boats that feature an engine (Plastics Insight, 2021).
- TPEs are another technically feasible alternative to PVC as this material can be processed into a wide range of flexibilities and strength properties. Like PVC, olefinic TPEs can be calendared[193] for the elaboration of flexible sheets, and then post form heat-welded to produce inflatable toys. This material is able to withstand extreme temperatures without degradation (Tickner, 1999).
- Polyolefins have been used for decades in the toy industry because of their ease of processing, durability and colourability. Indeed, polyolefin elastomers have better flexibility, clarity, and tensile strength compared to PVC (Tickner, 1999).

[193] Calendering is the process of smoothing and compressing a material during production by passing a single continuous sheet through a number of pairs of heated rolls (Brittanica, n.d.).

Table 7-23: Main characteristics and ratings of alternative inflatable toys and boats

Alternative	Resistance to temperature and weather extremes	Waterproof and chemical/ UV/resistant	Flexibility/resilience/ strength	Ease of processing	Appearance
TPE	High temperature range without degradation and good resistance to a wide variety of environments. Polyurethane boats are highly weather resistant after recent techncial improvements (Tickner, 1999).	Waterproof and good chemical, abrasion and UV resistance.	TPE materials can match or improve the characteristics of PVC in terms of strength and performance (Tickner, 1999). Experts classify Polyurethane as the strongest inflatable boat material as it is extremely puncture and abrasion-resistant (Inflatables Guide, n.d.).	Easily processed with conventional equipment. The two main fabrication methods are extrusion and injection moulding (Tickner, 1999).	Easily coloured and soft to touch (Tickner, 1999).
EVA	Resistance to harsh weather conditions compared to PVC (Plastics Insight, 2021).	Waterproof and good resistance to UV radiation. It has poorer chemical resistance and barrier properties (Plastics Insight, 2021).	Good flexibility, toughness and resilience with increased vinyl acetate (VA) content, however, leachable plasticizers or other additives are not needed (Tickner, 1999).	Uses standard processing equipment and techniques such as injection moulding and most extrusion processes (Tickner, 1999).	Good clarity with increased vinyl acetate (VA) content. Leachable plasticizers or other additives are not needed to improve appearance (Tickner, 1999).
Polyolefins	Some polyolefins are more resistance to extreme weather conditions and UV rays, and are therefore more durable than PVC (Anchor Travel, n.d.). CSPE boats are known to be more durable than PVC ones.	Waterproof and very good resistance to chemicals and abrasion. Easy to patch up in case of puncture (Inflatables Guide, n.d.).	Some materials from the polyolefin family surpass PVC in strength and flexibility. However, their performance is decreased if they are kept folded for long periods of time (Anchor Travel, n.d.).	Easy to process by injection, blow and extrusion moulding. However, building an inflatable boat using CSPE is more labour intensive compared to PVC as it s cold glued by hand (Inflatables Guide, n.d.).	Depends on the type of polyolefin. Some have a wide range of colourability and/or clarity (Tickner, 1999). Some are more limited, such as CSPE (but these are mostly used for boats). CSPE also has a porous surface and it has a tendency of easily getting dirty and losing its colour if left under the sun (Inflatables Guide, n.d.).

7.10.4 Economic feasibility and economic impacts

All of the PVC alternatives for inflatable toys and boats presented here are currently commercially available for use in these applications. Information on cost and durability was not identified for all materials (Table 7-24). The materials also have a different strength to weight ratios, meaning that less material can be used for products of similar strength, decreasing the material and therefore the price of the final product. However, specific information of the strength to weight ratio for these materials was not identified and therefore a quantitative comparison was not possible. Production costs are a further consideration. For example, even though EVA costs per tonne are higher than PVC, sealing temperatures are reduced, leading to a reduction in the fabrication time for flexible sheets and films and production efficiency. Similarly, EVA is 30% lighter than PVC, which gives the film a better yield and reduces transportation costs (Tickner, 1999).

Some polyolefins have been used for decades in the toy industry and are a preferred plastic for many applications because of their ease of processing and cost-effectiveness. Polyolefins have a low energy demand during the melting process, minimising production costs. However, this is not the case for CSPE inflatable boats, as this material requires a more labour-intensive production process compared to PVC. Offsetting this, CSPE are also considerably more durable (Anchor Travel, n.d.). Polyurethane boats have also recently become a popular choice by boat experts because of their performance and durability. However, inflatable boats made with this material are typically among the most expensive.

In conclusion, to effectively determine the economic feasibility of alternative materials for the production of inflatable toys and boats, it would be important to obtain data on the cost of the finished products using each of the alternative materials, considering their production costs as well as their durability. For inflatable toys, this data was not identified, as most manufacturers still use PVC and, in many cases, they do not specify the type of plastic used. For inflatable boats, those using PUR are the most expensive in the market, followed by CSPE. Currently, PVC boats are the cheapest available option, but noted as having lower durability (Anchor Travel, n.d.).

Table 7-24: Material costs and durability of PVC and alternative materials for inflatable toys and boats

Substance	Material cost (EUR/t) [194]	Durability
PVC	***€425 – €1,700***	Boat retailers have noted the durability as significantly lower than other available materials (such as polyurethane and certain polyolefins) (Tornado, n.d.).
TPE (in general)	€1,020 - €3,825	Unknown
PUR	No reliable information identified but PUR boats are the most expensive in the market.	PUR boats are the most durable in the market.
EVA	€850 - €1390	No reliable information identified.
Polyolefin (PP)	€1,650 - €2,000 (Plastic Portal, 2021a)	No reliable information identified.
Polyolefine (CSPE)	No specific information identified, but more expensive than PVC. An CSPE inflatable boat could be up to twice the price compared to a PVC boat.	More durable than PVC

7.10.5 Reduction of overall risk due to transition to the alternative

The above plastics use additives during the manufacturing process to enhance material properties, add characteristics (stability (heat and light), flexibility, flame resistance, colour and

[194] USD to EUR exchange rate of 0.85 used.

aesthetics, for example). However, PVC on average has used the greatest amount of additives of any commercial plastic, reaching up to 50% of the total weight of the material in toys (Deanin, 1975; Tickner, 1999). In comparison, additives used for the manufacturing of the alternative materials presented here typically comprise 0-2% of the polymer mixture (Deanin, 1975). Furthermore, as stated in a report on the toxicity of materials used in toys, alternative plastics are less likely to leach harmful substances, as the additives are bound more strongly to the polymer compared to PVC (Tickner, 1999). A further issue in the reduction of risk for this application will be the degree to which manufacturers have been using phthalates that are subject to a 2018 REACH restriction (in force since 2020). This restriction includes particular controls on childcare articles and toys, but permits concentrations of up to 0.1% w/w (European Chemicals Agency, n.d.).
EVA presents environmental concerns, including the potential use of chloride catalysts and other by products resulting from ethylene production (Hello Natural Living, n.d.). Most TPEs can be produced in a wide range of hardness without the use of additives. However, thermoplastic polyurethane (PUR) uses significant quantities of chlorine.

According to a study on alternative plastics for inflatable toys, polyolefins can be considered among the least dangerous petrochemical plastics in terms of their environmental impacts through production, use, and disposal (Tickner, 1999). Polyolefins are also extremely versatile with differing properties achieved without the use of plasticizers or other types of additives. For example, PE hardness can be changed by modifying hydrocarbon chain length or cross-linking (Ali Chamas, 2020). Additionally, polyolefin plastics can become biodegradable by creating weak links in the polymer, increasing capacity to degrade by bacteria and other microorganisms. It is petroleum based, however.

In terms of end of life options, EVA offers a wider range of possibilities, such as break-down recycling for re-using, upcycling, or repurposing for the manufacturing of new products (Tickner, 1999; Anchor Travel, n.d.). Polyolefins plastics can become biodegradable by creating weak links in the polymer, increasing capacity to degrade by bacteria and other microorganisms. The properties of TPE make it a highly recyclable material (Vanden Recycling n.d.). PUR can be recycled using chemical methods including alkaline hydrolysis and pyrolysis (GreenMax n.d.).

7.10.6 Availability

All of the alternative materials for inflatable toys and boats analysed here are currently commercially available. Data on the 2019 production and import volumes of these alternative plastics in the EU-27 was retrieved from the Eurostat PRODCOM database (European Commission, 2017; European Commission, 2021a) (Table 7-13). The data shows the EU-27 production capacity and imports for TPE PUR were almost 3 million and over 80 thousand tonnes, respectively. PP exceeds current PVC production and imports, while EVA is produced and imported in much smaller quantities. A key aspect to consider is that all of the alternative materials analysed here are used for several other applications beyond inflatable toys. This is also the case of PVC used for the production of inflatable toys, were only around 3% of the total PVC mass is used for this application. The fact that there are a number of alternative materials that would be technically feasible substitutions to PVC will help meeting an increase in the demand of these alternative materials if necessary. The main competing uses for polyolefins include packaging, electrical cables, piping and other construction applications, furniture, and medical appliances (Plastics Europe, n.d a), while EVA is used in packaging, automotive applications, electronics, and footwear (Plastics Insight, 2021), and TPE is used in adhesives, footwear, medical devices, household goods, and automobile parts (Omnexus, n.d.).

Table 7-25: Material production and import quantities

Substance		2019 production quantity (tonnes) [133]	2019 import quantity (tonnes) [134]
PVC	***Primary unmixed PVC***	***4,960,064***	***504,285***
	Non-plasticised mixed PVC	384,389	28,518
	Plasticised mixed PVC	796,204	47,560
TPE (in general)		No data	
TPE: Primary PUR		2,954,510	81,574
Primary EVA		400,000	99,922
Polyolefins: Primary PP		10,435,915	1,370,022

7.11 Conclusion on suitability and availability

There are a number of alternative materials that have the required technical characteristics for the production of inflatable toys and boats. In some cases, alternatives materials present improved performance characteristics compared to PVC (Tickner, 1999). Limited information on the costs of finished products using alternative materials was identified, but in the majority of cases, the cost of the alternative raw material is higher than PVC. But some alternatives, such as polyolefin elastomers, require less material to achieve the same strength than PVC films used in the production of inflatable toys.

For larger boats, the alternative materials would likely result in increased prices for the finished products, but this would likely be offset by improved durability. The suggested alternatives are currently commercially available in significant quantities. The literature indicates the alternatives presented here are likely to present a reduced risk to human health and the environment, when compared to soft PVC (Tickner, 1999), however effects of the 2018 restriction of several phthalates (European Parliament, 2018a) – including in childcare articles – aim to address several concerns.

7.12 Analysis of alternatives – Boots and soles of shoes

7.12.1 Key performance criteria

PVC is used in outsoles of different types of shoes and wellington boots. Whilst PVC has been used for more specialist ski boots, these are now usually made of other plastics including polyolefins, thermoplastic polyurethane and nylon (Martino Colonna, 2013).

PVC use is based on the following key factors:

- **Durability –** the boot/outsole must be resistant to abrasion damage and wear;
- **Traction –** depending on the type of boot or shoe, the material must provide grip to ensure safe and comfortable use;
- **Water and weather resistance –** outsoles and wellington boots must provide protection from water/ice and mud;
- **Insulation –** materials must provide good insulation to the wearer from the cold; and
- **Comfort –** lightweight, with a degree of breathability and flexibility.
- **Aesthetic factors** are often considered by consumers when assessing different materials used in footwear.

7.12.2 Substance ID and properties

A desktop review of materials used in boots and soles of shoes found that the main alternatives to PVC are natural rubber, polyurethane (PUR), thermoplastic polyurethane (TPU), ethylene vinyl acetate (EVA) and leather (Table 7-26).

Table 7-26: Potential alternatives to PVC in footwear

Alternative substance (or technology)	CAS Number	Application(s)	Key properties and Classification and Labelling Information
Natural rubber	9006-04-6	Boots, soles of shoes	A naturally occurring latex product obtained from rubber trees, formed of polymers of isoprene. It undergoes a vulcanisation process to enhance its toughness and is typically combined with fillers and plasticisers (Satta Srewaradachpisal, 2020). It is highly durable but generally heavier than other materials used in footwear (Safety Shoes Today, 2019).
Polyurethane (PUR)	9009-54-5	Boots, soles of shoes	PUR is a thermosetting polymer of organic units joined with urethane linkages. It possesses good flexibility and a high degree of durability, tear resistance and abrasion resistance (Plastics Insight, n.d a). PUR is manufactured by reacting a polyol with a diisocyanate or a polymeric isocyanate (American Chemistry Council , n.d a). The chemical group of isocyanates is linked with a variety of adverse respiratory impacts (Ian Kimber, 2014). Additives are often included to enhance the properties of the polymer.
Thermoplastic polyurethane (TPU)		Soles of shoes	TPU is a thermoplastic variant of polyurethane, and possess characteristics of both plastics and rubbers, including excellent tensile strength and elasticity (Omnexus, n.d.). It is manufactured through a reaction between diisocyanates and short-chain diols.
Ethylene vinyl acetate (EVA)	24937-78-8	Boots, soles of shoes	A halogen-free elastomeric copolymer of ethylene and vinyl acetate., with rubbery flexibility. EVA typically includes additives to enhance its properties.
Leather	-	Soles of shoes	Leather is a material produced by tanning animal hides to alter their protein structure, resulting in enhanced durability and resistance to decomposition. 80-90% of global leather tanning makes use of salts of trivalent chromium, chrome alum (CAS No: 10141-00-1), chromium (III) sulphate (CAS No: 10101-53-8) and chromium (III) hydroxide sulphate (CAS No: 12336-95-7). Chrome alum is known to cause serious eye and skin irritation (European Chemicals Agency, n.d a), while chromium (III) sulphate causes sever skin burns and serious eye damage, poses long-lasting toxicity to aquatic biota, and is harmful if swallowed (European Chemicals Agency, n.d b). Chromium (III) hydroxide sulphate can cause serious eye and skin irritation, can induce allergic skin reactions, and is harmful to aquatic life (European Chemicals Agency, n.d c). Chromium (III) can undergo oxidisation to form chromium (VI), which is a suspected carcinogen (Fitreach, n.d.). A restriction is in place for chromium VI compounds, including in leather (European Chemicals Agency, 2015). Additionally, a restriction on the placing on the market of textile, leather, hide and fur articles containing skin sensitising substances has been proposed by France and Sweden (European Chemicals Agency, n.d d). Environmentally benign tanning methods based on vegetable-derived tannin are available.

7.12.3 Technical feasibility

A review of the alternative materials to PVC for use in boots and soles of shoes suggests that there are several materials which fulfil the key technical criteria identified above (Table 7-27).

All of the materials assessed are currently used in commercially available footwear. Despite the advantages above, PVC has been noted as providing relatively poor grip, and inferior comfort due to its rigidity and breathability (Topbestshoes, 2021). The alternatives considered exhibit a similarly high level of durability to PVC, with two exceptions; EVA is prone to compression due to its low density (The Wired Runner , n.d.) and leather which - even with the required care - is not as durable as plastics and rubbers (Stridewise , n.d.). Footwear constructed with natural rubber can become brittle with time without the required care and treatment (Welly-King, n.d.).

The evidence indicates all alternatives are comparable to or outperform, PVC in terms of traction, resistance to water and adverse weather conditions, and thermal insulation. The exception is leather, which offers more limited grip on most surfaces, is susceptible to deformation when wet, and offers generally inferior protection to the wearer in colder conditions. The alternatives provide comparable or better comfort, given their flexibility and breathability.

Overall, several identified alternatives meet or exceed the standard of performance of PVC, with the exception of leather which is more fragile. While EVA is durable, it is prone to deformation through compression sustained through prolonged use, which could reduce the lifespan of footwear.

