Investigating options for reducing releases in the aquatic environment of microplastics emitted by (but not intentionally added in) products

Final Report

Simon Hann

Chris Sherrington

Olly Jamieson

Molly Hickman

Peter Kershaw

Ayesha Bapasola

George Cole

23rd February 2018

Report for DG Environment of the European Commission

Prime Contractor

ICF
Watling House
33 Cannon Street
London EC4M 5SB. UK

Tel: +44(0) 203 3096 4800
Web: www.icf.com

Technical Lead

Eunomia Research & Consulting Ltd
37 Queen Square
Bristol
BS1 4QS, UK.

Tel: +44 (0)117 9172250
Fax: +44 (0)8717 142942
Web: www.eunomia.co.uk

Document Control

For Eunomia	
Prepared by	Simon Hann
Approved by	Chris Sherrington 
For ICF	
Checked by	Andrew Jarvis

Disclaimer

Acknowledgements

Peer Reviewers:

Dr Richard Thompson, Plymouth University, UK
Peter Sundt, Mepex, Norway

Other Contributors:

Panayiota Apostolaki, MRAG, UK
Outi Setälä, Finnish Environment Institute (SYKE)
Julia Talvitie, Aalto University, Finland

Executive Summary

This report has been prepared for the European Commission by ICF in association with Eunomia and partners.

The objectives of the study were to:

- Critically assess all available information on sources, pathways and impacts of microplastics in the aquatic environment, as well as on actions aiming at their reduction;
- Provide the analysis for the Commission to identify relevant measures and policy options to reduce releases of microplastics in the aquatic environment including through product policies, improved sewerage and storm water collection, improved waste water treatment and any other relevant measures whether specific to the various sources of microplastics or horizontal measures;
- After validation of the proposed options by the Commission, analyse the social, economic and environmental impacts of the selected options
- Compare the impacts of the options with the baseline to enable the selection of most efficient and effective options, on the basis of which the Commission might propose preferred action(s)/options at EU level.

Microplastics' is a term commonly used to describe extremely small pieces of plastic debris in the environment resulting from the disposal and breakdown of products and waste materials. The concern about microplastics centres on their potential to cause harm to living organisms in the aquatic environment although this report also shows that microplastics are likely to accumulate in other environments such as rivers and soils. It is important, therefore, not to overlook the potential for negative impacts in these places as well.

This study is concerned only with microplastics that are created during the lifecycle of a product through wear and tear or emitted through accidental spills. Microplastics that are an intentionally added ingredient within a product or designed with the expectation that they could be emitted during their lifecycle are covered by the parallel study by Amec Foster Wheeler[1] in close cooperation with the authors of this report. Microplastics that may be generated as a result of poor or non-existent waste management or as a result of the degradation of larger ***plastic waste*** are covered by other initiatives under the Commission's Plastic Strategy[2] to reduce macro plastic litter (which can break down over time to form microplastics).

---

[1] Amec Foster Wheeler Environment & Infrastructure UK Limited (2017) *Intentionally added microplastics in products*, Report for European Commission (DG Environment), October 2017 http://ec.europa.eu/environment/chemicals/reach/publications_en.htm

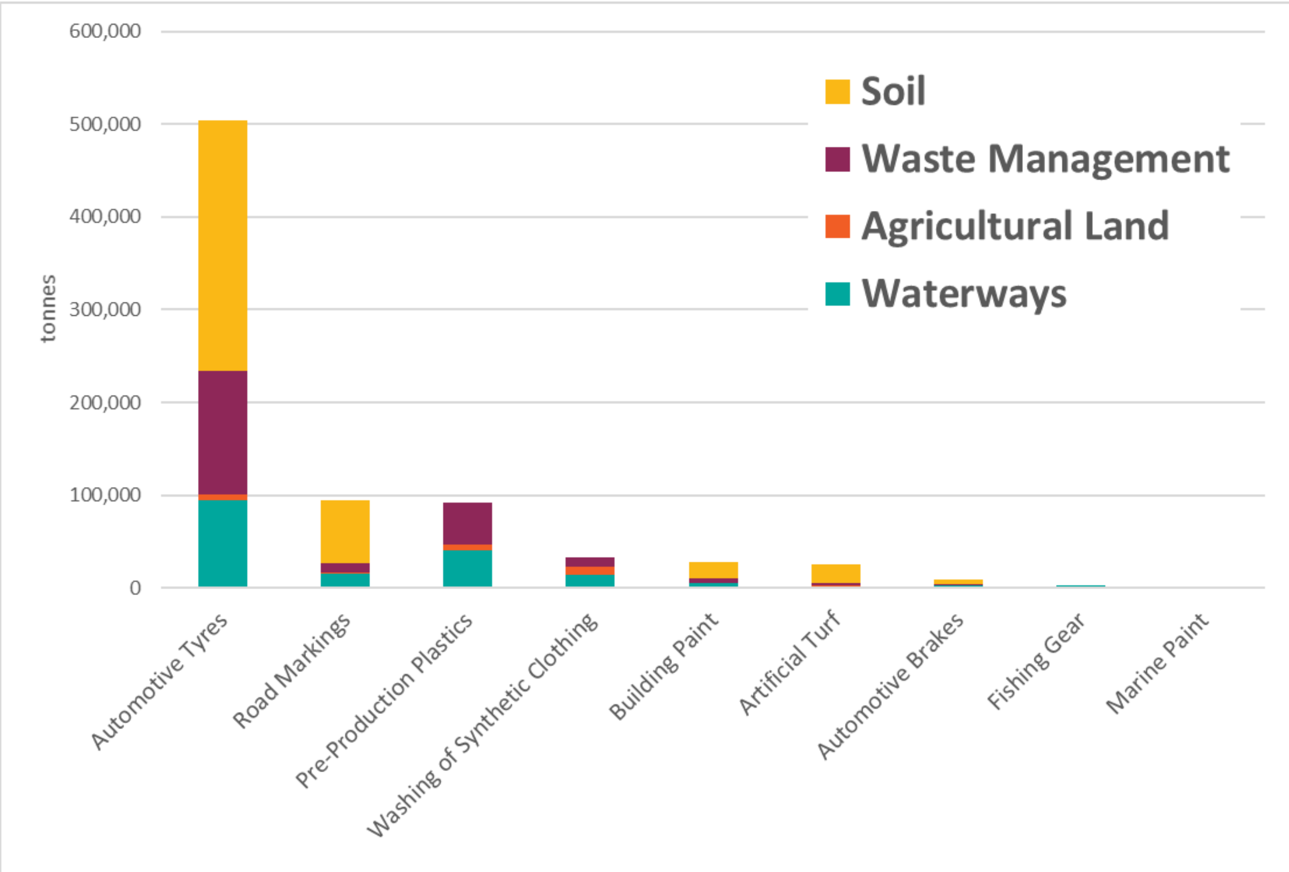
[2] https://ec.europa.eu/commission/publications/documents-strategy-plastics-circular-economy_en

E.1.1 Estimating Microplastics Emissions

Figure 1 shows the estimated generation of microplastics at source for the products identified during this study. This demonstrates that tyres, road markings, pre-production plastic pellets and washing of synthetic textiles are all large sources of microplastics emissions into the environment. Several other sources have been identified, but are found to emit less. Artificial turf is relatively small source, however emissions are from a relatively small number of large point sources (pitches) with annual emissions of 1—5 tonnes each, whereas automotive tyre wear emanates from millions of vehicles all throughout Europe's road systems.

Soil is the largest single sink and is largely comprised of microplastics washed or blown from roads. These also may, over time, be washed into waterways. Waste management includes microplastics collected during road sweeping and the various roadside storm water filtration devices; however, these devices are only effective if regularly emptied. Also included is wastewater treatment sludge destined for incineration or landfill—this accounts for around half of all sludge and in the case of landfill, may also provide a pathway for leaching microplastics to waterways. The other half of the sludge is applied to agricultural land along with any captured microplastics. The effects of this are yet to be established.

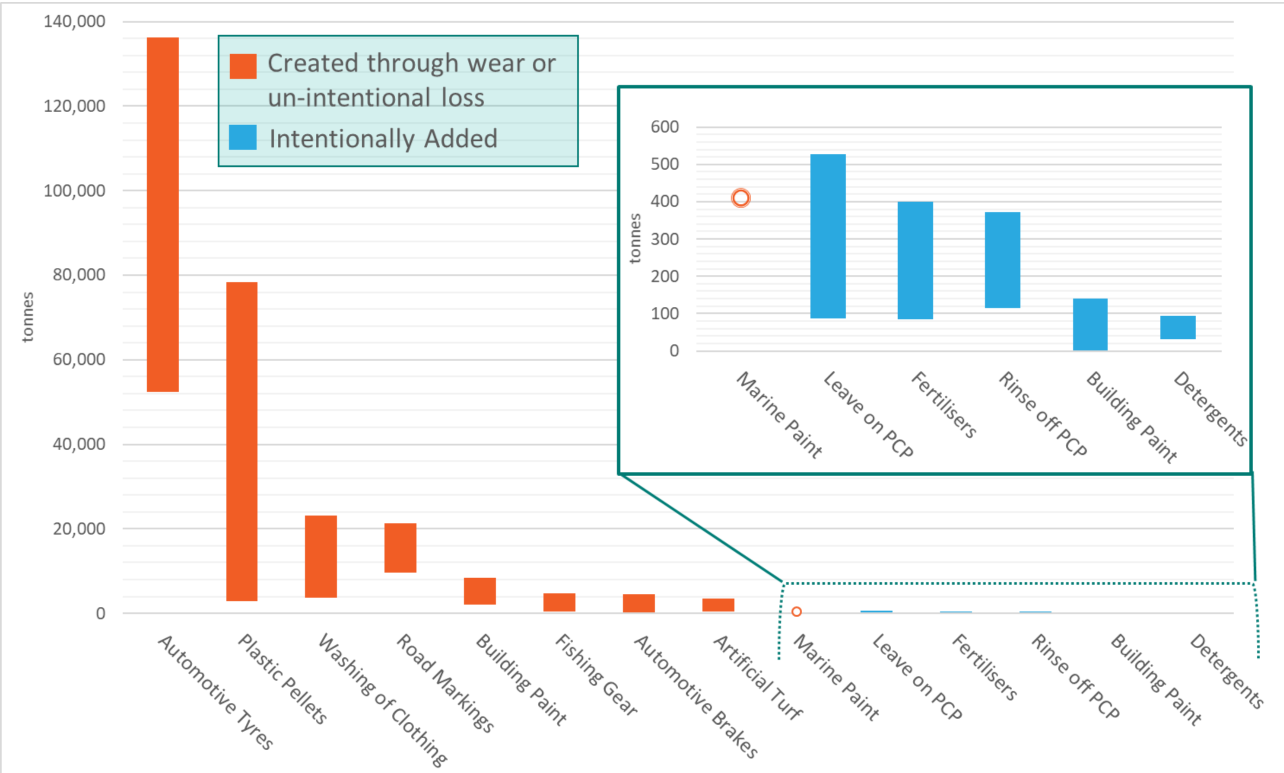
Figure 1 - Source Generation and Fate of Microplastics from Wear and Tear in the EU (midpoint)



Source: Eunomia modelling

Only a proportion of microplastic emissions end up reaching the aquatic environment. Figure 2 shows the modelling results of the pathways to surface waters. This demonstrates that tyre wear is potentially the largest source of microplastics entering the aquatic environment. Washing of clothing accounts for a greater proportion of microplastics entering the aquatic environment than initial microplastic emissions, as the pathway to surface waters is better defined with fewer opportunities to be captured—except in wastewater treatment (WWT) plants where capture is estimated to be between 53 and 84 per cent. This capture in WWT will vary greatly between countries, depending upon the WWT infrastructure, but the level of microplastics retention is still highly uncertain. Emissions of 'Intentionally Added' microplastics from the parallel study are also shown for comparison.

Figure 2 – Annual Emissions of Microplastics to Surface Water (Upper and Lower Ranges)[3]



Source: Eunomia and Amec Foster Wheeler modelling

[3] Marine paint has no range associated with its estimate as emissions are direct to the marine environment. Building paint is included twice, once for intentionally added losses and again for losses due to wear during the life of the paint.

E.1.2 Reduction Options

Table 1 provides a summary of the options analysed for this report. From these results it is clear that the largest reductions in both source emissions and emissions to surface water can be achieved through ***measures targeted at reducing emissions at source***. Supply Chain Accreditation for pre-production pellets is likely to have the largest reduction impact—600,000 tonnes cumulative reduction to surface waters between 2017 and 2035—and is also expected to be the most cost effective. This is a Regulation that stipulates best practices be demonstrated vertically throughout the supply chain including logistics and, with a high level of stakeholder support observed through this project, it is expected to be the most effective approach. However, it is important to note that the amount of pellet loss is subject to some uncertainty, therefore the reduction impacts also come with a reasonably high level of uncertainty. What is clear however, is that pre-production pellets are frequently found in significant numbers on European beaches, so there is a strong marine-litter prevention rationale for action targeted towards this source.

Similarly, source prevention for tyre wear abrasion is likely to have a large impact—a cumulative reduction in emissions to surface water of 500,000 tonnes. The amount of tyre wear generated at source has a reasonable level of certainty associated with it, but its pathways to various environments are currently not as well understood. Measure 3, using the Type Approval Regulation to remove the worst performing tyres from the market, and the combined measure (Type Approval plus including tyre abrasion rates on the EU tyre label) both appear to be relatively cost-effective in preventing emissions at source compared with other measures. The testing required to implement these measures is estimated to add between €0.03 and €1.43 onto the cost of a tyre. However, even the combined measure is only expected to reduce emissions to surface water by 33% (of tyre wear emissions). Therefore, it is also important to consider downstream measures such as capture in storm water, as this is expected to be the dominant pathway for microplastics emitted on roads. Costs for this are difficult to estimate as it is not known how much infrastructure would be needed to achieve a certain capture rate – this being strongly influenced by the level of traffic on particular roads – and thus primary research in this area is needed. That having been said, if storm water management is approached on a case by case basis by targeting hotspots for microplastics emissions, it is likely to cost more per tonne than preventative measures, but less than improvements to wastewater treatment (WWT) plants—the only other likely intervention point.

Source prevention measures for textiles are also likely to be cost effective if a self-certification process is used to govern the implementation of a maximum threshold for fibre release. This effectively removes the garments and fabrics from the market that emit the most fibres during washing. If (third party) testing of individual textile products is necessary to regulate this, the costs may begin to make downstream capture more appealing. Similar to tyre wear, however, such measures might also be expected to have limited impact (however this largely depends of where the maximum fibre release threshold can feasibly be set) and therefore downstream measures may also be necessary regardless of the cost effectiveness of source measures. For textiles the cost-effectiveness of capture at the washing machine via a filter or at a WWT plant appears to be very similar. However, there are some more subtle qualitative differences that suggest capture at the machine may be more favourable. Firstly, current WWT technology sequesters microplastics in sludge, which may simply transfer the issue for countries that apply sludge to land. Secondly, it should also be recognised that if any of the measures aimed at reducing the key sources of microplastics through WWT are implemented, the cost-effectiveness of any infrastructure improvements would decrease significantly. Tyre wear (25%), pre-production pellets (27%) and textiles (40%) are the largest contributors to microplastics loads in WWT and they all appear to have more cost-effective source prevention measures associated with them. For these reasons it may be more appropriate to investigate washing machine capture in the absence of proven cost-effective capture in WWT.

Table 1 – Summary of Measures (Emissions Using Midpoint Baseline Projections)

Measure		Cumulative Emissions 2017-2035 (tonnes)		Cumulative Reduction from Baseline 2017-2035 (tonnes)		Annual Cost per Tonne Prevented *at Source*
		Source Emissions	Surface Water Emissions	Source Emissions Reduction	Surface Water Emission Reduction	
Automotive Tyres						
Baseline		***11,200,000***	***2,100,000***	-	-	
Measure 2 -Tyre Label	Low	10,900,000	2,040,000	300,000	60,000 (3%)	Circa €11,000
	High	10,400,000	1,900,000	800,000	200,000 (19%)	Circa €4,000
Measure 3 -Type Approval		10,100,000	1,900,000	1,100,000	200,000 (10%)	Circa €3,000
Combined		8,700,000	1,600,000	2,500,000	500,000 (33%)	Circa €1,300
Pre-Production Plastics						
Baseline		***2,200,000***	***1,100,000***	-	-	
Measure 4 - Supply Chain Accreditation		800,000	600,000	1,400,000	600,000 (55%)	Circa €950
Measures 1-3 -Horizontal Measures		1,200,000	700,000	1,000,000	400,000 (36%)	Circa €1,400
Textiles						
Baseline		***600,000***	***250,000***	-	-	-
Measure 2 - Maximum Threshold	10%	500,000	210,000	100,000	40,000 (16%)	Self cert €500—20k Third Party €4k—100k
	20%	400,000	160,000	200,000	90,000 (36%)	
Measure 3 - Labelling		500,000	210,000	100,000	40,000 (16%)	
Measure 4 -Washing Machine Filter	Filter	300,000	130,000	300,000	120,000 (48%)	€50k—125k
	Cora Ball	500,000	220,000	100,000	30,000 (12%)	€41k—104k
	Guppy Friend	500,000	200,000	100,000	50,000 (20%)	€44k—112k
Wastewater Treatment						
Baseline		**-**	***600,000***	-		
Measure 2 - Improved WWT		-	400,000	-	200,000 (33%)	€45k—137k

Note:

1. **Emissions figures rounded to nearest 100,000 or 10,000 for those less than 100,000.**
2. **Wastewater treatment cost per tonne is per tonne reduction into surface water as it is not a source of microplastics.**
3. **All figures are rounded**

Contents

Glossary

Table 2 – Common Polymers

Short form	Full name
ABS	Acrylonitrile butadiene styrene
AC	Acrylic
EP	Epoxy resin (thermoset)
EPDM	Ethylene propylene diene monomer
PA	Polyamide 4, 6, 11, 66
PCL	Polycaprolactone
PE	Polyethylene
PE-LD	Polyethylene low density
PE-LLD	Polyethylene linear low density
PE-HD	Polyethylene high density
PET	Polyethylene terephthalate
PGA	Poly(glycolic acid)
PLA	Poly(lactide)
PP	Polypropylene
PS	Polystyrene
EPS (PSE)	Expanded polystyrene
PU (PUR)	Polyurethane
PVA	Polyvinyl alcohol
PVC	Polyvinyl chloride
SBR	Styrene-butadiene rubber
TPE	Thermoplastic elastomers

Table 3 - Common chemical additives in plastics

Short form	Full name	Examples of Function
BPA	Bisphenol A	a monomer used in the manufacture of polycarbonates and epoxy resins
DBP	dibutyl phthalate	anti-cracking agents in nail varnish
DEP	diethyl phthalate	skin softeners, colour and fragrance fixers
DEHP	di-(2-ethylhexyl)phthalate	plasticizer in PVC
HBCD	hexabromocyclododecane	flame retardant
NP	nonylphenol	stabilizer in food packaging and PVC
PBDEs	Polybrominated diphenyl ethers (penta, octa & deca forms)	flame retardants
NPEs	nonylphenol	stabilizer in PP, PS
phthalates	Phthalate esters	improve flexibility and durability

Table 4 - Common Organic Contaminants absorbed by plastics

Short form	Full name	Origin
DDT	dichlorodiphenyltrichloroethane	insecticide
PAHs	Polycyclic aromatic hydrocarbons	combustion products
PCBs	polychlorinated biphenyls	cooling and insulating fluids, e.g. in transformers

1.0 Introduction

This report has been prepared for the European Commission by ICF in association with Eunomia and partners. It presents the final report of the study on *"Investigating options for reducing releases in the aquatic environment of microplastics emitted by (but not intentionally added in) products"* (Specific Contract No.070201/2017/SFRA/749833/ENV.C2 under Framework Contract No. ENV.C.2/FRA/2016/0017).

The objectives of the study were to:

- Critically assess all available information on sources, pathways and impacts of microplastics in the aquatic environment, as well as on actions aiming at their reduction;
- Provide the analysis for the Commission to identify relevant measures and policy options to reduce releases of microplastics in the aquatic environment including through product policies, improved sewerage and storm water collection, improved waste water treatment and any other relevant measures whether specific to the various sources of microplastics or horizontal measures;
- After validation of the proposed options by the Commission, analyse the social, economic and environmental impacts of the selected options
- Compare the impacts of the options with the baseline to enable the selection of most efficient and effective options, on the basis of which the Commission might propose preferred action(s)/options at EU level.

1.1 Project Scope

1.1.1 Definition and scope of microplastics adopted for this study

'Microplastics' is a term commonly used to describe extremely small pieces of plastic debris in the environment resulting from the disposal and breakdown of products and waste materials. The concern about microplastics centres on their potential to cause harm to living organisms in the aquatic environment.

The following definition for microplastics is used in this study: **Synthetic polymer-based material that is not liquid or gas, in a size less than 5mm in all directions.** The source calculations in Section 2.1.2 consider that microplastics may contain non-polymeric additives, oils, fillers or other product aids. The mass of these inherent ingredients is included in the emission calculations because they form an inexorable part of the particle. However, external substances or materials attached to the outer surface of the microplastics during or after the use phase, such as road dirt to tyre wear particles, are not included in the emission calculations.

This study is concerned only with microplastics that are created during the lifecycle of a product through wear and tear or emitted through accidental spills. Other types of microplastics are out of scope and excluded from the analysis, examples being:

- microplastics that are an intentionally added[4] ingredient within a product or designed with the expectation that they could be emitted during their lifecycle.
- microplastics that may be generated as a result of poor or non-existent waste management or as a result of the degradation of larger plastic waste.

1.1.2 Geographical Scope

The geographical scope of this report is those countries in the European Union (EU) which may emit microplastics into the marine environment in one or more of the four seas under the following Regional Seas Conventions (RSC)[5];

- HELCOM—Baltic Sea
- Bucharest Convention—Black Sea
- Barcelona Convention—Mediterranean Sea
- OSPAR—North East Atlantic

Although there are countries outside the European Union (EU) that border these seas—and therefore may contribute to microplastics loading in their waters—this report focuses on actions that can be taken at EU level to reduce these emissions. Where data is available for countries bordering these seas, this is highlighted and incorporated (Iceland for fishing industry sources, for example). Where the term 'Europe' is used, this refers to the EU28 Member States as well as Norway and Switzerland unless otherwise stated.

[4] These are covered by the parallel study by Amec Foster Wheeler
[5] http://ec.europa.eu/environment/marine/international-cooperation/regional-sea-conventions/index_en.htm

2.0 Problem Definition

2.1 What is the Problem?

An overview of the current understanding around the general impacts of microplastics can be found in Appendix A.1.0. The following sections look at the presence and specific impacts of microplastics from different sources.

2.1.1 Influence of the source of microplastics on their impact

The input of microplastics into the aquatic environment is poorly quantified, but it is clear that several factors are involved. These include: land use, population density in river catchments and on the coast, the presence of ports, the sophistication of waste water collection and treatment, the presence and nature of specific land- and sea-based activities, seasonal variations in precipitation and the degree of urbanisation. In addition to the quantities released, an improved knowledge of the physical and chemical characteristics of different categories of microplastics is needed to provide a more reliable risk assessment of probable impact, and help guide future reduction measures. The following sections summarise some of the source categories considered, within the present study, of being significant in a European context. The categories reviewed are:

- Synthetic textiles
- Automotive Tyre and Brake Wear
- Paint flakes and coatings
- Pre-production plastics

2.1.1.1 Synthetic Textiles

Particle characteristics

Textile are manufactured using pure forms and combinations of natural and synthetic fibres. Common natural fibres include cotton, linen (flax), silk and wool. Common synthetic fibres include acrylic, polyester/PET (e.g. Terylene™, Dacron™), polyamide (nylon), acetate, and PPT (poly-paraphenylene terephthalamide i.e. Kevlar™). Rayon (viscose, cellulosic) falls into a separate category of fibres of a natural origin that are modified chemically and are therefore referred to as 'man-made' or 'semi-synthetic'.

Polyester dominates the synthetics market. It is commonly combined with cotton for comfort and improved wear. Additive chemicals are frequently used, for example to increase UV resistance in Kevlar™ ropes, reduce flammability in clothing and furnishings, and impart different colours.[6]

Exposure pathways

The main human exposure pathway, in the context of the present study, is expected be from inadvertent ingestion in seafood. Exposure by inhalation may occur due to the re-suspension of dried shoreline or riverbank sediments. The main concern is likely to be from direct physical toxicity in the lungs, but no published data of the toxicity of textile fibres has been found during this study.

[6] Hermabessiere, L., Dehaut, A., Paul-Pont, I., Lacroix, C., Jezequel, R., Soudant, P., and Duflos, G. (2017) Occurrence and effects of plastic additives on marine environments and organisms: A review, *Chemosphere*, Vol.182, pp.781–793

Textile fibres may be released into the aquatic environment during use, or released during washing. It is this latter category that is considered the most significant by far (Hernandez et al. 2017), and has received most attention. Textile fibres have been reported as being widely distributed in freshwater and saline environments, in particular close to urban centres, in water, sediments and biota. Fibres, from unspecified sources, have been observed in many commercial species of fish and shellfish (GESAMP 2015, 2016,).

Fibres may enter the aquatic environment in wastewater, by indirect runoff or by atmospheric deposition. Natural fibres can be expected to degrade relatively rapidly under normal environmental conditions, whereas synthetic fibres will persist and hence become available for ingestion. Fibres have been observed widely in shoreline and seabed sediments, suspended particulate samples and many forms of biota, including invertebrates, fish and birds (GESAMP 2015, 2016). However, it may be difficult to establish the source. Airborne fibres are likely to be the source of fibres found in products such as beer and honey (Liebezeit and Leibexeit 2013), and have been reported to cross-contaminate environmental samples during handling and analysis of microplastics from aquatic systems (GESAMP 2016).

The greater propensity for fibres to be retained within organisms, compared with sub-spherical forms, combined with the common presence of additives such as flame-retardants and UV stabilisers, suggests this category of microplastics has the potential for causing harm. However, the risk remains unquantified.

2.1.1.2 Automotive Tyre and Brake Wear

Particle characteristics

The environmental science communities' recognition of the debris from vehicle tyres as a source of microplastics is comparatively recent.[7,8,9] In contrast, the importance of this vector as a significant source of particulate matter in the terrestrial environment has been well reported. For example, in a review of non-exhaust traffic related emissions literature by the JRC many studies were found that show vehicles emit several mg of particulate matter per km travelled[10]. It is thought that least 50% is contributed by non-exhaust emissions, including tyre wear and brake pad dust.[11] If exhaust gas particulate emissions controls reduce as planned, then the relative contribution of non-exhaust particulate per vehicle will increase significantly.[12]

---

[7] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

[8] Verschoor, A., and et al. (2014) *Quick scan and Prioritization of Microplastic Sources and Emissions*, Report for National Institute for Public Health and the Environment (RIVM) (Netherlands), 2014

[9] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

[10] Grigoratos, T., and Martini, G. (2014) Non-exhaust traffic related emissions. Brake and tyre wear PM, *JRC Scientific and Policy Reports*

[11] Bukowiecki, N., Lienemann, P., Hill, M., et al. (2010) PM10 emission factors for non-exhaust particles generated by road traffic in an urban street canyon and along a freeway in Switzerland, *Atmospheric Environment*, Vol.44, No.19, pp.2330–2340

[12] Rexeis, M., and Hausberger, S. (2009) Trend of vehicle emission levels until 2020 ? Prognosis based on current vehicle measurements and future emission legislation, *Atmospheric Environment*, Vol.43, No.31, pp.4689–4698

Tyre debris is one components of road dust, and has been reported to contribute around 10-15% of this material in urban areas.[13] Road dust is characterised by relatively high concentrations of heavy metals, organic carbon (OC) and elemental carbon (EC), with associated health concerns. Tyre wear is related to vehicle speed, road surface roughness, braking and acceleration, and other factors that increase friction between the road surface and the tyre. Tyre wear is complex but can be expected to be higher in urban areas.[14,15]

Modern tyres are complex in construction and contain a large number of different compounds, including many types of rubber. Brake pads also consist of a mix of many materials, including polymers, metals and metallic compounds (Kazimirova et al. 2016).[16]

In the absence of accurate observations, it may be assumed that tyre debris will consist of a complex variety of polymers with associated combustion products, and in a variety of forms, including flakes, fibres and irregular fragments. The physical and chemical characterisation of wear fragments will be very challenging. However, it is likely that differences in the shape and composition of microplastics from this source will influence the environmental behaviour and the environmental and human impact.

It is also claimed by the European Tyre and Rubber Manufacturers' Association (ETRMA) that tyre wear is bound with road particles when emitted to become 'tyre and road wear particles' (TRWP). It is estimated that the particles contain around 50% tyre compound[17]. It is unclear whether the addition of asphalt material to the tyre particle will have a great or lesser, impact. However, it may have a greater propensity to settle into sediments due to being negatively buoyant. That being said, there is no current substantive evidence that all tyre particles are bound with road particles with independent research into this being inconclusive at this time.[18]

Exposure pathways

No study has been located that demonstrates the presence of microplastics from tyre debris, by direct visual observation, in the aquatic environment. Tyre debris has been identified as a significant component of road dust by chemical characterisation (Valotto et al. 2015). Road dust, containing tyre microplastics (with the attendant pollutant loading) may be re-suspended and transported in the atmosphere, washed into drains or simply run off onto the surrounding area.[19] Tyre fragments

---

[13] Harrison, R.M., Jones, A.M., Gietl, J., Yin, J., and Green, D.C. (2012) Estimation of the Contributions of Brake Dust, Tire Wear, and Resuspension to Nonexhaust Traffic Particles Derived from Atmospheric Measurements, *Environmental Science & Technology*, Vol.46, No.12, pp.6523–6529

[14] Amato, F., Pandolfi, M., Moreno, T., et al. (2011) Sources and variability of inhalable road dust particles in three European cities, *Atmospheric Environment*, Vol.45, No.37, pp.6777–6787

[15] Valotto, G., Rampazzo, G., Visin, F., et al. (2015) Environmental and traffic-related parameters affecting road dust composition: A multi-technique approach applied to Venice area (Italy), *Atmospheric Environment*, Vol.122, pp.596–608

[16] Kazimirova, A., Peikertova, P., Barancokova, M., et al. (2016) Automotive airborne brake wear debris nanoparticles and cytokinesis-block micronucleus assay in peripheral blood lymphocytes: A pilot study, *Environmental Research*, Vol.148, pp.443–449

[17] Kreider, M.L., Panko, J.M., McAtee, B.L., Sweet, L.I., and Finley, B.L. (2010) Physical and chemical characterization of tire-related particles: Comparison of particles generated using different methodologies, *Science of The Total Environment*, Vol.408, No.3, pp.652–659

[18] IVL Swedish Environmental Research Institute (2010) *Wear particles from road traffic - a filed, laboratory and modelling study*, June 2010

[19] Wijesiri, B., Egodawatta, P., McGree, J., and Goonetilleke, A. (2016) Understanding the uncertainty associated with particle-bound pollutant build-up and wash-off: A critical review, *Water Research*, Vol.101, pp.582–596

may pass directly into waterways and be transported to the sea, especially in coastal locations (Valetto et al. 2015), or collected and passed through wastewater treatment plants, the filtration efficiency of which can be expected to vary widely (HELCOM 2016).[20] Storm water is also an important pathway for tyre particles. A recently published report from Sweden[21] found that sediment in a storm water pump station contained an average 1,100 asphalt particles per kg dry weight. In northern Europe an additional route will be from snow that is collected from urban streets and dumped directly into coastal waters (HELCOM 2016). Seasonal changes in rainfall patterns will also affect the likelihood of tyre debris reaching the sea, especially due to episodic heavy rainfall.

It appears that the main exposure pathway to aquatic organisms will be through ingestion, and to humans through ingestion of contaminated foodstuffs and potentially through inhalation of re-suspended shoreline material. Most concern about non-exhaust particulate has focussed in the inhalation of fine particulate (PM10, PM2.5) and associated metals (e.g. Pb, Zn, Cu), particle-bound combustion products (PAHs) and other chemical species, rather than microplastics per se.[22] Brake dust has been shown to be both mutagenic and toxic, and the concentration of iron and copper in the dust may influence the toxicity by inducing oxidative stress.[23,24] In addition, brake dust contains nano-sized particles that have the potential to induce chromosome damage (Kazimirova et al. 2016).

2.1.1.3 Paint and Coatings Particle characteristics of road and maritime paints

Paints used for roads and marine applications are complex mixtures of polymers and ancillary compounds. Road markings typically utilise hot-melt paints, applied at temperature of around 180 – 200°C, and comprising approximately 15 – 25% binder and 75 – 85% filler.[25] The binder consists of synthetic resins to improve adhesion to the road surface, plasticizer and thermoplastic elastomers. The filler includes glass beads to improve reflectance and increase wear resistance, aggregates, extender (e.g. $CaCO_3$) and pigment. Pigments are added to improve visibility and indicate the purpose of the marking. White pigments include titanium dioxide (TiO_2) and zinc oxide (ZnO), while heat-yellowing lead is used for yellow paints. Titanium dioxide is favoured as it has high strength, opacity and UV resistance. The binder uses mixtures of co-polymers, 0 – 5% by weight, including PS, polybutadiene, polyisoprene and polyethylene/butylene (Conserva and Dupont, date unknown).

On the basis of these figures it appears that the composition of paint flakes from road markings may be dominated by non-plastic components, which might nevertheless be considered environmentally hazardous.

[20] HELCOM (2016) Report of the HELCOM stakeholder conference on marine litter, 9 March 2016

[21] Katja Norén, Kerstin Magnusson, and Fredrik Norén (2016) *Mikroskräp i inkommande och utgående renat avloppsvatten vid Arvidstorps reningsverk i Trollhättans kommun*, Report for IVL Svenska Miljöinstitutet, January 2016

[22] Pant, P., Baker, S.J., Shukla, A., Maikawa, C., Godri Pollitt, K.J., and Harrison, R.M. (2015) The PM10 fraction of road dust in the UK and India: Characterization, source profiles and oxidative potential, *Science of The Total Environment*, Vol.530–531, pp.445–452

[23] ibid

[24] Malachova, K., Kukutschova, J., Rybkova, Z., Sezimova, H., Placha, D., Cabanova, K., and Filip, P. (2016) Toxicity and mutagenicity of low-metallic automotive brake pad materials, *Ecotoxicology and Environmental Safety*, Vol.131, pp.37–44

[25] Conserva, V., and Dupont, M. (Unknown) Kraton polymers boost functional life of thermoplastic road marking paints

Ships and other marine structures made of metal, often have a coating of an epoxy-based paint, with a PUR-based overcoat, as epoxy is not resistant to ultra-violet light. PUR coatings are relatively easy to clean, which is an important consideration given the need to frequently remove biofouling. Anti-corrosive pigments are included to protect iron and steel. Traditionally red lead (Pb_3O_4) was used extensively, but this is being superseded by zinc phosphate, zinc chromate, zinc molybdate and barium metaborate, due to environmental concerns.[26] Coatings on ships need to have additional anti-fouling properties. Without this ships' hulls rapidly develop a biofilm and macroscopic organisms attach and grow. This results in a large increase in drag resistance, and hence fuel consumption. It also results in the hull transfer of non-indigenous organisms to locations where they can become invasive. For many years tributyltin (TBT) was used until evidence for the significant environmental impact, typically seen as inducing imposex in marine invertebrates, became overwhelming. A global ban on the application of TBT came into force in 2008, with European countries complying since 2003.[27] Since then attempts have focussed on finding alternatives that are as effective but with lower environmental consequences. Copper-based compounds have been used as the main alternatives but a great variety of metallic, non-metallic, polymeric and combination compounds have been utilised.[28]

Paint flakes from ships and other maritime vessels and structures will consist of a complex mix of polymers, anti-corrosive and anti-fouling compounds.

Exposure pathways

Paint flakes from road markings will be generated by general wear and tear, principally from vehicle tyres. They will then form an important component of road dust and enter the aquatic environment as described in an earlier. Exposure may occur through ingestion of paint flakes or other contaminants associated with the paint, such as metals. No data have been found to quantify this exposure pathway. Road marking flakes have been found in river sediments in the UK[29](at an average of 66 microplastic particles in every 100g of sediment). This highlights that rivers are not only a pathway to the marine environment, but also a sink for microplastics.

Paint flakes from maritime applications will tend to enter the aquatic environment directly, or via a short transfer from waterside facilities such as boatyard and dry docks. Vessels and other structures need to be re-coated periodically, often preceded by removal of existing coatings using abrasive powders. This can generate paint flakes that will readily leak into the aquatic environment.

The hulls of most vessels bear the scars of encounters with harbour walls, other vessels or buoys. This persistent wear and tear will cause the paint to deteriorate and flake. It is instructive to note

---

[26] New Zealand Institute of Chemistry (Unknown) *Paints and polymers*, Unknown, http://www.nzic.org.nz/ChemProcesses/polymers/10D.pdf

[27] Filipkowska, A., Z?och, I., Wawrzyniak-Wydrowska, B., and Kowalewska, G. (2016) Organotins in fish muscle and liver from the Polish coast of the Baltic Sea: Is the total ban successful?, *Marine Pollution Bulletin*, Vol.111, Nos.1–2, pp.493–499

[28] Tornero, V., and Hanke, G. (2016) Chemical contaminants entering the marine environment from sea-based sources: A review with a focus on European seas, *Marine Pollution Bulletin*, Vol.112, Nos.1–2, pp.17–38

[29] Horton, A.A., Svendsen, C., Williams, R.J., Spurgeon, D.J., and Lahive, E. (2017) Large microplastic particles in sediments of tributaries of the River Thames, UK – Abundance, sources and methods for effective quantification, *Marine Pollution Bulletin*, Vol.114, No.1, pp.218–226

that paint flakes observed in Manta trawl samples examined for the presence of microplastics originated from the two survey vessels involved[30].

A very clear example of the impact of maritime paint comes from the environmental response to the use of TBT, with widespread occurrence of imposex induction in invertebrates due to endocrine disruption. The impact of TBT is still being reported in European waters, some years after its use was discontinued (Filipkowski et al. 2016). One reason for its persistence is thought to be the presence of paint flakes in contaminated sediments, especially in harbours, marinas and around dry docks. It seems reasonable to assume that this exposure route will also be relevant for the later generations of maritime coatings.

Exposure to maritime paint flakes may occur through ingestion of particles, or may result in exposure due to the leaching of components in the paint, such as biocides, which have been shown to cause significant environmental harm.

2.1.1.4 Pre-Production Plastics

Particle characterisation

Pre-production plastic pellets are typically spherical or cylindrical, around 5mm in diameter, and are the form that polymers are produced and transported for use in the plastics manufacturing sector. The composition of these 'pellets' found in the environment reflects the production volumes of different types of polymer.

Exposure pathways

Plastics production and use is a global industry. Significant quantities of plastic resin pellets are shipped around the world. Accidental spillages are common, at sea (e.g. lost containers), in ports and around plastics manufacturing facilities. Significant quantities of pellets have been reported from the Danube, Rhine and Scheldt rivers (GESAMP, 2016). Pellets are frequently found as a component of shoreline microplastic surveys, and in biological specimens such as birds. Their initial large size, and spherical or cylindrical shape, will limit their propensity for being consumed and causing physical toxicity. As the pellets gradually become smaller, such as by abrasion on a river bed or shoreline, then the potential for ingestion is likely to increase.

Most plastics will absorb hydrophobic organic contaminants, and resin pellets have been used widely as a passive sampler of environmental contamination (GESAMP, 2015).

2.1.1.5 Conclusions

It is apparent that the term microplastics covers a wide range of particle sizes, shapes and chemical characteristics. Some major sources are characterised by distinct particle types, for example synthetic fibres from textiles and flakes from paints and protective coatings. However, in other instances there may be no one dominant particle type, as may be the case for tyre debris.

It is clear that fibres and sub-spherical particles will behave differently, in terms of their fate in the aquatic environment and in the nature and degree of the toxicological and ecotoxicological response. Potential chemical effects appear to be more likely following exposure to nano- and micro- particles derived from textile fibres, vehicle tyres, paint flakes and fragments of durable plastics due to the common inclusion of modifying chemicals with known toxic properties. A summary of the categories of microplastic by source, shape, composition and potential impact is

[30] Thomas Maes pers. comm., Cefas UK; Kara Lavender Law pers.comm., SEA USA

provided in Table 34, indicating estimated relative importance and confidence level. This table is incomplete but may serve to stimulate additional expert input. Additional information on the potential for ecosystem and human impacts of plastic-associated contaminants in marine biota and seafood will become available on publication of the FAO (in press) report[31].

On the basis of available published evidence in is not possible to quantify the risk from microplastic exposure to the human population or aquatic environment with a reasonable degree of certainty. There are significant gaps in knowledge, in particular concerning the presence and impact of micro-fibres and nano-size particulates in environmental settings. A better understanding and description of the many diverse sources of microplastics, combined with a better understanding of exposure pathways, will allow a more robust estimate of the probability of harm to the human population and the overall health of aquatic ecosystems. In the absence of direct evidence, it is necessary to use a risk-based approach, utilizing expertise from other disciplines such as medicine, pharmacology, toxicology, materials science and nano-sciences. This approach is further explored in the partner study by Amec Foster Wheeler. For policy making, the Precautionary Principal[32] is the key strategy used to cope with possible risks where scientific understanding is currently incomplete—the implications of this in the context of microplastics is discussed further in Section 5.1.

2.1.2 Who is Impacted?

Whilst there are significant concerns around the suspected environmental impacts of microplastics in the marine (and other) environments there are also wider impacts to consider which have yet to be studied in detail.

The full consequences of any potential long-term impacts at a species level are yet to be established but these could have social impacts as well as human health impacts (health impacts are described in Appendix A.1.4.1). A negative perception of seafood for health reasons could have a large impact on industries and countries that rely on fishing for income. The public response may or may not be proportional to the actual risk. There also may be a danger of fish population reductions which would also affect fisheries negatively; this is unlikely in the short term, however.

Tourism can be impacted in areas that are prone to accumulation of microplastic debris. Small islands that are directly in the path of strong currents often find their beaches covered in micro and macro plastics that were not emitted in the locality. The Canary Islands are one such example[33] of this, where a large proportion of the sand can often be comprised of microplastics. With constant accumulation from prevailing currents it is almost impossible to maintain a clean beach. This can have negative impacts on tourism for these places. Again, this magnitude of this problem has not been studied in any detail.

Other impacts associated with negative public perception may also develop over time. Consumers may begin to choose alternative products that reduce microplastics emissions. For example, choosing cotton for their garments rather than synthetic based fibres. Although this sort of widespread shift has yet to be seen it would have a large impact on the textiles industry. The knock-on impacts are also not yet established as there are other negative environmental impacts associated with, for example, cotton, such as water and land use—comparing microplastics impacts

---

[31] FAO, and UNEP (2017) *Microplastics in fisheries and aquaculture*, 2017, http://www.fao.org/3/a-i7677e.pdf
[32] http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=LEGISSUM%3Al32042
[33] Herrera, A., Asensio, M., Martínez, I., Santana, A., Packard, T., and Gómez, M. (2017) Microplastic and tar pollution on three Canary Islands beaches: An annual study, *Marine Pollution Bulletin*

with wider environmental impacts is almost impossible at this stage so identifying alternatives is fraught with uncertainty. Nevertheless, it is possible that public perception could lead to unexpected trade-offs in the future.

2.2 What are the Causes?

Although microplastics have been found in the environment for many years it is only since 2014 that attempts have been made to fully identify and quantify the sources. To date, five country level studies (from Norway[34], Germany[35], Denmark[36], Sweden[37] and the Netherlands[38]) have been conducted within Europe. These attempt to identify and quantify microplastics sources and emission pathways. There has also been one global level study from IUCN[39] and two European level studies; one by Eunomia[40] for the European Commission and one from OSPAR (yet to be published).

Analysis of these reports (shown in detail in Appendix A.2.0) shows that between them these studies cover all of the main sources of microplastics that are known currently. Based on this, the following products are identified as further priority investigation for this project as they are thought to be the largest sources that are within scope of this project;

- **Automotive tyres**
- **Pre-production plastic**
- **Synthetic Clothing**
- **Artificial sports turf**
- **Building paints**
- **Marine paints, and**
- **Road markings.**

Appendix A.2.0 also shows the long list of all sources of microplastics that have been identified. During the investigations for this study, further sources were also identified and quantified as they have either hitherto remained unquantified, or indications are that the source may be significant. These are:

- **Fishing and aquaculture** (quantified for Sweden)
- **Automotive brake dust** (not previously quantified)

A quantitative exercise was carried out for all of the identified sources—the source generation of microplastics before they are distributed into various environments. Where a range has been

[34] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

[35] Roland Essel, and et al. (2014) *Sources of microplastics relevant to marine protection*, Report for Federal Environment Agency (Germany), November 2014

[36] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

[37] Kerstin Magnusson, and et al. (2016) *Swedish sources and pathways for microplastics to the marine environment*, Report for Swedish Environmental Protection Agency, March 2016

[38] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

[39] IUCN (2017) *Primary microplastics in the oceans.pdf*, 2017

[40] Eunomia Research & Consulting (2016) *Study to support the development of measures to combat a range of marine litter sources*, Report for European Commission DG Environment, 2016

estimated, the mid-point is displayed. These figures are taken forward to the pathways analysis in Section 2.2.9. Tables showing the full results are available in Appendix A.3.8.7.

As part of this exercise, seven pathways were characterised for the expected places the microplastics will be deposited as they are generated;

- **Residential Sewerage**—entering foul water drains from residential and light commercial areas.
- **Urban Non-road Drains**—entering residential external drains not connected to roads.
- **Urban Roads**—deposited on roads classified as being in an urban environment
- **Rural Roads**— deposited on roads classified as being in a rural environment.
- **Highways**— deposited on roads classified as higher capacity arterial roads (motorways in the UK and autobahn in Germany for example)
- **Direct to Surface Water—**deposited directly into nearby surface waters.
- **Direct to Soil—**deposited directly on nearby soils.

The following sections discuss each of the emission sources in turn.

2.2.1 Automotive Tyres

This section provides a summary of the key points of the analysis of the quantity of microplastics generated in the EU28 by automotive tyres. The full literature review and calculation methodology is provided in Appendix A.3.2. Previous calculations of tyre wear-derived microplastic emissions at source are reviewed. The way in which these emissions have previously been divided amongst environmental compartments is analysed and updated estimates are calculated.

The chosen calculation approach estimates tyre wear emissions at source, whereby traffic activity data expressed as vehicle kilometres (vkm) is multiplied by a grams per-vkm rate of wear to calculate overall emissions of tyre particles. For eight Member States plus Norway[41] 2012 traffic data disaggregated by vehicle type was obtained from either Eurostat[42] or national data archives.[43] For the remaining Member States national total vehicle fleet traffic activity were retrieved from the OECD[44]. Then, to disaggregate this total traffic activity by vehicle type, the data were scaled using national Tyre-Sales data from 2016[45].

To derive wear deposited on urban, rural and highway roads, and to facilitate a more powerful environmental pathways analysis for tyre-derived microplastics, the wear rates presented in a 2016 guide from the Netherlands National Water Board (Water Unit)[46] were applied to the traffic data disaggregated by vehicle type and road type. Table 52 provides an estimate of the total tyre wear deposited on different European road types based on these data.

---

[41] France, Germany, Hungary, Latvia, the Netherlands, Poland, Romania, and the UK.

[42] Eurostat (2016) Road traffic on national territory by type of vehicle and type of road (million Vkm)

[43] Klein, J., Hulskotte, J., van Duynhoven, N., Hensema, A., and Broekhuizen, D. (2016) Methods for calculating the emissions of transport in the Netherlands

[44] OECD (2013) *"Road traffic, vehicles and networks", in Environment at a Glance 2013: OECD Indicators*, 2013

[45] ETRMA (2016) *European Tyre & Rubber Industry Statistics Edition*, 2016

[46] Deltares, and TNO Consulting (2014) *Emissieschattingen Diffuse bronnen Emissieregistratie: Bandenslijtage wegverkeer*, Report for Rijkswaterstaat - WVL, May 2014

Table 5: Application Wear Rates to Traffic Activity Data

European Traffic Activity by Vehicle Type (Millions of Vehicle Kilometres)						
Vehicle type	**Motorcycles and mopeds**	**Passenger cars**	**Buses**	**Goods vehicles <= 3.5 tonnes**	**Lorries**	**Total**
Highway	11,517	598,725	2,520	78,247	38,684	**729,693**
Urban	51,321	1,043,136	7,119	116,174	49,702	**1,267,451**
Rural	60,080	1,384,005	7,311	162,448	112,423	**1,726,267**
Wear Rates (g vkm-3)						
Highway	0.047	0.104	0.326	0.125	0.668	
Urban	0.060	0.132	0.415	0.159	0.850	
Rural	0.039	0.085	0.267	0.102	0.546	
Tyre Wear Emitted (Tonnes)						
Highway	541	62,267	822	9,781	25,841	**99,252**
Urban	3,079	137,694	2,955	18,472	42,246	**204,446**
Rural	2,343	117,640	1,952	16,570	61,383	**199,888**
Total	**5,964**	**317,602**	**5,728**	**44,822**	**129,470**	**503,586**

The ETRMA[47] also provided upper and lower bound tyre wear rates based on their current best expert judgement. These figures were not used in the final calculations, but applied to verify the accuracy of the results in Table 10. This total deposited tyre wear figure (503,586 tonnes) is not dissimilar to that calculated using the midpoint of the ETRMA wear rates (572,157 tonnes). It is therefore believed to represent a reasonable working estimate of the total deposited tyre wear, and it is this data that will be carried forward for pathways modelling as it is disaggregated by urban, rural and highway deposition—each road environment is likely to have different pathways to the aquatic environment.

This figure should not be confused with Tyre and Road Wear Particles (TRWP) which, as identified in Section 2.1.1.2 and claimed by the ETRMA, constitute an equal mix of tyre wear particles and particles worn from the road surface. As it is unclear currently whether the wear from roads should be included in the total figure (and the evidence for the ETRMA's claim not substantiated as yet), it is excluded for the purposes of this study. However, if the definition of TRWP is used, emissions would double to over 1 million tonnes.

Total microplastics generated from the wear of automotive tyres in the EU

503,586 tonnes per year

[47] Personal Communications with ETRMA (2017)

2.2.2 Automotive Brake Wear

The linings of vehicle brakes are worn by use. These linings are composed of binders, fibres, filler and friction modifiers which can include natural rubber and resins produced using synthetic polymers.[48] Particles produced through the wear of these materials that meet other criteria for definition as microplastics are therefore within the scope of this study.

The same approach as tyre wear can be used to estimate the quantity of microplastics generated. This involves use of traffic activity data (see Appendix A.3.2) and per-vehicle kilometre (vkm) wear rates to estimate emissions at source. These are also split by urban, rural and highway in the same way.

Per-kilometre wear rates were collected from a literature review of primary experimental research for passenger cars, light goods vehicle and lorries.[49] For light goods vehicles an average figure was available which was scaled by 25% up and down to arrive at lower and upper bound estimates respectively. For passenger cars and lorries, upper and lower bound estimates were averaged to arrive at an estimated midpoint rate. These derived wear rates were applied to the aforementioned traffic activity data. Although bus and motorcycle wear rates were not available (and are therefore excluded from the analysis) they only represent 3.8% of total European annual vehicle kilometres and so their impact on estimated emissions at source is unlikely to be significant.

The methodology of the European Environment Agency for estimating emissions of air pollutants[50] indicates that 50% of brake emissions are typically captured in and around the vehicle body. Estimates of the coarse fraction of brake wear were derived from a recent literature review of primary research.[51] There is some uncertainty as to the fraction of wear which is coarse and, as such, upper and lower estimates from literature cited by the review of 2% and 38% have been applied.

These assumptions are applied to the activity data to arrive at an estimated ***source emission of between 505 and 17,161 tonnes.***

Total microplastics generated from the wear of automotive brake wear in Europe--

505—17,161 tonnes per year

[48] Luhana et al. (2004) Characterisation of exhaust particulate emissions from road vehicles - Measurement of non-exhaust particulate matter

[49] Luhana et al. (2004) Characterisation of exhaust particulate emissions from road vehicles - Measurement of non-exhaust particulate matter

[50] Ntziachristos, L., and Boulter, P. (2016) *European Environment Agency - EMEP/EEA Air Pollutant Emission Inventory Guidebook - 1.A.3.b.vi-vii Road tyre and brake wear*, accessed 16 March 2017,

[51] Ntziachristos, L., and Boulter, P. (2016) *European Environment Agency - EMEP/EEA Air Pollutant Emission Inventory Guidebook - 1.A.3.b.vi-vii Road tyre and brake wear*, accessed 16 March 2017, http://www.eea.europa.eu/publications/emep-eea-guidebook-2016/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/1-a-3-b-vi

2.2.3 Pre-Production Plastics

Companies manufacturing plastic goods use a feedstock of plastic material which is then melted and formed into plastic products. The feedstock typically comes in the form of small pellets, although flake and powder forms are sometimes used. The pellets are very distinctive and easy to identify – and so when pellets are found in the marine environment, as they often are on beaches worldwide, there is little doubt that these originate from the plastics manufacturing supply chain. The issue of plastic pellets as marine litter has been raised as a priority by a number of prominent NGOs but there have been few rigorous studies to understand the source of the problem in detail.[52] Whilst the public campaigns focus on the pellet form it is likely that the flake and powder forms of pre-production plastics are also entering the marine environment through the same pathways, but they are harder to distinguish from other plastic particles and so are not so well documented.

Appendix A.3.6 contains further information on pre-production plastics.

Efforts to quantify the amount of pre-production plastics entering the marine environment typically apply a 'loss rate' to the quantity of this material handled. Robust empirical evidence to inform a 'loss rate' is scarce. Loss rates are likely to vary widely from facility to facility, and from region to region, and so establishing an accurate loss rate for the EU based on observed losses would be a considerable undertaking.

The loss rates used in previous studies range from 0.000003% to 1%, although for some there is little basis for the numbers used. [53] Previous studies also suffer methodological shortcomings in that a loss rate, typically representing losses at one facility, is applied to the total amount of pre-production plastics created or consumed. This implies that there is only one point of loss in the supply of this material. In fact, losses can occur at any point in the value chain and so each group of players must be considered independently and calculations should take account of the number of times that the material is handled.

The companies handling pre-production plastics can be categorised as follows:

- **Producers** who create the plastics material from oil, gas and other raw materials;
- **Intermediary facilities** that handle the material between the producer and processor, including compounders and master batch makers who make specialist mixes of plastics and additives, distributors, storage facilities;
- **Processors** who convert the pre-production plastics into manufactured products;
- **Off-site waste management** who handle commercial waste from the categories of company above; and
- **Shipping companies** who transport the material on boats.

Terrestrial logistics companies are employed to transport the material between locations in the supply chain. However, losses are understood to predominantly happen when the material is spilt during handling. Spills have been documented when loading and unloading material from trucks and

[52] See campaigns of Fauna and Flora International: http://www.fauna-flora.org/initiatives/reducing-plastic-pellet-loss/, Fidra: http://www.nurdlehunt.org.uk/, Surfers Against Sewage: https://www.sas.org.uk/campaign/mermaids-tears/, as well as the European Coalition to End Plastic Pellet Loss which is comprised of 13 non-governmental organisations (Fauna & Flora International [FFI], Fidra [FID], Plastic Soup Foundation [PSF], S.O.S. Mal de Seine [MDS], Norges Naturvernforbundet [NN], Seas At Risk [SAR], SurfRider Europe [SRE], Plastic Change [PCH], Legambiente [LEG], North Sea Foundation [NSF], Marine Conservation Society [MCS], Zero Waste Europe [ZWE] and Environmental Investigation Agency [EIA]).

[53] See summary of loss rates in Appendix A.3.6

rail but as these activities take place within the grounds of the companies listed above there is no need to include a separate category for transport in the calculations. However, transport companies are part of the supply chain, and that spills undoubtedly take place in their business operations and so they must be considered in any actions taken to address this issue.

Table 6 shows estimates of the losses of pre-production plastics in the EU. The basis for these calculations is outlined in Appendix A.3.6.

The upper estimate has been revised downwards (by around 30,000 tonnes) since the interim report as the number of intermediary facilities (i.e. handling stages) has been reduced from five to four based on conversations with logistics providers. The handling stages are expected to be an average of two for bulk transported product. Consultations did not yield an estimate for the smaller quantities, but it is expected to be much higher as the pellets are bagged from larger bulk material. Despite extensive talks with plastics industry stakeholders, no further data could be provided that would improve the accuracy of the figures presented in Table 6.

Table 6: Annual losses of pre-production plastics

	Material handled (tonnes)	Loss rate	Quantity lost (tonnes)
Producers	58,000,000[a] – 70,565,000[b]	0.010% - 0.040%[c]	5,800 - 28,226
Recyclers	6,896,340 – 7,662,600	0.010% - 0.040%[c]	690 – 3,065
Intermediary Facilities	52,925,399 – 265,026,636[d]	0.010% - 0.040%[c]	5,293 – 106,011
Processors	48,563,380[a] – 66,776,366[e]	0.010% - 0.040%[c]	4,856 – 26,711
Offsite Waste Management	1,079,950 – 7,984,111	0.010% - 0.040%[c]	108 – 3,194
Shipping	10,082,674[g]	0.001% - 0.002%[h]	141 - 225
Total			**16,888 – 167,431**

Notes:

a) **From Plastics Europe (2016) *Plastics – the Facts 2015: An analysis of European plastics production, demand and waste data*. Includes CH and NO.**

b) **From Eurostat External Trade Database (EASY COMEXT Interface) data, 2015**

c) **From Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015**, and Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014.

d) **Based on Plastics Europe (2016) *Plastics – the Facts 2015: An analysis of European plastics production, demand and waste data*, and European Commission Trade Export Helpdesk Statistics accessible. Lower estimate assumes the material is handled once, the upper estimate assumes the material is handled four times.**

e) **Based on Eurostat External Trade Database (EASY COMEXT Interface) data, 2015 and European Commission Trade Export Helpdesk Statistics at**

f) **Using material handled tonnages for producers, recyclers, intermediary facilities and processors and an estimate of the proportion of feedstock as waste from Eunomia (2016), *Report for Fidra on Study to Quantify Pellet Emissions in the UK*, March 2016.)—0.6% of material handled for low and 1.9% for high estimate.**

g) **Based on European Commission Trade Export Helpdesk Statistics accessible and Eurostat News release 184/2016, Half of EU trade in goods is carried by sea, 28 September 2016,**

h) **Based on Marine Insight (2014) *Survey: How Many Containers are Lost at Sea?*,**

Total microplastics generated from the loss of plastic pellets

16,888 – 167,431 tonnes per year

2.2.3.1 Waste Water Treatment Plastic Media

Small plastic pellets, known as 'biobeads', have recently been identified as a source of microplastics. These have been found on and around a number of beaches in the South West of England. Information pertaining to spills of these products is provided in a report by UK pressure group, the Cornish Plastic Pollution Coalition.[54] These are placed under the plastics pellets section due to their similarities in size and appearance.

Biobeads are known to be used extensively in wastewater treatment (WWT) plants throughout the UK as a filter media. Their use (and that of other similar plastic media) throughout Europe is not currently confirmed, although they have been sightings in other EU countries.

They are released into WWT effluents either via catastrophic failure of the metal retaining mesh in one-off spills or via unknown mechanisms that see a slow trickle of biobeads released. Although the mesh is supposedly designed to retain 100% of the biobeads, there has been a number of sightings downstream of WWT plants. Their appearance (outside of reported spills) is yet to be adequately explained by either the WWT companies or the material providers. They are also known to be released into the environment via mismanagement during transportation and storage (in a similar manner to that described for plastic pellets).

Although little is known about the full extent to which these products enter the environment, there are several indicators (summarised in Table 7). If these were scaled up from UK estimates to the European level by population equivalents this could be around 1,200—5,000 tonnes lost annually (not including one-off spills). However, this is highly speculative as the use of such media throughout Europe is not confirmed and the high concentrations found on beaches appear to be mostly confined to the South West of England. This is possibly due to a huge 2010 spill in the local area whose effects are still being felt. These figures are highly speculative at present and none of the European water or wastewater associations was able to provide any more insight into this as a potential issue at the EU level, although it was identified that similar products are in use in limited amounts.

Table 7 – UK Biobead Losses

Loss Mechanism	Losses	Tonnes
Known Spills	5.45 billion	164[b]
Annual Losses	1%	154—600[c]

Notes:

a. **Two known quantified spills of 5.4 billion and 5 million biobeads**
b. **Assumed to be 4mm Dia Sphere of density 905kg/m^3**
c. **Lower limit = 94 confirmed reactors each with 5.4 billion biobeads x 1% annual loss. Upper limit = total sales into the UK annually (assumed loss rate= replenishment rate)**

Source: Derived from Cornish Plastic Pollution Coalition (2017)

[54] Cornish Plastic Pollution Coalition (2017) *Biobead pollution on our beaches. What we know so far…*, October 2017

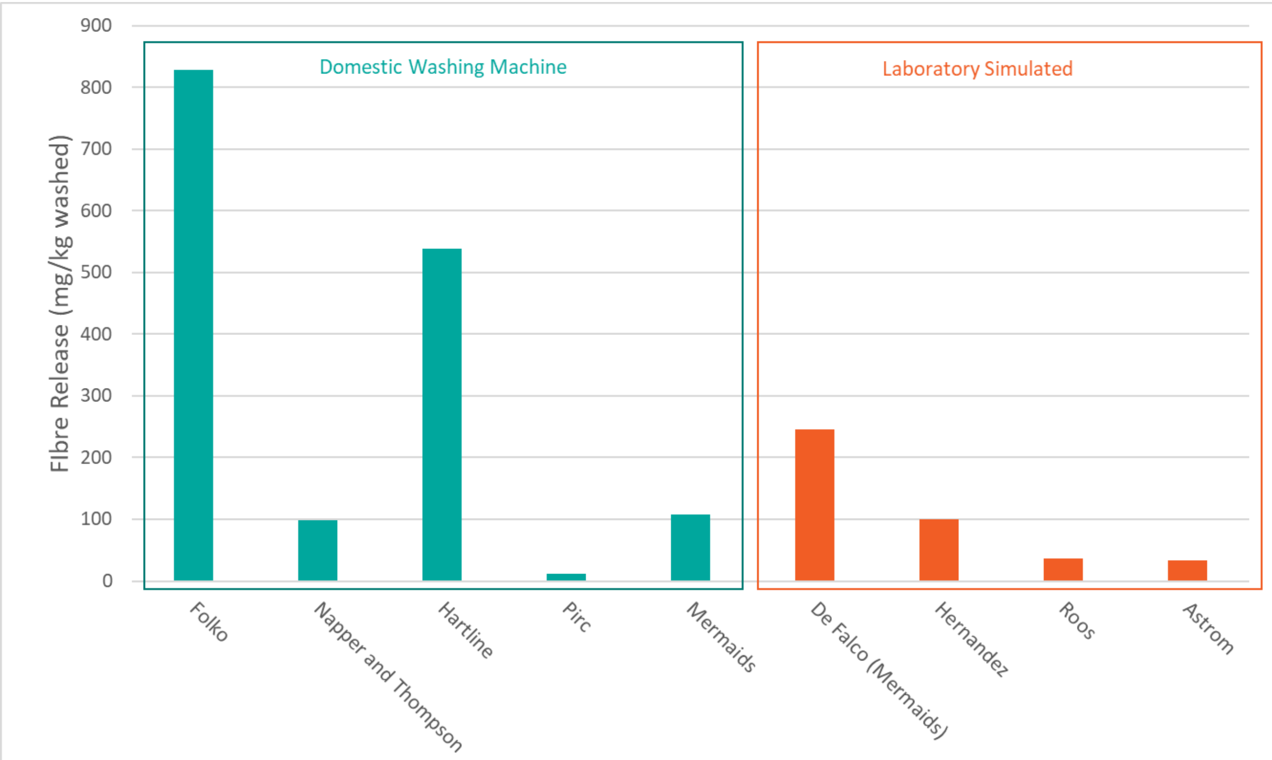
2.2.4 Synthetic Textiles

2.2.4.1 Clothing

There are now a range of studies which have attempted to quantify the release of microplastics from the washing of clothing. These studies can be broken into two main types. Firstly, there are the studies which, through their own experiments, have quantified the release of microfibres from textiles. Secondly, there are the studies which take these calculations and, by applying their own assumptions, attempt to upscale the results to national, international or global scales. A brief summary of the available literature and calculation assumptions is presented below – for the full literature review please see Appendix A.3.1.

Figure 3 shows a comparison between the fibre release studies that are publicly available as of November 2017. They are categorised by the washing apparatus used. Most of the more recent studies have been conducted in laboratory simulated conditions using test methods and equipment derived from colour fastness testing. Their applicability to real life fibre release is debatable (although the colour fast tests do have an ISO standard[55]), but these tests allow the isolation of specific factors that can affect fibre release and may be the basis for a standardised test.

Figure 3 – Polyester Fibre Release: Comparison of Study Results



Note: Where specific mg/kg release figures are not provided by the study they have been calculated separately. Where only fibre numbers and not mass is reported these are converted using a fibre length of 0.5mm and a Dtex of 1.1 resulting in a fibre weight of 0.8 micrograms. These values were chosen after consultation with textiles industry experts. Napper and Thompson release weight calculated using fibres/mg figures that are subject to an erratum to be published soon.

---

[55] https://www.iso.org/standard/51276.html

Upscaling the results of these tests to a European level is fraught with issues as none of the tests have recreated exactly what happens when a typical composition of clothing is washed under 'normal' circumstances. However, we can use the results to identify a likely range which will provide an idea of the magnitude of the issue.

The estimation of EU fibre release presented in this report has been updated from the calculations presented in the interim report. This is as a result of interactions with stakeholders from the textiles industry during stakeholder meetings and subsequent communications. The main concern with the results was the use of the assumption that fibre release would be in line with what was observed by Napper and Thompson[56]. This study found that fibre lengths were on average between five and seven millimetres in length based on the average length of only 10 sampled fibres. Applying this as conversion across all fibre releases may lead to a potential over estimation of fibre weight for those studies that only report numbers of fibres captured. An alternative approach was therefore adopted.

For the **upper** range of fibre release estimates the results of Hartline et al[57] (average of fibres released from aged and new clothing from a front-loading machine) are used to scale up to the European level. These results are chosen as they arguably represent a close approximation to a real-life scenario, as the whole garment was washed. The garments were also a polyester fleece which is thought to be particularly prone to fibre release, and therefore these results when scaled up would represent a reasonable ceiling value. Falko is not used as the results appear to suggest it is an outlier—there may have also been issues with controlling contamination.

For the **lower** range of fibre release estimates use is made of a journal article[58] published by the EU-funded Mermaids project[59] that summarises some of the project's results. The project produced a series of deliverable reports which, although they provide a significant amount of information and results, contain inconsistencies. The project team refused to acknowledge or address inconsistencies in the reports and therefore the only data that can be used is from the journal article.

The journal article only includes testing of polyester and polypropylene fabrics, whereas the full Mermaids project tested several other fabric types. Because of this, the scale of fibre release estimates are based on these two fabrics and scaled up for the other types. The article does find that—at least for polyester—woven fabrics release more fibres compared with knitted, however the released knitted fibres are thicker and longer and therefore are the mass of their release is greater. For the purposes of the current study the difference between woven and knitted fibre release is assumed to hold true for other fabrics, but this should be verified through comparative testing.

The results of this study also show large differences between the use of liquid and powder detergent; the latter being responsible for a greater release rate. As these two types of detergent

---

[56] Napper, I.E., and Thompson, R.C. (2016) Release of synthetic microplastic plastic fibres from domestic washing machines: Effects of fabric type and washing conditions, *Marine Pollution Bulletin*, Vol.112, Nos.1–2, pp.39–45

[57] Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., and Holden, P.A. (2016) Microfiber Masses Recovered from Conventional Machine Washing of New or Aged Garments, *Environmental Science & Technology*, Vol.50, No.21, pp.11532–11538

[58] De Falco, F., Gullo, M.P., Gentile, G., et al. (2017) Evaluation of microplastic release caused by textile washing processes of synthetic fabrics, *Environmental Pollution*

[59] http://life-mermaids.eu/en/

are used in roughly equal proportions throughout Europe[60], an average of the two release rates is taken to be representative.

The decision to base the lower range of fibre release on the results of De Falco et al. is because the data is provided is such a way that there are fewer assumptions needed to scale up. Very few studies accurately identify the size and shape of the released fibres, which is key to this exercise. As seen in Figure 3 the results are also in the same order of magnitude as the other laboratory simulations. As identified in the discussion in Appendix A.3.1, Pirc et al's[61] results seem excessively low, which may be the result of a fairly undemanding test. Indeed, its findings were consistent with those of De Falco et al. Both studies found polyester fibre release of 12mg/kg washed ***without detergent***, however De Falco et al. found fibre release to be up to 30 times higher when used with powdered detergent. Pirc et al. did not test with detergent, which is unlikely to reflect consumer behaviour.

Washing Load Composition

Assumptions have been made about the washing load composition and washing habits to upscale to the EU level. The Man Made Fibres Association assert that the presented composition of fibres are not representative of a typical washing load but could not substantiate this statement with further evidence or an alternative data source.

Data from a JRC report into the environmental improvement potential of textiles[62] splits EU sales of textiles by clothing type, fabric type and fabric construction. For the purpose of the calculations it is assumed that this percentage split of fabric types consumed represents the makeup of an average washing machine load in the EU.

The fibre type 'viscose' (often also known as rayon) is composed mainly of natural substances—primarily wood pulp— which are treated chemically to form artificially modified cellulose based fibres. This grey area between natural and synthetic makes it difficult to categorise the fibre as a microplastic. However, these fibres are suspected to persist in the marine environment despite their 'natural' origins[63,64]. Despite this, there are considerable uncertainties around the definitive identification of viscose[65]in environmental sampling as it is possible to be confused with natural cellulose fibres (cotton, flax, hemp etc.). This is part of the ongoing need for improvement in microplastic identification that was not necessarily the focus (or seen as important) at the time of some of the earlier sampling studies.