Table 7-27: Technical specifications of PVC and alternative substances used in footwear

Substance	Durability	Traction	Water and weather resistance	Insulation	Comfort
PVC	PVC offers good durability and resistance to abrasion (AirySole , 2019), and does not tear or break easily (IPC , 2019).	PVC non-slip performance is generally limited (AirySole , 2019; IPC , 2019).	Excellent resistance to water (AirySole , 2019; Shoes Consultant , 2019).	PVC offers excellent insulation (AirySole , 2019; Comforting Footwear, n.d.).	Offers limited ventilation, breathability and flexibility (Topbestshoes, 2021), but enables lightweight footwear (Shuperb , 2015).
Wellington boots and soles of shoes					
Natural rubber	Highly robust and durable (Hand Dyed Shoe Co., 2020). Rubber is longer lasting than PVC and offers greater durability (Shuperb , 2015). Wellington boots with a high natural rubber content are susceptible to embrittlement without the necessary care (Welly-King, n.d.).	Natural rubber generally offers good grip and slip resistance (Hiking and Fishing , n.d.; Comforting Footwear , n.d.), but is prone to sliding and slipping in frosty and icy conditions (Shoes Report , 2014).	Very resistant to the elements and suitable for use in all weather types (Hand Dyed Shoe Co., 2020).	Rubber in footwear will keep the wearer warm (AirySole , 2019) and provides good thermal protection (Shoes Report , 2014).	Rubber is heavier than other alternatives (The Wired Runner , n.d.), thus impairing comfort and limiting its suitability in running and sports shoes. Rubber provides a good degree of flexibility and shock absorption, but does not mould to the wearer's feet to the extent that leather does.
EVA	EVA can last for a long time with little damage (Running Shoes Online , 2021). When used in soles, EVA can compress and become flat after prolongued use due to its low density (The Wired Runner , n.d.).	EVA provides a solid grip on different surfaces and terrains (Running Outfitters, 2021).	Due to its closed-cell structure, EVA is non-absorbent and offers a high degree of water resistance (YAPI, 2019).	EVA does not conduct as much heat as other materials and is a better insulator in colder conditions (Running Shoes Online , 2021; Giesswein , 2019).	A very lightweight material offering good cushioning, shock absorption and flexibility (Comforting Footwear, n.d.; Shoes Report , 2014).
PUR	PUR is highly abrasion-resistant but can become brittle in advanced age (Uvex-Safety, 2019). PUR soles are widely considered as the most durable (Klaveness , n.d.; Running Shoes Online , 2021).	PUR provides good grip and prevents slipping (Running Shoes Online , 2021; Uvex-Safety, 2019).	PUR is highly waterproof (Comforting Footwear, n.d.; Polyurethanes , 2021 a).	PUR provides good thermal insulation (Shoes Report , 2014).	PUR offers superb shock absoprtion and is a lightweight material (Comforting Footwear, n.d.). Where used in soles, PUR is flexible (Shoes Report , 2014). PUR is supple, and PUR wellington boots are 35% lighter than PVC or rubber boots (Welly-King, n.d.).
Soles of shoes					

TPU	TPU is study and highly resistant to abrasion, offering greater durability than PU (Safety Shoes Today , 2019a; Badger Australia , 2018) R.	TPU provides a high level of slip resistance on uneven and slippery surfaces (Badger Australia , 2018).	TPU is highly resistant to water and weather conditions (Kaliber , 2021)	Provides good protection against low temperatures (Safety Shoes Today , 2019a).	TPU is flexible and lightweight (Safety Shoes Today , 2019a; Badger Australia , 2018).
Leather	Treated with care, leather outsoles can be resistant to long-term wear and damage, but they are generally not as durable as plastics and rubbers (Stridewise , n.d.).	Leather outsoles can result inferior grip on a variety of surfaces, including concrete and carpet (Standard Handmade, 2018). Leather soles cannot be shaped to provide a wide base thus limiting their application in walking and hiking boots (Stridewise , n.d.).	When worn in wet weather, leather soles can deform, which can require application of protective sprays and impregnations (Beckett Simonon , 2021; Shoes Report , 2014).	Leather outsoles offer greater breathability to the wearer and are well-suited to warmer climates, but do not offer the same level of insulation as rubber and plastic in colder conditions (Stridewise , n.d.).	Possesses natural elasticity (AirySole , 2019) and offers a high level of breathability (Shoes Report , 2014). Leather is a natural membrane and can mould to the shape of the wearer's foot (Beckett Simonon , 2021).

7.12.4 Economic feasibility and economic impact

All of the alternatives to PVC reviewed in this assessment are currently commercially available, indicating they are economically viable in at least some footwear applications. A targeted review of a number of online marketplaces was conducted in order to determine the key economic effects and a search of literature on the materials was completed to ascertain lifespans. Table 1-28 displays market prices obtained via desktop review for wellington boots and outsole components for the various alternatives considered, as well as an indication of lifespans, where such information could be found. Cost information was obtained for pairs of wellington boots as well as outsole components for shoes. The data available indicate that all alternative materials used in wellington boots present an increase in cost compared to PVC. EVA boots are generally closest to PVC in cost, natural rubber boots typically start at over twice the starting price of PVC boots, and PUR is substantially costlier than PVC; described as a premium product by one specialist vendor (Welly-king , 2021).

The price of outsole components made from natural rubber, EVA, PUR and TPU are all comparable to one another, and slightly higher than costs for PVC components. Leather outsoles typically cost considerably more. The various plastic materials assessed alongside natural rubber are all as durable if not more so than PVC. EVA is susceptible to deformation resulting from prolonged use, which may shorten the usable lifespan of EVA footwear. Leather is noted as especially susceptible to deformation and damage (Table 7-27) and is noted to have a limited lifespan of up to two years of "casual use".

Table 7-28: Technical specifications of PVC and alternative substances used in footwear

Substance	Market prices [195, 196]	Lifespan
PVC	Prices for adult wellington boots typically start at approximately €13. Outsole components typically start at approximately €0.60.	Highly durable (see Table 7-27).
Wellington boots and soles of shoes		
Natural rubber	Prices for adult wellington boots typically start at approximately €30. Outsole components typically start at approximately €0.85.	Highly durable (see Table 7-27).
EVA	Prices for adult wellington boots typically start at approximately €18. Outsole components typically start at approximately €0.85.	Noted as a durable material, but prone to deformation with prolongued use [341] (see Table 7-27).
PUR	Prices for adult wellington boots typically around €40. Outsole components typically start at approximately €0.85.	Highly durable (see Table 7-27).
Soles of shoes		
TPU	Outsole components typically start at approximately €0.85.	Highly durable (see Table 7-27).
Leather	Outsole components typically start at approximately €13.	With casual use, 1-2 years (Nicks Boots , 2020).

7.12.5 Reduction of overall risk due to transition to the alternatives

PUR is typically produced using a process where isocyanates are reacted with a polyol (Plastics Insight 2021). Once formed, PUR is inert and presents no health risk to humans. However, the isocyanates used in their production are recognised as potentially carcinogenic with links to

[195] Wellington boots prices determine from welly-king.co.uk. GBP to EUR exchange rate of 1.17 used.

[196] Outsoles component prices determined from alibaba.com. USD to EUR exchange rate of 0.85 used.

asthma and pulmonary sequelae (Plastics Insight, n.d a). As such, PUR presents occupational hazards during the production phase of footwear.

As part of production of a leather outsole, animal hides undergo a tanning process to make them firm. Up to 90% (Alta Andina, n.d.) of the leather produced worldwide is tanned using chrome salts, particularly chrome alum (CAS No: 10141-00-1), chromium (III) sulphate (CAS No: 10101-53-8) and chromium (III) hydroxide sulphate (CAS No: 12336-95-7). Chrome alum is known to cause serious eye and skin irritation (European Chemicals Agency, n.d a), while chromium (III) sulphate causes sever skin burns and serious eye damage, poses long-lasting toxicity to aquatic biota, and is harmful if swallowed (European Chemicals Agency, n.d b). Chromium (III) hydroxide sulphate can cause serious eye and skin irritation, can induce allergic skin reactions, and is harmful to aquatic life (European Chemicals Agency, n.d b). These substances present a potential health hazard to workers involved in leather production (depending on prevailing regulatory standards and PPE). In addition, the chrome-based tanning process produces toxic wastewater that can result in significant environmental impacts (Axess Wallets , n.d.). Plant-based tanning processes make use of vegetable-derived tannin (Tannins.org, 2019) and bypass the need for hazardous and environmentally damaging chrome slats, but vegetable-based tanning makes up a small portion of global leather production.

Concerns associated with additives remain for some of the other plastics used as alternatives to PVC in footwear, and switching to polymers such as EVA, PUR and TPU as well as natural rubber (which is also susceptible to leaching additives, including phthalates (Imanda Jayawardena, 2016)) would present similar concerns. Environmentally benign formulations of TPU, marketed as “EcoTPU”, have been commercially available for use in footwear since 2012 and are 60% based on renewable plant origin material (Resimol , n.d.). Note, data on the additives used in EcoTPU formulations was not identified. However, it is not clear how widely used EcoTPU is in footwear at present.

The presence of additives in alternative materials presents the same obstacles to recycling currently observed with PVC, with incineration and landfill the more feasible routes for disposal. At present, recycling of EVA formulations is limited in scale (Greenmax, n.d.). The limited recycling options available for PUR from footwear at present generally yield a low-quality, low-value product, although there is appetite within footwear manufacturing to expand recycling options (Ecco, 2020). EVA, PUR and TPU are halogen-free substances, and provided that halogen-based additives are not used to enhance their properties, they do not pose the same risk as PVC in this regard.

7.12.6 Availability

All of the alternatives to PVC use in footwear in this assessment are currently commercially available. Material production and import data for 2019 for the EU-27 have been obtained from the Eurostat PRODCOM database (European Commission, 2021; European Commission, 2021b) (Table 7-29). The quantity of PVC used in clothing, including boots and soles of shoes, is around 2% of the total virgin PVC volume on the market. While the quantities of alternative materials produced and imported is significantly smaller than the quantities of PVC available annually, boots and soles of shoes constitute a very small portion of the PVC used. Taking this into account, additional demand would likely be able to be met without undue delay, depending on the timescale of a phase-out. The main competing uses for EVA are packaging, automotive applications, electronics, and footwear (Plastics Insight, n.d.). PUR is used principally in thermal insulation, furniture and padding and coatings for materials and surfaces (Townsend Chemicals, 2019).

Table 7-29: Material production and import quantities for alternatives to PVC in footwear

Substance		2019 production quantity (tonnes) (European Commission, 2021b)	2019 import quantity (tonnes) (European Commission, 2021a)
PVC	***Primary unmixed PVC***	***4,960,064***	***504,285***
	Non-plasticised mixed PVC	***384,389***	***28,518***
	Plasticised mixed PVC	***796,204***	***47,560***
Natural and modified natural polymers, in primary forms (including alginic acid, hardened proteins, chemical derivatives of natural rubber)		No data	
Primary EVA		400,000	99,922
Primary PUR		2,954,510	81,574
Leather	**Leather of bovine animals, without hair, whole**	506,373	337,334
	Leather of bovine animals, without hair, not whole	183,823	61,945
	Leather of equine animals, without hair	169	2,162
	Leather of other animals, without hair on	20,692	812

7.12.7 Conclusion on suitability and availability for alternatives

The analysis of alternatives to PVC in footwear has identified a number of technically feasible alternatives. These are natural rubber, EVA, PUR and TPU. While leather is used in soles for specialised footwear and fashionwear, its technical qualities likely mean it is an unsuitable substitute to PVC soles in many cases. The available market data suggest that in wellington boots, alternative materials present potentially significant additional costs when compared to PVC. Sole components made from alternatives are generally similarly priced to one another (with the exception of leather soles, which are substantially more expensive), but slightly higher than PVC.

With the exception of leather, alternatives offer a similar degree of durability as PVC and would not lead to additional costs through a shortened average lifespan. Alternative plastics face many of the same risks as PVC with regard to hazardous additives, although all assessed plastics are halogen-free and would not generate the same toxic emissions as PVC from combustion (see Section 7.12.5) provided that any additives used are also halogen-free. PUR presents potential occupational hazards to employees working in its production through isocyanates, and leather tanning using chrome-based techniques are a risk to human health and aquatic life – although this use of Chromium VI in leather articles coming into contact with the skin is now restricted in the EU (European Chemicals Agency, n.d.). The volumes of alternative materials currently placed on the EU-27 market are likely sufficient enough to ensure that additional demand arising from a PVC phase-out could be met through increased production and/or imports.

7.13 Analysis of alternatives: Automotive (dashboards and artificial leather)

7.13.1 Key performance criteria

The average car today has more than 1,000 plastic components and roughly 12% of them are made of soft PVC. The full range of components include cup holders, arm rests, door handles and panels as well as dashboard covers, console components and knobs (Griffin-Rutgers, n.d.). This section focusses on two specific internal uses; PVC vinyl coverings, including artificial leather, on dashboards and seats. Note that PVC can also be blended with acrylonitrile butadiene styrene (ABS) to form sheets which are also used in automobiles for interior coverings. The key performance criteria for the PVC articles and for any replacement materials are as follows (Nitin Girdhar Shinde, 2020; PVC Forum - South East Europe, n.d.; Plastics Today, 1999; Car Safety Design Features, n.d.; Vinyl Council Australia, n.d.):

- The material should be **lightweight**; plastic dashboards and other components - along with a wide variety of other components and technological developments - have contributed to weight reductions and fuel efficiency improvements in automobiles.
- It should provide **fire/flame retardancy properties**
- It needs to **withstand high temperatures and exposure to sunlight**, including via larger windscreens common in modern cars.
- Some manufacturer information suggests PVC provides important safety features when used in dashboards. It is unclear how much of this is delivered by the PVC cover, rather than the foam underlay, which is usually made from Polyurethane. But any material should be **able to absorb the high-impact energy in the event of a collision**, avoiding sharp edges and surfaces and protecting occupants and other road users (AZO Materials, 2001).
- Commercial literature also indicates PVC coverings provide more general **convenience and aesthetic benefits**. These include that they are flexible/formable, so can take up minimum space; aiding with noise and vibration dampening in the interior of the car and that they prevent scratches, providing a smooth and glossy effect (PVC Forum - South East Europe, n.d.).

7.13.2 Substance ID and properties (or Description of alternative technique)

Desktop research identified several alternatives for PVC in dashboard and/or seat coverings, discussed below.

Table 7-30: Potential alternatives to PVC in automotive interiors

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
Leather	Animal hides are dried, cleaned tanned, rolled and stretched, before applying a matt or glossy finish. This final stage is often carried out by the seat manufacturer or car upholstery company (One 4 Leather, n.d.). The EC restricted chromium VI above certain concentrations in leather, this includes car seats (and steering wheel covers). A further restriction on skin sensitisers used in textiles (including leather upholstery) is currently being considered (European Chemicals Agency, n.d.; European Chemicals Agency, 2020)	Some chemical treatments are used in both the cleaning and tanning stages, alongside high levels of water use. If it contains sensitising chemicals, leather can cause contact dermamtius/allergies (Costa, 2019). Although the EC has taken action on e.g leather articles.
Cloth or natural fibres.	Typically nylon or polyester (Simoniz, 2020) (Polyethylene terephthalate (PET), CAS No: 25038-59-9) (Independent	There are no harmonised classifications for these substances.

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
	Commodity Intelligence Services , 2010). Nylon-66 is the most common polymer type CAS No 32131-17-2). Natural fibres such as kenaf, hemp, flax, jute, alongside thermoplastic / thermoset matrices are also noted (Shinde & Patel, 2020).	
Acrylonitrile butadine styrene (ABS) (AC Plastics Inc, n.d.; Omnexus , n.d.; Smith, 2018)	ABS sheet provides a sleek/glossy finish. It is made from three polymers: • Acrylonitrile (CAS: 107-13-1) a synthetic monomer produced from propylene and ammonia. • Butadiene (CAS 106-99-0): is produced as a by-product of ethylene production from steam crackers. • Styrene (CAS 100-42-5): is manufactured by dehydrogenation of ethyl benzene.	Acrylonitrile is toxic if swallowed/in contact with skin/inhaled, may cause cancer, is toxic to aquatic life with long lasting effects, is a highly flammable liquid and vapour, causes serious eye damage, causes skin irritation, may cause an allergic skin reaction and may cause respiratory irritation (European Chemicals Agency, n.d.). Butadine may cause genetic defects, may cause cancer and is an extremely flammable gas (European Chemicals Agency, n.d.).Styrene causes damage to organs through prolonged or repeated exposure, is a flammable liquid and vapour, causes serious eye irritation, is harmful if inhaled, is suspected of damaging the unborn child and causes skin irritation (European Chemicals Agency, n.d.).
Polypropelyne (PP) (Total Pipe Systems, n.d.; Automotive Plastics, n.d.)	PP (CAS: 9003-07-0) is one of the most common plastics used in cars, which includes integrated dashboard units. The propylene monomer is subjected to heat and pressure in the presence of a catalyst (British Plastics Foundation, n.d.). A hypothetical option would involve dispensing with the PVC covering for the dashboard; this is common in "lower segment" cars (Shinde & Patel, 2020). Polypropylene seat coverings are commercially available (Helligar, 2016) and literature suggests PP textile covers in automotive applications are "emerging" (British Plastics Foundation, n.d.).	There are no harmonised classifications for the substance (European Chemicals Agency, n.d.). PP is obtained via cracking naptah (light oil distillate) (British Plastics Federation, n.d.).
Thermoplastic olefin (TPO/TPE-O)	TPO compounds are resin blends of polypropylene (PP) and un-crosslinked Ethylene Propylene Diene Monomer (EPDM) rubber and polyethylene (PE) (Hexpol, n.d.).	Ethelylene (CAS: 74–85-1) is flamable and may cause drowsiness/dizzyness (European Chemicals Agency, n.d.). Propylene/ propene (CAS 115-07-1) is also extremely flammable (European Chemicals Agency , n.d.).
ABS/PC Blend	Some literature refers to blends of ABS and Polycarbonate (PC - CAS: 25037-45-0) being used for car dashboards, however it is assumed this is for the dash unit, rather than coverings.	Bisphenol A (CAS: 80-05-7) has been used in polycarbonate plastic for many years. It can damage fertility and has been identified as a substance affecting the hormonal systems of humans and animals. In addition, it damages eyes and may cause allergic skin reactions and respiratory irritation. There is a call for evidence for a much broader restriction on its use, beyond a restriction already in force on its use in thermal paper (European Chemicals Agency, n.d.).

7.13.3 Technical feasibility

There are two alternative approaches to the use of PVC in automotive interiors. First redesign of components to avoid the need for PVC where multiple materials comprise one assembled part and second, substitution of PVC to alternative materials. As such, there are a number of technically feasible alternatives to PVC use in the seat/dash components. Several have been commercially available and have been installed by a range of downstream users and consumers, in some cases for many years.

The use of PVC in internal automobile components was phased out by a major car manufacturer from as early as 1995, - all use of PVC by that company was expected to cease by 2000. Other major manufacturers have made public statements regarding phase out "from 2010" (ENDS Report, n.d.), or to reduce its use, using both new design solutions as well as alternative materials to do so based on ecological and economic criteria (Greenpeace International, 2003). In the United States, phase out of PVC by one company was driven in part by concerns over durability (they found it cracks, warps and fades too quickly), that it was heavier than alternatives and resulted in fogging from leaching of plasticisers (note this was a publication from 2003, it has not been identified in other publications and it is not clear if this problem persists) (Greenpeace International, 2003).

Table 7-31: Technical feasibility of alternatives to PVC in automotive interiors

Alternative substance (or technology)	Lightweight	Fire/flame retardancy	Temparature/UV resilience	Other
Leather	Yes, weight depends on thickness. Leather is widely used in automotive interiors.	Offers natural and relatively high resistance to fire, but it is generally a flammable material. Leather seats pass fire standards in cars at present and aircraft for example (Leather Dictionary , n.d.).	Extended exposure can lead to fading and drying, unless it is maintained. Experiments have observed improved protection with antioxidants (tocopherol and butylhydroxytoluene (BHT)) (All Aces, n.d.).	Durable and easy to clean and maintain, although can require some treatment. Can tear (Simoniz, 2020).
Cloth or natural fibres	Yes, weight influenced by material and thickness. Various cloths are widely used in automotive interiors.	Flammable, but prexisting standards are widely used, although some manufacturers have tightened requirements to stipulate slower burn rates (Satra Technology, n.d.).	Extended exposure to direct sunlight can lead to fading. Window tints, or covers can mitigate this (Sunbusters Winder Tinting Limited, n.d.).	Comfortable, durable and affordable. Can stain and retain odour (Simoniz, 2020).
Acrylonite butadine styrene (ABS)	Yes. Widely used in automotive interiors.	It is noted that "ordinary" grades can burn easily (Omnexus , n.d.) and non fire retardent, without additives (McClarin Composite Solutions, n.d.). But are currently used which adhere to existing safety standards.	ABS does not resist ultraviolet degradation unless UV stabilizer is added to the resin before extrusion (McClarin Composite Solutions, n.d.).	High heat resistance, good impact and stain resistance. Mechanically strong and stable over time. Recyclable (Omnexus , n.d.).
Polypropelyne (PP)	Yes, low density relative to other common plastics (Creative Mechanisms, 2016).	Flammable. Burns with a relatively low amount of smoke, flame retardents may be used (Kiron, n.d.).	High degradation rates when exposed to the sun whic can weaken the plastic (Zande, 2015).	Durable, maleable, a good insulator with high tensile strength (Mayco International, 2020) and impact and stiffness balance (British Plastics Foundation, n.d.).
Thermoplastic olefin (TPE-O)	Yes, used in car interiors at present (World Industrial	The susbtance itself is described as combustable (Hoehn Plastics, n.d.), also as	Yes, although may require a lining of calcium, calcium carbonate or talc	Impact and scratch resistant (Corrosionpedia, 2020). Ford of Europe stated

Alternative substance (or technology)	Lightweight	Fire/flame retardancy	Temparature/UV resilience	Other
	Reporter, 2013).	flame retardant. Currently used, adhering to existing safety standards.	(Corrosionpedia, 2020).	this was a candidate material for phase out of PVC (Greenpeace International, 2003).
ABS/PC Blend	Yes. Widely used in automotive exteriors and interiors, often under a textile covering.	As above for ABS. Flame redardent grades of PC exist (British Plastics Federation , n.d.).	As above for ABS. PC Maintains rigidity up to c.140°C (British Plastics Federation , n.d.).	PC has very high impact strength (Rogers, 2015). Not suitable for seating coverings.

7.13.4 Economic feasibility and economic impacts

There are several economically feasible alternatives to PVC in seat/dash components. These are commercially available and are currently used by some manufacturers. The key issues include the unit price for alternatives versus PVC as well as the durability of the article. The evidence suggests that whilst some of more expensive, others are comparable, as is durability. The data above suggests that the use of PVC for interior use, specifically in dashboards and seating, could have decreased further than the volumes estimated in chapter 3, given the development of alternatives by some major manufacturers. Whilst PVC was estimated to have been used in nearly all new car interiors in 2006, that figure had decreased to 73% by 2012, with manufacturers stating it has continued to fall. Some major manufacturers have claimed to have eliminated its use entirely (BBC, 2016). Despite this, public market analysis indicates that PVC is still the dominant material (Bregar, 2019) with demand expected to grow, driven by continuing efforts to improve fuel economy from reduced vehicle weight (ITB Group, n.d.).

Table 7-32: Economic feasibility of alternatives to PVC

Alternative substance (or technology)	Unit Price	Durabilty and other notes
Leather	A very wide variety of price data are available which reflect product ranges. Qualitative information indicates these are more expensive than PVC, however some car buying guides indicate they can be equivalent price (Classic Motoring , 2019).	Without maintenance it can wear faster than PVC (Classic Motoring , 2019).
Cloth or natural fibres	A very wide variety of price data are available which reflect product ranges. Qualitative information indicates these are often cheaper than leather.	Durable, although differs based on material. Whilst can stain, it can be cleaned.
Acrylonite butadine styrene (ABS)	Prices vary, but public data suggests between \$10,000 to \$16,000 per tonne (2017) (Plastics Insight, n.d.). The same source indicates PVC prices around \$2,000 to \$2,500 per tonne (Plastics Insight, n.d.).	Strong, durable with high impact properties. Manufacturer literature suggests heat and UV stabilisers can be added (Omnexus , n.d.). Is used for dashboards or covers and provides a shiny aesthetic effect, but heat shrinkage has been noted (Shinde & Patel, 2020).
Polypropelyne (PP)	Prices vary, but public data suggests between \$3,500 to \$1,200 per tonne (2017) (Plastics Insight, n.d.). Other sources suggests >£1,000 per tonne for the polymer (British Plastics Federation, n.d.).	Highly, durable with good chemical and fatigue resistance. PP dashboards are "becoming increasingly achievable,[with] PP film cushioning, film skins, and powder slush moulding and even blow moulded parts with integral PP textile covers are emerging"" (British Plastics Foundation, n.d.). Is being used in place of PVC for covered dashboards (Shinde & Patel, 2020). Good UV resistance.

Alternative substance (or technology)	Unit Price	Durabilty and other notes
Thermoplastic olefin (TPE-O)	Suited to ”high volume low cost applications” (British Plastics Federation, n.d.). Extensively used in the automotive industry (Polymer Properties Database, n.d.). Literature referring to the use of TPO (in roofing), suggests that it is less expensive than PVC (Roofing Calculator , 2021).	Used where there is a requirement for increased toughness and durability (British Plastics Federation, n.d.; Hexpol, n.d.). Is being used in place of PVC for covered dashboards (Shinde & Patel, 2020).
ABS/PC Blend	As above for ABS. Market data suggest prices for Polycarbonate (Manufacturers in China) at around $3,000 - $4,000 per tonne (2017-2018) (Independent Commodity Intelligence Services, 2018).	Durable and extremely tough. Used for various internal and external components including control panels (Chauhan, et al., 2019). Good UV resistance (Szeteiová, 2010). Not suitable for seating.

7.13.5 Reduction of overall risk due to transition to the alternative

Additives of concern in automotive interiors include phthalate plasticisers used in PVC seat fabrics (including artificial leather), instrument panels and interior trim (including dashboards). Tests undertaken by the NGO the ecology centre indicate that both drivers and passengers can be exposed to phthalates in cars via inhalation of air and dust. They note frequent exposure to heat and UV increases exposure levels and may exacerbate risk. Emission of dioxins from vehicle fires is a further, more limited, concern (Gearhart & Posselt, 2006).

The use of several alternatives is likely to increase in the event of a phase out of PVC. The alternatives however are not free from risk. Two involve use of ABS, either on its own or as part of a blend with Polycarbonate (PC). The latter is commonly manufactured with Bisphenol A, a suspected endocrine disrupter (see section 7.13.2). However, PC is not likely be suitable for many of the flexible articles where PVC has been used. ABS and/or thermoplastic olefins pose several risks, including cancers and genetic defects and can be combined with flame retardants (Pardon My Plants, 2014). The balance of evidence suggests that the other potential materials, beyond those above, are likely to be of lower risk.

In terms of recycling, leather and cloth (nylon and/or polyester) (Sustain Your Style, n.d.; Textiles Environment Design, n.d.) are recycled. Whilst, general recycling rates of PP have historically been low, recent technological development (commercialised at scale in July 2019) indicates some of these underlying issues are being remedied and rates could improve (Chasan, 2019); recycling rates of plastics in end of life vehicles in France are as high as 63% for some PP components and up to 42% for ABS (Aigner et al., 2020). Note, the corresponding detail for PVC has not been identified. ABS is recyclable, via froth flotation, grinding, melting and reforming. Most municipalities do not recycle ABS, however (Vanden, n.d.). Using TPO in place of PVC in car interiors, driven by the possibilities of recycling was discussed in technical journals as early as 1993 (Joshi, 1993). The chemical substances used in manufacturing or processing for several materials (leather, ABS, PC) pose circular economy challenges (Leather Sustainability, 2018).

7.13.6 Availability

The alternatives are commercially available in significant quantities. For several materials, detailed information on EU production volumes is available via PRODCOM. This illustrates the scale of existing supply. For other materials, volume data is more limited but estimates of the financial value of markets are available, again providing indications of scale. PRODCOM data for the EU (28) for 2019 show the following production tonnages:

- Polypropylene - over 10 million tonnes[197], with global demand of 45 million tonnes (Creative Mechanisms, 2016)
- Acrylonitrile - 500,000 tonnes[198]
- Ethylene - 10 million tonnes[199]
- Leather - well over 600,000 tonnes[200].

The global polyolefin market was estimated at nearly 150 million tonnes in 2016 (Market Analysis Report, 2018). No quantitative data has been identified on typical timescales for phase out from the car manufacturer information described above. The main competing uses of polypropylene are industrial applications, consumer goods and furniture (Omnexus, n.d.), while acrylonitrile is used in clothing and textiles, sportswear, carpets and upholstery (Public Health England, 2017).

7.13.7 Conclusion on suitability and availability for alternatives to PVC in automotive interiors

Alternatives to PVC in automotive seats/dashboards are technically and economically viable . They are commercially available and have been so for many years, in large volumes. An overall reduction in human health and environmental risk compared to continued PVC use is likely from the majority of alternative materials or plastics due to reduced use of additives including phthalate plasticisers (see Section 7.13.5). However, the potential for increased demand for PC and ABS presents occupational health risks related to input chemicals used in their production as well as circular economy concerns from chemical substances used in processing and treating. This makes an overall comparison of the net change in risks challenging without more detailed LCA data.

7.14 Analysis of alternatives: Medical applications (blood bags, medical devices and gloves)

7.14.1 Key performance criteria

The two main applications for PVC in medical applications are flexible containers (e.g. blood bags) and tubing (general medical tubing including endotracheal tubing, nasal cannulas). Other medical uses include oxygen masks, IV infusion sets and blister packaging for medication (PVCMed, n.d.). The key performance criteria for the PVC articles and for any replacement materials are as follows (Shinde & Patel, 2020; South East Europe PVC Forum, n.d.; Plastics Today, 1999; Car Safety Design Features, n.d.; Vinyl Council Australia , n.d.):

- The articles need to be **elastic and flexible**, but with **good tensile strength** so they do not tear.
- Blood Bags require blood to be **safely stored** (a period of 49 days is referenced in the literature) (Lozano & Cid, 2013; University of Birmingham, n.d.).
- The articles need to be **heat resistant** to enable sterilisation by steam prior to use (Fosmedic, 2017).
- They also need to be **resistant to chemicals** that may be present (e.g. cleaning fluids etc).
- One of the perceived advantages of PVC gloves is **avoidance of latex allergies** (Shield Scientific, 2008).
- The alternatives need to be **biocompatible** and would already have or be able to obtain approval for Medical Device use and a Conformité Européenne (CE) mark under the Regulations on Medical Devices (EU 2017/746) (European Medicines Agency , n.d.).

[197] 20165130 - Polypropylene, in primary forms
[198] 20144350 – Acrylonitrile
[199] 20141130 – Ethylene
[200] 15113200 - Leather, of bovine animals, without hair, not whole and 15113100 - Leather, of bovine animals, without hair, whole

7.14.2 Substance ID and properties (or Description of alternative technique)

Desktop research identifies several alternatives for PVC in these medical applications. These are discussed below.

Table 7-33: Potential alternatives to PVC in medical applications

Alternative substance (or technology)	Notes (CAS Number if relevant)	Key properties and Classification and Labelling Information
Polyethelyne (PE) and PMPC (Lloyd, 2004)	An alloy of PE (CAS No: 9002-88-04) and biocompatible poly(2-methacryloyloxyethyl phosphorylcholine) (PMPC). CAS No: 67881-98-5). The mixed polymer system is produced by dissolving PE and PMPC in xylene and n-butanol, respectively (Chemical Book, n.d.).	PMPC may cause skin sensitisation (European Chemicals Agency , n.d.). Ethylene (Cas No: 74-85-1) is extremely flammable and may cause drowsiness and dizziness (European Chemicals Agency, n.d.).
Polypropelyne (Lloyd, 2004)	A thermoplastic made from propylene monomer PP (CAS: 9003-07-0).	PP has been cited as an alternative in tubing (Raumedic , n.d.). There are no harmonised classifications for the substance. PP is obtained via cracking naphtha (light oil distillate). See table 7-30.
Polyurethane (Lloyd, 2004)	A rigid or flexible plastics, based on diisocyanates (TDI and MDI) and polyols (Polyurethanes, n.d.). CAS No: 9009-54-5 – or 68083-75-0 for the prepolymer.	The prepolymer has several notified classifications under the C&L inventory, including skin and respiratory sensitisation (European Chemicals Agency, n.d.).
diisononylester of cyclohexanedicarboxylic acid (DINCH) (Lozano & Cid, 2013)	This is an alternative plastisiser which has been used in place of DEHP, for example. CAS Number (166412-78-8) (Special Chem, n.d.). This would not enable replacement of PVC as such.	No hazards are classified in REACH Registrations (European Chemicals Agency , n.d.).
Silicone rubber (Lloyd, 2004)	Raw materials include silicon, methanol and hydrogen chloride. Products include silicone polymers. Medcial uses include tubing (American Chemistry Council, n.d.).	Silicone polymers can contain Octamethylcyclotetrasiloxane (D4) – CAS 555-67-2; Decamethylcyclopentasiloxane (D5) – CAS: 541-02-6. These substances are the subject of a possible restriction when used on their own or in formulations for consumer and professional products. An earlier restriction for wash-off cosmetic products has been in effect since January 2020. This is based on concerns over PBT/vPvB properties (European Chemicals Agency, n.d.). ECHA recommended the substances for authorisation in April 2021 covering other uses (European Chemicals Agency, n.d.).
Nitrile (Health Care Without Harm, n.d.)	Nitrile gloves are widely used and made from synthetic (Nitrile butadiene rubber- NBR) rubber (Omni International Corp, n.d.).	NBR a copolymer from acrylonitrile and butadiene. Acrylonitrile enhances chemical resistance, and butadiene creates flexibility and tear resistance (Omni International Corp, n.d.).
Ethylene vinyl acetate (EVA)	Copolymer of ethylene and vinyl acetate. The weight percent vinyl acetate usually varies from 10 to 40%, with the remainder being ethylene (Monmouth Rubber & Plastics Corp, n.d.).	References cite use in medical tubing, alongside SBR rubber (Raumedic , n.d.; Health Care Without Harm, n.d.)

7.14.3 Technical feasibility

The evidence suggests that technical feasible alternatives to PVC are available for use in a range of medical tubing uses as well as for gloves. Their suitability for use in blood bags is more problematic at present. However, alternatives are in development which appear to offer acceptable functionality for this use. These include alternative plastics as well as an alternative plasticiser to DEHP..