---

[60] AISE (International Association for Soaps, Detergents and Maintenance Products) (2014) AISE Consumers Habits Survey Summary

[61] U. Pirc, M. Vidmar, A. Mozer, and A. Kržan (2016) Emissions of microplastic fibers from microfiber fleece during domestic washing, *Environ Sci Pollut Res*

[62] JRC (2014) *Environmental Improvement Potential of Textiles (IMPRO-Textiles)*, Report for European Commission, January 2014

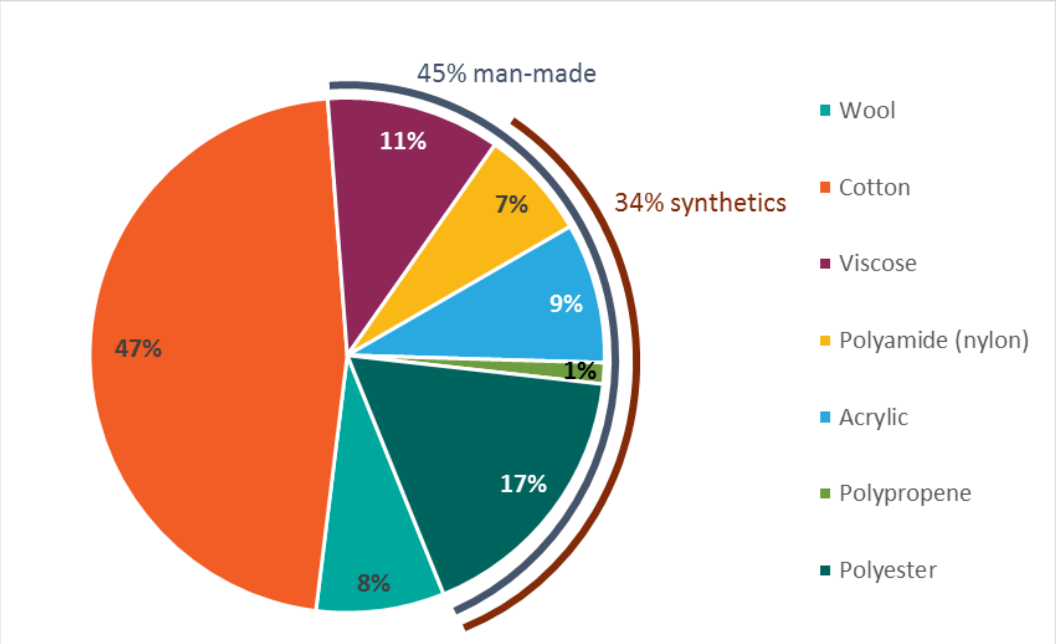
[63] Woodall, L.C., Sanchez-Vidal, A., Canals, M., et al. (2014) The deep sea is a major sink for microplastic debris, *Royal Society Open Science*, Vol.1, No.4, p.140317

[64] Lusher, A.L., McHugh, M., and Thompson, R.C. (2013) Occurrence of microplastics in the gastrointestinal tract of pelagic and demersal fish from the English Channel, *Marine Pollution Bulletin*, Vol.67, Nos.1–2, pp.94–99

[65] Comnea-Stancu, I.R., Wieland, K., Ramer, G., Schwaighofer, A., and Lendl, B. (2017) On the Identification of Rayon/Viscose as a Major Fraction of Microplastics in the Marine Environment: Discrimination between Natural and Manmade Cellulosic Fibers Using Fourier Transform Infrared Spectroscopy, *Applied Spectroscopy*, Vol.71, No.5, pp.939–950

Despite this uncertainty, cellulose-based fibres have been sampled in the environment and therefore specific claims that rayon is biodegradable in sea water are questionable. Sea water is the least aggressive environment for biodegradation and there is no current European or international standard for this. Some rayon products have been approved under the Vincotte OK Biodegradable marine label[66], but this is a private certification and largely based on the defunct ASTM D 7081 standard[67]. Despite the standard being withdrawn in 2014, no replacement has been developed—this demonstrates the difficulty in positively determining biodegradability in the marine environment. Testing to this withdrawn standard takes place in sea water at 30°C. This high temperature is used to accelerate testing, but there is no current agreed maximum threshold for the length of time it is acceptable for such materials to reside in the oceans. Claims for biodegradability in the marine environment should always, therefore be viewed with caution due to the lack of agreed standardisation. Based on this uncertainly it is important that viscose is not completely discounted until it can be ruled out as a persistent marine contaminant.

Figure 4 – European Clothing Sales by Fabric Type



Source: Derived from JRC (2014)

The release of fibres from the washing of viscose have been included in overall microplastics estimations and, as shown in Table 43 in Appendix A.3.1, these fibres account for 24% of 'man-made' fibre clothing sales—second only to polyester with 38%. They may therefore be a significant source of microplastics if viscose is categorised as such. Following this, it is assumed that an average wash load is made of 45% man made fabrics, 34% of these would be fully synthetic. A breakdown of the European clothing sales by fabric type is shown in Figure 4.

Fibre Releases per Wash

To represent the release from an average load of washing, the data for fibre release is combined with estimations of the average washing load composition. It is presumed that the average washing

---

[66] AIB-Vinçotte International (2017) *OK biodegradable Marine OK biodegradable Soil and OK biodegradable Water Conformity Marks*, 2017, http://www.okcompost.be/data/pdf-document/okb-mate.pdf
[67] https://www.astm.org/Standards/D7081.htm

machine capacity in the EU is 5.4kg[68]. Calculations for the *lower release* estimation are given in Table 8. The higher release is calculated by replacing the figures in the mass fibres released column with the ones calculated from Hartline (538 mg/kg).

The total release of fibres from the washing of an average load is therefore calculated at between 0.5 and 1.3g — or 3.2 to 17 million fibres.

Table 8 - Calculating the Release in an Average Washing Load (Lower Estimate)

Fabric Type and Construction		% of average load [a]	Mass (mg) fibres released/ kg washed [b]	Total mg released per kg washed	Release in average 5.4 kg wash (mg)
Viscose	W	3.7%	319	12	64
	K	7.2%	203	15	80
Polyamide	W	1.5%	242	4	20
	K	5.3%	154	8	45
Acrylic	W	0.3%	251	1	4
	K	8.4%	159	14	73
Polypropylene	W	0.2%	100	0.2	1
	K	1.2%	64	1	4
Polyester	W	8.4%	174	15	80
	K	8.6%	317	28	149
				Total release in 5.4kg wash (mg)	**520**
				Total release in 5.4kg wash (g)	**0.52**

Key:
W = Woven K = Knitted

Notes:
a) **Percentages taken from JRC (2014). See Table 43** in Appendix A.4.0**.**
b) **See Appendix A.4.0.**

Fibre Release at EU Level

To upscale the calculations to a European level (EU 28 countries plus Norway and Switzerland), Pakula and Stemminger's[69] data on the average number of wash cycles carried out per household

[68] Mermaids (2017) *Report on localization and estimation of laundry microplastics sources and on micro and nanoplastics present in washing wastewater effluents. Deliverable A1.*, May 2017
[69] Christiane Pakula, and Rainer Stamminger (2010) Electricity and water consumption for laundry washing by washing machine worldwide, *Energy Efficiency*, Vol.3, No.4, pp.365–382

per country was used, along with current figures on the number of households per country[70,71]. The Mermaids project also found that only 90% of European households have a washing machine and therefore they scaled accordingly; it is assumed that the remaining 10% of households attend commercial laundrettes. This provides the total number of domestic washes carried out each year by the 30 countries, a total of 33.8 billion (Appendix A.3.1.2, Table 46).

Consideration was also given to how consumers fill their washing machines. AISE's consumer survey[72] stated that 84% of people surveyed (approximately 200 from each of the 23 European countries taking part) wash with a full load. Following this, the assumption has been made that the remaining 16% of consumers are washing with a half load. The inclusion of this factor along with the multiplication of the estimated fibre release per wash and the number of washes gives an upper estimated release of ***40,530 tonnes per year and a lower estimate of 8,493 tonnes per year*** (See Table 9).

Table 9 – Fibre Release from Domestic Washing Machines in Europe

% of washing machine full	Full Load	Half Load
% of consumers [a]	84%	16%
Number of washes [b]	28.4 billion	2.7 billion
Releases per Wash (g) [c]	0.5—1.3	
Corresponding release of fibres (tonnes) Upper estimate	37,396	3,524
Corresponding release of fibres (tonnes) Lower estimate	14,771	1,407
Total release per year (tonnes) Upper estimate	**40,958**	
Total release per year (tonnes) Lower estimate	**16,177**	

Notes:
a) **AISE (2014)**
b) **Eurostat and UNECE with Mermaids assumption of 90% market penetration of washing machines.**
c) **See Appendix A.4.0.**

To account for commercial laundering, an additional 14% is added onto the release figures. This figure was calculated from data provided in Mermaids[73], which shows that around 25 million tonnes of commercial washing is carried out in Europe each year. This equates to an additional 14% of the

[70] Eurostat *Eurostat - Data Explorer. Number of private households by household composition, number of children and age of youngest child*, accessed 7 June 2017, http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=lfst_hhnhtych&lang=en
[71] UNECE *Private households by Household Type, Measurement, Country and Year*, accessed 7 June 2017, http://w3.unece.org/PXWeb2015PXWeb2015/pxweb/en/STAT/STAT__30-GE__02-Families_households/08_en_GEFHPrivHouse_r.px/
[72] AISE (International Association for Soaps, Detergents and Maintenance Products) (2014) AISE Consumers Habits Survey Summary
[73] Mermaids (2017) *Report on localization and estimation of laundry microplastics sources and on micro and nanoplastics present in washing wastewater effluents. Deliverable A1.*, May 2017

total washed domestically[74]. This assumption is similar to the 10% addition used in Lassen's[75] calculations.

The final estimated range for the microplastics generated from the washing of synthetic clothing is therefore ***18,430—46,175 tonnes per year.*** This equates to a release of between 100 and 600 quadrillion individual fibres (assuming 0.08µg per fibre).

Total fibres released from the washing of clothing in Europe:

18,430—46,175 tonnes per year

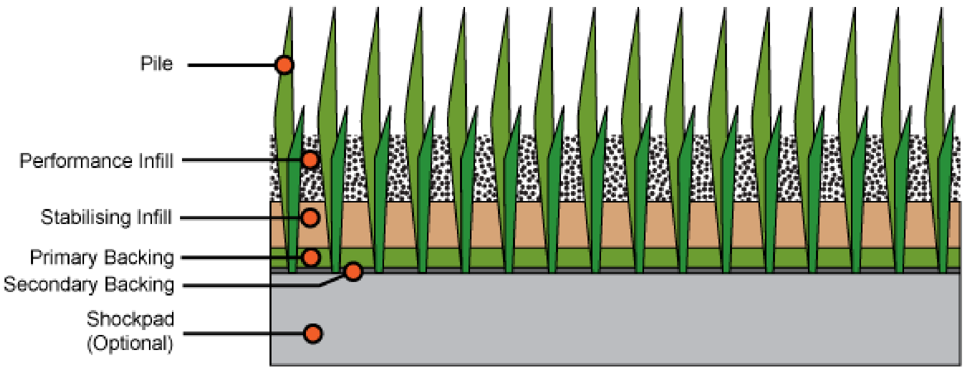
(between 100 and 600 quadrillion fibres)

2.2.5 Artificial Sports Turf

Artificial turf has many uses, although the main focus in this report is that used for sports (mostly contact sports) that include football (soccer), rugby and American football. All these sports require the turf to absorb impacts to help prevent injury and to mimic the feel of natural turf. This is usually achieved by the application of 'infill' material. This material is usually polymeric and in the form of small particles <5mm in size. It is distributed throughout the turf surface just below the artificial grass pile. Figure 5 shows a typical artificial turf composition of what is known as 3rd Generation (3G) turf design. The stabilising infill is usually sand, to help the pile to retain its shape. The performance infill is laid on top. Although the performance infill can be made from organic alternatives such as cork and coconut husk, the majority of the market uses rubber crumb from recycled tyres—often referred to simply as SBR (styrene-butadiene rubber) infill despite the original tyre composition often being more complex than containing purely SBR. As well as SBR, there are also virgin elastomer alternatives such as EDPM and TPE.

Artificial turf for domestic applications is unlikely to contain plastic infill material as it is both costly and unnecessary for this purpose. The pile fibres may wear or break and form microplastics, but this is expected to be minimal compared with sports turf that is subject to a great deal more abrasion. Non-contact sports such as hockey and tennis also have different requirements and cannot usually be played on 3G surfaces due to the need for much shorter pile. The use of 2nd Generation (2G) turf is more often used, which only incudes the stabilising sand layer.

Figure 5 – Typical 3G Turf Composition



[74] 33.8 million washes per year at 5.4kg each equals 182 million tonnes washed in households.
[75] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

With fibres ranging from 40—65 mm[76], the identified wear rate would equate ***to fibre losses of 0.5—0.8 % annually***—around 10 times less than estimated in the Danish study. The pile loss is calculated based upon an average pile weight of 1.4 kg/m^2 from FIFA data[77] one over 3,000 pitch installations (this is compared with 0.8 kg/m2 from the Danish study).

Results from FIFA data[78] on installed football turf showed that SBR is installed at an average density of 16 kg/m^2. This means that there is around 120 tonnes of SBR in each full size pitch (106x71m[79]). A loss of between 1.5 and 5 tonnes per year (the upper and lower estimates of previous studies identified in Appendix A.3.4.1) ***equates to between 1—4% of the total infill installed.*** This correlates with the amount of infill top-up that was commonly reported by turf manufacturers for this study—typically around 3% per year was stated.

At this point it is assumed that all infill used to top-up a pitch is to replace that which is lost to the environment. However, it is also the case the infill may be topped up as a result of compaction. This is a process by which the infill settles and is compacted through use. Infill can then be added on top to provide the correct level once more. However, Loughborough University have studied[80] this phenomenon and concluded in both laboratory and real-life experiments, that proper maintenance can reduce compaction almost entirely. Various de-compaction methods can be employed to disturb the infill and return it to its previous levels. This would suggest that 'losses' due to compaction are only apparent in pitches that are not well maintained. Data is not available for this and consequently it is difficult to estimate the true impact of compaction on potential loss calculations. However, the range of 1—4% losses is likely to incorporate the variety of maintenance that may be seen throughout Europe—1% losses, therefore representing pitches that continue good maintenance procedures.

The number of pitches installed was estimated by the European Synthetic Turf Organisation (ESTO) in a market report[81] provided to this study. The report surveyed football associations from across Europe to estimate the number of full sized and small training pitches installed in 2012. The survey also asked them to estimate the number that will be built by 2020. Although not all FAs were able to estimate this future scenario, data was extrapolated from these few across Europe. Estimates for pitch numbers are found in A.3.4.4. An estimate for installed rugby pitches was also provided in the ESTO report for Europe as a whole. Although artificial turf use in rugby is growing fast, it currently only represents 2% by surface area installed.

It is estimated that a total of 51,616 pitches exist in Europe with an installed area of 112 million square meters. Using the infill density of 16.1 kg/m2 the total infill estimated to be installed in Europe is 1.8 million tonnes.

Table 10 shows the infill loss rate applied to the total installed infill which shows that between ***18,000 and 72,000 tonnes could be lost per year***. The pathways and sinks for infill are different to

---

[76] Confidential data provided by FIFA.
[77] ibid
[78] ibid
[79] The FA Guide to Artificial Grass Pitches, May 2010
[80] Fleming, P.R., Forrester, S.E., and McLaren, N.J. (2015) Understanding the effects of decompaction maintenance on the infill state and play performance of third-generation artificial grass pitches, *Proceedings of the Institution of Mechanical Engineers, Part P: Journal of Sports Engineering and Technology*, Vol.229, No.3, pp.169–182
[81] ESTO (2016) *Market Report Vision 2020*, 2016

the other microplastics sources i.e. there is another step before they enter one of the pathways identified in Section 2.2.8. This intermediate pathway step is therefore discussed in this section of the report and the results fed into the pathways model.

There is currently only one study which has attempted to create a mass balance for infill in artificial turf. The study, from the Netherlands[82], looked at three local pitches containing SBR infill and one containing TPE. This study is discussed further in Appendix A.3.4.1. The results of the study are inconclusive and may not be representative however, some indicative release figures can be used to determine where the infill goes. Transport by players was calculated at around 4% of losses and releases to surface water were also around this level (2—3%, with one notable exception at 75%). One this basis, an estimate of 5% is used for both. The remainder is split evenly between soils/grass and waste disposal.

A further pathway for infill material of snow removal is also identified in Norway[83]. In England, the FA recommend that snow removal or playing is not undertaken[84]. However, this is not practical for countries that see snow cover for much of the winter— this includes much of Norway, Sweden and Finland as well as parts of Eastern Europe such as Estonia, Latvia, Lithuania and Poland. These countries account for around 15% of the installed turf (from Appendix A.3.4.4). This, therefore may be a significant issue in these countries, but not on a European scale.

Table 10 – Artificial Turf Infill Losses

		Lower	Upper
Infill Loss [b]	**%**	1%	4%
	Tonnes	18,026	72,105
Pathways and Sinks			
Waste Disposal	45%	**8,112**	**32,447**
Surface Drains [a]	5%	**901**	**3,605**
Internal drains [a]	5%	**901**	**3,605**
Soil/Grass	45%	**8,112**	**32,447**
Total	**100%**	**18,026**	**72,105**

Notes:

a) **Indicative figures derived from Annet Weijer, and Jochem Knol (2017). Internal drains are from player transported infill into either a changing room or to their homes. It is unclear how much of this would actually end up in the drains at present.**

b) **Losses from overall installed infill from Table 60**.

Total microplastics generated from artificial sports turf pitches-

18,000 – 72,000 tonnes per year

[82] Annet Weijer, and Jochem Knol (2017) *Verspreiding van infill en indicatieve massabalans*, Report for Branchevereniging Sport en Cultuurtechniek, May 2017

[83] Mepex (2016) *Primary microplastic- pollution: Measures and reduction potentials in Norway*, April 2016

[84] Personal communication with English FA.

2.2.6 Paints and Coatings

Paint and coatings microplastics generation have been identified for four sources;

- Building paints
- Marine paints
- Road markings; and
- Automotive paints.

The OECD emission scenario document on the coatings industry[85] was the only source of emissions factors for the wear and tear of all paint sources except road markings during the writing of the interim report. It has been used and referenced by several of the other country level microplastics emission studies as well as Eunomia's previous study.

Emission estimates have been revised from the ones given in the interim version of this report. This is based on new information provided in a report by CEPE[86] in response to data queries and technical questions posed by this study's authors. The cooperation of the paint industry is welcomed and has led to improved data and assumptions used in the revised calculations.

The basic approach by CEPE to calculation differ subtly from other attempts to quantify paint emissions using total paint sales volumes:

- For 'wear loss', calculations were based on the volumes applied as top-layer and therefore subject to weathering (primers and other base coats are excluded)
- For 'removal losses' the volumes that relate only to those surfaces that are subject to sanding or blasting.

The authors of this study concur with this approach which helps to narrow down the specific paints that may be causing microplastic pollution.

Paints are also considered in the parallel study by AMEC for intentionally added microplastics. As these are also covered in the present study it is important to make the distinction in the relative project scopes clear. The AMEC study sets out the following distinction:

> *"In case of solid plastics that are added to paints, varnishes, lacquers, and (powder) coatings it is possible that particles are emitted into the environment while a person applies the product onto a surface or afterwards by rinsing brushes and paint rolls.* ***Once the product layer is dried or cured, the particles should become embedded completely and become an integral part of the layer. Considering wear and tear over time, however, it is assumed that this product category can be a source for secondary microplastics."***

This present study deals only with the highlighted sentence.

2.2.6.1 Building Paints

Market segment calculations for the architectural/decorative paints market are shown in Appendix A.3.5.2.

---

[85] OECD (2009) *Emission Scenario Document On Coating Industry (Paints, Lacquers and Varnishes)*, 2009

[86] CEPE (2017) *Micro-plastics emitted from 'wear and tear' of dried paints. The view of the paint industry.*, September 2017

Interior Paints

For interior paints it is assumed that the only pathway to surface water is through the washing of brushes and paint rollers in sinks after use for water based paints. As the paint, in its 'wet' form is considered an 'intentionally added' microplastic for the purposes of this project, it is therefore out of scope. However, the emission is quantified to provide further context for the microplastics generated from wear (See Appendix A.3.5.2). This is estimated to be **around 3,500 tonnes per year washed into household drains**.

Exterior Paints

The losses during removal were originally derived in part from the OECD emissions factors for marine paint as none were given for removal of decorative paints. CEPE have since provided their own emission factor of 5% (adjusted to 4% in this study as only 80% of coatings are believed to be used for maintenance) from the sanding and removal of paint and have stated that there is no data on the use of extraction or capture systems for external building surfaces. Importantly, the largest market segment of wall paints is assumed by CEPE to not be a source of maintenance derived microplastics as these are not typically sanded. However, the scraping of peeling or cracking paint before applying new paint is likely to happen to a certain extent, therefore applying a zero emission factor to this is potentially underestimating the problem. Due to this uncertainly an emission factor of 1—4% is used for these paints.

Wear losses were also found in OECD emission report which estimate these to be 3% for building paints. This has been revised down to 2.5% by CEPE. Importantly CEPE also state that the polymer fraction of the paint will be subject to a chemical transformation by photolytical and hydrolytical degradation throughout the life of the coating and will finally be volatilized as carbon dioxide, water and nitrogen. This is estimated to remove 67% of the polymer although this only accounts for around 3,000 tonnes of the total particles released.

Table 11 shows how these emission factors are applied to the sales data which provides as estimated microplastics generation of ***21,100—34,900 tonnes per year*.**

To allow analysis of the pathways that these particles may take, this figure is further separated by the expected split between urban and rural deposition. Eurostat[87] estimate that 43% of the population of the EU live in urban areas. Whilst this does not necessarily mean that buildings are distributed in the same way, it is a useful basis for estimation. The Netherlands microplastics emissions study[88] calculated the urban/rural split directly from the distribution of housing within the Netherlands and classified 66% classified; Eurostat estimate the population of the Netherlands to be 73% in urban areas—a 7% difference. As housing density is often higher in urban areas this is to be expected and can be used to adjust the European figure accordingly. ***This provides a split of 39% urban and 61% rural***.

[87] Eurostat (2016) *Urban Europe Statistics on cities, towns and suburbs 2016 edition*, 2016

[88] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

Table 11 – Building Paint Lifetime Losses

Market	Solid Coating Applied (tonnes)	Polymer in Solids	Polymer degradation	Remaining Coating	Removal Losses		Wear Losses	
					%	tonnes	%	tonnes
Exterior Walls	458,600 [a]	20% [b]	67%	397,145	1—4 % [c]	4,600—18,300	2.5% [d]	9,900
Exterior wood, metals etc.	78,400 [a]	50% [b]	67%	52,122	4 % [c]	3,100	2.5% [d]	1,300
Exterior Wood Varnish	37,000 [a]	95% [b]	67%	13,449	4% [c]	1,500	5% [d]	672
Total	1,137,650			462,716		**9,200—23,000**		**11,900**

Notes:

a) **Calculated from market data in Appendix A.3.5 provided by CEPE.**
b) **CEPE (2017)**
c) **Removal losses from CEPE (2017) cited as 5%, but adjust to 4% as CEPE estimate that only 80% of coatings are used in maintenance applications.**
d) **CEPE (2017)**
e) **Numbers rounded therefore totals may not add up.**

Total microplastics generated from the wear of exterior building paints is-

21,100—34,900 tonnes per year

2.2.6.2 Marine Paints

Marine paints can be broadly split into two different types;

- Protective coatings; and
- Anti-fouling coatings.

Although there is no specific sale data publicly available that disaggregates the total marine paint sales by application (seen in Table 60 in Appendix A.3.5.3), the European Chemical Agency estimate that 25,000 tonnes[89] of anti-fouling paint are sold in the EU and the OECD[90] estimates that commercial and pleasure craft account for around 95% of total demand. This correlates somewhat with data supplied by CEPE on marine paint sales[91].

[89] European Chemicals Agency (2013) *ANNEX XV RESTRICTION: REPORT AMENDMENT TO A RESTRICTION: SUBSTANCE NAMES: CADMIUM AND ITS COMPOUNDS - Paints*, October 2013
[90] OECD (2005) *Emission Scenario Document on Antifouling Products*, 2005
[91] CEPE (2017) *Micro-plastics emitted from 'wear and tear' of dried paints. The view of the paint industry.*, September 2017

The OECD has produced two emission scenario reports that are relevant to this sector and form the only independent data sources available for the emissions of paint during maintenance, use and removal. The first 2009 OECD report[92] covers different paint applications and provides emission scenarios for protective (non-antifouling) paint. A second 2005 OECD report[93] focuses specifically on anti-fouling paint used in commercial and recreational craft. This report provides figures separately for commercial and recreational craft along with average and worst-case scenarios. However, the report does not provide emission factors for weathering during use or during the end of life. As the application of paint and its subsequent spillage directly in the sea is assumed to be considered a spill of 'intentionally added' microplastics these figures are also not used. Appendix A.3.5.3 describes a further calculation for this emission in order to put the wear-based emissions in context.

CEPE have provided their own emission factors[94] to this project which are used in preference to the OECD factors. These figures are applied to sales data also provided by CEPE (which is similar to data from other sources). The results are shown in Table 12. This provides an estimate of **emissions to surface water of 946 tonnes.** This is four times less than the results of the interim report. This is because CEPE has provided a more thorough breakdown of the uses of marine paint which cannot all be assigned the same emission factor. For example, wear from interior paints in not included which accounts for around 25% of marine paint sales. As detailed in the building paints section there is also an element of polymer degradation that reduces its availability to wear. All of the factors that contribute to this estimate are shown in more detail in Appendix A.3.5.3.

Table 12 – Emissions to Surface Water from Marine Paint

	Commercial (t)	Recreational (t)	Total (t)
Paint Applied-	76,440 [a]	14,560 [a]	91,000
Weathering	177	44	222
Sanding	422	550	972
Total Emissions to Water			**1,194**
Total Emissions to Water from 'intentionally added' [b]			**1,993—4,525**

Notes:

*a) **CEPE***

*b) **These are emissions of 'uncured' paint directly into the marine environment during application***

Total microplastics generated from wear of marine paints (direct to surface waters)-

1,194 tonnes per year

[92] OECD (2009) *Emission Scenario Document On Coating Industry (Paints, Lacquers and Varnishes)*, 2009
[93] OECD (2005) *Emission Scenario Document on Antifouling Products*, 2005
[94] CEPE (2017) *Micro-plastics emitted from 'wear and tear' of dried paints. The view of the paint industry.*, September 2017

2.2.6.3 Road Markings

The two most common road markings in Europe are solvent based—where the pigment and binder are suspended in an organic solvent—and thermoplastic markings—also known as 'hot melt' coatings—where heat is applied to increase the viscosity and allow the coating to be applied to a road surface before drying quickly. Water borne paints and cold plastic road markings are also used to a lesser extent.

As well as polymer binders, a large proportion of the coatings often comprises fillers that provide wear resistance (aggregates) and increase tyre grip and reflectiveness (glass beads). As per the definition used in this report, all ingredients additional to the polymer that make up the solid component of the material are considered to be microplastics when they are worn away.

Both previous microplastics source studies from Mepex[95] and Eunomia[96] assume that all paint use represents the paint that is worn away. Eunomia assigned a small amount to new roads. The Danish study[97] took a different approach by estimating the amount that is used in the reapplication of existing road markings (15—25%) and the amount for new and resurfaced roads (75-85%). The only other evidence found for the amount of paint used to renew existing lines is from another Okopol report which found that around 85% of road marking systems are used for re-painting in Germany[98]. These figures are confirmed by CEPE who state that 80% of road markings are applied for maintenance purposes.[99]

The amount worn away before repainting can be estimated by using guidelines for the renewal of road markings. In the UK the guidelines appear to vary depending upon the responsible authority. National guidance[100] for highways suggest that a visual wear limit of 70% is achieved before renewal. Several cities[101,102,103] specify that only 30% wear should be evident before renewal—reflecting the increased requirement for highly visible road markings in cities. There are obvious issues with this, as this is a very subjective approach. To combat this, the UK highways guidance has since been updated to use a visual scoring assessment to compare with example pictures. Nevertheless, these figures are useful indicators as to the likely wear that will occur before repainting and may even underestimate the wear due to several reports suggesting the condition of road markings throughout Europe is not satisfactory.

---

[95] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

[96] Eunomia Research & Consulting (2016) *Study to support the development of measures to combat a range of marine litter sources*, Report for European Commission DG Environment, 2016

[97] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

[98] Okopol (2009) *Implementation and Review of Directive 2004/42/EC - PART 1: MAIN REPORT, ANNEXES 1-25*, Report for European Commission, November 2009

[99] CEPE (2017) *Micro-plastics emitted from 'wear and tear' of dried paints. The view of the paint industry.*, September 2017

[100] The Highways Agency (2007) *Inspection and Maintenance of Road Markings and Road Studs on Motorways and All-Purpose Trunk Roads*, 2007

[101] http://www.wiltshire.gov.uk/highway-inspection-manual.pdf

[102] https://www.york.gov.uk/download/downloads/id/3326/annex_cpdf.pdf

[103] https://www.hackney.gov.uk/media/2771/highways-maintenance-policy/pdf/Highways-Maintenance-Policy

Table 13 shows the calculation for the emissions of microplastics from road paints ***at source*** are derived using the previously stated data and assumptions and sales data shown in Appendix A.3.5.4. This leads to an estimated emission of ***94,358 tonnes per year***.

Table 13 – Calculating Microplastics at Source from Road Markings

	Urban	Rural	Highway	Total Remaining
Used for Maintainance[1]	80%			**192,992**
Material Used[1]	97%			**187,202**
Polymer Decomposition[1]	See Appendix A.3.5.4			**173,452-**
Material Wear[3]	30%	70%	70%	**4,358**
Material Wear at Source	20,294	71,636	2,428	

Notes:
1. ***CEPE.***
2. ***From Eurostat road length data averaged for seven EU countries***
3. ***From guidance on wear observed before renewal from UK.***

Total microplastics generated from wear of road markings-

94,358 tonnes per year

2.2.7 Fishing Gear

A complete literature review and associated calculation tables can be found in Appendix A.3.7.

Research into microplastic formation from fishing gear during use is particular sparse as attention is usually focused on preventing or recovering lost fishing gear. Some research suggests that the quantity of microplastic released may not be that significant, at least in certain regions, as the products are replaced before they are too badly degraded.[104]

[104] Magnusson et al. (2016) Swedish sources and pathways for microplastics to the marine environment; after Sundt et al. (2014).

A recent study investigated degradation of polymer ropes at 10m depth in Scottish coastal waters.[105] A rate of mass loss of 0.4—1% per month was observed. The rate varied depending upon the polymer. The study indicates that polyethylene and polypropylene do not wear as much as nylon. However, the study emphasises that the degradation of marine plastics is highly dependent on the context in which they are found and so it would be unwise to assume that the results are representative of degradation of fishing and aquaculture gear in use. Further research is required in this area in order to inform estimates of microplastic emissions from these sources. Indeed, similar techniques could be applied to measure the rate of plastic degradation and emission of microplastic particles from fishing and aquaculture gear in use and establish the correlation with the principal degradation factors.

There is also a lack of data on the on fishing nets sold, used, discarded and lost. This is compounded by spurious statistics such as a 2009 FAO report[106] repeatedly being quoted as the saying that 640,000 tonnes of fishing gear are lost every year, when this refers to what is currently residing on the sea floor.

Despite this, an attempt has been made to ascertain the magnitude of this issue. Prodcom data suggests that 28,571 tonnes of fishing nets were used (sold minus exports plus imports) in the EU in 2015 (see Table 77 in Appendix A.3.7). Data for Norway and Iceland is incomplete, but if scaled by reported live catch weight, they account for a further 19,000 tonnes (see Table 76A.3.7).

Using the estimated loss rate from Sweden of 1—10%[107], a ***total loss of 478—4,780 tonnes*** direct to the ocean is therefore estimated. This estimate is highly speculative, and both the loss rate and the fishing net data are very uncertain at this stage.

Total microplastics generated from fishing net wear and emitted directly into oceans-

478—4,780 tonnes per year

2.2.8 Microplastic Pathways and Sinks

Once microplastics are released into the environment there are several pathways they can take towards the aquatic environment and several sinks on these pathways that can retain them. The following section details these and looks at the data that can be used to model the movement of microplastics through these pathways. There is still considerable uncertainty about how microplastics move around urban and rural environments, especially when they are not directly released into sewerage systems.

The main pathways to the aquatic environment are via:

- Wastewater Treatment infrastructure; including
    - Combined sewer overflows; and

[105] Welden and Cowie (2017), Degradation of common polymer ropes in a sublittoral marine environment
[106] UNEP, and FAO (2009) *Abandoned, Lost or Otherwise Discarded Fishing Gear*, 2009
[107] Ibid

    - WWT plants effluent;
- Run-off from roads directed into sewers (some of which is directed towards WWT plants); and
- Direct emissions into surface waters.

The main sinks for microplastics along these pathways are:

- **Sewage sludge** and consequently application to agricultural land;
- **Road-side sedimentation** devices and consequently residual waste treatment;
- **Road cleaning** with the resultant debris ending up in residual waste treatment;
- **Porous asphalt** which can trap particles under its surface; and
- **Soils** near to microplastics deposition zones.

Wind-blown microplastics directly entering water bodies are also a potential pathway but the data needed to quantify this are lacking. It would require knowledge of microplastics deposition patterns as well as an understanding of how windblown microplastics can move towards nearby water bodies.

2.2.8.1 Treatment and Capture of Waste Water and Run-off

As identified by all of the other country and EU level studies on microplastic emissions, wastewater treatment (WWT) plants and the sewerage infrastructure are key pathways in which microplastics may travel. The plants themselves are also often identified as point sources for microplastic emissions, however there are numerous points in the sewerage system where microplastics can be captured or emitted. The key factors which influence this are discussed in the following section but consist of:

- Retention rate of microplastics in wastewater treatment plants;
- The type of sewage treatment employed in each country;
- The type of sewerage systems in place (combined or separate); and
- Capture of microplastics at or near the roadside.

WWT Microplastics Retention

Retention rates of microplastics in WWT plants have been the subject of increased study over the past three years. Whilst this has led to an increased understanding of the subject, there is still a considerable way to go before there can be any level of certainty about retention rates. Each WWT plant studied is unique in both the population it serves and therefore the proportion and type of microplastics moving through it, and also the technologies used to treat the wastewater. None of the mainstream treatment processes currently used in the EU are specifically designed to capture microplastics. Because of these factors it is unwise to apply findings from one study to other plants especially from other countries. The methodological approaches in each study are also different, with no common standard for testing being applied. Many studies merely capture a point in time and therefore do not account for the large differences in load throughout the year.

A summary of the studies published since 2015 which have calculated a retention rate are shown in Table 81 in Appendix A.3.8.1. The rates range from 17% —99.7%, depending upon the treatment type. The majority of testing has been undertaken on plants with tertiary treatment. There is no specific definition of tertiary treatment and therefore the types of processes employed can vary, but is usually specified when chemicals need to be removed before effluent is discharged into sensitive ecosystems. Secondary treatment normally refers to a biological process used to remove dissolved and suspended organic compounds. This is then removed as sludge. Primary treatment is the most basic - sewage is pumped into settlement tanks where heavy solids sink and are pumped away and buoyant solids (oils etc.) are skimmed off. Most studies of WWT plants with some form of tertiary treatment show a retention rate of >90%, with the notable exception of the most recent study from the Netherlands which found an average of 72%.

One of the areas that requires further exploration and standardisation to improve the robustness of the available evidence is the method through which microplastics are identified. This is particularly important so as not to overestimate the number of microplastics in influents or effluents. For example Ziajahromi et al[108] discovered that between 22% and 90% of the suspected microplastics were determined to be not plastic particles after FT-IR analysis.

Consequently to this, a range of retention rates (from 50—90%) have been applied by the country level microplastics emissions studies. Older studies typically cited the higher rates identified at the time of around 90% whereas latter studies have recognised that this is unlikely in practice.

Sewage Treatment in European Countries

For estimations of WWT retention rates at the EU level it is important to take into account the variations between countries as only 56% of the population is connected to tertiary treatment and 77%[109] are connected to secondary or higher treatments. Figure 6 shows this for all EU countries.

To develop an EU level estimate of WWT retention rates this data is applied to the maximum and minimum retention rates observed for each treatment type in the WWT plant studies identified previously in Table 81 (Appendix A.3.8.1). This results in an **average retention rate in Europe of between *53% and 84%.*** Although this retention rate is based on data for retention by number rather than mass, it is used to represent mass in this study. It is recognised that weight may play a significant part in microplastic retention. However, without further research into the full range of densities, sizes and shapes it is not practicable to include this level of complexity at this time.

Individual country retention rates are shown in Table 82 in Appendix A.3.8.1 along with the calculations and assumptions used in Table 83.

A statement from EurEau[110] the “voice of Europe’s drinking water and waste water service operators”[111] states that they believe these retention ranges do not represent reality and suggests that an average removal rate of up to 90% should be used. However, the statement also asserts that “…the knowledge level of micoplastics [sic] [in wastewater treatment] is still too underdeveloped to define sound policy options.” This suggests that it is still too early to conclude that across Europe—including countries that are struggling to meet the UWWT Directive—the retention rates are all performing to the highest observed standards. The benefit of using a retention rate range is that if evidence comes to light in future that allows a more accurate assessment, it is still likely that it will

---

[108] Ziajahromi, S., Neale, P.A., Rintoul, L., and Leusch, F.D.L. (2017) Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics, *Water Research*, Vol.112, pp.93–99

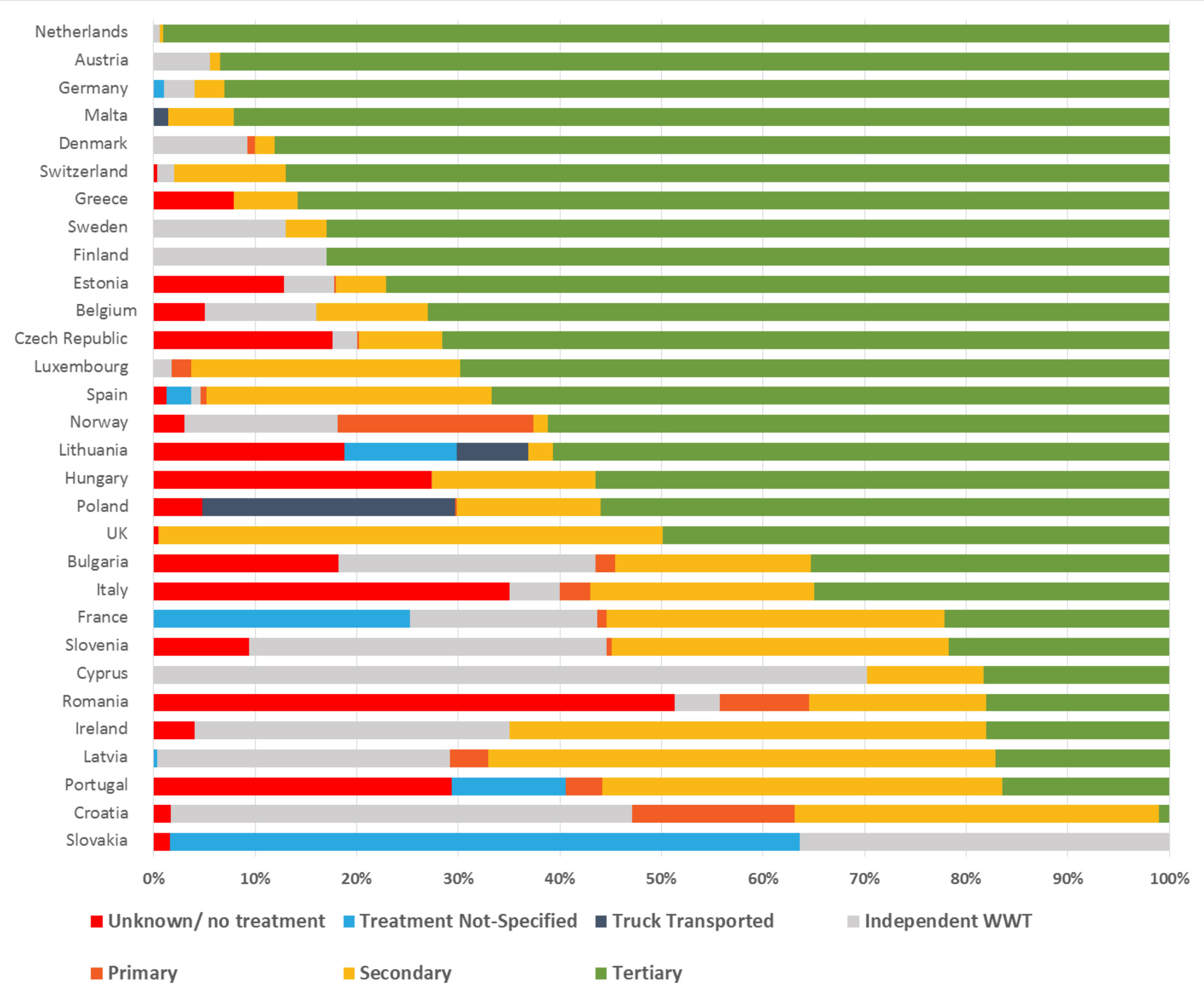
[109] Own calculations using data from Eurostat (Population connected to wastewater treatment plants from update 03.04.17)

[110] EurEau, 28th November 2017, EurEau comments on the Eunomia report

[111] http://www.eureau.org/index.php/who-we-are

fall inside the proposed range. If, for example, retention rates are shown to be closer to 90%, the upper range would be considered more valid for policy making.

Figure 6 - Population Connected to WWT by Treatment Type



Source: Data extrapolated from Eurostat from latest reported data from each country (2009—2013, 2005 for Cyprus)

Sewerage Systems

Another factor which will affect the proportion of microplastics that are retained in WWT systems is the type of sewage transport system that is employed. These are generally split into two systems;

- **Combined—** foul waste water and storm water travel in the same pipe and are treated in a WWT plant unless the system is overloaded, in which case raw sewage is deposited directly into water bodies via combined sewage overflows (CSOs)
- **Separate—** foul waste water and storm water are carried separately. CSOs are not often needed as storm water does not go through a WWT plant. Equally storm water is therefore not often given any form of treatment before entering water bodies.

A further hybrid system is also beginning to be installed whereby during rain events, storm water is initially treated as this contains the largest amount of pollutants.

Table 80 in Appendix A.3.8.1 shows some of the few sources of information on sewerage infrastructure which demonstrates the variability which ranges from 12—70% combined systems. There will also be a large difference between urban and rural areas whereby many cities in Europe still have a legacy of combined sewers that are many decades old.

There is generally very little data available on the proportion of sewerage system types in European countries. According to Eureau, there is no official data on this at a European level. It estimated that there are 650,000 CSOs across Europe[112] but was unable to provide estimates of CSO release rates for any country. Each one of these CSO's is a potential point source for microplastic emissions from untreated effluent. The proportion of wastewater discharged from CSOs is even less well known, despite its potential to contribute hugely to pollution and microplastics emissions in water bodies. If a high level of 90—95% of pollutants (and microplastics) are removed during WWT, CSOs could account for up to 50% of pollution if just 5% of waste water is directly discharged. During stakeholder discussions on this subject it was suggested by representatives from the Netherlands and the WWT industry that 5% would be considerably underestimating CSO release rates. This agrees with a recent WWF report[113] on UK wastewater infrastructure which found up to 14% of CSOs are reported to be discharging into rivers on a weekly basis.

Considering this uncertainty, ***a figure of 50:50 has been used between combined and separate systems*** throughout this report when microplastics are known to enter a sewerage system. A ***CSO release rate of 10%*** was used for assumed discharge directly to water bodies.

2.2.8.2 Run-off Treatment and Sedimentation

Unlike wastewater, water from road run-off – whilst coming under the Water Framework Directive (WFD) – does not have any specific treatment requirements.[114] The goal of the WFD is that member states should ensure 'good ecological and chemical status'[115] of water bodies, but road run-off is not specifically targeted for measures or treatment levels. Because of this, countries interpret and mitigate the issues of pollutants from road run-off differently; it is therefore also difficult to assess whether these practices are compliant with the WFD.

It is generally considered that some form of treatment of the run-off is required if the vehicle traffic exceeds 15,000 vehicles per day (known as ADT – Annual Daily Traffic).[116] This is unlikely to be the case on many rural roads, but would usually be reached in highways and urban areas.[117]

The effectiveness of the capture of suspended particles of various methods of run-off treatment from roads is a key consideration. Swales, infiltration basins, wetlands, sedimentation ponds are all

---

[112] Personal communication with Eureau (June 2017)

[113] https://www.wwf.org.uk/updates/40-rivers-england-and-wales-polluted-sewage

[114] Dr Sondre Meland, (2016) *Management of contaminated runoff water: current practice and future research needs*, Report for Conference of European Directors of Roads, April 2016

[115] http://ec.europa.eu/environment/water/water-framework/info/intro_en.htm

[116] Dr Sondre Meland, (2016) *Management of contaminated runoff water: current practice and future research needs*, Report for Conference of European Directors of Roads, April 2016

[117] Data from the UK's Department for Transport indicates that 15,000 ADT would be unlikely to be reached on many of the 'main roads' in rural areas, (Wales, excluding the cities of Cardiff and Swansea only reaches an average of 12,000 ADT and this excludes minor roads which are not measured), whereas urban areas would easily exceed this (London has an average of 26,000 ADT on it roads, with over 100,000 ADT on some of the major trunk roads/highways). http://www.dft.gov.uk/traffic-counts/download.php

common methods used near roads to manage run-off and reduce pollutants entering water courses as part of sustainable drainage systems (SuDS). This also includes microplastics (under the guise of 'suspended solids'), although this is rarely their specific intended purpose. Data on their effectiveness is sparse. The Design Manual for Roads and Bridges from Highways England estimates that such measures are over 60% efficient at removing suspended solids from road run-off.[118] A Swedish storm water watershed modelling tool (Stormtac)[119] uses data collected from a large number of studies to estimate the retention rates of various treatment systems (See Table 84 Table 84in Appendix A.3.7). Retention efficiencies of 70 – 90% are observed for the more common treatments. These are ranges for well-performing systems, however, and as one study in Sweden found, the efficiency can vary from 40-90%[120] for retention ponds (a common treatment method in Sweden). Therefore the existence of the treatment installation does not necessarily indicate that its retention rate will be optimal. Indeed, the study also goes on to demonstrate that monitoring of pollutant loads and retention was sub-standard in 70% of the 27 ponds in the study.

The other more ubiquitous type of sediment capture device in Europe is the gully pot. These are installed along many urban and rural roadsides to prevent sediment from blocking drains or slowing down the flow through increased hydraulic resistance; in the worst cases such blockages can trigger CSO discharges unnecessarily.[121] They are situated directly below roadside drains and must be emptied regularly (once or twice per year) to prevent blockages. The general design has remained the same for many years although plastic is used more often recently as it can be more versatile and easy to handle than moulded concrete gully pots.

Sediment capture rates are thought to be good, with urban roads generating 200g per m^2 per year of which 90g is retained in the gully pot—around 45%. A study of pollutant removal with gully pots from Germany in 1990[122] also found a removal efficiency of 10—40%. The efficiency is also know to relate to particle size with particles smaller than 0.05 mm having a capture rate of <50% rising to >90% for particles greater than 0.3 mm.[123] Tests such as these are usually carried out with sand rather than microplastic particles therefore these results could only reasonably applied to negatively buoyant polymers.

Based on the above discussion, ***a retention rate of 40—80% of microplastics in roadside sedimentation devices*** is proposed. This is to cover the range of retention efficacies for all technologies and the expectation that many will not be maintained fully, leading to lower retention rates.

---

[118] Highways England (2006) *Design Manual for Roads and Bridges - Vegetated Drainage Systems for Highway Runoff*, 2006

[119] http://www.stormtac.com/index.php Stormtac database updated 19/3/2017

[120] Persson, J., and Pettersson, T.J.R. (2009) Monitoring, sizing and removal efficiency in stormwater ponds, Vol.2009, No.4

[121] S.Arthur (1999) Sediment transport in sewers a step towards the design of sewers to control sediment problems, *Proc. Instn Civ. Engrs Wat., Marit. & Energ*, No.136

[122] Matthias Grottker (1990) Pollutant removal by gully pots in different catchment areas, *The Science of Total Environment*, No.93, pp.515–522

[123] A. Bolognesi, A. Casadio, A. Ciccarello, M. Maglionico, and S. Artina (2008) Experimental study of roadside gully pots efficiency in trapping solids washed off during rainfall events, paper given at 11th International Conference on Urban Drainage, 2008

Urban, Rural and Highway Emissions

It is also important to determine whether there are differences in microplastic emission fates depending upon where they are emitted. Seven scenarios have been developed for this study.

Residential waste water is simply water that is washed away in households directly down the drains and is mostly send directly to WWT plants. Non-road drains are similar, but include some form of sedimentation device. Urban, rural and highway run-off are different in that the emission sources of the microplastics are diffuse and therefore they will not all be washed into the sewerage system. Some will be captured in porous asphalt or in road cleaning, but most will either enter some form of storm management where they may settle out, or they will become part of the nearby soil. This can happen either by rainfall run-off or windblown.

Although several microplastic emissions studies have attempted to estimate this, there are no formal methods or models for doing so, therefore each approach is different. A more complete discussion on the subject is provided in Appendix A.3.7. A distribution is proposed in Table 14.

Table 14 - Distribution percentages for microplastics to compartments

Geography	Soil	Surface Water	Sewers
Urban	30%	-	70%
Rural	80—90%	10—20%	-
Highway	40%	-	60%

2.2.8.3 Porous Asphalt

Porous asphalt is used primarily to allow surface water to drain away quickly from the surface of the road and either into storm drains or by slow infiltration into surrounding soils. Although it is more expensive than standard asphalt, it is claimed that its use can prove less expensive as it is offset by the reduce need for storm water management.[124]

The use of porous asphalt also varies greatly throughout Europe. The Netherlands has over 95%[125] of its highways made from this, but this is not the norm in the rest of Europe. Data from the European Asphalt Association (EAPA) for 10 European countries shows that annual production of porous asphalt represents between 0.008% and 9% of the surfacing market in those countries[126]. Most countries are below 1% with the notable exception of Italy and Netherlands with 7% and 9% respectively. Based on this, an estimate of ***5% is used in the pathways model to represent average porous asphalt use in Europe's highways***—this is considered to be a high estimate to represent a best case capture. It is also assumed that this is not used in Urban and rural roads.

[124] http://www.asphaltpavement.org/index.php?option=com_content&view=article&id=359&Itemid=863

[125] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

[126] EAPA (2016) *Asphalt in Figures 2015*, 2016

The capture of microplastics also varies in the literature. In the Netherlands it is estimated to be around 40%, whereas the Stormtac database[127] suggests 90% based on the results of four studies from the US. In the current model 90% is used to represent a best case capture of microplastics. Combined with the estimate of porous asphalt use, a ***microplastic capture rate of 4.5%*** is arrived at for highways. This uses factors that demonstrate the best case capture of microplastics so that its current impact is not underestimated—its impact is likely to be much lower until porous asphalt is more widely used.

2.2.8.4 Road Cleaning

As there is no data available for the potential for road sweeping to capture tyre-wear derived microplastics, a set of variables was created and likely figures inputted to demonstrate the potential impact road sweeping might have under these assumptions. Figures were chosen based on consultation with local experts, but with an emphasis on overestimating the factors in order to show a 'best case' scenario for the amounts of particular matter that are gathered from road sweeping.

It was assumed that rural roads are never swept/cleaned, urban roads with a high footfall would be cleaned regularly and that all highways are cleaned at some point in the year. Urban streets with high footfall were assumed to represent 10% of total urban roads.

Next, to derive the portion of wear that is removed by rainfall on roads that are swept an estimated average number of rainfall days per year for Europe was derived.

To calculate what portion of the dust deposited on dry days is captured in road sweeping an estimate was made as to the frequency of sweeping. Urban roads were assumed to be swept six days a week and all highways were assumed to be swept once per year. Next, a factor for the efficiency of mechanical street sweepers in removing road dust was derived from the literature.

The results of **4.7% capture on urban roads and 0.3% on highways by road cleaning** demonstrates that even assuming an optimistic scenario the modelling suggests that sweeping activities capture only a small proportion of total European emitted tyre-wear derived microplastics. Even when assuming extremely high values for key factors, for example that urban roads which are swept are swept every day of the year, that all highways in Europe are cleaned once per month and that rainfall runoff removal of wear is only 50% on rainfall days, the capture rate on both urban and highway roads does not exceed 8%.

2.2.8.5 River Transport

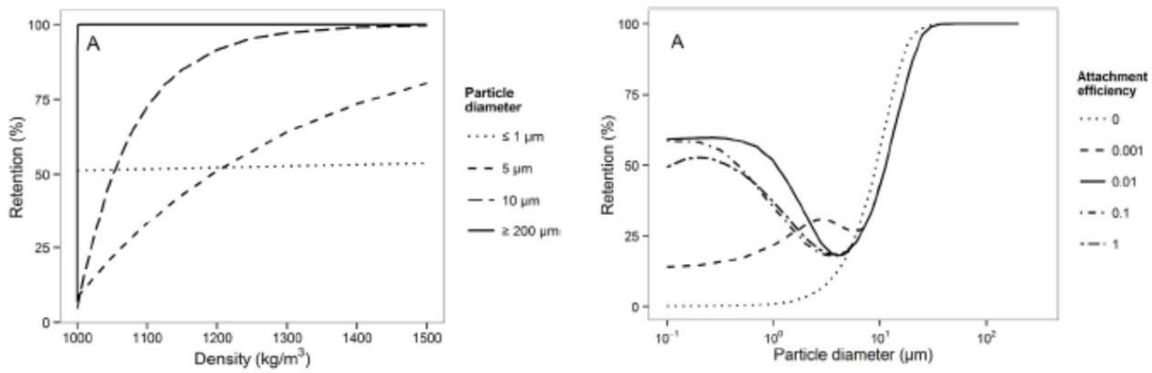
There is very little research into how microplastic particles and fibres would behave once they have entered into rivers. A recent study by Besseling[128] attempted to model the fate of microplastics in freshwater systems and found that particles larger than 100 µm were unlikely to travel further than 10-20kms from the point of emission and that all particles would be retained within 900km. Particles between one and 10 µm were modelled to be the least likely to be retained and therefore travel the furthest—see Figure 7.

---

[127] http://www.stormtac.com/index.php Stormtac database updated 19/3/2017

[128] Besseling, E., Quik, J.T.K., Sun, M., and Koelmans, A.A. (2017) Fate of nano- and microplastic in freshwater systems: A modeling study, *Environmental Pollution*, Vol.220, pp.540–548

Figure 7 – Effect of Particle Size and Density on Retention in Sediments



Source: Besseling (2017)

Applying these findings to the emissions modelling in the present study is very difficult due to the need to characterise all the potential emission by size and density. The location that a particle enters a river and therefore the distance to the ocean is also key to this calculation. This is a potentially straight forward exercise for WWT effluent where all these factors can be directly measured, but diffuse sources of tyre and paint particles would be much harder to quantify. It also may not be appropriate to assume such a high rate of settling from a single statistical modelling exercise.

In the case of tyre wear, the study results suggest that there would be a considerable deposition of 'tyre and road wear particles' (TRWP) in sediments of rivers across Europe. Indeed, this assertion agrees with the findings of a tyre industry funded study that took several sediment samples from watersheds across the world, including the Seine in France[129]. TRWP were detected in over 92% of the Seine samples (100% in three of the four sample sites) with a mean of around 5µg per gram of sediment up to around 10µg (a 1% concentration by weight)—five times higher than sites in the US and Japan. Similar concentrations were also found in roadside soil samples.

Using size characterisation data of TRWP from Kreider et al[130] and assuming a density of 1180 kg/m^3, Besseling's model would suggest that fewer than 10% of TRWP would remain suspended in rivers. There are a great many factors that would influence this the distance travelled and the river topology so this is a highly speculative estimate at present.

This was also investigated in a study[131]by the ETRMA submitted as evidence in the closing stages of the current project (December 2017). The study modelled watershed deposition transport of TRWPs for the Seine and Scheldt river basins. The study's results largely agree with the spread of tyre wear throughout the environment that has been modelled as part of this project—a split of around 50:50

[129] Unice, K.M., Kreider, M.L., and Panko, J.M. (2013) Comparison of Tire and Road Wear Particle Concentrations in Sediment for Watersheds in France, Japan, and the United States by Quantitative Pyrolysis GC/MS Analysis, *Environmental Science & Technology*, Vol.47, No.15, pp.8138–8147

[130] Kreider, M.L., Panko, J.M., McAtee, B.L., Sweet, L.I., and Finley, B.L. (2010) Physical and chemical characterization of tire-related particles: Comparison of particles generated using different methodologies, *Science of The Total Environment*, Vol.408, No.3, pp.652–659

[131] ETRMA - European Tyre & Rubber Manufacturers Association (2017) *Preliminary Tyre and Road Wear Particle Environmental Fate Assessment*, November 2017

between deposition in roadside soils and run-off for example. Whilst the current project estimates emissions to surfaces waters of around between 8 – 15% of the total emitted ***tyre wear,*** the ETRMA estimates this to be higher at 17 – 19%, albeit as ***TRWP***. Of those particles that enter waterways they also estimate that 8 – 11% are transported down-river into the estuaries (~2% of total generated particles). However, both this and the previous speculative estimate are based on the assumption that the particles are ***all*** TWRP at a density of 1180 kg/m^3. As previously discussed, the evidence for this being the case for all tyre particles is not conclusive and therefore it is likely that at least a proportion of the particles will be purely tyre wear with a density of around 800 kg/m^3,[132] these particles are more likely to be buoyant and therefore travel towards the oceans in greater quantities. It is clear that particle size and density will play a large role in a particles' distribution throughout the environment, therefore it will be important to establish this for tyre and other microplastic particles if their movement towards the marine environment can be fairly assessed. There is no evidence currently that can lead to the conclusion that there is little or no risk of this at present.

The tyre industry's own studies[133,134] also claim that TRWP in sediments poses a low risk to sediment dwelling aquatic organisms at the concentrations of 1%. This disagrees with previous studies (summarized by Wik and Dave[135]) although the former studies are the only ones to date that have specifically studied effects within fresh water sediment.

Questions remain about potential long term effects to organisms and as it has been established that a large proportion of the TRWP may end up in riverine sediments, there is also substantial potential for the concentrations to increase significantly. No thresholds for exposure to higher concentrations have been established at this point[136]. The lack of field studies is also concerning.

River sediments are not static and not necessarily an end-point for microplastics. Rivers will transport sediment downstream at varying levels depending upon the velocity and size of the flow. These are either carried as suspended particles in the water column or as simply pushed along the bottom of a waterway—the latter accounting for around 10% of sediment load[137]. For example, the Danube discharges sediment into the Black Sea at a rate of 4—6 million tonnes per year. If only a small percentage of this is comprised of microplastics, the movement of sediment could be a substantial mechanism for the transport of microplastics into the marine environment as well.

It is unclear whether the results from Besseling's modelling study can be applied to textiles as they do not have uniform dimensions and may behave differently to particles. Fibre lengths have been recorded to be anywhere from 20µm to over 5mm in length, but with a thickness (diameter) of 10-20 µm. Polyester has a density of 1,380kg/m^3, therefore is well within the density range proposed

---

[132] Sofi, A. (2017) Effect of waste tyre rubber on mechanical and durability properties of concrete – A review, *Ain Shams Engineering Journal*

[133] Marwood, C., McAtee, B., Kreider, M., Ogle, R.S., Finley, B., Sweet, L., and Panko, J. (2011) Acute aquatic toxicity of tire and road wear particles to alga, daphnid, and fish, *Ecotoxicology (London, England)*, Vol.20, No.8, pp.2079–2089

[134] Panko, J.M., Kreider, M.L., McAtee, B.L., and Marwood, C. (2013) Chronic toxicity of tire and road wear particles to water- and sediment-dwelling organisms, *Ecotoxicology (London, England)*, Vol.22, No.1, pp.13–21

[135] Wik, A., and Dave, G. (2009) Occurrence and effects of tire wear particles in the environment – A critical review and an initial risk assessment, *Environmental Pollution*, Vol.157, No.1, pp.1–11

[136] Wik and Dave (2009) suggest that the Predicted No Effect Concentration (PNEC) in sediment to be 0.6% based upon extrapolation from surface water data.

[137] Milliman, J. D. (2001). River Inputs. In Encyclopedia of Ocean Sciences (pp. 2419–2427). Elsevier.

by Besseling for sediment retention, however as fibres are found in considerable quantities both in river sediments and further down-stream[138] into the oceans it is unlikely that Basseling's model is a true reflection of reality.

For other more homogenous microplastics it may be easier to assess the likelihood of their transportation downstream into oceans. For plastic pellets the density of the material is likely be the most important factor. Those plastics that are neutral or positively buoyant, such as polyethylene and polypropylene, would be expected to be transported easily—these account for around 50% of plastics in the European market. This reflects the findings of a sampling study along the River Rhine[139] which found a high abundance of buoyant plastic 'spherules'. Interestingly the study also managed to sample many negatively buoyant plastic particles in large sizes (their sampling net mesh was 333 µm therefore only relatively large particles could be captured) with polystyrene being the most abundant throughout the study despite it density being 1,040 kg/m^3. This suggests that there are far more complex interactions than can be effectively modelled at this time.

In conclusion, linking the estimates of microplastics releases into fresh water to what may be transported to the marine environment is fraught with uncertainty at this time. It is likely that a large amount of sedimentation will occur, however this may also eventually be transported downstream. The high concentrations of TRWP already found in sediments it therefore a potential cause for concern.

---

[138] Flavia Salvador Cesa, Alexander Turra, and Julia Baruque-Ramos (2017) *Synthetic fibers as microplastics in the marine environment: A review from textile perspective with a focus on domestic washings*, April 2017

[139] Mani, T., Hauk, A., Walter, U., and Burkhardt-Holm, P. (2015) Microplastics Profile Along the Rhine River, *Scientific Reports*, Vol.5, p.17988

2.2.9 Results of Pathway Modelling

The factors identified in the previous section are combined to create a mass flow model for the proportion of microplastics that are estimated to enter waterways from the different entry points of residential/industrial foul water, and run-off in urban, rural and highway locations. An upper and a lower estimate of the proportion of microplastics that will enter surface waters is given for any microplastic emission that is expected to end up in one or more of these five pathway scenarios. For example if 100 tonnes of microplastics enter a residential drain system, between 18 and 48 tonnes are expected to reach surface waters. A further 26 to 41 tonnes are expected to be applied to agricultural land and disposed of in residual treatment (usually landfill or incineration).

The results from Section 2.2.1 to 2.2.6 (full tables in Appendix A.3.0 and A.3.8.7) are entered into the pathways mass flow model to provide results in

Figure 8 (using the midpoint of all results). Figure 9 shows the upper and lower estimates for each microplastic emission to waterways.

Table 15 shows the results of the data quality assessment for each microplastic s source at generation and emitted to surface waters. The full assessment can be found in Appendix A.7.0. This demonstrates that not all sources can be quantified with the same level of accuracy; for example, clothes washing has data available on source generation and a defined pathway, whereas building paints have neither and rely on assumptions primarily.