A 2003 Greenpeace report provides a summary of experience of PVC substitution in the Medical sector (Greenpeace International, 2003). Whilst dated, this serves to illustrate actions taken in advance of some of the more recent alternatives listed below. These include:

- **Austria:** "The Paediatric Clinic began phasing out PVC in 2001. Currently, nearly all medical products with so- called invasive uses, such as catheters and tubing used in the Paediatric Clinic, are PVC-free. Pacifiers, IV bags, blood filters, respiratory therapy equipment, feeding tubes and other tubing used at the Clinic are made from alternative materials. PVC is used for only a few non-invasive products, such as urinary bags, since no alternatives are on the market yet. However, this was the case for only about 3 % of all products used, and even with those, hospital management expects that PVC-free products of the same performance quality will be on the market "within one to two years".
- **Denmark**: Grenaa Central Hospital have phased out/substituted 95 per cent. of their PVC usage.
- **Germany:** German medical products companies (two examples were listed in the report) "offer several products (incl. complicated ones like dialysis sets (consists of bag, tubing etc.) labelled as 'PVC free'.
- **Japan:** The report cites a company which switched from PVC to PP for dialysing fluid bags in 1999, later developing a new PP based polymer.
- **United States:** A supplier of PVC free IV bags reported increasing market share, having switched from PVC in its products.

Table 7-34: Technical feasibility of alternatives to PVC in medical applications

Alternative substance (or technology)	Flexibilty and strength	Heat and chemical resistance	Other notes.
Polyethelyne (PE) and PMPC	PE has been used in various medical devices, is impact resistant and durable (BMP Medical , n.d.). In combination with PMPC the alloy "could be used in place of PVC" (Ishihara, et al., 2004).	Does not fade, retain dangerous bacteria and can withstand harsh cleaning agents (BMP Medical , n.d.).	Use of PE alone in blood-contacting applications is limited, due to blood clot on the material surface. The addition of PMPC markedly reduces this (Lloyd, 2004).
Polypropelyne	Used in medical devices. Good mechanical performance, durable and can be re—used (BMP Medical , n.d.).	Used where steam sterilised medical devices are necessary (BMP Medical , n.d.).	More commonly used in syringes, surgical textiles, IV solutions bags and labware; with around 10% in injection moulded devices (Akre, 2012).
Polyurethane	Flexible. Used for general purpose medical gloves, tubing various injection moulded devices as well as e.g. catheters or artificial hearts. Durable with good strength and abrasion resistance (American	Chemical resistance depends on material type (Gallagher, n.d.). PU application are used in application that need to sustain very high temperatures. Fillers may be used to increase heat resistance (Amado, 2019). PU coated gloves are	A versatile material with good biocompatibility (American Chemistry Council, n.d.).

Alternative substance (or technology)	Flexibilty and strength	Heat and chemical resistance	Other notes.
	Chemistry Council, n.d.).	marketed as heat resistant gloves (Safety Gloves, 2021).	
DINCH	An non phthalate plasticiser developed specifically for use in medical applications (Hexamoll DINCH, n.d.).	Can be sterilised without effect on tensile elongation (comparable to DEHP) (Hexamoll DINCH, n.d.).	It is noted as suitable for use in blood bags, storing of plasma, for gloves as well as infusion/IV treatment (Hexamoll DINCH, n.d.).
Silicone Rubber	Good tear strength and flexibility (Shin Etsu Silicone, n.d.).Used in medical applications (American Chemistry Council, n.d.).	Yes. Can withstand over 150 Degrees Centigrade indefinitely, steam, oils and solvents (Shin Etsu Silicone, n.d.).	Can withstand repeated sterilisation (American Chemistry Council, n.d.).
Nitrile	Not as elastic or flexible as their latex, but more durable and resistant to chemicals. Suited to medical use as puncture-resistant (Ventyv, 2020).	Resistant to chemicals (Ventyv, 2020). Nitrile gloves are marketed as heat resistant (although these are combined with other materials, including neoprene) (Grainger, n.d.).	Avoids risk of latex allergy reactions (Ventyv, 2020).
EVA	Flexible (rubbery), good low temperature flexibility (-70'C), good chemical resistance (British Plastics Foundation, n.d.).	Good chemical resistance (Omnexus, n.d.). EVA resin has a melting point of 75 – up to 96 Degrees Centigrade (Plastics Insight, n.d.).	Transparent, high friction coefficient (British Plastics Foundation, n.d.).

7.14.4 Economic feasibility and economic impacts

There are economically feasible alternatives to PVC. These are commercially available and currently used. These are typically more expensive, per tonne, than PVC.

Table 7-35: Economic feasibility of alternatives to PVC in medical applications

Alternative substance (or technology)	Unit Price	Other notes
EVA, PE, or PP	- EVA prices vary between c. $2,000 per tonne to $5,000 (2017) (Plastics Insight, 2021). - For PP – as noted above whilst prices vary, but public data suggests between $3,500 to $1,200 per tonne (2017). Other sources suggests >£1,000 per tonne for the polymer (Table 7-32). - PU: Prices appears relatively stable but differ significantly between production locations. From $5,000 per tonne (India) to $12,000 per tonne (Japan), 2017 data (Plastics Insight, n.d a).	PVC-free bags made of (EVA), multilayer polyethylene or polypropylene are described as cost-effective and technically competitive with PVC bags. Hospitals can lower per-unit costs via bulk purchasing minimising price differentials, especially for bags (Health Care Without Harm, n.d.; Health Care Without Harm, n.d.).
DINCH	DINCH plasticiser is commercial available from around $2,000 per tonne (Alibaba, 2021).	Note: DINCH is an alternative to existing phthalate plasticisers, and could still be used in PVC.
Silicones rubber	Global average of $13,300, per tonne for formulated silicone products used in	Silicones used for healthcare are comparatively more expensive on average

Alternative substance (or technology)	Unit Price	Other notes
(polymers)	healthcare applications (American Chemistry Council, n.d.).	than other silicone products.
Nitrile	Prices vary widely, but are commercially available from £10.00 to c. £25.00 per 100 (Just Gloves, 2021).	Nitrile gloves are more costly than PVC (although there has been a significant cost reduction in recent years) they are also more durable (Health Care Without Harm, n.d.).

Information provided during stakeholder consultation provided further information based on practical experience of phase out of PVC in the healthcare sector in Norway and Sweden. This indicated that several medical components are in the process of being phased out, with the exception of blood bags; issues persist with substituting PVC use in blood bags due to their reliance on the phthalate DEHP to interact with the red blood cells and extend the usable lifespan of the blood (Lozano and Cid 2013). The phase out did result in increased prices – which differed by component. This is discussed further in chapter 8.

7.14.5 Reduction of overall risk due to transition to the alternative

A 2019 chemicals manufacturer presentation indicated that at that time PVC had the largest share of medical plastics (40%). It stated that 85% of disposable medical devices are made of PVC and that disposal is fundamental to reduce the occurrence of transfusion-transmitted diseases (pre-sterilized, single-use devices). Moreover the manufacturer expected annual growth in the PVC in medical devices market of 12% (it is not clear if these data are global, or relate to specific global regions) (Hexamoll DINCH, n.d.). More specifically, data indicates DEHP/DINP may represent between 22% to 44% of total glove composition (Shield Scientific, 2008).

Particular concerns have been raised over the presence of phthalate plasticisers in medical products, given the risk of transfer to patient bodies compounded by the likely vulnerability of a patient at the time of receiving treatment (Health Care Without Harm, n.d.). Of these, Di(2-ethylhexyl) phthalate (DEHP) is the predominant plasticizer added to PVC; whilst wider use of this additive will decrease reflecting the REACH restriction, medical devices were excluded from the scope of the restriction. In medical devices it can comprise between 20% and 40% of the final polymer weight. DEHP can leach into, e.g., solutions. The rate can depend on temperature, storage time, solution flow rate as well as the amount of DEHP in the PVC product. Studies on infants in neonatal intensive care undergoing extensive, long-term and/or repetitive treatment can have much higher exposure than the average adult (Perinatol, 2011). Note the effects of the 2018 REACH restriction on several phthalates, including DEHP, in consumer articles could be expected to reduce risk in this sector, depending on the precise phthalate plasticisers used (European Chemicals Agency, n.d.).

In terms of the alternative materials (or plasticisers) identified, a 2011 study reviewed several alternatives to PVC in medical uses. It noted that although there are toxicity data on the raw materials used in the production of EVA, neither carcinogenicity nor other relevant health data relating to EVA was identified. Similarly, no published data on health effects of PUR was identified. In terms of PP, no evidence of carcinogenicity *in vitro* was identified. But respiratory risk was identified and data on absorption, distribution or excretion of propylene in humans was not available, nor was data on long-term effects. PE and ethylene have low toxicity and has been widely used in medical settings. A further technical study cited in the review examined leaching of DEHP from several PVC co-extruded lines laminated with either PUR or PE to prevent DEHP leaching, plus non-PVC perfusion lines, and found that "co-extruded PVC/PU and PVC/PE lines leached comparable levels of DEHP to pure PVC lines when used with a lipid emulsion. Pure PE perfusion lines, however, leached only a negligible fraction of DEHP (0.23 µg m^3)”. This suggests replacing PVC and PVC co-extruded lines with pure PU or PE alternatives may reduce risk (Perinatol, 2011).

In terms of DINCH, manufacturers state tests for reproductive and developmental toxicity as well as for repeated use does not show any adverse effects (Hexamoll DINCH, n.d.). In 2006, the European Food Safety Authority approved DINCH for a wide variety of food contact applications. A 2013 review noted that toxicological studies in animals have shown no evidence of developmental or reproductive toxicity or endocrine disruptive properties, but at high doses, thyroid hyperplasia and signs of renal toxicity were reported in rats (Silva, et al., 2013). In terms of silicones, as noted in Table 7-33 the presence of D4 and D5 in polymers as manufacturing impurities gives rise to concern based on PBT/VbVP properties.

In terms of recycling, EVA can be recycled although only some municipalities accept it (The Bow Index, n.d.). Some data sources indicate recycling rates of EVA are extremely low (Green Max Intco Recycling, n.d.). PU is more widely recycled, via chemical or mechanical methods as are various forms of PE and PP (Green Max Intoco Recycling, n.d.). Nitrile gloves can be recycled provided they are not contaminated from use. It is understood via stakeholder consultation that the majority of medical waste is incinerated (Unigloves, n.d.) (see Section 6.3.4 for further details on hospital waste incineration). Silicone can be recycled but may need specialist facilities (EarthHero, 2018).

7.14.6 Availability

According to the ECHA C&L inventory PMPC is used in small quantities only (less than 1 tonne) (European Chemicals Agency, n.d.), although PE is widely used and available in significant quantities. The global market for PP was estimated at between 45 million tonnes (2016) and up to 65 million tonnes by 2020 (Creative Mechanisms, 2016). Around 28,000 tonnes of formulated silicone products are sold into healthcare applications globally, some 11,300 tonnes into Europe. The global market for all formulated silicone applications is over 2 million tonnes (2013 data) (American Chemistry Council, n.d.). CE-marked Medical Devices based on DINCH plasticisers are available in Europe, Asia and North America (Hexamoll DINCH, n.d.). As of 2014 there was a manufacturing capacity from one manufacturer of over 200,000 tonnes per year (Hexamoll DINCH, n.d.). No quantitative data has been identified on typical timescales for phase out from the hospitals/health services described above. The main competing applications for silicone are personal care products, electronics, the aviation sector, construction, kitchenware, paints and coatings, and sporting goods and apparel (ChemicalSafetyFacts, n.d.).

7.15 Conclusion on suitability and availability for alternatives to PVC in medical applications

The evidence suggests that technical feasible alternatives to PVC are available for use in a range of medical tubing uses as well as for gloves. Stakeholders have indicated their suitability for use in blood bags is more problematic at present. However, alternatives are in development which appear to offer acceptable functionality for this use and other examples of successful phase out have been identified. These alternatives are typically more expensive, per tonne, than PVC. They include both alternative materials as well as non-phthalate plasticisers.

7.16 Overall conclusions on suitability and availability of possible alternatives for PVC

Overall conclusions on the suitability and availability of all possible alternatives considered in the Analysis of Alternatives are below. These are based on their technical and economic feasibility, capacity for reducing the overall risk and their availability.

Table 7-36: Analysis of Alternatives to PVC - Overview

Sector	Application(s)	Alternatives Assessed	Technical Feasibility	Economic Feasibility	Comparative Hazard and Risk	Availability	Uncertainties and quality of information
Construction	Pipe and Pipe Fittings (Drinking water, waste and rainwater and e.g. irrigation and natural gas conveyance)	HDPE, MDPE, LDPE, PP, Glass fibre reinforced PP composite, PE-X, PB, ABS, Polyamide/Nylon, Galvanised Steel, Stainless Steel, Copper, Yellow Brass, Aluminium, Cast Iron, Ductile Iron, Concrete and vitrified clay.	Requirements differ significantly by downstream use. Overall, durable alternatives, with water, chemical and UV resistance have been identified in these applications which are currently used.	There are economically viable alternatives to PVC pipes. Although the average costs per metre are greater, some alternatives offer comparable durability, break rates, weight and flexibility for repair.	Generally long lifetimes, but widely varying depending on source. Transition to alternative plastics may result in leachate of harmful substances and increases in finite raw material and energy use where metal alternatives are used. Uncertainty over recyclability of non-plastic alternatives.	Yes, although increases in production and/or import volumes may be necessary.	Uncertainties remain on the precise net effect on hazard and risk given the range of uses. Limited data on risk of PVC and alternatives leachate into water and soil. Limited data on costs associated with ease of repair/in situ repair.
Electronics	Cables	PE, CPE, PP, PUR, TPE, mPPE, FEP, ETFE, PTFE, PFA, PEX, CP, EPDM, EVA, Silicone Rubber.	Plastic or synthetic rubber alternatives are available and in use, but each perform differently.	Material costs vary; some are comparable whilst others are significantly more expensive. Costs for articles (I,e cables) reflect this, although the differences likely to be lower.	**Some alternatives are fluoropolymers and may be affected by ongoing regulatory action, others contain harmful substances posing recycling challenges. Additives still likely to be required in several cases.**	Alternatives widely available. Increases in production and/or import volumes may be necessary	Lack of reliable economic and cost information on every alternative and it is recognised that specific performance requirements differ significantly within the sector.
Construction	Window Frames	Timber, aluminium, fibreglass and clad-wood	Alternatives currently commercially available and commonly used, each with specific advantages and disadvantages.	Whilst unit costs are typically greater, durability is comparable or longer.	Alternatives generally present lower health and environmental risks, but are not free of concerns and greater use will involve trade-offs associated with energy use and	Alternatives are commercially available, although may result in an increased consumption of natural resources (e.g. wood).	Information generally of good quality.

Sector	Application(s)	Alternatives Assessed	Technical Feasibility	Economic Feasibility	Comparative Hazard and Risk	Availability	Uncertainties and quality of information
					natural resources consumption (wood).		
Construction	Flooring	Wood, laminate, linoleum, tiles, stone, rubber, carpet and cork and bio based/recycled plastic.	Various materials are in use; each has specific advantages and disadvantages for specific uses.	Alternatives are commercially available and whilst unit costs and lifetimes uses differ, in several cases they are comparable to PVC.	An overall reduction is likely, although this is likely to differ between alternatives, some of which have potential for adverse effects.	Yes, although increases in production and/or import volumes may be necessary.	Uncertainties remain on the precise net effect on hazard and risk given the range of products, treatments and coatings.
Packaging	Containers (Food packs, blister packs	PET, PE (HDPE, LDPE, LLDPE), PP, PS, PA, COC/PO, EVOH, PVDC, Bioplastics, Aluminium, Paper, Ceramics and Glass.	Alternatives plastics, materials and/or bioplastics are technically feasible for a range of applications. Non-plastics can be used in some circumstances.	Some non-plastics alternatives are more expensive, either to transport or manufacture, and may require product redesign, but various alternatives available and in use.	The risks differ depending on the material used and greater use involves trade-off. Plastic alternatives contain leachable additives and films are problematic to recycle..	Alternatives are commercially available, although novel bioplastics available in smaller quantities at present.	Uncertainties remain on the precise net effect on hazard and risk given alternative plastics.
Toys	Inflatable toys, rafts, balls, pools, rubber boats)	TPE, EVA, Polyolefins.	Alternatives available, in use and comparable to PVC.	Commercially available and in use. Whilst there is limited information on costs, the alternatives generally appear to be more expensive, but more durable.	Evidence suggests that alternatives do not use phthalates and additives are typically used in lower concentrations (% w/w), with wider recycling potential. Specific concerns with some alternatives (e.g. chlorine use). The 2018 restriction would be expected to reduce risks from specific phthalate plasticisers.	Commercially available in large volumes. Increases in production and/or import volumes may be necessary.	**Poor, limited detail on alternative materials. Materials used in many products is not clear.**
Automotive	Dashboard and seat covers including artificial leather	Leather Cloth or natural fibres; ABS, PP, TPE-O, ABS/PC blend.	Alternatives are in use and achieve technical functionality.	Alternatives are commercially available.	An overall reduction is likely, but significant concerns with ABS and PC, for which demand may increase. The 2018	Yes, although increases in production and/or import volumes may be necessary.	Good level of detail and manufacturers are developing novel alternatives from a range of materials.