Table 15 – Data Quality Summary

Source	Data Quality/Uncertainty Assessment Score	
	Generation at Source (score) [a]	Emitted to Surface Waters [b]
Automotive Tyres	9	Split, between urban, rural and highways—rural pathways particularly uncertain
Washing of Clothing	8	Direct to sewers with data available on retention rates.
Artificial Turf	12	Dispersion throughout the environment highly uncertain
Pellets	13	Dispersion throughout the environment highly uncertain
Fishing Gear	18	Direct to surface waters
Marine Paint	8	Direct to surface waters
Building Paint	7	Split, between urban, rural and highways—rural pathways particularly uncertain
Road Markings	8	Split, between urban, rural and highways—rural pathways particularly uncertain

■ = High **Certainty,** ■ **= Medium Certainly,** ■ **low certainty**

a) **Scores generated in Appendix A.7.0** for generation a source
b) A qualitative score is applied to emissions to surface water

Figure 8 – Microplastic Emissions Sinks ordered by releases to waterways (Midpoint)

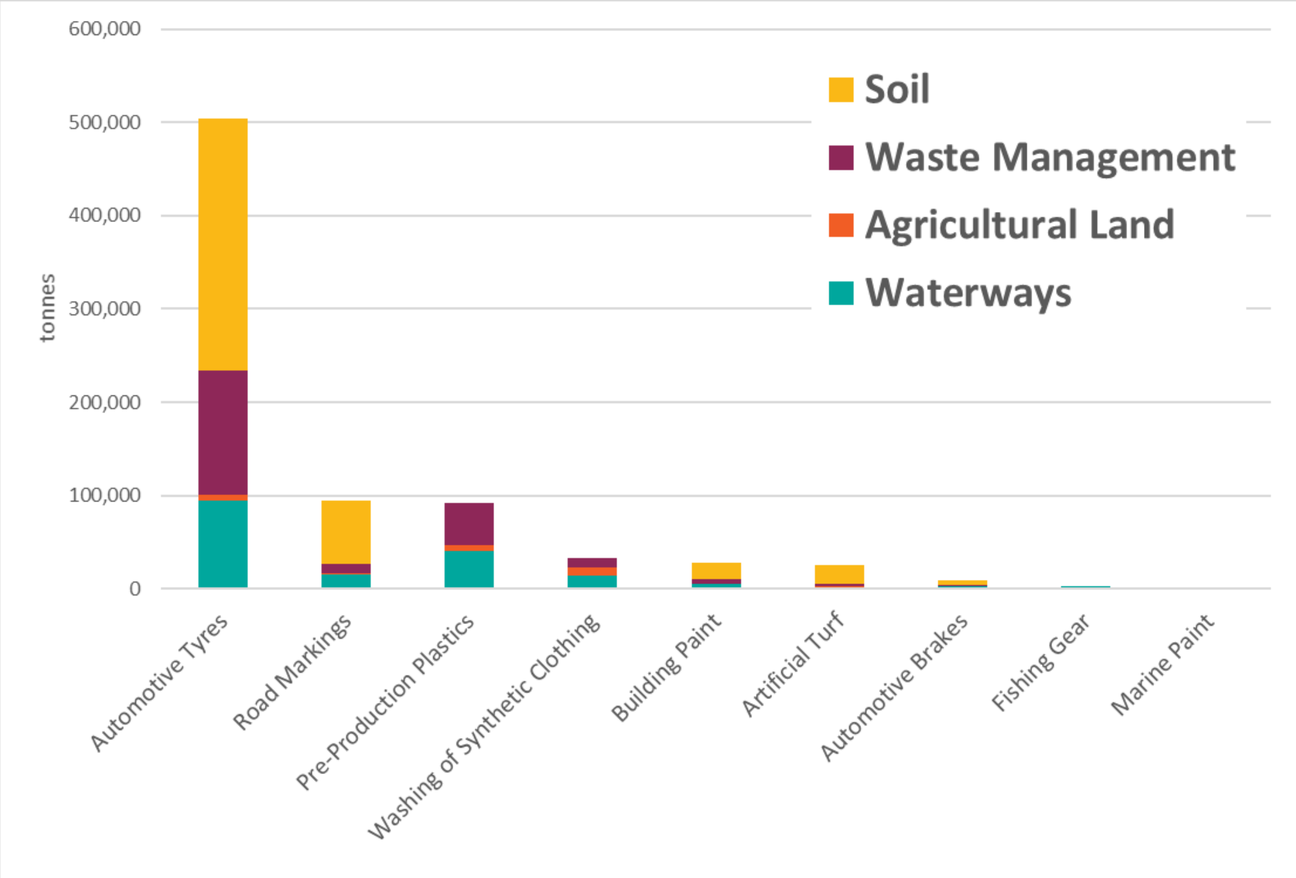


Figure 9 – Microplastic Emissions to Surface Waters Upper and Lower Estimates

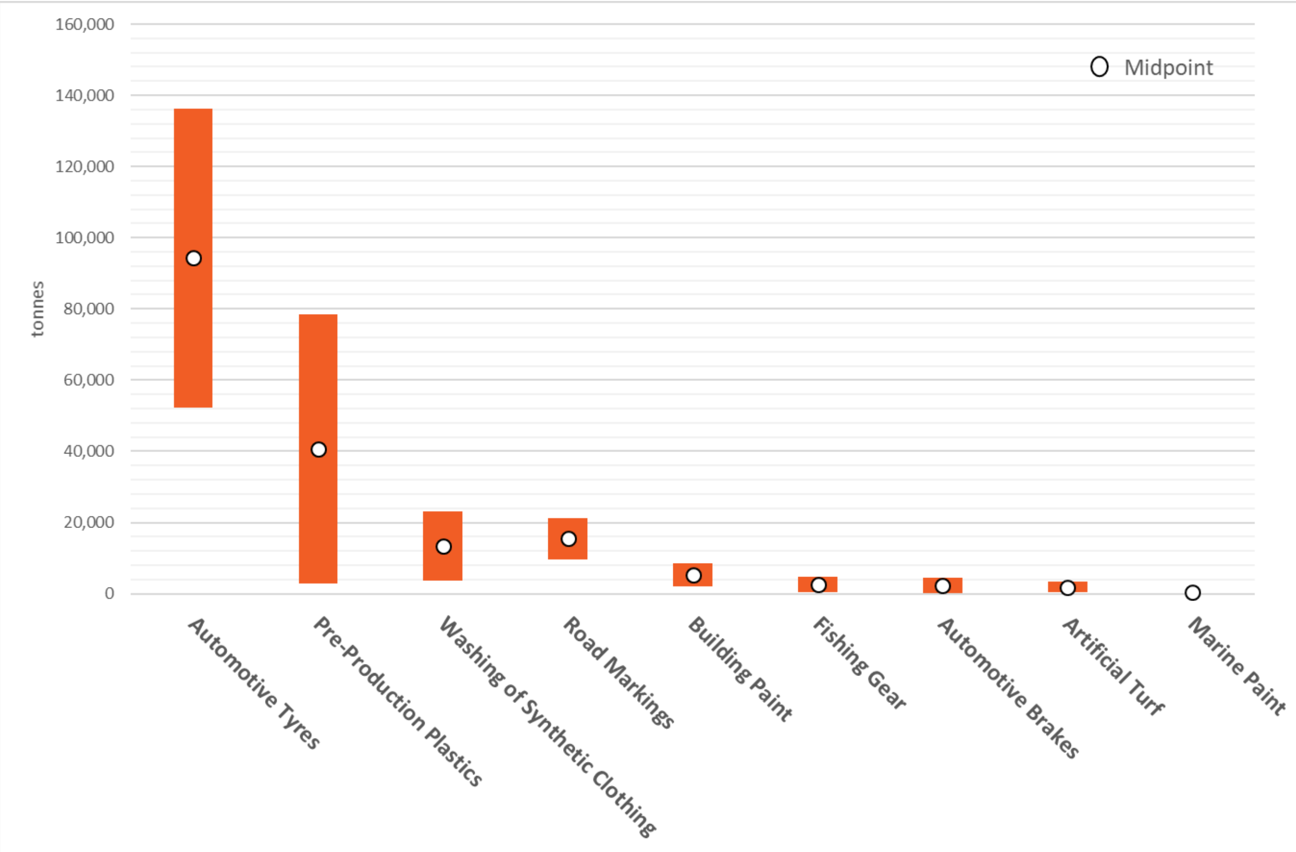
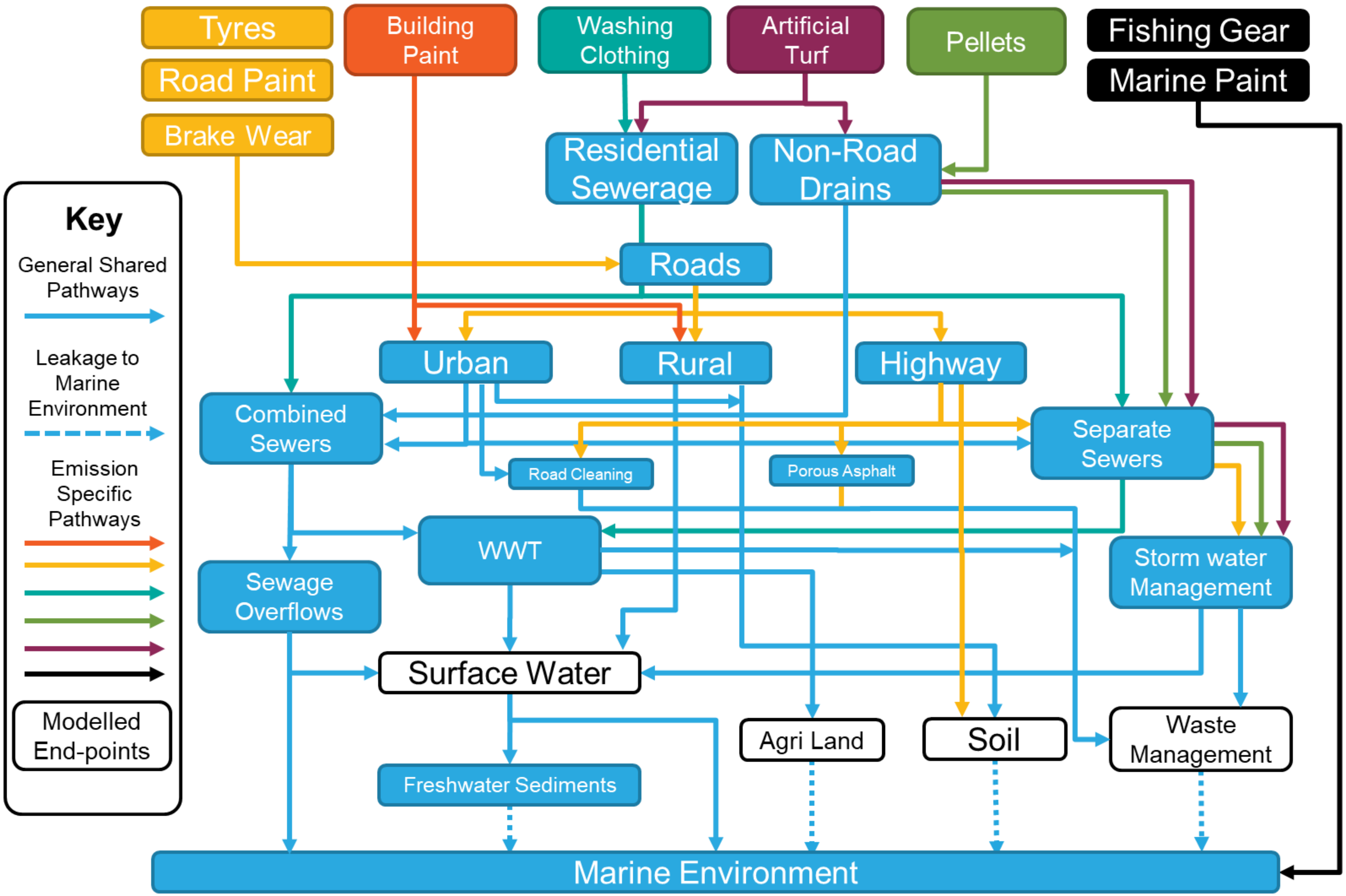


Table 16 – Annual Microplastics Emissions to Surface Waters

Source	Upper (tonnes)	Midpoint	Lower (tonnes)
Automotive Tyres	136,000	94,000	52,000
Pellets	78,000	41,000	3,000
Washing of Clothing	23,000	13,000	4,000
Road Markings	21,000	15,000	10,000
Building Paint	8,000	5,000	2,000
Fishing Gear	5,000	2,600	500
Automotive Brakes	5,000	2,000	100
Artificial Turf	3,000	2,000	300
Marine Paint	400	400	400
Total	**280,600**	**176,300**	**71,800**
Note: All Figures are rounded therefore totals may not add up			

Figure 10 shows the full complexity of the pathways identified for all of the key microplastics sources. The four modelled end-points are highlighted with surface waters being the key transport method to the marine environment. There was insufficient data to model this transfer (as discussed in Section 2.2.8.5), however current evidence suggests that the marine environment is not the only environment that can be damaged by microplastics—riverine fauna is also at risk. It is important to recognise that leakage from other sinks such as soil and waste management (including sludge application to land) is also likely to be a small but not insignificant source of microplastic releases to the marine environment.

Figure 10 – Microplastic Emission Pathway Model Graphical Representation



3.0 Problem Drivers

In the following sections, the problem drivers are considered by source, followed by cross-cutting problem drivers relating to capture through road drainage infrastructure, street cleansing, and waste water treatment plant. Of the sources initially investigated, we subsequently identify problem drivers and associated objectives for:

- automotive tyre wear;
- pre-production plastic pellets;
- synthetic clothing;
- road markings;
- building paint; and
- artificial turf.

These have been selected because (with the exception of artificial sports turf) our analysis shows that they are by far the most significant contributors to loss at source, and releases to the aquatic environment.

Artificial sports turf, while one of the smaller contributors, is the source projected to grow most rapidly in percentage terms in the years out to 2035. A further reason for its inclusion that there is a comparatively small number of sources (around 50,000 pitches using artificial turf in the EU), each producing comparatively large quantities of microplastics (infill loss per pitch is estimated at up to 5 tonnes of infill per annum).

3.1 Automotive Tyre Wear

Wear from vehicle tyre treads is an inevitable effect of their use. The treads of some tyres are understood to abrade (abrasion rates being measured in mg/km) under standard conditions at a higher rate than others (independent of external factors such as driving style and road surface that may affect actual tread abrasion rates). There is no standard way of communicating the tread abrasion rate of tyres to consumers. However, the tread abrasion rate is one of the factors that affects the distance that can be covered before the tyre has to be replaced, known as 'mileage'.

Mileage was identified by car drivers as second only in importance to wet grip in a survey reported in a recent study for the European Commission.[140] The authors of the study note that 'mileage is a common parameter used to express the durability of tyres as a distance in miles or kilometres', and that the mileage of a tyre is directly correlated to the tyre wear factor (amount of tread lost per kilometre). The authors also note that abrasion (i.e. the removal of materials from the tyre when it interacts with the road surface) 'is related to tyre mileage, since both are linked closely to the tyre wear.'

[140] Viegand Maagøe A/S (2016) Review Study on the Regulation (EC) No 1222/2009 on the Labelling of Tyres, Final Report to DG ENER of the European Commission, March 2016. Available at http://www.labellingtyres.eu/downloads/Final_report-Review_study_on_labelling_of_tyres.pdf

The Commission's own Impact Assessment from 2008 on the labelling of tyres also notes the importance that consumers place on this aspect, stating that:[141]

> *Market surveys in addition show that wear (i.e. "long lasting tyre") is the most important criteria in consumers purchasing decision.*

It's important to note that the mileage of the tyre depends on both the tread abrasion rate in mg/km, *and* the depth of the tread. For example, as shown in Table 17, the two tyres A and B might have the same mileage, but depending on the tread depth and the abrasion rate, the amount of tyre wear particles released over the tyre life can vary considerably.

From the consumer perspective, mileage is likely to be the most important aspect, while from the perspective of reducing tyre wear particle emissions, the abrasion rate is more important.

Table 17: Relationship between Tread Depth, Abrasion Rate and Mileage

	Tyre A	Tyre B
Mileage	80,000 km	80,000 km
Tread Depth	5 mm	10 mm
Abrasion Rate	1mg/km	2mg/km
Tyre wear particles released over tyre life	80,000 mg	160,000 mg

Source: ETRMA illustrative figures

Whether or not tyre wear particles are captured before entering sewers, surface water or soil, depends on the nature of the road drainage infrastructure, and the methods and the nature and frequency of street cleaning.

The problem drivers in respect of automotive tyre wear can broadly be divided into problems that relate to the nature of the tyre itself (including its use) and those that relate to the road surface, the drainage infrastructure and the nature and frequency of street cleansing. The latter are considered in Section 3.7.

While automotive tyre wear is unavoidable, there is insufficient incentive, be it financial, regulatory, or reputational, for manufacturers to develop tyres that abrade at lower rates while maintaining the other important attributes valued by consumers such as wet grip and fuel efficiency.

All things being equal, making tyres that abrade at lower rates would mean that manufacturers would sell fewer tyres over a given period.[142] This acts as a disincentive to further reduce abrasion

[141] European Commission (2008) Commission Staff Working Document: Accompanying document to the Proposal for a Directive of the European Parliament and of the Council on Labelling of Tyres with Respect to Fuel Efficiency and Other Essential Parameters – Impact Assessment, 13.11.2008

[142] Viegand Maagøe A/S (2016) Review Study on the Regulation (EC) No 1222/2009 on the Labelling of Tyres, Final Report to DG ENER of the European Commission, March 2016. Available at http://www.labellingtyres.eu/downloads/Final_report-Review_study_on_labelling_of_tyres.pdf

rates. The EU tyre label currently focuses on three elements - fuel efficiency, wet grip and noise. These factors will therefore influence consumer choice to a greater extent than wear rates.

There is no regulation on the minimum rate at which tyres can abrade, and the issue of automotive tyre wear as a possible contributor to marine microplastics has only recently emerged, meaning there has been no demand for such a minimum level of abrasion rate in order to address microplastics.

A lack of awareness among consumers of the problem itself and of the extent to which tyres from different manufacturers may be contributing to the problem (as it is not represented in the EU tyre label) means that this is not a reputational issue.

Drivers are unaware of the problems of vehicle tyre wear - potentially the lack of 'visibility' of vehicle tyre wear particles contributes to this - and of the ways in which they can mitigate it via modal shift or applying eco-driving techniques.

The problem drivers relating to the tyre, and its use, can thus be summarised as follows:

- Insufficient financial, regulatory or financial incentives for tyre manufacturers to develop tyres that abrade at lower rates
- Insufficient information to encourage behavioural change

3.2 Pre-production Plastics

Pre-production plastics take the form of small pellets, powders and flakes (the generic term ‘pellet’ is used in the discussion that follows). These are accidentally spilled at various points in the pre-production chain (i.e. at any step prior to incorporation in a plastic component or product). Their small size makes it difficult (and relatively costly) to clean up, and hence even after clean-up efforts they find their way through drains and surface waters to the marine environment. Pellets are frequently found on beaches around the EU, and their distinctive nature means that they are readily recognised.

Best practice measures to prevent pellet spills and ensure high levels of capture of spilt pellets, have already been developed in the form of Operation Clean Sweep (OCS), which is promoted by Plastics Europe and other plastics industry trade bodies. However, the number of companies signing up to OCS is low, and there is no way of checking that those that have signed up have actually put in place measures that would be regarded as best practice.

The small size of pre-production plastics means that they are easily spilt. If spilled within a facility, to avoid causing workers to slip, there will be a health and safety reason for clearing up those in a location likely to cause such an accident.[143] However, beyond this, there is insufficient incentive, be it financial, regulatory, or reputational, for firms to clean up all pellets, or indeed adopt the well-established best practice measures that would reduce the likelihood of spills and improve capture.

On the financial side, the value of the pellets does not justify the effort it would take to clean them all up, especially as once spilled, they are considered as contaminated, and therefore a waste.[144] The external costs associated with the loss of such pellets into the environment is not incorporated into the market price, albeit this would need to be very high indeed to stimulate the adoption of best

[143] Eunomia (2016), Report for Fidra on Study to Quantify Pellet Emissions in the UK, March 2016

[144] As demonstrated in Eunomia (2016), Report for Fidra on Study to Quantify Pellet Emissions in the UK, March 2016

practice measures. The implementation of best practice measures would thus present a cost to the business that will not be recovered through the value of the avoided lost material.

On the regulatory side, as an emerging issue, there is no legal requirement for any facilities to put in place best practice measures, and those facilities (the larger ones) that are visited by environmental regulators will not be inspected on this issue. The majority of facilities are too small to receive such visits from environmental regulators.[145] While theoretically it may be possible to prosecute an individual facility, there appears to be little appetite for this among regulators, identifying the source of an emission is very difficult, and the deterrent effect would be minimal.[146] Absence of a requirement to implement best practices, in the form of regulation or some other form of requirement, is arguably the most significant of the problem drivers.

From a reputational perspective, consumers are insufficiently aware of the issue to demand action, and as most of the supply chain is not 'public facing', there is an insufficient reputational driver to adopt best practices. NGOs are already engaged in significant efforts to raise awareness of the problem, such as the Great European Nurdle Hunt (2nd to 5th June 2017), but it's a very different issue from, for example, microbeads in cosmetics where consumers engage directly with brands that incorporate microplastics as ingredients.[147]

The problem drivers can thus be summarised as:

- The small size of pre-production plastics (as pellets, powders and flakes) means they are prone to loss and difficult (and costly) to recover
- There is insufficient financial, regulatory or reputational incentives for all of the actors in the pre-production plastics supply chain to implement best practice measures to reduce loss

3.3 Synthetic Clothing

Synthetic clothing fibres are known to be released when the clothes are washed. As washing machines do not capture these fibres, they typically enter the wastewater treatment system, from where a proportion may be captured in sludge, and others enter the aquatic environment. The sludge may subsequently be applied to land, or incinerated.

Although one of the more well-researched sources of microplastics, there is still much that is unknown about the relative influence of the numerous factors affecting rates of microfibre loss. Alongside a general lack of awareness of such factors, there is currently a lack of incentives for manufacturers to design clothing in a way that reduces the likelihood of loss, or to capture microfibres within washing machines. There is also a wider issue of consumer awareness as to the mitigating measures that can be adopted by individuals.

Problem drivers relating to the nature of the clothes placed on the market and consumer purchasing decisions can be summarised as follows:

- A lack of awareness to date amongst manufacturers/brands and retailers of the issue, the scale of the issue, and possible mitigating measures that can be taken in the design and manufacture of synthetic garments

---

[145] See Eunomia (2016), Report for Fidra on Study to Quantify Pellet Emissions in the UK, March 2016
[146] See Eunomia (2016), Report for Fidra on Study to Quantify Pellet Emissions in the UK, March 2016
[147] See http://www.nurdlehunt.org.uk/euronurdlehunt.html

- Insufficient financial, regulatory or reputational incentive for manufacturers/brands and retailers to produce and sell clothing that sheds fewer, or no, synthetic microfibres
- Consumer purchasing habits mean that the use phase of clothing is relatively short

Problem drivers relating to washing machine use can be summarised as follows:

- A lack of consumer awareness as to the actions they can take to minimise the loss of synthetic fibres when washing clothing
- Insufficient financial, regulatory or reputational incentive for washing machine manufacturers to develop methods to capture synthetic fibres

3.4 Road Markings

Road markings contains polymers, and as markings wear throughout their lifetime, microplastics are released. Whether or not road markings microplastics are captured before entering sewers, surface water or soil, depends on the nature of the road drainage infrastructure, and the methods and the nature and frequency of street cleaning.

The problem drivers in respect of road markings can broadly be divided into problems that relate to the road markings itself, and those that relate to the drainage infrastructure and the nature and frequency of street cleansing. The latter are considered in Section 3.7.

There is a general lack of awareness among road authorities of the issue of road markings wear as a source of microplastic emissions, along with a lack of understanding of the relative rates of loss of microplastics from different types of road markings. While options to enable safe use of road space without recourse to road markings exist, these are not widely used, and awareness of these options as genuine alternatives is low.

Problem drivers relating to road markings can be summarised as follows:

- A lack of awareness among road markings users of the issue
- A lack of understanding of the relative rates of loss of microplastics from different types of road markings
- Insufficient consideration given to alternatives to road markings.

3.5 Building Paint

Building paint contain binders, which are polymers, which form a matrix to hold the pigment in place. Through its lifetime, and when removed at end of life – if not adequately captured and sent for appropriate disposal, microplastics can be released. Whether or not building paint microplastics are captured before entering sewers, surface water or soil, depends on the nature of the road drainage infrastructure, and the methods and the nature and frequency of street cleaning.

The problem drivers in respect of building paint can broadly be divided into problems that relate to the building paint itself, and those that relate to the drainage infrastructure and the nature and frequency of street cleansing. The latter are considered in Section 3.7.

There is a general lack of awareness among the public and contractors of the issue of building paint wear as a source of microplastic emissions. Accordingly, adequate precautions may not be taken to prevent the loss to the wider environment of old paint when it is removed prior to application of new paint. There is currently no clear alternative to the use of polymers as binders.

Problem drivers relating to building paint can be summarised as follows:

- The use of polymer as binders means that building paint is a source of microplastic emissions

- A lack of awareness among contractors and the general public of the issue meaning that adequate precautions may not always be taken when removing old paint leading to release of microplastics to the environment

3.6 Artificial Turf

Polymeric infill from artificial sports turf can be inadvertently removed by players (when attached to their clothing or footwear), and also through maintenance activities such as snow clearance in some countries. It may then enter drains, soil, or surface water, or be removed as part of waste collection.

The potential for the polymeric infill from artificial sports turf to contribute to the problem of marine microplastics has only been relatively recently identified. Best practice measures can be taken to reduce the loss of infill from individual pitches, and alternative infill materials are available. However, at present there is a lack of financial, regulatory, or reputational incentives for pitch operators to implement best practice measures, or switch to alternative infill material.

The problem drivers in respect of artificial sports pitches can broadly be divided into those that relate to inadequate capture of infill, and those that relate to the use of alternatives.

As an emerging issue there is a lack of awareness to date amongst pitch operators that loss of infill can contribute to marine microplastics. As SBR in particular is relatively cheap compared to other costs associated with the construction and maintenance of artificial sports pitches, there is an insufficient financial case for preventing loss, and switching to natural infill alternatives such as cork, would be costly. Regulators, pitch users and the public, are also unaware of the issue, and thus there is no regulatory of reputational driver for pitches to prevent loss of polymeric infill, or use alternatives. Finally, in the absence of ‘design, build, and maintain’ contracts installers do not have an incentive to minimise lifetime costs through avoiding purchase of 'top-up' infill to replace that which is lost (albeit the infill is relatively cheap).

Problem drivers relating to artificial turf can be summarised as follows:

- Insufficient financial, regulatory or reputational incentive for pitch operators to implement best practice measures in specifying the facility and managing its use
- Insufficient financial, regulatory or reputational incentive for pitch operators to use alternative infill material
- Insufficient financial incentive upon installers of artificial sports pitches to design in such a way as to minimise likelihood of infill loss

3.7 Capture of Microplastics on Roads

Our analysis has suggested that roads are a pathway for automotive tyre wear, road paint and building paint. Such microplastics can thus potentially be captured through the drainage infrastructure (e.g. gully pots or sustainable drainage systems (SuDS), or through street cleansing.

While the potential for capture exists, there is a lack of awareness among authorities of the potential role of drainage infrastructure (including SuDS) and street cleansing to capture microplastics. This is due to the emerging nature of the issue. Accordingly, there is a lack of understanding of the effectiveness of different techniques in capturing microplastics, and of what ‘best practice’ would look like.

Notwithstanding the lack of awareness, there is no financial or regulatory incentive for road authorities to enhance the provision of the street cleansing or drainage infrastructure to capture microplastics.

The problem drivers relating to capture of microplastics on roads can be summarised as follows:

- A lack of awareness of the issue of automotive tyre wear, road paint and building paint as a source of microplastics, and of the potential role of road drainage infrastructure and street cleansing in capturing microplastics, and understanding of best practice techniques for doing so.
- Insufficient financial or regulatory incentives to capture automotive tyre wear, road paint and building paint through road drainage infrastructure and street cleansing

3.8 Capture of Microplastics in Wastewater Treatment

Some microplastics that enter the sewage system can be captured in wastewater treatment (WWT) facilities. Such microplastics are typically captured within sludge, of which approximately 50% across Europe is applied to land and 50% incinerated.

The existence of combined sewer overflows means that some microplastics may enter the aquatic environment directly without any being removed. For wastewater that does go to treatment facilities, there is no standard method for measuring the capture rate of microplastics. This means that results from studies undertaken to date on different types of treatment facilities are not directly comparable, and it is thus not possible to identify best practice in removal of microplastics from effluent. There is no known method of removing microplastics from sludge.

The problem drivers relating to capture of microplastics in wastewater treatment can be summarised as follows:

- Spills from CSOs mean some microplastics enter the aquatic environment without being subject to any treatment
- There is no standard approach to measuring capture rate of microplastics in wastewater treatment facilities meaning it is not possible to identify best practice in removal from effluent

There is no known method of removing microplastics from sewage sludge.

4.0 Baseline Scenario

The baseline scenario is based upon a 'business as usual' attitude to microplastics emissions. There are no initiatives which are expected to significantly affect microplastics emissions at this time. For each emission source there are drivers that may increase or decrease the level of emissions (beyond general market trends), however in most cases data is not available to model the effects of these fully.

Figure 11 presents the results of the baseline scenario analysis. Where an upper and a lower emissions estimate has been calculated the average is given. For the sources in question, the ***microplastics emitted to surface waters*** are estimated to increase by around 27% by 2035. Specific figures and calculation methodology are presented in Appendix A.3.8.7. Most emissions see increase of 20-30% over this time which is generally related to projected economic growth. This in itself may be an overestimation for sources such as paints that are considered to be a mature market in the EU. Emissions from the washing of clothing are the only source that is projected to decline (by 5%) due to the increases linked to population growth (which is predicted to be relatively slow) being less than the improvements to WWT plants that are planned as part of the Urban Wastewater Treatment Directive. These include a 12% increase in tertiary treatment across the EU by 2013. Artificial turf is the other anomaly with emissions expected to double by 2035 due to the large projected increases in the number of installations over this time.

The ETRMA[148] has identified a number of factors that are expected to influence the scale of tyre wear emissions. It specifies:

- **Drivers for reduced tyre wear emissions**
  - *"The legal requirement of the installation of Tyre Pressure Monitoring Systems (TPMS) on cars since 2014 and most likely on HDVs (Heavy Duty Vehicles) in the next specific legislation review, is an example of short term solution that helps to keep the pressure maintenance of tyres up to the best performing status, thus reducing the wear rate of the tread due to wrong inflation pressure."*

  - *"The increase of automaticity and connectivity of vehicles will definitively represent a step forward to also reducing the "driving behaviour" effect on tyre wear. Traffic and start-stop conditions of current typical city journey will be reduced. If on top of this we also add the increased use of car sharing transport model, we can even predict a non-linear relation between population's increased needs of mobility and corresponding number of cars on the roads."*

  and

- ***Drivers for increased tyre wear emissions***
  - *"The electrification of transportation will also affect the wear rate of tyres. Tyres on EVs (electric vehicles) get to see extremes due to some of the features on an electric vehicle. Electric vehicles tend to weigh about 20 to 30% more than their internal combustion counterparts mostly due to the batteries. They also deliver instant torque.*

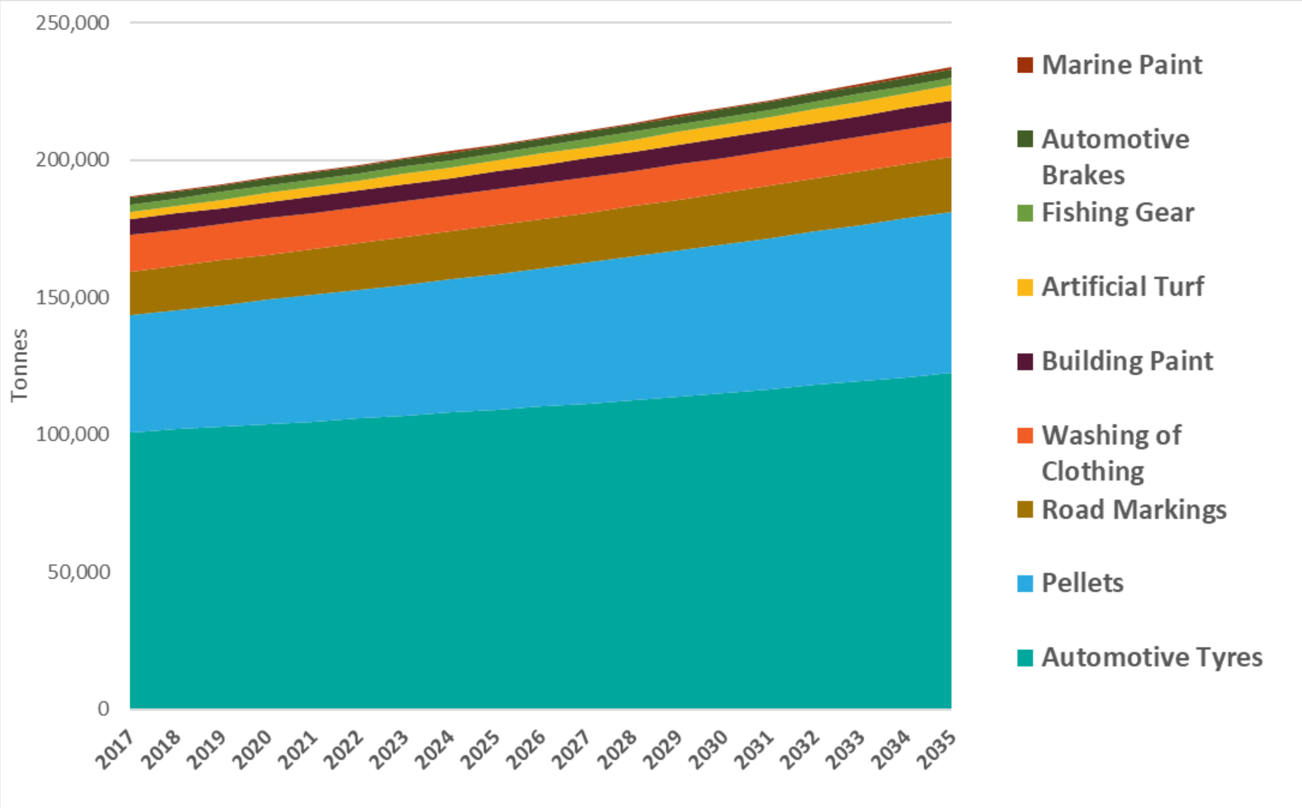
148 Personal communication with ETRMA

Vehicle weight is expected to play a role in emission factors, since there is a positive correlation between weight and non-exhaust emissions. Accelerating hard at low speeds will also affect tyre life. However, more research is needed into the exact impact EVs additional weight and instant torque has on emission factors."

These factors have been modelled for the tyre wear baseline which shows that TMPS will reduce emissions by 5.5% of that time, but the increase in electric vehicles is projected to almost offset this with an increase of 4%. The overall growth of 21% is largely due to the projected increase in car ownership over this time which leads to a 35% increase in vehicle km[149]. This is also the largest source increase by tonnage.

Pre-production plastics (pellets) are expected to increase in line with economic growth by 38% and therefore see the second largest increase by tonnage.

Figure 11 – Projected Microplastics Emissions to Surface Water Growth 2017-2035



[149] Tetraplan A/S (2009) *Traffic flow: Scenario, Traffic Forecast and Analysis of Traffic on the TEN-T, Taking into Consideration the External Dimension of the Union.*, Report for European Commission, December 2009

5.0 Objectives

Section 2.0 has provided a problem definition, including detail on the:

- estimated annual flow of microplastics into the aquatic environment from each source; and
- current state of knowledge on impacts and who is impacted.

Section 3.0 has identified the problem drivers – why these flows occur.

In this section we identify the objectives. The Better Regulation Toolkit notes that after the analysis of the problem, both general and specific objectives can be set.[150] These are defined as follows:[151]

- general objectives – these are Treaty-based goals which the policy aims to contribute to – they describe the high level ambition of intervention; and
- specific objectives – these set out concretely what the policy intervention is meant to achieve. They should be broad enough to allow consideration of all relevant policy alternatives without prejudging a particular solution.

5.1 General Objectives

As an emerging issue, the more we learn, the greater the apparent cause for concern about the damage that is being done to the terrestrial, freshwater and marine environments from microplastics. Accordingly, a strong argument can be made that the precautionary principle should be applied. We may not fully understand the impacts of microplastics in the terrestrial, freshwater or marine environment, but as shown in Task 1, we know the impacts are negative, and expect that furthering our understanding will highlight new and potentially more severe impacts. This strongly suggests that we should do all that can reasonably be done, within bounds of acceptable cost, to address the problem.

The precautionary principle is enshrined in EU Law. Article 191(2) of the Treaty on the Functioning of the European Union (TFEU) states that:[152]

> *Union policy on the environment shall aim at a high level of protection taking into account the diversity of situations in the various regions of the Union. It shall be based on the* ***precautionary principle*** *and on the principles that* ***preventive action should be taken****, that* ***environmental damage should as a priority be rectified at source*** *and that* ***the polluter should pay.***[153]

Accordingly, the general objectives adopted for this study are as follows:

- To reduce microplastic leakage into the environment;
- Reducing the emissions of microplastics at source as a priority; and

[150] Specifically under Tool #13: How to Set Objectives

[151] Noting that *operational* objectives are set after identifying the preferred option(s), and, being defined in terms of the deliverables of policy actions are typically option-specific.

[152] OJEU (2012) Consolidated Version of The Treaty on the Functioning of the European Union, Official Journal of the European Union, 26th October 2012, available at http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:12012E/TXT&from=EN

[153] Emphasis added

- For the polluter pays principle to be respected in the case of the microplastic sources.

5.2 Specific Objectives

The specific objectives for the key sources and pathways are identified in below;

- **Automotive Tyre Wear**
    - Ensure there is a sufficient financial, regulatory and/or reputational incentive for manufacturers to develop tyres that abrade at lower rates
- **Pre-production Plastics**
    - Ensure there is a sufficient financial, regulatory and/or reputational incentive for actors in the pre-production plastics supply chain to implement best-practice measures to reduce loss
- **Synthetic Clothing**
    - Increase awareness amongst manufacturers/brands and retailers of the issue and of mitigating measures that can be taken in the design and manufacture of synthetic garments
    - Ensure sufficient financial, regulatory and/or reputational incentives for manufacturers/brands and retailers to produce and sell clothing that sheds fewer, or no, synthetic microfibres
    - Ensure widespread consumer awareness as to the actions they can take to minimise the loss of synthetic fibres when washing clothing
    - Ensure sufficient financial, regulatory and/or reputational incentives for washing machine manufacturers/brands to develop methods to capture synthetic microfibres
- **Road Markings**
    - Increase understanding of the relative rates of loss of microplastics from different types of road markings
    - Increase awareness among road markings users of the issue of microplastics from road markings
    - Increase awareness among road authorities of alternatives to road markings
- **Building Paint**
    - Encourage research into the use of alternatives to polymers as binders in building paint
    - Increase awareness among contractors and the general public of the problem of microplastics from removing old paint and of mitigating actions they can take
- **Artificial Turf**
    - Ensure there is a sufficient financial, regulatory and/or reputational incentive for pitch operators to implement best practice measures in specifying the facility and managing its use
    - Encourage the uptake of contracts whereby installers are also financially responsible ongoing supply of replacement infill to increase the incentive to design for the prevention of infill loss
    - Ensure a sufficient financial, regulatory or reputational incentive for pitch operators to use alternative infill material where this is environmentally preferable
- **Capture of Microplastics on Roads**
    - Increase awareness of the issue of automotive tyre wear, road paint and building paint as a source of microplastics, and of the potential role of road drainage infrastructure and street cleansing in capturing microplastics, and understanding of best practice techniques for doing so.

    - Ensure sufficient financial and/or regulatory incentives to capture automotive tyre wear, road paint and building paint through road drainage infrastructure and street cleansing
- **Capture of Microplastics in Wastewater Treatment**
    - Seek to minimise spills from CSOs to avoid microplastics entering the aquatic environment without being subject to any treatment
    - Develop a standard approach to measuring the capture rate of microplastics in wastewater treatment facilities making it possible to identify best practice in removal from effluent
    - Encourage research into preventing microplastics from being contained within sewage sludge

6.0 Identification and Screening of Policy Options

A 'longlist' of measures considered for the respective sources, and in relation to capture on roads and wastewater treatment was developed. In line with Tool #14 of the Better Regulation Toolkit, this longlist was subsequently screened according to the following criteria:

- Legal feasibility
  - Options must represent the principle of conferral. They should also respect any obligation arising from the EU Treaties (and relevant international agreements) and ensure respect of fundamental rights. Legal obligations incorporated in existing primary or secondary EU legislation may also rule out certain options
- Technical feasibility
  - Technological and technical constraints may not allow for the implementation, monitoring and/or enforcement of theoretical options
- Previous policy choices
  - Certain options may be ruled out by previous Commission policy choices or mandates by EU institutions
- Coherence with other EU policy objectives
  - Certain options may be ruled out early due to poor coherence with other general EU policy objectives
- Effectiveness and efficiency
  - It may already be possible to show that some options would uncontrovertibly achieve a worse cost-benefit balance than some alternatives
- Proportionality
  - Some options may clearly restrict the scope for national decision making over and above what is needed to achieve the objectives satisfactorily
- Political feasibility
  - Options that would clearly fail to garner the necessary political support for legislative adoption and/or implementation could also be discarded
- Relevance
  - When it can be shown that two options are not likely to differ materially in terms of their significant impacts or their distribution, only one should be retained

The results of the screening exercise is shown in Appendix A.5.0. After the screening exercise, the measures were presented to relevant stakeholders during three stakeholder meetings held in Brussels as well as subsequent telephone conferences and personal communications (an overview of these meetings is provided in the Consultation Synopsis Report for this project). This narrowed down the feasibility of some of the key measures which were then agreed with the Commission for further analysis of their impacts. The results of the Open Public Consultation (OPC) were also used to further refine these decisions (taking into account that the measures presented in the OPC were chosen early in the project and do not fully reflect the final options that were taken through for analysis). The full results of the OPC can be found in the Consultation Summary for this project and key data points and results are referred to in the current report where appropriate. In the sections below, we show the measures selected for further analysis.

7.0 Assessment of the Impacts of Options

The following sections provide an overview of the chosen measures, which can be combined together to create an option, and their impacts. Impacts are assessed by estimating the emission reduction potential (both at source and projected to enter waterways) and looking at the potential costs for these reductions as a ***cost per tonne prevented at source***[154]. The benefits associated with reduced emissions both at source and to waterways cannot be fully quantified. However, the known negative impacts on marine life, and areas of concern that are receiving increased scientific investigation in respect of microplastic emissions from these sources (such as impacts on soil fauna and freshwater fauna) are described in Section 2.0. Thus, in reducing emissions relative to the baseline, the extent to which these negative impacts are experienced will also be lower.

This is an area of great public concern, and there will thus be a benefit, albeit one not amenable to quantification, in that the public will be reassured that concrete action is being taken to address the emission of microplastics from these sources. Where co-benefits are likely, these are identified.

A full discussion of the various measures and options is provided in Appendix A.6.0. The key points are presented below.

7.1 Automotive Tyre Wear

The measures taken forwards for detailed analysis are as follows:

- **Development of a standard measure of tyre tread abrasion**
    - Such a test will be used to determine the rate at which different tyres abrade (mg/km) under standard conditions. While factors *external* to the tyre such as vehicle weight, driving style, road conditions and level of inflation all have a bearing on real world rates of abrasion, such a test will provide details on the factors that are within the control of tyre manufacturers.
    - Such a test will of itself not lead to any reduction in microplastic emissions from vehicle tyres, but it will be the basis for the subsequent measures for tyres detailed below:
- **Inclusion of tyre tread abrasion rates in the EU Tyre Label Regulation (EC/1222/2009)** (once a standard measure of tyre tread abrasion has been developed)
    - Using the standard A-G rating, this *demand-side* measure would ensure that consumers are adequately informed about the likely rate of tyre tread abrasion for each tyre placed on the market.
- **Using the Type Approval Regulation (EC/661/2009) to restrict the worst performing tyres (in respect of tyre tread abrasion) from the market** (once a standard measure of tyre tread abrasion has been developed)

[154] Although cost per tonne realised into the marine environment is also another potential cost indicator, cost per tonne prevented at source was chosen due to the increased concern for microplastics accumulation in other environments (terrestrial and riverine) and the increasing uncertainty of the estimates further along their projected pathways.

- Similar to the approach used in respect of rolling resistance, this *supply-side* measure would restrict access to the European market to those tyres that meet and exceed this threshold for tread abrasion.

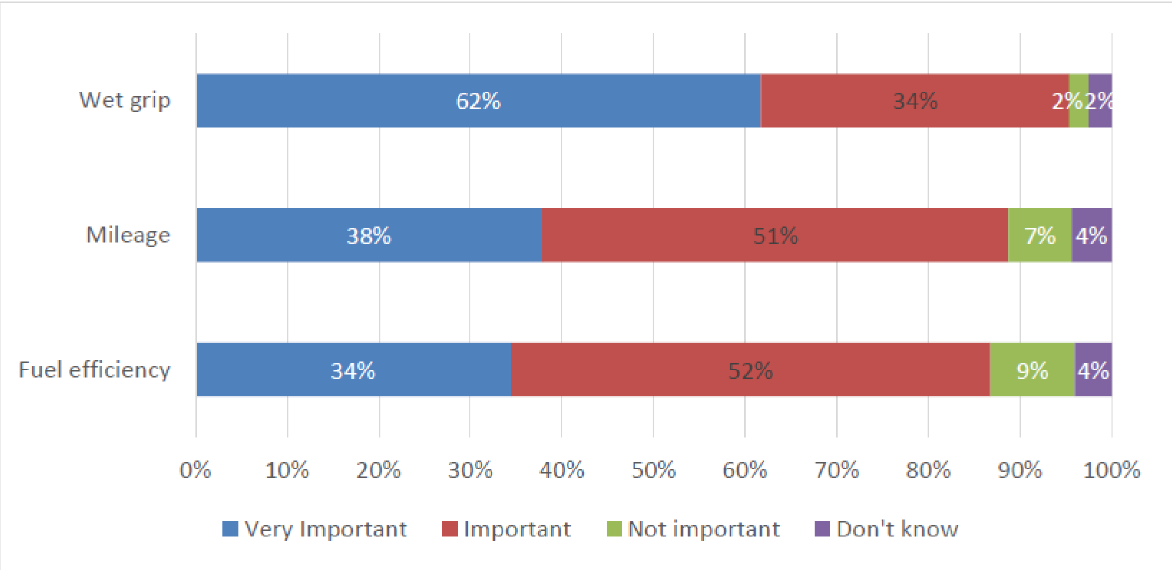
In the sections below:

- We note the strong consumer demand for information about tyre tread abrasion rates;
- Identify the relevant impacts associated with the selected measures; and
- Seek to quantify the costs and the benefits.

Consumer Demand for Information

There is strong consumer demand for information about tyre wear, as illustrated in the findings of a study for DG ENER in March 2016. The relative importance of wet grip, fuel efficiency, and 'mileage' (reflecting the durability of the tyre) as indicated in the C1 end-user survey (which includes consumers defined as private persons buying tyres for their own private cars, as well as leasing companies buying tyres for their lease cars) are shown in Figure 12.[155]

Figure 12 - C1 End-user Rating of Fuel Efficiency, Mileage and Wet Grip Importance



Source: Viegand Maagøe A/S, 2016

The authors of the study note that 'mileage is a common parameter used to express the durability of tyres as a distance in miles or kilometres', and that the mileage of a tyre is directly correlated to the tyre wear factor (amount of tread lost per kilometre). The authors also note that abrasion (i.e. the removal of materials from the tyre when it interacts with the road surface) 'is related to tyre mileage, since both are linked closely to the tyre wear.'

[155] Viegand Maagøe A/S (2016) Review Study on the Regulation (EC) No 1222/2009 on the Labelling of Tyres, Final Report to DG ENER of the European Commission, March 2016. Available at http://www.labellingtyres.eu/downloads/Final_report-Review_study_on_labelling_of_tyres.pdf

The Commission's own Impact Assessment from 2008 on the labelling of tyres also notes the importance that consumers place on this aspect, stating that:[156]

> *Market surveys in addition show that wear (i.e. "long lasting tyre") is the most important criteria in consumers purchasing decision.*

Accordingly, in the absence of labelling of tyre tread abrasion rates, there is a market failure arising from the lack of information for end-users. This market failure affects:

- **End-users** (consumers, companies or municipalities owning small or larger fleets such as leasing companies, and road transport operators) who do not benefit from the savings they could obtain from the use of tyre with lower tread abrasion rates;
- **Tyre producers** who have more difficulties in obtaining a return on their investments in R&D to reduce the tyre tread abrasion rate; and
- **Society as a whole** resulting from a reduced rate of tyre tread abrasion which is expected to reduce the rate at which particles are generated from the usage of tyres, with associated benefits in terms of air quality and marine microplastics.

Labelling of tyre tread abrasion rates could also improve competition between tyre producers while providing a level playing field for all. Producers may both have incentives to provide better-performing tyres on the market and benefit from reduced barriers to entry as brand reputation may lose its importance compared to objective tyre performance characteristics. New entrants will be able to demonstrate that they produce well-performing tyres in respect of tread abrasion rates.

7.1.1 Measure 1 – The Development of a Standard Measure of Tyre Tread Abrasion Rate

The development of a standard measure of tyre tread abrasion is an essential pre-condition for the successful introduction of either of the proposed subsequent measures, namely:

- Inclusion of tyre tread abrasion rates in the EU Tyre Label Regulation (EC/1222/2009); and
- Using the Type Approval Regulation (EC/661/2009) to restrict the worst performing tyres (in respect of tyre tread abrasion) from the market.

Accordingly, we identify the costs associated with the development of a standard measure of tyre tread abrasion rate, and to whom these costs might accrue, but we do not consider any benefits. These will be considered in respect of the application of the standard measure of tyre tread abrasion rate in the two subsequent measures.

As explained in Appendix A.6.1 it is possible to test for abrasion, and tyre manufacturers already perform their own tests. Road based tests cost €5,000 to €10,000, or potentially up to €40,000 per tyre model, depending on the distance travelled. [157,158]

---

[156] European Commission (2008) Commission Staff Working Document: Accompanying document to the Proposal for a Directive of the European Parliament and of the Council on Labelling of Tyres with Respect to Fuel Efficiency and Other Essential Parameters – Impact Assessment, 13.11.2008

[157] Personal communication with Dr Stuart Cook (Director of Research), Paul Brown (Head, Advanced Materials & Product Development), Dr Andy Chapman (Senior Research Fellow), and Dr Pamela Martin (Advanced Materials and Product Development), Tun Abdul Razak Research Centre, October 2017

[158] Personal communication with Joerg Burfien, Head of Global Standards and Regulations, Continental AG

The average cost of testing per individual tyre placed on the market will be between €0.03 and €1.43, but higher for some less popular models, and lower for the most popular models (as the cost of testing for a given model will be shared between fewer/more tyres actually placed on the market).

While testing for tyre tread abrasion already takes place, this is not based on standardised test procedure. To enable the development of a tyre label rating for tyre tread abrasion a standardised test procedure would have to be developed and undertaken. One challenge would be in getting the major brands to agree to a standardised test procedure. They will all have invested in developing their own approach to determining the likely rate of tyre tread abrasion for their own tyres, and may be reluctant to switch over to a different method. In large part such resistance may be because they would prefer to use their own approach in order to have continuity of data with that previously gathered. There may also be a concern that in divulging their own preferred test method, which presumably will be close to their own current approach, they may be giving away sensitive data that they feel gives them a competitive advantage at present.

Given that there will be time and resource implications associated with the development of, and agreement on, a standardised test method, a high level estimate of €500,000 to €1.5 million is made to account for the costs. We anticipate these would fall largely on industry. The assumptions underpinning this estimate is included in Appendix A.6.1.

In developing a standardised test method for tyre abrasion there is also the potential to amend the wet grip, rolling resistance and external noise tests, by incorporating them within the new test for tyre abrasion. Testing for rolling resistance, wet grip and external noise is currently undertaken on new tyres.[159] However, there is no testing for the performance against any of these criteria over the lifespan of a tyre, meaning the consumer does not have any indication as to the extent to which these properties (of wet grip, rolling resistance and external noise) will vary over the tyre's lifetime. There is an argument that such information is of importance to the consumer. Indeed, the UK's Automobile Association notes, in its advice to drivers, that:[160]

> *Wet grip in particular gets worse as the tread on your tyres wears*

Given that safety (and rolling resistance and external noise) is of concern to motorists throughout the lifetime of the tyre, and not just when it is new, there is arguably a case for the wet grip and other tests to be revised and incorporated within a new test for tyre tread abrasion rates.

For example, if the test were to be conducted over 20,000 km, wet grip, rolling resistance and external noise could each be tested at the outset, then at 10,000 kms and at 20,000 kms. Such testing would appear likely to give a better indication of the lifetime performance of the tyre.

7.1.2 Measure 2 - Include Tyre Tread Abrasion Rates on the Tyre Label

Including tyre tread abrasion rates on the tyre label is a demand-side measure. It will inform consumer choice and hence is expected to lead to demand for lower abrasion rate tyres.

---

[159] See https://www.blackcircles.com/general/tyre-labelling/tyre-testing
[160] See https://www.theaa.com/driving-advice/safety/tyre-life-and-age

As the costs of display of the labelling scheme, as identified in the 2008 Impact Assessment are considered to be marginal on a per tyre basis, it is anticipated that the *additional* costs associated with inclusion of tyre tread abrasion rates will be similarly marginal.[161]

Given that no standardised test is currently available to illustrate the distribution of tyre tread abrasion rates across the current stock of tyres on the EU market, it is not possible to fully characterise the range in abrasion rates across all tyre models. However, as shown in Appendix A.6.1.3. It is possible to obtain an indicative estimate of the distribution using data from the Uniform Tire Quality Grading (UTQG) test used to measure tread wear in the United States. On this basis we assume, arguably conservatively, that the worst performing tyres exhibit double the abrasion rate (mg/km) of the best performing tyres.

The existence of a rating for abrasion rate on the tyre label could lead to two effects:

1) A move by consumers towards existing tyre models that exhibit a lower abrasion rate; and
2) A move on the part of producers towards the manufacture of tyres that have a lower abrasion rate than current models.

The combined effect of these will be an overall reduction in the average rate of tyre abrasion. However, the speed of the market transformation will depend upon a number of factors including:

- The number of tyre models exhibiting higher than average performance in respect of abrasion rates (i.e. lower abrasion rates than the average);
- The cost of these models relative to other models on the market; and
- The performance of these models in respect of the other attributes detailed in the tyre label (i.e. their performance in terms of wet grip, rolling resistance, and external noise).

If the annual improvement (i.e. reduction) in tyre tread abrasion rates across the stock of tyres in use were of a similar magnitude to that seen over the period 2013 – 2015 in respect of fuel efficiency (rolling resistance coefficient) and wet grip, then we might expect an annual 1% reduction in the rate at which the stock of tyres in the EU abrades.[162] This is described in Figure 13 and Table 18 as 'Tyre Label (Low).'

A more rapid shift may be achievable. If the measurements for wet grip, rolling resistance and external noise were revised to be part of a 20,000km tyre abrasion rate test, then the combined effect may be to increase the rate of market transformation. The rationale for assuming a higher level of market transformation under such circumstances is that in order to score highly in wet grip, rolling resistance and external noise over the lifetime of a tyre, a tyre that abrades at a lower rate might be desirable (in order to maintain these characteristics as much as possible over the 20,000km of the test). Accordingly, if this were the case, tyres that abrade at lower rates might score well on other attributes that are rated on the tyre label, thus increasing demand for such tyres.[163]

[161] European Commission (2008) Commission Staff Working Document: Accompanying document to the Proposal for a Directive of the European Parliament and of the Council on Labelling of Tyres with Respect to Fuel Efficiency and Other Essential Parameters – Impact Assessment, 13.11.2008

[162] Viegand Maagøe A/S (2016) Review Study on the Regulation (EC) No 1222/2009 on the Labelling of Tyres, Final Report to DG ENER of the European Commission, March 2016. Available at http://www.labellingtyres.eu/downloads/Final_report-Review_study_on_labelling_of_tyres.pdf

[163] Revising the tests in this way, to operate alongside a new test for tyre abrasion, may offer a number of possible co-benefits in terms of lifetime improvements in wet grip, rolling resistance and external noise.

For illustrative purposes, we assume a 2% per year shift under such circumstances (capped at 20%). This is described in Figure 13 and Table 18 as ‘Tyre Label (High).’

Given that consumers are likely to be more interested in the overall ‘mileage’ of a tyre under standard conditions, and that the mileage relates to both the tread abrasion rate and the tread depth (see Section 2.2.1), it may be sensible to show both side by side on the tyre label.[164]

A possible co-benefit of reduced tyre abrasion, as explained in Appendix A.6.1.2, may be in relation to air quality. This would occur if reduced rates of abrasion also meant reduced levels of airborne particulate matter (PM_{10} and $PM_{2.5}$) from tyres. However more detailed testing is required to improve current knowledge, and determine the nature of any potential changes in emissions to air from tyre abrasion.

7.1.3 Measure 3 - Using the Type Approval Regulation to restrict the worst performing tyres (in respect of tyre tread abrasion) from the market

Restricting the availability of tyres with the highest rates of abrasion under the type-approval legislation is a supply-side measure. It is expected to shift industry production towards lower abrasion rate tyres.

In the absence of a standardised test and resulting performance data in respect of tyre abrasion, and on the distribution of tyre abrasion rates across the EU market, it is not, at present, possible to identify where a ‘reasonable’ threshold for permitted tyres abrasion rate may lie.

Determining such a threshold point, and providing supporting justification for it, would have to take into account the other performance criteria of relevance alongside tyre abrasion rates.

Accordingly, for illustrative purposes, we provide an indication of the effect of using the Type Approval Regulation in 2020 to prevent the worst performing tyres from being placed on the market, such that the effect is a 10% drop in the tonnage of tyre wear abraded at source. We further illustrate the effect of a similar incremental restriction coming into place in 2025.

7.1.4 Impacts of Measures on Tyre Abrasion

Figure 13 shows the illustrative reduction potential of the tyre measures relative to the baseline out to 2035. The ‘Combined’ measure represents the illustrative effects of both the Measure 2 Tyre Label (High) and Measure 3 Type Approval measures (which both incorporate Measure 1 as a pre-condition).

Table 18 shows the cumulative tonnage reductions in emissions at source, and final entry to surface waters, for each of the measures. As previously noted, because of a lack of data on the effectiveness of the measures it is not a straightforward matter to identify costs per tonne prevented at source. However, for the two tyre label measures, we identify average annual costs per tonne prevented at source over the period out to 2035, based on the average cost per tyre of the testing procedure. The cost effectiveness of the measures is highly dependent upon the effectiveness of the measures. For the type approval regulation, and combined measures, the estimates of cost-effectiveness are very tentative, and should be treated solely as illustrative of what the costs would be if the assumed reductions were achieved, and with no additional costs incurred beyond that of testing. In reality,

[164] The risk of simply combining the two into a ‘mileage’ figure is that the industry response may be to simply increase tread depth, which will do nothing to reduce the abrasion rate.

there may be costs associated with redesign of tyres, but we cannot yet know the extent to which this will take place in response to the introduction of the measures. Furthermore, tyres that abrade at a lower rate may be more expensive, although this could be offset to the consumer by a tyre longer lifetime.

Figure 13 - Reduction Potential of Tyre Measures

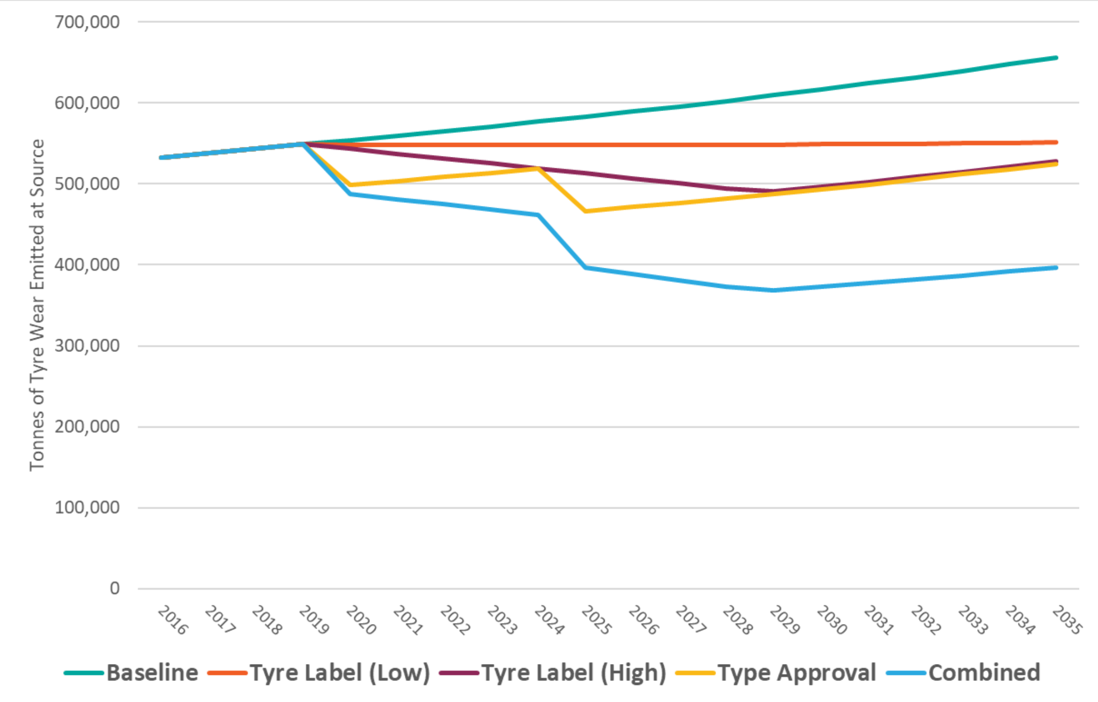


Table 18 - Reduction Potential of Tyre Measures

Measure		Cumulative Emissions 2017-2035 (tonnes)		Cumulative Reduction from Baseline 2017-2035 (tonnes)		Annual Cost per Tonne Prevented *at Source*
		Source Emissions	Surface Water Emissions	Source Emissions Reduction	Surface Water Emission Reduction	
Baseline		***11,200,000***	***2,100,000***	-	-	
Measure 2 - Tyre Label	Low	10,900,000	2,040,000	300,000	60,000 (3%)	Circa €11,000
	High	10,400,000	1,900,000	800,000	200,000 (8%)	Circa €4,000
Measure 3 -Type Approval		10,100,000	1,900,000	1,100,000	200,000 (10%)	Circa €3,000
Combined		8,700,000	1,600,000	2,500,000	500,000 (22%)	Circa €1,300

Note: Emissions figures rounded to nearest 100,000 or 10,000 for those less than 100,000.

7.2 Pre-Production Plastics

The measures taken forwards for detailed analysis are as follows:

- **Amending the Polymer Production BREF to include best practice pellet loss prevention measures as BAT**
  - Member States to address prevention and control of releases of micro-plastics to water as part of the revision of permits of plastics production plants required by the IED to align industry practices with Best Available Techniques; Include in future work on BREFs regarding water pollution from the plastics industry the identification of BAT for prevention and control of releases of micro-plastics to water.
  - This approach could incorporate the best practice measures as described in Operation Clean Sweep - and subsequently verified and enhanced through an expert group – into the Polymer Production Best Available Technique (BAT) Reference Document (BREF) under the Industrial Emissions Directive (IED).
  - All polymer producers in Europe would thus be required to implement BAT in respect of preventing pellet loss, and would be subject to regulation and potential enforcement action as per other aspects of their Environmental Permit
- **Regulation on the Transport of Pellets**
  - This would be a new regulation specifically covering the transport of pellets from and to facilities. All operators undertaking such transportation would be required to implement best practice approaches, again derived from expert knowledge, and further developing the approaches already pioneered by industry via Operation Clean Sweep.
- **Regulation on Plastic Converters**
  - The circa 50,000 plastics converters in the EU are mostly SMEs to whom the polymer production BREF does not apply. This new regulation would thus require all plastic converters in the EU to implement best practice measures to prevent pellet loss. Environmental regulators in each Member State would be required to ensure adherence to the Regulation
- **Regulation Requiring Supply Chain Accreditation of Adherence to Best Practice**
  - This regulatory measure would require those placing plastics on the market (large businesses in the first instance) to ensure their entire supply chain demonstrates best practice in the prevention of pellet loss.
  - Akin to the way in which the Timber Regulations operates, adherence to best practice can be demonstrated through the use of accreditation bodies that certify adherence to best practice criteria. This would involve regular audits along the entire supply chain to ensure that best practice measures to prevent pellet loss, and clean up any pellet spills, are being implemented and adhered to.
  - This measure would include anyone directly placing plastics products on the market that were manufactured outside the EU, thus ensuring a level-playing field between the EU plastics producers and converters, and those outside of the EU wishing to sell in.

In the sections below:

- We consider the structure of the pre-production plastics supply chain in the EU, and what this means in respect of possible measures;
- Identify the relevant impacts associated with the selected measures; and
- Seek to quantify the costs and the benefits.

Structure of the Plastics Industry

Polymer producers only represent a small proportion of the companies within the European plastics industry. The Polymer Production Best Available Technique Reference Document (BREF) under the Industrial Emissions Directive (IED) is focused on polymer producers. As stated in the BREF:[165]

> *Polymer companies produce a variety of basic products, which range from commodities to high added-value materials and are produced in both batch and continuous processes covering installations with a capacity of some 10000 tonnes per year up to some 300000 tonnes per year.*
>
> *The basic polymers are sold to processing companies, serving an immense range of end-user markets.*

Plastics Europe notes that the European plastics industry comprises 60,000 companies, ***mainly small and medium enterprises in the converting sector***. Polymer producers are represented by Plastics Europe, converters are represented by European Plastics Converters (EuPC) and machine manufacturers are represented by EUROMAP.[166]

Plastics Europe notes that its members are among the most important polymer producers in the world, and indicates that 54 companies are members.[167] EUROMAP represents around 1,000 companies.[168]

EuPC represents close to 50,000 companies, and states that:[169]

> *Plastics converters (sometimes called "Processors") are the heart of the plastics industry. They manufacture plastics semi-finished and finished products for an extremely wide range of industrial and consumer markets - the automotive electrical and electronic, packaging, construction and healthcare industries, to name but a few.*
>
> *Plastics Converters buy in raw material in granular or powder form, subject it to a process involving pressure, heat and/or chemistry and apply design expertise to manufacture their products. They often undertake additional finishing operations such as printing and assembly work to add further value to their activities*

Accordingly, the vast majority of the companies in the sector are small and medium sized enterprises (SMEs).

Given the European Commission's desire to minimize regulatory burden on SMEs, this presents some interesting challenges when considering the appropriate policy response(s).[170] It is informative, in this regard, to understand the views of plastic converters about the current

---

[165] European Commission (2007) Reference Document on Best Available Techniques in the Production of Polymers, August 2007, available at http://eippcb.jrc.ec.europa.eu/reference/BREF/pol_bref_0807.pdf
[166] Plastics Europe (2017) The European Plastics Industry, available at http://www.plasticseurope.org/plastics-industry.aspx
[167] Plastics Europe (2017) Our Members, available at http://www.plasticseurope.org/plastics-industry/our-members.aspx
[168] EUROMAP (2017) About EUROMAP, available at http://www.euromap.org/about-us/about-euromap
[169] EuPC (2017) EuPC homepage, available at http://www.plasticsconverters.eu/
[170] See European Commission (2011) Minimizing Regulatory Burden for SMEs: Report from the Commission to the Council and the Council and the European Parliament, 23.11.2011, available at http://ec.europa.eu/smart-regulation/better_regulation/documents/minimizing_burden_sme_en.pdf

regulatory environment. A recent report on the competitiveness of the European plastics converting industry, produced for EuPC, offers some useful insights.[171]

The authors of the report undertook a survey of a representative sample of 326 EU plastics converters from 19 European countries and more than 20 expert interviews with mostly senior company representatives. The authors, in presenting their findings note the view that:

> *The bureaucratic and regulatory framework conditions within the EU are assessed as mostly stable for plastics converters. Nevertheless, cost burdens from direct taxes or necessary effort to comply with domestic and EU-driven regulations and requirements have worsened substantially compared to previous years. This development poses a massive threat to the competitiveness of EU plastics converters. Still, most converters expect a further worsening of the situation.*

This is not a surprising view to be expressed, given the desire of any industry to avoid further regulation. However, of greater relevance to the question of preventing pellet loss, the authors go on to report the view of plastic converters that:

> *The level of fragmentation from domestic legislation, regulations and bylaws, driving the framework conditions, is assessed as too high and still far from a perfectly harmonized European single market. The root cause for this fragmentation can be found within the member states. EU legislation, by nature aiming at a legislative level playing field, is slowly or sometimes even not adopted to national law by the member states. Other factors further pushing the level of fragmentation are different domestic bylaws and authorities charged with the enforcement of legislation. Thus, companies need to adjust to these differences within the EU market with additional administrative effort. Key drivers for this fragmentation on a national level are different requirements for consumer safety, the use of raw materials, for processing technologies and approvals to sell different plastic products.*

This strongly suggests that any policy measures that seek to reduce the loss of pellets from converters should be consistently applied in order to safeguard the functioning of the European single market, in order to minimise the impact on SMEs.

Loss Rates at Different Stages of the Supply Chain

While it is not possible to identify a specific figure, we suspect that percentage losses at plastics producers are likely to be towards the lower end of the 0.01-0.04% range (as identified in Section 2.2.3), while those at converters (including intermediary facilities) may well be towards the higher end of this range. This is for the following three reasons:

- Current level of regulatory attention;
- Level of public scrutiny; and
- Engagement with Operation Clean Sweep

On the first of these, polymer producers, which tend to be large in size, are already regulated under the Industrial Emissions Directive. While pellet loss prevention measures are not specifically included within the Polymer Production BREF, one might reasonably expect that facilities that are

[171] Dr. Wieselhuber & Partner GmbH (2016) Competitiveness of the European Plastics Converting Industry: A European Industry Study. Report to EuPC, June 2016, available at https://www.agoria.be/www1.wsc/webextra/prg/nwAttach?appl=enewsv6&enewsdetid=189927&attach=Attach110523001.pdf

more closely regulated may well have better operational procedures. This is not something that can be demonstrated, but discussions with some stakeholders indicate a view that this is the case.

Secondly, to the extent that the pre-production plastics supply chain is visible to the public, it is the larger producers that will be more well-known, and have in place more stringent Corporate Social Responsibility (CSR) reporting. Again, this suggests a greater effort may be expended on maintaining reputations.

Thirdly, and related to the last point, engagement with Operation Clean Sweep varies. Specifically, Plastics Europe's promotion of Operation Clean Sweep has been considerable in recent years and months. It is understood that 50 percent of Plastics Europe members to whom OCS is applicable have signed the pledge. By volume, this accounts for the majority of plastics production in Europe and the target is for 100% coverage by 2017. By contrast, we understand from some NGOs that EuPC (European Plastics Converters), the trade association for the circa 50,000 plastic converters in the EU, has not been open to engagement with them to the same extent as has Plastics Europe.

This view was supported by stakeholders during the workshop on preventing the loss of pre-production plastics (pellets, powders and flakes) held in Brussels on 27th September 2017.

Costs of Preventing Pellet Loss

The physical changes and improvements to management practices required to prevent pellet spills in the first place, and ensure that any spills are promptly cleaned up or captured, are well characterised. A number of specialist companies already provide services to assist in identifying where pellet management practices are sub-standard, and how pellet loss may be most cost-effectively prevented.

The costs of putting in place the required changes to prevent pellet loss are presented in detail in Appendix A.6.2.2. Over the period from 2020 to 2035, the average annualised cost is estimated to be €390 per year per tonne prevented, assuming 100% prevention of pellet loss is achieved.

The unit costs of physically implementing the best practices to prevent pellet loss are likely to be similar, regardless of the way in which firms are encouraged or required to implement them. However, there are clear differences in the impacts on the European plastics sector depending on the specific measures selected to lead to uptake of best practice. These are discussed in the following sections.

7.2.1 Measure 1 - Amending the Polymer Production BREF

Amending the Polymer Production BREF to require best practice pellet loss prevention measures would only affect EU polymer producers, many of whom may already have taken action to address pellet loss, and for whom the loss rate as a proportion of pellets handled is thought to be lower than for plastics converters.[172] On its own, amending the Polymer Production BREF would thus be expected to result in a smaller overall reduction in pellet emissions than would a regulation on converters.

Even if implemented in parallel with other horizontal measures (i.e. a regulation on the transport of pellets and a regulation on converters – Measures 2 and 3) there is a risk of practices not being

[172] It is understood that 50 percent of Plastics Europe members to whom Operation Clean Sweep is applicable have signed the pledge. By volume, this accounts for the majority of plastics production in Europe and the target is for 100% coverage by 2017.

coherent at the point of loading (and indeed unloading) of pellets. That is to say, a polymer producer could be following what it understands to be best practice, and a haulier could also be following what it understands to be best practice, but they may not be adopting procedures that are compatible in reality.[173]

This option, in common with other horizontal measures, would focus solely on facilities based in the EU. There would be no requirement for those importing pellets to the EU to have implemented best practice to prevent pellet loss at their own facilities. This was felt by stakeholders to be a significant disadvantage of this option, as it would place a financial burden on EU industry that would not be experienced by those operating outside of the EU.

7.2.2 Measure 2 - Regulation on the Transport of Pellets

While some hauliers may specialise in transporting pellets (and powders and flakes), others may only transport them very infrequently, and in very small numbers. Identifying those to include in the regulations (if a *de minimus* threshold is to be applied) will not be straightforward. To achieve the greatest reduction in pellet loss, there should be no *de minimus* threshold. However, this would raise concerns over whether the regulation were a proportionate response to the problem, given that all hauliers would have to be covered unless they were able to prove that they do not, and will never, carry plastic pellets.

More significant is the challenge of ensuring that a regulation on the transport of pellets (and the approaches adopted by hauliers, particularly in respect of loading and unloading) is compatible with the approaches taken by the polymer producers from whom they collect, and the converters to whom they deliver. As noted by a haulier who has been closely involved in the roll-out of Operation Clean Sweep best practices, “Pellet-loss prevention only works if there’s total co-operation up and down the supply chain.”[174] Supply chain practices run vertically, and integration between these stages is key, in particular on reaching agreement as to the process for co-operation when things go wrong, i.e. how to clean-up spillages quickly and effectively.

Finally, this option, in common with other horizontal measures would focus solely on facilities based in the EU. There would be no requirement for those transporting pellets outside of the EU that may end up being imported to EU converters, or indeed made into finished goods and imported into the EU, to have implemented best practice to prevent pellet loss during transportation. This would place a financial burden on EU industry that would not be experienced by those operating outside of the EU.

7.2.3 Measure 3 - Regulation on Plastic Converters

Introducing such a regulation on EU plastics converters (in isolation) would be likely to lead to a greater reduction in pellet loss than amending the polymer production BREF (in isolation), given that the majority of plastic converters are thought to have only taken minimal action, if any, to address pellet loss.

Under such a regulation, the cost of regulation and enforcement would fall to the national regulators, albeit they could then recover costs through the imposition of fees on regulated industry. As reported in the workshop, there is concern among stakeholders that regulators may not

---

[173] Personal communication with Iain Mitchell, Managing Director, John Mitchell Haulage & Warehousing, October 2017

[174] Personal communication with Iain Mitchell, Managing Director, John Mitchell Haulage & Warehousing, October 2017

have the industry expertise to identify what might be best practice in specific circumstances, and would thus choose from a pre-determined list of options. It was felt that there may be a tendency for those inspecting plastics converters to simply go through a checklist and identify where things have, or have not, been done, and would not necessarily be able to offer guidance on how best to achieve best practice. It was also felt this meant an increased risk of being fined or given other punitive sanctions.[175]

However, in common with the other horizontal measures, the risk of practices not being coherent at the point of loading and unloading of pellets would remain. Furthermore, there would be no requirement for plastics converters outside of the EU whose finished goods may be imported into the EU, to have implemented best practice to prevent pellet loss. This would place a financial burden on EU industry that would not be experienced by those operating outside of the EU.

7.2.4 Measure 4 - Regulation Requiring Supply Chain Accreditation

This is a regulatory measure that tackles the entire supply chain (unlike measures 1 to 3, which individually only tackle one part of the supply chain) through working up from the end users of the plastic items (e.g. brand owners who place plastic on the market such as Danone, Coca Cola etc.), all the way up to the top of the supply chain.

This measure would require those placing plastics on the market (i.e. the brand owners) to ensure that their entire plastics supply chain, including all logistics operations, has implemented best practice measures to prevent pellet loss. These best practice measures would build on those developed in Operation Clean Sweep guidance, with an improved emphasis on the safe transport of pre-production plastics. Measures identified as 'best practice' for the purposes of the Regulation would be agreed and endorsed by an expert group (comprising representatives of industry, NGOs, regulators and the European Commission – perhaps hosted by the JRC).

The brand owners would be able to demonstrate their compliance with this best practice through the use of one of a number of accredited, independent, privately operated certification organisations, with independent audit, repeated annually, ensuring continued compliance.

The measure is explained in more detail in Appendix A.6.2.6.

Such an approach would ensure the vertical integration in pellet management practices at the interface between the different stages, such as producers to transporters, and from transporters to converters. This should also apply to water companies that handle 'biobeads' or similar biomedia which are similar in shape and form to pellets, but are also likely to benefit from the improved best management practices associated with OCS.

As is clear from discussion with stakeholders, for pellet loss prevention measures to be successful vertical integration of best practices is essential. Also, due to the incorporation of extra-EU supply chains (as the focus is on plastic goods placed on the EU market, regardless of where they are made), there is no disadvantage for EU producers, transporters and converters relative to those outside the EU who are selling to the EU market

[175] As noted in the workshop, one would ideally need at least ten years of plastics industry experience in order to able to effectively advise on what best practice investments and changes in practices would be most appropriate at a specific site

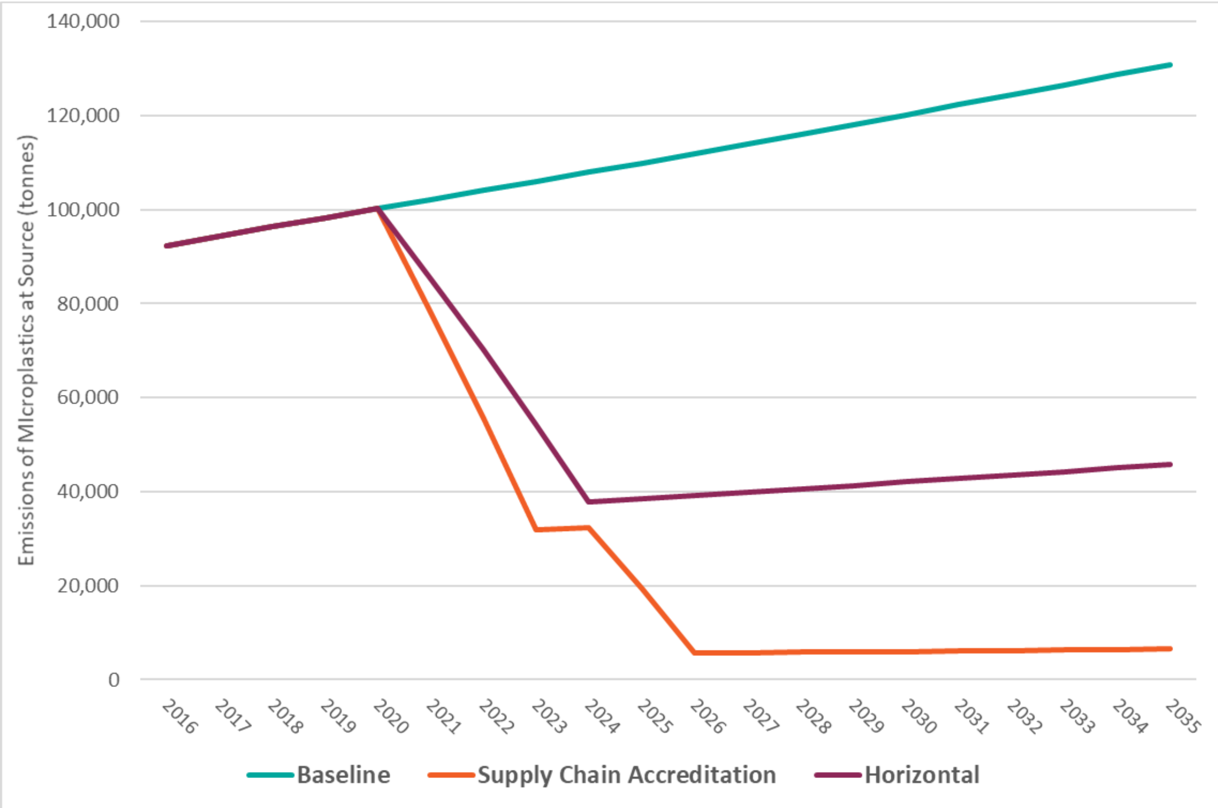
7.2.5 Impacts of Measures on Pre-Production Plastics

Figure 14 shows the anticipated reduction potential for the Regulation Requiring Supply Chain Accreditation, and the three 'Horizontal' measures combined.

We assume that the regulation requiring supply chain accreditation will lead to a 95% reduction in emissions where it is implemented.[176] As explained in further detail in Appendix A.6.2.6, it is assumed to first be applied to all large firms placing more than 5,000 tonnes of plastic per year on the EU market, and their supply chains. This process is completed by 2023, at which point we estimate a 70% reduction in overall emissions.

Then in 2024, the requirement is extended to all those placing plastics on the market, leading to an overall reduction relative to the baseline of 95% by 2026. For the horizontal measures, we assume a lower effectiveness, of 65%, due to the likelihood of practices not being harmonious at hand-over points in the supply chain. Full coverage under these measures is assumed to be achieved by 2024.

Figure 14 - Reduction Potential of Pellet Measures



It is important to note that the Regulation Requiring Supply Chain Accreditation will lead to 'extra-territorial' benefits, outside of the EU. However, these are not accounted for in the impact analysis. Such reductions would therefore be additional.

Of the horizontal measures, if considered in isolation, it is likely that amending the Polymer Production BREF will deliver the smallest reduction in pellet loss. This is for two reasons. Firstly, as

[176] This is based on discussions with an industry expert who explained that implementing required measures would effectively 'seal' a plastic converters facility, meaning close to zero pellet loss. However, given that some pellet loss is inevitable, and to account for the occasional, inevitable spillages, within the supply chain, we make the (perhaps conservative) assumption that only 95% of loses will be prevented.

explained in Section 7.2.1, many producers are thought to have already taken action to address pellet loss. Secondly, as indicated in Table 6, the tonnages handled annually by producers (58-75 million tonnes per annum) are much smaller than those handled by intermediary facilities and processors/converters (100 to 330 million tonnes per annum). However, due to the uncertainty involved, indicative reduction potentials for each horizontal measure in isolation have not been represented graphically.

Table 19 shows the indicative effectiveness and cost-effectiveness of the measures. It's important to recognise that the tonnages presented here are from 2017 to 2035, rather than from 2020, when the measures are assumed to be implemented. The annual cost per tonne prevented at source, over the years 2020 to 2035, takes account of both the physical interventions required to prevent pellet loss, and the costs of inspection and verification.

As explained in Section A.6.2.2, over the period from 2020 to 2035, the average annualised cost (associated with physical interventions) is estimated to be €390 per year per tonne prevented, assuming 100% prevention of pellet loss is achieved. Therefore:

- For the three horizontal measures, with the assumed effectiveness of 65%, the annualised cost per tonne prevented (for the physical measures) will be circa €601 (€390/65%).
- For the regulation requiring supply chain accreditation, with the assumed effectiveness of 95%, the annualised cost per tonne prevented (for the physical measures) will be circa €411 (€390/95%).

As explained in Appendix A.6.2.6, the average annualised cost (associated with upfront and ongoing audits under the regulation requiring supply chain accreditation) is calculated to equate to €516 per year. Therefore, assuming, perhaps generously that the cost of inspection and verification (per tonne handled) is the same under the horizontal measures:

- For the three horizontal measures, with the assumed effectiveness of 65%, the annualised cost per tonne prevented (for inspection and verification) will be circa €793 (€516/65%).
- For the regulation requiring supply chain accreditation, with the assumed effectiveness of 95%, the annualised cost per tonne prevented (for inspection and verification) will be circa €543 (€516/95%).

Combining the annualised costs for the physical measures, and for inspection and verification, gives the following:

- For the three horizontal measures, the total annualised cost per tonne prevented will be circa €1,394 (€601+€793)
- For the regulation requiring supply chain accreditation, the total annualised cost per tonne prevented will be circa €954 (€411 +€543)

It can thus be seen that the supply chain accreditation approach is expected to be more effective, and more cost-effective. However, in acknowledgement of the uncertainties involved in such estimates, we round the estimates of the annualised cost per tonne prevented at source as shown in Table 19.

Table 19 - Costs and Reduction Potential of Pellet Measures

Measure	Cumulative Emissions 2017-2035 (tonnes)		Cumulative Reduction from Baseline 2017-2035 (tonnes)		Annual Cost per Tonne Prevented *at Source*
	Source Emissions	Surface Water Emissions Reduction	Source Emissions Reduction	Surface Water Emission Reduction	
Baseline	***2,200,000***	***1,100,000***	-	-	
Measures 1-3 - Horizontal Measures	1,200,000	700,000	1,000,000	400,000 (36%)	Circa €1,400
Measure 4 - Supply Chain Accreditation	800,000	600,000	1,400,000	600,000 (55%)	Circa €950
Note: Emissions figures rounded to nearest 100,000 or 10,000 for those less than 100,000.					

7.3 Synthetic Clothing

The measures taken forwards for detailed analysis are summarised in the following sections. Full discussion and detailed calculations are provided in Appendix A.6.3.

The options taken forwards for detailed analysis are as follows:

- **Development of a test standard to determine in a consistent manner the rate of fibre release from clothing during washing (and tumble drying)**
    - Such a test standard would likely be carried out on small samples under laboratory conditions rather than on whole garments in standard washing machines.
    - Part of the development of this standard would be to identify which factors affect release of different fibres, and the relative influence of each factor.
    - Such a test will of itself not lead to any reduction in microplastic emissions from clothing, but it will be the basis for subsequent measures detailed below:

- **Setting a Maximum Threshold for Fibre Release, possibly with a new Regulation in line with the Ecodesign Directive (2009/125/EC)**
    - On development of a test standard, manufacturers of clothing would be required to submit samples of the fabrics used for testing before placing on sale in the EU.
    - The samples must be below a maximum threshold of fibre release in order for the clothing to be placed on the EU market.
    - The threshold will be developed based on the testing of a wide range of fabrics that are available on the market.
- **Development of a label for fibre release from washing of clothing to be included under the Regulation for labelling and marking of the fibre composition of textile products (EU/1007/2011).**

- On development of a test standard, manufacturers of clothing would be required to include a label attached to the product indicating the relative level of fibre release during washing.
- Using the standard A-G rating, this *demand-side* measure would ensure that consumers are adequately informed about the relative rate of fibre release for clothing placed on the market.

In addition, we consider a further measure that relates to the EPR approach to covering the costs of microplastics from a range of sources within wastewater treatment facilities. Such EPR measures are discussed more broadly in Section 7.4, but EPR measures to capture microplastics from synthetic clothing using filters *within* washing machines are covered under the synthetic clothing measures.

7.3.1 Measure 1 - Development of a Standard Measurement for Fibre Release

As more textiles samples are subjected to varying tests for fibre release during washing it has become apparent that there is a need for standardisation in this regard. Researchers are beginning to ascertain which factors are most import in designing a test and therefore obvious methodological improvements can be made. It is, however, problematic to compare studies and develop European level release estimates based on current findings. The many different ways in which samples can be tested and the multiple factors which affect fibre release mean that the observed ranges are currently very large.

A standardised comparative test may require a different approach to one that is designed to capture and characterise all fibres released. For example, to simplify and speed up testing, a large filter mesh size (~100um) could be used if previous test work has shown that comparison can be accurately made between fabrics using this size filter—i.e. if using a smaller filter is likely to yield the same comparative results. For comparative tests, absolute fibre release count is less important.

Washing using a fabric sample may be a more reproducible method of creating a standard test than testing whole garments. Greater control is possible and samples can be directly compared. Small changes can be made (to the way in which edges are finished, for example) which will lead to isolation of the best practices that reduce fibre loss. These tests may be less useful for the calculation of the *absolute* fibre release (on an EU scale), but potentially more useful for comparison between samples in order to set a standard.

Costs for such a test are difficult to estimate at this point. However, a significant amount of work needs to be carried out in order to develop such a standardised test procedure. The Mermaids project[177] cost over €1 million - albeit its focus was not on developing a standard – and it is expected that a similar amount would need to be spent on developing and agreeing a standard test, but with wider textiles industry support and engagement during the process.

There is already an *ad-hoc* working group chaired by DG GROW where current progress on this issue is being shared. No formal project proposals have been shared as yet, but a tentative voluntary agreement by a cross-sectoral group of textile stakeholders is reportedly being formulated. It is understood that this agreement does not currently include any proposals or commitment to actions that would lead to a reduction in fibre release during washing. This being the case, it is therefore

[177] http://life-mermaids.eu/en/

important to investigate some of the potential options that could be adopted; whether they be voluntary or mandatory.

The costs for an individual test are not known at present, but illustrative costs of between €1,000 and €5,000 are used, as described in Section 7.3.2, to indicate a plausible scenario for the potential costs to the textiles industry.

7.3.2 Measure 2 - Setting a Maximum Threshold

After the creation of a standardised test method, it should be possible to compare fabrics placed on the market for their tendency to release fibres during washing. On this basis it would therefore also be possible to determine a fibre release range and create a threshold that removes the worst performing products from sale.

This threshold could either be adopted as a new Regulation (as it is important that this is harmonised across Europe, and would also apply to all items placed on the market, including imports) or as part of a voluntary agreement. As part of this, there would need to be certification requirements. These could either be;

- A declaration of conformity as part of a self-certification process; or
- An independent testing regime for all clothing/textile products on the market.

Clearly the latter has the potential to be costlier, however non-compliance with the former would not be identified unless some form of random spot testing were also applied within Member States. It would appear to be unnecessary to impose such spot testing procedures if a set of parameters can be identified that can establish that particular fabrics/constructions should be restricted. A declaration of conformance to these restrictions may be the requirement in a similar way to the administration of the RoHS Directive[178] which primarily affected the use of lead-based solder in electronics—another complex global supply chain.

In this case, certain fibre types or constructions could be restricted and the certification would merely be required to confirm compliance with these restrictions. Exceptions could be made—in the case of new innovations— if the standard test is applied and it is shown that fibre release is below the threshold.

It is likely that a new Regulation would be required to enforce the maximum threshold. A voluntary agreement is in the very early stages of being discussed, but with no current focus on reduction methods. If voluntary reductions are agreed, it is unclear how effective they would be due to the fragmented nature of the industry with many players overseas in Asia. A voluntary agreement may also lead to a financial advantage for those that do not commit on a voluntary basis, meaning there would not be a level playing field.

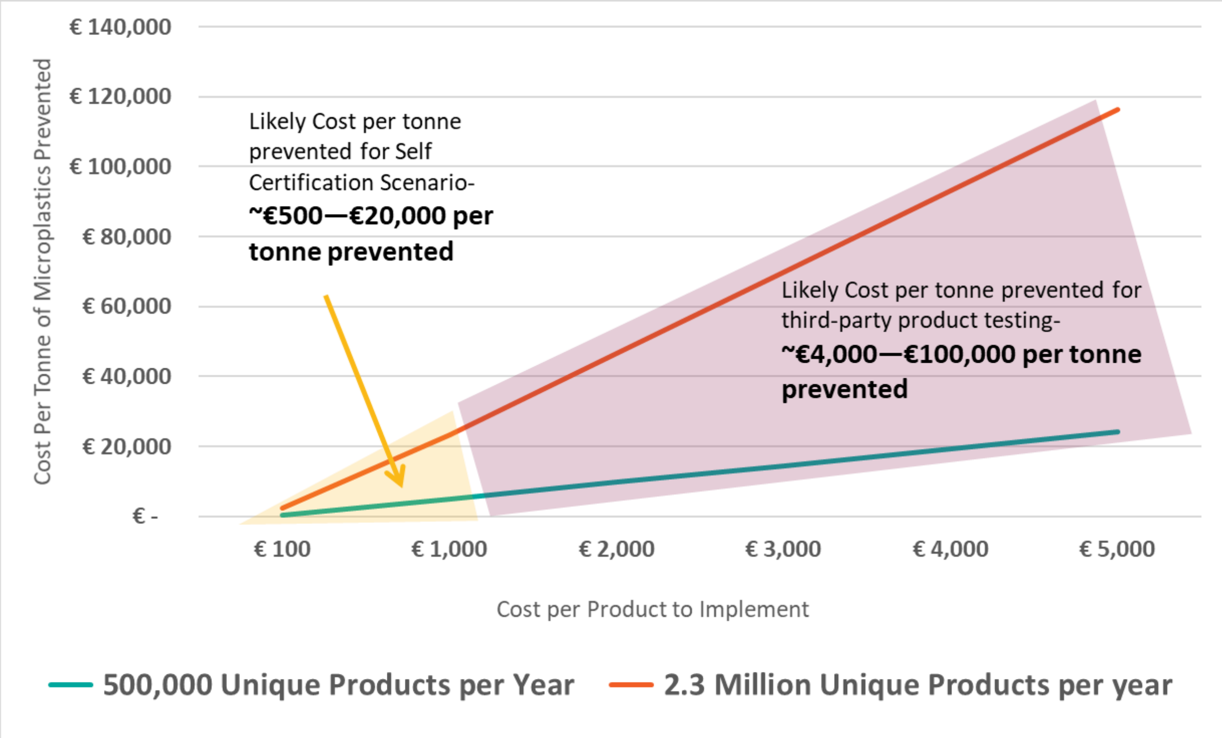
The costs to industry for the introduction of the threshold are difficult to assess due to lack of industry data. A possible cost range can be identified to provide an illustration of the potential costs. Clothing item sales data can be used to estimate costs based on the estimated number of 'unique products' on the market—these would be products that require their own certification and this number will vary depending upon the definition of unique product. For example, whether different colour variations are classed as the same product.

[178] http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2011:174:0088:0110:EN:PDF

Annual costs are therefore estimated to be from around €0.5 billion (around 500,000 unique products costing €1,000 each to test), up to a maximum of over €11 billion (around 2.3 million unique products costing €5,000 each to test). The mean overall cost is around €3 billion and with 3.2 million tonnes of man-made clothing placed on the EU market every year this would equate to an ***additional cost of €0.90 per kg.***

Figure 15 shows that the cost effectiveness per tonne of microplastics prevented could differ between the two certification and testing regimes. The self-certification approach is clearly preferable in this instance (with costs for each product certification assumed to be around 10% of a third-party test). Whether the certification process can function this way in practice will be determined by the development of the test method and whether it lends itself to the self-certification process.

Figure 15 – Costs per tonne of Microplastics Prevented by Certification Type



7.3.3 Measure 3 - Development of Product Labelling

There are two ways of including a label which can be used to achieve different outcomes;

- **A Sewn in label**—containing washing and user guidance which can be referred to on an ongoing basis; and
- **A Removable Label**—containing information that is designed to provide environmental information and influence buying decisions.

There may be scope to include one or both of these labels.

As information on the sewn-in label is already mandatory it would be straightforward to amend the current labelling Regulation (EU 1007/2011[179]) with extra requirements. The effectiveness of this to

---

[179] http://eur-lex.europa.eu/legal-content/en/TXT/?uri=CELEX%3A32011R1007

influence consuming washing practices, relative to the increased burden on manufacturers (i.e. the potential for the label to increase in size) is not known.

The inclusion of a specific additional label may be designed in line with the well-recognised and understood labelling systems for energy using products (Energy Labelling Directive 2010/30/EU[180]) and tyres. This may be in the form of a removable card tag attached prominently on the outside of the garment similar to the tags that are often provided to provide information for more technical garments of branding on mid-high range clothing. This would incorporate an A-G rating based upon the expected level of fibre release. It may be argued, however, that a label specifically for fibre release is disproportionate when compared with other environmental impacts associated with clothing and textile manufacture. In the case of energy labelling for products such as washing machines and refrigerators, the energy use of the product is highlighted to cause the most environmental damage over the life of the product—hence why the energy label was created. There is currently no evidence to suggest that the biggest environmental impact for textile products is fibre release—or indeed, even a method in existence for comparing the impacts of fibre release with those of climate change or resource depletion for example.

Similar to introducing the maximum threshold, there is also the issue of whether the garment is individually tested or is placed on the A—G scale based upon the fabric type and construction method. In either case the burden of introducing such a label is significantly decreased if the information is already available through the maximum threshold introduction. For this reason, it is recommended that the threshold be introduced first.

7.3.4 Measure 4 - Extended Producer Responsibility

Extended Producer Responsibility (EPR) is a policy approach under which producers are given a significant responsibility – financial and/or physical – for the treatment or disposal of post-consumer products. In this context there would be a requirement for the textiles industry to pay for the implementation of initiatives that capture fibres before they are released into the aquatic environment. This can be achieved at the two key points in emission pathways that fibres from clothing are known to pass through; the washing machine and the wastewater treatment (WWT) plant.

Funding of improved WWT is discussed in more detail in Section 7.4 which addresses measures specific to the wastewater industry. In this section, the effectiveness of capture at the washing machine is discussed.

There are a number of potential capture systems that have been proposed or developed recently that are designed to work with a washing machine. They broadly come under two categories;

- An in-built washing machine effluent filter; and
- A device placed in the washing machine drum which is independent of the washing machine itself.

It is difficult to determine the effectiveness of devices that are designed to be placed into a washing machine drum by the user. To assess this in terms of the best case for reduction potential, it is assumed that one would be supplied with every purchase of a new washing machine. Similarly, this is compared against the introduction of a compulsory filter which is built into the effluent outlet of

---

[180] http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32010L0030

all new washing machines. Early indications from the designers of such products suggest a fibre capture rate of between 26 and 80 per cent.

The overall effectiveness is also determined by whether the product is actually used (correctly) during washing. For an in-built filter this is assumed to be 100% of the time. No data is available for the in-drum devices, but for the basis of the calculations it is assumed that these will be used between 40 and 80 per cent of the time depending upon how much effort is needed to use the device. Specific products currently being trialled to capture loose fibres include: a filter, by start-up 'Planetcare', that is integrated into the washing machine; The Cora Ball[181], designed to be placed into the drum with the clothing and is free to move around; and the Guppy Friend[182] that requires the user to place synthetic clothing within a bag.

The cost per tonne captured for the three examples of these devices are between €40—125 thousand with an overall effectiveness of 21 to 80 percent for washing machines that use the devices.

7.3.5 Impacts of Synthetic Clothing Measures

Table 20 shows the projected reduction in microplastics emissions from the introduction of a filtration technology in the washing machine and introducing a maximum threshold for fibre release. These are shown graphically over time in with the assumption that any Regulation would be introduced from 2020.

Measure 1, the development of a test standard is also not assessed for its impact on its own as it will not result in reductions in itself, but is a necessary prerequisite for the subsequent measures.

For Measure 2, the maximum threshold demonstrates two levels at 10% and 20%. The costs vary considerably, but suggest that if a self-certification process is viable, this would likely be the most cost-effective way of reducing fibre releases. Critically, however, it is unlikely to reduce the releases to zero.

For Measure 3 there is a lack of data to determine the costs of the introduction of a labelling system. As it would likely be a physical label there would be additional product costs, but depending upon the nature of the label these may be relatively low, but would affect low cost products more. In this scenario the label would provide information to the consumer on fibre release rates therefore all of the costs associate with testing and or certification in Measure 2 would be applicable. It may be broadly comparable to the impact expected from the tyre label—around a 10% reduction in emissions—but the dynamics of the interaction with the consumer is complex. On one hand the clothing label may have a larger influence as it would be more visible to every consumer compared with the tyre label that is often not seen. Conversely, unless the consumer finds a link between fibre release and garment quality, they may not find themselves incentivised to choose the better performing product.

Measure 4 suggests that an integrated washing machine filter is likely to be the most effective at capturing fibres before they can be released into sewers. The costs are higher, but are highly speculative at the present time. If product costs were to increase significantly above these, it would be a less viable prospect.

[181] http://coraball.com/
[182] http://guppyfriend.com/en/

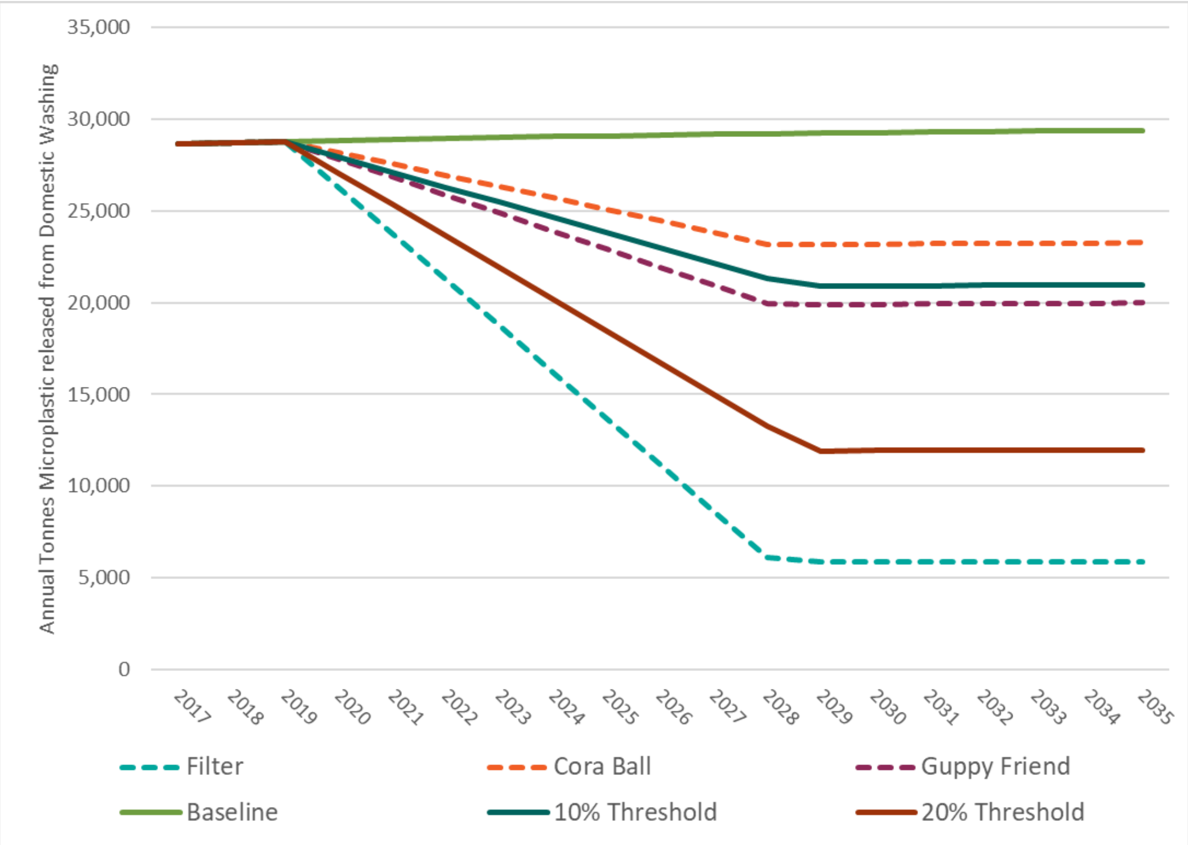
Table 20 - Costs and Reduction Potential of Synthetic Clothing Measures

Measure		Cumulative Emissions 2017-2035 (tonnes)		Cumulative Reduction from Baseline 2017-2035 (tonnes)		Annual Cost per Tonne Prevented *at Source*
		Source Emissions	Surface Water Emissions	Source Emissions Reduction	Surface Water Emission Reduction	
Baseline		***600,000***	***250,000***	-	-	-
Measure 2 - Maximum Threshold	10%	500,000	210,000	100,000	40,000 (16%)	Self cert €500—20k Third Party €4k—100k
	20%	400,000	160,000	200,000	90,000 (36%)	
Measure 3 - Labelling		500,000	210,000	100,000	40,000 (16%)	
Measure 4 - Washing Machine Filter	Filter	300,000	130,000	300,000	120,000 (48%)	€50k—125k
	Cora Ball	500,000	220,000	100,000	30,000 (12%)	€41k—104k
	Guppy Friend	500,000	200,000	100,000	50,000 (20%)	€44k—112k

Note:

1. **Emissions figures rounded to nearest 100,000 or 10,000 for those less than 100,000.**

Figure 16 - Reduction Potential of Synthetic Clothing Measures 2, 3 and 4 (10% threshold also indicate reduction for Measure 3 – Labelling)



7.4 Wastewater Treatment

The measures taken forwards for detailed analysis are as follows:

- **Development of a test standard for the quantification (both in mass and number) of the microplastics in the influent, effluent and sludge output of wastewater treatment plants.**
  - Such a test would:
    - take account of possible contamination from microfibres in the atmosphere;
    - use filters as small as practicably possible to make sure the smallest microplastics are measured;
    - sample a high enough proportion of the influent/effluent so as to be confident that the sampling is representative; and
    - be conducted over a time period that would allow accurate estimation of annual microplastic loads.
  - Where possible the test will also be used to identify where the microplastics have come from. This may be impossible for some types, but microplastics from paint, tyres and clothing fibres are likely to be unique enough to be identifiable.
  - Such a test will of itself not lead to any additional capture of microplastics through WWT, but it will be the basis for subsequent measures detailed below or in the context of a future review of the UWWT Directive
- **Development of an EPR Scheme such that the sources responsible for microplastics in WWT cover the respective costs of remedial action**
  - This may be administered and be applied differently from country to country and will rely on sampling at a local level. This will lead to different industries being required to contribute differently within each member state.
  - Similarly, the fee would go towards (and the level of the fee would be determined by) the most appropriate means of mitigation, whether that be, for example:
    - Adding additional (existing or novel) treatments to WWTPs
    - Improving roadside capture; or
    - Increasing road cleaning activity

Two distinct EPR measures are identified, with one targeted at capture of microplastics via waste water treatment, and the other targeted at capture of microplastics via storm water treatment.

7.4.1 Measure 1 - Development of a test standard for the quantification of the microplastics in Wastewater Treatment

The development of a standard test method that can be used to characterise the microplastics entering and leaving a WWT plant is critical to the application of any further measures for microplastic reduction in WWT. Although several studies from across Europe as well as the US and Australia have attempted the measure the occurrence of microplastics in WWT there are often key methodological differences that not only make comparison of result difficult, but also highly speculative when used to scale up emissions beyond the plant in question.

The differences in wastewater effluent data occurs mainly due the differences in sampling volumes, mesh-sizes of the filters and material characterization (visual identification vs. Fourier-transform infrared spectroscopy (FT-IR)). These are the most crucial methodological steps that affect the results. Potentially the best methods currently devised for wastewater effluent microplastics measurement use large sample volumes (>/=1 m^3), a small mesh-size for filtration (<10 µm) and automated material analyses.

Data from the wastewater influent and sludge is very preliminary and reliable, comparable data is likely to take considerably longer to obtain. The extraction of microplastics from influent and sludge is very difficult (due to filter clogging), leading to very small sample sizes.

Textile fibres are relatively easy to identify from wastewater samples due to their shape and colours. FT-IR analyses can identify synthetic fibres relatively well, but natural textile fibre materials are more challenging (although not a natural fibre this is also an issue for 'man-made' fibres such as viscose). The provenience of other microplastics is more difficult as their characteristics (size, shape, material type, colour, etc.) may be identified, but it can often be far more challenging to positively trace these back to an emission source.

The identification of specific materials may be more or less important depending upon what sort of measure the results are supporting. There are two main ways that a test standard can be used to support legislative measures;

- To identify whether upstream measures are effective at reducing microplastics; and
- To identify which sources of microplastics are contributing the most to microplastics loads through WWT plants.

The former may not require 100% positive identification as there is no 'penalty' attached to it. The test would be undertaken merely to identify whether policy measures have the desired effect. The latter would require a high degree of certainty around the identification of the source of the microplastics as this would directly link to any EPR payments (the EPR scheme itself is discussed further in the following sections). It is unclear at present, whether this is technically possible or cost effective. The costs of either approaches are also not known at present.

The implication would be that each WWT plant would need to sample their influent, effluent and sludge on a regular basis (unless it can be ascertained that a few plants are representative of the rest of the system). The scope could follow the UWWT Directive by limiting to WWT plants servicing towns and cities over 2,000 inhabitants. According to the European Environment Agency, there were around 19,000 such plants in 2010[183]. If each one of these plants spend €10,000 per year on testing, the annual cost of this for the EU would be €190 million. This is highly speculative without further information on the costs of test regimes which can only be accurately assessed once the testing is scoped and developed. The costs of such testing would have to be incorporated into any of the subsequent reduction measures.

7.4.2 Measure 2 - Improved Wastewater Treatment EPR

Under the assumption that it is possible to achieve measure 1 with a robust method for quantification and identification of microplastics in WWT it then becomes possible to assign the costs of improved capture to the associated polluting industries.

The costs were estimated by Eureau to be between €0.08-0.20 per cubic metre of wastewater treated per year using one of the current technologies that is suspected to enable close to 100% capture of microplastics in WWT plants. However, this approach requires subsequent disposal of the collected microplastics, which end up trapped in sludge. This is often applied to agricultural land and therefore can still enter the environment and potentially over the longer term the marine

[183] https://www.eea.europa.eu/themes/water/water-pollution/uwwtd/waste-water-infrastructure/urban-waste-water-treatment-plants

environment. This also does not take into account possible technological advancements in the capture of microplastics in WWT plants. Several initiatives are underway in the EU to develop technologies to specifically tackle this issue, but none are commercially proven at this stage. Wasser 3.0, an initiative from University of Koblenz-Landau in Germany has developed one such promising technology which has the potential to isolate and remove microplastics before sludge. The cost and practical implementation on a large scale is yet to be established (more information in Appendix A.6.4.4).

However, there are currently existing plants that have tertiary treatment[184] and therefore the costs should be applied to improving those that do not. There are no comprehensive official datasets for the volume of wastewater treated by country, but the overall estimates from Eureau can be split by population equivalents (PE) for each country. The Eurostat data used to model the microplastics retention rates in Section 2.2.8.1 is used to ascertain the capacity gap for those that do not currently have tertiary treatment. This suggests that there are between 10 and 16 million cubic meters of wastewater that could benefit from an upgrade to increase retention. This would cost €0.76—3.14 billion per year assuming the unit costs from Eureau. However, the baseline calculations in Section 4.0 show that the compliance with the Urban Waste Water Treatment Directive will increase tertiary treatment by around 12% by 2035. This means that additional spending is lessened and therefore costs are reduced to €0.6—2.4 billion. With an average cost of €1.49 billion per year the basis of an EPR scheme can be developed.

Costs to improve WWT can be applied to the different product groups based on their proportional contribution. In practice this would rely upon accurate identification and characterisation of the particles entering into WWT as described in the previous measure. Presently this example uses the estimated tonnages that are modelled to enter WWT. Annual costs range from €10 million for building paint to just over €1 billion for textiles. Full results can be found in Appendix A.6.4.2.

One potential method of administering an EPR scheme would be via a fee applied to each product based on the level of contribution to microplastics entering WWT. It would be expected that the fee would vary between Member States as different industries are likely to contribute differently between countries. It is therefore important to have representative microplastic data from WWT plants in each of the countries.

For most of the products a fee per kg placed on the market appears to be the most appropriate. This is the case for textiles, pellets and paints. The weight of a tyre itself has no direct link to wear rate, albeit the overall weight of the vehicle does, and therefore it is more appropriate (and administratively simpler) to apply the fee per unit (tyre) sold. Tyres have been split into two groups as the average wear rate per km per tyre is considerably higher for truck tyres—by volume truck tyres are around 5% of the market, but they contribute to 26% of tyre wear emissions. They also cost more, therefore a higher fee can be applied while resulting in a similar proportional increase in the product price as for a car tyre.

In most cases the fee is less than 1% of the product cost for most sources and often as low as 0.1%. The cost per tyre is increased by as much as 7% for the lowest value tyres on the market. Full results can be found in Appendix A.6.4.2.

---

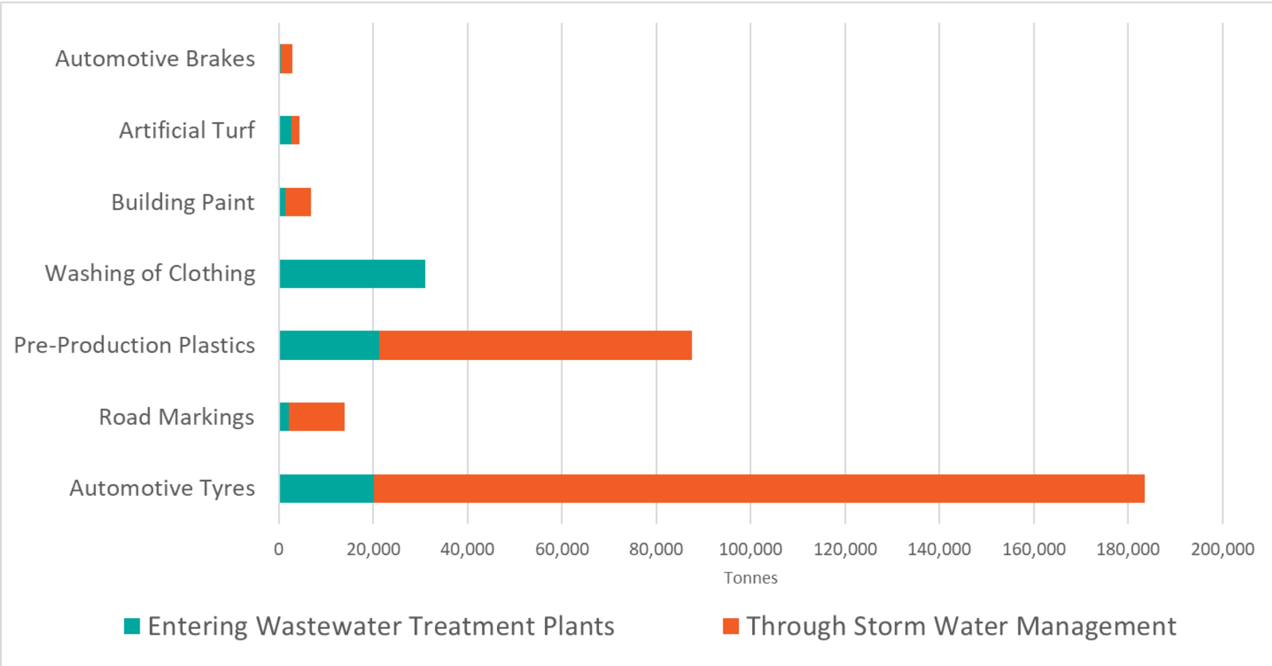
[184] It is recognised that the term 'tertiary' is very broad and is not well defined, however it is assumed that tertiary treatment in whichever form it takes will increase microplastics capture rates.

Artificial turf is different to the others as the infill remains in service for around 10 years and does not inherently become an environmental issue as part of its product life, i.e. unlike tyres which are designed to wear during their life. The application of a fee on the cost of the infill would add around €3,000 (~1% of the total installation costs) to the installation cost of a typical football pitch. This money is likely to be better spent introducing on-site mitigation and containment measures which would reduce the burden on downstream captures mechanisms such as WWT.

7.4.3 Measure 3- Improved Storm Water Treatment EPR

Around 10% of total microplastic source emissions are estimated to end up in waterways through storm water. For Automotive tyres 52% of the emissions to waterways (10% of total tyre emissions) are via storm water run-off. Around 250 thousand tonnes of microplastics are estimated to go through storm water—this is greater than the amount estimated to go through WWT plants (see Figure 17, Appendix A.3.8.7 for data table). However, in this case there are a huge number of potential points in which these can enter waterways. This is compared with the relatively few WWT plants.

Figure 17 – Microplastics Entering Water Management Systems (midpoint)



Recent studies suggest that wetlands can provide an effective method of capturing microplastics in storm water effluent. The cost of this could be around €0.06 per cubic meter—less than that of upgrading WWT—however, unlike WWT, it is much more difficult to estimate how much storm water would need to be treated to remove a certain amount of microplastics. There is almost no sampling data to help with this calculation and it is also likely to be very site specific.

For this reason, it will be very unlikely that full coverage could be achieved via this method. It may also be impractical to link all storm water sources to wetlands especially in urban areas adjacent to rivers. However, improvements to storm water capture may be more cost effective in urban areas where the concentration of emissions is highest—with 40-50% of the total run-off emissions. Alternative roadside capture methods should therefore be investigated for these areas.

Hotspots for emissions would have to be identified. This could initially be carried out by simply looking for the roads which have the most traffic over the course of a year. The run-off from these should be sampled to ascertain the concentration levels that are present. A key question that would also need to be answered is what level of concentration is deemed high enough to install mitigation measures. This decision is likely to be made at a Member State level based on local sampling.

Given the significant uncertainty in the nature of the most appropriate mitigation measures, the degree to which different mitigation measures would need to be adopted and the scale at which these may be rolled out, it is not feasible to establish an estimate of the likely cost-effectiveness of Measure 3.

7.4.4 Impacts of Wastewater Treatment Measures

Impacts have been calculated only for Measure 2 as there is not enough data to provide impacts and costs for the other two measures at present. Measure 1 is also simply a prerequisite to Measure 2 or a way of verifying the effectiveness of other measures. It therefore has no impact in itself. Table 21 shows the reduction potential relative to the baseline which demonstrates that improving the WWT infrastructure could reduce microplastics release to surface waters by circa 50% at a cost of €45,000—137,000 per tonne. Figure 18 demonstrates this graphically; the baseline shows a gradual reduction as current infrastructure improvements come online, and a steady improvement from 2020 is shown for the increases associated with implementing the high capture levels specified in this measure.

Figure 19 shows this impact disaggregated by emission source; demonstrating that the three main sources of clothing, tyres and pellets will see the largest reductions.

Table 21 - Costs and Reduction Potential of WWT Improvement Measures

Measure	Cumulative Emissions 2017-2035 (tonnes)	Cumulative Reduction from Baseline 2017-2035 (tonnes) Emissions to Surface Water	Cost per Tonne Captured at WWT
Baseline	***600,000***	-	-
Measure 2 - Improved WWT	400,000	200,000 (33%)	€45k—137k
Note: Emissions figures rounded to nearest 100,000 or 10,000 for those less than 100,000.			

Table 22 shows the cost per unit that may be applicable if an EPR scheme is implemented to fund proposed improvements to WWT infrastructure. This assumes the annualised cost of €1.5 billion is shared amongst the products relative to their current estimated contribution towards microplastics loading in WWT. In most cases the product increase would be less than 1% of the sales cost although this would vary greatly depending upon the price point of the product—particularly for textiles. This does demonstrate that the increase in product cost is not unrealistic, but as microplastics is only one element of the end-of-life of each one of these products it may not be proportional to the impact (if compared with other end-of-life environmental impacts). This is yet to be established.

Figure 18 - Reduction Potential of WWT Improvement Measures

Table 22 - Costs per Unit for Wastewater Treatment EPR

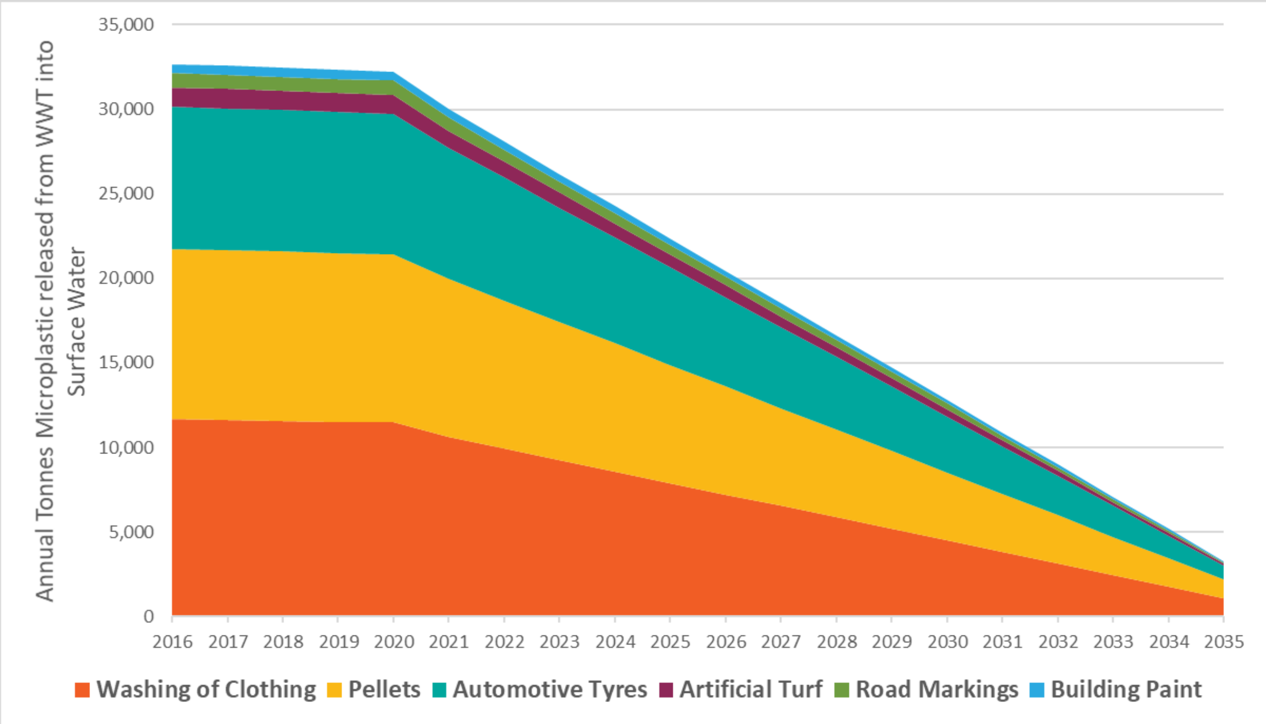
Source		Annual Sales	Unit	Costs per unit
Washing of Synthetic Clothing		7,571,600,000	kg	~€0.10
Pre-Production Plastics		70,565,000,000	kg	€0.01 - €0.001
Automotive Tyres [1]	Car	0.28	billion units	~€1.30
	Lorry	0.01	billion units	~€10
Artificial Turf		110,195,280	sqm	~€3,000 per full sized pitch
Road Markings		30,000,000	kg	~€1.30
Building Paint		1,137,650,400	kg	€0.01 – 0.02

Notes:

1. The costs for tyres are split between lorry and passenger car tyres. Although lorry tyre sales account for 5% by number they account for 26% by contribution to tyre wear particles. This is why the cost for these tyres is great (but likely to be similar as a proportion of tyre costs).

2. See individual emissions sections of the report for annual sales data sources.

Figure 19 - Reduction Potential of WWT Improvement Measures – Emission Sources



8.0 Comparison of Options

In this section we look at the potential to combine measures into options for each emission source (where possible). The impacts of the different options on different stakeholders are considered, along with a comparison of the overall likely effectiveness in reducing microplastics emissions, and estimates of cost-effectiveness.

It's worth noting upfront that different criteria are used to compare measures and options that relate to different sources. For example, one set of criteria are used to compare the different tyre options, but the pellet options are compared using a different set of criteria. This is because they are very different markets and thus different priorities or barriers exist. This is one of the issues of assessing many differing products under the banner of 'microplastics'.

Comparison is made, in Section 8.5, of the potential scale of reduction that might be expected through the different measures, and the anticipated cost per tonne of microplastic loss prevented.

8.1 Automotive Tyre Wear

For automotive tyre wear, rather than a series of competing measures, the identified measures have the potential to work together. The attributes of the selected measures are shown in Table 23.

Table 23: Comparison of Measures

Measure		Addressing a Knowledge Gap	Effect on Competition	Potential for Market Transformation	Coherence with Other Measures
1	**The Development of a Standard Measure of Tyre Tread Abrasion Rate**	This will address the key gap in knowledge that currently prevents the development of Measures 1 & 2	N/A in isolation, but will be the basis for subsequent measures that will have positive effects	N/A in isolation but will permit development of subsequent measures that will lead to market transformation (albeit it may stimulate R&D investment prior to the development of subsequent measures	This is a fundamental step in permitting the development of subsequent measures
2	**Include Tyre Tread Abrasion Rates on the Tyre Label**	Positive – This will address a key market failure - the knowledge gap currently faced by consumers when purchasing tyres	Positive – will improve competition and allow new entrants to demonstrate their attributes	Medium-High, given that abrasion relates to tyre 'mileage' which is of importance to consumers. If the other tyre label tests are also reconfigured to be more representative of performance over the tyre's lifetime the rate of transformation will be higher	This is fully coherent with other measures. It provides a demand-side pull, using consumer information and preferences to bring about a shift towards tyres that abrade at lower rates

3	**Using the Type Approval Regulation to restrict the worst performing tyres (in respect of tyre tread abrasion) from the market**	N/A – will simply involve removal of the worst performing tyres (in respect of abrasion rates) from the market	It may reduce consumer choice if implemented without adequate warning. Therefore adequate warning should be given	Medium – High, depending on the level at which the restriction is imposed	This is fully coherent with other measures. It provides a supply-side restriction to complement the demand-side pull of the Tyre Label

It is also clear that certain steps need to be taken first, and subsequent steps then should be considered in light of the outcomes of earlier steps. Accordingly Option 1 and Option 2, as presented in Table 24 are not mutually exclusive options. Instead, Option 2 could be introduced to complement Option 1.

Conclusion

There is strong consumer demand for information about tyre wear, and thus tyre abrasion rates. In the absence of labelling, the identified market failures relating to end-users, tyre producers, and society as a whole will continue. Furthermore, the relative contribution to airborne PM from tyres is likely to increase in the coming years as the move towards electrification of the vehicle fleet continues.

It is therefore recommended that:

In Support of Option 1 and 2-

1) A standard measure of tyre tread abrasion be developed as a priority – possibly through the use of a 20,000km tyre abrasion test for each tyre model (and variants thereof as deemed appropriate) that is placed on the market.
2) The merits of revising the test methods for wet grip, rolling resistance and external noise by incorporating them within a 20,000 km tyre abrasion test, should also be further explored alongside the development of a standard measure of tyre tread abrasion.
3) At the same time further scientific investigation should take place, ideally publicly funded, into the extent and characteristics of particulate matter released to the air from abrasion of vehicle tyres. Importantly, a better understanding is necessary of whether a shift towards tyres that exhibit a lower rate of abrasion (as demonstrated through the standard measure of tyre tread abrasion) will be accompanied by:
    a. An overall reduction in emissions across all particle sizes (i.e. both $PM_{2.5}$ and PM_{10}; or
    b. An overall reduction in emissions of all PM_{10} by mass, but an absolute increase in the amount emitted as the more damaging fine particles (i.e. $PM_{2.5}$)
4) As long as the air quality impacts from a move towards tyres that abrade at a lower rate will not be negative, and on the basis that the standard measure of tyre tread abrasion allows sufficient confidence to accurately determine appropriate bands, the inclusion of tyre abrasion rates within the tyre label should be progressed; and

In Support of Option 2-

5) The possibility of using the Type Approval Regulation to restrict the worst performing tyres in respect of tyre tread abrasion from the market should be further considered in light of the

results from the standard measure of tyre tread abrasion. This may indicate immediate opportunities to achieve a significant reduction in tyre abrasion by targeted removal from the market of the worst performing tyres. Conversely, there may be merit in first observing the effectiveness of the inclusion of tyre abrasion rates on the tyre label over several years, in order to consider whether this alone has brought about sufficient market transformation and then considering whether further progress could sensibly be achieved in a proportionate manner through the Type Approval Regulation.

Table 24 – Summary of Tyre Options

Option	Measures	Description
Option 1 – Tyre Label	1 & 2	**Development of a Standard Measure of Tyre Tread Abrasion Rate followed by Inclusion of Tyre Tread Abrasion Rates on the Tyre Label** The ability to implement this Option will depend upon whether a standard measure of tyre tread abrasion rate can be developed, and if so, whether the measurement is sufficiently accurate to determine with confidence banding for tyre tread abrasion within the Tyre Label. Whether or not implementation of the Option, or at least the inclusion of the Tyre Tread Abrasion rate within the Tyre Label, is desirable from a societal perspective, depends on the nature of any changes in emissions to air of particulate matter from tyres as a result of efforts to reduce the rate at which tyres abrade If implemented, the effectiveness of the measure will likely be increased if the testing procedure for wet grip, rolling resistance and external noise is amended, through incorporation within the testing procedure for tyre abrasion
Option 2 – Tyre Label plus Type Approval Regulation	1, 2 & 3	**Development of a Standard Measure of Tyre Tread Abrasion Rate followed by Inclusion of Tyre Tread Abrasion Rates on the Tyre Label, plus using the Type Approval Regulation (EC/661/2009) to restrict the worst performing tyres (in respect of tyre tread abrasion) from the market** In addition to the points above in relation to Option 1, a decision on whether or not, and if so how, to restrict certain tyres from the market under the Type Approval Regulation could only be made following the introduction of a standardised test, meaning that detailed information would be available as to performance in respect of abrasion rates for all tyres on the EU market. The merit of introducing such a restriction might vary depending on the extent to which Option 1 had already delivered, and were anticipated to continue to deliver, market transformation in respect of tyre abrasion.

8.2 Pre-Production Plastics

In this section we consider the impacts of the different options on different stakeholders, along with the overall likely effectiveness, and cost-effectiveness of the options. In so doing we have considered each of the horizontal measures (amending the Polymer Production BREF and Regulations on the Transport of Pellets, and on Converters) in isolation, as well as in combination.

A comparison of the performance of each of the four options in respect of a number of criteria is shown in Table 25. The relative performance of the horizontal measures (1 to 3) is first considered against each criteria and then a comparison drawn with the anticipated performance of the regulation requiring supply chain accreditation (4):

Vertical integration: The first criteria is the extent of vertical integration of any best practice measures and procedures within the supply chain. This was noted as a key element in effective delivery of pellet-loss prevention by stakeholders. For the three horizontal measures, both in isolation and in combination, the extent of vertical integration is low, leading to a real risk of inconsistent methods being adopted. In addition, this could potentially hamper any clean-up efforts where spills occur at the point of collection (from producers) or delivery (to converters) of pellets, with a possible focus on attribution of blame rather than on timely and comprehensive clean-up.

Scope: The second is scope, which relates strongly to the likely effects on competition, and also to the scale of impact (the latter also depending on the extent of vertical integration and thus co-operation up and down the supply chain). For the horizontal measures the scope is limited to producers and converters within the EU.

Competition**:** The third is competition, which is closely related to scope. The three horizontal measures, focusing predominantly on EU-based production, transport and conversion would mean EU producers, transporters and converters facing costs that would not be incurred by supply chains outside the EU. This would provide an advantage to supply chains based outside of the EU that, without countervailing measures, would still be able to place plastic items manufactured outside the EU on the EU market.

Scale of impact: The fourth is scale of impact in reducing pellet loss, which relates both to the extent of vertical integration (and thus the effectiveness of best practice measures and procedures to prevent pellet loss) and the scope. The horizontal measures would lead to either limited (in the case of amending the Polymer Production BREF), or moderate positive impacts in reducing pellet loss, given that the focus is primarily within the EU.

In contrast to the three horizontal measures (measures 1 to 3), against all of the above criteria, the Regulation Requiring Supply Chain Accreditation performs more strongly given that:

- It is designed to focus on and reinforce the vertical integration of best practices within the supply chain, ensuring the effectiveness of best practice measures and procedures to prevent pellet loss;
- Being focused on plastic items placed on the EU market, its scope would cover supply chains within and beyond the EU;
- Due to the incorporation of extra-EU supply chains (as the focus is on plastic goods placed on the EU market, regardless of where they are made), there is no competitive disadvantage for EU producers, transporters and converters relative to those outside the EU who are selling to the EU market; and
- The scale of impact would be high due to both the focus on vertical integration of best practice measures, and the reach of the supply chain regulation beyond the EU. Indeed, if loss rates (per tonne) from production, transport and conversion of pellets from facilities outside the EU were greater than those inside the EU, the unit impact of this approach would be even greater.

Table 25: Comparison of Measures

Measure		Vertical Integration	Scope	Effects on Competition	Scale of Impact
1	**Amending the Polymer Production BREF**	Low - leading to the risk of inconsistent methods being adopted	Limited - focused solely on EU-based producers	Negative - EU producers would bear costs not faced by producers outside the EU	Limited - would only apply to EU-based producers, the majority of whom are understood to already be taking action, or planning to do so shortly
2	**Regulation on the Transport of Pellets**	Low - leading to the risk of inconsistent methods being adopted	Moderate - focused primarily on transport within the EU, but would include imports and exports of pellets	Negative - transporters within the EU would bear costs not faced by transporters outside the EU	Moderate - while focused primarily on transport within the EU, it would also include imports and exports of pellets
3	**Regulation on Plastic Converters**	Low - leading to the risk of inconsistent methods being adopted	Limited - focused solely on EU-based converters	Negative - EU converters would bear costs not faced by converters outside the EU	Moderate - it is understood that converters have taken less action to date to address pellet loss than producers
4	**Regulation Requiring Supply Chain Accreditation**	High - designed to reinforce vertical integration of best practices within the supply chain	Extensive - would apply to plastic products placed on the EU market so would therefore cover extra-EU supply chains	Neutral - as the measure would apply to all plastics placed on the EU market it would apply equally to supply chains outside the EU	High - the positive impact in reducing pellet loss is expected to be far greater under this option as it will also apply to supply chains outside of the EU

In addition, there are three other key strengths associated with the Regulation Requiring Supply Chain Accreditation. These are:

1) Cost-effectiveness;
2) The extent to which the polluter pays; and
3) The potential to show strong European leadership on preventing this source of marine plastic pollution, and the possibility of such a supply-chain oriented approach being adopted in other jurisdictions.

Cost-effectiveness depends both on effectiveness and costs. Due to the focus on vertical integration of best practice measures and procedures within the supply chain – noted by stakeholders as a key element in effective delivery of pellet-loss prevention – the regulation requiring supply chain accreditation is expected to be the most effective of all options in preventing pellet loss. The costs should also be lower than for other options, and certainly much lower than the closest alternative in terms of scope, which would be the adoption of all three horizontal measures. For the combined adoption of all three horizontal measures, it would be necessary to amend a BREF and introduce two regulations, compared to a single regulation under the regulation requiring supply chain accreditation. Furthermore, it was strongly felt by industry stakeholders that it would be much cheaper to address the issue of pellet loss via the regulation requiring supply chain accreditation, with private firms competing to offer advice, auditing and certification rather than an approach (which would be required for the horizontal measures) whereby regulators directly visited facilities. The regulation requiring supply chain accreditation was also perceived to be a much more 'business friendly'.

Polluter pays: The regulation requiring supply chain accreditation would place a greater proportion of the overall costs on industry (the polluter) than would be the case with the three horizontal measures. The latter would all involve direct regulation by Member State environmental regulators. While in principle, regulator costs could be recovered under the horizontal measures, the 'lighter tough' role of regulators under the regulation requiring supply chain accreditation means that as well as costs being expected to be lower overall, a greater proportion would be borne by 'polluters'.

Effects outside Europe: the regulation requiring supply chain accreditation will lead to improvements being made at production and conversion facilities, and by transporters, in countries outside the EU, including those in countries that have been identified as major direct sources of marine plastic pollution.[185] Accordingly, the regulation requiring supply chain accreditation will deliver tangible benefits in such locations, and could provide a template that could be adopted in other jurisdictions.

Table 26 - Summary of Pre-Production Plastics Options

Option	Measures	Description
Option 1 – Horizontal Measures	1, 2 & 3	**Amending the Polymer Production BREF to include best practice pellet loss prevention measures as BAT, *plus* a Regulation on the Transport of Pellets, *plus* a Regulation on Plastic Converters** Implementing these horizontal measures as an Option would only address pellet losses within the EU. This would mean that there would be a lack of a level playing field as supply chains outside the EU would not be covered, while products from such supply chains would still be allowed to be placed on the market. The lack of vertical integration in pellet loss prevention would also lead to a clear risk of spillages occurring at 'handover' points due to a lack of harmony in handling and clean up practices between the different stages in the supply chain.
Option 2 – Regulation Requiring Supply Chain Accreditation	4	**Regulation Requiring Supply Chain Accreditation** This Option would ultimately cover all plastics placed on the EU market, meaning a level playing field with supply chains outside of the EU, as well as extra-territorial benefits from preventing pellet loss beyond the EU. With a focus on supply chains, rather than on horizontal 'stages', this Option would reinforce vertical integration of best practices within the supply chain.

[185] Jambeck, J.R., Geyer, R., Wilcox, C., et al. (2015) Plastic waste inputs from land into the ocean, *Science*, Vol.347, No.6223, pp.768–771

8.3 Synthetic Clothing

Table 27 shows a comparison of the selected measures based on: their effectiveness at reducing microplastics emission, their cost, their implementation processes and their impacts on the affected industries (textiles, clothing, appliance manufacturers etc.). There is no clear preferred measure at this stage as there are too many evidence gaps to make assessments. The development of a standard measure is the only one which can be readily implemented given the current state of knowledge. Subsequent measures will rely on the results of the investigations that will form part of measurement development process.

Table 27 – Selected Measures Comparison for Synthetic Clothing

Measures		Reduction Effectiveness	Cost	Implementation	Impacts on Industry(s)
1	Development of Standard Measurement	Ineffective in itself (although will aid in further understanding and awareness), but is necessary for subsequent measures	Low - Research and development costs shared by industry	This requires cross-industry support and cooperation to develop. It is important that efforts are coordinated	Low – the textiles industry is already looking into this.
2	Setting Maximum Threshold	Medium to High – the effectiveness is dependent upon the threshold level which can only be set when a standard measure is available	Low to High depending upon compliance level. Self cert is low cost and a more likely outcome.	Requires significant product testing to ascertain how to measure and set the level of the threshold	High - Potential for certain fibre types/construction techniques to be effectively banned.
3	Development of Product Labelling	Low – Consumer awareness will increase, but purchasing decisions are unlikely to be significantly affected.	Low – Depending on whether labelling is separate or integrated	Requires significant product testing to ascertain what can be put on the label(s).	Low – The industry already attaches various labels; however low-cost items may be affected more.
4	Extended Producer Responsibility through supply of washing machine filtration device	Medium to High Effectiveness depending on the nature and integration of the technology	Medium - depending upon effectiveness of product.	Requires collaboration with washing machine manufacturers and agreement on how the filtration device will be funded. Unclear, at this stage, whether technologies are effective.	Potential to have high impact on washing machine manufacturers for redesign.

The measures with the greatest likely impact are the setting of a maximum threshold (2) and the funding of a washing machine filter through EPR (4). They both could result in a large reduction in microplastics emission to the aquatic environment. Product labelling is likely to have a much lower impact.

In terms of feasibility, the situation is reversed. Introducing product labelling can be implemented without too much further research (if a simple advice label is all that is required). However, the setting of a maximum threshold and the design of a filter require a significant amount of work to implement.

The options packages presented in Table 28 show what may be possible when combining measures. Options 1 and 2a both require the development of a standard test method, although a simple advice label in option 2b could be achieved without a standard test being developed. Option 3 may be the preferred option if the technical feasibility of creating a test standard is found to be too challenging, or its application becomes too costly.

Table 28 - Summary of Synthetic Clothing Options

Option	Measures	Description
Option 1 – Textiles Industry Action	1 &2	**Development of a standardised measurement procedure followed by the setting maximum threshold.** Many unknowns at this stage, but could be industry lead as a voluntary commitment or as a defined maximum threshold for fibre release as a gate to market.
Option 2a – Awareness Raising	1 & 3	**Development of a standardised measurement procedure followed by development of product labelling.** Product labelling would consist of a comparative fibre release label similar in nature to the current energy label system.
Option 2b – Awareness Raising	3	**Development of product labelling** Can also potentially be a low-cost option (without a standard measurement procedure) if it is limited to washing and care advice rather than a comparative fibre release label.
Option 3 – At Source Capture via EPR	4	**Extended Producer Responsibility through supply of washing machine filtration device.** Standard measure not required for implementation, but further research is needed to develop technologies. Potentially high effectiveness.

In terms of effectiveness it is also important to recognise that Option 1 is unlikely to eliminate all fibre release and therefore a method to capture the fibres may still be necessary. Upstream capture in the washing machine is likely to be easier to implement than a large-scale waste water treatment improvement programme. There are also a number of opportunities for the released fibres to be lost to the environment between the washing machine and WWT; these include losses in pipe work and discharges via CSOs. In this way Option 1 is likely to be more effective than any WWT options for the capture of textile fibres.

On this basis it is recommended that two streams of research are immediately undertaken:

- **To Support the implementation of Option 1** – Research into fibre release (with a specific brief to focus on development of a comparative fibre release test standard). This would require cooperation between industry stakeholders who will lead and collaborate in a transparent manner; and secondly,
- **To Support the implementation of Option 3** – Provision of support for research into the design of a filter integrated into washing machines—either for existing design solutions or proposals for new approaches. Again, the focus should be on collaborative development. This would involve the appliance manufacturing sector with technical input from the textile sector.

8.4 Wastewater Treatment

Table 29 shows a comparison of the selected measures based on their effectiveness at reducing microplastics emission, their cost, their ease of implementation, and their impacts on the WWT industry.

Table 29 – Selected Measures Comparison for Wastewater Treatment

Measures		Reduction Effectiveness	Cost	Implementation	Impacts on Industry(s)
1	Development of Test Standard	Ineffective in itself (although will aid in further understanding and awareness), but is necessary for subsequent measures	Low - Research and development costs shared by industry	This requires cross-industry support and cooperation to develop. It is important that efforts are coordinated	Low – several organisations and researchers are looking into this already.
2	Improved Wastewater Treatment EPR	Medium to High – Could be most effective for textile fibres, but other sources such as tyres do not primarily go thorough WWT	Likely to be High when utilising current technology.	Requires significant upgrades to existing infrastructure over a number of years	High – Investment in WWT infrastructure is long term and changes can be costly and disruptive.

3	Improved Storm Water Treatment EPR	Unclear at this stage. Will be most effective in a targeted approach. Will have the largest effect on capture of tyre wear particles.	Unclear at this stage. If high emission areas can be targeted effectively the costs may be low	Requires investigative work to determine the sites that will have the greatest impact	Low – The industry already attaches various labels; however low-cost items may be affected more.

Measure 1 is a pre-requisite for the successful implementation of measure 2. The two measures are therefore combined into Option1 – the development of a standardised measurement procedure followed by improved WWT through EPR funding. As measure 1 is a prerequisite for measure 2, the ability to achieve this option is determined by the issues of quantification and identification of microplastics in WWT as discussed.

As identified, a great deal of research has been conducted into capture rates of microplastics in WWT in recent years. Test procedures are beginning to be standardised as researchers learn more about what is needed to perform accurate sampling. The development of a standard testing protocol that WWT plants can conduct to determine the microplastics load is a key step toward an EPR scheme. The main barrier to this is that it is very difficult to identify the source of the microplastics in effluents. This is a key step if costs are to be assigned accordingly. It remains to be seen whether this is possible for some sources; for example, differentiating between tyre wear particles and artificial turf infill would certainly be challenging, especially in a cost effective and repeatable manner.

Table 30 shows how the measures are combined into the two proposed options.

Table 30 - Summary of WWT Options

Option	Measures	Description
Option 1 – WWT EPR	1 &2	**Development of a standardised measurement procedure followed by improved WWT through EPR funding.**
Option 2 – Storm water EPR	3	**Improved Storm Water Treatment through EPR funding**

Option 2 is primarily focused on addressing the largest fraction of the largest source of microplastic emissions from tyres. Although the proposed options for the tyre industry will reduce these emissions, there will always be wear as long as there are tyres. It is therefore important to combine one or more of the tyre wear options with an effective capture system. Storm water being the main pathway for tyre wear to enter waterways means that this should addressed. However, the lack of data around where the tyre wear will enter water courses prevents any detailed analysis of this at present. It is therefore recommended that further research is conducted which is focused specifically on identifying and measuring instances of tyre wear in storm water. Sampling different road types with varying levels of traffic density will be the first step to identifying potential hotspots that could be mitigated with improved storm water management such as a wetland. Existing storm water management infrastructure should also be further investigated for its effectiveness in capturing microplastics and any potential impact of microplastic accumulation in these areas.

On this basis it is recommended that three streams of research are undertaken;

- **To Support the implementation of Option 1** – Research into the capture of microplastics in WWT. Whilst there are several pools of research being carried out on this subject, there is no strategic EU level working group on the subject. This will likely lead to duplication of work. It is recommended that The Commission facilitate the forming of such a group so that research can be aligned and shared more effectively and the development of a standard can be expedited.
- There are several European based projects that are looking at the problem of microplastics in WWT plants and how to remove them from the effluent. The most promising ones also seek to prevent microplastics from becoming part of sewage sludge as well (Such as Wasser 3.0, described in Appendix A.6.4.4). This is important as there are no current viable methods or even speculated methods for removal of microplastics from sludge. Comments at the WWT stakeholder workshop suggested this is very unlikely to be possible at reasonable cost. Therefore, for any microplastics removal technology to be viable it should incorporate all of the following characteristics in which the current technologies that are part of the Option 1 fall short in most, if not all respects:
    - Be more cost effective than upstream reduction measures;
    - Be capable of retrofitting to exiting plants throughout Europe;
    - Be scalable to different sized plants; and
    - Prevent microplastics from entering both effluent and sludge if the sludge is applied to land and not incinerated.[186] Simply increasing the capture of microplastics in sewage sludge, and then applying this sludge to land, may well lead to negative consequences to soil fauna. This is an area of possible impact that is currently not well understood.
- **To Support the implementation of Option 2 -** Research into the effectiveness of storm water management systems in capturing microplastics. Expanding on our existing understanding of how microplastic behave in storm water is key to providing the correct guidance for the building and maintenance of road and urban infrastructure.