Sector	Application(s)	Alternatives Assessed	Technical Feasibility	Economic Feasibility	Comparative Hazard and Risk	Availability	Uncertainties and quality of information
					restriction would be expected to reduce risks from specific phthalate plasticisers.		
Household	Clothing (Wellingtons, soles of shoes)	Natural rubber PUR, TPU, EVA, Leather.	Alternatives are in use and can achieve technical functionality.	Alternatives are commercially available; some likely to result in increase in costs.	Issues differ by material used. Presence of additives in alternative plastics, although bio-based/recycled materials are available.	Yes, although increases in production and/or import volumes may be necessary.	Generally good information available, including on prices.
Medical	Blood bags, medical devices, gloves	PE/PMPC, PP, PU, (DINCH (alternative plasticiser), silicone rubber.	Alternative materials (and plasticisers) exist and examples of successful PVC phased out, with the exception of blood bags, but alternatives in development.	Economically viable alternatives are commercially available and currently used.	Reduction in risk expected to human health, drive by use of phthalate plasticizer in PVC. Silicone polymers may contain D4/D5. Most medical waste is incinerated. The 2018 restriction would be expected to reduce risks from specific phthalate plasticisers.	Most alternatives widely available, although some are currently used in low volumes.	Generally good information, much more detail available on human health risks, given the nature of use.

8. PHASE OUT SCENARIOS AND SOCIO-ECONOMIC EFFECTS

Key Messages – Phase Out Scenarios and Socio-Economic Effects

Approach and Phase-Out Options

This section considers potential options to accelerate or mandate an EU-wide phase-out of PVC. The key socio-economic effects of a potential phase-out are qualitatively assessed, drawing on work in previous chapters. This examines available data on the human health and environmental, economic and wider economic and social impacts. Socio-economic impacts were explored in more detail for three applications (flooring, medical and automotive uses) based on data availability.

Several options were outlined, which could be undertaken alongside an industry data collection exercise to fill current data gaps and uncertainties. This could generate more detailed application-specific data on additives use, potential speed of substitution, application-specific exposure and risks, as well as costs. The options include taking no further action; agreeing a voluntary phase out programme with industry, to regulatory action. These options include – but are not limited to – an ordinary legislative procedure; further REACH restrictions on specific additives used in PVC or on PVC itself, based on specific use(s) or all uses. This would require proof that the risk from the specific use(s) of PVC is not "adequately controlled". Several options depend on current reforms being considered to the operation of REACH, for example, the potential application of the "essential use concept". The analysis suggests that further data would be useful targeted risk-based approach.

Overall Messages

- **Economic impacts:** Potential alternatives comprise either alternative materials, often those used before PVC or that have retained some market share as PVC use has expanded; a variety of alternative plastics; more novel substances such as phthalate free plasticisers or bioplastics. In some specific cases it may be possible to re-design the article to accommodate alternative materials. Overall, there are two main options in the event of a phase out. It is expected, however, that a restriction in any one application would, in practice, result in a combination of the two.
- **A)** Replace PVC with alternative materials or substances if a feasible alternative (or mix of alternatives) is available or identified after further development. Investment in R&D, alongside batch/sample production for testing to ensure product functionality and hence acceptability by the downstream users and final consumers in specific applications may be needed. Timescales for reformulation are not specific and given the vast range of applications it has not been possible to identify the total number of product lines affected, but the number is likely to be substantial. Adopting alternatives requires several steps. The technical challenges and performance implications and costs would differ across applications.
- **B)** If the above is not technically or economically feasible, or the companies are unable to make investment decisions based on their current knowledge of alternatives, then the affected product lines, revenues and employment in the EU will cease. Under this option exports outside the EU would also cease.

- A key issue are distributional effects. In many cases, completely different upstream supply chains will be affected by a potential phase out. Decreases in demand, likely associated with significant effects to that sector, offset by increases in the manufacture and sale of alternatives. Manufacturers as well as wholesalers and distributors, who manufacture or supply several different materials, would be less affected under this scenario but would likely incur increases in prices and costs.
- **Social impacts:** Loss of markets that cannot be replaced by alternatives would have implications for employment. Net effects in downstream sectors are not clear, but significant effects are expected in the PVC manufacturing sector depending on their portfolios and the scope and speed of action.
- **Wider economic impacts:** Current exports of virgin PVC and PVC-containing products would likely be lost, depending on the scope of action. The scale of required reformulation efforts would likely preclude other R&D activities.

8.1 Introduction

This section first considers different potential options to accelerate or mandate an EU wide phase-out of PVC and/or of PVC additives. The key economic, social and wider economic effects of a potential phase out are then considered. This section draws on various earlier work packages. The scope and timing of any potential action has not yet been determined and no recommendations are made.

As discussed in Section 2.3, the externalities to human health and the environment from PVC and its additives have raised concern for many years. Various human health and environmental risks can arise from PVC as well as the additives used in it, when manufactured, used and disposed of inside the EU, as well as globally via exported PVC, articles and waste. Contaminants in PVC can affect both the potential uses of recycled or reused PVC in a circular economy, alongside ongoing potential exposure via secondary materials. In addition, there are concerns associated with landfilling and incineration of PVC waste, as discussed in Section 6.4.1.2 and Section 6.3.1.2, respectively.

Potential action would be driven by several different policy objectives at European level, including the Plastics Strategy (European Commission 2018), the European Green Deal (European Commission 2019); the Circular economy action plan (European Commission n.d.); the Chemicals strategy for sustainability for a toxic-free environment (CCS) (European Commission 2020) as well as protection of human health and the environment under Regulation (EC) No 1907/2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH).

The European Commission has taken action on some PVC additives to date. This includes a proposed restriction on lead compounds in PVC (European Chemicals Agency 2017) (see Annex 1.1); and a restriction adopted in 2018 on placing on the market on articles containing four phthalates for indoor environments and direct exposure (European Chemicals Agency 2018). See legal act here (European Commission 2018). Action under Restriction of Hazardous Substances in Electrical and Electronic Equipment (RoHS) have also restricted uses of heavy metals and several phthalates in these sectors – of particular relevance to the discussion on cables (European Commission 2011).

Further, action on some of the alternative materials that may be used in greater quantities in the event of a PVC phase out are also being considered. This includes a broad restriction on various uses of PFAS being prepared by the Competent Authorities of Germany, the Netherlands, Norway, Sweden and Denmark (RIVM n.d.).

8.2 Potential options

A phase-out of PVC could avoid human health and environmental hazards associated with PVC (see Section 2.3) by reducing or eliminating sources of the risk and accelerating the development of alternatives. There are several options, of differing scale and scope, from do nothing to a full restriction of all uses. Several potential scenarios are outlined below, including a business as usual (BAU) scenario with no further action, as well as potential REACH restrictions, either targeting specific risks and/or specific applications - or on all applications. This is not an exhaustive list and do not imply any one option is being considered, but intended to illustrate a range of potential options. The timing and scope of action will significantly influence the socio-economic effects. These are outlined below in reverse order of ambition in Figure 44, and discussed further below.

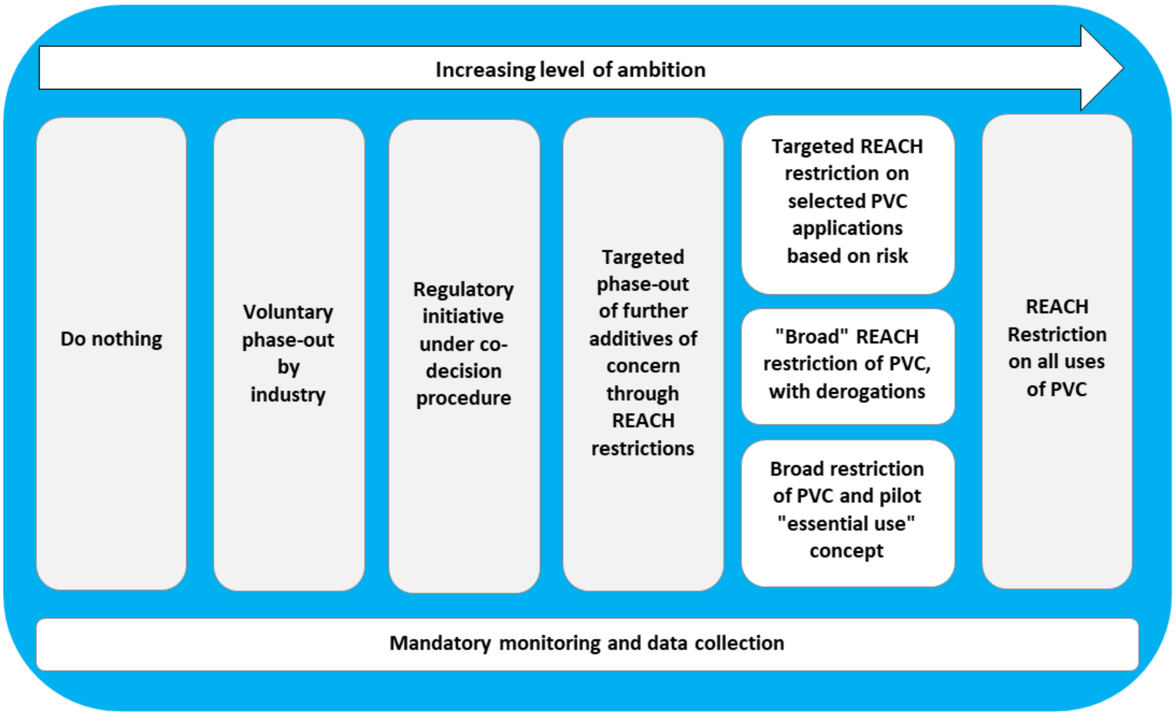

Figure 44. Illustrative overview of potential actions

- **Do nothing/ business as usual (BAU):** in case of no further regulatory action, use of PVC would be expected to continue to evolve with market demand. The evolution of use would likely differ with each application. Overall, the market assessment in Section 3.5 indicates that in the absence of further action, the volumes of PVC produced in Europe, would either stabilise at current levels or continue to decline. The volumes placed on the market in Europe would likely continue to increase; the share imported from outside the EU, particularly from China could be expected to grow. This is consistent with commercial market forecasts, which predicted continued growth in the global PVC market, before the economic disruptions associated with COVID-19.

- At the same time, other actions, such as the restriction on DEHP, alongside wider technological and environmental improvements, potential developments in incineration technologies, may serve to reduce the risks of PVC whilst retaining its use. Close monitoring of the effects of the 2018 restriction on phthalates would be important to observe, including the extent to which this results in the development of alternative plasticisers and/or a faster and wider PVC phase out. Rates of recycling under the VinylPlus framework have continued to increase, particularly PVC window frames (see Section 4.2.3).

- **Voluntary phase out:** A further step would involve voluntary agreement with industry, with defined targets and published results. The agreement could cover, for example, recycling targets, removal of further additives and/or phase out of specific applications. Pooling of R&D investment in faster development of alternatives may also be discussed. Other options along these lines may include use of ecolabelling with further information either on the physical package and/or digitally. The data on where European PVC applications are increasing or declining is undermined by differences in categorisation, but the evidence indicates use in packaging is declining in Europe, for example.

- •The ordinary legislative procedure (formerly the co-decision procedure (referred to here as "**co-decision**") is the general rule for adopting legislation at European Union level involving the European Parliament and the Council of the European Union with equal rights and obligations. A legislative proposal is made, via the Commission, with up to three readings by Council and Parliament. The co-legislators have to agree to adopt and approve identical text.

- **Targeted phase out of further additives of concern:** One of the concerns with PVC relates to the additives, particularly plasticisers. Further REACH restriction(s) on these, rather than on PVC itself could be considered. This could focus on: a wider number of additives of concern in PVC and their grouping; a lower concentration permitted in some or all articles, and/or with fewer derogations in existing restrictions (such as that on DEHP). This would be a continuation on the approach adopted to date, including cadmium in PVC and phthalates in consumer articles. It would not remove all concerns associated with PVC manufacture and disposal (e.g. incineration). It could incentivise the faster development of alternative additives (e.g. plasticisers) of lower toxicological concern. Some applications of PVC may become technically infeasible where particularly high volumes of additives are needed in the PVC article and/or when alternatives are not available or could be developed.

The remaining options would include restrictions on PVC and/or further action on the additives used in it. This would require proof that the risk posed to Human Health or the Environment is "not adequately controlled"[201], such as:

- **Targeted risk-based Restriction and transition to alternatives (R1):** A REACH restriction on selected applications of PVC focussing on particular sources of environmental and human health concern and where alternatives are available. This would benefit from additional data being created, via a mandatory monitoring programme, discussed below, or on the basis of information submitted during the public consultation stages of the restriction development. Criteria listed in section 8.4 could be used to do this.

- **Broad REACH restriction with derogations (R2):** This would involve a restriction on placing PVC on the market in the EU and would include PVC imports. Derogations (time-limited or not) for specific applications of virgin and recycled PVC could be included either proposed by the dossier submitter or on the basis of information submitted by stakeholders in the public consultations. Subject to proof that the risk was not adequately controlled, this places the burden of proof in providing further information to justify those derogations, based on risk and socio-economic aspects on industry.

- **Broad restriction, piloting use of the "essential use concept" (R3):** This would be a similar approach to above, but uses a different set of criteria to justify continued use. – Subject to the results of ongoing work by the European Commission on the potential application the "essential use" concept - such an approach could be piloted or adopted for PVC. This concept focusses on ending use of harmful substances specifically where their use is deemed non-essential for society or when safer alternatives exist. Cousins et al. (2019) applied the concept to PFAS and proposed three criteria, based on the Montreal Protocol:

[201] See Point 29 in the REACH Regulation (https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32006R1907&from=en

- **Non-essential:** "Uses that are not essential for health and safety, and the functioning of society. The use of substances is driven primarily by market opportunity". Examples here may include: toys, clothing or packaging.
- **Substitutable:** "Uses that have come to be regarded as essential because they perform important functions, but where alternatives to the substances have now been developed that have equivalent functionality and adequate performance, which makes those uses of the substances no longer essential". Examples here may include: pipes, window frames, flooring, automotive interiors and some medical applications.
- **Essential:** "Uses considered essential because they are necessary for health or safety or other highly important purposes and for which alternatives are not yet established". This status was anticipated to be regularly reviewed. Examples here may include: specific medical applications, such as blood bags.

- **Total phase out and transition to alternative substances (R4)**: A REACH restriction on all applications of PVC. This would be a higher level of ambition.

- **Mandatory monitoring phase to effectively target phase out based on risk:** Any one of the approaches outlined above could be undertaken alongside a data collection exercise with industry, Member States, and other relevant stakeholders. This could be initiated in advance of/alongside any restriction proposal which will take some time to be develop, be consulted on and to implement. This would focus on generating more detailed application specific data on additives used, on more detailed information on the speed of substitution, application specific exposure risk, EU imports by application and supply chain stage etc. A restriction could be initiated with a broad scope, potentially including derogations at a later stage if sufficient evidence became available that the risks were adequately controlled for specific applications, or vice versa.

8.3 The need for action

A potential phase out of PVC may affect a large number of downstream sectors and involve several trade-offs. The supply chains affected are global. It is not clear which articles that are placed on the EU market incorporate PVC manufactured from outside the EU, although this comprises an increasing share. There are differing levels of detail on specific applications of PVC, on recycling rates, on the specific additives used in different products and downstream sectors. The advantages and disadvantages of taking action are below.

Table 8-1: Pros and cons of action on PVC

Advantages	Disadvantages
• Address a material which has given rise to various concerns, including but not limited to additives of concern and which is associated with adverse human and environmental effects at all lifecycle stages, including end of life; • Take action on a material where an increasing share of the volumes placed on the EU market is manufactured outside it, under lower environmental and regulatory standards, particularly in China; • Depending on the scope of action, send a clear market signal to manufacturers of PVC , manufacturers of alternatives to increase production and expand product portfolios, and to downstream users and consumers in the EU and beyond to seek alternatives to PVC; • Support circular economy goals via the removal of contaminants in secondary materials and	• The costs incurred to upstream manufacturers, downstream users and consumers alongside some technical drawbacks in certain applications; • Remaining uncertainties including on specific volumes of use, differences in hazards and risks posed by PVC use in different applications, make targeted action challenging; • Risk of regrettable substitution where, for example, PVC is replaced by increased use of other problematic materials, including plastics and/or additives, mitigating the overall benefits. • Limited empirical data to date on the ex-poste effects of relatively recent action on several phthalates known to be used in PVC. • Uncertainties in current recycling technologies and techniques – which may improve current

Advantages	Disadvantages
aiding recycling of post consumer PVC, at least in some uses; • Address a source of EU and non-EU plastics waste; and • Avoid ongoing health and environmental costs of inaction from use and disposal of PVC.	environmental and economic feasibility of PVC recycling (see Section 5.2). • A decrease in the range of options available to consumers in certain applications, and limited options for blood bags.