186

8.5 Final Conclusions

Table 31 provides a summary of the options presented in the previous sections. From this it is clear that the largest reductions in both source emissions and emissions to surface water can be achieved through measures targeted at reducing emissions at source. Supply Chain Accreditation for pre-production pellets is likely to have the largest reduction impact—600,000 tonnes cumulative reduction to surface waters between 2017 and 2035—and is also expected to be the most cost effective. However, it is important to note that the amount of pellet loss is subject to some uncertainty, therefore the reduction impacts also come with a reasonably high level of uncertainty. However, what is clear is that pre-production pellets are frequently found in significant numbers on European beaches[187] (this is despite typical beach litter surveys not looking for or counting pellets due to their small size), so there is a strong marine-litter prevention rationale for action targeted towards this source.

Similarly, source prevention for tyre wear abrasion is likely to have a large impact—a cumulative reduction in emissions to surface water of 500,000 tonnes. The amount of tyre wear generated at source has a reasonable level of certainty associated with it, but its pathways to various environments are currently not as well understood. Measure 3, using the Type Approval Regulation to remove the worst performing tyres from the market, and the combined measure (Type Approval plus including tyre abrasion rates on the EU tyre label) both appear to be relatively cost-effective in preventing emissions at source compared with other measures. The testing required to implement these measures is estimated to add between €0.03 and €1.43 onto the cost of a tyre. However, even the combined measure is only expected to reduce emissions to surface water by 33% (of tyre wear emissions). Therefore, it is also important to consider downstream measures such as capture in storm water, as this is expected to be the dominant pathway for microplastics emitted on roads. Costs for this are difficult to estimate as it is not known how much infrastructure would be needed to achieve a certain capture rate – this being strongly influenced by the level of traffic on particular roads – and thus primary research in this area is needed. That having been said, if storm water management is approached on a case by case basis by targeting hotspots for microplastics emissions, it is likely to cost more per tonne than preventative measures, but less than improvements to wastewater treatment plants.

Source prevention measures for textiles are also likely to be cost effective if a self-certification process is used to govern the implementation of a maximum threshold. If (third party) testing of individual textile products is necessary to regulate this, the costs may begin to make downstream capture more appealing. Similar to tyre wear, however, such measures might also be expected to have limited impact (however this largely depends of where the maximum fibre release threshold can feasibly be set) and therefore downstream measures may also be necessary regardless of the cost effectiveness of source measures. For textiles the cost-effectiveness of capture at the washing machine via a filter or at a WWT plant appears to be very similar. However, there are some more subtle qualitative differences that suggest capture at the machine may be more favourable. Firstly, current WWT technology sequesters microplastics in sludge, which may simply transfer the issue for countries that apply sludge to land. Secondly, it should also be recognised that if any of the measures aimed at reducing the key sources of microplastics through WWT are implemented, the cost-effectiveness of any infrastructure improvements would decrease significantly. Tyre wear

[187] Fidra (2017) *The Great European Nurdle Hunt*, June 2017

(25%), pre-production pellets (27%) and textiles (40%) are the largest contributors to microplastics loads in WWT and they all appear to have more cost-effective source prevention measures associated with them. For these reasons it may be more appropriate to investigate washing machine capture in the absence of proven cost-effective capture in WWT.

Table 31 – Summary of Measures (Emissions Using Midpoint Baseline Projections)

Measure		Cumulative Emissions 2017-2035 (tonnes)		Cumulative Reduction from Baseline 2017-2035 (tonnes)		Annual Cost per Tonne Prevented *at Source*
		Source Emissions	Surface Water Emissions	Source Emissions Reduction	Surface Water Emission Reduction	
Automotive Tyres						
Baseline		***11,200,000***	***2,100,000***	-	-	
Measure 2 -Tyre Label	Low	10,900,000	2,040,000	300,000	60,000 (3%)	Circa €11,000
	High	10,400,000	1,900,000	800,000	200,000 (19%)	Circa €4,000
Measure 3 -Type Approval		10,100,000	1,900,000	1,100,000	200,000 (10%)	Circa €3,000
Combined		8,700,000	1,600,000	2,500,000	500,000 (33%)	Circa €1,300
Pre-Production Plastics						
Baseline		***2,200,000***	***1,100,000***	-	-	
Measure 4 - Supply Chain Accreditation		800,000	600,000	1,400,000	600,000 (55%)	Circa €950
Measures 1-3 -Horizontal Measures		1,200,000	700,000	1,000,000	400,000 (36%)	Circa €1,400
Textiles						
Baseline		***600,000***	***250,000***	-	-	-
Measure 2 - Maximum Threshold	10%	500,000	210,000	100,000	40,000 (16%)	Self cert €500—20k Third Party €4k—100k
	20%	400,000	160,000	200,000	90,000 (36%)	
Measure 3 - Labelling		500,000	210,000	100,000	40,000 (16%)	
Measure 4 -Washing Machine Filter	Filter	300,000	130,000	300,000	120,000 (48%)	€50k—125k
	Cora Ball	500,000	220,000	100,000	30,000 (12%)	€41k—104k
	Guppy Friend	500,000	200,000	100,000	50,000 (20%)	€44k—112k
Wastewater Treatment						
Baseline		**-**	***600,000***	-		
Measure 2 - Improved WWT		-	400,000	-	200,000 (33%)	€45k—137k

Note:

4. **Emissions figures rounded to nearest 100,000 or 10,000 for those less than 100,000.**
5. **Wastewater treatment cost per tonne is per tonne reduction into surface water as it is not a source of microplastics.**
6. **All figures are rounded**

9.0 Additional Research Needs and Actions

Although there are many sources of microplastics identified in this study, not all have positive and actionable measures associated with them. This is mostly due to a lack of data or knowledge around the nature and spatial distribution of these microplastic releases. For some of the key sources, such as tyre wear, fibres from washing of textiles and pellet loss, these have either been positively identified in the marine (and other) environments, or their emissions—if not their precise pathways—are well understood. For other sources, such as paint wear and fishing gear, less is known. This is a significant barrier towards proposing and designing effective policy measures. Therefore, the following sections outline some of the research gaps and potential policy measures that could be implemented once this information or data is obtained.

9.1 Fishing Gear

Although there has been a large focus on lost fishing gear and its contribution to both macro and microplastic pollution there has been significantly less work on the microplastic generation during use. There are three key pieces of information that are necessary to quantify the problem and begin to understand whether there are effective measures that can be put in place;

1) The amount of fishing gear used and how often it is used and replaced;
2) The amount that is worn off during use and the nature of the wear; and
3) Whether different materials/constructions used in fishing activities are more prone to wear.

The first issue of use data has been an ongoing problem for some time, but data may be improved in response to increased focus on fishing gear within the EU Commission’s 2018 Plastics Strategy. The strategy proposes to target measures for reducing the loss or abandonment of fishing gear at sea.

Whilst large pieces of fishing gear are known to break off during use, the nature of smaller microplastic wear and tear is not well understood. Fishing gear is also a broad term for a large number of products ranging from the fibres in ropes and nets to particles chipped off of buoys or similar. The small amount of research in this area to date has focused on nets, but losses are very much theoretical. Field research would need to be carried out to determine how much is lost through wear with the experiments designed to isolate this from macro-sized losses—this may be very difficult to achieve when nets have been significantly bio-fouled during use. Such experiments may bring forth recommendations for materials or constructions that are less prone to wear. Early indications suggest this may be the case and various mechanisms could be employed to incentivise the use of materials less prone to degradation.

9.2 Paints and Coatings

The emissions of paint particles from ships, buildings and roads are largely theoretical. Whilst these surfaces are known to wear during their lifetime, the exact nature of the resulting particles is yet to be established. Several studies have found larger paint particles downstream of potential sources, but it is likely that most particles will be very small and difficult to identify in the environment. The paint industry itself, whilst cooperating with this project’s aims, does not appear to have studied the subject until recently and only in response to increasing concern. Further engagement with the paint industry would be imperative to help understand the issue further whilst also attempting to identify these particles in the environment.

It will be particularly important to determine the nature of wear from road markings as this was estimated to be one of the larger emission sources. As the estimates are largely theoretical at this stage—in terms of the nature of the particles given off rather than its existence as a source—it was not possible to determine specific measures to address these. It is likely, however that most of the measures aimed at reducing tyre wear or capturing tyre wear particles at the roadside will also combat emissions from road markings. This would largely be dictated by the proportion that is worn away by abrasion from tyres rather than weathering. However, the targeted approach to storm water management in high traffic areas is also likely to improve the situation.

One key area which requires further investigation is whether different types of road marking material and application method (solvent, thermoplastic or cold plastic etc.) are likely to perform better from a wear perspective. Although this is likely to be the case it is not a simple task to substitute for better performing (abrasion resistant) materials. Different materials are favoured in different countries and associated industries are often localised to the particular region. Road types, traffic levels, visibility requirements, drying speed and ease of maintenance all play a role in material and application method choices.

Similarly, marine paint has not been addressed with specific measures. With further information from CEPE, the original release estimates have been significantly reduced. However there has been no empirical data gathered to support the assertions of the paint industry, therefore this source of microplastics should not be dismissed at this stage—especially these are direct emissions to the oceans. It is also likely that marine coatings in their uncured form during application could be a larger source of microplastic emissions during application, however this was out of scope of the current report (and was also not investigated in the parallel AMEC study). It is recommended that specific field experiments are carried out to determine the rate and nature of paint particles that are worn away during use. This may help to determine whether specific mitigating measures can be developed which could include improved maintenance procedures to prevent damaged paint surfaces from releasing particles.

9.3 Artificial Turf

There appears to be little awareness in the artificial turf industry or amongst pitch owners/managers of the potential for infill to be a source of environmental problems. Whilst the relative size of the problem is thought to be small compared with other emission sources it is one that is growing the fastest. There are also significant issues with the handling and disposal of infill at the end of life which, whilst out of scope for this study, adds to the potential for this material to end up in the environment.

Mitigation measures are potentially simple to achieve if implemented during the design and construction of the field. These are similar to ones employed at factories as part of Operation Clean Sweep (OCS) for pellet loss mitigation; traps for drains both inside and out, good housekeeping with spills regularly cleaned up and a site designed to prevent infill from migrating outside of the pitch area, are all simple but effective measures. On this basis it may be useful for the industry to adopt or adapt elements of OCS.

Whilst it is difficult to justify measures at an EU level, individual countries should be encouraged to specify this best practice for the procurement of new pitches—as many pitches are funded with public money. Implementation of this in countries such as the Netherlands, France and Germany — which account for the majority of pitches—would have a significant impact on the issue.

9.4 Biobeads (Biomedia)

Whilst there is emerging recognition of biobeads or similar biomedia used in WWT, no specific measures were developed. It is clear that the handling of these products can lead to spills and therefore inclusion within measures to reduce pre-production pellet loss should be considered. Losses during use are claimed by the WWT industry to be near zero, but these have been found in large quantities on UK beaches. A report[188] from Surfrider published very recently (February 2018) suggests this problem could be wider spread than the UK as these distinctive pellets have been found throughout the French coastline and in particular the Bay of Biscay. The extent to which these emissions occur and their proportion relative to other sources is, as yet, unknown. However, the European WWT industry should recognise the issue, examine how widespread their use is and develop mitigation measures.

[188] Surfrider Foundation Europe (2018) *Pollution des plages et des cours d'eaux par les biomédias, supports en plastique de prolifération bactériologique utilisés dans le traitement des eaux usées*, February 2018

APPENDICES

A.1.0 Microplastics Impacts

A.1.1 Background

Microplastics have been released into aquatic environment in very large quantities since large-scale production and use of plastics began in the 1950s. Consequently, they have been observed in freshwater bodies and throughout the global ocean, in water, sediments and biota. This raises the question of whether there are potential negative effects on aquatic organisms, sufficient to cause an impact at the population level. It has also prompted concerns about the exposure of humans to microplastics, principally though ingestion of contaminated foodstuffs.

Such concerns were first expressed in the early 1970s. A small number of publications reported the presence of small plastic fragments in plankton nets, and cited their potential to act as 'sponges' of organic contaminants already present in the aquatic environment[189,190] and as potential vectors for delivery of additional contaminant burden to marine organisms. The topic then remained largely dormant until about ten years ago. Over the past decade there has been a considerable expansion of interest among researchers, industry sectors, policy makers and politicians. This has been accompanied by an almost exponential increase in the number of scientific publications on various aspects of the issue.

The impact of microplastics on organisms depends on the physical and chemical characteristics of the particles, the exposure pathway(s) and nature of the hazard. These determine the degree of risk of effects occurring in the aquatic environment and the risk to human health.[191,192,193] Chemical hazards include exposure to the wide variety of potentially hazardous substances associated with some types of plastics and applications, included during manufacture.[194] There is also a potential risk from hydrophobic contaminants present in the environment, such as PCBs and PAHs, that may be absorbed by plastics following release. Given the multiplicity of particle types and compositions, the number of potential pathways and the varied nature of the potential

[189] Carpenter, E.J., and Smith, K.L. (1972) Plastics on the Sargasso Sea Surface, *Science*, Vol.175, No.4027, pp.1240–1241

[190] Carpenter, E.J., Anderson, S.J., Harvey, G.R., Miklas, H.P., and Peck, B.B. (1972) Polystyrene spherules in coastal waters, *Science (New York, N.Y.)*, Vol.178, No.4062, pp.749–750

[191] GESAMP (2016) *Sources, fate and effects of microplastics in the marine environment: part 2 of a global assessment*, December 2016

[192] United Nations Environment Programme (UNEP), and UNEA (2016) *Marine plastic debris and microplastics – Global lessons and research to inspire action and guide policy change*, 2016

[193] EFSA Panel on Contaminants in the Food Chain (CONTAM) (2016) Presence of microplastics and nanoplastics in food, with particular focus on seafood, *EFSA Journal*, Vol.14, No.6

[194] Hansen, G.J.A., Vander Zanden, M.J., Blum, M.J., et al. (2013) Commonly Rare and Rarely Common: Comparing Population Abundance of Invasive and Native Aquatic Species, *PLoS ONE*, Vol.8, No.10

physical and chemical hazards, assigning risk is challenging. It is, however, necessary to inform any decisions on possible exposure reduction measures.

This section of the report summarises the physical and chemical characteristics of microplastics in relation to the major sources and categories (e.g. vehicle tyre wear, textile fibres, fisheries-related, damaged durable plastics, paint flakes), based on published information, and examines whether different types of particles may be expected to vary in the effects they exhibit.

This is a very active area of research, with new papers appearing in the peer-reviewed literature every week. There have been several major assessments undertaken of the sources, fate and effects of microplastics by international and European bodies[195,196,197] and on behalf of individual governments.[198,199,200,201] There remains considerable uncertainty on whether microplastics pose a significant risk of harm to aquatic organisms or humans. The text below provides an overview of our current understanding, and highlights the factors that influence the extent of any impact.

A.1.2 Physical and Chemical Characteristics of Microplastics

Although microplastics have come to be defined loosely as plastic particles < 5 mm in diameter, the term covers a very wide range of particle sizes, including nano-sized[202]. Microplastics also exhibit a wide variety of aspect ratios, including near-spherical, sub-spherical, irregular pieces, flakes and fibres. This reflects variability in the sources, physical and chemical properties and subsequent transformations that have occurred in the environment. The aspect ratio influences behaviour and impact on organisms. Such variability in form needs to be considered when interpreting the results of field observations and experimental studies that refer to a particle diameter, without defining

[195] GESAMP (2015) *Sources, Fates and Effects of Microplastics in the Marine Environment: A Global Assessment*, 2015, http://www.gesamp.org/data/gesamp/files/media/Publications/Reports_and_studies_90/gallery_2230/object_2461_large.pdf

[196] GESAMP (2016) *Sources, fate and effects of microplastics in the marine environment: part 2 of a global assessment*, December 2016

[197] EFSA Panel?on Contaminants in the Food Chain (CONTAM) (2016) Presence of microplastics and nanoplastics in food, with particular focus on seafood, *EFSA Journal*, Vol.14, No.6

[198] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

[199] Mepex (2016) *Primary microplastic- pollution: Measures and reduction potentials in Norway*, April 2016

[200] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

[201] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

[202] GESAMP (2015) *Sources, Fates and Effects of Microplastics in the Marine Environment: A Global Assessment*, 2015, http://www.gesamp.org/data/gesamp/files/media/Publications/Reports_and_studies_90/gallery_2230/object_2461_large.pdf

the aspect ratio. For example, a spherical particle with a diameter of 1.0 mm can be expected behave rather differently from a fibre of length 1.0 mm and diameter <0.1 mm, both in the environment and when interacting with organisms. In addition to size and shape, the density of the polymer will influence the transport and fate of the microplastics particle in the environment. Of the most common polymers, only PE and PP will float in pure water. Most EPS will float in the ocean but PS, PVC, PA, PET and PE are denser than seawater, and will sink in the absence of flotation (See Table 32 in Appendix A.1.0).

The behaviour of microplastics in the aquatic environment is influenced by both density and shape, in addition to the dynamics of the receiving environment. For example, the behaviour of a PET fibre lies somewhere between that of a typical mineral grain (density approximately 2.65) and that of a PE particle (density 0.01 – 0.95). For this reason, PET fibres released from textiles during washing will tend to accumulate preferentially on river banks and shorelines[203] due to their rapid deposition from water. They will be more liable to wind-induced resuspension, on drying out, than the surrounding mineral grains, thus presenting a potential exposure pathway to humans by inhalation.

Many different polymer compositions and combinations are used in the manufacture of thermoplastic and thermo-set plastics. These are in addition to the familiar common polymers in relatively pure form (PE, PS, PVC, PP, PA, PES), which account for most of the volume production, something that is reflected in environmental surveys.[204]

Lithner et al. (2011)[205] presented a comprehensive review of 55 polymers in terms of the hazard ranking of their component monomers and potential effects, including on release to the environment. Thirty-one polymers were placed in the highest IV and V hazard categories. These were members of the polymer families' polyurethanes, polyacrylonitriles, polyvinyl chloride, epoxy resins and styrenic co-polymers (ABS, SAN and HIPS).

Additional substances are often included in the polymer to impart desirable properties during manufacture or use. These include: plasticisers to provide flexibility, flame-retardants (e.g. PBDEs), stabilisers, antioxidants, pigments, UV stabilisers and monomers such as bisphenol A (e.g. used in the manufacture of PC) and vinyl chloride.[206] Additives can represent a very significant fraction of the plastic. For example, PVC can contain up

---

[203] Brown, D.M., Wilson, M.R., MacNee, W., Stone, V., and Donaldson, K. (2001) Size-Dependent Proinflammatory Effects of Ultrafine Polystyrene Particles: A Role for Surface Area and Oxidative Stress in the Enhanced Activity of Ultrafines, *Toxicology and Applied Pharmacology*, Vol.175, No.3, pp.191–199

[204] GESAMP (2015) *Sources, Fates and Effects of Microplastics in the Marine Environment: A Global Assessment*, 2015, http://www.gesamp.org/data/gesamp/files/media/Publications/Reports_and_studies_90/gallery_2230/object_2461_large.pdf

[205] Lithner, D., Larsson, Å., and Dave, G. (2011) Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition, *Science of The Total Environment*, Vol.409, No.18, pp.3309–3324

[206] Hermabessiere, L., Dehaut, A., Paul-Pont, I., Lacroix, C., Jezequel, R., Soudant, P., and Duflos, G. (2017) Occurrence and effects of plastic additives on marine environments and organisms: A review, *Chemosphere*, Vol.182, pp.781–793

to 80% by weight of the plasticiser bis(2-ethylhexyl) phthalate (DEHP).[207] PVC accounts for 73% of the total global additives production[208]. Additives are often of low molecular weight and not chemically bound within the polymer. This leads to their leaching into the surrounding environment, including into tissue. Unfortunately, many additives have toxic properties, so an unintended consequence of their use is the potential to induce ecotoxicological and toxicological effects.[209]

A.1.3 Overview of Occurrence and Impacts of Microplastics

Evidence based on environmental observations

There is a rapidly growing body of evidence showing the occurrence of microplastics in rivers, lakes, urban water bodies, shorelines, seabed sediments, surface waters, water column and sea ice. All the major polymer types are represented (Hidalgo-Ruz et al. 2012, GESAMP 2016). GESAMP (2016) presented a very comprehensive review of reported instances of microplastic occurrence, arranged by Phylum. It included many species of fish and shellfish of commercial importance. There has been a concerted effort to elicit evidence of the degree and mechanisms of physical and chemical effects on aquatic organisms. But, the mere presence of microplastics in the gastro-intestinal tract of organisms sampled from the environment should not be taken to imply consequent physical or chemical impact. There is some limited evidence of the transfer of additive chemicals from ingested microplastics to fish and seabirds, specifically PBDEs where it is possible to 'fingerprint' and differentiate characteristic congeners of PBDEs in the plastic in the gut and in the prey food.[210,211] The presence of phthalates in skin biopsies of fin whales has been interpreted as showing the transfer from ingested particles (Fossi et al. 2017), although this has not been definitively proven. With these few exceptions, it has been very difficult to quantify the exposure of organisms to chemicals associated with microplastics. Chemicals associated with plastics abound in the freshwater and marine environment. Distinguishing the relative contributions of microplastics and overall environmental contamination will be extremely difficult. In addition, the presence of a contaminant should not imply that there will an impact. What is completely missing is direct evidence of harm under environmental conditions,

[207] Tickner, J.A., Schettler, T., Guidotti, T., McCally, M., and Rossi, M. (2001) Health risks posed by use of Di-2-ethylhexyl phthalate (DEHP) in PVC medical devices: a critical review, *American Journal of Industrial Medicine*, Vol.39, No.1, pp.100–111

[208] Lithner, D., Larsson, Å., and Dave, G. (2011) Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition, *Science of The Total Environment*, Vol.409, No.18, pp.3309–3324

[209] Wright, S.L., and Kelly, F.J. (2017) Plastic and Human Health: A Micro Issue?, *Environmental Science & Technology*

[210] Tanaka, K., Takada, H., Yamashita, R., Mizukawa, K., Fukuwaka, M., and Watanuki, Y. (2013) Accumulation of plastic-derived chemicals in tissues of seabirds ingesting marine plastics, *Marine Pollution Bulletin*, Vol.69, Nos.1–2, pp.219–222

[211] Rochman, C.M., Kurobe, T., Flores, I., and Teh, S.J. (2014) Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment, *Science of The Total Environment*, Vol.493, pp.656–661

certainly at a population level (such as a significant decline in reproductive success due solely to exposure to microplastics).

A range of new techniques is being developed to identify the physical & chemical characteristics of microplastics in field samples more effectively and with greater certainty.[212] This will greatly assist in developing improved risk models.

Evidence based on experimental studies

Direct evidence of the impact of microplastics on aquatic organisms has been obtained from experimental studies. These have yielded very valuable information about the uptake and internal relocation of plastics particles, revealing cross-membrane transfer, inflammatory responses, and effects on growth rates, as described extensively by GESAMP (2015, 2016). There is also some limited evidence of trophic transfer through the food chain (Outi et al. 2014). Reported impacts have been most convincing with experiments involving particles in the nano to 10s µm size range. However, many feeding experiments have used particle concentrations far exceeding those to which the organisms are exposed in the natural environment. In addition, most experiments examining the uptake and internal transfer of microplastics in experimental systems have used microspheres of defined dimensions. Such particles can be obtained commercially, with desired properties such as fluorescence. These studies have yielded valuable information about the influence of particle size on the relocation of particles, such as crossing cell membranes. But, for example, there has been a lack of information about the influence of particle shape, especially fibres[213] which is known to be a significant variable. In addition, many experimental studies have used particles with a diameter of a few µm or less. The presence of particles in this size range has not been studied in most environmental samples, partly due to analytical constraints. The greatest challenge remains interpreting results from experimental studies in terms of likely effects in the aquatic environment.

Evidence of impacts on humans

There is no direct evidence of effects in humans of environmental exposure to microplastics. Assessing the risk of harm from exposure to microplastics relies on gathering evidence from a range of other disciplines, including medicine (pharmacology, orthopaedics, physiology), toxicology, materials science and nano-sciences (Lithner et al. 2011, Wright and Kelly 2017). This includes exposure by ingestion and inhalation. There is growing evidence of the influence of particle size, shape and surface properties on the transfer and effects of particulates within humans, from in vivo studies, including synthetic polymers.

---

[212] Maes, T., Jessop, R., Wellner, N., Haupt, K., and Mayes, A.G. (2017) A rapid-screening approach to detect and quantify microplastics based on fluorescent tagging with Nile Red, *Scientific Reports*, Vol.7, p.44501

[213] Cole, M. (2016) A novel method for preparing microplastic fibers, *Scientific Reports*, Vol.6, No.1

A.1.4 Estimating impacts – general considerations

A.1.4.1 Human impacts

Exposure pathways by ingestion

Humans may come into contact with microplastics via inhalation or ingestion. Microplastics have been observed in a number of foodstuffs, including fish, shellfish and sea salt[214,215,216] indicating that ingestion of small particles is inevitable. Ingestion of larger sub-spherical microplastics appears less likely to lead to direct physical effects, provided egestion occurs relatively quickly. Microplastics > 150 µm are unlikely to be able to cross the human gastrointestinal lining, but there is concern that persorption (physical engulfing) of particles < 150 µm can occur via M cells in the Peyer's Patches[217], as has been demonstrated for starch granules.[218] Toxicity is thought to be linked to inflammation due to the persistent nature of the polymer and surface chemistry (Wright and Kelly 2017). The extent to which micro-fibres can cross cell membranes remains unclear. Hussain et al. (2011) has reported particles < 150 µm and nano-size particulates crossing the gut lining, and nano-polymers are widely used for drug delivery. Nano-sized PS particles have been shown to be taken up by macrophages and induce cellular damage in human systems (Geiser et al. 2005).[219] But, according to the European Food Safety Authority (EFSA), there is insufficient data to carry out a reliable risk assessment for nano-sized plastic particles (EFSA, 2016). The risk of significant physical or chemical toxicological effects from the ingestion of microplastics will depend on: the size range of the particles ingested the surface properties of the particles; the number of particles the body is exposed to; and, the efficacy of the body's defence mechanisms to identify and eliminate particles from the body. A comprehensive review of the potential risks due to ingestion, and inhalation, of microplastics has been published recently (Wright and Kelly 2017).

A number of studies have reported the presence of microplastics in commercial and non-commercial fish and shellfish species, although in relatively low numbers (Lusher et al. 2013, GESAMP 2016), and in seafood on sale in markets (Rochman et al. 2015). A comprehensive review of the literature is presented in GESAMP (2016). Of possible concern is the contaminant burden acquired through the consumption of food, especially seafood, contaminated as a result of food chain effects. Many of the chemicals

---

[214] Rochman, C.M., Tahir, A., Williams, S.L., et al. (2015) Anthropogenic debris in seafood: Plastic debris and fibers from textiles in fish and bivalves sold for human consumption, *Scientific Reports*, Vol.5, p.14340

[215] Yang, D., Shi, H., Li, L., Li, J., Jabeen, K., and Kolandhasamy, P. (2015) Microplastic Pollution in Table Salts from China, *Environmental Science & Technology*, Vol.49, No.22, pp.13622–13627

[216] Van Cauwenberghe, L., and Janssen, C.R. (2014) Microplastics in bivalves cultured for human consumption, *Environmental Pollution*, Vol.193, pp.65–70

[217] Wright, S.L., and Kelly, F.J. (2017) Plastic and Human Health: A Micro Issue?, *Environmental Science & Technology*

[218] Nagele, W., Müller, N., Brugger-Pichler, E., and Nagele, J. Persorption of Plant Microparticles after Oral Plant Food Intake

[219] Froehlich, E., Meindl, C., Roblegg, E., Ebner, B., Absenger, M., and Pieber, T.R. (2012) Action of polystyrene nanoparticles of different sizes on lysosomal function and integrity, *Particle and Fibre Toxicology*, Vol.9, No.1, p.26

associated with plastics usually are present in relatively high concentrations in the surrounding environment, making it difficult to ascribe the precise source of contamination to natural prey or exposure through the gills for example.[220] One exception to this general rule concerns certain chemicals included during manufacture, such as bisphenol A and brominated flame retardants, which may be present in high concentrations (GESAMP 2015). Several studies have reported the transfer of PBDEs from fragments of durable plastics into the adipose tissue in birds and fish (Tanaka et al. 2013, Rochman et al. 2013, 2014).[221,222,223]

The risk to humans of exposure will depend on: the concentrations of contaminants in the target organs; the fraction of the organism that is consumed (e.g. muscle tissue, whole organism); and, the quantity consumed. But, it is considered that the risk from contaminated seafood consumption, either due to the presence of microplastics in the foodstuff or from exposure to microplastics during the lifecycle of the organism, is low, on the basis of current knowledge, in comparison with other exposure pathways (UNEP 2016, FAO in press). Freshwater or saltwater fish and shellfish containing microplastics, or microplastic-associated chemical contamination originating outside Europe may reach the European market, due to the nature of the global food trade. Further discussion is considered out of scope of the present study, although there is a clear need for more information about this source.

Exposure pathways by inhalation

Humans may be exposed to microplastics by inhalation, following release to the aquatic environment by the drying out of river banks and shoreline sediments and subsequent airborne re-suspension, or from wave-formed aerosols. High concentrations of textile fibres have been reported from urban freshwater and marine shoreline sediments, reflecting the input of wastewater and limited transport as a consequence of the particles' higher density. No reported data have been identified that quantify this exposure route. Generally, concern for direct physical toxicity is focussed on inhalation of fine particulate for which there is an extensive literature concerning inorganic particles (e.g. silica dust, asbestos fibres, nano particles). The issues of concern are likely to be similar for inhaled fine plastic particles, and the literature has been reviewed recently by Wright and Kelly (2017).

The human body has evolved mechanisms to capture and either immobilise or eject fine sub-spherical particles inhaled into the lung. This is achieved by the process of phagocytosis, in which fine particles become engulfed by macrophages and excreted in

---

[220] ECsafeSEAFOOD (2017) *Micro-plastics and the associated contaminants in various environmental compartments and biota. Deliverable D2.3. Priority environmental contaminants in seafood: safety assessment, impact and public perception.*, 2017

[221] Tanaka, K., Takada, H., Yamashita, R., Mizukawa, K., Fukuwaka, M., and Watanuki, Y. (2013) Accumulation of plastic-derived chemicals in tissues of seabirds ingesting marine plastics, *Marine Pollution Bulletin*, Vol.69, Nos.1–2, pp.219–222

[222] Rochman, C.M., Hoh, E., Kurobe, T., and Teh, S.J. (2013) Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress, *Scientific Reports*, Vol.3

[223] Rochman, C.M., Lewison, R.L., Eriksen, M., Allen, H., Cook, A.-M., and Teh, S.J. (2014) Polybrominated diphenyl ethers (PBDEs) in fish tissue may be an indicator of plastic contamination in marine habitats, *Science of The Total Environment*, Vol.476–477, pp.622–633

the respiratory system via the lymphatic system.[224] The mechanisms of entry, internal transfer and impact of fibres in the human body has focussed on asbestos fibres. Long or large fibres are not effectively removed by this process, leading to chronic inflammation. This facilitates the transfer of asbestos fibres, for example, through cell membranes. The precise routes of further transport are a matter of debate, but it is thought that longer fibres will become trapped at the lymph stomata, leading to a greater likelihood of carcinogenesis.[225] It does appear that the shape of fine particulate is significant, with fibres showing a greater propensity to induce toxicity.[226] However, it has not been possible to identify published research, in the medical or nanomaterial literature, specifically on the possible impact of nano- and micro-fibres exposure in the human body, derived from environmental plastics.

A.1.4.2 Environmental impacts

Field-based observations and inferences

Microplastics have been observed in a wide variety of habitats and reported in over 100 species of marine biota, and in a smaller number of freshwater biota. A comprehensive literature review of field observations of microplastics in marine organisms, covering a wide range of trophic levels and feeding traits, is provided as an annex in GESAMP (2016). There may be effects on some individual organisms, at the concentrations observed, but the evidence remains unclear. Effects at a population level appear unlikely, on the basis of evidence published to date. But, there is a dearth of information about the presence and possible effects of nano-sized plastics, which may be more likely to induce toxicity (See Table 35 in Appendix A.1.0).

Laboratory-based studies

Most knowledge about the possible ecosystem effects has been gained from laboratory-based experimental studies. These are a very useful way of advancing our knowledge, but some caution must be exercised in the interpretation of the results.

A comprehensive literature review of the details and results of laboratory-based experiments, exposing a wide range of marine and freshwater organisms to microplastics, is provided as an annex in GESAMP (2016). A summary of key effects related to exposure to nano- and fine microplastics has been compiled as part of a study commissioned by FAO, and is reproduced in Table 35 in Appendix A.1.0. Researchers used a number of end points, including growth rate, cell damage, retention efficiency, larval success, fecundity, internal redistribution and chemical transfer. There have been a limited number of studies investigating trophic transfer. Significant negative impacts

---

[224] Nagai, H., and Toyokuni, S. (2012) Differences and similarities between carbon nanotubes and asbestos fibers during mesothelial carcinogenesis: Shedding light on fiber entry mechanism, *Cancer Science*, Vol.103, No.8, pp.1378–1390

[225] Poland, C.A., Duffin, R., Kinloch, I., et al. (2008) Carbon nanotubes introduced into the abdominal cavity of mice show asbestos-like pathogenicity in a pilot study, *Nature Nanotechnology*, Vol.3, No.7, pp.423–428

[226] Stoehr, L.C., Gonzalez, E., Stampfl, A., Casals, E., Duschl, A., Puntes, V., and Oostingh, G.J. (2011) Shape matters: effects of silver nanospheres and wires on human alveolar epithelial cells, *Particle and Fibre Toxicology*, Vol.8, No.1, p.36

were reported in many of the studies. However, in most cases, exposure was to particle concentrations far in excess of realistic environmental conditions. Thus, the experimental studies have demonstrated potential mechanisms and effects, but it cannot be concluded that there are actual effects under natural conditions. An additional challenge is to deduce potential impacts at a population level.

A further complication to assessing the applicability of laboratory-based experimental results is that spherical particles are used in most cases. This is for reasons of commercial availability and desire to control for size effects, but results in a significant mismatch between the prevalence of micro-fibres observed in the environment and the availability of experimental toxicity data (Cole 2016). Given that the aspect ratio is known to influence toxicity (Stoehr et al. 2011), recent advances in method development to produce consistent fibres for experimental studies (Cole 2016) are to be welcomed.

A.1.5 Additional Tables

Table 32: Densities of common polymers found in the aquatic environment

Resin type	Common applications	Density
Polyethylene	Plastic bags, storage containers,	0.91–0.95
Polypropylene	Rope, bottle caps, gear, strapping	0.90–0.92
Pure water		*1.00*
Polystyrene (expanded)	Cool boxes, floats, cups	0.01–1.05
Average seawater		*1.025*
Polystyrene	utensils, containers	1.04–1.09
Polyvinyl chloride	Film, pipe, containers	1.16–1.30
Polyamide or Nylon	Fishing nets, rope	1.13–1.15
Poly(ethylene terephthalate)	Bottles, strapping	1.34–1.39
Polyester resin + glass fibre	Textiles, boats	>1.35
Cellulose Acetate	Cigarette filters	1.22–1.24

Source: Adapted from Andrady (2011)[227]

[227] Andrady, A.L. (2011) Microplastics in the marine environment, *Marine Pollution Bulletin*, Vol.62, No.8, pp.1596–1605

Table 33: Common polymers, typical applications, hazard ranking and potential for microplastic generation by shape category. Selected monomers and hazard rankings

Polymer	Typical applications	Monomer	Hazard level	Hazard score	Potential for microplastic generation
PAN (co-monomer example)	Acrylic fibres, clothing, yacht sails, fire-resistant textiles	Acrylamide	V	22,240	Fibres from washing, wear and tear
PUR (examples)	foam insulation, carpet underlay, durable wheels, elasticated sports clothing	Propylene oxide	V	20,061	Fibres and fragments from wear and tear
ABS	Wastewater pipes, 3D printing, automotive parts	1,3-butadiene	V	20,001	Fragments from damage, mechanical abrasion and fine aerosol (3D printing)
Epoxy (example)	Metal coatings, electrical insulators, structural adhesives, undercoat for automotive and marine paints, marine repair resins	4,4'-methylenedianline (MDA)	V	13,200	Paint flakes
PVC (plasticised example)	Plumbing, electrical insulation	Benzyl butyl phthalate (BBP) *plasticiser*	V	11,100	Fragments produced during installation or removal
PVC (unplasticised uPVC)	Construction (e.g. window frames), drainage pipes	Vinyl chloride	V	10,001	Fragments produced during installation or removal
PC	Glazing in construction and aviation, data storage discs, drinking vessels	Bisphenol A	IV	1,210	Flakes and fragments due to damage
PMMA	Impact-resistant glazing	Methyl methacrylate	IV	1,021	Fragments due to damage
PA (nylon 6)	Textiles (clothing, carpets)	ε-caproamide	II	50	Fibres due to washing and wear and tear

PS	Disposable food and drink containers and cutlery	Styrene	II	30	Fragments and flakes due to wear and tear and damage
EPS	Construction insulation, fresh food storage (e.g. fish), 'takeaway' containers, flotation devices (e.g. aquaculture floats)	Styrene	II	30	Fragments and flakes due to wear and tear, damage during installation and removal
HDPE	Drinks bottles, bottle caps, piping, storage containers	Ethylene	II	11	Fragments and flakes due to wear and tear
LDPE	Plastic bags, food wrap, food and drink cartons, snap-on lids	Ethylene	II	11	Flakes due to wear and tear
LLDPE	Plastic bags, food wrap, food and drink cartons, flexible tubing	Ethylene	II	11	Flakes due to wear and tear
PP	Potable plumbing, textiles (clothing, carpets), rope, sanitary products, sutures	Propylene	I	1	Fibres due to washing, wear and tear
PVAc	Paper coating, adhesives, sanitary products, water-soluble bags	Vinyl acetate	I	1	Flakes (relatively short-lived)
PET	Textiles (clothing), drinks bottles, packaging trays	Dimethyl terephthalate	Low[1]		Fibres from washing, fibres and flakes from wear and tear
PLA	Biodegradable food containers, 3D printing, sanitary products, mulch films	Lactic acid	Low		Fibres and flakes from wear and tear (relatively short-lived)

Notes:

1. ***Low level of concern for Human Health and the Environment indicated by the European SIDS initial assessment reports/profiles***

Source: Lithner et al. *(2011)*

Table 34: Categories of microplastic by source, shape, composition and potential impact, indicating estimated relative importance and confidence level, based on published evidence

Category	Sub Category	I/C[1]	Shape [2] FB/FL/S	Polymer	Additives[1]	Exposure pathway[3]	Potential human impact[4]	Potential environmental impact	Relative importance[5] H/M/L	Confidence level H/M/L
Artificial Turf	**Infill**	I	S	R PP		IN/A	Physical damage to lungs	Impairment of growth and reproductive success	L	L
	Fibres	C	FB	PE	UVP	IN/A	Physical damage to lungs	Impairment of growth and reproductive success	M	L
Vehicles	**Tyres**	C	FB/S	R	M CB	IN/A	Physical damage to lungs	Impairment of growth and reproductive success	H	M
	Brake dust	C	S	?	M	IN/A	Physical damage to lungs, PAHs,	Impairment of growth and reproductive success	M	L
Synthetic Textiles	**Synthetic clothing**	C	FB	PET PA PAN-Acrylic	FR	IN/A	Physical damage to lungs	Impairment of growth and reproductive success	H	H
	Carpets	C	FB	PA	FR	IN/A	Physical damage to lungs Exposure to endocrine disrupting chemicals (flame-retardants)		M	L

	Cleaning cloths	C	FB			IN/A			L	L
	Hygiene products	C	FB			IN			L	L
Pre-Production Plastics	**Pellets/powders/flakes**	I	FL/S	all		IN		Impairment of growth and reproductive success	M	H
	3D printing	I	S			IN/A			L	L
	Plastics recycling	C	FB/FL/S	all		IN/A		Impairment of growth and reproductive success	M	L
Agriculture	**Mulch films**	C	FL			IN/A			M	L
	Baling films	C	FL			IN/A			L	L
Paints/coatings	**Building**	I/C	FL			IN/A			L	L
	Road	I/C	FL			IN/A			L	L
	Marine (commercial and domestic)	I/C	FL		M	IN	Exposure to endocrine disrupting chemicals	Exposure to endocrine disrupting chemicals Impairment of growth and reproductive success	M	M[6]
Marine	**Fishing gear**	C	FB/S	PA PP		IN/EN			M	L
	Aquaculture	C	FB/S	PA PP		IN/EN	Exposure to endocrine disrupting chemicals	Exposure to endocrine disrupting chemicals Impairment of growth and reproductive success	M	L

Others	**Shoe sole wear**	C	S			IN/A			L	L
	Insulation	I	S	EPS		IN			L	L
	Furniture	I	S	EPS		IN			L	L
	Kitchen utensils	C	S	PS		IN			L	L
	Buildings	C	S	PP PVC		IN/A			L	L
	Indoor dust	C	FB/S	PA PE PP		IN/A	Exposure to endocrine disrupting chemicals		M	L
	Fragmentation of litter and unmanaged waste						Contaminated seafood		H	M

Notes:

1. **I – intentionally added, C – created during use**
2. **Types: FB – fibres, FL – flakes, S – spherical/sub-spherical**
3. **Additives: M – metals, CB – carbon black, UVR – UV retardation, FR – flame resistance (e.g. Vinyl chloride monomer in acrylic),**
4. **Exposure pathway: IN – ingestion, A – inhalation from atmosphere, EN - entanglement**
5. **Based on current published evidence**
6. **Relative importance as a source will be vary locally, regionally & seasonally**

Table 35 - Summary of micro- and nano-sized particle behaviour relevant to humans

Microplastics (0.1 – 5000 µm)	Nanoplastics (1 – 100 nm)
> 150 µm - no absorption	
< 150 µm - in lymph (absorption ≤ 0.3%)	
= 110 µm - in portal vein	
< 1.5 µm - access into organs	
	≤ 100 nm - access to all organs, translocation of blood-brain and placental barrier

Source: Compiled by Chelsea Rochman, adapted from FAO (in press). Permission being sought

A.2.0 Microplastic Source Identification

The following section analyses the current studies that attempt to quantify microplastics emissions, and identifies the main sources.

Table 36 shows the scope of these studies with the product sources of microplastics, set in order of how many studies have researched them and to what level. For example, all studies have looked in depth at automotive tyres, whereas only two studies have mentioned but not quantified pharmaceuticals as a microplastic source. Not all studies have the same scope, therefore an attempt has been made to compare the studies using;

- **microplastic sources**: the generation of microplastics at source;

Which is in contrast to;

- **microplastic emissions:** the amount of microplastics that are emitted to the aquatic environment.

Most studies attempt to quantify the former, but the latter is particularly difficult due to the levels of uncertainly involved. Table 37 shows the sources in the same order as Table 36, but with the estimated ***microplastic source quantities*** shown in grams per capita for comparison. This shows that in general, the sources that have received the most attention in these studies are also the largest sources of microplastics. The exception is personal care products (PCPs) which have received a lot of attention, but are often found to be less significant as a source of microplastics.

Table 36 – European Microplastics Sources and Emissions Studies

Quantified as emissions to water (score = 3), Quantified as sources of microplastics (score = 2) (as amount manufactured and/or emitted with pathways not explicitly identified/quantified), Identified but not quantified (score = 1) (no data)—Higher scores have more sources data available and/or better defined pathways.

Year of study	Country	Automotive Tyres	Marine Paint	Pre-Production Plastics	Textiles	Personal Care Products	Building/ decorative	Road Markings	Artificial Turf	Abrasives	Detergents	Fishing and Aquaculture	Indoor Dust	Waste and Recycling	Footwear	Agriculture	Pharmaceutical
2014/16	Norway[228, 229]	2	3	2	2	2	2	2	3	1		1	2	2		1	
2014	Germany[230]	2		2	2	2				1	2						1
2015	EU[231]	3	3	3	3	3	3	3		1	1			1			
2015	Denmark[232]	3	3	3	3	3	3	3	3	3	1	1		1	3	1	
2016	Sweden[233]	2	3	2	3	3	2	2	2	1		3	3	1		1	1
2016	Netherlands[234]	3	3				3				3						
Total Score		15	15	13	13	13	13	10	8	7	7	5	5	5	3	3	2

[228] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014
[229] Mepex (2016) *Primary microplastic- pollution: Measures and reduction potentials in Norway*, April 2016
[230] Roland Essel, and et al. (2014) *Sources of microplastics relevant to marine protection*, Report for Federal Environment Agency (Germany), November 2014
[231] Eunomia Research & Consulting (2016) *Study to support the development of measures to combat a range of marine litter sources*, Report for European Commission DG Environment, 2016
[232] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015
[233] Kerstin Magnusson, and et al. (2016) *Swedish sources and pathways for microplastics to the marine environment*, Report for Swedish Environmental Protection Agency, March 2016
[234] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

Table 37 – Results from European Microplastics Sources and Emissions Studies (grams per capita of microplastics produced from source)

■ Largest emission source(s) (Score = 3), ■ Second largest emission source(s) (Score = 2), ■ Third largest emission source(s) (Score = 1)

Year of study	Country	Population (M)	Automotive Tyres	Marine Paint	Pre-Production Plastics	Textiles	Personal Care Products	Building/ decorative Paint	Road Markings	Artificial Turf	Abrasives	Detergents	Fishing and Aquaculture	Indoor Dust	Waste and Recycling	Footwear
2014/16	Norway	5.2	864	140	86	134	8	77	61	288	-	-	-	86	78	-
2014	Germany	82.1	730-1,351	-	256-2,556	1-5	6	-	-	-	<1.22	<1.22	-	-	-	-
2015	EU	510	920	2-8	100	31-94	18-24	24-56	235							
2015	Denmark	5.7	736-1,051	7-84	1-9	35-175	2-5	0.4-1	18-105	79-277	0.1-0.44	-	-	-	-	18-175
2016	Sweden	9.8	1,370	49-138	31-54	18-203	6	13-25	-	233-396	-	-	0.4-4.7	0.09-1.7	-	-
2016	Netherlands[1]	5.2	1,019	12				29				0.15				
Total Score			18	5	6	6	2	3	4	6	-	-	-	1	1	1

1. The study from the Netherlands focuses on three emission sources as a result of a previous prioritisation exercise based upon the; 1) volume of the emission, 2) feasibility of measures and 3) action perspectives for consumers. Although textiles and cosmetics were also identified as a priority sources, textiles were being investigated by the EU 'Mermaids' project and cosmetics has already received attention and voluntary commitments

Table 38 shows the long list of microplastics sources that are either created during use or intentionally added. The knowledge status is that before this study was conducted.

Table 38 – Long List of Microplastics Sources

Category	Sub Category	Intentionally added/Created during use	Knowledge Status
Artificial Turf	Infill	Intentionally added	Although the 'infill' material is intentionally added and the plastic grass fibres would become microplastics through wear, it was felt that because Eunomia already have experience and contacts in this area that they can deliver this category.
	Fibres	Created during use	
Vehicles	Tyres	Created during use	Tyres is a well-known microplastics source as a result of wear. Tyre wear data is available, but the pathway data is less developed.
	Brake dust	Created during use	Brake dust has also been highlighted in the past although the exact composition of brake pads is not known at present.
Synthetic Textiles	Synthetic clothing	Created during use	Direct pathway to surface water identified and a great deal of research is being undertaken in this area
	Carpets	Created during use	This is less known with no direct pathways identified as yet
	Cleaning cloths	Created during use	Similar to clothing, but no research currently
	Hygiene products	Created during use	It is not known how much of these products contain plastic fibres
Pre-Production Plastics	Pellets/powders/flakes	Intentionally added	Although these could be considered intentionally added, again Eunomia have significant experience in this area and will address this category.
	3D printing	Intentionally added	Relates to the feedstock for 3D printing which can be in powder form but to what extent is not known currently.
	Plastics recycling	Created during use	Very little data exists on this, but it may be covered under the measures for pellets in terms of improving the handling to reduce loss.
Oil and gas	Drilling fluids	Intentionally added	A fairly un-researched source, but has been determined as intentionally added in the form of 'microbeads'.
Agriculture	Mulch films	Created during use	May be a significant source for terrestrial plastics, but the pathways to water are not well established. May also incorporate biodegradable/oxo-degradable plastics. Conventional plastics may be excluded due to being considered as mismanaged waste rather than
	Baling films	Created during use	
	Fertilisers (nutrient prills)	Intentionally added	Little is known of these other than they are a plastic coating that would be considered a microplastic both before and after the encased fertiliser has dissolved. This is therefore a microplastic by design and deemed intentionally added.
Paints/coatings	Building	Both	

	Road	Both	For most kinds of paint there are two states in which the paint could possibly become microplastic; at the solvent stage before setting and after the product is set and is either removed or abraded. In recognition of this, AMEC will focus on the first state and the chemical compositions and Eunomia will focus on the latter state.
	Marine (commercial and domestic)	Both	
	Anti-skid powder	Intentionally added	Microplastics are manufactured and intentionally added to paint to increase friction in yacht decks.
	Laser printer inks	Intentionally added	It is unclear presently what this particular source refers to other than the inference that printer toner may contain microplastics.
Personal Care Products	Rinse off	Intentionally added	This has received the most attention and industry data is available.
	Leave on	Intentionally added	This has received less focus and requires increased industry engagement to make progress.
Detergents	Commercial	Intentionally added	Identified but not fully investigated as intentionally added in detergent products. Likely to be mostly for commercial applications. Plastic beads for commercial Dishwashing ('Power granules') supposed to be used on ships. Supposed to be a closed system with no emissions.
	Domestic	Intentionally added	
Marine	Fishing gear	Created during use	Both of these sources are theoretical losses due to abrasion. Data is likely to be non-existent as there is very little data on the use of fishing gear and losses in general.
	Aquaculture	Created during use	
Industrial Abrasives	Abrasive media	Intentionally added	This has been previously identified, but not quantified and should be investigated further as an intentionally added source.
Others	Shoe sole wear	Created during use	Little is known about this source, but is due to wear and tear whilst walking
	Dentist polish	Intentionally added	This needs to be investigated further as an intentionally added source.
	Insulation	Intentionally added	Small polystyrene balls are often used as an insulation product in buildings. Their application may be controlled, but building demolition may be an issue.
	Furniture	Intentionally added	Small polystyrene balls are often used inside furnishings to provide a filling.
	Kitchen utensils	Created during use	Wear of kitchen utensils during use.
	Buildings	Created during use	The wear of plastic building materials (not paint) during use.
	Indoor dust	Created during use	A generic term that can include a number of different sources of microplastic from in the home. Some of this may already be included under 'carpets'.

A.3.0 Microplastic Source Quantification and Pathways Summary Data

Table 39 summarises the upper lower and midpoint estimates for emissions of microplastics to surface waters and includes figures from the parallel study on intentionally added microplastics for comparison.

Table 39 – Annual Microplastics Emissions to Surface Waters

Source	Upper (tonnes)	Midpoint	Lower (tonnes)	Source Data Year
Automotive Tyres	136,000	94,000	52,000	2012
Pellets	78,000	41,000	3,000	2015
Washing of Clothing	23,000	13,000	4,000	2016
Road Markings	21,000	15,000	10,000	2015
Building Paint	8,000	5,000	2,000	2013
Fishing Gear	5,000	2,600	500	2015
Automotive Brakes	5,000	2,000	100	2012
Artificial Turf	3,000	2,000	300	2012
Marine Paint	400	400	400	2013
Leave on PCP	526	-	86	-
Fertilisers	400	-	85	-
Rinse off PCP	373	-	114	-
Building Paint	141	-	0.40	-
Detergents	94	-	30	-
Total	**300,000**		**72,500**	

Note:

1. **Figures in orange are 'intentionally added' microplastics taken from the parallel study for comparison.**
2. Data for the calculation of emissions comes from different years for each emission source. The results are normalised to 2017 for the baseline calculations using the midpoint.
3. All Figures except for those from 'intentionally added' products (highlighted in orange) are rounded therefore totals may not add up

Table 40 summarises the results of the pathways analysis showing there are several ways for microplastics to get to surface waters. This is summarised by emissions from WWT plants (g), through storm water (j) or direct (D). Summing these (g + j + D) provides the midpoint emissions to surface waters in Table 39. Overall midpoint *source emissions* for each product are found by summing A + B + C—these are the totals shown in the graph of Figure 8 in the main report.

The results suggest that the most significant sink are soils directly adjacent to roads. Storm water management carries the largest amount of microplastics although, if automotive tyres are discounted the split between WWT and storm water becomes similar in magnitude. More detailed calculations and methodology are found in Appendix A.3.8.7.

Table 40 – Microplastics Pathways and Sinks (tonnes) (midpoint estimates)

Source	Entering Waste Water Treatment Plants A = g+h+i	Surface Water from WWT (g)	Local Residual Waste Manage-ment (h)	Agri-Soil (i)	Through Storm Water Manage-ment B = j+k	Surface Water from Storm Water (j)	Storm Water Sediment-ation e.g. gully pots (k)	Direct to Soil (C)	Direct to Surface Water (D)	Road Cleaning Waste Manage-ment (E)	Direct to Marine Environ-ment (F)	Total Source Emissions (A+B+C)
Automotive Tyres	20,034	8,346	5,907	5,780	163,440	49,349	114,091	269,153	36,661	14,298		503,586
Road Markings	1,989	828	586	574	11,991	3,205	8,785	67,906	11,408	1,064		94,358
Pre-Production Plastics	21,324	9,857	5,796	5,671	66,147	25,928	40,219		4,604		183	92,259
Washing of Clothing	30,919	11,763	9,683	9,474					1,627			32,546
Building Paint	1,241	539	354	347	5,510	1,596	3,914	17,593	3,131	508		27,984
Artificial Turf	2,632	1,091	779	762	1,680	590	1,091	20,280	227			24,819
Automotive Brakes	456				2,326			4,910	923	218		8,833
Fishing Gear											2,629	2,629
Marine Paint											411	411
Total	78,595	32,637	23,229	22,728	251,094	81,811	169,284	379,842	58,582	16,089	3,224	787,425

A.3.1 Synthetic Textiles

A.3.1.1 Clothing

It is now widely recognised that the washing of synthetic textiles is one of the major sources of microplastic pollution. Synthetic textiles include materials such as polyester, acrylic and polyethene and are found in a range of clothes. When washed, the abrasion of these textiles causes microplastics to be released in the form of small synthetic fibres, known as microfibres. These are discharged into washing machine effluent and, through wastewater, can make their way into the aquatic environment.

Understanding Sources and Pathways

The idea that clothing is contributing to microplastic pollution was first presented, all be it very briefly, in 2004. Thompson et al's[235] research focused on microplastic pollution in the sediment of UK beaches, finding synthetic polymers in most (76.6%) of the sediment samples taken. The report suggested that the breakdown of clothing could be a potential source but did not go into more detail.

It wasn't until Browne et al's 2011 study[236] was published that the link between the washing of clothes and microfibre contamination was recognised. When sampling sediment from shorelines worldwide, Browne's study found high concentrations of microfibres near densely populated areas and at sewage disposal sites. Recognising that the microfibres found were largely made of the materials used in clothing, such as acrylic and nylon, Browne et al. proposed that an important source could be the washing of clothes. Through direct sampling of the effluent from front-load domestic washing machines after washing synthetic textiles, Browne et al. found evidence for this theory and states that a single garment can release >1,900 fibres per wash. The study details that the items washed in the experiments were polyester blankets, fleeces and shirts, though the specific material makeup of the textiles is not given in detail.

The recognition of this pathway for microfibres set the course for a range of studies, the majority of which either rely on Browne's data, or at least reference the study's findings. Dubaish and Leibezeit[237] focused on microplastic pollution in the North Sea. The study found that through washing a polyester garment, between 0.033–0.039 % of the garments weight was lost through microfibre release. Although this is potentially a useful figure for comparison, the paper does not give any details of the experimental conditions. Details such

---

[235] Richard Thompson, Ylva Olsen, Richard P. Mitchell, et al. (2004) *Lost at Sea: Where Is All The Plastic?*, Report for Brevia, June 2004

[236] Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., and Thompson, R. (2011) Accumulation of microplastic on shorelines worldwide: sources and sinks, *Environmental Science & Technology*, Vol.45, No.21, pp.9175–9179

[237] Dubaish, F., and Liebezeit, G. (2013) Suspended Microplastics and Black Carbon Particles in the Jade System, Southern North Sea, *Water, Air, & Soil Pollution*, Vol.224, No.2, pp.1–8

as the washing machine type used or the composition of the garment washed would have been valuable, as these are likely to have impacted the study's findings.

More recent studies have investigated the effects of additional factors such as fabric type or aging on the rate of microfibre release. Folkö[238] used a front-loading washing machine to wash a fleece shirt (100% polyester) and a sports sweater (57% polyamide and 43% polyester) four times each, finding that they lost a total of 0.46% and 0.1% of their initial mass respectively. These percentage losses are significantly higher than those found by Dubaish and Leibezeit[239], but without the methodology of the latter, it is hard to determine why. Folkö's study found that fibre mass released decreased with each consecutive wash: the fleece lost 64% of the total emitted fibre mass during the two first washing cycles and the sweater lost 75% during the first washing cycle. Through washing the different garments, the findings enable a comparison of fabric types, demonstrating a higher release from the 100% polyester garment, rather than that which was a blend of materials.

Pirc et al. [240] also investigated the influence of consecutive washing, a useful comparison with Folkö's study[241]. The research found that release started at 189.5 mg per kg of material washed and then stabilised at 12 mg per kg. This decrease in microfibre release with consecutive washing is the same correlation as found by Folkö[242] but is a much lower release. This could be due to Pirc washing a blanket rather than a garment, as the latter is likely to have more joins where fibres are likely to be loose. The study also had a short 15 minute wash at a low temperature, factors which the Mermaids project[243] found are likely to significantly reduce the rate of microfibre release.

Petersson and Roslund's 2015 study[244] found that a worn material resulted in more shedding. Although the details of the mechanical aging process are unclear in the report and raw data is not given, this correlation does, however, fall in line with that found by Hartline et al[245] who researched fabric aging in more detail. To age garments the study used what they call an 'industry testing procedure' and washed the garments in a top-loading commercial heavy-duty washer through a 24 hour no spin wash cycle, then dried them in a home-style dryer and, finally, dusted them to remove loose fibres. The study found that, on

---

[238] Amanda Folkö (2015) *Quantification and characterization of fibers emitted from common synthetic materials during washing*, Report for Käppala, 2015

[239] Dubaish, F., and Liebezeit, G. (2013) Suspended Microplastics and Black Carbon Particles in the Jade System, Southern North Sea, *Water, Air, & Soil Pollution*, Vol.224, No.2, pp.1–8

[240] U. Pirc, M. Vidmar, A. Mozer, and A. Kržan (2016) Emissions of microplastic fibers from microfiber fleece during domestic washing, *Environ Sci Pollut Res*

[241] Amanda Folkö (2015) *Quantification and characterization of fibers emitted from common synthetic materials during washing*, Report for Käppala, 2015

[242] Amanda Folkö (2015) *Quantification and characterization of fibers emitted from common synthetic materials during washing*, Report for Käppala, 2015

[243] Mermaids (2017) *Report on localization and estimation of laundry microplastics sources and on micro and nanoplastics present in washing wastewater effluents. Deliverable A1.*, May 2017

[244] Hanna Petersson, and Sofia Roslund (2015) *A survey of polyester fiber emission from household washing*, June 2015

[245] Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., and Holden, P.A. (2016) Microfiber Masses Recovered from Conventional Machine Washing of New or Aged Garments, *Environmental Science & Technology*, Vol.50, No.21, pp.11532–11538

average, mechanically aged garments released 25% more fibres. They expect that this was mainly caused by the fraying that was visible on the aged garments.

Comparing these studies, it can be hypothesised that a garment will initially loose a high volume of fibres which are loose from manufacturing, as demonstrated in Folkö's[246] study. This amount will decrease over successive washes until the garment becomes 'worn', which is the point at which the fabric is starting to become damaged and frayed, leading to an increased amount of microfibre release with the increased 'wear and tear' of the product. This is demonstrated in Hartline[247] and Petersson and Roslund's[248] studies, where efforts were made to 'age' the garments, rather than just washing them successively.

Through reading recent literature, it is clear that the type of fabric being washed significantly influences the rate of microfibre release, though findings tend to differ between studies. Napper and Thompson[249] washed three different types of fabrics: 65% polyester/35% cotton blend, 100% acrylic and 100% polyester. The study washed a 20cm by 20cm square from the selected garments in a front-load machine. After recording the releases, the study estimated that through washing 6kg of material a polyester/cotton blend would release significantly fewer fibres (137,951) than acrylic (728,789) or polyester (496,030). The finding that pure polyester released the most microfibres is consistent with Folkö's paper[250], previously discussed. Napper and Thompson's experiments used varying combinations of softener and conditioner but found no statistically significant correlations between these variables and the number of fibres released.

Petersson and Roslund[251] also stressed the importance of fabric construction, stating that we should take care to avoid washing materials which have a combination of a high pitch, high gauge and worn material, to limit microplastic release.

An additional factor which is touched on by several studies is the influence of washing machine type on the rate of microfibre release. Hartline[252] found a 7x greater release of fibres from top-load machines as opposed to front-load. The study theorised that this higher release was likely due to the central agitator of the top-load washing machine which moves clothes particularly vigorously through the water. It is likely that this more rigorous washing

---

[246] Amanda Folkö (2015) *Quantification and characterization of fibers emitted from common synthetic materials during washing*, Report for Käppala, 2015

[247] Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., and Holden, P.A. (2016) Microfiber Masses Recovered from Conventional Machine Washing of New or Aged Garments, *Environmental Science & Technology*, Vol.50, No.21, pp.11532–11538

[248] Hanna Petersson, and Sofia Roslund (2015) *A survey of polyester fiber emission from household washing*, June 2015

[249] Napper, I.E., and Thompson, R.C. (2016) Release of synthetic microplastic plastic fibres from domestic washing machines: Effects of fabric type and washing conditions, *Marine Pollution Bulletin*, Vol.112, Nos.1–2, pp.39–45

[250] Amanda Folkö (2015) *Quantification and characterization of fibers emitted from common synthetic materials during washing*, Report for Käppala, 2015

[251] Hanna Petersson, and Sofia Roslund (2015) *A survey of polyester fiber emission from household washing*, June 2015

[252] Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., and Holden, P.A. (2016) Microfiber Masses Recovered from Conventional Machine Washing of New or Aged Garments, *Environmental Science & Technology*, Vol.50, No.21, pp.11532–11538

caused greater abrasion of the textiles and thus greater microfibre release. Bruce[253] also found higher emissions of microfibres from top-load machines, though he estimates that the extent of this difference was around 430%, a significantly higher figure than Hartline.

As is clear from the preceding literature, studies that have quantified microfibre release demonstrate different methodologies and different metrics when presenting their findings. Some, for example, give findings in the number of fibres released per washing of a garment [254] whereas others may define the release in terms of mg released per garment washing [255] [256] [257]. This difference is likely due to microfibre release being a new area of research and the methodology not yet being standardised.

Table 41 summarises the studies that have attempted to simulate and quantify fibre release from clothing during washing. Several studies[258,259] have been omitted due to their lack of methodological detail. This includes Browne's[260] early work into the subject that arguably triggered much of the subsequent research, but was designed purely to test the theory rather than to provide definitive quantification. Many estimates have been based on his work, but more recent studies provide a more robust methodology.

However, many studies were also simply designed to prove the occurrence of fibre release and make steps towards identifying which factors affect this. It is essential to recognise that none of these studies were specifically designed to be scaled up the national or international estimates of fibre release, nor were they designed to become a standardised test. Earlier tests also provided very little information on the construction of the fabrics tested. Whilst this is not an issue when attempting to ascertain whether fibres are released, it does not help to determine whether there are aspects of the construction that mitigate or aggravate release. Certainly, many of these tests were also designed in isolation in an emerging scientific field—a key future step is therefore to begin to harmonise these methodologies.

---

[253] Nicolas Bruce, and et al. (2016) *Microfibre Pollution and the apparel industry*, Report for Patagonia, June 2016

[254] Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., and Thompson, R. (2011) Accumulation of microplastic on shorelines worldwide: sources and sinks, *Environmental Science & Technology*, Vol.45, No.21, pp.9175–9179

[255] Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., and Holden, P.A. (2016) Microfiber Masses Recovered from Conventional Machine Washing of New or Aged Garments, *Environmental Science & Technology*, Vol.50, No.21, pp.11532–11538

[256] U. Pirc, M. Vidmar, A. Mozer, and A. Kržan (2016) Emissions of microplastic fibers from microfiber fleece during domestic washing, *Environ Sci Pollut Res*

[257] Dubaish, F., and Liebezeit, G. (2013) Suspended Microplastics and Black Carbon Particles in the Jade System, Southern North Sea, *Water, Air, & Soil Pollution*, Vol.224, No.2, pp.1–8

[258] Dubaish, F., and Liebezeit, G. (2013) Suspended Microplastics and Black Carbon Particles in the Jade System, Southern North Sea, *Water, Air, & Soil Pollution*, Vol.224, No.2, pp.1–8

[259] Dris, R., Gasperi, J., Saad, M., Mirande, C., and Tassin, B. (2016) Synthetic fibers in atmospheric fallout: A source of microplastics in the environment?, *Marine Pollution Bulletin*, Vol.104, Nos.1–2, pp.290–293

[260] Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., and Thompson, R. (2011) Accumulation of microplastic on shorelines worldwide: sources and sinks, *Environmental Science & Technology*, Vol.45, No.21, pp.9175–9179

There are still a number of methodological differences between the studies which are also identified in Table 41 (highlighted in red and yellow text). Some may have a large influence and/or not necessarily reflect common washing practices and others are consistently highlighted as highly influential for fibre release;

- Possible Influences of fibre release;
  - High spin speeds (~1,400 rpm)
  - Low spins speeds (~600rpm)
  - Short washing duration (<30 minutes)
  - High Washing temperatures
- Known large influences of fibre release;
  - Lack of detergent use (water only)
  - Large mesh filter sizes (>50 µm)

In a written statement, the Man Made Fibre Association dismissed all of the current research by citing the flaws in each methodology. Whilst these flaws are, of course, important to recognise, the statement also proposes that Pirc et al. should be considered the best method so far. This study found the lowest fibre release, however it has issues of its own (the lack of detergent use, for example) and it is not particularly wide ranging in the types of material studied.

Indeed, it's findings were consistent with those of De Falco et al[261]. Both studies found polyester fibre release of 12mg/kg washed without detergent, however De Falco et al. found fibre release to be up to 30 times higher when used with powdered detergent. Pirc et al. did not test with detergent which is unlikely to reflect consumer behaviour. There is therefore no specific reason that its findings should be regarded as more significant than others.

There appears to be universal agreement amongst studies that looked into the issue that detergents increase fibre release. The exact reason for this is unknown, but it is speculated that the surfactants helps to lubricate already released fibres. The release figures used for the EU estimates are based on studies that have use detergents as this is intuitively more likely to be the case.

Not enough is known about the exact mechanisms of fibre release that previous work can be dismissed in its entirety especially when there is a well-documented methodology and assumptions. It is clear that very large number are released however there is uncertainty about the absolute numbers. For these reasons the range that is calculated in this report represents an analysis of the most likely range of emission that the current evidence suggests.

---

[261] De Falco, F., Gullo, M.P., Gentile, G., et al. (2017) Evaluation of microplastic release caused by textile washing processes of synthetic fabrics, *Environmental Pollution*

Table 41 – Method Comparison of Clothes Washing Studies

Red highlights = methodological issue, Yellow highlights = unconfirmed methodological issue

Study	Publisher	Samples	Washing device	Washing Conditions	Filter	Count/weighing
Napper & Thompson [262]	Journal Article	20x20 cm cut and sewn. PES and Acrylic	Domestic Front Loading Washing Machine	1hr 15min 1400rpm	25 µm	Weighed fibres using precision balance
Pirc [263]	Journal Article	PES fleece blankets (12 x 70) 320g	Domestic Front Loading Washing Machine	15 min 30°C 600rpm no detergent	200 µm	Weighed fibres
Hartline (Patagonia) [264]	Journal Article (industry funded)	5 fleece jackets	Domestic Front Loading Washing Machine (also top loading, but not relevant for EU)	29 to 41°C warm cycle, 1200rpm	333 and 20 µm	Calculated mass of filter by photographing
Folko [265]	Masters Project	PES Fleece shirt and 'sports sweater' PA and PES	Domestic Front Loading Washing Machine	30min 40°c 1400 rpm no detergent	20 µm	Weighed filters

[262] Napper, I.E., and Thompson, R.C. (2016) Release of synthetic microplastic plastic fibres from domestic washing machines: Effects of fabric type and washing conditions, *Marine Pollution Bulletin*, Vol.112, Nos.1–2, pp.39–45

[263] U. Pirc, M. Vidmar, A. Mozer, and A. Kržan (2016) Emissions of microplastic fibers from microfiber fleece during domestic washing, *Environ Sci Pollut Res*

[264] Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., and Holden, P.A. (2016) Microfiber Masses Recovered from Conventional Machine Washing of New or Aged Garments, *Environmental Science & Technology*, Vol.50, No.21, pp.11532–11538

[265] Amanda Folkö (2015) *Quantification and characterization of fibers emitted from common synthetic materials during washing*, Report for Käppala, 2015

Mermaids (report B4) [266]	EU Funded Project	Woven polyester (sample not described)	Domestic Front Loading Washing Machine	30°c 400prm 60°c 1400rpm	43-48 µm	Filter dried and weighed
Hernandez[267]	Journal Article	Polyester jersey fabric cut 30 x 10cm 'tailored' pre-washed	Laboratory machine	40°c 45 min 10 stainless steel balls	0.45 µm	Fibre mass calculated from size/number
Roos[268]	Swedish Research Project	fleece fabric from recycled polyester and woven fabric for outer layer of jacket Cut or ultrasonically welded edges.	Laboratory machine	40°c 60 min cycle with and without detergent 25 metal balls	100 µm, 5 µm and 0.65 µm	Automatic fibre identification software
Aström[269]	Masters Project	Cut with laser cutter 10x10 cm Prewashed	Laboratory machine	30 min 60°c 25 metal balls	1.2 µm	Fibres counted through microscope

[266] Mermaids (2017) *Report on localization and estimation of laundry microplastics sources and on micro and nanoplastics present in washing wastewater effluents. Deliverable A1.*, May 2017

[267] Hernandez, E., Nowack, B., and Mitrano, D.M. (2017) Polyester Textiles as a Source of Microplastics from Households: A Mechanistic Study to Understand Microfiber Release During Washing, *Environmental Science & Technology*, Vol.51, No.12, pp.7036–7046

[268] Sandra Roos, Oscar Levenstam Arturin, and Anne-Charlotte Hanning (2017) *Microplastics shedding from polyester fabrics*, Report for Mistra Future Fashion, 2017

[269] Linn Åström (2016) *Shedding of synthetic microfibers from textiles*, dissertation submitted at Göteborg University, Institute for Biology and Environment, January 2016

De Falco et al[270] (Mermaids)	Journal Article (EU Funded Project)	Woven polyester Cut to 9 x9.3 cm Cotton sewn edges	Laboratory machine	30°c 400prm 60°c 1400rpm 45—90 min 45°C and 60°C degree cycles 10 steel balls	5 µm	Filter dried and weighed

[270] De Falco, F., Gullo, M.P., Gentile, G., et al. (2017) Evaluation of microplastic release caused by textile washing processes of synthetic fabrics, *Environmental Pollution*

Upscaling the Research

With an increasing awareness of the issue of microplastics, there have been several attempts to upscale the research to calculate microfibre release from the washing of clothes on a national, European and even global level.