8.4 Prioritisation criteria for risk-based REACH restriction

To mitigate disadvantages and focus action on the areas of greatest environmental and human health benefits, several prioritisation criteria could be considered. These draw on the previous chapters in the current study. The existing data required for these prioritisation criteria are of mixed quality. The collection of additional mandatory information – potentially alongside the initiation of regulatory action under REACH, which may take several years - could be considered so as to better target the scope of action.

A PVC phase out scenario could use the following prioritisation criteria to focus on PVC placed on the EU market:

- In the greatest volumes of use, with the highest PVC waste volumes and volumes to landfill/incineration.
- Where feasible alternatives have been identified. The different timescales for phase out/review periods could reflect this.
- On PVC risks that have not or cannot be addressed by other means, for example via action on specific additives such as DEHP.
- Applications with the highest risk of PVC service life exposure (identified through a combination of exposure pathway, duration/frequency of exposure and product lifetime)
- Applications where current recycling rates are lower, and/or where future recycling potential is judged to be lower, where the CE potential of the recycled PVC is greater and where risk from co-mingling and cross contamination during recycling is highest.

Based on the earlier work packages, the quality of existing data for this prioritisation is assessed below.

Table 8-2: PVC Phase out prioritization assessment criteria

	Notes on prioritisation	Data source and data quality?
Volumes of use	A phase out should focus on those applications with the greatest volume of use (taking into account other factors such as service life exposure, additives, CE potential, waste and recycling).	Good data exist on volumes of virgin PVC per sector, although the categories are relatively broad. Differences in data categorisation and definitions used make time series comparison of changes in volumes of use within specific applications difficult. Global data lack granularity and comprehensive quantitative data on additives is not available.
Availability, suitability and costs of alternative	A phase out should focus on those applications where feasible alternatives have been identified and timescales for phase out could reflect this.	Detailed assessments have been undertaken, on alternatives in a selection of PVC applications, as a sample (see Sections 7.4 to 7.14).
Service life exposure *(pathway, duration/frequency, product lifetime)*	A phase out should focus on applications with the highest risk of PVC service life exposure (identified through a combination of exposure pathway, duration/frequency of exposure and product lifetime)	It has not been possible to generate evidence on potential exposure pathway, duration, frequency and number of people exposed / environmental compartments impacted. Neither has it been possible to differentiate risk in specific applications (see chapter 2).
Current PVC recycling rates	A phase out should focus on PVC applications where current	Data on recycling rates is limited in detail and does not differentiate between pre- and post-

	Notes on prioritisation	Data source and data quality?
	recycling rates are lowest (alongside other factors, discussed below).	consumer waste and different product groups/applications (see Section 4.2).
Recyclability and potential in a CE	A phase out should focus on PVC applications where future recycling potential is lowest and where the CE potential of the recycled PVC is lowest (alongside other factors, discussed below).	It has not been possible to differentiate specific applications of PVC with low future recycling potential, and/or low CE potential, although some forecasts are presented in Section 5.5 based on a number of assumptions. This also reflects future technological development in recycling methods, including chemicals recycling.
Risk of co-mingling and cross contamination	A phase out should focus on PVC applications where risk from co-mingling and cross contamination during recycling is highest) (alongside other factors, discussed below).	Packaging waste streams are known to have significant co-mingling (see Section 5.1.2.1), but otherwise it has not been possible to identify specific PVC applications (or where recycled PVC inputs) have high co-mingling/cross contamination potential.
Waste – % to landfill and incineration	A phase out should focus on those applications with the highest PVC waste volumes and volumes to landfill/incineration	The disposal routes for PVC waste arising from automotive, WEEE, packaging and medical applications have been quantitatively scoped (see Section 6.2.2); other waste streams, including construction and demolition waste, could not be quantified. It is necessary to note that the scoping is based on assumptions and significant underlying uncertainties. As such, It has not been possible to reliably differentiate specific PVC applications by disposal route.
Socio-economic costs	A phase out should focus on those applications where the economic, social and wider costs of substitution are lowest.	Whilst volume and average price data provides an estimate of market segments, it has not been possible to differentiate the various PVC products in terms of typical product margins. It is also not know how many product lines may be affected in each of the applications/sectors, or costs of reformulation for example.

8.5 Socio-economic effects

This section examines the main economic impacts associated with possible restriction scenarios. Social, wider economic and distributional effects are also assessed. For simplicity, we focus on the expected effects associated with broad restrictions (R1-4), compared to the effects of taking no action (BAU). The analysis in this section relates to all of the PVC applications covered in the report. Given the range of applications, of possible alternatives, a quantitative assessment of the socio-economic costs and benefits of this action has not been possible.

8.5.1 Economic Impacts

The overall conclusion from the analysis of alternatives in the previous sections suggests that potential alternatives exist in the majority, if not all, PVC applications that have been assessed in detail. These alternatives comprise:

- alternative materials, often those used before PVC or that have retained some market share as PVC use has expanded;
- a variety of alternative plastics;
- Novel products such as phthalate free plasticisers or bioplastics.
- In some specific cases it may be possible to re-design the article to accommodate alternative materials.

Current market shares of PVC differ by application. Some trend patterns have been established via consultation; PVC use in bottles has decreased in the EU and the overall share of total PVC used in construction has increased over the past decade, for example. In some applications, such as flooring, non-PVC alternatives are actively marketed to the end consumer as sustainable alternatives. But there is limited detailed data on trends in individual applications.

Where substitution has begun and/or where PVC retains a decreasing market share, it can be assumed that some of the more challenging aspects of new product development (reformulation, testing, downstream user and consumer acceptance, marketing and regulatory approvals, where necessary) would have been overcome, for at least some specific uses in that application. Information gained during the conducted Focus Group on alternatives– again for a specific flooring application – indicates that the process of product development has taken in the order of three years, between the initial decision and a commercially available product. This made allowance for iterations of the formulation after early tests. In others, cost increases that were experienced when alternatives to PVC were adopted reduced somewhat thereafter as production volumes rose and greater economies of scale were achieved.

A key issue are distributional effects. In many cases, completely different upstream supply chains will be affected by a phase out of PVC. Demand for PVC decreasing, likely associated with significant effects to that sector, offset by increases in the manufacture and sale of alternatives (wood, linoleum, plastics etc). Manufacturers as well as wholesalers and distributors, who manufacture or supply several different materials, would be less affected under this scenario but would likely incur increases in prices.

8.5.2 Response scenarios

Overall, there are two main options in the event of a phase out. These would apply in all applications subject to a restriction. It has not been possible to place monetary estimates on the net economic costs and benefits, for the reasons stated in Table 8-2. It is expected, however, that a restriction in any one application would, in practice, result in a combination of options 1 and 2 below.

1. **Replace PVC with alternative materials or substances if a feasible alternative (or mix of alternatives) is available or identified after further development.**
    - In some cases, this will require significant shifts in the supply chain from one material to another (e.g. from PVC to aluminium or wood). In others, it may require use of alternative plastics and/or the development/adjustment of new formulations. For the latter, investment in R&D, alongside batch/sample production for testing to ensure product functionality and hence acceptability by the downstream users and final consumers in specific applications may be needed.
    - Timescales for reformulation are not specific and given the vast range of applications it has not been possible to identify the total number of product lines affected, but the number is likely to be substantial.
    - This would involve losses in revenue and earnings to PVC manufacturers both in the EU and beyond who currently supply the EU market. These could be offset by increased sales of alternative materials by those same firms, depending on the product portfolio.

    - Adopting alternatives requires several steps. The key stages required would be similar, but the challenges and technical implications and costs would differ across applications.

- Purchasing alternative raw materials, taking account of different prices and any differences in volumes required to achieve the same / similar function.
- Several of the alternatives evaluated consist of natural substances (albeit with various treatments), such as wood, rubber, leather, cork. In the many cases alternative plastics are likely to be used. The alternatives identified are generally commercially available and already in use.
- Several of the alternatives are more expensive on a unit basis. Sunk costs in past investments in specific machinery, capital costs associated with changes in processing/machinery and operational cost changes associated with any loss of production efficiency may be incurred.
- The number of product lines that may need to be tested, reformulated, re-marketed, submitted for regulatory approval and produced may pose substantial challenges; financially and in terms of staff, R&D and production capacity for some parts of affected supply chains. This will likely affect manufacturers' judgements on whether to continue to invest in some product lines.
- The above would pose specific challenges for SMEs.

2. **If the above is not technically or economically feasible, or the companies are unable to make investment decisions based on their current knowledge of alternatives, then the affected product lines, revenues and employment in the EU will cease**. Under this option exports outside the EU would also cease.

8.5.2.1 Economic effects in downstream PVC applications

For each of the PVC applications considered in chapter 7, the main source of economic costs are described below. Three applications (flooring, medical and automotive applications) are considered in greater detail, with case studies. They were selected on the basis of one or more of the prioritisation criteria outlined in Section 8.4.

8.5.2.2 Flooring

<u>Business as usual</u>
Demand for new and replacement flooring will be influenced by the rate of construction development, both residential and commercial. The sector was severely impacted in 2020 particularly as a result of the COVID-19 crisis, but recent forecasts expected the market to rebound, with 4-5% growth in 2021 (Business Wire 2021). Supply constraints are currently a concern (Condon 2021). Global market forecasts expected PVC flooring market to grow strongly to 2023 (Dawande 2018). Consultation has indicated that at least some new alternative formulations to PVC are being developed, with sales of these increasing alongside continued availability of established alternative materials such as wood, stone or laminate. It is likely that PVC will remain a key flooring material.

The following sections discuss the health and environmental, and economic impacts on the flooring industry associated with any phase-out scenario.

<u>Health and environmental impacts</u>
Increased demand for alternative materials in the event of a phase out of PVC poses some concerns, but these differ with the materials concerned (see Section 7.7.5). These include deforestation, the presence of paints and solvents, pigments or drying agents, and formaldehyde in certain laminate flooring products manufactured in Asia. Others' such as linoleum, stone, ceramic or latex, comprise natural ingredients.

As discussed in Section 7.7.5, additives of concern in flooring including phthalates can be released into indoor dust. Transition to alternative materials free of additives of concern could lead to reduced human health impacts where people come into contact with flooring.

Economic impacts

A range of alternative materials are used currently; several are marketed as sustainable alternatives to PVC. The unit costs differ significantly depending on material, but there are a wide range of alternatives at different price points as detailed in Section 7.7.4. The key driver of costs are the durability and unit costs for flooring. Consultation has indicated some concerns with alternatives to PVC required for applications with very high footfall and where high standards of cleanliness are required, such as in hospitals. But it is noted that alternative PVC free formulations are commercially available that have been developed for these applications, including for commercial and healthcare settings. Those incurring additional costs would include a range of businesses, commercial property developers and homeowners.

Case Study

Case Study: Experiences of developing non-PVC polymer product lines in the flooring sector
A flooring manufacturer described their experiences in branching into production of non-PVC polymer flooring. As a major manufacturer of laminate flooring, the business observed increased customer demand for polymer-based flooring and decided to expand into this area. This decision followed initial small-scale milling of PVC sheets for an original equipment manufacturer (OEM) customer, but the business opted not to continue. It was concluded then that alternatives exist with less of a damaging environmental impact. The business also faced difficulties in disposing of PVC milling dust waste and were unable to recycle it and had little choice but to ship it back to China. Polyolefins were identified as a suitable alternative to PVC in flooring, with good scope for recycling and products in a circular economy. Polypropylene (PP) in particular does not require plasticisers, unlike PVC. In addition to its relative environmental performance, adopting PP product lines would allow the business to differentiate itself in the polymer-based flooring market. The company invested several tens of million Euro in developing a polyolefin flooring production capacity for several million m2 of product. A second smaller investment was made, reflecting the company's confidence in the future development of the market. The first product lines were commercially available around three years after the initial decision was taken to produce polyolefin-based flooring products. The business experienced some limited concern from some customers initially, but suppliers wanted to have a non-PVC polymer option in their product portfolios to provide customers with greater choice. It is now widely accepted in product portfolios and the company reported that all existing customers (in the DIY business in Germany) have ordered their non-PVC product. The business reports selling increasing volumes of their PVC-free product each year and distributing in more and more countries worldwide. The products are available in a range of hard and soft options. The company did note a specific application where non-PVC products would require further development and are not available today. However, this is a small part of the market and this specific product is available from an alternative manufacturer. The company judged it would be able to develop such a product in PP if required. The non-PVC product is slightly more expensive. However, as production volumes have increased, costs have come down. The greatest costs faced were in adapting manufacturing processes as PVC is generally much easier to process than polyolefins. The advantage of formulating a polyolefin-based product has been the ease of recycling the product, and the ease of using recycled polyolefins as a raw material. The business has reported that they are able to regrind all of their waste and off-cuts with virtually no material loss. At present, approx. 75% of the polymer raw material content is recycled, and there is potential for this to increase in future. The business has established a partnership with a German recycling company to secure a reliable source of raw material. The decision to produce polyolefin flooring necessitated the acquisition of a new production lab and required new employees with the relevant knowledge and specialism. While there was a period of 'optimisation' to improve the technical performance of the PVC-free flooring, the business claimed that today the products are comparable to PVC flooring in terms of technical functionality.

8.5.2.3 Automotive (dashboard and seat coverings)

Business as usual

A number of automobile manufacturers have made voluntary commitments to phase out PVC use in internal automobile components or have already discontinued its use (Greenpeace International 2003, ENDS Report n.d.). However, market analysis indicates that PVC is likely to remain a dominant material in automotive interior components (Bregar, 2019), and will experience a resurgence in the future, driven by continued expansion of the Chinese automotive sector and the cost/performance balance of PVC compounds (ITB Group, n.d.). Consequently, despite actions by individual automotive manufacturers to limit use of PVC in vehicle interiors, it is likely that PVC will remain a key material used for this purpose in the absence of any restrictions on PVC.

The following sections set out the health and environmental, and economic impacts of PVC phase-out scenarios on the interior automotive component sector.

Health and environmental impacts

The use of several alternatives is likely to increase in the event of a phase out of PVC. The alternatives however are not free from risk. Two involve use of ABS, either on its own or as part of a blend with Polycarbonate (PC). The latter is commonly manufactured with Bisphenol A. However, PC is not likely be suitable for many of the flexible articles where PVC has been used. Some of the raw materials used in ABS and/or thermoplastic olefins pose several risks. The balance of evidence suggests that the other potential materials, beyond those above, are of lower risk. Recycling rates of alternatives to PVC are increasing, and there are a variety of technical options for recycling PP, ABS and thermoplastic olefins (see section 7.13.5). A phase-out of PVC and transition to alternative materials could therefore improve the recyclability of the affected components.

Economic impacts

The key economic impacts in the automotive sector are the same as those described in Section 8.5.1. The substitution route may involve either product redesign so that specific coatings/coverings are no longer needed (i.e. single material dashboard units) or the use of alternative coverings (such as leather seats).

A study into the costs of different vehicle components revealed that the vehicle body (including interior and exterior parts, electronics and systems) accounts for 11-20% of the cost of a car (Fries, et al., 2017); interior components will in turn account for a fraction of this portion. Substituting internal PVC components for potentially costlier PVC-free alternatives is therefore unlikely to result in significant additional costs to the price of an individual car, although costs may be significant to specific part suppliers.

A number of car manufacturers have already taken voluntary steps to eliminated PVC in car interiors or have already discontinued its use (Greenpeace International 2003), suggesting that substitution does not pose an insurmountable economic challenge. PVC is also used in other parts of a vehicle, including exterior body components, the underbody, and cabling insulation, which may also be impacted by a PVC restriction. Interior components are therefore one of a number of issues that manufacturers would have to address in the event of a restriction. When considered in combination, the costs of substituting different components may be more significant. Some alternative materials offer enhanced technical performance; in the United States, phase-out of PVC by one company was driven in part by concerns over its durability and its weight (Greenpeace International 2003).

8.5.2.4 Medical applications (blood bags, devices and gloves)

Business as usual

Limited data are available to accurately determine the trends in PVC consumption in the medical sector, but short-term forecasting indicates that the global PVC market value is expected to continue increasing, and this is anticipated to be driven in part by increased use of PVC in a growing number of healthcare applications (see Section 3.5.1). It is unclear whether this growth is driven by activity in Europe or the rest of the world. A review of literature (Greenpeace International 2003, Health Care Without Harm 2004, Health Care Without Harm 2007) and consultation undertaken with stakeholders from hospital procurement on 27th September 2021 indicated that an increasing number of hospitals and healthcare procurement bodies have pursued efforts to discontinue PVC use in gloves and other applications; the focus group revealed that in a survey of ten European hospitals, all but one had already transitioned to PVC-free medical gloves, for example.

Success in substituting PVC has varied considerably across different medical applications, with some uses (notably blood bags, where the blood needs to be stored for an extended period, enabling stocks to be managed and transferred) has been more challenging. It is ubndersood some alternatives are being developed for this use; however, these appear to be currently available in more limited volumes (and price data has not been identified for this specific use). It is unclear to what extent these largely voluntary phase-out initiatives have impacted overall PVC use in the medical sector. Considering the uncertainty in trends of PVC use in healthcare, it is not possible to reliably predict future trends in the absence of any further action or restrictions. However, it is likely that PVC use will continue to increase in some medical application areas, at the same time as continued voluntary uptake of PVC-free alternatives.

The following sections discuss the health and environmental, and socio-economic impacts of a phase-out of PVC in medical applications.

Health and environmental impacts

Use of alternative plastics in direct physical contact with patients poses similar issues to PVC with regard to transfer of toxic additives, notably plasticisers such as phthalates, to the body. Describing their decision to phase out PVC use, Sykehusinnkjøp HF (the Norwegian Hospital Procurement Trust, discussed further below) noted that part of the decision was precautionary; by eliminating PVC from their supply chains, the Trust seeks to avoid any potential future risks associated with additives used in place of phthalates. Note: phthalates are currently subject to a REACH restriction, although medical applications are exempt.

For hygiene purposes, the majority of disposable medical appliances are incinerated. As detailed in Section 6.3.4.2, PVC’s chlorine-based structure makes combustion of hospital waste is a source of toxic dioxins and furans emission, although these emissions have decreased significantly. A transition to non-chlorine-based alternatives including EVA, PE, PP and nitrile (see Section 7.14) could lead to a reduction in impacts from waste incineration emissions. A considerable amount of health care waste is disposed of after single use, so action would represent a shorter term reduction in PVC waste.

Economic impacts

The key economic impacts in the medical sector are the same as those described in section 8.5.1. Any phase-out scenario targeting PVC will result in a loss of business and jobs for primary PVC producers, offset by increased demand for alternative materials (see Section 7.14), and greater economic opportunities for businesses that manufacture them. Where production of alternatives in

the EU is not feasible or cost competitive, then supply chains are likely to shift to non-EU producers.
Section 7.14.4 notes that EVA, PE and PP alternatives in the medical sector are cost-effective compared to PVC, but that silicone rubber and nitrile may present more significant increased material costs. Additional production and material costs are likely to be passed on to public and private healthcare providers, although several are significant bulk purchasers. Sykehusinnkjøp HF stated that substituting PVC tubing products resulted in a temporary 30% increase in unit costs, caused by both materials and loss of economies of scale, which was regained as volumes increased. The Trust noted that the costs for the affected products represented a small portion of their overall budgets (see case study below). This, in turn suggests that costs passed on to private sector patients, or via greater private health insurance costs, would be marginal. This would therefore present an increase in upfront investment to prevent the risk of downstream human health damage.

In procuring PVC-free products, hospitals and patients may initially experience some drawbacks in technical performance of new products compared to their previous PVC product lines. These would need careful testing to ensure patient safety is not compromised.

<u>Case Studies</u>

Case Study: Sykehusinnkjøp HF (Norway Hospital Procurement Trust), Norway

Norway's hospital procurement is managed by a unified body; the Norwegian Hospital Procurement Trust. The trust, working alongside other similar organisation in Sweden, had ongoing concerns over the use of PVC in medical applications for several years. Their primary concern has been the use of additives, specifically phthalates, including DEHP, in a healthcare setting. Whilst the Trust recognises action has been taken under REACH on some phthalates, their preference is to remove uncertainty over potential regrettable substitutions for the future. More general concerns from PVC during manufacture and end of life are also relevant, as the vast majority of waste is incinerated and due to challenges with recycling medical products, as well as legacy contamination from harmful additives.

From 2015, the Trust initiated a phase out of PVC products, from medical goods the Trust procured. This included a requirement for non PVC products in the tenders' award criteria, incentivising suppliers to use alternatives. In addition, the Trust lab-tested products to verity that information provided in tenders were correct. Products developed with alternatives were tested with clinicians before acceptance, and gradually met with obligatory requirements to be PVC-free in the following tenders.

The Trust reported no pushback from its suppliers on the concept of switching. The Trust has engaged suppliers and responded that manufacturers in the supply chain benefit from supply certainty to facilitate investment in alternative products. The Trust did note some initial reluctance in some supply chains with switching, due to familiarity with existing product lines, but the response from medical staff was positive, following testing of the products developed with alternatives to PVC. This testing stage was very important, given technical failure of parts may have serious consequences and the Trust makes use of contract breaks to manage this risk.

In substituting PVC products, the Trust encountered a temporary increase in 30% in the unit costs of tubing products. Approximately 15% of this increase was attributed to increased material costs from the use on PVC alternatives. The remaining 15% was attributed to increased production costs, this, in turn, reflected the relatively small volumes and this was expected to fall back as higher volumes are manufactured. Anecdotal information from Sweden indicates that prices paid for PVC and non PVC medical tubing was equivalent as larger volumes were ordered. For some products, there were no price differences between PVC and PVC-free alternatives.

Whilst the Trust have some challenging targets for incremental cost efficiencies in procurement year on year and can exert some leverage on suppers due to their size, the price of these medical supplies represent a small proportion of their overall budgets. The primary concern of the Trust was quality and safety.

Where suppliers had to develop wholly new formulations for non PVC products, the price increase was higher, closer to 30-40%. As above, these were expected to fall as production increased.

The Trust acknowledged that almost all products were " a little bit more expensive" and the technical quality – whilst not placing any risk on the patients – was "a little bit inferior", but were fit for purpose nonetheless, with positive feedback form clinicians on the ground.

The Trust will work closely with suppliers during the contractual period by exploring options to innovate and develop new products where they don't currently exist. In anticipation of further substitution of PVC

Case Study: Sykehusinnkjøp HF (Norway Hospital Procurement Trust), Norway
products the Trust is carrying out mapping research with suppliers to ascertain precisely which products are using PVC. To date, the Trust has encountered a "a very few" examples where PVC substitution has not been technically feasible. This has included PVC-free blood bags. The issue in this case was due to the shelf life for storage, but this could potentially be overcome by more frequent replacement. A European Union funded project (PVCfreeBloodBag.eu 2015) is underway to support the development of alternatives to PVC blood bags. Similarly, an alternative product has been piloted at Karolinska University Hospital, but is understood to not yet be in production. Other problematic substitutions were very specific and to date have included specific flexible tubing and challenges with the fit of face masks. In 2021, the Trust conducted a review of chemicals and materials in its products to establish a baseline for monitoring progress. The review found that for medical commodities 23% of products include PVC, of which 26% contain phthalates that are on the Trust's Restricted Substances List. The RSL includes the phthalates on the Reach Candidate List and Restriction List and those identified as SVHCs, but not yet included on the Candidate List. Where phthalates were present, 89% were DEHP. If the PVC-products were phased out, it would reduce products with candidate list substances with 80 % and those with phthalates with 84 %. The Trust expressed a view that "clear guidelines", either through the European Union Ecolabel, or preferably regulatory action - ideally with timelines - on PVC would provide a clear market signal to industry, enable production volumes of PVC-free products to increase, costs to come down, and the portfolio to expand.

Case Studies: PVC phase out in selected EU hospitals
• **Vienna Hospital Association, Austria:** The Vienna Hospital Association oversees 18 hospitals, nursing homes and geriatric care centres. The Association adopted a policy of discontinuing PVC use from packaging, building materials and medical devices in 1992. Since then, Glanzing Paediatric Hospital and Preyer Paediatric Hospital have almost completely eliminated PVC and phthalate use. An audit of PVC use at the hospitals revealed that the share of PVC products had declined from 14.6% to 0.37% (Glanzing) and 9.8% to 0.9% (Preyer). As of 2004, almost all invasive medical products (including pacifiers, IV bags, blood filters, tubing) were PVC-free. The remaining PVC use is in non-invasive products where no feasible alternatives exist. In 2004, hospital management anticipated that PVC-free products for these outstanding applications would be available within the following two years**.** (Health Care Without Harm, 2004) • **Na Homolce Hospital, Czechia:** The hospital switched to PVC-free IV bags as of November 2003. Initial costs of PVC-free bags were higher, but the hospital managed to negotiate on prices. After 3 years the hospital pharmacy completely switched to PVC-free IV bags, including alternatives such as PE, PA, PP. Since substituting IV bags, the hospital has gone on to substitute IV lines (Health Care Without Harm, 2007). Sources note that substituting IV bags bears relatively little additional cost. (Health Care Without Harm, 2004) • **Grenaa Hospital, Denmark:** Grenaa Hospital began eliminating PVC in 1988. By 2002, it had substituted 95% of its PVC products and was working to produce PVC-free nasal cannulas, suction catheters and oxygen masks with PVC-free tubing (MedWaste, 2001). The hospital reported that many of the alternatives are more cost effective (GreenBiz, 2002). (Greenpeace International, 2003). It should be noted that more specific costs data was not identified.

8.5.3 Socio-economic effects in other applications

8.5.3.1 Pipes

Near term economic forecasts indicate that demand for PVC in the construction industry, including piping, will increase in the coming years (Mordor Intelligence 2020) (see Section 3.5.1). The evidence suggests that whilst there are some differences in costs per metre, and in weight and flexibility, the main driver of cost would be the initial investment for material and siting the pipe, frequency of breakage and ease of repair. Repairing pipes in situ with a "sleeve" or similar, without excavation/removing a section of pipe is a key issue for non-plastic materials. The costs of product redevelopment and/ or replacement would be borne by public water utilities and/or private water companies, depending on the prevailing model in each Member State. Costs would be incurred in the form of delays by the general public if breakages occur and in situ repair was not possible or where wider shutdowns were required. The long service life of these components indicates that effects on waste reduction would occur in the longer term (c. 20 -30 years or more) and the presence of legacy PVC pipes would remain problematic for many years.

8.5.3.2 Cables

Data on future trends in PVC use in cables in the absence of further action is limited. Alternative plastics (including potentially new formulations not covered in the AoA) are the most likely to be used. Each has different technical performance and material unit costs differ significantly. The costs for some are comparable with PVC while others, likely to be used in specific circumstances, are significantly higher. The major costs in this application are likely to be reformulation and testing to ensure that individual product lines fulfil the specific technical and safety requirements for conditions of use. These costs will be borne by cable/jacket providers and ultimately by a range of downstream users that includes automotive and ICT/EEE part suppliers and manufacturers, building developers/owners and tenants. Potential alternatives include fluoropolymers that may be the subject of ongoing regulatory action under REACH.

8.5.3.3 Window Frames

Near term economic forecasts indicate that demand for PVC in the production of window frames will increase in the coming years (Mordor Intelligence 2020) (see Section 3.5.1). Alternative materials are widely available and used, although these are typically more expensive to purchase. The evidence on the average longevity of PVC frames compared to alternative materials– which would offset these initial costs where lifetime use was longer - is disputed, but in many cases, these are replaced in advance of the maximum lifetime use. Costs would be borne by building developers/owners and, potentially, tenants. Energy efficiency standards would need to be met by these alternatives.

8.5.3.4 Packaging

Stakeholder discussion indicated that PVC demand in packaging has declined significantly in recent years, especially for use in bottles, but continues to be widely used in blister pack applications (see Section 7.9.1); in the absence of further action, it is likely that demand will continue to decline in future. Several plastics, emerging bioplastic and non-plastic alternatives are available and used in this application currently. A wider transition to non-PVC alternatives may result in an increase in manufacture and/or transportation costs as well as packaging redesign where alternative materials (e.g. recyclable paper/card) are incorporated. These costs would be incurred by packaging manufacturers and potentially passed on to the final retailer, although it is less clear if these could then be passed to the final consumer.

8.5.3.5 Toys

Data on future trends in PVC use in toys in the absence of further action are limited. Alternative plastics are available and used currently; these are typically more expensive, but the evidence suggests are also more durable. Additional costs from reformulation or cessation of specific low value product lines would be incurred by toy manufacturers and potentially passed on to the final consumer. The comparatively short service life of many of these components suggest this application presents a shorter-term opportunity for waste reduction.

8.5.3.6 Clothing (shoes, wellingtons)

Limited data are available to determine future use of PVC in footwear applications in the absence of further action. Alternatives are generally available and can achieve technical functionality. The use of some materials is likely to result in an increase in costs, incurred by footwear /sole suppliers, with the potential for some or all of the increase to be passed to the consumer.

8.6 Effects on downstream users and final consumers

The preceding analysis suggested that in certain applications the range of products available may decrease and/or the cost of reformulated/alternative products may increase (as is the case with some alternatives for use in cables; see Section 7.6.4). The technical functionality of some products is expected to be inferior, including those used in medical applications (see Section 7.14.3). The extent to which this may occur depends on the scope and speed of action.

8.7 Social Impacts

Any loss of the market that could not be replaced by alternatives would have implications for employment. As above, it is not clear that significant net effects would be felt in downstream sectors, but significant effects within the companies that manufacture PVC would be expected to occur, depending on their portfolios and on the scope and speed of action.

8.8 Wider Impacts

The main wider economic impacts of a restriction would relate to international trade and the competitiveness of the EU. Current exports of virgin PVC and PVC containing products would be lost, depending on the scope of action. The scale of the required reformulation effort would likely prevent other R&D activities that would otherwise have occurred. These would be offset by gains in human health and environmental damage avoided, and, depending on the scope and scale of the phase-out, potentially increased recycling and circular economy potential for recycled and reused products.

The extent of wider macroeconomic impacts will be significantly affected by whether non-EU manufacturers of PVC or PVC based articles could still place these goods on the EU market.

9. CONCLUSION

9.1 Overall conclusions

The main aim of this report is to reflect on the role of PVC as a material in the context of the EU European Green Deal and, more specifically, the envisaged transition to a circular economy.
To this end, the report identifies and describes findings and uncertainties, particularly under a chemicals angle, concerning PVC production, recovery and end of life treatment. The report takes into account environmental, human health, economic, legislative, and technical aspects, in order to enable the consideration of policy measures which ensure maximum recycling rates of PVC, but at the same time minimize environmental impacts and human health risks associated with management of PVC waste.
The following sections provide an overview of main findings concerning the fate and environmental impacts of PVC and its waste streams throughout the different life cycle phases. A final concluding section explicitly addresses main encountered data gaps and uncertainties, in order to put findings into perspective and indicate further research needs.

9.2 The chemistry and characteristics of PVC

Given its many applications, a variety of different types of additives are used in PVC to achieve the desired properties. Based on the main applications and concentration of plasticisers two types of PVC are differentiated:

- Flexible (or plasticized) PVC; and
- Rigid (or unplasticized) PVC.

As the additives are essential for the functions of PVC, it is necessary that PVC additives remain in the polymer. However, additives can be released from plastic products during their life cycle, leading to potential risks to human health and the environment.

<u>Migration of additives</u>

Within the plastic matrix, the vast majority of additives are not covalently bound. For some plasticisers e.g., phthalates but also for certain metals like lead it is well known that they migrate from the matrix (Danish Environmental Protection Agency 2016; Mercea et al. 2018; Polcher et al. 2020; Wiesinger, Wang, and Hellweg 2021).

Some PVC- related additives have been found in environmental matrices, like surface waters in close distance to landfills (Wowkonowicz and Kijeńska 2017), and also humans (Jamarani et al. 2018; Przybylińska and Wyszkowski 2016).

The extent of the migration varies greatly depending on the specific additive (effect of branching, molecular weight, end-group functionality, and polydispersity) and its initial concentration in the PVC (Danish Environmental Protection Agency 2016; ECHA 2019). In addition, the presence of several additives at the same time in PVC can influence the release potential of some of the additives (Mercea et al. 2018).

Some PVC-relevant plasticisers of the group of phthalates are, due to their reprotoxic effects and adverse endocrine effects, under scrutiny within the context of various EU regulations such as REACH, the RoHS Directive, and the Toy Safety Directive. For other additives, including those which are not under regulatory scrutiny, comparably less information is available. However, in order to assess the risk from hazardous additives in PVC the release potential/migration and subsequent bioavailability need to be known. One result of this study is that knowledge concerning these aspects is, for most of the additives, not (publicly) available. It is therefore difficult to make general statements regarding the migration of additives out of a polymer matrix and their bioavailability.