The first study to attempt this was a report for the Norwegian Environment Agency (2014) by Mepex [271] which estimated release on a national level. The report calculated that the microfibre release from the washing of textiles in Norway was 0.12kg microfibres, per capita, per year, into washing machine effluent. The calculations throughout this study were based on Browne [272] and Dubaish and Liebezeit's [273] original estimations of fibre release, and upscaled using population statistics. To convert Browne's calculations of number of fibres into weight of fibres, the study used 'decitex' which presumed that the length to weight ratio for polyester and nylon fibres is about 300grams/ 10,000 meters.

Essel et al. (2015) [274] also reference Browne et al's[275] estimations of fibre release. The study uses different assumptions, however, to make its calculations, and presumes that everyone in the population owned one fleece which decreased in weight by 1% to 5% through washing. The calculations in the study estimated the release of microfibres from Germany to be between 80–400 tonnes per year and Europe to be between 500 and 2,500 tonnes per year.

Lassen et al. (2015) [276] calculated the release of microfibres from textiles in Denmark. Their workings were based on Browne [277] and Dubaish and Leibezeit's [278] calculations of release and used the assumption that 50% of all textiles being washed in the country were synthetic. The paper proposed that the annual release from the countries washing of textiles is between 200–1,000 tonnes per year.

---

[271] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

[272] Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., and Thompson, R. (2011) Accumulation of microplastic on shorelines worldwide: sources and sinks, *Environmental Science & Technology*, Vol.45, No.21, pp.9175–9179

[273] Dubaish, F., and Liebezeit, G. (2013) Suspended Microplastics and Black Carbon Particles in the Jade System, Southern North Sea, *Water, Air, & Soil Pollution*, Vol.224, No.2, pp.1–8

[274] Roland Essel, and et al. (2014) *Sources of microplastics relevant to marine protection*, Report for Federal Environment Agency (Germany), November 2014

[275] Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., and Thompson, R. (2011) Accumulation of microplastic on shorelines worldwide: sources and sinks, *Environmental Science & Technology*, Vol.45, No.21, pp.9175–9179

[276] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

[277] Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., and Thompson, R. (2011) Accumulation of microplastic on shorelines worldwide: sources and sinks, *Environmental Science & Technology*, Vol.45, No.21, pp.9175–9179

[278] Dubaish, F., and Liebezeit, G. (2013) Suspended Microplastics and Black Carbon Particles in the Jade System, Southern North Sea, *Water, Air, & Soil Pollution*, Vol.224, No.2, pp.1–8

Magnusson et al. (2016) [279] also used previous research to quantify microfibre release, but this time for Sweden. The study used estimations of release from Browne et al[280], Folkö[281] and Dubaish and Leibezeit[282], converting them into the same metric to calculate an upper and lower range of microfibre release per load of washing. The study estimated that the total annual synthetic fibre discharge was between 195 and 2,216 tonnes per year. When this study was revised in 2017[283] the calculation was updated to include Napper and Thompson's data[284].

In a report for the European Commission by Eunomia (2016) [285] the EU's microfibre release from textiles was estimated to be between 1,747 and 52,400 tonnes per year. This study used Browne's estimations as a basis, converting his calculated number of fibres released into weight, using the same dtex as the earlier Mepex study[286]. This presumed that every 10,000m of fibre would weigh 300g.

A recently published report for the International Union for Conservation of Nature (IUCN)[287] was the first to quantify microplastic release on a global scale, using two approaches to calculate release. The first approach uses the estimated number of washes per country and calculates per country release into washing machine effluent. The second approach combines yearly textiles sales data with typical losses over the lifetime of a synthetic textile cloth. For the first approach, the estimation is split into an optimistic 300mg and pessimistic 1500mg loss per kg wash. For the second approach, the estimation is given as optimistic and pessimistic scenarios: corresponding to 0.74% and 5% microfibre loss over a lifetime (based on Essel et al's, 2015 [288] paper). The report used an average of the central scenarios in both approaches for its calculations. The final estimate is that synthetic textiles account for 34.8% of the global release of microplastics into the oceans.

The limitation of these upscaling reports is that they tend to be based on just one or two smaller studies and rely on singular assumptions to scale beyond the scope of these studies. The consequences of variables such as washing conditions, fabric type and fabric aging are complex and not yet fully understood and these uncertainties make it difficult to accurately quantify microfibre release from the washing of clothes on a large scale.

---

279 Kerstin Magnusson, and et al. (2016) *Swedish sources and pathways for microplastics to the marine environment*, Report for Swedish Environmental Protection Agency, March 2016

280 Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., and Thompson, R. (2011) Accumulation of microplastic on shorelines worldwide: sources and sinks, *Environmental Science & Technology*, Vol.45, No.21, pp.9175–9179

281 Amanda Folkö (2015) *Quantification and characterization of fibers emitted from common synthetic materials during washing*, Report for Käppala, 2015

282 Dubaish, F., and Liebezeit, G. (2013) Suspended Microplastics and Black Carbon Particles in the Jade System, Southern North Sea, *Water, Air, & Soil Pollution*, Vol.224, No.2, pp.1–8

283 Kerstin Magnusson, and et al (2017) *Swedish sources and pathways for microplastics to the marine environment*, March 2017

284 Napper, I.E., and Thompson, R.C. (2016) Release of synthetic microplastic plastic fibres from domestic washing machines: Effects of fabric type and washing conditions, *Marine Pollution Bulletin*, Vol.112, Nos.1–2, pp.39–45

285 Eunomia Research & Consulting (2016) *Study to support the development of measures to combat a range of marine litter sources*, Report for European Commission DG Environment, 2016

286 Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

287 IUCN (2017) *Primary microplastics in the oceans.pdf*, 2017

288 Roland Essel, and et al. (2015) *Sources of microplastics relevant to marine protection*, Report for Federal Environment Agency (Germany), 2015

Mermaids[289] is the most recently published report attempting to estimate the release of microfibres on a large scale. Making for a useful comparison with Eunomia's report, they estimate this release at a Europe-wide level. Mermaids worked on the assumption that a 100% polyester load would release 3.9g of microplastics. Assuming that 90% of households in the EU own washing machines, and on average, carry out a wash 3.2 times a week, the total emissions from Europe were calculated at 29,215 tonnes per year. This value is similar to the final figures calculated by Eunomia.

Table 42 shows the final estimations made by all the upscaling studies mentioned.

Table 42 - Comparison of Studies

Study	Geography	Total emissions calculated per year (tonnes)	Per Capita per Year
Mepex (2014)[290]	Norway	276 to 315 (0.12 kg per capita)	0.12 kg
Essel et al. (2015)[291]	Germany Europe	Germany: 80 to 400 Europe: 500 to 350	0.005—0.001 kg
Magnusson et al. (2015)[292]	Sweden	195 to 2,216	0.02—0.225 kg
Magnusson et al, revised (2017) [293]	Sweden	8 to 945	0.001—0.096 kg
Lassen et al. (2015)[294]	Denmark	200 to 1000	0.035—0.175 kg
Eunomia (2016)[295]	EU	15,800 to 47,600	0.03—0.091 kg
Mermaids (2017)[296]	Europe	29,215	0.05 kg

[289] Mermaids (2017) *Mitigation of microplastics impact caused by textile washing processes. C2 report.*, 2017
[290] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014
[291] Roland Essel, and et al. (2014) *Sources of microplastics relevant to marine protection*, Report for Federal Environment Agency (Germany), November 2014
[292] Kerstin Magnusson, and et al. (2016) *Swedish sources and pathways for microplastics to the marine environment*, Report for Swedish Environmental Protection Agency, March 2016
[293] Kerstin Magnusson, and et al (2017) *Swedish sources and pathways for microplastics to the marine environment*, March 2017
[294] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015
[295] Eunomia Research & Consulting (2016) *Study to support the development of measures to combat a range of marine litter sources*, Report for European Commission DG Environment, 2016
[296] Mermaids (2017) *Mitigation of microplastics impact caused by textile washing processes. C2 report.*, 2017

A.3.1.2 Calculation Methodology (supplemental information)

Composition of the Average Washing Machine Load

Data from a JRC report into the environmental improvement potential of textiles[297] splits EU sales of textiles by clothing type, fabric type and fabric construction. For the purpose of the calculations it is assumed that this percentage split of fabric types consumed represents the makeup of an average washing machine load in the EU. Applying this assumption to the data does not factor in the different washing frequencies of different garments, however it is assumed that on an EU level the different textiles from sales will broadly be indicative of the composition of a washing load. This is an improvement over some previous estimates which use the global fibres composition; this includes items such as carpets which often use different fibre types. Using a composition for European clothing fibres is likely to be more accurate.

Using the JRC's data it was estimated that the average washing load was composed of 45% man-made fabric types. This estimation is consistent with other research[298,299] though slightly on the conservative side.

The release of fibres from the washing of viscose have been included in overall microplastics estimations, and as shown in Table 43 these fibres account for 24% of 'man-made' fibre clothing sales—second only to polyester with 38%—and therefore may be a significant source of microplastics if viscose is categorised as such. Following this, it is assumed that an average wash load is made of 45% man made fabrics, 34% is fully synthetic.

Table 43 - Fabric Sales by Weight in the EU

Fabric Type	Percentage of EU fabric sales by weight		
	Woven	Knitted	Total
Wool	2.9%	5.1%	8%
Cotton	17.5%	29.1%	47%
Silk	0.04%	0.003%	0%
Flax	0.3%	0.1%	0%
Viscose	3.7%	7.2%	11%
Polyamide (nylon)	1.5%	5.3%	7%
Acrylic	0.3%	8.4%	9%
Polypropylene	0.2%	1.2%	1%
Polyester	8.4%	8.6%	17%

Source: *JRC (2014)*

[297] JRC (2014) *Environmental Improvement Potential of Textiles (IMPRO-Textiles)*, Report for European Commission, January 2014

[298] IUCN (2017) *Primary microplastics in the oceans.pdf*, 2017

[299] Food and Agriculture Organisation of the United Nations (2013) *World Apparel Fiber Consumption Survey*, July 2013

Calculation of Synthetic Fibre Release

The raw data from De Falco et al. is shown in Table 44. As this data does not include estimates of releases for all fibre types these have been derived in Table 45 based on the following hypotheses;

- The ratio between knitted and woven fibre release for polyester holds true for the other fibres; and,
- The release rate of non-polyester fibres is similarly influenced by detergent type.

Table 44 – Fibre Release results Summary of Experiments Carried out by De Falco et al. (2017)

Material	Density (g/cm^3)	Length µm	diameter µm	Fibres Released/kg			Mg Fibres Released/kg		
				Water	Liquid Detergent	Powder Detergent	Water	Liquid Detergent	Powder Detergent
PEP (knitted polyester)	1.4	478	20	60,000	1,138,000	1,920,000	13	235	399
PEC (woven polyester)	1.4	340	14	162,000	1,273,000	3,538,000	12	92	255
Polypropylene	0.9	339	19	172,000	640,000	1,676,000	17	57	146

Table 45 – Calculating Fibre Release Rates for Each Material

☐ = Results Taken from De Falco et al. (2017), ☐ = Results extrapolated from De Falco et al. (2017)

Material		Fibre/kg	Density[2]	Length µm[3]	diameter µm[3]	xsectional area	Fibre weight	mg fibre/kg
Viscose	Woven	2,405,500	1.5	500	15	0.00018	0.00013	319
	Knitted	1,529,000	1.5	500	15	0.00018	0.00013	203
Polyamide	Woven	2,405,500	1.1	500	15	0.00018	0.00010	242
	Knitted	1,529,000	1.1	500	15	0.00018	0.00010	154
Acrylic	Woven	2,405,500	1.2	500	15	0.00018	0.00010	251
	Knitted	1,529,000	1.2	500	15	0.00018	0.00010	159
Polypropylene	Woven	1,158,000[1]	0.9	339	19	0.00028	0.00009	100
	Knitted	736,056	0.9	339	19	0.00028	0.00009	64
Polyester	Woven	2,405,500$_1$	1.4	340	14	0.00015	0.00007	174
	Knitted	1,529,000[1]	1.4	478	20	0.00031	0.00021	317

Notes:

1. **Taken from De Falco et al. (2017): Average of washing with liquid and powered detergent**

http://www.minifibers.com/documents/Choosing-the-Proper-Short-Cut-Fiber.pdf

A comprehensive size distribution by Hernandez[300] shows that the vast majority of fibres lie between 50 and 1000 µm with a roughly even distribution throughout these sizes. The figure of 500 µm is therefore used for fibres where no data exists. This is also consistent in magnitude with evidence submitted by the Man Made Fibres Association whose own conversions used 1000 µm, however this may also be on the high side when considering the body of research available.

[300] Hernandez, E., Nowack, B., and Mitrano, D.M. (2017) Polyester Textiles as a Source of Microplastics from Households: A Mechanistic Study to Understand Microfiber Release During Washing, *Environmental Science & Technology*, Vol.51, No.12, pp.7036–7046

Data Gaps

Due to the complexities surrounding the release of microfibres from the washing of textiles and a lack of research in certain areas, there are several factors that have not been incorporated into the calculations.

Washing temperature is known to influence microfibre release, with high temperatures damaging the structure of fabrics casing greater release[301]. AISE[302] states that the average washing temperature for Europe is 40.9°C whilst Mermaids[303] used the assumption that the average washing temperature for Europe is 42.6°C. There is currently a lack of data on this subject so it has not been focused on in the calculations.

Further research is also needed on the type of washing machines owned in Europe. The calculations are based on the release of microfibres from front-load machines as it seems likely that this type will be owned by the majority of consumers. There will, however, be a proportion of consumers who own top-loading machines, from which the rate of microfibre release is known to be significantly greater[304,305].

The effects of softener and detergent have been touched on by various reports[306,307,308]. The effect could be significant with up to 30 times more release between water only and detergent and 2—3 times more between liquid and powdered detergent[309]

The effect of fabric aging is also a subject area which requires further research. Hartline [310] found that when washing textiles, 'mechanical' aging increased the release of microfibres by 25%. The influence of aging could have a significant increase on rates of release but further research is needed on different fabric types and their rates of wear before this can be factored into estimations.

[301] Mermaids (2017) *Report on localization and estimation of laundry microplastics sources and on micro and nanoplastics present in washing wastewater effluents. Deliverable A1.*, May 2017

[302] AISE (International Association for Soaps, Detergents and Maintenance Products) (2014) AISE Consumers Habits Survey Summary

[303] Mermaids (2017) *Mitigation of microplastics impact caused by textile washing processes. C2 report.*, 2017

[304] Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., and Holden, P.A. (2016) Microfiber Masses Recovered from Conventional Machine Washing of New or Aged Garments, *Environmental Science & Technology*, Vol.50, No.21, pp.11532–11538

[305] Nicolas Bruce, and et al. (2016) *Microfibre Pollution and the apparel industry*, Report for Patagonia, June 2016

[306] Napper, I.E., and Thompson, R.C. (2016) Release of synthetic microplastic plastic fibres from domestic washing machines: Effects of fabric type and washing conditions, *Marine Pollution Bulletin*, Vol.112, Nos.1–2, pp.39–45

[307] Mermaids (2017) *Report on localization and estimation of laundry microplastics sources and on micro and nanoplastics present in washing wastewater effluents. Deliverable A1.*, May 2017

[308] U. Pirc, M. Vidmar, A. Mozer, and A. Kržan (2016) Emissions of microplastic fibers from microfiber fleece during domestic washing, *Environ Sci Pollut Res*

[309] De Falco, F., Gullo, M.P., Gentile, G., et al. (2017) Evaluation of microplastic release caused by textile washing processes of synthetic fabrics, *Environmental Pollution*

[310] Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., and Holden, P.A. (2016) Microfiber Masses Recovered from Conventional Machine Washing of New or Aged Garments, *Environmental Science & Technology*, Vol.50, No.21, pp.11532–11538

EU Washing Habits

Table 46 - Total Number of Washes in the EU (EU 28 + Norway and Switzerland)

Country	Wash cycles per year[311]	Households[312]	Total washes / year
Austria	164	3,864,000	633,696,000
Belgium	165	4,692,600	774,279,000
Bulgaria	165	2,742,800	452,562,000
Croatia	177	1,480,800	262,101,600
Cyprus	177	315,100	55,772,700
Czech Republic	165	4,690,400	773,916,000
Denmark	165	2,387,300	393,904,500
Estonia	165	573,400	94,611,000
Finland	165	2,640,500	435,682,500
France	165	29,138,500	4,807,852,500
Germany	164	40,536,500	6,647,986,000
Greece	177	4,410,700	780,693,900
Hungary	165	4,148,600	684,519,000
Ireland	177	1,728,300	305,909,100
Italy	165	25,797,200	4,256,538,000
Latvia	177	835,700	147,918,900
Lithuania	165	1,391,600	229,614,000
Luxembourg	165	233,600	38,544,000
Malta	177	151,200	26,762,400
Netherlands	165	7,722,600	1,274,229,000
Poland	177	14,225,100	2,517,842,700
Portugal	177	4,080,200	722,195,400
Romania	177	7,470,000	1,322,190,000
Slovakia	177	1,845,600	326,671,200
Slovenia	177	889,200	157,388,400
Spain	165	18,444,200	3,043,293,000
Sweden	140	4,825,000	675,500,000
United Kingdom	165	28,646,900	4,726,738,500
Norway	165	*2,316,647*[313]	*382,246,755*
Switzerland	165	*3,576,648*[314]	*590,146,920*
		Total EU washes per year	**37,541,304,975**

[311] Christiane Pakula, and Rainer Stamminger (2010) Electricity and water consumption for laundry washing by washing machine worldwide, *Energy Efficiency*, Vol.3, No.4, pp.365–382

[312] Eurostat *Eurostat - Data Explorer. Number of private households by household composition, number of children and age of youngest child*, accessed 7 June 2017, http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=lfst_hhnhtych&lang=en

[313] UNECE *Private households by Household Type, Measurement, Country and Year*, accessed 7 June 2017, http://w3.unece.org/PXWeb2015PXWeb2015/pxweb/en/STAT/STAT__30-GE__02-Families_households/08_en_GEFHPrivHouse_r.px/

[314] ibid

Cleaning Cloths

The revised calculation for cleaning cloths shows that the microplastic release is around 90% lower than previous estimates therefore it has been removed from the main report as it is no longer considered a key source.

Cloth sales in Europe have been obtained from a JRC report [315] looking into the improvement potential of textiles (the same dataset used in the clothing estimates in Section 2.2.4.1). Lassen et al. suggest that cloths usually come in two types: a polypropylene/viscose mix and polyester. The JRC data also suggests this may be the case as these are the dominant synthetic fibres, but with cotton accounting for over half the overall sales.

As Lassen et al. have not provided justification for their fibre release estimates, an alternative approach to quantification is suggested. Fibre release rates per kg washed are taken from Table 45 (woven PP and Polyester) based on fibre release observed by De Falco et al. (2017) are applied to the tonnage sales. The JRC data states that cloths will be washed 100 times during their life. It is unclear whether this includes the daily rinsing that would occur rather than cleaning in a washing machine. It is also unlikely that most cleaning cloths would last 100 washes. However, this is potentially offset by the likely higher release due to abrasion from use and the fact that they are almost always in contact with water. Estimated ***fibre releases are 1,288 tonnes per year*** which is all assumed to go to household sewers. This is around 90% lower than the results from the interim report due to update fibre release rates being used.

Table 47 – Fibre Releases to Sewers of Cleaning Cloths in Europe

Fibre Type	Sales (tonnes)	Fibre Release Rate (mg/kg)	Fibre Release over 100 Washes [c] (tonnes)
PP/viscose	72,620 [b]	100	727
Polyester	32,240 [b]	174	560
Total	104,860	**17,449**	**1,288**

Notes:

a) **Taken from Table 45 (woven PP and Polyester) based on fibre release observed by De Falco et al. (2017)**
b) **Sales figures derived from JRC (2014) Environmental Improvement Potential of Textiles (IMPRO-Textiles), Report for European Commission.**
c) **100 washes is assumed to include the daily rinsing and fibre loss due to abrasion.**

[315] JRC (2014) *Environmental Improvement Potential of Textiles (IMPRO-Textiles)*, Report for European Commission, January 2014

A.3.2 Automotive Tyre Wear

A.3.2.1 Literature review

Reports attempting national-level estimations of tyre wear-related microplastic release for Norway[316], Germany[317], Denmark[318], Sweden[319] and The Netherlands[320] have been published since 2014. More recently, since 2016, international institutions with a purview over marine management, nature conservation or environmental management more generally have conducted or commissioned studies attempting to identify and quantify sources.[321] Some have also tentatively attempted identification of pathways to the marine environments and estimates of mass flow.[322,323]

Although a focus on microplastics from tyre wear has only developed since around 2014, there has been a policy and academic interest in the impact of tyre wear particulates for far longer, upon which much of the limited analysis of tyre wear-related microplastics is based.

In the following sections the calculation of microplastic emissions at source is reviewed, followed by an analysis of how these emissions have previously been divided amongst environmental compartments, before updated estimates are calculated.

Quantification of Microplastic Emissions at Source

Broadly speaking two approaches have been employed previously for the estimation of emissions at source; the earliest known mention of which is Blok, 2005.[324] These methodologies are;

- The emissions approach and;
- the tyre sales/recycled tyres approach.

The details of how these two methods have been applied in practice in the literature are now explained.

The Emissions Approach

Traffic activity figures for a specified time period, for the country in question, expressed as vehicle kilometres driven are multiplied by a per-vehicle kilometre rate of wear. The average polymer content of tyre tread is sometimes then applied to the resulting estimate of total tyre material

---

[316] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

[317] Essel et al. (2015) *Sources of microplastics relevant to marine protection in Germany*, 2015

[318] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

[319] Kerstin Magnusson, and et al. (2016) *Swedish sources and pathways for microplastics to the marine environment*, Report for Swedish Environmental Protection Agency, March 2016

[320] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

[321] GESAMP (2015) *Sources, Fates and Effects of Microplastics in the Marine Environment: A Global Assessment*, 2015

[322] IUCN (2017) *Primary Microplastics in the Oceans: a Global Evaluation of Sources*, Report for International Union for Conservation of Nature, 2017

[323] Eunomia Research & Consulting (2016) *Study to support the development of measures to combat a range of marine litter sources*, Report for European Commission DG Environment, 2016

[324] Blok, J. (2005) Environmental exposure of road borders to zinc, *Science of The Total Environment*, Vol.348, Nos.1–3, pp.173–190

emissions where only the synthetic polymer portion of the tyre is being considered microplastic. All reviewed national- and international-level reports attempting to quantify emissions at source have applied an emissions approach.

Adjustments for additional factors

Acknowledging the broad range of factors which can influence wear rates, various studies have attempted to add an additional level of nuance to estimation methodologies in calculating emissions at source. The following additional factors have been most commonly incorporated or noted as potentially influential;

- **Vehicle type/Tyre type:**
  Tyre wear rates specific to vehicle types for application in an emissions approach have been applied in nearly all national-level estimations.

- **Speed and driving style:**
  Both our previous study and Verschoor et al. (2016) differentiate wear rates for driving carried out in simplified settings such as urban, rural and highway, based on studies suggesting that differences in road surfaces and the required frequency of braking, acceleration and turning result in varying loss of material. This division across road types also supports a more powerful analysis of what environmental compartments tyre wear is subsequently distributed to.

- **Vehicle loading/weight:**
  In its proposed methodology for the calculation of non-exhaust particle emissions from vehicles, the European Environment Agency includes a load-correction factor for Heavy Duty Vehicles[325] as vehicle weight has been shown to influence tyre wear rates.[326]

Selection/derivation of wear rates

Reports attempting national- to European-level estimates of microplastic source emissions by the emissions approach rely upon a small body of secondary and primary research for their wear rates, small differences in which can result in thousands of tonnes difference at national or European scale. It is therefore important to identify a reliable literature source of such a critical factor. To assist us in establishing a likely range for wear rates the European Tyre and Rubber Manufacturers Association has supplied their most up-to-date wear rates to compare figures in the literature against. For passenger cars they cite 0.05 - 0.25 g vkm^{-3} and for Lorries/Trucks 0.6 – 1.00 $gvkm^{-3}$. They were however unable to provide a methodology for derivation of their figures.

[325] Ntziachristos, L., and Boulter, P. (2016) *European Environment Agency - EMEP/EEA Air Pollutant Emission Inventory Guidebook - 1.A.3.b.vi-vii Road tyre and brake wear*, accessed 16 March 2017, http://www.eea.europa.eu/publications/emep-eea-guidebook-2016/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/1-a-3-b-vi

[326] Timmers, V.R.J.H., and Achten, P.A.J. (2016) Non-exhaust PM emissions from electric vehicles, *Atmospheric Environment*, Vol.134, pp.10–17

The derivation of emissions factors/wear rates for existing national- to European-level estimations is now evaluated with the aim of identifying a suitable set of emissions factors, or a universally applicable average, for use in the present study.

National tyre wear estimations in both the 2015 Denmark study and 2014 Norwegian study (for Heavy transport) are based upon wear rates derived from an informal document presented by an unnamed "expert from the Russian Federation".[327] This document cites as the source of its wear rates a 2003 study[328] on the carcinogenic risk posed by tyre wear particles. It is not clear from where the expert from the Russian Federation derives these wear rates, as neither of the two sources cited appear to include the figures, and the paper does not appear to be reporting on any primary research.

For passenger cars, the Norwegian national estimates are based upon a wear factor of 0.1g vkm^{-3} from a single UK study conducted in 2004.[329] However, Luhana et al. (2004) concluded that the wear rate for front-wheel drive cars (0.074g/vkm) was most representative due to problems encountered during the testing of rear-wheel-drive vehicles and the market dominance of front-wheel drive cars.

The review of microplastic sources to the Swedish marine environment, conducted Magnusson et al*.* in 2016, calculated national-level estimates having derived tyre rubber-specific (i.e. excluding other tyre component materials) emissions factors for buses (0.7g vkm^{-3}) and cars (0.05g vkm^{-3}) from a literature review conducted by the National Swedish Road and Transport Research Institute in 2001.[330] However, the wear rate selected from the literature review originated from a single 1987 study[331] upon which Gustafsson (2001) makes no comment as to the quality or reliability of. Furthermore, as the 1987 study appears to be unavailable for purchase in digital or print form the reliability of the wear rates could not be verified.

Having previously identified tyre wear as one of the three largest potential sources of microplastics in the Netherlands, Verschoor et al*.* (2016) calculated national-level emissions at source based on the 2015 method of the Dutch Pollutant Release and Transfer Register.[332] This catalogue of national inventory emission calculation methodologies cites as its source a 2008 guide from the Netherlands National Water Board (Water Unit) by Ten Broeke et al*.*[333] The authors of the 2008 guide derive their wear rates from a literature review of primary tyre wear rate research carried out between 1971 and 2004. They disaggregate their wear rates by road type the wear occurs on (urban, rural or highway) and nine vehicle types but do not *fully* describe the methodology disaggregating by vehicle type.

---

[327] GRPE (2013) Particulate Matter Emissions by Tyres - Transmitted by the expert from the Russian Federation
[328] Hesin (2003) *Carcinogenic risk of car tires*, accessed 16 March 2017, http://web.archive.org/web/20050223023324/http://www.miet.ru/struct/44/Art9.htm
[329] Luhana et al. (2004) Characterisation of exhaust particulate emissions from road vehicles - Measurement of non-exhaust particulate matter
[330] Gustafsson, M., Blomqvist, G., Gudmundsson, A., et al. (2008) Properties and toxicological effects of particles from the interaction between tyres, road pavement and winter traction material, *Science of The Total Environment*, Vol.393, Nos.2–3, pp.226–240
[331] Lindstrom, and Rossipal(1987) *Emissioner från landsvägs- och järnvägstrafik*, Stockholm, Sweden: Dept. of Land Improvement and Drainage
[332] Klein et al. (2015) Methods for calculating emission from transport in Netherlands
[333] Broeke et al (2008) Road traffic tyre wear - Emission estimates for diffuse sources

Finally, there is the methodology of the European Environment Agency[334] which is proposed as;

"A common basis of calculating and comparing non-exhaust particle emission in different countries."

This methodology presents PM10 and 2.5 wear rates for motorcycles, passenger cars, light duty vehicles and heavy duty vehicles, with the potential to disaggregate wear rates for all vehicle classes by speed and, additionally, for heavy transport by loading/weight where such data is available. The wear rates in the 2016 EEA methodology have been derived by averaging of wear rates collated by a 2004 peer-reviewed literature review[335] of 35 studies conducted between 1942 and 2000, having excluded out-dated values and outliers. For all of the reviewed studies the EEA also supply the calculated total material wear rate, the relevant factor from a microplastics perspective. By averaging the figures from those studies the EEA judged not to be out of date or outliers one can derive an estimated total material wear rate for passengers cars, motorcycles, light-duty vehicles and heavy-duty vehicles.

This overview highlights the difficulty of selecting appropriate wear rates when sources of previously-used figures cannot be traced and thus are not verifiable based on publicly available documents. Table 48 details all the wear rates reported in the studies evaluated above for ease of comparison.

A.3.2.2 The Tyre Sales/Recycled Tyre Approach

The alternative method for calculating emissions at source is the *Recycling/Sales Approach* whereby either tyre sales figures or records of the number of tyres collected for recycling for a set area are combined with a lifetime mass loss figure. The average polymer content of tyre tread is then sometimes applied to the resulting estimate of total tyre material emissions where only the synthetic polymer portion of the tyre is being considered microplastic. Three national- to international-level reports have adopted this approach. These are the Norwegian study by Sundt et al. (2014), the Danish study by Lassen et al. (2015), and the global-scale IUCN study (2017). Across these studies the most important factor for which there is some discrepancy in the literature is the lifetime loss rates.

The 2014 Norwegian study assumes a 10-15% lifetime loss rate but does not appear to provide any reference to support this figure. Subsequently, the 2015 Danish study also applied a 10-15% loss rate citing primary literature, their own modelling, the Dutch Environment Agency and Sundt et al. (2014). The IUCN report however assumes a 10-25% loss citing three sources;

- the 2015 German report by Essel et al. (2015)[336] which does not appear to contain any lifetime loss rate;
- the 2016 Swedish report by Magnusson et al. which provides a loss rate of 20%, and;

[334] Ntziachristos, L., and Boulter, P. (2016) *European Environment Agency - EMEP/EEA Air Pollutant Emission Inventory Guidebook - 1.A.3.b.vi-vii Road tyre and brake wear*, accessed 16 March 2017, http://www.eea.europa.eu/publications/emep-eea-guidebook-2016/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/1-a-3-b-vi

[335] Councell, T.B., Duckenfield, K.U., Landa, E.R., and Callender, E. (2004) Tire-Wear Particles as a Source of Zinc to the Environment, *Environmental Science & Technology*, Vol.38, No.15, pp.4206–4214

[336] Essel et al. (2015) *Sources of microplastics relevant to marine protection in Germany*, 2015

- the 2014 Norwegian report by Sundt et al..

Overall lifetime loss statistics appear to be relatively well agreed upon within the literature although, as with literature applying the emissions approach, a lack of clarity surrounding how some key parameters have been derived exists. Table 49 details the lifetime loss statistics and calculated total deposited wear reported in the studies evaluated above for ease of comparison.

Definition of particles in scope

There is discrepancy in the literature as to what portion of matter emitted when tyres are worn constitutes microplastics. Reports quantifying microplastic emissions at source have variously included in scope;

- all coarse tyre material that is polymer;
- all coarse tyre material (i.e. including additives and other materials which make up part of the composition of tyres);
- all coarse material emitted including the road dust (grit, sand etc.) that worn tyre material becomes bound up with when emitted; and
- multiple variations upon these categorisations.

This is important because in defining the problem of microplastic release what material is included in scope leads to hugely different estimates of the total mass of emissions at source. For example, in our previous report we assumed a 50% polymer share in tread and so treated 50% of total coarse material worn from tyres as being microplastics. However, had we assumed that all material that leaves the tyre during wear is microplastics, as Verschoor et al. (2006) have, our total emissions calculated for Europe would have been twice as much.

In the present report we treat as in scope all material larger than PM10 and smaller than 5mm which is lost from tyres through wear while driving.

Previously reported estimates of emissions at source

Table 48 details the key statistics that have been used to estimating microplastics emissions from tyres based on an emissions approach in previous national- to international-level reports and their total deposited wear.

Table 48: Previously reported estimates of emissions at source based on an Emissions approach

Study reference	Wear Rates used - g/vkm	Geography	Particles in Scope	Emissions (Tonnes year^{-1})	Emissions (grams per Capita)
Sundt et al., 2014, 2016	Car: 0.10 Heavy Transport: 0.178	Norway	A 60% polymer share of the tyre tread material.	4,500 (excl. buses) 5,000 (incl. buses)	864 960
Essel et al., 2015	Car: 0.09 Truck: 0.70 Tractor trailers: 1.2 Bus: 0.70	EU level scaled up from Germany	All plastic particles with a diameter of >1 micrometre and <5 millimetres. Note: Wear rates and total emissions are for total tyre wear.	Germany: 60,000 – 111,000 Europe: 375,000 – 693,750	730 - 1,351 737 - 1364
Lassen et al., 2015	Car: 0.033 Light commercial: 0.051 Commercial: 0.178	Denmark	Considers total particle releases as microplastics.	1,915	336
Verschoor et al., 2016	Car (Urban; Rural; Highway): 0.132; 0.085; 0.104 Van (Urban; Rural; Highway): 0.159; 0.102; 0.125 Truck (Urban, Rural, Highway): 0.850; 0.546; 0.688	The Netherlands	All tyre tread wear particles were considered to be microplastics.	17,300 (2012)	1,019
Eunomia, 2016	Car (Urban; Rural; Highway): 0.158; 0.079; 0.079 Van (Urban; Rural; Highway): 0.190; 0.095; 0.95 Truck (Urban, Rural, Highway): 0.785; 0.393; 0.393	EU scaled up from Netherlands	A 50% polymer share of the tyre tread material.	Netherlands: 7,726 Europe: 232,777	920
Magnusson et al., 2016	Car: 0.05 Bus: 0.70	Sweden	Rubber portion of tyre wear.	13,000	1370

Notes:

1. No IUCN report figure is included because while they calculated relative contributions of various sources to total microplastics emissions from all sources to the marine environment they did not state total emissions at source.

Table 49: Previously reported estimates of emissions at source based on a sales/recycling approach

Study reference	Lifetime loss statistic applied	Geography	Total emissions (Tonnes year^{-1})	Emissions (grams per Capita)
Sundt et al., 2014	10-15%	Norway	5,700	1,094
Lassen et al., 2015	10-15%	Denmark	4,200 - 6,600	736-1,051

Notes:

No IUCN report figure is included because while they calculated relative contributions of various sources to total microplastics emissions to the marine environment they did not state total emissions at source.

Estimation of Emissions at Source Conclusions

The two approaches to estimating emissions at source applied in the existing literature each have their advantages and disadvantages. The key parameters needed to carry out the sales/recycling approach such as lifetime mass loss rates of tyres have less uncertainty surrounding them, are more-widely agreed upon, than those for the emissions approach. The uncertainty surrounding accurate per-vkm wear rates for tyres likely stems from the broad range of factors, internal and external to the tyre, which can influence wear rates and as such definitive figures are not available for all vehicle classes and local contexts. This uncertainty is not helped by a lack of clarity in parts of the literature as regards the methodology by which wear rates have been derived. Notwithstanding this uncertainty, the wear rates collated in Table 48 from traceable sources lie within those bands suggested by the ETRMA. There are thus literature-based ranges of wear rates for different vehicle classes and road types and midpoints likely to be broadly representative which can be applied as reasonable estimates for the calculation of European-level emissions figures. Furthermore, of the two approaches to estimating emissions at source the emissions approach is pre-eminent owing to the potential it provides to calculate emissions across broad road categories, i.e. Urban, Rural and Highway. This enhanced understanding of where wear is produced should support a more powerful subsequent analysis of environmental fate and thus the development of more targeted approaches to mitigation. As a result, it is the emissions approach that will be our primary method of modelling emissions at source.

A.3.2.3 Calculation methodology (supplemental information)

The methodology by which estimates of emissions at source are now outlined.

This section models the transport of tyre wear-derived particles as far as the roadside at which point the microplastic pathways and sinks are described in Section A.3.8.

Traffic Activity Data

For eight Member States plus Norway[337] 2012 traffic data disaggregated by vehicle type was available from either Eurostat[338] or national data archives.[339] For the remaining Member States national total vehicle fleet traffic activity were retrieved from the OECD[340]. Then, to disaggregate this total traffic activity by vehicle type, the data were scaled using national Tyre-Sales data from 2016[341]. See Table 53 in Appendix A.3.2.4 for the sales data. To verify the accuracy of this approach (using tyre sales as a proxy for vehicle movements) the proportions of total national traffic activity represented by each vehicle group in the Member States for which this data was readily available were compared to the proportions of total tyre sales represented by each vehicle group for the same countries. The average error was 3.54%. Table 54 Appendix A.3.2.4 contains notes detailing the source of each Member State's traffic activity data. Traffic activity for each vehicle type derived from this disaggregation was then summed for all Member States to arrive at an estimate of total traffic activity for Europe disaggregated by vehicle type. See Table 50 for the results of this.

Traffic data disaggregated by vehicle type *and* road type (urban, rural and highway) from 2013 was available from Eurostat for only 4 Member States[342]. These figures were therefore averaged to provide a set of factors which could be applied to the total European traffic activity by vehicle type in Table 50. This provides European-level traffic activity disaggregated by both road type and vehicle type. See Table 50 for the results of this calculation.

---

337 France, Germany, Hungary, Latvia, the Netherlands, Poland, Romania, and the UK.

338 Eurostat (2016) Road traffic on national territory by type of vehicle and type of road (million Vkm)

339 Klein, J., Hulskotte, J., van Duynhoven, N., Hensema, A., and Broekhuizen, D. (2016) Methods for calculating the emissions of transport in the Netherlands

340 OECD (2013) *"Road traffic, vehicles and networks", in Environment at a Glance 2013: OECD Indicators*, 2013

341 ETRMA (2016) *European Tyre & Rubber Industry Statistics Edition*, 2016

342 The UK, Poland, the Netherlands and Norway.

Table 50 — Calculation of Member State Vehicle Kilometres Disaggregated by Vehicle Type and Road Type

European Traffic Activity by Vehicle Type (M Vkm)						
Vehicle Type	**Motorcycles and mopeds**	**Passenger cars**	**Buses**	**Goods vehicles <= 3.5 tonnes**	**Lorries**	**Total**
Millions of Vehicle Kilometres[1]	122,918	3,025,866	16,951	356,868	200,809	**3,723,411**
Apportionment across Road Types						
Relative (percentage) [2]						
Highway	9%	20%	15%	22%	19%	**9%**
Urban	42%	34%	42%	33%	25%	**34%**
Rural	49%	46%	43%	46%	56%	**46%**
Absolute (Millions of Vehicle Kilometres) [3]						
Highway	11,517	598,725	2,520	78,247	38,684	**729,693**
Urban	51,321	1,043,136	7,119	116,174	49,702	**1,267,451**
Rural	60,080	1,384,005	7,311	162,448	112,423	**1,726,267**

Notes:

1. **Derived from;**
   a. **Eurostat (2016);**
   b. **National archives (Klein et al., 2016); or**
   c. **the disaggregation of Member State total-vehicle fleet data provided by the OECD (2013) or Eurostat using tyre sales data provided by the ETRMA disaggregated by vehicle type. See Table 54 for a more detailed breakdown of how the traffic activity disaggregated by road type is derived.**
2. **Derived through averaging data available from Eurostat for The UK, Poland, The Netherlands and Norway**

Wear Rates

The ETRMA[343] provided upper and lower bound tyre wear rates based on their current best expert judgement. The ranges are particularly wide, to account for differing;

- tyre characteristics (radius/width/depth);
- tyre constructions;
- vehicle characteristics such as weight, distribution of load, location of driving wheels, engine power, electronic braking systems, suspension type and state of maintenance;
- road surface characteristics; and
- vehicle operation such as speed, acceleration and cornering.

These factors were applied to the traffic activity data disaggregated by vehicle type to arrive at an estimate of total tyre wear deposited in Europe. As the ETRMA could only provide estimated wear rates for cars and trucks (lorries) these were scaled before being applied, or where appropriate were directly applied, to traffic data for motorcycles, buses, and goods vehicles ≤ 3.5 tonnes. Additionally, reasonable mid-points were estimated for each of the vehicle-type wear rate ranges. See Table 51 for the results of these calculations.

The analysis outlined in Table 51 suggests an estimated lower bound of around 329, 631 tonnes, a midpoint of around 572,157 tonnes and an upper bound of around 1,117,270 tonnes. To derive wear deposited on urban, rural and highway roads, in order to facilitate a more powerful environmental pathways analysis for tyre-derived microplastics, the wear rates presented in a 2016 guide from the Netherlands National Water Board (Water Unit) (Deltares and TNO, 2016) were applied to the traffic data disaggregated by *both* vehicle type and road type. See Table 52 for the estimated total tyre wear deposited on different European road types based on these data.

This total deposited tyre wear figure (503,586 tonnes) is not dissimilar to that calculated using the midpoint of the ETRMA wear rates (572,157 tonnes). It is therefore believed to represent a reasonable working estimate of the total deposited tyre wear, and it is this data that will be carried forward for pathways modelling as it is disaggregated by urban, rural and highway deposition—each road environment are likely to have different pathways to the aquatic environment.

[343] Personal Communications with ETRMA (2017)

Table 51: Application of ETRMA wear rates to European Traffic Activity

European Traffic Activity by Vehicle Type (Millions of Vehicle Kilometres)						
Vehicle type	**Motorcycles and mopeds**	**Passenger cars**	**Buses**	**Goods vehicles <= 3.5 tonnes**	**Lorries**	**Total**
Millions of Vehicle Kilometres	122,918	3,025,866	16,951	356,868	200,809	**3,723,411**
Wear Rates (g vkm-3)						
Lower	0.025[3]	0.05[1]	0.6[4]	0.125[5]	0.6[1]	
Upper	0.075[3]	0.25[1]	1[4]	0.375[5]	1[1]	
Average	0.05[3]	0.1[2]	0.8[4]	0.25[5]	0.8	
Tyre Wear (Tonnes)						
Lower	3,073	151,293	10,170	44,609	120,485	**329,631**
Upper	9,219	756,466	16,951	133,826	200,809	**1,117,270**
Mid-Point	6,146	302,587	13,560	89,217	160,647	**572,157**

Notes:
1. **Wear rates provided by the ETRMA.**
2. **0.1g vkm^{-3} was selected as the midpoint for passenger cars on the basis that the majority of previously reported wear rates in the primary literature lie around this figure. This is illustrated by** Figure 20 **in Appendix** A.3.2.4**, a diagram produced by Boulter (2005)** *[344]* **depicting the spread of wear rates for light-duty vehicles found in a literature review conducted by Councell et al. (2004)***[345]* **and additional published values identified by Boulter. 0.1g vkm^{-3} remains an appropriate midpoint when outdated valued are excluded.**
3. **Due to their lower weight motorcycles were assumed to have a midpoint wear rate equal to the lower bound of passenger car wear rates. Upper and Lower bounds for motorcycles were estimated to be 50% higher and lower than this midpoint respectively.**
4. **Bus wear rates were assumed to be the same as those for Lorries.**
5. **Due to their greater weight Goods vehicles <= 3.5 tonnes were assumed to have a midpoint wear rate equal to the upper bound of passenger car wear rates. Upper and Lower bounds for Goods vehicles <= 3.5 tonnes were estimated to be 50% higher and lower than this midpoint respectively.**

[344] Boulter, P., and et al. (2004) *Measurement of non-exhaust particulate matter*, Report for EUROPEAN COMMISSION Directorate General Transport and Environment, 2004

[345] Councell, T.B., Duckenfield, K.U., Landa, E.R., and Callender, E. (2004) Tire-Wear Particles as a Source of Zinc to the Environment, *Environmental Science & Technology*, Vol.38, No.15, pp.4206–4214

Table 52: Application of Ten Broeke et al. (2016) Wear Rates to Traffic Activity Data

European Traffic Activity by Vehicle Type (Millions of Vehicle Kilometres)						
Vehicle type	**Motorcycles and mopeds**	**Passenger cars**	**Buses**	**Goods vehicles <= 3.5 tonnes**	**Lorries**	**Total**
Highway	11,517	598,725	2,520	78,247	38,684	**729,693**
Urban	51,321	1,043,136	7,119	116,174	49,702	**1,267,451**
Rural	60,080	1,384,005	7,311	162,448	112,423	**1,726,267**
Wear Rates (g vkm-3)						
Highway	0.047	0.104	0.326	0.125	0.668	
Urban	0.060	0.132	0.415	0.159	0.850	
Rural	0.039	0.085	0.267	0.102	0.546	
Tyre Wear Emitted (Tonnes)						
Highway	541	62,267	822	9,781	25,841	**99,252**
Urban	3,079	137,694	2,955	18,472	42,246	**204,446**
Rural	2,343	117,640	1,952	16,570	61,383	**199,888**
Total	**5,964**	**317,602**	**5,728**	**44,822**	**129,470**	**503,586**

A.3.2.4 Data Tables

Table 53: European Tyre Sales Data 2012 (tonnes)

	Passenger cars	Goods vehicles <= 3.5 tonnes	Lorries	Motorcycles and mopeds
Austria	4,878,564	520,981	209,460	170,934
Belgium + Lux	4,142,191	374,609	301,319	137,631
Bulgaria	914,784	156,314	80,403	6,296
Croatia	968,710	87,585	62,546	27,756
Cyprus	188,280	19,729	10,536	7,727
Czech Republic	3,479,965	388,746	206,515	84,576
Denmark	2,111,697	243,162	110,489	32,545
Estonia	347,161	39,796	26,248	9,296
Finland	2,613,946	321,499	107,646	62,508
France	26,532,535	2,523,966	882,156	1,164,845
Germany	43,271,118	3,524,852	1,228,043	1,562,874
Greece	2,351,706	163,497	111,952	282,174
Hungary	1,761,458	231,426	126,691	31,672
Ireland	1,124,689	86,091	89,230	16,001
Italy	18,602,187	1,448,828	1,011,351	1,871,841
Latvia	372,517	51,739	32,804	7,087
Lithuania	572,057	63,613	75,747	4,138
Malta	16,247	1,373	215	0
Netherlands	6,960,421	538,564	299,958	249,825
Poland	9,210,752	1,241,072	661,430	87,141
Portugal	1,974,796	189,015	161,729	31,579
Romania	1,981,417	436,882	260,912	4,861
Slovakia	1,417,493	140,625	69,792	16,054
Slovenia	990,010	73,605	53,122	29,467
Spain	12,427,482	838,081	670,483	699,577
Sweden	3,661,085	326,919	198,520	106,667
UK	15,425,102	1,388,014	776,337	657,255

Source: ETRMA (2016) European Tyre & Rubber Industry Statistics Edition 2016

Table 54: Millions of Vehicle Kilometres (A) multiplied by Wear Rates (B) to arrive at Tonnes of Tyre-Derived Microplastics Emitted Annually (C) – A x B = C

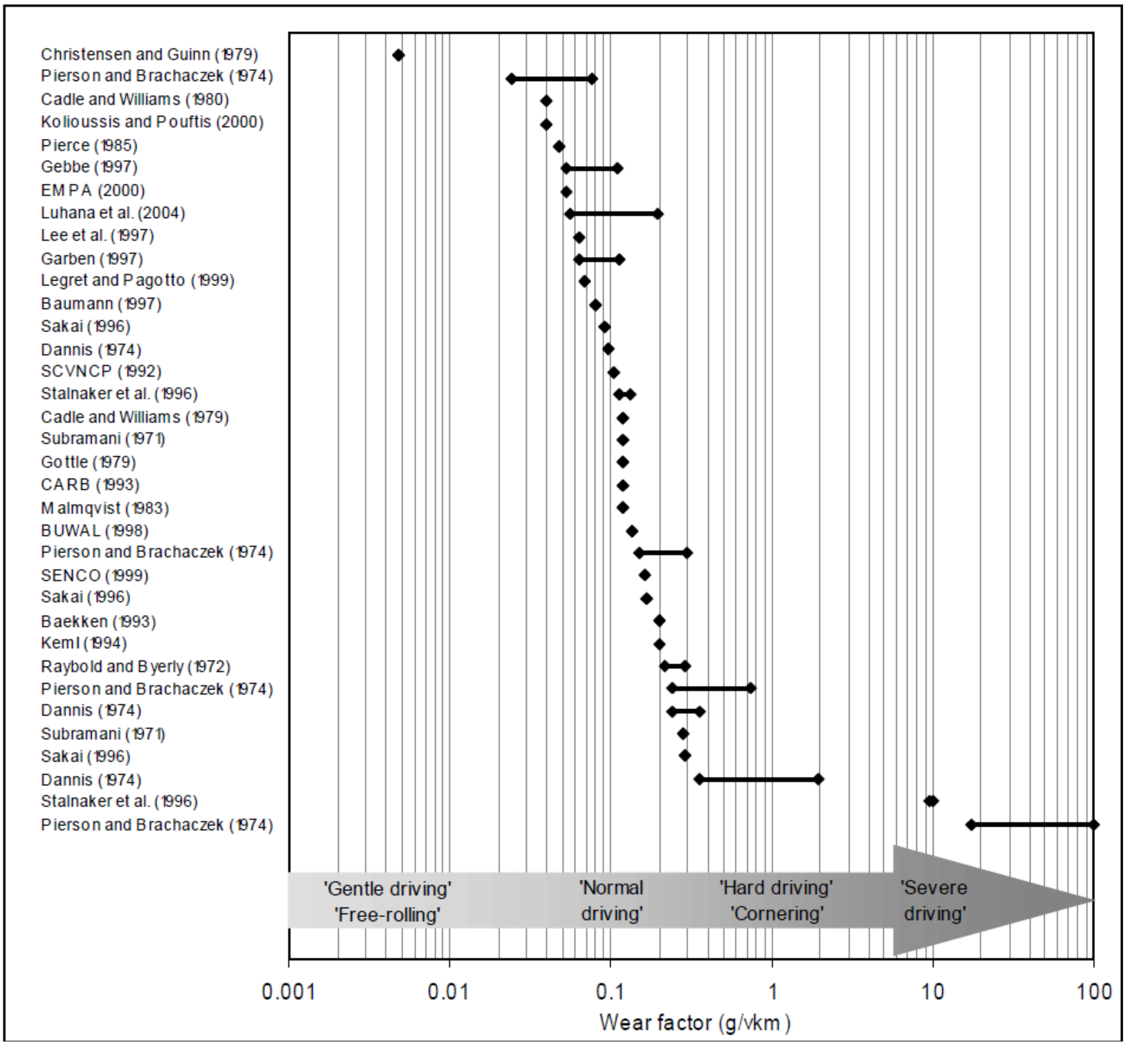
	Millions of Vehicle Kilometres (A)						Wear Rates (Grams per Vehicle Kilometre) (B)					
	Motorcycles and mopeds	Passenger cars	Buses	Goods vehicles <= 3.5 tonnes	Lorries	Total	Motorcycles and mopeds[7]	Passenger cars8	Buses[9]	Goods vehicles <= 3.5 tonnes[10]	Lorries[11]	Total
							0.05	0.10	0.80	0.20	0.80	
							Tonnes of Tyre-Derived Microplastics Emitted Annually (C)					
Austria[1]	2,282	65,142	-	6,957	2,797	77,178	114	6,514	-	1,391	2,237	10,257
Belgium + Lux[2]	3,096	93,179	-	8,427	6,778	111,480	155	9,318	-	1,685	5,423	16,581
Bulgaria[3]	-	-	-	-	-	-	-	-	-	-	-	-
Croatia[4]	189	17,995	300	0	1,833	20,317	9	1,800	240	-	1,466	3,515
Cyprus[3]	-	-	-	-	-	-	-	-	-	-	-	-
Czech Republic[2]	1,089	44,802	-	5,005	2,659	53,555	54	4,480	-	1,001	2,127	7,663
Denmark[2]	592	38,433	-	4,426	2,011	45,462	30	3,843	-	885	1,609	6,367
Estonia[2]	213	7,944	-	911	601	9,668	11	794	-	182	481	1,468
Finland[5]	1,080	45,161	-	5,554	1,860	53,655	54	4,516	-	1,111	1,488	7,169
France[4]	13,932	426,280	3,280	92,878	27,110	563,480	697	42,628	2,624	18,576	21,688	86,212
Germany[4]	16,253	595,045	3,084	46,935	33,930	695,247	813	59,504	2,467	9,387	27,144	99,315
Greece[2]	7,916	65,970	-	4,586	3,140	81,613	396	6,597	-	917	2,512	10,423
Hungary[4]	411	24,253	636	4,946	4,731	34,977	21	2,425	509	989	3,785	7,729
Ireland[2]	532	37,371		2,861	2,965	43,728	27	3,737	-	572	2,372	6,708

Italy[2]	44,941	446,617	-	34,785	24,281	550,624	2,247	44,662	-	6,957	19,425	73,291
Latvia[4]	-	8,234	222	977	1,471	10,904	-	823	177	195	1,177	2,373
Lithuania[2]	56	7,750		862	1,026	9,694	3	775	-	172	821	1,771
Luxembourg[3]	-	-	-	-	-	-	-	-	-	-	-	-
Malta[3]	-	-	-	-	-	-	-	-	-	-	-	-
Netherlands[6]	4,880	103,122	543	16,649	7,008	132,203	244	10,312	434	3,330	5,606	19,927
Poland[4]	3,676	166,095	2,062	15,784	18,958	206,575	184	16,610	1,650	3,157	15,166	36,766
Portugal	1,266	79,178	-	7,578	6,484	94,507	63	7,918	-	1,516	5,188	14,684
Romania[4]	576	33,430	1,827	983	7,755	44,571	29	3,343	1,462	197	6,204	11,234
Slovakia[2]	154	13,640	-	1,353	672	15,820	8	1,364	-	271	537	2,180
Slovenia[2]	458	15,397	-	1,145	826	17,826	23	1,540	-	229	661	2,452
Spain[2]	11,526	204,751	-	13,808	11,047	241,131	576	20,475	-	2,762	8,837	32,650
Sweden[2]	1,909	65,523	-	5,851	3,553	76,836	95	6,552	-	1,170	2,842	10,660
UK[4]	4,551	386,678	4,400	66,436	25,005	487,070	228	38,668	3,520	13,287	20,004	75,707
Norway[4]	1,339	33,876	597	7,172	2,308	45,292	67	3,388	478	1,434	1,846	7,213
Switzerland[3]	-	-	-	-	-	-	-	-	-	-	-	-
Total	**122,918**	**3,025,866**	**16,951**	**356,868**	**200,809**	**3,723,411**	**6,146**	**302,587**	**13,560**	**71,374**	**160,647**	**554,314**

Notes:

1. **Traffic activity disaggregated by road type from Eurostat (2016) summed and then disaggregated using Tyre Sales data (ETRMA, 2016).**
2. **Member State total vehicle fleet traffic activity for 2011 taken from OECD (2013) disaggregated by vehicle types using tyre sales data (ETRMA, 2016).**
3. **Data not available.**
4. **Traffic activity disaggregated by vehicle type from Eurostat (2016).**
5. **Partial breakdown of Member State traffic activity by vehicle type available from Eurostat. This data is summed and then disaggregated using Tyre Sales data (ETRMA, 2016).**
6. **Data from national archives (Klein et al., 2016)**
7. **Derived from ranges of wear rates suggested by the ETRMA. Due to their lower weight motorcycles were assumed to have a midpoint wear rate equal to the lower bound of passenger car wear rates. Upper and Lower bounds for motorcycles were estimated to be 50% higher and lower than this midpoint respectively.**
8. **Derived from ranges of wear rates suggested by the ETRMA. 0.1g vkm-3 was selected as the midpoint for passenger cars on the basis that the majority of previously reported wear rates in the primary literature lie around this figure. This is illustrated by Figure 3, a diagram produced by Boulter (2005) depicting the spread of wear rates for light-duty vehicles found in a literature review conducted by Councell et al. (2004) and additional published values identified by Boulter. 0.1g vkm-3 remains an appropriate midpoint when outdated valued are excluded.**
9. **Derived from ranges of wear rates suggested by the ETRMA. Bus wear rates were assumed to be the same as those for Lorries.**
10. **Derived from ranges of wear rates suggested by the ETRMA. Due to their greater weight Goods vehicles <= 3.5 tonnes were assumed to have a midpoint wear rate equal to the upper bound of passenger car wear rates. Upper and Lower bounds for Goods vehicles <= 3.5 tonnes were estimated to be 50% higher and lower than this midpoint respectively.**
11. **Mid-point of a range of Lorry Wear rates suggested by the ETRMA.**

Figure 20: Diagram from Boulter et al., 2005 – cited as "Wear factors for light-duty vehicles (adapted from Councell et al., 2004)



Sources: A) Boulter, P. (2005) A review of emission factors and models for road vehicle non-exhaust particulate matter; B) Councell, T.B., Duckenfield, K.U., Landa, E.R., and Callender, E. (2004) Tire-Wear Particles as a Source of Zinc to the Environment, Environmental Science & Technology, Vol.38, No.15, pp.4206–4214; ETRMA (2017) Personal Communication.

A.3.3 Automotive Brake Wear

A.3.3.1 Calculation Methodology

Traffic activity data calculated for estimating emissions at source of tyre wear-derived microplastics is applied in this modelling of brake wear.

Per-kilometre wear rates were collected from a literature review of primary experimental research for passenger cars, light goods vehicle and lorries.[346] For light goods vehicles an average figure was available which was scaled by 25% up and down to arrive at lower and upper bound estimates respectively. For passenger cars and lorries upper and lower bound estimates were available which were averaged to arrive at an estimated midpoint rate. These derived wear rates were applied to the aforementioned traffic activity data (see Appendix A.3.2.4). Although bus and motorcycle wear rates were not available they only represent 3.8% of total European annual vehicle kilometres and so the impact on estimated emissions at source is unlikely to be significant.

With regards microplastic emissions, not all material lost from brakes through wear is in scope. Firstly, not all material worn from the brake lining is emitted to the environment as some is trapped within the vehicle in areas such as the brake drums and wheels. The methodology of the European Environment Agency for estimating emissions of air pollutants[347] notes that vehicle-specific features will dictate what portion of wear is trapped, but cite 50% as being typically applicable. Additionally, as with tyre-wear derived microplastics, it is only those particles of emitted wear which are >10µm and <5mm in size, the coarse fraction, which are in scope. Estimates of coarse fraction of brake wear were derived from a recent literature review of primary research.[348] There is some uncertainty as to the fraction of wear which is coarse and, as such, upper and lower estimates from literature cited by the review of 2% and 38% have been applied. See Table 55 for the application of these additional factors to generated wear.

There is no factor to account for the fraction of the worn brake material that is made of polymers because, as with tyre wear-derived microplastics, it is assumed that the constituent materials are bound together when worn from the brake friction material and so the entire particle is treated as in scope. This assumption may be altered in the future.

The analysis outlined suggests an estimated lower bound of around 505 tonnes and an upper bound of around 17,161 tonnes annual emissions at source of brake wear-derived microplastics. These results are highly sensitive to assumptions around the fraction of wear that escapes the vehicle over its lifetime, including at the end-of-life stage which does not appear to be included in the 50% escape rate applied.

---

[346] Luhana et al. (2004) Characterisation of exhaust particulate emissions from road vehicles - Measurement of non-exhaust particulate matter

[347] Ntziachristos, L., and Boulter, P. (2016) *European Environment Agency - EMEP/EEA Air Pollutant Emission Inventory Guidebook - 1.A.3.b.vi-vii Road tyre and brake wear*, accessed 16 March 2017,

[348] ibid

Table 55: Application of Brake Lining Wear Rates to European Traffic Activity

	Traffic Activity (Thousands of vkm) [1]			Total
Vehicle Type	**Passenger cars**	**Goods vehicles <= 3.5 tonnes**	**Lorries**	
	3,025,866	356,868	200,809	**3,583,543**
	Wear Rates (g vkm^{-3}) [2]			
Low	0.011	0.022	0.047	
Medium	0.020	0.036	0.084	
High	0.016	0.029	0.066	
	Total Wear (Tonnes)			
Low	33,285	7,762	9,438	**50,484**
High	60,517	12,936	16,868	**90,322**
Medium	46,901	10,349	13,153	**70,403**
	Total emissions assuming 50% entrapment in vehicle (Tonnes) [2]			
Low	16,642	3,881	4,719	**25,242**
High	30,259	6,468	8,434	**45,161**
Medium	23,450	5,175	6,576	**35,202**
	Coarse Fraction			
	Assuming 2% of emissions are coarse (Tonnes) [2]			
Low	333	78	94	**505**
High	605	129	169	**903**
Medium	469	103	132	**704**
	Assuming 38% of emissions are coarse (Tonnes) [2]			
Low	6,324	1,475	1,793	**9,592**
High	11,498	2,458	3,205	**17,161**
Medium	8,911	1,966	2,499	**13,377**

Notes:

1. ***Calculated according to the method described in the Appendix A.3.2.3***
2. ***Derived from Ntziachristos, L., and Boulter, P. (2016)***

A.3.4 Artificial Sports Turf

A.3.4.1 Literature Review

A Swedish government study[349] estimated that between three and five tonnes of infill is needed annually to preserve the properties of an 11-a-side football pitch. With 1,400 football fields in Sweden, the study therefore estimated 2,300—3,900 tonnes of infill are lost each year. No estimates are given for other types of sports pitches. The study also makes no estimate as to the proportion of these losses that would end up in surface water.

A study[350] from the Danish Environmental Protection Agency also estimated that three to five tonnes are 'consumed' for each pitch every year, however due to material settling estimated that only half of this is released to the environment. The study also estimated that around 5—10% of the grass pile (weighing 0.04-0.08kg/m^2) is also released every year. The study estimates that 5—20% of released material ends up in in WWT plants, of which 3—6% is released into the surface water.

The study also highlighted the following release pathways that would allow the plastic particles to travel towards surface waters:

- Release to surrounding soil area;
- Release to paved areas surrounding the field, and subsequently release to sewerage system via grates (includes releases from shoes and clothing);
- Release of infill particles to the indoor environment, as the particles get stuck in sports bags, shoes and clothing where they 1) are removed by vacuum cleaning or 2) are released to sewerage system via discharges from washing machines; and
- Release to drainage via drainage water. The fate of the drainage water is: 1) downward seepage; 2) release to sewerage system; or 3) release to nearby streams due to heavy rainfall.

A study[351] for the Norwegian Environment Agency discusses the findings of the Danish and Swedish studies and used data from both to form their own estimate of a loss of 10% of the infill per pitch per year, of which 50% is 'lost to nature'. The study also suggests that the losses to surface waters may be much higher in Norway dues to the harsher climate and poorer waste water treatment solutions. The Norwegian estimate of infill loss (10%) is much higher than other studies (3—5%) which is largely attributed to the practice of snow removal which also captures some of the infill. Figure 21 shows examples of how infill is routinely piled up alongside pitches during snow removal and in some extreme cases this happens directly adjacent to watercourses.

[349] Kerstin Magnusson, and et al. (2016) *Swedish sources and pathways for microplastics to the marine environment*, Report for Swedish Environmental Protection Agency, March 2016
[350] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015
[351] Mepex (2016) *Primary microplastic- pollution: Measures and reduction potentials in Norway*, April 2016

Figure 21 – Infill Accumulation from Snow Removal in Norway

Source: Mepex (2014)

None of the studies distinguish between the types of infill, although as SBR is the most common it is assumed that these studies refer to this type. There is no evidence to suggest that other oil-based materials will be more likely to be emitted—the density of TPE is 0.8—1.22[352] g/cm^3, EPDM is around 1.1[353] g/cm^3, and SBR is around 1.2[354] g/cm^3. As water density is 1 g/cm3, this makes all of these materials (except for the lower density TPEs) negatively buoyant i.e. they will sink in water. This means that it is likely that the material type will make little difference to the likely loss rate.

The Danish study[355] attempted to quantify the loss of the pile fibres and assumed a 5—10% loss per year, but this seems unrealistic due to the fact that a pitch is usually expected to last around 10 years, at which point it is unlikely to have lost 50—100% of the fibres. A study by Loughborough University[356] found that annual data collected from over 165 pitches in the UK found that the mean pile loss was 0.32 mm per year. The mean age of the pitches was 4.8 years, but the maximum age was around 15 years old. The older pitches were also found to have a greater pile loss per year with the maximum loss being 2.4 mm per year. With the pile on 3G pitches ranging from 40—65 mm, even the shortest pile length (40 mm) combined with the highest loss rate (2.4 mm/yr.) would only reach the equivalent of just over the lower (5%) Danish loss rate estimate. The Loughborough study does point out that the use rate of the pitches is not recorded in the data, but the relatively large sample size would reflect the full range of typical quality and usage. It also only looked at the measured wear rate, but does not account for full fibre loss when it is ripped from the backing or when lose fibres are released during the first few years of use.

---

[352] http://www.apstpe.com/maxelast-tpe/

[353] http://www.polyhedronlab.com/services/rubber-testing/epdm-rubber-testing.html

[354] http://www.stargum.pl/en/granules/sbr_granules

[355] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

[356] Sharma, P., Fleming, P., Forrester, S., and Gunn, J. (2016) Maintenance of Artificial Turf – Putting Research into Practice, *Procedia Engineering*, Vol.147, pp.830–835

Table 56 summarises the loss estimates from the three Scandinavian countries which vary depending upon the assumptions used to calculate them. This assumes a standard 11- a-side 106 x 71 meter football pitch.

Table 56 – Estimated Microplastic Emissions from Artificial Turf per Year

Country	Infill Loss (kg/m3)	Pile Loss (kg/m3)	Total Loss Per Pitch (tonnes)
Sweden	0.4—0.67	n/a	3—5
Norway	1.59	n/a	12
Denmark	0.2—0.33	0.07—0.14	Infill = 1.5—2.5 pile = 0.5—0.9

There is currently on one study which has attempted to create a mass balance for infill in artificial turf. The study, from the Netherlands[357], looked at three local pitches containing SBR infill and one containing TPE.

The results of the mass balance are extremely variable and the methods are not described fully, which makes firm conclusions difficult to ascertain. In all cases the infill added to the pitches did not equal that which was estimated to be lost (i.e. the mass did not balance). In one case, four times more infill was applied than lost. In another, none was applied and 300kg lost. There is little information given about the maintenance schedules for the pitches or any other reasons for this disparity.

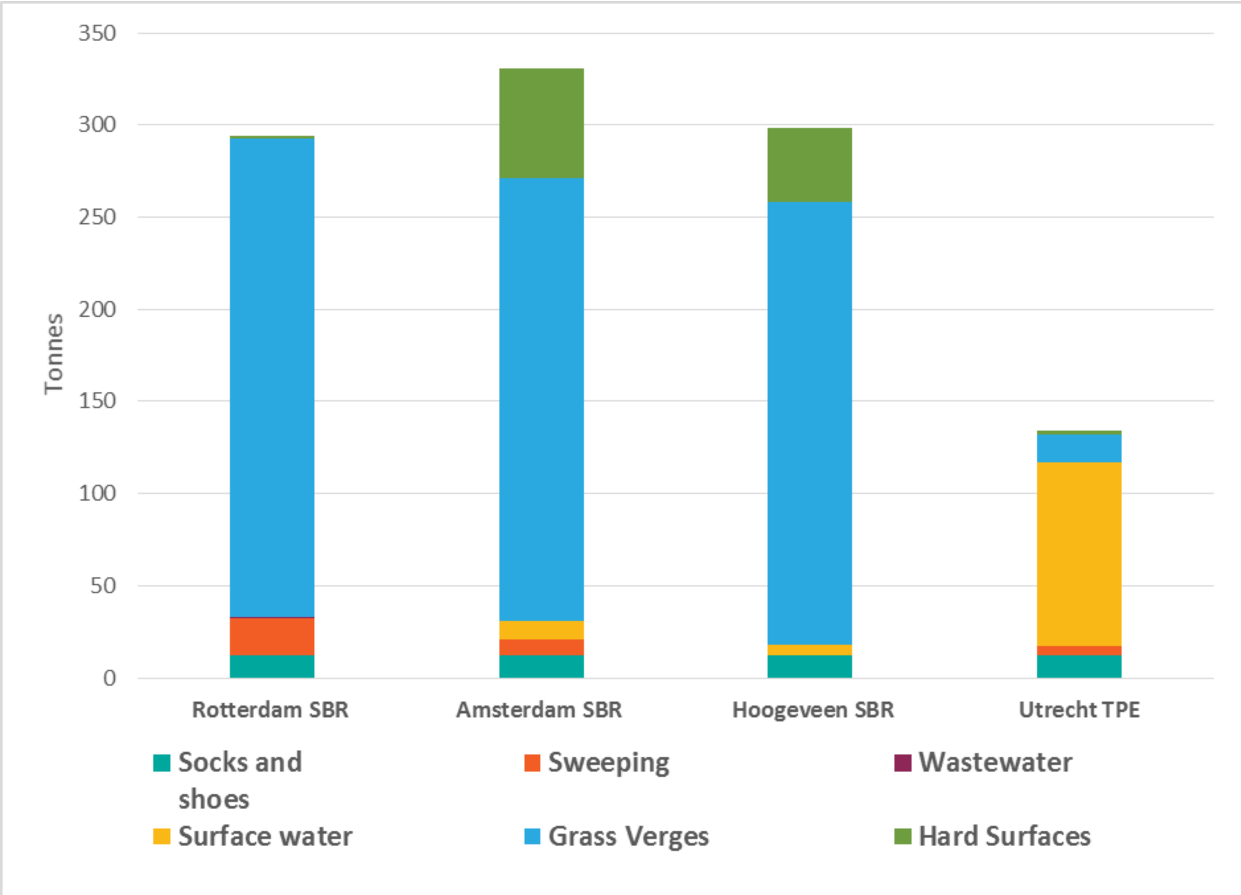
The estimates for one pitch show that 75% of losses go to surface water (see Figure 22), whereas the other three show almost zero. The report suggest that the majority will end up in nearby grass verges, but this is very situational dependent. Very little is was also found to be swept up. This is in stark contrast to observations made during a visit by the current study authors to a pitch in the UK. Most of the infill was found within the hard standing 2—3 meters from the pitch edge and this is regularly swept away (and in close proximity to surface drains). Similarly, the estimates of 12kg per year losses related to infill transported by player's socks and shoes is very low compared with observations—conversations with the English FA suggest that during matches where it rains, the infill will stick to the players' clothing at a much higher rate. The Dutch study estimated based on one sample taken on a dry day.

The small sample size and the conflicting results show that this issue is one that may be very dependent of the local situation and therefore applying any of the results across Europe is unwise at this stage.

---

[357] Annet Weijer, and Jochem Knol (2017) *Verspreiding van infill en indicatieve massabalans*, Report for Branchevereniging Sport en Cultuurtechniek, May 2017

Figure 22 – Measured Losses of Infill to Different Compartments



Source: Generated using data from Annet Weijer, and Jochem Knol (2017)

A.3.4.2 Artificial Turf Market Data

The market for 3G turf has grown considerably in recent years, with the number of full sized 3G football pitches in Europe trebling between 2006 and 2012—this growth is also expected to continue into 2020 at least.[358] Table 57 shows that the dominant use for infill globally is SBR in contact sports. This accounts for 98% of the global demand for infill.

Table 57 – Global Infill demand for artificial turf in 2015 ('000 tonnes per year)

	SBR	EPDM	TPE	Coated sand/ SBR	Other	Total
Contact sport	1,265	4.7	12.9	3.4	9.1	1,286
Non-contact sport	2	-	-	-	-	2
Leisure/DIY	3.6	-	-	0.1	0.2	3.7
Landscaping	0.8	-	-	-	-	0.8
Total	**1,271.4**	**4.7**	**13**	**3.5**	**9.3**	**1,292.5**

Source: AMI Consulting Via European Chemicals Agency[359]

A.3.4.3 Artificial Turf Example Site

Figure 23 shows photographs from an artificial turf pitch installed in the UK in 2015. They demonstrate the extent to which infill can migrate around the grounds of the pitch and surrounding areas. The changing rooms were found to be covered in infill material after a match had been conducted the previous day. This was, in part, due to the wet weather during the match which apparently increases the amount of infill that sticks to players' clothing. Infill was also found all throughout the hard standing nearby and in close proximity to surface drains. Brushes are used at the exit/entrance to the pitch to clean infill from the players' boots. This is reportedly swept away and disposed of with the residual waste. There was also evidence of the infill migrating towards the edges of the pitch where little activity occurs. The edges are lever with the surround area which also allows the infill to easily transfer out of the pitch.

[358] ESTO (2016) *Market Report Vision 2020*, 2016

[359] European Chemicals Agency (2017) *An Evaluation Of The Possible Health Risks Of Recycled Rubber Granules Used As Infill In Synthetic Turf Sports Fields*, February 2017

Figure 23 – Infill Movement from Artificial Turf Pitches

Clockwise from top left: Changing rooms after a match conducted during rain, Infill build-up near adjacent buildings, Brushes used to remove infill from players' boots, Build-up of infill towards the edges of the pitch.

Source: Eunomia

A.3.4.4 Artificial Turf Installation Data

Table 58 shows that calculation method for estimating the total amount of infill that is currently installed in football and rugby pitches in Europe. The data for the number of pitches installed was estimated by the European Synthetic Turf Organisation (ESTO) in a market report[360] provided to this study. The report surveyed football associations from across Europe to estimate the number of full sized and small training pitches installed in 2012. The survey also asked them to estimate the number that will be built by 2020. Although not all FAs were able to estimate this future scenario, data was extrapolated from these few across Europe. Estimates for pitch numbers are found in A.3.4.4. An estimate for installed rugby pitches was also provided in the ESTO report for Europe as a

[360] ESTO (2016) *Market Report Vision 2020*, 2016

whole. Although artificial turf use in rugby is growing fast, it currently only represents 2% by surface area installed.

Table 58 – Calculation Method for Total Installed Infill in Europe

Pitch	Length (m)	Width (m)	Area (m^2)	Number Pitches	Installed Area (m^2)	%
Full Size Football[1]	106	71	7,526	11,459[5]	86,240,434	77%
Small Football[2]	30	20	600	39,925[5]	23,954,846	21%
Rugby[3]	116	74	8,584	232[5]	1,991,488	2%
				Total	112,186,768	100%
				SBR Infill Installed Density (kg/m^2)	16.1[4]	
				Total Installed Infill (t)	**1,802,626**	

Notes:

1. ***Football pitches can vary in size although the English Football Association specified a pitch of 106x71m in 2010. This has since been updated to 116x76m in 2012 but the earlier figure is used to account for legacy installations.***

2. ***Small football pitches are assumed to be at least this size according to data from ESTO.***

3. ***Recommended size according to the Rugby Football Union (RFU). World Rugby specifies a much larger range of possible pitch sizes.***

4. ***Figure calculated by confidential data provided by FIFA as an average of the SBR infill in pitches installed under the FIFA Quality Programme which accounts for around 20% of full sized pitches in Europe.***

5. ***Figures calculated from data provided by ESTO (see Appendix A.3.4.4 for full country breakdown).***

Table 59 – Artificial Turf Football Pitch Installations in Europe

Country	Large Football (2012)	Small Football (2012)	Total Installed Area (m^2)
Austria	186	648	1,788,666
Belgium	280	976	2,692,615
Bulgaria	10	35	96,165
Croatia	19	66	182,713
Cyprus	5	17	48,082
Czech Republic	187	652	1,798,282
Denmark	146	509	1,404,007
Estonia	9	31	86,548
Finland	207	721	1,990,612
France	2,157	7,515	20,742,754
Germany	3,053	10,637	29,359,123
Greece	40	139	384,659
Hungary	18	63	173,097
Ireland	48	167	461,591
Italy	394	1,373	3,788,894
Latvia	15	52	144,247
Lithuania	6	21	57,699
Luxembourg	10	35	96,165
Malta	8	28	76,932
Netherlands	1,450	5,052	13,943,901
Poland	100	348	961,648
Portugal	400	1,394	3,846,593
Romania	20	70	192,330
Slovakia	49	171	471,208
Slovenia	10	35	96,165
Spain	350	1,219	3,365,769
Sweden	475	1,655	4,567,829
UK	797	2,777	7,664,337
Norway	897	648	8,625,985
Switzerland	113	976	1,086,663
Total Pitches	**11,459**	**39,925**	**110,195,280**

Source: ESTO and own calculations

A.3.5 Paints

The section has revised estimates from the ones given in the interim version of this report. This is based on new information provided in a report by CEPE[361] in response to data queries and technical questions posed by this study's authors. The correlation of the paint industry is welcomed and has led to improved data and assumptions used in the revised calculations.

A.3.5.1 Paint Market Data

There is data available for the amount of paint demand in Europe from various sources which all have similar outcomes. IRL produce market reports for this sector for Eastern[362], Central[363] and Western[364] Europe by individual country which has been combined and summarised in Table 60. This covers the year 2013 for Central and Western Europe and 2011 for Eastern Europe and excludes Cyprus, Luxemburg and Malta. In the following sections of this report, marine and architectural/decorative paints, as well as automotive paints are of most relevance. Road marking paints are also discussed, however this data does allow identification of the current market for these paints.

Table 60 - Paint Demand EU28 + NO, CH (excl. Cyprus, Luxemburg and Malta)

Sector	Paint Demand (tonnes)	Market Share
Architectural/Decorative	**4,213,520**	**62%**
General Industrial	951,440	14%
Automotive OEM	**339,800**	**5%**
Industrial Wood	339,800	5%
Powder	339,800	5%
Protective	203,880	3%
Automotive Refinish	**135,920**	**2%**
Marine	**135,920**	**2%**
Plastic Coatings	135,920	2%
Total	**6,796,000**	**100%**

Source: IRL (2014, 2013)

CEPE have also provided sales data for architectural/decorative and marine paints directly to the authors of this study on a confidential basis. It is confirmed that the data does not significantly

[361] CEPE (2017) *Micro-plastics emitted from 'wear and tear' of dried paints. The view of the paint industry.*, September 2017

[362] IRL (2013) *A PROFILE OF THE EASTERN EUROPEAN PAINT INDUSTRY*, February 2013

[363] IRL (2014) *A PROFILE OF THE CENTRAL EUROPEAN PAINT INDUSTRY*, November 2014

[364] IRL (2014) *A PROFILE OF THE WEST EUROPEAN PAINT INDUSTRY*, June 2014

deviate from the IRL data but provides more detail around the specific applications within the market segments (external/external application for example).

A.3.5.2 Building Paints

Various estimates around the proportion of unused paint have been published. The OECD report suggests 25% and 3% for DIY and trade respectively[365] and a WRAP report[366] in the UK estimated the same for DIY but 1.5% for trade. The trade figures appear to be rarely disputed, however the Ecolabel[367] study reports that stakeholders believed that used paint in the DIY sector was closer to 10%. The Dutch paint association estimate between 10 and 16%[368]. Based on these sources an unused paint proportion of 15% and 3% will be used for DIY and trade respectively—this also agrees with estimates from CEPE[369].

There are also potentially differences between the professional trade market and the DIY market. The split between the trade and DIY markets for decorative paints could vary from 30—70%. During market research for a revision of the indoor and outdoor paints EU Ecolabel[370] data was found that suggests the DIY market accounts for 41% of decorative paint sales in Europe. CEPE[371] estimate this to be closer to a 50:50 split. Combined with the data on unused paint for these sectors therefore shows that 8% remains unused in total (or 9% using the CEPE split).

A further factor is also necessary to disaggregate the overall sales data; the split between interior and exterior paint. Whilst there is no publicly available European data for this, both Sweden and Denmark have sales figures for interior paints published in the Nordic Eco-label[372]. Compared with the overall sales estimates from IRL[373], we find that interior decorative paint accounts for 37% for both countries. This also agrees with the Danish microplastics[374] study calculated from different data sources. The representativeness of this for Europe as a whole is debatable, however. Conversations with CEPE and their own report suggest that interior paints account for 75%. This is a vast difference and also has a large bearing on the results—the amount of architectural paint assigned to outdoor consumption is reduced by almost two thirds compared with the estimates shown in the interim report. The large difference may be accounted for by the expectation that Nordic countries may use

---

[365] OECD (2009) *Emission Scenario Document On Coating Industry (Paints, Lacquers and Varnishes)*, 2009

[366] WRAP (2013) *Product Opportunity Summary: Paints & varnishes*, April 2013

[367] Oakdene Hollins (2012) *Revision of EU European Ecolabel and Development of EU Green Public Procurement Criteria for Indoor and Outdoor Paints and Varnishes - Ecolabel Background Report*, Report for JRC, June 2012

[368] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

[369] CEPE (2017) *Micro-plastics emitted from 'wear and tear' of dried paints. The view of the paint industry.*, September 2017

[370] Oakdene Hollins (2012) *Revision of EU European Ecolabel and Development of EU Green Public Procurement Criteria for Indoor and Outdoor Paints and Varnishes - Ecolabel Background Report*, Report for JRC, June 2012

[371] CEPE (2017) *Micro-plastics emitted from 'wear and tear' of dried paints. The view of the paint industry.*, September 2017

[372] Nordic Ecolabelling (2015) *About Nordic Ecolabelled Indoor paint and varnishes*, January 2015

[373] IRL (2014) *A PROFILE OF THE WEST EUROPEAN PAINT INDUSTRY*, June 2014

[374] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

proportionally more external paint due to the harsher climates that necessitate re-coating at a greater frequency.

This estimate is backed up by a third-party source from a market report published in the European Coating Journal. Results from this publication show that interior paints account for 73%[375]. Applying this figure to the decorative paints market total from IRL (Table 60) finds an exterior applied paint tonnage similar to unpublished figure from CEPE.

The final market segment calculations are shown in Table 61.

Table 61 – Decorative Paints Market Segmentation

Market		Proportion	Paint Sales (tonnes)
Interior		73% [b]	3,160,140
	Trade	59%	1,870,743
	DIY	41% [a]	1,289,397
Exterior		27%	1,137,650
	Trade	59%	673,468
	DIY	41% [a]	464,183
Total			4,213,520 [c]

Notes:

a) **Oakdene Hollins (2012) Criteria for Indoor and Outdoor Paints and Varnishes - Ecolabel Background Report**

b) EUROPEAN COATINGS JOURNAL, (2017).

c) **IRL (2014)**

Interior Paints

For interior paints it is assumed that the only pathway to surface water is through the washing of brushes and paint rollers in sinks after use for water based paints. ***As the paint, in its 'wet' form is considered an 'intentionally added' microplastic for the purposes of this project, it is therefore out of scope.*** However, the emission is quantified to provide further context for the microplastics generated from wear.

It is also assumed that professional painters will not wash brushes and rollers and will discard them after use[376,377] therefore only the DIY market is affected. A certain proportion of the paint, when dried on the wall, may also flake off and become part of general household dust. The extent to

---

[375] (2017) ARCHITECTURAL COATINGS MARKET REPORT, *EUROPEAN COATINGS JOURNAL*, No.01

[376] OECD (2009) *Emission Scenario Document On Coating Industry (Paints, Lacquers and Varnishes)*, 2009

[377] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

which this takes place and the amount that would subsequently end up in surface waters is not known. It is also not known what proportion of the interior paint market is comprised of water-based paints, however it will be assumed that this is 100% as a worst case example.

These calculations show that around 3,500 tonnes of paint is expected to be washed into household drains from brush cleaning per year.

Table 62 – Interior Paint Emissions to Surface Water Calculation

Market	Tonnage	Paint Used	Polymer Content	Rinsing Losses
DIY	1,289,397[a]	85% [b]	20% [c]	1.6%
		540,137 t	81,021 t	**3,507t**

Notes:

a) **See Table 61**
b) **Estimated based on several data sources, see main text body.**
c) **CEPE**

A.3.5.3 Marine paints

A 2005 OECD report[378] focuses specifically on anti-fouling paint used in commercial and recreational craft. This report provides figures separately for commercial and recreational craft along with average and worst case scenarios. However, the report does not provide emission factors for weathering during use or during the end of life. As the application of paint and is subsequent spillage directly in the sea is assumed to be considered a spill of 'intentionally added' microplastics these figures are also not used. This emission source is calculated here in order to put the wear-based emissions in context. In the case of commercial application of antifouling paint for maintenance, the report states the worst case emission rate (35%) is a realistic scenario when this is undertaken in a floating dock. The close proximity to the sea for direct emissions appears to be more likely for commercial ships than for recreational craft which can easily be removed from the water and painted inside or on a hard standing. This is not usually possible for larger ships. The end of life removal of paint is given as 100% split between different compartments with no further information to allow disaggregation.

Table 63 and shows the emission ranges from the antifouling OECD report and Table 64 shows the ranges for protective coatings.

Table 63 - Emissions Estimates for Antifouling Coatings

<table>
<tr><th></th><th></th><th>Disposal</th><th>Surface Water</th><th>Land (soil)</th><th>Sewage Treatment</th></tr>
<tr><td rowspan="2">Commercial</td><td>Application</td><td></td><td>7—35%</td><td></td><td></td></tr>
<tr><td>End of Life</td><td colspan="2">0—100%</td><td></td><td></td></tr>
<tr><td rowspan="2">Recreational</td><td>Application</td><td></td><td></td><td colspan="2">2.5—6%</td></tr>
<tr><td>End of Life</td><td></td><td colspan="3">0—100%</td></tr>
</table>

Source: OECD (2005)

Table 64 - Emissions Estimates for Marine Coatings (non-antifouling)

	Surface Water	Land (Soil)	Disposal
Application	1.8%	1.8%	31.5%
Weathering	1%	-	-
End of life	3.2%	3.2%	57.5%
Total	**6%**	**5%**	**89%**

Source: OECD (2009)

Using the emission estimates from the above tables and sales data from CEPE, losses via direct paint emissions during application of the paint are estimated. (Table 65). Although these are considered 'intentionally added' emissions for the purposes of this report, this is used to put the wear emissions into context.

[378] OECD (2005) *Emission Scenario Document on Antifouling Products*, 2005

Table 65 – Calculated Emissions of 'Intentionally Added' Marine Paint Microplastics to Surface Waters

Paint Type		Commercial	Recreational
Antifouling	Solid Paint Used (t)[1]	11,466	6,370
	Losses to SW[3]	7.5—35%	2.5—6%
	Solid Content[2]	75%	75%
	Losses to SW (t)	**645 – 3,010**	**119 - 287**
Marine Protective	Paint Used (t)[1]	64,974	8,190
	Solid Content[2]	75%	75%
	Losses to SW[4]	1.8%	1.8%
	Losses to SW (t)	**877**	**111**
Total 'intentionally added'		**1,752—4,284**	

Notes:
1. **CEPE**
2. **CEPE**
3. **OECD (2005)**
4. **OECD (2009)**

As emission factors for wear and sanding have been provided by CEPE, these are used in preference to the OECD emission factors. The sales figures and assumptions provided by CEPE[379]are shown in Table 66.

The table details the factors that are used to ascertain the total emissions which equal 1,194 tonnes. The data and calculations provided by CEPE found the total to be 716. The difference lies where emission factors have been left out or excluded due to lack of knowledge/data on the subject. Zero rating these emissions could lead to underestimation, therefore figures similar to the other emission estimates have been added. This does not appear to make a large difference to final figure which is in the same order of magnitude.

---

[379] CEPE (2017) *Micro-plastics emitted from 'wear and tear' of dried paints. The view of the paint industry.*, September 2017

Table 66 – Marine Paint Emissions Calculations

Application	Tonnes Sold	Used Paint	Solid Content	Polymer Content	Remaining Polymer at End of Life	Weathering Emission factor	Weathering Emissions (tonnes)	Sanding Emission Factor	Sanding Emissions (tonnes)	Total Emissions (tonnes)
Leisure	**14,560**									
-Professional	**9,464**									
Anti Fouling	3,312	100%	75%	40%	33%	0.5%*	9	1.0%	18	27
Interior Applications	946									
Superstructure	1,107	100%	75%	40%	33%	0.5%	3.04	0.5%*	3.0	6
Hull	4,429									
Above Waterline	*1,329*	100%	75%	40%	33%	0.5%*	4	0.5%*	3.6	7
Below Waterline	*3,100*	100%	75%	40%	100%	0.5%	12	1.0%	23	35
-DIY	**5,096**									
Anti Fouling	3,058	85%	75%	40%	33%	0.5%*	7	25.0%	357	364
Interior Applications	204									
Superstructure	734	85%	75%	40%	33%	1.0%	3.42	1.0%*	3.4	7
Hull	1,101									
Above Waterline	*330*	85%	75%	40%	33%	1.0%*	1.54	12.5%*	19.3	21
Below Waterline	*771*	85%	75%	40%	100%	1.0%	4.91	25.0%	123	128
Commercial	**76,440**									
Anti Fouling	11,466	100%	75%	40%	33%	0.5%*	31.5	1.6%	101	132
Interior Applications	21,441									
Superstructure	8,707	100%	75%	40%	33%	0.5%	23.90	0.08%*	3.8	28
Hull	34,826									
Above Waterline	*8,707*	100%	75%	40%	33%	0.5%	24	0.08%*	3.8	28
Below Waterline	*26,120*	100%	75%	40%	100%	0.5%*	98	1.60%	313	411
Total	**91,000**						**222**		**972**	**1,194**

Source: Data and assumptions provided by CEPE

Where CEPE was unable to supply estimated emission factors these have been estimated using similar factors. These are marked *

A.3.5.4 Road Markings

According to a 2011 report[380] by the Okopol Institute reviewing the impact of a European Directive limiting VOC content on road markings, the most common road markings in Europe are solvent based along with thermoplastic markings—also known as 'hot melt' coatings—where heat is applied to increase the viscosity and allow the coating to be applied to a road surface before drying quickly. Water borne paints and cold plastic road markings are also used to a lesser extent.

As well as polymer binders, a large proportion of the coatings is often comprised of fillers which provide wear resistance (aggregates) and increase tyre grip and reflectiveness (glass beads). As per the definition used in this report, all ingredients additional to the polymer that make up the solid component of the material are considered to be microplastics when they are worn away.

Both the solvent based and waterborne paints dry in a similar way to other paints as the carrier evaporates off. The proportional weight of the solids (the fraction that will dry and remain on the road surface) within the paint vary, but examples of around 50[381,382]—98%[383] have been identified for water borne paints and around 75%[384,385] for solvent based. Cold plastic systems contain around 80—85%[386] solids and thermoplastic systems require heat to make them flow rather than a solvent, therefore they contain 100% solids.

The amount worn away before repainting can be estimated by using guidelines for the renewal of road markings. In the UK the guidelines appear to vary depending upon the responsible authority. However national guidance[387] for highways suggest that a visual wear limit of 70% is achieved before renewal. Several cities[388,389,390] specify that only 30% wear should be evident before renewal—reflecting the increased requirement for highly visible road markings in cities. There are obvious issues with this, as this is a very subjective approach. To combat this, the UK highways guidance has since been updated to use a visual scoring assessment to compare with example pictures. Nevertheless, these figures are useful indicators as to the likely wear that will occur before repainting and may even underestimate the wear due to reports suggesting the condition of road

[380] Okopol Institute (2011) *Report on Potential Scope Extension of the Directive Covering Road Markings*, May 2011
[381] https://www.firwood.co.uk/pdf/TDS2501.pdf
[382] www.swarco.com
[383] Okopol (2009) *Implementation and Review of Directive 2004/42/EC - PART 1: MAIN REPORT, ANNEXES 1-25*, Report for European Commission, November 2009
[384] ibid
[385] www.swarco.com
[386] Okopol (2009) *Implementation and Review of Directive 2004/42/EC - PART 1: MAIN REPORT, ANNEXES 1-25*, Report for European Commission, November 2009
[387] The Highways Agency (2007) *Inspection and Maintenance of Road Markings and Road Studs on Motorways and All-Purpose Trunk Roads*, 2007
[388] http://www.wiltshire.gov.uk/highway-inspection-manual.pdf
[389] https://www.york.gov.uk/download/downloads/id/3326/annex_cpdf.pdf
[390] https://www.hackney.gov.uk/media/2771/highways-maintenance-policy/pdf/Highways-Maintenance-Policy

markings throughout Europe is not satisfactory. A report[391] for the Swedish National Road and Transport Research Institute concluded;

> *"…in most regions, the share of road markings fulfilling the requirements regarding dry road markings in the regulations was less than 50 per cent. For wet-road markings, the corresponding figure was 21 per cent."*

The Road Safety Markings Association from the UK also found that around 40% of markings needed immediate replacement[392]. This suggests that—at least in these two countries—minimum standards for road marking replacement are not being met, and therefore more may be worn off before replacement.

The IRL data for paint demand in Europe does not disaggregate by enough to identify road paint. However, CEPE have provided[393] sales data along with solids content of the different marking materials which is shown in Table 67.

Table 67 – Road Markings Sales in EU28 + NO, CH for 2015

Road Paint Type	Road Paint (tonnes)[1]	Solids Content[2]	Road Paint Applied to Roads (tonnes)
Solvent	60,000	75%	45,000
Water Based	8,000	78%	6,240
Thermoplastics	160,000	100%	160,000
Cold Plastics	30,000	100%	30,000
Total	**258,000**		**241,240**

Notes:
1. **CEPE**
2. **CEPE**

[391] Sven-Olof Lundkvist, Jonas Ihlström, and Mohammad-Reza Yahya (2013) *Condition assessment of road markings 2012 summary of the results from all regions in Sweden*, Report for Swedish National Road and Transport Research Institute, 2013

[392] European Union Road Federation (2014) *Marking the way towards a safer future: An ERF Position Paper on how Road Markings can make our road safer*, 2014

[393] CEPE (2017) *Micro-plastics emitted from 'wear and tear' of dried paints. The view of the paint industry.*, September 2017

CEPE have also provided data on the polymer content of the road markings and, in line with building paints, a degradation factor as the polymer surface oxidises before it is worn off (Table 68).

Table 68 – Road Marking Polymer Content and Degradation

Road Paint Type	Polymer Content	Degradation Rate
Solvent	13%	50%
Water Based	13%	50%
Thermoplastics	16%	64%
Cold Plastics	35%	50%

Table 69Table 13 shows the calculation for the emissions of microplastics from road paints ***at source*** are derived using the previously stated data and assumptions. This leads to an estimated emission of between ***137,000—160,000 tonnes per year***.

Table 69 – Calculating Road Markings Microplastics at Source

		Urban	Rural	Highway	Total Paint Remaining
New Roads[1]	**Upper**	1%	1%	1%	**256,094**
	Lower	15%	15%	15%	**219,878**
Road Type Split[2]		19%	81%	1%	-
Paint Wear[3]		30%	70%	70%	-
Paint Wear at Source	**Upper**	14,434	144,476	1,109	**160,020**
	Lower	12,393	124,045	952	**137,391**

Notes:

4. Upper estimate from derived Eurostat figures for total road lengths (average increase between 2000—2014), the lower is from one data point suggesting that 85% of road markings for German roads are for resurfacing.

5. From Eurostat road length data averaged for seven EU countries

6. From guidance on wear observed before renewal from UK.

A.3.5.5 Automotive Paints

A further source of emissions of paint from wear is also identified in the OECD emissions scenario document[394]. This has yet to be identified as a source of microplastics from any of the existing literature. It provides emissions factors for both new vehicles (OEM) at 3.4% and for the repainting of damaged or crashed vehicles (refinishing) at 6.5%.

Since the publication of the interim report, CEPE have provided their own analysis[395] of this emission source and state:

> *"There is absolutely no evidence for such a high loss, especially when comparing this figure with the 1 % default expectation and the 1 % (1.5 of applied NV) assumption for ships and airplanes."*

CEPE therefore suggest 0.5% as an emission factor, but also include other factors that reduce the emission rate such as;

- only the clear coat is expected to degrade which accounts for 12% of the market, and
- 35% is wasted due to overspray.

Based on this new information the contribution from automotive paint has been revised downward to 98 tonnes (from ~12,000 tonnes) and is no longer considered a significant source.

The calculations for annual paint wear are shown in Table 69 along with the calculations for the deposition environments in Table 70.

Table 70 – Automotive Paint Wear Calculations

	OEM	Refinish
Market Size (tonnes) [a]	339,800	135,920
Clear Coat [b]	12%	12%
Overspray [b]	35%	35%
Solid Content [b]	53%	53%
Lifetime Wear Rate [b]	0.5%	0.5%
Total Wear (tonnes)	70	28
	98	

Notes:
a) **See Table 60**
b) **CEPE**

[394] OECD (2009) *Emission Scenario Document On Coating Industry (Paints, Lacquers and Varnishes)*, 2009
[395] Personal Communication with Jan van der Meulen, CEPE

Table 71 – Deposition of Automotive Paint

	Urban	Rural	Highway
Split [a]	34%	46%	20%
Totals	**33**	**46**	**19**

Notes:
b) **See Table 50 in automotive tyre wear Appendix section for splits.**

A.3.6 Pellets and other Pre-Production Plastics

Pellets are a form of primary microplastic defined in ISO 472:2013 as a "small mass of preformed moulding material, having relatively uniform dimensions in a given lot, used as feedstock in moulding and extrusion operations".[396] They are commonly also known as nibs, nurdles, pre-production plastic pellets, plastic resin pellets and virgin resin pellets. The lentil-sized pellets (usually <5mm) are used as raw material in the production of plastic products and are therefore manufactured and shipped worldwide by the plastics manufacturing and conversion industry.

During this process, many pellets are lost due to spillages and do not get cleaned up (termed "pellet loss"). These pellets can then go on to enter the wider environment through a number of direct and indirect pathways (termed "pellet release"). In these sections, we set out to identify the sources and pathways for pellet loss and release, as well as quantify the scale of such losses in the EU 28. The impacts of pellet release on the environment are beyond the scope of this study but are well documented in other research.

It should be noted that pre-production plastics are also manufactured in the form of flakes, liquids and powders, though the literature and data often do not distinguish between these forms of raw material. Flake and powder are therefore assumed to be included within this analysis of plastic pellets, though they are not explicitly investigated. This is not unreasonable, given that pellets are the most commonly used/ manufactured form of plastic raw material in the EU.

Literature Review

Pellets used in the manufacture of plastics have been identified as a source of marine microplastic pollution since the 1980s.[397] The issue was first investigated as a cause for concern in the USA where the Resin Pellet Task Force was established, and, in the early 1990s, the US Environmental Protection Agency carried out the first study to provide a "comprehensive assessment of the sources, fate, and effects of pellets in the aquatic environment, and to determine what can be done to control and prevent their release to the environment".[398]

A direct result of these findings was the initiation of Operation Clean Sweep (OCS) in 1991, aimed at committing the plastics industry to the total containment, or recapture, of pellets. Since then, the OCS has expanded into an international voluntary programme, adopted as the Zero Pellet Loss initiative in the EU. Neither initiative has published any figures relating to their success, nor are there any industry figures available on the likely magnitude of current pellet loss. Despite this lack of concrete data, it is widely accepted that there has been a decreasing trend in pellet release over the last few decades due to improved handling procedures, though the material continues to be reported and monitored in varying concentrations in marine systems worldwide.

In Europe, reports attempting to quantify pellet loss and release at various points in the plastics value chain have been published in Germany, Denmark, UK, Norway, Sweden and the Netherlands

---

[396] ISO 472:2013 – Plastics - Vocabulary.

[397] A.T. Pruter (1987), Sources, quantities and distribution of persistent plastics in the marine environment, Marine Pollution Bulletin, 1987

[398] United States Environmental Protection Agency (1992), *Plastic Pellets in the Aquatic Environment: Sources and Recommendations*, Office of Water (WH-556F) EPA 842/B-92/010, December 1992

since 2014[399]. In addition, the 'European Coalition to End Plastic Pellet Loss' collates existing knowledge on pellet loss and works to monitor pellet pollution in Italy, France and Belgium[400], and work has been done by the Austrian Environment Agency to quantify pellet pollution flows along the River Danube into the Black Sea.[401] Nurdles have also been reported during beach surveys and surface water sampling in several other Member States, including Greece, Spain, Cyprus[402] and Malta[403]. Therefore, the evidence base supporting concrete estimates of pellet loss and release in the EU is still very much under development, and further research will be required before such an estimate can be derived for each member state. Nevertheless, the existing body of literature on pellet loss and release to the environment is reviewed in these sections, in order to inform the quantification of these microplastic emissions.

Sources and Pathways for Pellet Release

A number of direct and indirect pathways for pellet release to the aquatic environment have been identified in the literature. These include:

1) Direct pellet release due to pellet loss at industrial units located along waterways, port facilities, or due to container loss or spillage at sea; and
2) Indirect pellet release from land based loss of pellets (during handling, transport and waste management along the plastics supply chain) that enter the aquatic environment through waste and storm water systems.

Each of these points of pellet loss and pathways for release in the European context are discussed in the following sections.

Pellet Loss and the Plastic Industry Value Chain

Studies to quantify pellet loss to date have focussed on losses at a maximum of four key points in the plastics industry value chain:

1) **Producers** who create polymers and extrude resin pellets from powders or liquids. Pellets can be lost at this stage due to spills during handling, loading and unloading, as well as leakage from containers and storage silos.
2) **Transporters**, including loading and unloading, or accidental loss from railcars, lorries or shipping containers (due to unsuitable packaging, spills and so on) that transfer pellets from producers to processors. Studies in the literature estimate loss from one such journey. In transit losses are likely to be significant only in the shipping sector, where entire containers f pellets can be lost at sea. With regards to terrestrial transporters, spills have been documented when loading and unloading material from trucks and rail, but as these activities take place within the grounds of the producers, processors, intermediaries and waste managers, there is no need to include a separate category for transport in the calculations. However, it must be noted that transport companies are part of the supply

[399] These are summarised with sources provided in Table 72 below.
[400] http://www.nurdlehunt.org.uk/european-nurdle-hot-spots.html
[401] https://www.bmlfuw.gv.at/wasser/wasserqualitaet/donauplastik2015.html
[402] Cózar A, Sanz-Martín M, Martí E, González-Gordillo JI, Ubeda B, Gálvez JÁ, et al. (2015) Plastic Accumulation in the Mediterranean Sea. PLoS ONE 10(4): e0121762. https://doi.org/10.1371/journal.pone.0121762
[403] http://www.independent.com.mt/articles/2010-05-30/news/alarming-number-of-plastic-nurdles-found-on-maltese-beaches-275269/

chain, and that spills undoubtedly take place in their business operations and so they must be considered in any actions taken to address this issue.

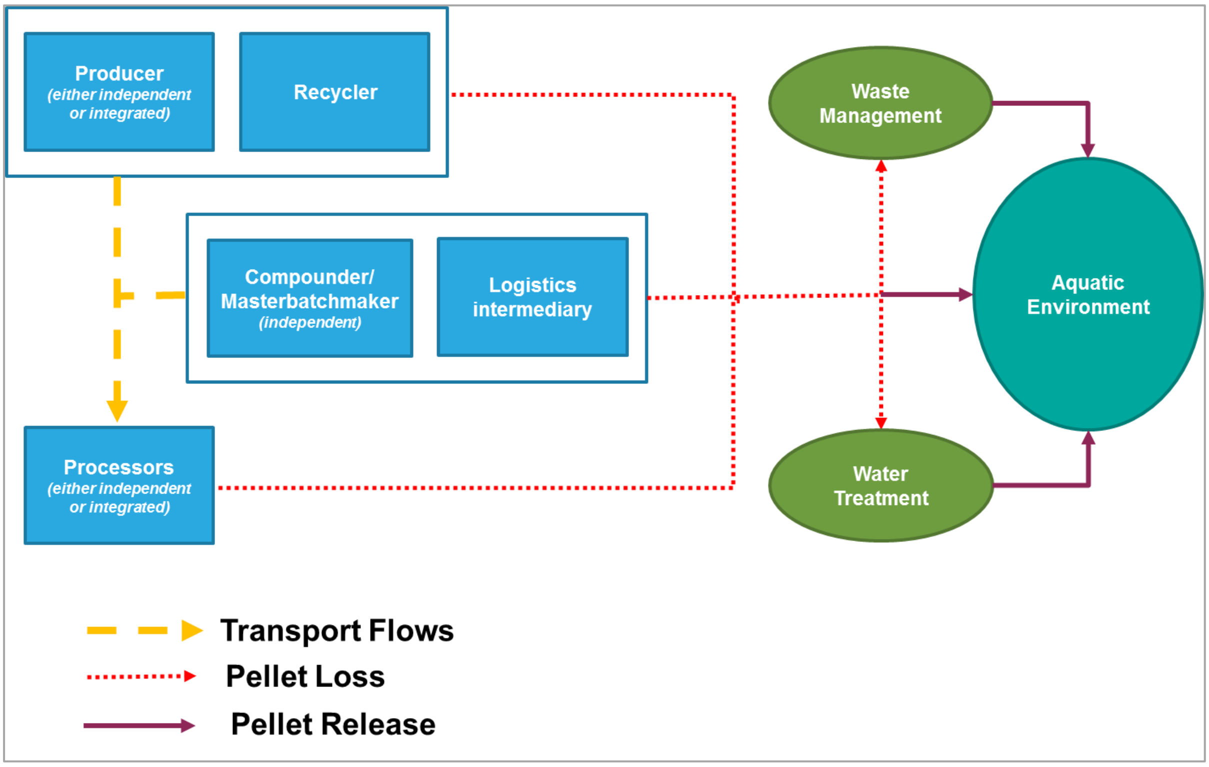
3) **Processors (or converters)**, who melt and remould plastic pellets (usually compounds) into final plastic products, Pellets can be lost at this stage during handling, transfer, storage, and conveyance along the processing line.
4) **Waste management**, at which stage pellet loss mostly occurs during storage for disposal when they are either disposed of with mixed residual waste, or are blown away from bins stored outside. Producers, processors and intermediary facilities typically employ commercial waste management firms to handle their waste, and their pellet waste is most commonly put into a skip. However, skips are not designed to contain such small waste items and have holes in the bottom to let out water, which is a route by which pellets can escape. Pellets can also be spilt when transferring the waste into the skip. At the facility of the processor in question, the waste boxes and bags are put in a single bag, which is sealed and then put in the skip, and sent to landfill.[404]
5) However, it is widely accepted that pellets may be lost wherever they are handled. As such, it is important to understand the wider plastics industry value chain in order to identify all possible points of loss. This wider industry value chain includes:
6) **Compounders/ masterbatch makers** who melt blending plastics with other additives to change the physical, thermal, electrical or aesthetic characteristics of the plastic pellets produced. It is unclear whether these are included as producers or processors in the literature to date. In reality, although the compounding process is often integrated forwards or backwards into the processing or pellet production stages, smaller, independent compounders using virgin resin pellet as a raw material are also active in the supply chain as intermediaries.
7) **Recyclers**, who sort, clean and process waste plastics (predominantly packaging) into recycled plastic pellets and compounds (it is unclear whether or not these are included among pellet producers in the literature to date). Mechanical recycling involves several stages of destructive mechanical process (including grinding, washing, separating, drying, re-granulating and sometimes compounding) during which material losses of powder and shredded plastics varies according to the system in place, but is likely to be significant. Similar to producers, the pellets produced as an outcome of the recycling process can be lost due to spills during handling, loading and unloading, as well as leakage from containers and storage silos. For this reason, and because recyclers have not been identified in prior literature on pellet loss rates, the same assumptions are judged to apply to recyclers as to producers for the purposes of this work.
8) **Logistics suppliers**, providing intermediary services to the stakeholders above, aside from transporters i.e. including warehousing, redistribution, packaging etc. These intermediary points are important as they represent additional stages at which pellets are handled, and can therefore be lost.
9) The wider plastics industry value chain is shown in Figure 24 below. It can be seen that in the most simplified scenario, pellets can transported directly from the producer (with an integrated system for compounding) to the processor. In reality, however, the supply chain is far more complex, with additional stages of warehousing, distribution, port/ depot handling

---

[404] Eunomia (2016), Report for Fidra on *Study to Quantify Pellet Emissions in the UK*, March 2016

and compounding often being undertaken via both terrestrial and water based transportation routes within the EU. Recyclers also have a significant role to play as producers of recycled pellets that then follow the same supply chain as virgin counterparts. Pellets can be spilled at any or all of these handling points in the supply chain, and thereby released in to the aquatic environment directly, or through waste management and water treatment (both wastewater and storm water) systems.

Figure 24: Pellet Flows Across the Supply Chain



Estimation of Pellet Loss Rates

Although the identification of all points of potential pellet loss is important, understanding the quantities of pellets lost at each point in the various stages along the supply chain is even more crucial, and far more difficult. As mentioned above, studies that have attempted the estimation of such loss rates to date have focussed on a maximum of four points in the supply chain. These are discussed in this section, with a comparative summary of findings presented in Table 72 below.

As can be seen from the summary, the pellet loss estimates derived by Mepex in 2014 for Norway (referred to as the Mepex study) and by the Danish EPA in 2015 are the most widely referenced in subsequent work by Eunomia for the EU (2016) and UK (2016), as well as the Swedish EPA in 2016. However, despite the fact that the Mepex and Danish EPA studies represent the most reliable estimates to date, they are based on limited evidence and have limited application in other contexts.

The Mepex study, for example, bases its estimate of the pellet loss from transport on losses of solid powders given in an OECD (2009) report. However, "powders handle very differently to pellets and a much greater rate of loss is expected, as corroborated by the experience of Algalita visiting facilities in California. Mepex states that this emission factor is a worst case scenario for what remains, or gets spilt, from transferring material from different transport containers. The authors found no evidence of the effectiveness of spill control measures for the transferral process, but assumed that 90% of spills would be contained and 10% would be lost to the environment. The basis for the estimate of pellet loss from processors appears to be more reliable. This figure is calculated from measurements of pellets found in effluent from a Norwegian polystyrene plant. However, this only represents data from one specific site." [405]

The Danish EPA study, on the other hand, based its estimates on survey findings of pellet loss from individual processing facilities as reported by members of the Danish Plastics Federation. However, as all respondents were already signed up to Operation Clean Sweep, it was assumed that the 0.001% (of raw materials) loss rate found via the survey represented the lower rate of pellet loss, with a factor ten times higher (i.e. 0.01%) assumed to be the average bound for estimation.

In addition to these five studies using Mepex and Danish EPA estimates of pellet loss, Nova Institut estimated a loss rate between 0.1% and 1% of total European plastics production. However, this estimate was derived based as resource efficiency in production, and the loss rate therefore would also include other forms of process waste (offcuts, pellets discarded due to subpar quality and so on) aside from loss to the environment due to spillage that is not cleaned up. In addition, the papers used as base evidence for these rates do not specify that pellet loss is even a variable in the resource productivity study conducted, as stated in a previous review of the study:

> *There is no indication that pellet loss has been considered when calculating these resource efficiency figures and so there is no justification for even using them as an 'upper bound' estimate of losses (including other losses such as waste). Whilst it is tempting to think that pellet losses could be measured simply by comparing the weight of the material bought to the weight of the final product it is unlikely this would be done with the precision necessary to capture the*

[405] Eunomia personal correspondence with Algalita in Eunomia (2016), *Report for Fidra on Study to Quantify Pellet Emissions in the UK*, March 2016.

likely marginal losses that come about through pellet loss unless this was the specific aim of the monitoring exercise.[406]

Finally, additional estimates have been provided by the Boomerang Alliance in Australia (1% of pellet production lost) and by IUCN (0.000003/0.00001/0.0001 %). However, while the former is a high level estimate and is not based on empirical evidence, the latter wrongly cites Eunomia's previous work as the source of its loss rates and does not provide any further analysis. As a result, neither of these estimates is suitable for application in this study.

Table 72 - Summary of Literature

Author and Year	Area of Study	Estimate of Pellet Loss	Basis of Estimate
OECD (2009)[407]	USA	The emission factor for dust emissions from transferring solid powders is estimated at 5 kg per tonne (0.5%).	This was the default emission factor as found in a previous USEPA (2006) model to estimate dust releases from transferring solid powders, using data from industries including paint and varnish formulation, plastic manufacturing, printing ink formulation, rubber manufacturing, and chemical manufacturing.
Nova Institut (2014) [408]	Germany	0.1 – 1.0% of total plastics production	Estimates of resource efficiency comparing how much raw material is needed to make a tonne of manufactured product.
Mepex (2014) [409]	Norway	0.09% of total plastics production, (0.05% from transport and 0.04% from processors)	The transport estimate is based on the OECD (2009) emission factor for dust emissions from transferring solid powders and an assumption that 10% of this will not be contained by spill control measures. A Norwegian reprocessor provided the estimate of 0.04%.

[406] Eunomia (2016), Report for Fidra on Study to Quantify Pellet Emissions in the UK, March 2016.
[407] OECD (2009) Emission Scenario Document On Adhesive Formulation, 2009
[408] Roland Essel, and et al. (2014) Sources of microplastics relevant to marine protection, Report for Federal Environment Agency (Germany), November 2014
[409] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

Author and Year	Area of Study	Estimate of Pellet Loss	Basis of Estimate
The Danish Environmental Protection Agency (2015) [410]	Denmark	On average 0.01% of raw material consumption at plastics facilities. Maximum 0.0013% of raw material consumption for processors that have joined OCS.	Estimates provided by processors who have joined OCS in a survey undertaken by the Danish Plastics Federation. The figures represent loss to sewage from within the companies' area (incl. unloading from trucks that deliver raw materials). The authors adjust the potential for bias in the provision of this information by assuming the *average* facility will lose ten times as many pellets.
Boomerang Alliance (2015) [411]	Australia	1% of domestic production, relating to a medium scenario of nurdle loss in domestic production and transport.	The source of this estimate is not given in the paper – not based on empirical evidence.
Eunomia (2016) [412]	EU	0.04% losses of domestic production from production, of which 0 – 57% will be captured in waste water treatment. 0.05% losses of domestic production from transport, of which 10 – 50% will be captured in in some way before they reach the oceans.	Both pellet loss figures are taken from the Mepex study. The waste water capture is calculated from 63% of EU population being connected to tertiary waste water treatment. In the best case 90% of microplastics are captured in these facilities and in the worst case, no microplastics captured. Capture of losses from transport is an assumption reflecting the likelihood that pellet spills that occur during transport—especially oceanic—will not be captured in a waste water treatment system

[410] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

[411] Boomerang Alliance (2015), Submission into Senate Inquiry on the Threat of Marine Plastic, October 2015

[412] Eunomia (2016), Report to DG Environment on Study to support the development of measures to combat a range of marine litter sources, January 2016

Author and Year	Area of Study	Estimate of Pellet Loss	Basis of Estimate
Eunomia (2016)[413]	UK	0.001 – 0.01% loss at each stage (four stages studied – producers, processors, storage and transport, offsite waste management).	Loss rates based on Danish EPA (2015). The lower bound of this range assumed that every UK facility loses no more pellets than the Danish processors reported that they lost. The Danish EPA study assumes that the average facility loses ten times more than the best performing, but this provided the highest rate of pellet loss reviewed that could be used in the study. In lieu of better data, and supported by personal communication with a Scottish processor, this estimate was therefore used for the worst performing facility, i.e. the upper bound figure.
Swedish Environmental Protection Agency (2016)[414]	Sweden	Pellet loss calculated at two points – a 0.04% emission factor is assumed from plastic pellet production, and a lower and upper estimate of 0.0005% - 0.01% loss rate is estimated from pellet handling at processors. The latter are estimated as net emission figures (i.e. emissions to the environment).	The pellet loss from production figures are taken from the Mepex study. The handling figure is based on Danish EPA (2015).
IUCN (2017)[415]	Global	Losses are computed at four stages: production of primary plastics, manufacturing of plastics, transport on land (for domestic uses of plastics products) and water (for interregional trade of plastics products), as well as plastic end-of-life. Optimistic/central/pessimistic: 0.000003/0.00001/0.0001 % of microplastics losses per stage	Loss rates are wrongly stated to be based on Fidra 2016. No further basis for the range of loss rates is provided.

[413] Eunomia (2016), Report for Fidra on Study to Quantify Pellet Emissions in the UK, March 2016
[414] Swedish Environmental Protection Agency (2016), Swedish sources and pathways for microplastics to the marine environment, March 2016
[415] IUCN (2017), Primary Microplastics in the Oceans: A Global Evaluation of Sources, February 2017

Market Trends in EU

The European plastics industry is estimated by trade associations to consist of approximately 60,000 companies; including ~1,000 machine manufacturers[416], ~50,000 processors[417], 1000 recyclers[418] and the remainder representing pellet producers.[419] Additionally, 700 compounding sites have also been identified across Europe.[420] These figures appear to represent only those companies that are members of trade associations; the actual number of actors in the market is likely to be larger.

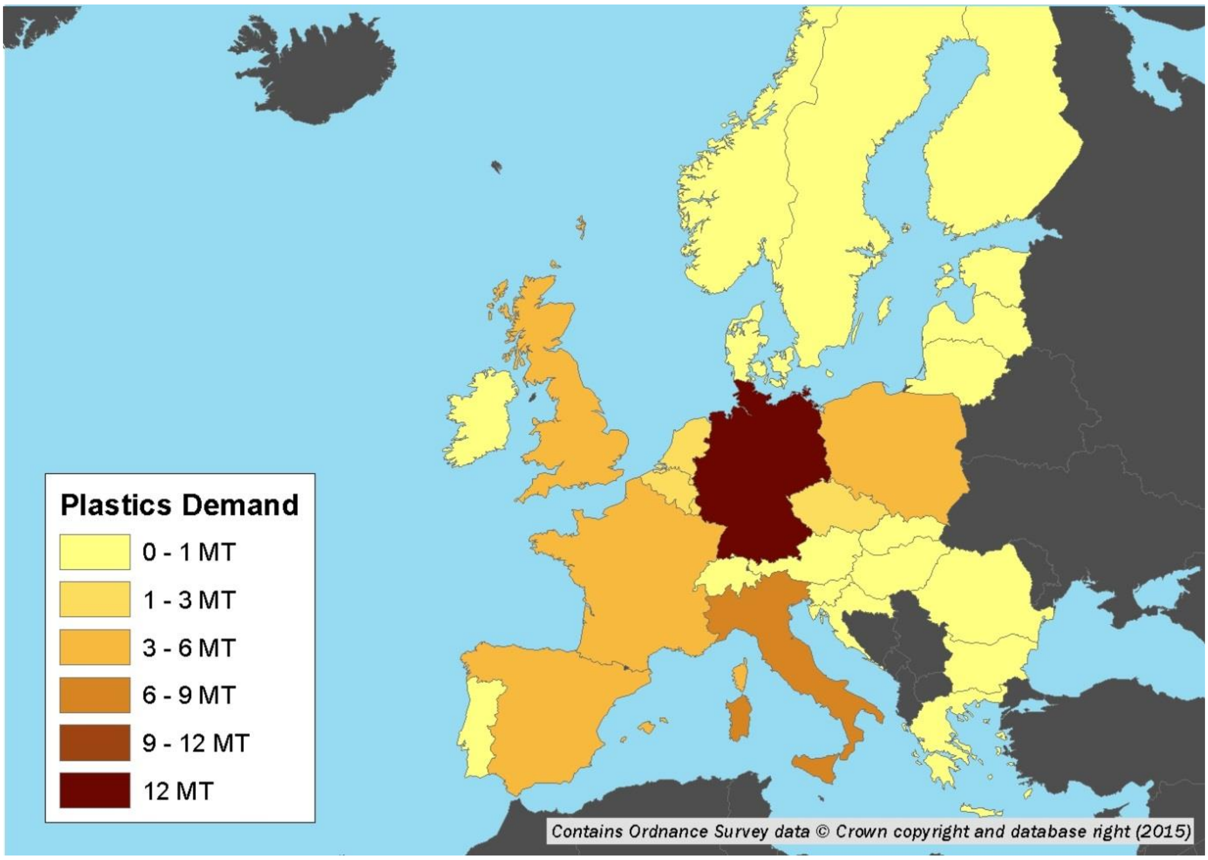
Plastics Europe Data

In 2016, Plastics Europe estimated that 322 million tonnes of plastic materials were produced globally, of which roughly 58 million tonnes were produced in Europe (EU-28 plus Norway and Switzerland)[421].

In addition, the quantity of plastic materials demanded for processing was estimated at approximately 49 million tonnes in Europe. Of this, 70% of demand is also shown to be concentrated in six countries (Germany, Italy, France, Spain, UK and Poland). Extrapolating this data, Figure 25 below provides an indication of the scale of primary plastic demand across all EU member states.

[416] Euromap webpage, *The Industry at a Glance*, accessible at http://www.euromap.org/markets/the-industry-at-a-glance. Accessed on 16th May 2017.
[417] EuPC (Association of European Plastics Converters) webpage, accessible at http://www.plasticsconverters.eu/ . Accessed on 16th May 2017.
[418] Plastics Recyclers Europe webpage, *Facts and Figures*, accessible at http://www.plasticsrecyclers.eu/facts-figures. Accessed on 14th June 2017.
[419] Plastics Europe (Association of Plastics Manufacturers) webpage, *The European Plastics Industry*, accessible at http://www.plasticseurope.org/plastics-industry.aspx. Accessed on 16th May 2017.
[420] 2016, *Applied Market Information (AMI) Directory of Compounders and Master batch Producers* via NetComposites News online, 19th April 2017, accessible at http://netcomposites.com/news/2016/april/19/ami-report-european-compounding-industry-growth-ahead-of-polymer-demand/. Accessed on 16th May 2017.
[421] Includes plastic materials (thermoplastics and polyurethanes) and other plastics (thermosets, adhesives, coatings and sealants). Does not include the following fibers: PET-, PA-, PP- and polyacryl-fibers.

Figure 25: Plastics Demand in Europe 2015



Data Source: Plastics Europe

In terms of recycling, 7.7 million tonnes of plastics were estimated to be recycled (29.7% of 25.8 million tonnes of all EU post-consumer plastic waste in official waste streams) in 2014. Finally, the data shows that significant quantities of plastic manufacturing material were imported into and exported from Europe, resulting in a positive trade balance of more than 16.5 billion euros in 2015.[422] As recycling in the EU is governed by various regulations around reporting, it is assumed that these figures are reasonably accurate.

No data is provided on either the individual contributions of each EU MS to the production or trade figures, or regarding the number of producers, processors, compounders, and recyclers in each MS, or the quantities of primary plastic materials transported via road, rail and sea. These significant data gaps make a precise estimation of the distribution of pellet loss across the EU impossible.

Eurostat Data

Using Eurostat import-export data for the EU 28 (excluding Norway and Switzerland) on a range of CN-codes assumed to represent primary pellets, the positive trade balance with extra-EU countries was found to correspond to total exports of 11.9 million tonnes, and imports of 8.2 million tonnes of plastic pellets from/ to non-EU countries in 2015. This is shown in Table 9-73 below. Eurostat further estimates that 51% of all extra-EU imports are carried by sea, while 48% of such exports were

[422] Plastics Europe (2017), Plastics – the Facts 2016: An analysis of European plastics production, demand and waste data, 2017

represented by shipment.[423] These two pieces of data can be used to derive an estimate of primary plastics shipped for Extra-EU import and export. It should be noted, however, that this estimate excludes shipments of primary plastics within the EU.

Table 9-73: Import/ Export Quantities of Primary Plastics from EU28

Code	Product Description	Import Qty (Kg)	Export Qty (Kg)
3901	Polymers of ethylene, in primary forms	3,207,966,000	2,698,914,000
3902	Polymers of propylene or of other olefins, in primary forms	1,274,360,000	1,697,219,000
3903	Polymers of styrene, in primary forms	470,815,000	774,319,000
3904	Polymers of vinyl chloride or of other halogenated olefins, in primary forms	442,868,000	1,595,716,000
3905	Polymers of vinyl acetate or of other vinyl esters, in primary forms; other vinyl polymers in primary forms	216,079,000	366,479,000
3906	Acrylic polymers in primary forms	300,354,000	884,471,000
3907	Polyacetals, other polyethers and epoxide resins, in primary forms; polycarbonates, alkyd resins, polyallyl esters and other polyesters, in primary forms	1,579,991,000	1,970,118,000
3908	Polyamides in primary forms	274,682,000	535,551,000
3909	Amino-resins, phenolic resins and polyurethanes, in primary forms	201,824,000	1,130,241,000
3910	Silicones in primary forms	37,406,000	124,681,000
3911	Petroleum resins, coumarone-indene resins, polyterpenes, polysulphides, polysulphones and other products specified in note 3 to this chapter, not elsewhere specified or included, in primary forms	175,962,000	193,232,000
Total		**8.18 billion**	**11.9 billion**

Source – Extracted from European Commission Trade Export Helpdesk Statistics accessible at http://www.exporthelp.europa.eu/thdapp/display.htm?page=st%2Fst_Statistics.html&docType=main&languageId=en ; accessed on 16th May 2017.

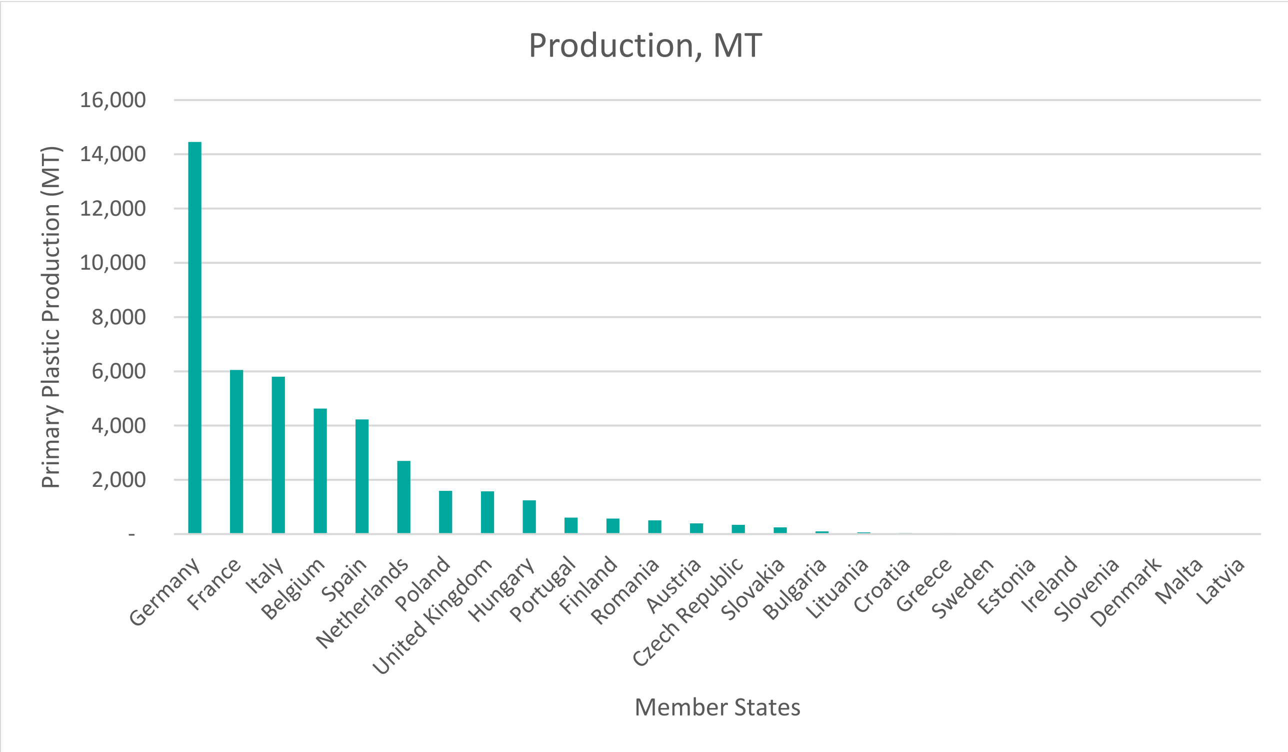
Similar product-code specific data is available on the quantities of primary plastics produced in each EU28 MS in 2015, however, large chunks of this is confidential and has been suppressed.[424] No alternative and easily accessible sources of country level data on pellet production and processing

[423] Eurostat News release 184/2016, *Half of EU trade in goods is carried by sea,* 28 September 2016, accessible at http://ec.europa.eu/eurostat/documents/2995521/7667714/6-28092016-AP-EN.pdf Accessed on 14th June, 2017.
[424] Eurostat External Trade Database (EASY COMEXT Interface) data, 2015, accessed at http://epp.eurostat.ec.europa.eu/newxtweb/. Accessed on 11th May 2017.

quantities were identified. The data available via Eurostat is shown in the chart in Figure 26 below, to give an indication of market share among MSs.

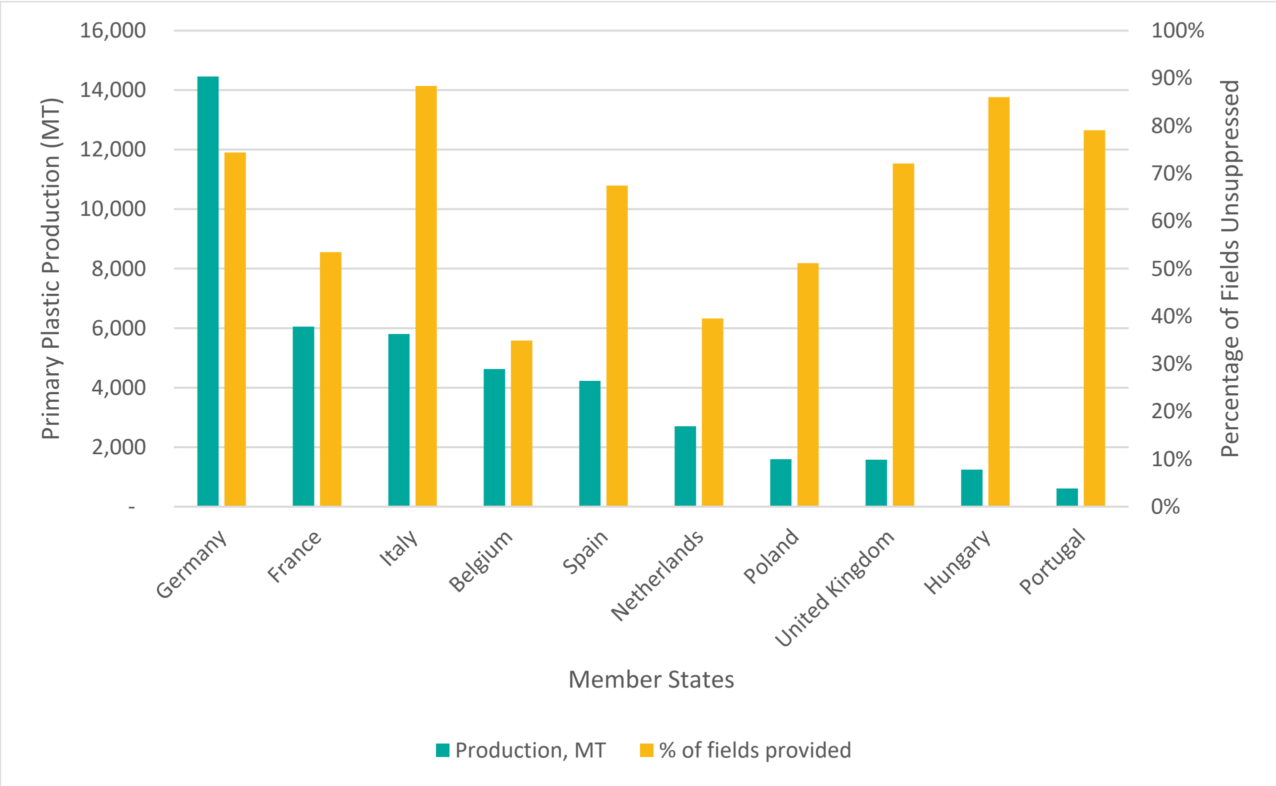
In addition, Figure 27 provides production statistics for the 10 MSs producing the highest tonnages of primary plastics, alongside an indicator of the comprehensiveness of data available for each MS (percentage of data fields unsuppressed). It can be seen that the data is not comprehensive, and therefore cannot be used as a reliable source for quantification. The data indicates a total EU-28 primary plastic production figure of 70.5 million tonnes in 2015, of which only 45.2 million tonnes is accounted for in the supressed MS data. Using the EU total figure together with the import export data in Table 9-73, an estimate can be derived for the total quantity of plastics processed in the EU 28 (production + imports – exports) in 2015 of 66.7 million tonnes.

Figure 26: EU Primary Plastic Production by MS



Source: Eurostat External Trade Database (EASY COMEXT Interface) data, 2015

Figure 27: Primary Plastic Production and Data Reliability for Top 10 MS



Source: Eurostat External Trade Database (EASY COMEXT Interface) data, 2015

Calculation Methodology

Table 6 shows estimates of the losses of pre-production plastics in the EU. The basis for these calculations is outlined below.

Table 74: Annual Losses of Pre-Production Plastics

	Material handled (tonnes)	Loss rate	Qty lost (tonnes)
Producers	58,000,000[a] – 70,565,000[b]	0.010% - 0.040%[c]	5,800 - 28,226
Recyclers	6,896,340 – 7,662,600	0.010% - 0.040%[c]	690 – 3,065
Intermediary Facilities	52,925,399 – 331,283,295[d]	0.010% - 0.040%[c]	5,293 – 132,513
Processors	48,563,380[a] – 66,776,366[e]	0.010% - 0.040%[c]	4,856 – 26,711
Offsite Waste Management	1,079,950 – 9,274,260[f]	0.010% - 0.040%[c]	108 – 3,710
Shipping	10,082,674[g]	0.001% - 0.002%[h]	141 - 225
Total			**16,888 – 194,450**

Notes:
a. **From Plastics Europe (2016) *Plastics – the Facts 2015: An analysis of European plastics production, demand and waste data*. Includes CH and NO.**
b. **From Eurostat External Trade Database (EASY COMEXT Interface) data, 2015**
c. **From Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015**, and Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014.
d. **Based on Plastics Europe (2016) *Plastics – the Facts 2015: An analysis of European plastics production, demand and waste data*, and European Commission Trade Export Helpdesk Statistics accessible**
e. **Based on Eurostat External Trade Database (EASY COMEXT Interface) data, 2015 and European Commission Trade Export Helpdesk Statistics at**
f. **Using material handled tonnages for producers, recyclers, intermediary facilities and processors and an estimate of the proportion of feedstock as waste from Eunomia (2016), *Report for Fidra on Study to Quantify Pellet Emissions in the UK*, March 2016.**
g. **Based on European Commission Trade Export Helpdesk Statistics accessible and Eurostat News release 184/2016, Half of EU trade in goods is carried by sea, 28 September 2016,**
h. Based on Marine Insight (2014) *Survey: How Many Containers are Lost at Sea?*,

The amount of pre-production plastics handled by each of the key groups of company in the supply chain can be estimated with a good degree of confidence. Plastics Europe reports that 58 million tonnes of pre-production plastics were produced in Europe in 2015 and that plastics demand, i.e. the quantity consumed by plastics processors, was 49 million tonnes.[425] Eurostat data indicates that

[425] Plastics Europe (2016) *Plastics – the Facts 2015: An analysis of European plastics production, demand and waste data*. Includes CH and NO but these countries only account for 2% of plastics demand in Europe.

71 million tonnes were produced in the same year, with a demand of 67 million tonnes.[426] These two datasets therefore provide our low and high estimates for the volumes of pre-production plastics produced and demanded within the EU. Plastics Europe also estimates that 7.7 million tonnes of plastics were recycled in the EU in 2014. In the absence of further data, it is assumed that this figure remained constant in 2015 and this forms the upper range of estimation for the volumes handled by recyclers. This is reduced by 10% to provide the lower estimate of 6.9 million tonnes of pre-production plastics handled by producers, under the assumption that up to 10% of the material collected for recycling may be lost during the process of recycling and re-granulation.

The material produced in Europe but shipped out of the EU is likely to have a relatively small proportion of its supply chain within the EU borders. Material that is not exported out of the EU is likely to have a longer supply chain of EU companies and therefore more points of handling where spills and losses can occur. To estimate the quantity of material handled by intermediary facilities we therefore conservatively subtract the export tonnage from production figures. In the low range estimates we conservatively assume that this material is on average handled only once between producers and processors. In the high range estimates, based on Eunomia's understanding of the industry, we assume it may be handled up to five times on average between these points.

Producers, processors and intermediary facilities typically employ commercial waste management firms to handle their waste. The Eunomia study to quantify losses in the UK estimated around 1.3% of material handled at a facility may end up in waste management, based on measurement of pellet spills outside facilities in the US, average throughput at UK facilities and spills indoors at facilities in the US. [427] In the absence of better data for the EU it is assumed that a similar figure applies. A figure of half this value is used for the low range and a figure of 1.5 times this value is applied for the high range estimate. The quantity of material handled by shipping companies is taken from Eurostat, summing data on the quantity of primary plastics imported and exported from the EU28.

Loss rates published in previous studies are known to have issues of reliability and care has been taken to review each source to assess its suitability for use in this study with a preference given to values based on empirical data or reported by facilities. The Danish EPA study into microplastics provides the most reliable estimate of a loss rate from these facilities as it is based on survey responses from plastics facilities that have signed up to OCS. [428] The authors then assume that a facility not signed up to OCS may on average lose as much as ten times more material. In the study into measures to address pellet loss in the UK, Eunomia optimistically assumed that the lower bound of loss rates in the UK would match those in the Danish EPA study. However, pellet loss prevention measures, such as OCS, are much less widely adopted in the European plastics industry and so a similar loss rate cannot be justified in this context.[429] Instead, the loss rate for non-OCS signees of 0.01% is likely to be more representative. The Mepex study into microplastics found a loss rate of 0.04% based on measurements of pellets found in effluent from one Norwegian

---

[426] Based on Eurostat External Trade Database (EASY COMEXT Interface) data, 2015 http://epp.eurostat.ec.europa.eu/newxtweb/ and European Commission Trade Export Helpdesk Statistics accessible at http://www.exporthelp.europa.eu/thdapp/display.htm?page=st%2Fst_Statistics.html&docType=main&languageId=en

[427] Eunomia (2016), Report for Fidra on Study to Quantify Pellet Emissions in the UK, March 2016

[428] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

[429] European Coalition to End Plastic Pellet Loss (2016), Microplastic Pellet Loss: Preventing Pollution Through The EU Plastics Strategy

processor. [430] These two values are therefore taken as the high and low range of loss rates. No studies to date have investigated loss rates at intermediary facilities or offsite waste management facilities and so in the absence of better data the same loss rate is applied.