Classification of mixtures

PVC often contains various hazardous additives which are added in concentrations above the limits listed in Annex I to the EU CLP Regulation. As such, the resulting PVC mixture would need to be classified as hazardous if the relevant PVC additive is classified as hazardous or if one of the substances it contains has hazardous properties, and is added to the mixture in a concentration above the relevant limit values set out in Annex I of the CLP-Regulation. An exception to this can be applied if it can be proven through adequate and robust data that the substance is not bioavailable. For this publicly available SDS were screened and in some cases, it was mentioned that the additives are encapsulated in the polymer matrix and migration is negligible. If an additive does not migrate out of the polymer matrix it is not bioavailable and as such the mixtures is not considered to be hazardous in accordance with Article 12 of the CLP-Regulation. However, this statement needs to be proven with adequate and robust data provided by the supplier. Available data on the migration of most additives is not available and the data from the supplier, which proves, that the additive is not bioavailable, does not need to be included in the MSDS. As such it cannot be assessed whether the statement that the additives are bound to the polymer matrix and do not migrate is true or false. Without concrete data, a more precautionary approach concerning the classification of PVC may be preferable from an environmental and human health perspective.

9.3 Market for PVC and PVC products

Eurostat data indicate that approximately 6 million tonnes of primary PVC were manufactured in 2019 in the EU27. Of this:

- 4.9 million tonnes were unmixed rigid PVC;
- 0.4 million tonnes were rigid PVC; and
- 0.8 million tonnes were soft PVC mixed with other substances.

In addition, a total of approximately 5 million tonnes are placed on the EU market in 2019. Production volumes of unmixed PVC in Europe have declined between 2008-2019, with production of soft and rigid PVC having remained largely stable since 2008. The market value for the volumes of primary PVC produced in the EU27 was just under €5 billion in 2019. Taken together the values of primary and processed PVC placed on the market in the EU27 was in the order of €13 billion in 2019.

Whilst data are not precise due to differences in categorisation, the largest volumes of PVC are applied in construction (pipes and fittings, window frames), medical and healthcare applications and packaging. Cables for electronics used the largest volumes of PVC based on 2020 data from the European Council of Vinyl Manufacturers (ECVM).

Information on global PVC production is less detailed, but was estimated at 55 million tonnes in 2016, with China accounting for almost half of the volume. Estimated demand is somewhat lower at 45 million tonnes in 2018, although there is uncertainty around these numbers. Recent years have seen a steady increase in global production capacity, predominantly driven by China. Pre-pandemic, the global PVC market was expected to continue to grow at a rate of 3.5%, with China driving the majority of the growth. Simple market projections based on historical data suggest that PVC production could either stabilise in line with trends between 2013-2019 or continue to decline in line with trends between 2008-2019.

9.4 PVC waste recycling

It is assumed that currently 2.9 million tonnes of pre- and postconsumer PVC waste are generated per year in the EU (2.4 million tonnes post-consumer PVC waste). Currently, there are more than 100 operations in Europe which recycle PVC pipes, profiles, flooring, coated fabrics and membranes.

As for any waste stream, collecting PVC waste separately and cleanly is a precondition for high-quality recycling. While sector-specific take-back systems and separate collection are successfully applied to post-consumer waste from the construction and building sector (e.g. windows, pipes) in most of the EU Member States, the (limited) available information suggests that separate collection remains a challenge in practice for most post-consumer PVC waste streams. PVC waste from packaging, automotive and medical waste is rarely separately collected (and thus recycled less).

Mechanical recycling is currently the only relevant process for pre- and postconsumer PVC waste in the EU. No significant chemical recycling of PVC is done at industrial scale in the EU; while chemical recycling generally can be considered a technology in development, chemical recycling specifically of PVC is not practiced in the EU (except in pilot plants or in research projects) and in most initiatives of chemical recycling, PVC is not the targeted material.

In 2020, 730,000 tonnes of (pre- and post-consumer) PVC waste have been recycled within the VinylPlus program (regional scope is EU-27 plus Norway, Switzerland and the UK). About 60% of the recycled materials by VinylPlus consist of rigid PVC (i.e. profiles, pipes and fittings). A 35% majority of PVC recyclate is redirected into new windows and profiles, whereas 15% is directed into traffic management products and 13% into pipes. Window waste, which is the biggest PVC waste stream, seems to be the only PVC waste stream that can be considered recycled in closed loop or at least partly.

Separation and sorting technologies were optimised in the last years, however contaminants, composites, laminates or other complex product composition still pose a problem for conventional mechanical recycling.

Moreover, restricted or unwanted additives (e.g. cadmium and lead stabilisers or phthalate plasticisers such as DEHP) are an issue for PVC recycling, as no feasible recycling technology exists to remove such substances from PVC waste. In the future, non-conventional mechanical recycling processes like selective dissolution could be an option to remove contaminants and/or legacy additives from PVC waste, however currently no feasible recycling technology exists to remove such substances from PVC waste.

In the absence of feasible decontamination processes, the presence of restricted or unwanted additives in recyclate could in theory require finding a balance between the aim for a toxic-free environment and calls for an increased recycling in the Circular Economy. Equally importantly a discussion may be required about different other scenarios, reaching from a targeted risk-based restriction to a total phase out of PVC applications. In pipe and window production, but also in some flooring companies, PVC recyclate is inserted between or under layers of virgin PVC (co-/tri-extrusion). Apart from the case of pipes, for which this requirement applies due to REACH, it is not clear to what extent this is only done for aesthetical reasons or also to protect the environment from substances (e.g legacy additives) in the recyclate or to dilute hazardous substances with virgin material.

9.5 PVC waste disposal, other recovery, import and export

Due to the overall lack of data concerning the quantities of PVC directed towards disposal and recovery, a quantitative scoping of five major PVC-relevant waste streams (ELVs, C&D waste, WEEE, packaging and household waste and medical waste) was performed to analyse the relevance of different waste treatment options. Where possible, the shares directed towards the following options were assessed:

- Disposal, including: i) disposal through incineration without energy recovery and ii) disposal on landfills; and
- Recovery, including: i) energy recovery, ii) recycling and iii) reuse.

Based on the comparison with the results of another study focusing on the quantitative scoping of PVC in different waste streams, the following can be concluded:

- Between 1.719.334 and 2.435.000 tonnes PVC waste are produced per year.

- Between 73.1 – 77.7 % are directed towards recovery.
- The majority of PVC waste is energetically recovered (41.9 - 53.1 %).
- The mechanical recycling rate ranges between 24.6 and 31.2 %.

The analysis demonstrated that incineration and landfilling represent the main non-recycling treatment routes for PVC and that the treatment of PVC waste strongly depends on the specific waste stream and its properties. Where PVC waste is energetically recovered, the majority is treated in MSW incineration plants and a relatively small share is used as RDF. Nevertheless, major data gaps exist concerning the fate of PVC present in the analysed waste streams and the quantities directed towards different incineration facilities or landfill types.

As regards incineration, relevant incineration facilities for the treatment of PVC waste include

- MSW incineration facilities with and without energy recovery;
- medical and hazardous waste incineration plants; as well as
- co-incineration plants (cement kilns and blast furnaces).

Both non-hazardous and hazardous streams of PVC waste are subject to incineration. In this regard, it is relevant to note that incineration represents the main treatment method for hazardous PVC waste.

Due to the improvement of air pollution control technologies, impacts on human health and the environment from the incineration of PVC waste are rather related to the solid incineration residues than to air pollution. Concerns exist regarding the leaching of soluble salts, heavy metals or dioxins from solid incineration residues which were landfilled or processed otherwise (e.g. used as raw material). Available data on the economic impacts indicates that the treatment of PVC by incineration might increase maintenance costs and costs associated with the disposal of solid residues. However, several commercially available initiatives for the management – including for instance resource recovery or thermal treatment – of solid residues are available.

As regards landfilling, stakeholder input indicates that PVC waste is mainly disposed of at landfills for non-hazardous waste, while smaller quantities are disposed of in landfills for hazardous or inert waste. As indicated in section 4.2, a highly relevant question in this regard concerns the classification of PVC waste as hazardous or non-hazardous. Such classification may inter alia depend on the extent to which migration and bioavailability can or cannot be established for relevant additives contained in a relevant PVC waste stream. Chapter 2 notes migration of additives from plastics can take place, since such additives are not covalently bound in the plastic matrix. However, data on migration and bioavailability of PVC additives is limited. As such, from a precautionary perspective, further assessment of the classification of PVC as hazardous or non-hazardous may be necessary for the determination of the most suitable disposal route.

Within the EU, waste management practices and landfilling costs vary considerably between Member States resulting in different amounts of PVC being landfilled. As regards the environmental and human health impacts, available literature suggests that phthalates used as additives in soft PVC might be detected in considerable concentrations in landfill leachate, while leaching of heavy metals from PVC additives (stabilizers) or emissions to landfill gas attributable to PVC can be considered less relevant. Another potential risk related to landfilling of PVC waste are dioxin emissions resulting from landfill fires.

Illegal disposal options of mixed or hazardous waste containing PVC include uncontrolled burning, illegal recycling, illegal trade and illegal dumping. The uncontrolled burning of waste, an increasing phenomenon, containing PVC leads to emissions of various pollutants which are harmful to the environment or human health, such as dioxins. Further impacts on human health and the environment can occur due to illegal landfilling of waste containing PVC.

Due to the existing differences concerning the recycling infrastructure and cost structure for waste treatment, significant quantities of PVC waste are shipped legally within the EU but also imported from and exported to non-EU countries. Recent developments in the legal framework on transboundary shipments of plastic waste (including PVC waste) from the EU to third non-OECD countries (see Annex 1.1) may lead to considerable changes in shipment volumes and practices. Available information suggests that hazardous plastic waste is often mixed with non-hazardous waste, in illigal act in itself, to disguise illegal export. As a result, the mixed plastic waste is not suitable for recycling anymore due to hazardous characteristics.

9.6 PVC alternatives

PVC applications considered in the analysis of alternatives include:

- Pipes and pipe fittings;
- Window frames;
- Cables;
- Flooring;
- Packaging;
- Inflatable toys;
- Boots and soles of shoes;
- automotive uses (dashboards and artificial leather); and
- Medical applications.

The analysis indicates that there are economically viable and technically feasible alternatives in the vast majority, if not all, applications assessed in detail where PVC is currently used. These alternatives are not without technical drawbacks, the extent of which differs between applications. Additional costs are associated with their use in several cases. The alternatives identified fall into three groups:

- Alternative non-plastic materials (such as wood, leather, cloth, etc.);
- Alternative plastics (here a wider variety of plastics were identified; some of these contain additives which pose similar risks to those used in PVC and others which are the subject of potential regulatory action); and
- More novel alternatives (such as bioplastics), about which limited application-specific information has been obtained.[202]

The vast majority of the alternative materials identified are commercially available, often placed on the EU market in significant volumes at present. The balance of evidence suggests that the human health and environmental risks associated with a transition would decrease, although identifying net effects is challenging and differs between applications.
The following main conclusions per considered application can be highlighted:

- Technically feasible plastic and metal alternatives are available for use in pipes and fittings and can be adopted to meet different piping requirements. Costs of alternatives range between broadly comparable to more expensive to PVC when considering unit costs and lifetime although differences based on the specific needs of the application could affect installation, maintenance and repair.
- Alternatives to PVC window frames are commercially available, typically at greater cost albeit with generally longer lifetimes.
- A number of technically and economically viable alternatives exist for commercial and residential flooring, as well as hospital flooring.
- Technically feasible plastic alternatives exist for cables at varying costs. Plastic alternatives may pose similar health and environmental risks to PVC.
- A variety of non-plastic, plastic and emerging bioplastic alternatives are available for use in packaging, and the use of PVC in packaging has declined substantially in Europe over the last decade.
- Technically feasible alternatives in children's toys are commercially available. These are typically at higher cost but can offer greater durability.

[202] Some alternative plasticisers (i.e. phthalate free) were identified.

- Alternative plastics and leather are currently used for footwear but are generally more expensive.
- Alternatives are already in use in automotive interiors and several manufacturers have already started reducing/phasing out PVC or have made public statements on their desire to do so where possible. Alternatives range from comparative in price to more expensive than PVC components.
- Alternatives to PVC are available in medical applications, with examples of hospitals and trusts committing to phasing out its use, although challenges exist in substituting its use in blood bags. At present, alternatives are typically more expensive than PVC products.

9.7 PVC phase out scenarios

The analysis considers different potential options to implement or accelerate an EU-wide phase-out of PVC, and the key economic, social and wider economic effects of such options. A 'Do nothing' base scenario would see no further regulatory action and PVC use continuing to develop with market demand. On the other side of the spectrum of possibilities, a total phase-out and transition to alternative substances could be implemented through a REACH restriction on all uses of PVC. Note that this would require uncontrolled risks to human health and/or the environment to be proven.

A number of options have been identified between these two scenarios, including targeted risk-based restrictions under REACH or a potential future application of the "essential use" concept to determine whether specific applications of PVC should continue to be permitted – depending on ongoing analysis on the potential application of that concept in REACH. Depending on the scope, any phase-out scenario could lead to a combination of response scenarios in product supply chains. Replacement of PVC with alternative materials or substances where feasible alternative(s) are available or identified after further development. Where technically and economically feasible alternatives are unavailable, affected product lines, revenues and employment in the EU will cease.

Socio-economic impacts associated with these scenarios are considered across a number of focused applications, with more detailed consideration of impacts in the medical, automotive and flooring sectors. Impacts include:

- Reduced environmental and human health risk through elimination of the source of the risk associated with PVC additives. This is dependent, however, on the scope and timescales of actions, the applications covered and the progress of parallel regulatory action which may affect the use of some alternatives;
- Loss of revenues of primary PVC producers may be significant, depending on the timing and scope of action. However at the level of the EU as a whole, this could be offset by increased sales of alternatives. In some cases, this may require significant shifts in the supply chain from one material to another (e.g. from PVC to aluminium or wood). In others, it may require use of alternative plastics and/or the development/adjustment of new formulations.
- Potential increases in material costs where alternatives are more expensive than PVC, on a unit basis. Alongside reformulation and other costs (testing, R&D, marketing, product approval etc), some or all of the additional costs may be passed on to end consumers by manufacturers. This may make goods more expensive at the point of sale, although it is very difficult to quantify this and the extent will be influenced by the scope and timing of action;
- Potential loss of employment accompanying loss of market in PVC production;
- Potential loss of PVC exports to third countries; and
- Drawbacks in product performance where alternatives do not meet the technical performance of PVC.

In several applications, there are downstream users, or other actors in the supply chain which have voluntarily substituted PVC in their products, or are in the process of doing so, reflecting ongoing concerns about their environmental and human health effects. Examples can be found in the automotive, flooring and healthcare sectors.

9.8 Reflection on data gaps and further research

While the findings presented in previous sections provide various insights and conclusions concerning the PVC life cycle, attention should also be paid to various identified data gaps. The considerable gaps in data concerning all PVC life cycle phases may raise various questions and challenges for future EU policy measures, most notably from a precautionary perspective.
With regard to the chemistry of PVC and associated risks to human health and the environment, the lack of publicly available data on release potential, migration and consequent bioavailability of hazardous additives constituted an important limitation of the conducted analysis. In addition, only limited data could be identified concerning the classification of PVC mixtures containing hazardous additives by manufacturers.

With regard to the current market of PVC, different demand datasets use different sector classifications and differing levels of granularity. Comparing and corroborating datasets is not straightforward and has resulted in uncertainty over volumes of PVC used in different sectors.
With regard to PVC waste generation volumes, it should be noted that in 1999/2000, generated PVC waste volumes were estimated at 4.1 million tonnes. Studies in 1999 and 2000 projected a steady increase in PVC waste generation (e.g., 6.2, 6.4, or 7.2 million tonnes in 2020). Currently, it is assumed that between 2.5 and 2.9 million tonnes of PVC waste are generated. This would represent a decrease in waste volumes since 1999/2000 which is not consistent with the 1999/2000 projections. Identified data has not provided a clear explanation for this disparity.
With regard to the recycling of PVC, it is important to note that most quantitative data on recycled volumes originates from the VinylPlus initiative. While this initiative seems to cover the majority of recycling operations in the EU, verification of data reliability on the basis of alternative sources has been challenging. In addition, the available data from VinylPlus is not detailed concerning, e.g., the origin of PVC waste fractions as pre- or post-consumer.

With regard to the disposal of PVC, it should be noted that data concerning the quantities of PVC waste directed towards different waste treatment options is limited. This has implications for the calculation of PVC waste disposal routes, which have been based on broad assumptions in this report. In addition, it should be noted that data on illegal disposal of PVC (e.g. through open burning or dumping) is very scarce and, if available, largely anecdotal. The lack of data on illegal disposal poses important limitations to the comprehensive mapping of the fate of non-recycled PVC in the EU.

Finally, with regard to the assessed phase out scenarios for PVC, whilst volume and average price data provides an estimate of market segments, it has not been possible to differentiate the various PVC products in terms of typical product margins. It is also not known how many product lines may be affected in each of the applications/sectors, or costs of reformulation.
Annex 7.1 provides a more comprehensive and detailed overview of identified data gaps, indication of the implications of these gaps for research, as well as suggestions on how these gaps can be addressed through future research and other measures.

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