The Marine Insight website reports that 120 million containers were shipped globally in 2013.[431] Over the preceding six year period 1,679 containers were lost annually on average, but losses increased during the last three years of that period raising the average to 2,683 containers a year. These figures correspond to a loss rate of 0.001% and 0.002% and are used for the low and high ranges of loss rate in the calculations.

[430] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

[431] Marine Insight (2014) *Survey: How Many Containers are Lost at Sea?*, http://www.marineinsight.com/shipping-news/survey-how-many-containers-are-lost-at-sea/

Losses at Member State Level

A Plastics Europe report outlines the level of demand for pre-production plastics, which may be indicative of the distribution of the plastics industry as a whole.[432] Table 75 shows the scale of member state losses of pre-production plastics if they were apportioned on this basis. If this were representative of losses in Europe it is clear that the majority of material would be lost in Germany, Italy, France, Spain, and the UK which represent 64% of total demand in Europe.

Table 75: Losses of Pre-Production Plastics Apportioned on the basis of Demand

Country	% of Total Demand	Losses - Low	Losses - High
Germany	24.6%	4,154	47,835
Italy	14.3%	2,415	27,806
France	9.6%	1,621	18,667
Spain	7.7%	1,300	14,973
United Kingdom	7.5%	1,267	14,584
Poland	6.3%	1,064	12,250
Belgium & Lux.	4.6%	769	8,852
Netherlands	4.2%	703	8,096
Czech Republic	2.4%	401	4,615
Austria	2.1%	355	4,086
Sweden	1.8%	302	3,480
Portugal	1.8%	296	3,405
Switzerland	1.6%	276	3,178
Hungary	1.6%	269	3,102
Romania	1.5%	250	2,875
Greece	1.4%	230	2,648
Finland	1.2%	197	2,270
Denmark	1.1%	191	2,194
Slovakia	0.9%	158	1,816
Bulgaria	0.8%	138	1,589
Ireland	0.5%	92	1,059
Norway	0.5%	79	908
Slovenia	0.5%	79	908
Croatia	0.5%	85	984
Lithuania	0.4%	72	832
Latvia	0.4%	59	681
Estonia	0.3%	46	530
Cyprus & Malta	0.1%	20	227
Europe Total	100%	16,888	194,450

Source: Plastics Europe and own Calculations

[432] Plastics Europe (2016) Plastics – the Facts 2015: An analysis of European plastics production, demand and waste data

A.3.7 Fishing and Aquaculture

Plastic products are common in the fishing and aquaculture sectors. In many ways, these industries are reliant on plastic material to provide affordable, lightweight and durable equipment. As the majority of business activities in these sectors are conducted in the marine environment these products are exposed to weathering conditions from sunlight and salt water that can degrade the plastic and cause it to fragment over time. Degradation factors acting on the plastic include UV radiation, changes in temperature, moisture, oxidation and biological attack. Some products degrade faster than others, particularly where they are also subject to abrasion from contact with the sea floor, movement through the sea, cleaning activities, or, in the case of rope, line and nets, friction from interaction with other gear. These forces undoubtedly degrade the plastic products over time and have the potential to release microplastic particles directly into the marine environment.

Some research suggests that the quantity of microplastic released may not be that significant, at least in certain regions, as the products are replaced before they are too badly degraded.[433] This assumes that the material degrades in a non-linear fashion, that in the initial period very little microplastic is released and that the plastic products are replaced before they reach the point at which the rate of microplastic emissions increases significantly. The way in which weathering degradation factors interact suggests that they may well act in a non-linear fashion. For example, UV radiation is often the dominant weathering factor that degrades plastics outdoors, which in itself has been found to act in a non-linear manner with the greatest impact happening in the first months of exposure in the benthic environment.[434] [435] Stabilisers can be added to plastic material to reduce susceptibility to UV damage. However, additives such as these can be removed by moisture, which attacks the bonds between the polymer and the additive, or by microbiological attack, which targets additives over plastic polymer. Similarly, elevated temperatures and UV radiation accelerate the oxidation process.

Abrasion, when acting alone, is more likely to cause a linear degradation of material. Of course, when combined with weathering factors the net effect will become non-linear, in that as the plastic is weakened by weathering it will become more susceptible to abrasion. It therefore follows that the relative exposure of the plastic material to these different degradation factors will determine how much microplastic is released.

The total quantity of microplastic released by products used in the fishing and aquaculture industries is dependent upon:

- The relative weight of plastic product in use or storage in the marine environment;
- The disposal / replacement rate;
- Where the product is used and stored, as this will affect the exposure to:
    - UV,
    - Other weathering factors discussed above, and

[433] Magnusson et al. (2016) Swedish sources and pathways for microplastics to the marine environment; after Sundt et al. (2014).
[434] ASM International (2003) Characterization and Failure Analysis of Plastics
[435] Welden and Cowie (2017), Degradation of common polymer ropes in a sublittoral marine environment

- Abrasion; as well as
- Susceptibility to degradation factors to which it is exposed.

On this basis, nets pose an elevated risk of degradation due to their size, exposure to weathering from the sea and air, and abrasion in use. Gear used to protect bottom trawl nets, such as dolly ropes and rock hoppers, are also high risk due to the strong friction forces they are subject to as they are dragged over the sea floor. However, it is not clear what portion of the material lost would be classified as microplastics – abrasion in particular could cause larger fragments to be lost.

A recent study investigated degradation of polymer ropes at 10m depth in Scottish coastal waters.[436] A rate of mass loss of 0.4—1% per month was observed which varied depending upon the polymer. The study indicates that polyethylene and polypropylene do not wear as much as nylon. However, the study emphasises that the degradation of marine plastics is highly dependent on the context in which they are found and so it would be unwise to assume that the results are representative of degradation of fishing and aquaculture gear in use. Further research is required in this area in order to inform estimates of microplastic emissions from these sources. Indeed, similar techniques could be applied to measure the rate of plastic degradation and emission of microplastic particles from fishing and aquaculture gear in use and establish the correlation with the principal degradation factors.

There is no scientific basis upon which to estimate of the rate of microplastic emissions from fishing and aquaculture gear as there is little or no empirical evidence and the factors that may cause microplastic emissions exhibit complicated interactions that are likely to cause significant variation in the rate of loss. There is also a lack of data on fishing nets sold, used, discarded and lost. This is compounded by spurious statistics such as a 2009 FAO report[437] repeatedly being quoted as the saying that 640,000 tonnes of fishing gear are lost every year, when this refers to what is currently residing on the sea floor.

Despite this, an attempt has been made to ascertain the magnitude of this issue. Prodcom data suggests that 28,571 tonnes of fishing nets were used (sold minus exports plus imports) in the EU in 2015 (see Table 77). Data for Norway and Iceland is incomplete, but if scaled by reported live catch weight, they account for a further 19,000 tonnes (see Table 76). Swedish estimates for fishing net use stand at 464 tonnes. Scaling the EU production data by catch to Sweden suggests over 2,000 tonnes is used. This would account for the fact that the Swedish estimate is based on fishing gear recovered for recycling and will therefore be underrepresenting the issue.

Using the estimated loss rate from Sweden of 1—10%, a ***total loss of 478—4,780 tonnes*** direct to the ocean is therefore estimated. This estimate is highly speculative however, and both the loss rate and the fishing net data are very uncertain at this stage.

---

[436] Welden and Cowie (2017), Degradation of common polymer ropes in a sublittoral marine environment

[437] UNEP, and FAO (2009) *Abandoned, Lost or Otherwise Discarded Fishing Gear*, 2009

Table 76 – Estimating Fishing Gear Use by Catch Weight

Country	Catch, Live Weight (tonnes) [a]	Proportion	Fishing Gear Use (tonnes) [b]
Belgium	24,463	0.3%	136
Bulgaria	8,747	0.1%	49
Denmark	868,890	10.1%	4,824
Germany	251,268	2.9%	1,395
Estonia	70,753	0.8%	393
Ireland	234,772	2.7%	1,303
Greece	64,431	0.7%	358
Spain	901,512	10.5%	5,005
France	497,435	5.8%	2,762
Croatia	72,264	0.8%	401
Italy	191,634	2.2%	1,064
Cyprus	1,475	0.0%	8
Latvia	81,305	0.9%	451
Lithuania	72,432	0.8%	402
Malta	2,437	0.0%	14
Netherlands	364,990	4.2%	2,026
Poland	187,051	2.2%	1,038
Portugal	185,217	2.2%	1,028
Romania	4,843	0.1%	27
Slovenia	191	0.0%	1
Finland	153,394	1.8%	852
Sweden	202,946	2.4%	1,127
United Kingdom	701,769	8.2%	3,896
Total (EU28)	**5,146,234**	**59.8%**	**28,571**
Iceland	1,317,153	15.3%	7,313
Norway	2,146,074	24.9%	11,915
Overall Total	**8,609,461**	**100%**	**47,798**

Notes:

a) Eurostat

b) Figure for EU28 used to scale by country according to catch and to upscale for Iceland and Norway

Table 77 – Fishing Net Usage in EU 28 (Prod-EXp+Imp)

PRODCOM Classification	Tonnes (EU 27)
13941233 - Made-up fishing nets from twine, cordage or rope of man-made fibres (excluding fish landing nets)	23,685
13941235 - Made-up fishing nets from yarn of man-made fibres (excluding fish landing nets)	4,886
Total	**28,571**

Source: Eurostat

Table 78: Evaluation of Fishing Gear for Relative Risk to Emit Microplastics

Product	Use and storage	Relative abundance, by weight, of gear in use at any one time	Exposed to UV	Exposed to other weathering	Exposed to abrasion	Susceptibility to degradation factors	Typical replacement rate	Relative risk in terms of microplastics
Rope	On deck	Medium	Yes	Yes	Yes	Medium	Unknown	Medium
Nets	In the sea and on deck	High	Yes	Yes	Yes	Medium	Unknown	High
Fishing line	In the sea and on deck	Low	Yes	Yes	Yes	Medium	Unknown	Low
Fishing lures / light-sticks (FADs)	Mostly in the sea	Low	Yes	Yes	Low	Medium	Unknown	Low
Floats	In the sea and on deck	Medium	Yes	Yes	Low	Medium High - if unprotected EPS	Unknown	Medium
Dolly ropes	In the sea and on deck	Low	Yes	Yes	Strong abrasion	Medium	High	Medium
Rock hoppers & similar bottom trawl gear	In the sea and on deck	Low	Yes	Yes	Strong abrasion	Medium	High	Medium
Bait boxes / packaging	On deck	Low	Yes	Yes	No	Medium High - if unprotected EPS	Replaced when bait is used up?	Low

Product	Use and storage	Relative abundance, by weight, of gear in use at any one time	Exposed to UV	Exposed to other weathering	Exposed to abrasion	Susceptibility to degradation factors	Typical replacement rate	Relative risk in terms of microplastics
Other commercial product packaging – films, plastic bottles etc.	On deck	Low	Low – if put in waste containment quickly	Low – if put in waste containment quickly	No	Medium	High	Low
Sails	On deck	High	Yes	Yes	Low	Medium	Low?	Medium
Crab / Lobster pots	Mostly in the sea	Medium	Reduced – as under the sea	Some factors also reduced	Abrasion with sea floor?	Medium	Low – can function for a long time – not a highly technical product	Medium
Plastic sheeting	On deck	Medium	Yes	Yes	No	Medium	Low??	Medium
Boat paint and anti-fouling paint	Mostly in the sea	Low	Reduced – as just under the surface	Yes	High – friction from moving through the sea. Also when cleaned / abrasion blasted.	Medium	High???	Medium
MSW type waste from crew	On deck (inside?)	Medium	No	No	No	Medium	High	Very low

A.3.8 Pathways and Sinks Analysis

A.3.8.1 Waste Water Treatment Retention Rates

Table 79 – Microplastics retention rates applied in country level studies

Study Geography	Retention Rates Applied
Norway (2014)	90%
EU (2015)	0—57% (90%)
Denmark (2015)	>300 µm: 94-97% 20-300 µm: 75-85%
Netherlands (2016)	50% (10—90%)
Ospar (2017) (unpublished)	72%

Table 80 – Sewerage System Types by Country

Country	Combined Sewage Systems	Combined Sewage Overflow (CSO) Releases
UK[438]	70%	-
Norway[439]	14%	5 – 10%
Denmark[440]	38%	4%
Netherlands[441]	70%	0.5%
Sweden[442]	12%	1.53%
Germany[443]	43%	-
France[444]	"majority combined"	5%

[438] Marine Conservation Society (2011) Combined Sewage Overflow Postion Paper
[439] Mepex (2016) *Primary microplastic- pollution: Measures and reduction potentials in Norway*, April 2016
[440] Carsten Lassen *Microplastics. Occurrence, effects and sources of releases to the environment in Denmark.*
[441] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016
[442] Kerstin Magnusson, and et al. (2016) *Swedish sources and pathways for microplastics to the marine environment*, Report for Swedish Environmental Protection Agency, March 2016
[443] Christian Berger, Christian Falk, Friedrich Hetzel, Johannes Pinnekamp, Silke Roder, and Jan Ruppelt (2016) State of the Sewer System in Germany - Results of the DWA survey 2015, *Korrespondenz Abwasser, Abfall*, pp.15–17
[444] M.G. Carleton (1990) Separate and Combined Sewers - Experience in France and Australia, 1990

Table 81 – WWT Plant Microplastic Retention/Emission Studies

Study	Retention Rates	Treatment Level	
Leslie et al., 2017[445]	72%	Tertiary	7 Plants in Netherlands
MST Microplastic in Danish wastewater, 2017[446]	99.7%	Tertiary	10 Plants in Denmark
Ziajahromi et al., 2017[447]	>90% 29% 17%	Tertiary Secondary Primary	3 plants in Australia
Mintenig et al., 2017[448]	97%	Tertiary	1 plant in Germany
Michielssen et al. 2016[449]	99% 96%	Tertiary Secondary	3 plants is USA
Murphy et al., 2016[450]	98% 78%	Secondary Primary	1 plant in UK
Carr et al., 2016[451]	90%	tertiary	7 plants in USA
Talvitie et al., 2015[452]	99.8%	Tertiary	1 plant in Finland

Table 82 shows the maximum and minimum retention rates that are used to model the European retention rates. Although these studies use the retention rate by number of particles it is also

---

[445] Leslie, H.A., Brandsma, S.H., van Velzen, M.J.M., and Vethaak, A.D. (2017) Microplastics en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota, *Environment International*, Vol.101, pp.133–142

[446] The Danish Environmental Protection Agency (2017) *Microplastic in Danish wastewater Sources, occurrences and fate*, March 2017

[447] Ziajahromi, S., Neale, P.A., Rintoul, L., and Leusch, F.D.L. (2017) Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics, *Water Research*, Vol.112, pp.93–99

[448] Mintenig, S.M., Int-Veen, I., Löder, M.G.J., Primpke, S., and Gerdts, G. (2017) Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging, *Water Research*, Vol.108, pp.365–372

[449] Michielssen, M.R., Michielssen, E.R., Ni, J., and Duhaime, M.B. (2016) Fate of microplastics and other small anthropogenic litter (SAL) in wastewater treatment plants depends on unit processes employed, Vol.2, No.6, pp.1064–1073

[450] Murphy, F., Ewins, C., Carbonnier, F., and Quinn, B. (2016) Wastewater Treatment Works (WwTW) as a Source of Microplastics in the Aquatic Environment, *Environmental Science & Technology*, Vol.50, No.11, pp.5800–5808

[451] Carr, S.A., Liu, J., and Tesoro, A.G. (2016) Transport and fate of microplastic particles in wastewater treatment plants, *Water Research*, Vol.91, pp.174–182

[452] Talvitie, J., Heinonen, M., Pääkkönen, J.-P., Vahtera, E., Mikola, A., Setälä, O., and Vahala, R. (2015) Do wastewater treatment plants act as a potential point source of microplastics? Preliminary study in the coastal Gulf of Finland, Baltic Sea, *Water Science and Technology: A Journal of the International Association on Water Pollution Research*, Vol.72, No.9, pp.1495–1504

recognised that weight may play a significant part in microplastic retention. However, without a further research into the full range of densities, sizes and shapes it is not practicable to include this level of complexity. These are combined with Eurostat data on the proportion of the population that are connected to WWT types. The results of this are shown in Table 83 for all countries.

Table 82 – Maximum and Minimum Microplastics Retention Rates Observed in Literature (by number)

	Primary	Secondary	Tertiary	Not Specified	Independent	Truck Transport	Unknown/ no treatment
Max Retention Rates	78%[1]	98%[1]	99.7%[2]	50%[5]	50%[5]	50%[5]	0%[6]
Min Retention Rates	17%[3]	29%[3]	72%[4]	50%5	50%[5]	50%[5]	0%[6]

Notes:
1. **Murphy et al., 2016*[453]***
2. **MST Microplastic in Danish wastewater, 2017*[454]***
3. **Ziajahromi et al., 2017*[455]***
4. **Leslie et al., 2017*[456]***
5. **A default value of 50% is used for treatment with no associated data. This accounts for 12% of the EU population.**
6. **A default value of no capture is assumed which accounts for around 9% of the EU population.**

[453] Murphy, F., Ewins, C., Carbonnier, F., and Quinn, B. (2016) Wastewater Treatment Works (WwTW) as a Source of Microplastics in the Aquatic Environment, *Environmental Science & Technology*, Vol.50, No.11, pp.5800–5808

[454] The Danish Environmental Protection Agency (2017) *Microplastic in Danish wastewater Sources, occurrences and fate*, March 2017

[455] Ziajahromi, S., Neale, P.A., Rintoul, L., and Leusch, F.D.L. (2017) Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics, *Water Research*, Vol.112, pp.93–99

[456] Leslie, H.A., Brandsma, S.H., van Velzen, M.J.M., and Vethaak, A.D. (2017) Microplastics en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota, *Environment International*, Vol.101, pp.133–142

Table 83 - European WWT Coverage and Estimated Average Microplastic Retention Rates

P= primary, S=secondary, T=Tertiary, NS=Not Specified, IND= Independent, TT=Truck Transported, UKN=Unknown/no treatment,

Country	Proportion of Pop Covered by WWT Type (Eurostat)								Average Microplastic Retention Rate by WWT Type and Coverage							
	P	S	T	NS	IND	TT	UKN	Σ	P	S	T	NS	IND	TT	UKN	Avg.
Maximum Retention Rates by Technology									78%	98%	99.7%	50%	50%	50%	0%	-
Minimum Retention Rates by Technology									17%	29%	72%	50%	50%	50%	0%	-
Netherlands	-	0%	99%	-	1%	-	-	100%	-	0.2%	85.1%	-	0.3%	-	-	86%
Malta	-	6%	92%	-	-	1%	-	100%	-	4.1%	79.1%	-	-	0.7%	-	84%
Germany	-	3%	93%	1%	3%	-	-	100%	-	1.9%	79.8%	0.5%	1.5%	-	-	84%
Austria	-	1%	94%	-	6%	-	-	100%	-	0.6%	80.3%	-	2.8%	-	-	84%
Switzerland	-	11%	87%	-	2%	-	0.3%	100%	-	7.0%	74.7%	-	0.9%	-	-	83%
Denmark	1%	2%	88%	-	9%	-	-0.1%	100%	0.4%	1.2%	75.7%	-	4.6%	-	-	82%
Sweden	-	4%	83%	-	13%	-	-	100%	-	2.5%	71.3%	-	6.5%	-	-	80%
Finland	-	-	83%	-	17%	-	-	100%	-	-	71.3%	-	8.5%	-	-	80%
Luxembourg	2%	27%	70%	-	2%	-	-	100%	0.9%	16.8%	59.9%	-	0.9%	-	-	79%
Greece	-	6%	86%	-	-	-	7.9%	92%	-	4.0%	73.7%	-	-	-	-	78%
Spain	1%	28%	67%	2%	1%	-	1.3%	99%	0.3%	17.8%	57.3%	1.2%	0.5%	-	-	77%
Belgium	-	11%	73%	-	11%	-	5.0%	95%	-	7.0%	62.7%	-	5.5%	-	-	75%
UK	0%	50%	50%	-	-	-	0.4%	100%	0.0%	31.5%	42.8%	-	-	-	-	74%
Estonia	0%	5%	77%	-	5%	-	12.8%	87%	0.0%	3.2%	66.2%	-	2.5%	-	-	72%
Norway	19%	1%	61%	-	15%	-	3.0%	97%	9.2%	0.9%	52.5%	-	7.6%	-	-	70%
Poland	0%	14%	56%	-	-	25%	4.8%	95%	0.0%	9.0%	48.1%	-	-	12.5%	-	70%
Czech Republic	0%	8%	72%	-	2%	-	17.6%	82%	0.1%	5.2%	61.5%	-	1.2%	-	-	68%
Latvia	4%	50%	17%	0%	29%	-	-0.1%	100%	1.8%	31.8%	14.8%	0.2%	14.5%	-	-	63%
Lithuania	-	2%	61%	11%	-	7%	18.8%	81%	-	1.5%	52.1%	5.5%	-	3.6%	-	63%
France	1%	33%	22%	25%	19%	-	-	100%	0.4%	21.1%	19.0%	12.6%	9.3%	-	-	62%
Ireland	-	47%	18%	-	31%	-	4.0%	96%	-	29.8%	15.5%	-	15.5%	-	-	61%
Hungary	0%	16%	57%	-	-	-	27.3%	73%	0.0%	10.2%	48.5%	-	-	-	-	59%
Cyprus	-	12%	18%	-	70%	-	-	100%	-	7.3%	15.7%	-	35.1%	-	-	58%
Slovenia	1%	33%	22%	-	35%	-	9.4%	91%	0.2%	21.1%	18.6%	-	17.6%	-	-	58%
Bulgaria	2%	19%	35%	-	25%	-	18.2%	82%	0.9%	12.3%	30.3%	-	12.7%	-	-	56%
Croatia	16%	36%	1%	-	45%	-	1.7%	98%	7.6%	22.8%	0.9%	-	22.7%	-	-	54%
Slovakia	-	-	-	62%	36%	-	1.6%	98%	-	-	-	31.0%	18.2%	-	-	49%
Italy	3%	22%	35%	-	5%	-	35.0%	65%	1.4%	14.0%	30.0%	-	2.5%	-	-	48%
Portugal	4%	39%	16%	11%	-	-	29.3%	71%	1.7%	25.0%	14.1%	5.7%	-	-	-	46%
Romania	9%	18%	18%	-	4%	-	51.3%	49%	4.2%	11.1%	15.5%	-	2.2%	-	-	33%

A.3.8.2 Storm Water Retention Rates

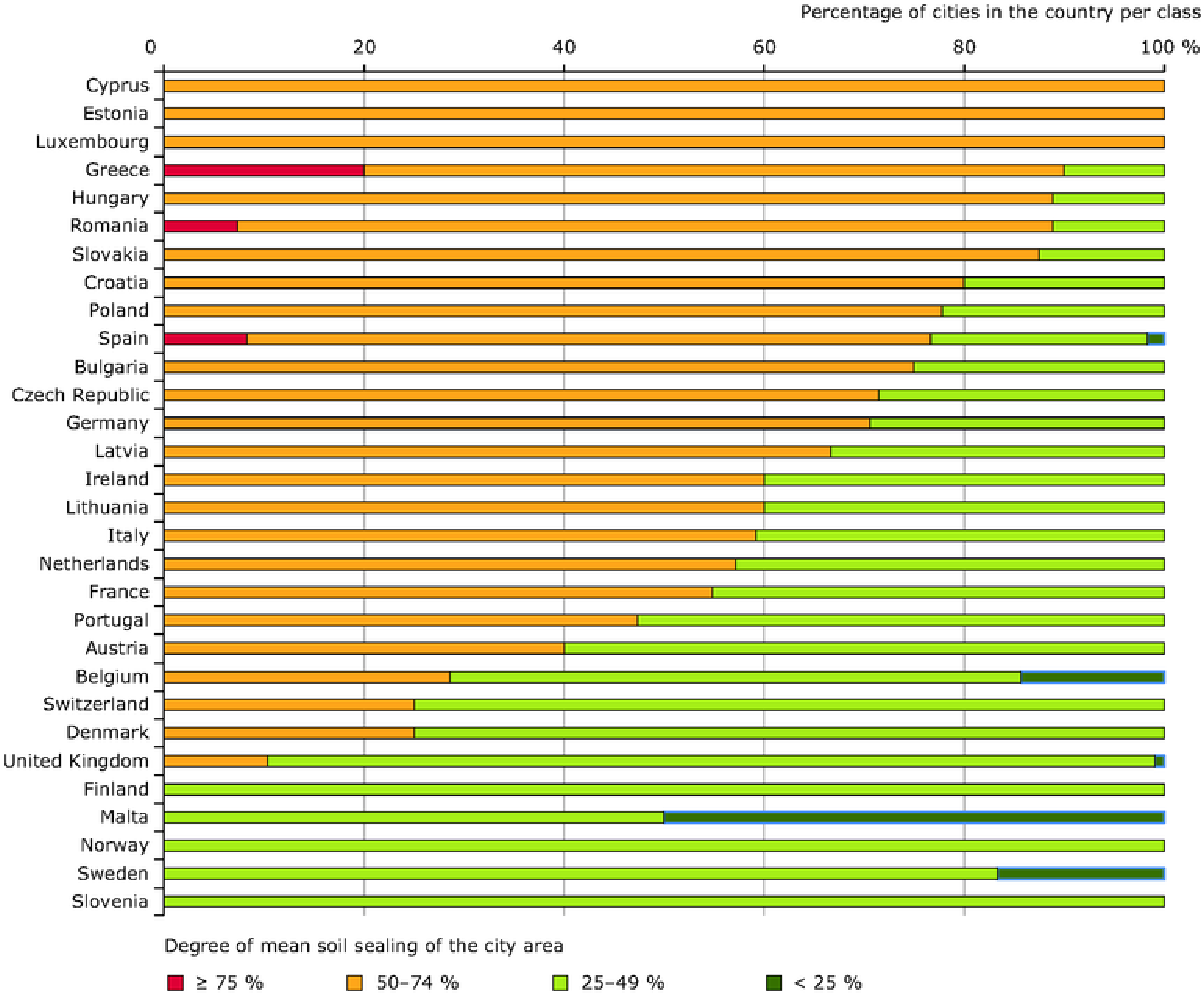
Table 84– Storm Water Treatment Suspended Solids Retention Rates

Run-off Treatment	Suspended Solids Retention Rates
Wet Pond	80%
Wetland	85%
Swale/ Open Ditch	70%
Infiltration trench	90%
Storm drainage filter	5.5%
Retention basins	75 - 80%
Bio-retention filter	80%
Permeable pavement	90%

Source: http://www.stormtac.com/index.php Stormtac database updated 19/3/2017

A.3.8.3 Soil Sealing

Figure 28 - Degree of mean soil sealing in City Area in Europe



Source: European Environment Agency (2012)[457]

[457] https://www.eea.europa.eu/data-and-maps/figures/degree-of-mean-soil-sealing

A.3.8.4 Urban, Rural and Highway Road Pathways

It is also important to determine whether there are difference in microplastic emission fates depending upon where they are emitted. For this study, five key emission scenarios are created to demonstrate this;

- Residential wastewater (foul water);
- Urban road run-off;
- Rural road run-off;
- Highway road run-off; and,
- Non-road run-off.

Residential waste water is simply water that is washed away in households directly down the drains and is mostly send directly to WWT plants. Non-road drains are similar, but include some form of sedimentation device. Urban, rural and highway run-off are different in that the emission sources of the microplastics are much dispersed and therefore they will not all be washed into the sewerage system. As previously discussed, some will be captured in porous asphalt or in road cleaning, but most will either enter some form of storm management where they may settle out, or they will become part of the nearby soil. This can happen either by rainfall run-off or windblown.

Although several microplastic emissions studies have attempted to estimate this, there are no formal methods or models for doing so, therefore each approach is different. A more complete discussion on the subject is provided in Appendix A.3.7. The following is a summary of the main data sources used to estimate where microplastics are expected to go after their emission.

Dutch microplastics emissions[458] estimates are based upon their national emissions inventory for road traffic wear.[459] Table 85 shows the estimated pathways for tyre wear particles from this study which is largely based on expert judgement assumptions.

Table 85 - Netherlands Emission Inventory distribution percentages for tyre particulates to compartments

Geography	Soil	Surface Water	Sewers
Urban	40%	-	60%
Rural	90%	10%	-
Highway	90%	10%	-

Source: TNO (2016)

[458] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016
[459] Deltares, and TNO Consulting (2014) *Emissieschattingen Diffuse bronnen Emissieregistratie: Bandenslijtage wegverkeer*, Report for Rijkswaterstaat - WVL, May 2014

A 60%—40% allocation to soils and sewers respectively is applied to the coarse fraction of tyre wear generated on urban roads. The study used a GIS overlay analysis for the Netherlands to establish that 50% of the land connected to sewerage is also paved and this is used as a starting point for the distribution between sewers and soil. They also highlight several factors which would increase or decrease the chance of a particle ending up in sewers—shown in Table 86—and conclude that 60% will end in sewers (as a provisional value). In this present study street cleaning is taken into account (see Section 2.2.8.4) therefore this factor can be discounted. The evaporation of water will occur, but in urban environments there is not expected to be many places for particles to travel to outside of the road or nearby paved areas. The particles could also be washed into sewers during the following rain events. Around 50% of European urban areas are also known to be soil sealed[460] (i.e. impermeable services) therefore there is no soil in these areas for microplastics to settle on. The other 50% is intuitively more likely to comprise of residential gardens and parklands rather than areas immediately adjacent to the roadside. The intensifying factors of the proximity of the source to the drains and leaching into sewers are therefore judged to be far more influential.

Table 86 – Factors Affecting Particle Transport by Storm Water

Factor	Chance of ending up in sewers
Some water evaporates	⇩ Decreased
Particles are deposited close to or on roads	⇧ Increased
Street cleaning	⇩ Decrease
Leaching into sewers	⇧ Increase

Source: Adapted from TNO (2016)

The rural pathways are the least understood—not least because the definition of rural can encompass a large variety of conditions—but, there is no real data to support the assumption that 10% of microplastic will end up in surface water.

TNO applied the same factor for rural and highway, although it is clear from the research for this study that highways are far more likely to include storm drains and water management strategies. Because of this, the emission distribution is judged to be more akin to the urban environment, albeit with more opportunity to be blown into nearby soils.

Based on the preceding discussion, a revised distribution is therefore proposed in Table 87

Table 87 - Distribution percentages for Microplastics to compartments

Road Geography	Soil	Surface Water	Sewers
Urban	30%	-	70%

[460] https://www.eea.europa.eu/data-and-maps/figures/degree-of-mean-soil-sealing

Rural	80—90%	10—20%	-
Highway	40%	-	60%

A.3.8.5 Road Cleaning

As there is no data available for the potential for road sweeping to capture tyre-wear derived microplastics, a set of variables was created and likely figures inputted to demonstrate the potential impact road sweeping might have under these assumptions. Figures were chosen based on consultation with local experts, but with an emphasis on overestimating the factors in order to show a 'best case' scenario for the amounts of particular matter that are gathered from road sweeping.

Table 88 outlines the assumptions applied for estimating tyre wear captured by road sweeping, and Figure 29 details the calculation.

It was assumed that rural roads are never swept/cleaned, urban roads with a high footfall would be cleaned regularly and that all highways are cleaned at some point in the year. Urban streets with high footfall were assumed to represent 10% of total urban roads.

Next, to derive the portion of wear that is removed by rainfall on roads that are swept an estimated average number of rainfall days per year for Europe was derived. This was achieved by averaging data for the seven countries within which we estimated the most tyre-wear derived microplastics to be produced[461]. It was assumed that on rainfall days no road sweeping targeting dust occurs, as rainfall is effective in suppressing road particulate matter[462], and that 100% of dust is transferred to roadside runoff capture mechanisms by runoff.

To calculate what portion of the dust deposited on dry days is captured in road sweeping an estimate was made as to the frequency of sweeping. Urban roads were assumed to be swept six days a week and all highways were assumed to be swept once per year. Next, a factor for the efficiency of mechanical street sweepers in removing road dust was derived from the literature. Applying the fraction of wear deposited during dry days to the percentage of days in a year that cleaning occurs and the efficiency of road sweeping machines provides as estimate of the wear captured by road sweeping on the day it is deposited.

This leaves a fraction of wear that is not captured due to the lower than 100% efficacy of road sweeping machines and the probability that some dry days will coincide with days when cleaning is not being carried out. However, it is assumed that ultimately 100% of coarse tyre-derived microplastics emitted in a year are removed from the road surface by either rainfall runoff or road sweeping. Therefore the remaining fraction is divided between the two removal mechanisms, according to the percentage they transport of the wear that is removed on the day it is deposited.

The results of 4.7% capture on urban roads and 0.3% on highways by road cleaning demonstrates that even assuming an optimistic scenario our modelling suggests that sweeping activities capture only a small proportion of total European emitted tyre-wear derived microplastics. Even when assuming extremely high values for key factors, for example that urban roads which are swept are swept every day of the year, that all highways in Europe are cleaned once per month and that rainfall runoff removal of wear is only 50% on rainfall days, the capture rate on both urban and highway roads does not exceed 8%.

---

[461] Excluding Poland for which data was unavailable; Italy, the UK, France, Germany, Spain and The Netherlands
[462] Amato, F. The scientific basis of street cleaning activities as road dust mitigation measure

Figure 29: Equation for Calculating the Fraction of Urban Wear Captured by Road Sweeping

$$\frac{A - (A \times B) \times C \times D}{(A - (A \times B) \times C \times D + A \times B)} \times A = E$$

Table 88: Capture by Road Sweeping (Fraction of total emissions)

	Urban	Highway
Wear Available for Capture by Road Sweeping		
% of roads that are regularly swept[1 A]	10%	100%
Wear Transported from Roads that are Swept by Rainfall Runoff		
Percentage of days in a year that are rainfall days[1 B]	33%	
Wear Captured by Road Sweeping Activities on Roads that are Swept		
Percentage of days in year that road sweeping occurs[2 C]	85.5%	0.3%
% efficiency of material capture by road sweeping[3 D]	51%	
Capture by Road Sweeping Assuming that 100% of Deposits are Removed Each Year		
Fraction of wear that is removed from roads that are regularly swept by rainfall runoff	53%	99.7%
Fraction of wear that is removed from roads that are regularly swept by road sweeping	47%	0.3%
Wear removed by rainfall runoff	5.3%	99.7
Wear captured by road sweeping [E]	4.7%	0.3%

Notes:

1. **Derived from data from national meteorological agencies for Italy, the UK, France, Germany, Spain and The Netherlands collated on www.currentresults.com .**
2. **Opinion of local experts on road sweeping; Urban roads 6 days a week and highways once a year.**
3. **Derived from literature reviews of the effectiveness of road sweeping machines carried out by**
   a. ***Amato, F. The scientific basis of street cleaning activities as road dust mitigation measure***
   b. ***Calvillo, S.J., Williams, E.S., and Brooks, B.W. (2015) Street Dust: Implications for Stormwater and Air Quality, and Environmental Management Through Street Sweeping, in Whitacre, D.M., (ed.), Reviews of Environmental Contamination and Toxicology Volume 233 (2015) Cham: Springer International Publishing, pp.71–128***
   c. ***Mepex (2014) Sources of microplastic pollution to the marine environment, Report for Norwegian Environment Agency, April 2014***
4. ***Note that figures will not add up in all instances due to rounding of decimal place figures.***

A.3.8.6 Road Deposition Pathways Background

Introduction

Many tyre wear microplastic studies identify potential pathways for tyre wear leaving the road environment, but only four national- to global-scale reports reviewed go beyond pathways identification to attempt comparatively detailed quantification of mass flows in either relative or absolute terms to a range of environmental compartments. These are the 2015 Danish study[463], the 2016 Dutch study[464], our previous 2016 study for DG Environment[465] and the 2017 IUCN report.[466]

Tyre wear is generally assumed to be distributed to a different profile of pathways and in varying quantities dependent on whether it is generated on urban or rural roads, with occasional further differentiation to take account of differences for highway roads. Of the attempts to quantify pathway mass flows, the Danish report stands out as the only study not to use figures derived using an emissions approach in modelling, nor to apply a different set of pathways dependent upon the road type that wear is generated on.

Due to these differences in approach, and to introduce some of the challenges characteristic of efforts to quantify mass flows of tyre wear from the road, the pathways assumptions of the Danish report will be reviewed first. Subsequently, the methodologies of reports identifying pathways for urban, rural and highway driving separately are reviewed.

The Danish Study Approach

The Danish inventory of the occurrence, effects and sources of microplastics performed by Lassen et al. in 2015 assumes that tyre wear is transported from the road environment either by wind or in rainwater runoff. The author's estimate a range of distribution factors for different areas depending on the coverage of sewerage systems and apply these factors to their estimates of emissions at source calculated using a sales approach. A key assumption of this approach is that area coverage of sewage system types accurately reflects the proportion of total driving taking place in these areas, which ultimately determines tyre wear emission. They note that of the paved road area in Denmark, 38% is connected to a combined sewer system, 45% is connected to a separate storm water system and 16% is not connected to any form of sewerage or storm water system.

For areas without sewerage connection, it is assumed that 95-98% of emissions will be distributed to soil (by what mechanism is not mentioned) and the remaining 2-5% to surface waters by wind. This assumption does not appear to be literature based. For areas with any sewage connection the authors adopt an assumption that 50-70% of emissions on roads are distributed to soil while the remaining 30-50% reaches sewers, citing a 2002 study investigating sources of heavy metals in

---

[463] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

[464] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

[465] Eunomia Research & Consulting (2016) *Study to support the development of measures to combat a range of marine litter sources*, Report for European Commission DG Environment, 2016

[466] IUCN (2017) *Primary Microplastics in the Oceans: a Global Evaluation of Sources*, Report for International Union for Conservation of Nature, 2017

urban wastewater in Stockholm.[467] This source adapted the finding of another paper[468] that 37% of the expected Zinc load in the runoff waters from a major rural highway was actually observed, which led to the conclusion that;

"37% of Zinc [is] removed by runoff water, with the remainder being dispersed in the atmosphere as dust."

It is worth noting that the expected load of Zinc was estimated for all sources on the road, including brake wear and erosion of the road barriers, of which tyre wear made up less than half. This calls into question the legitimacy of applying a 37% removal by runoff to tyre wear only as it is not known whether contributions from other sources could be relatively under- or over-represented in runoff.

This overview of the approach of Lassen et al. highlights an important point about the derivation of distribution factors of tyre wear to pathways through the environment; that the body of literature interrogating tyre wear microplastics emissions is a relatively young, still-developing body of literature. As a result key parameters, such as factors for distributing wear between transport mechanisms, are frequently based on best guesses and informed assumptions due to a lack of empirical data. Alternatively, proxies for the environmental fate of tyre wear, such as knowledge of the distribution of heavy metals, especially Zinc, are used in the absence of sufficient measured data. This lack of empirical data specific to tyre wear, however, need not preclude a pragmatic approach such as Lassen et al*.* have adopted whereby the best available data is applied to establish likely orders of magnitude of mass flows with an acknowledgment of the scope for improvement in the future.

Urban driving

Pathways identification

Figure 30: illustrates the pathways that tyre wear produced in urban road environments may take through the environment according to the existing literature on tyre wear microplastics.

There is general agreement across the literature addressing tyre wear microplastics that the key mechanism by which tyre wear is removed from the urban road environment are suspension in rainfall run-off and in wind-blown air. These pathways are assumed in national studies for Sweden[469], the Netherlands[470] and Norway,[471] as well as by studies attempting to identify pathways applicable across Europe[472] and the world.[473]

---

[467] Sörme, L., and Lagerkvist, R. (2002) Sources of heavy metals in urban wastewater in Stockholm, *Science of The Total Environment*, Vol.298, Nos.1–3, pp.131–145

[468] Legret, M., and Pagotto, C. (1999) Evaluation of pollutant loadings in the runoff waters from a major rural highway, *Science of the Total Environment*, Vol.235, No.1, pp.143–150

[469] Kerstin Magnusson, and et al. (2016) *Swedish sources and pathways for microplastics to the marine environment*, Report for Swedish Environmental Protection Agency, March 2016

[470] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

[471] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

[472] Eunomia Research & Consulting (2016) *Study to support the development of measures to combat a range of marine litter sources*, Report for European Commission DG Environment, 2016

[473] IUCN (2017) *Primary Microplastics in the Oceans: a Global Evaluation of Sources*, Report for International Union for Conservation of Nature, 2017

Regarding the destination of rainfall run-off there is discrepancy in the literature. Although most studies allocate some fraction of run-off to urban roadside soils, the 2017 IUCN report assumes that 100% of urban run-off is transported to some sort of sewerage system. While many models do not describe further pathways steps for tyre wear spread to urban soils by run-off, the Netherlands study, which derives its pathways estimations from Ten Broeke et al. (2008) assumes a fraction subsequently leaches back out and is then carried in run-off to sewers or to surface waters. All studies reviewed agree that a fraction of tyre wear in urban run-off will be distributed to sewage treatment.

Regarding the environmental fate of the airborne fraction of tyre wear microplastics, a variety of different approaches have been taken. The allocation of tyre wear to environmental compartments in our previous study is based upon the analyses carried out in the 2008 Netherlands National Water Board study. For urban roads, this study assumes that the 5% PM10 fraction of tyre wear remains airborne and describe no further environmental pathways steps. Although Verschoor et al. (2016) also derive their identification of pathways from Ten Broeke et al. (2008), they do acknowledge that some of the airborne fraction may eventually reach surface water but do not describe intermediate pathway steps. Of the studies reviewed, only the IUCN study included a factor for the allocation of airborne particulate matter to the marine environment, although intermediate pathway steps between suspension in air and deposition to the marine environment are not detailed.

Another potential pathway for the removal of tyre wear from the urban road environment is its direct capture during road sweeping and cleaning. In urban areas of Norway regular road sweeping is carried out to collect dust generated from the spreading of sand and use of studded tyres during winter. It has been noted that this process might also collect tyre wear, although the mass of polymer particles within this collected dust had not been estimated as of 2016.[474]

One country-specific factor is the use of highly porous asphalt. This road surface type is characterised by greater void space meaning it acts as a filter in which tyre particles are captured to varying extents. The build-up of dust in pores reduces the desired functions of porous asphalt, such as faster infiltration of storm water which lowers the risk of aquaplaning. To resolve this porous asphalt is regularly cleaned, for example approximately twice a year in the Netherlands,[475] and the collected debris is either incinerated or treated as hazardous waste.

National inventories of microplastic sources and scoping studies of measures to reduce emissions for Norway and Sweden[476, 477 and 478] have noted that during winter a fraction of tyre wear produced in urban areas will be emitted directly to snow on roads. Mepex (2016) note that in urban areas in Norway,

"snow is frequently moved away on dumpers to designated deposit areas, or dumped directly in the sea to get it off the streets."

---

474 Mepex (2016) *Primary microplastic- pollution: Measures and reduction potentials in Norway*, April 2016

475 A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016

476 Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

477 Mepex (2016) *Primary microplastic- pollution: Measures and reduction potentials in Norway*, April 2016

478 Kerstin Magnusson, and et al. (2016) *Swedish sources and pathways for microplastics to the marine environment*, Report for Swedish Environmental Protection Agency, March 2016

This might therefore represent a significant pathway to the marine environment for certain member states.

Figure 30: Pathways for particles produced by wear on urban roads

Solid lines indicate pathways that are included in all reviewed models of microplastic flow and dashed lines indicate those pathways that have only been included in a few of the reviewed study models, either because they are country-specific, were considered insignificant by some studies, or due to a lack of consensus between studies as to whether they represent a pathway.

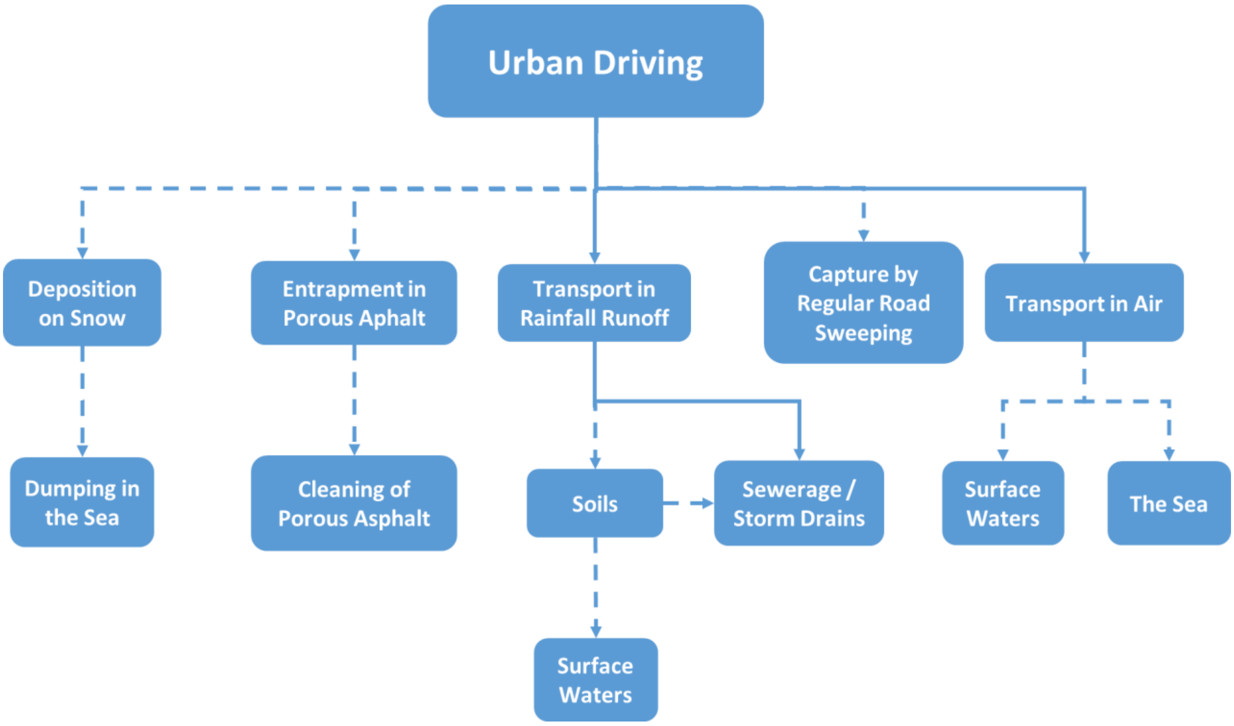


Figure 30 depicts the pathways thus far suggested in the literature for the release of microplastics from urban tyre wear. Those studies that provide some detailed quantification of mass flows along these pathways in either absolute or relative terms will now be reviewed.

Pathways quantification

Only three national- to global-scale reports reviewed have gone beyond identification of urban tyre wear pathways to explicitly attempt detailed quantification of tyre wear microplastics flows in either relative or absolute terms to a range of environmental compartments. Figures for the IUCN report could not be included as final distribution to environmental compartments could not be calculated from the figures available in their report.

Table 89 outlines the fraction of urban-generated tyre wear they allocate to various pathways. Figures for the IUCN report could not be included as final distribution to environmental compartments could not be calculated from the figures available in their report.

Table 89: Pathway Mass Flows Estimated for Tyre Wear from Urban Roads

Study (Geography)	Pathways	Mass flow
Verschoor et al., 2016 (Netherlands)	**Emitted to air**	5%
	Emitted to surface waters via sewers	32.5%
	Captured in Sewage sludge	25%
	Emitted to soil	37.5%
	Directly emitted to surface waters	0%
Eunomia et al., 2016 (Netherlands and Europe)	**Emitted to surface water via sewers**	26%
	Capture by sewage treatment	34%
	Emitted to soil	40%
	Directly emitted to surface waters	0%

Notes:

1. ***Figures for the IUCN report could not be included as final distribution to environmental compartments could not be calculated from the figures available in their report.***

The allocation of tyre wear microplastics to environmental compartments applied in both our previous report and the Netherlands report by Verschoor et al. is based upon the analyses carried out in the 2008 Netherlands National Water Board (Water Unit) study by Ten Broeke et al.. This study assumes that all particulate matter (particles ≤ 10 µm) emitted from tyres will be emitted to the atmosphere, and describes no further pathways steps for this fraction. A 60% - 40% allocation to soils and sewers respectively is applied to the remaining coarse fraction of tyre wear generated on urban roads. This distribution is largely based upon expert judgement, but Ten Broeke et al. (2008) do outline some limited quantitative analysis to back up their assumption.

Firstly, using a GIS overlay analysis for the Netherlands, they establish that 50% of the land connected to sewage systems is also paved but go on, however, to note that it cannot be automatically assumed from this that 50% of deposited coarse material will reach sewers. With this in mind, the authors discuss the potential use of run-off coefficients (a simple statistic describing the percentage of water falling in an area connected to sewerage that actually reaches the sewer system). They note a 2005 study which recorded runoff coefficients for the service areas of two Dutch wastewater treatment plants of 50% and 90%.

Ultimately, in the face of a lack of robust data, and noting the difficultly of reaching a quantitatively well-founded decision, the study proposes the assumption that 50% of tyre wear microplastic emissions in urban areas will arrive directly in sewers, and that a further 10% that is initially emitted to soils will leach back out and ultimately reach sewers. These values do not appear to be explicitly linked to any particular piece of analysis and the authors describe them as provisional until better measured data is available.

The 2017 IUCN report which seeks to quantify global pathways flows treats the airborne fraction of tyre wear in a different way. It assumes that 10% of tyre wear generated on any road type is distributed by wind citing a 2015 study by Wang et al.[479] into the sources, transport and deposition of iron the atmosphere. It is unclear why this figure has been selected as representative of the distribution of tyre wear particles by wind. Of this 10% distributed by wind, the IUCN report assumes that 100% of airborne fraction is deposited in the sea. No reason is given for assuming that 100% of the airborne fraction reaches the marine environment.

The study goes on to assume that all remaining urban emissions are distributed to sewers (either separate or combined), and assumes an optimistic scenario of 50% of all roads being connected to sewerage. The authors of the IUCN report estimated their central and pessimistic scenarios using an own-computed dataset which is not publicly available.

Rural and Highway Driving

Pathways Identification

Figure 31 illustrates the pathways that tyre wear produced on rural and highway road types may take through the environment according to the existing literature on tyre wear microplastics.

As with urban roads there is consensus that rainfall run-off and windblown drift are principal mechanisms by which tyre wear is removed from the rural/highway road types. The profile of subsequent destinations to which these transport mechanisms deposit tyre wear differs to those described for urban roads.

For example, unlike with urban roads there is a consensus in the reviewed tyre wear literature that a portion of tyre wear carried from rural/highway roads by rainfall run-off will reach soils. In many of the reviewed studies subsequent pathways steps from rural/highway soils are not explicitly described due to soils being treated as a final sink where microplastics are trapped or the difficulties of estimating onward flow from soil. The Netherlands study, however, does acknowledge that run-off from soils is likely to carry tyre wear microplastics onward to surface waters. Additionally, both Eunomia (2016) and Verschoor et al. (2016) ultimately base their distribution of tyre wear from rural/highway roads to soils on a 2005 report by Blok (2005)[480] examining the exposure of the roadside environment in the Netherlands to Zinc emissions from traffic-related sources. This study notes that in the Netherlands, the topsoil adjacent to the roadside is normally removed every 7-20 years and treated as solid waste or hazardous waste.

Regarding runoff to sewerage from rural/highway roads, there appears to be discrepancy within the literature. For example, in its global assessment the IUCN suggests that some portion of particulates produced by tyre wear on rural/highway roads will reach a form of sewerage system and Sundt et al., note that some Norwegian highway runoff is collected and treated. However, the Netherlands Water Board study upon which, as previously mentioned, the distribution of road runoff in our previous report and the Netherlands report is based, defines non-urban areas specifically as those which are not connected to sewage systems. This difference is not likely not to be based upon a

---

[479] Wang, R., Balkanski, Y., Boucher, O., et al. (2015) Sources, transport and deposition of iron in the global atmosphere, *Atmospheric Chemistry and Physics*, Vol.15, No.11, pp.6247–6270

[480] Blok, J. (2005) Environmental exposure of road borders to zinc, *Science of The Total Environment*, Vol.348, Nos.1–3, pp.173–190

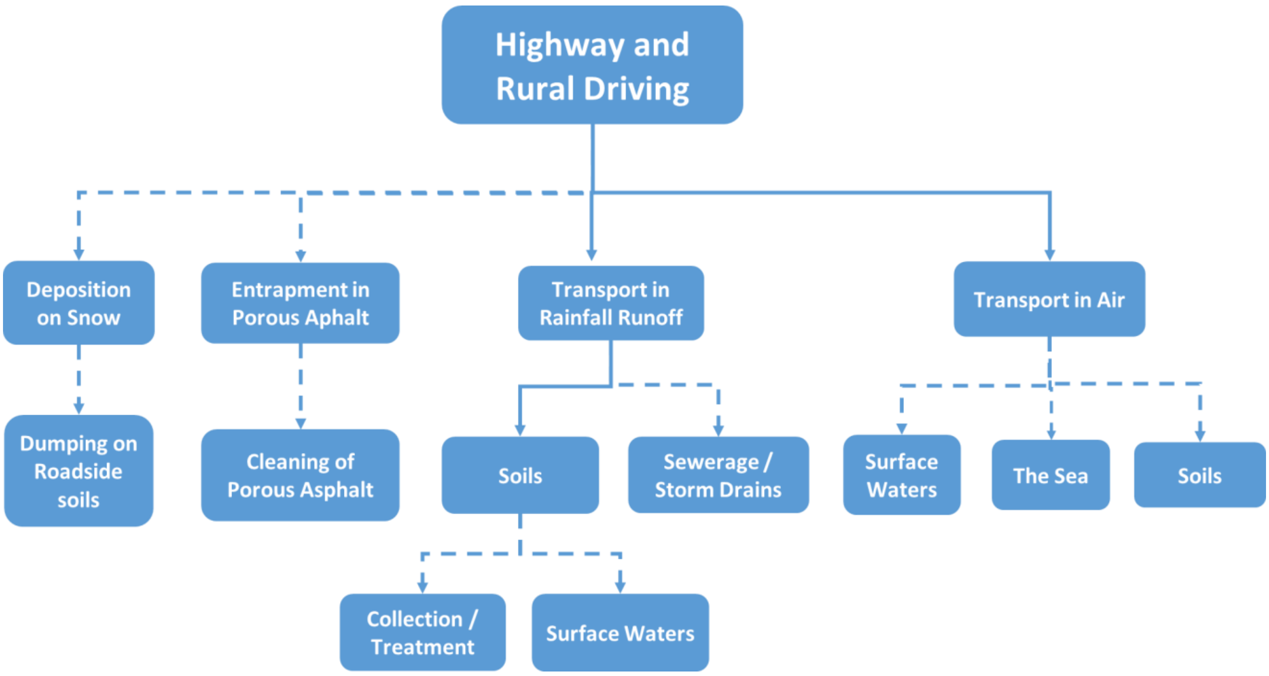
disagreement as to whether any highways or rural roads are connected to sewerage, but simply differing data availability for coverage of sewage connection for the geographies in question.

For the airborne fraction, the same variety of pathways are identified as for urban roads with one exception. Both Eunomia (2016) and Veschoor et al. (2016) base their distribution of the airborne fraction for rural/highway roads on the 2005 Dutch zinc exposure study by Blok. This study assumes that some portion of solids deposited on rural and highway roads is transported by drift to soils. Therefore, although it is not explicitly stated in either reports, Verschoor et al. and Eunomia apply distributions derived from an analysis which assumes some windborne allocation to soil.

As is the case with urban roads, porous asphalt coverage in some European countries means that entrapment in the void space of the road surface and subsequent removal during cleaning and treatment is identified as a pathway for highway roads.[481] Finally, studies for Norway and Sweden[482, 483 and 484] have noted that for rural/highway roads direct deposition to snow represents a pathway and Mepex (2016) note that outside urban areas snow is deposited at the roadside where it can be assumed that on melting, trapped tyre wear might be emitted to roadside soils.

Figure 31: Pathways for particles produced by wear on Rural/Highway roads

Solid lines indicate pathways that are included in all reviewed models of microplastic flow and dashed lines indicate those pathways that have only been included in a few of the reviewed study models, either because they are country-specific, were considered insignificant by some studies, or due to a lack of consensus between studies as to whether they represent a pathway.



[481] A. Verschoor et al. (2016) *Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear*, Report for National Institute for Public Health and the Environment (Netherlands), July 2016
[482] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014
[483] Mepex (2016) *Primary microplastic- pollution: Measures and reduction potentials in Norway*, April 2016
[484] Kerstin Magnusson, and et al. (2016) *Swedish sources and pathways for microplastics to the marine environment*, Report for Swedish Environmental Protection Agency, March 2016

Pathways Quantification

As is the case with urban roads, the only reports to attempt detailed quantification of tyre wear microplastic flows from rural/highway roads in either relative or absolute terms to a range of environmental compartments are our previous report to DG ENV, the Netherlands report and the IUCN report. Table 90 outlines the fraction of rural/highway tyre wear they allocate to various pathways. Figures for the IUCN report could not be included as final distribution to environmental compartments could not be calculated from the figures available in their report.

Table 90: Pathway Mass Flows Estimated for Tyre Wear from Rural/Highway Roads

Study (Geography)	Pathways	Mass flow
Verschoor et al., 2016 (Netherlands)	**Emitted to air**	5.0%
	Emitted to surface water via sewers	0.0%
	Captured in Sewage sludge	0.0%
	Emitted to soil	85.5%
	Directly emitted to surface waters	9.5%
Eunomia et al., 2016 (Netherlands and Europe)	**Emitted to surface water via sewers**	4.3%
	Capture by sewage treatment	5.7%
	Emitted to soil	90%
	Directly emitted to surface waters	0%

Notes:

1. Figures for the IUCN report could not be included as final distribution to environmental compartments could not be calculated from the figures available in their report.

The allocation of tyre wear microplastics to environmental compartments applied in both our previous report and the National emissions inventory for the Netherlands is based upon the analyses carried out by Ten Broeke et al. (2008). Regarding the airborne fraction, Ten Broeke et al. assume that the PM fraction of tyre wear, five percent of total emisisons, becomes airborne and do not describe further pathways steps.

This study derived the relative share of coarse tyre wear particulates removed from the road environment by runoff and drift from the aforementioned Dutch roadside zinc exposure study by Blok. As a large part of the literature quantifying environmental flows of microplastics from tyres uses the figures derived from this study as the basis of divisions of flows between environmental compartments, their derivation merits scrutiny.

Blok (2005) concludes that in a year around one third of zinc emissions on roads are carried away by drift and the other two thirds by run-off. Later in his study, Blok restates the relative allocation as,

"about 70% is removed from the road surface by run-off during storm events and enters in a small zone directly along the edge of the paved surface where infiltration of water occurs…the other 30%…is distributed by drift of fine solids over longer distances".

In the formulation of the Netherlands National Emissions inventory guidance for road traffic tyre wear, Ten Broeke et al. (2008), draw on Blok's 70% / 30% allocation of zinc emissions between runoff and drift and apply this distribution to tyre wear in general. Blok also claims that one third of all emitted solids on roads are distributed by air over a distance less than 15m from the carriageway, and that 75% of this drift is deposited at a distance less than six metres. Ten Broeke et al. applied this assumption, that 25% (75% of one third) of total solid emissions on roads are distributed by drift less than 6m from the road edge, to tyre wear. Then, assuming that less than 50% of this area equal to or less than 6 metres from the road edge would be made up of ditches (for drainage), they half the 25%, rounding down, and settle on a figure of 10% of tyre wear from rural and highway roads being emitted directly to surface waters. The remaining 15% is assumed to be emitted to soils along with the 5% assumed to travel 6-15m, as is the 70% originally estimated by Blok to be removed by run-off to the road environment, totally 90%.

It is this distribution, 90% to soil and 10% to surface waters, that both Eunomia and Verschoor et al. ultimately apply in subsequent analyses of pathway flows of coarse particulates from rural and highway roads.

The IUCN report (2017), as with urban roads, assumes that 10% of all emissions to non-urban roads are released in an airborne fraction and assumes a 100% deposition rate to marine environment based on a 2015 study by Wang et al.[485] For the remainder of rural emissions the IUCN assume that 3.5% of emissions at source in rural areas are emitted to the marine environment, but do not describe intermediate pathways steps nor a methodology for arriving at this figure.

Identification and Quantification of Pathways Conclusions

This review has highlighted the large range of pathways identified for tyre wear leaving the road environment. It appears to be agreed that for all road types the principal mode of transport from the road environment is rainfall runoff with additional important pathways being some entrapment in road surfaces, capture by road cleaning activities and removal by wind of particulate matter. More esoteric transport mechanisms specific to Member States, due to factors such as climate and road management regimes, have also featured in identification of pathways but rarely in quantification. Additionally, the destination and quantity of flows along said pathways is often treated as varying dependent on whether wear occurs on urban, rural or highway roads.

Another important output of this literature review is the understanding that, as a relatively young and still-developing body of literature, limited empirical data, informed assumptions and proxies must be relied upon in the absence of sufficient measured data specific to tyre wear for determining key parameters such as factors for distributing wear between transport mechanisms. The final load calculated as reaching various environmental compartments including the marine environment varies between studies according to this imprecise division of wear between likely pathways. However, as with the estimation of emissions at source, this uncertainty does not preclude a pragmatic approach in which available data is applied to establish likely orders of magnitude of mass flow, ranges of emissions to different environmental compartments and reasonable midpoints to arrive at figures which are of satisfactory reliability for the development and prioritisation of mitigating policy options.

---

[485] Wang, R., Balkanski, Y., Boucher, O., et al. (2015) Sources, transport and deposition of iron in the global atmosphere, *Atmospheric Chemistry and Physics*, Vol.15, No.11, pp.6247–6270

A.3.8.7 Pathways Modelling Results Tables

Table 91 – Pathways and Sinks Modelling

*Summed for surface water release, ** Summed for capture in sludge, ***Summed for Disposal to residual waste + 50% captured in sludge

	Entry points - pathways									
Sinks	Residential (foul water)		Storm Water Run-off							
			Urban		Rural		Highway		Non-Roads Drains	
Direct* [a]	**-**		**-**		**10—20%**		**-**		**-**	
Porous Asphalt* [b]**	**-**		**-**		**-**		**4.5%**		**-**	
Road Cleaning* [c]**	**-**		**5%**		**-**		**0.3%**		**-**	
Soil [a]	**-**		**30%**		**80—90%**		**38.2%**		**-**	
Sewer	**100%**		**65%**		**-**		**57%**		**100%**	
Sewer - Combined [d]	**50%**		**32.5%**				**0%**		**50%**	
Combined Overflow* [d]	*5%*	*5%*	*3.3%*	*3.3%*	-		-		*5%*	*5%*
Treated Effluent* [e]	*21%*	*7%*	*7.7%*	*0.5%*	-		-		*11.7%*	*.8%*
Treated Sludge [e]**	*24%*	*38%*	*8.7%*	*2.75%*	-		-		*13.3%*	*4.2%*
Sedimentation* [f]**	-	-	*13%*	*26%*					*20%*	*40%*
Sewer - Separate [d]	**50%**		**32.5%**				**57%**		**50%**	
Treated Effluent* [e]	*23%*	*8%*	-	-	-		-	-	-	-
Treated Sludge [e]**	*27%*	*42%*	-	-	-		-	-	-	-
Untreated*	-	-	*20%*	*7%*	-		*34%*	*11%*	*30%*	*10%*
Sedimentation* [f]**	-	-	*13%*	*26%*	-		*23%*	*46%*	*20%*	*40%*
Releases to Surface Waters (Σ*)	**49%**	**20%**	**31%**	**10%**	**20%**	**10%**	**34%**	**11%**	**47%**	**16%**
Captured in Sludge (Σ)**	**51%**	**80%**	**9%**	**3%**	**-**	**-**	**-**	**-**	**13%**	**4%**
Applied to Agricultural Land (50% sludge)	**25%**	**40%**	**4%**	**1%**	**-**	**-**	**-**	**-**	**6%**	**2%**
Waste Management (Σ*)**	**26%**	**41%**	**36%**	**59%**	**-**	**-**	**28%**	**50%**	**47%**	**83%**

Notes:

a) **Direct emission, see main body text.**
b) **Estimated at 90% capture rate with 5% of European highways using it. See Section 2.2.8.3.**
c) **Road cleaning See Section 2.2.8.4.**
d) **Estimated at 5% lost direct to surface waters through CSO's. A split of 50:50 between separate and combined sewer systems.**
e) **Estimated capture of microplastics in WWT to be between 53% and 84%.**
f) **40—80% of microplastics that end up here are assumed to be captured (see section 2.2.8.2). Example Calculation: 65% of urban emissions end up in sewers, 50% of this ends up in a separate sewer and 40—80% of this settles out in sediment— 65% x 50% x 40% =13%.**

Table 92– High Release Scenario Pathway Results (tonnes released at source)

Source	Residential Sewerage	Run-off				Direct to Surface Water	Direct to Soil	Total
		Urban	Rural	Highway	Non-Road Drains			
Automotive Tyres		204,446	199,888	99,252				**503,586**
Washing Clothing	46,662							**46,662**
Cleaning Cloths	1,288							**1,288**
Artificial Turf	3,625				3,625		32,447	**39,698**
Pre-Production Plastics					167,206	225		**167,431**
Fishing Gear						4,780		**4,780**
Marine Paint						411		**411**
Building Paint		13,552	21,311					**34,863**
Road Markings		20,294	71,636	2,428				**94,358**
Automotive Paint		33	46	19				**98**
Automotive Brakes		5,557	8,172	3,431				**17,161**
Total	**51,575**	**243,882**	**301,053**	**105,131**	**170,831**	**5,417**	**32,447**	**910,337**

Table 93- Low Release Scenario Pathway Results (tonnes released at source)

Source	Residential Sewerage	Run-off				Direct to Surface Water	Direct to Soil	Total
		Urban	Rural	Highway	Non-Road Drains			
Automotive Tyres		204,446	199,888	99,252				**503,586**
Washing Clothing	18,430							**18,430**
Cleaning Cloths	1,288							**1,288**
Artificial Turf	914				914		8,112	**9,939**
Pre-Production Plastics					16,945	141		**17,086**
Fishing Gear						478		**478**
Marine Paint						411		**411**
Building Paint		8,204	12,901					**21,105**
Road Markings		20,294	71,636	2,428				**94,358**
Automotive Paint		33	46	19				**98**
Automotive Brakes		163	240	101				**505**
Total	**20,632**	**233,140**	**284,711**	**101,801**	**17,858**	**1,030**	**8,112**	**667,285**

Table 94- Midpoint Release Scenario Pathway Results (tonnes released at source)

Source	Residential Sewerage	Run-off				Direct to Surface Water	Direct to Soil	Total
		Urban	Rural	Highway	Non-Road Drains			
Automotive Tyres		204,446	199,888	99,252				**503,586**
Washing Clothing	32,546							**32,546**
Cleaning Cloths	1,288							**1,288**
Artificial Turf	2,270				2,270		20,280	**24,819**
Pre-Production Plastics					92,075	183		**92,259**
Fishing Gear						2,629		**2,629**
Marine Paint						411		**411**
Building Paint		10,878	17,106					**27,984**
Road Markings		20,294	71,636	2,428				**94,358**
Automotive Paint		33	46	19				**98**
Automotive Brakes		2,860	4,206	1,766				**8,833**
Total	**36,103**	**238,511**	**292,882**	**103,466**	**94,345**	**3,224**	**20,280**	**788,811**

A.4.0 Baseline Calculations

Table 95 shows the microplastics emissions from the baseline year of 2017. This is may be different from the calculation year where pre-2017 data is used for each source. For example, automotive tyre data comes from 2012 and is scaled to 2017 using the method explained in the table notes.

Table 95 - Microplastics Emissions Projections using Midpoint Estimates (2017-2035)

Source	Microplastics Emitted to Surface Water (tonnes)					
	2017	2020	2025	2035	Change from 2017	
Automotive Tyres [a]	100,820	103,780	109,034	122,480	21%	22,630
Washing of Clothing [b]	13,296	13,229	13,083	12,694	-5%	-622
Artificial Turf [c]	2,777	3,242	3,990	5,577	101%	2,954
Pellets [d]	42,611	45,179	49,464	58,697	38%	17,012
Fishing Gear [b]	2,642	2,660	2,684	2,710	3%	75
Marine Paint [d]	442	469	515	612	39%	181
Building Paint [d]	5,661	6,008	6,584	7,822	38%	2,285
Road Markings [a]	15,856	16,484	17,589	20,031	26%	4,379
Automotive Brakes [a]	2,433	2,530	2,702	3,080	27%	679
Total	186,557	193,601	205,665	233,727	27%	49,576

Notes:

a) Tyres wear, brake wear particle generation and road paint wear is predicted to increase in line with vehicle/km driven. This is estimated to increase by 35%[486] between 2005 and 2030. A proportional increase is then extrapolated out to 2035.

b) Clothing are expected in increase in line with population growth (OECD[487]) rather than clothing sales as more clothing may not necessarily lead to more washing, but a greater population will. Fishing gear is also projected to increase in line with population.

c) Estimates extrapolated from ESTO data[488] for 2020 forecasts—8.5% growth per year—followed by a more conservative estimate of growth of 2% thereafter.

d) Pellet loss, marine and building paint emissions are projected to increase in line with GDP forecasts. For marine paints this is a simplification due to growth not being governed by GDP, but in trends in the cargo industry. This is not necessarily a linear increase. Building paints are a mature market in the EU which may also move slower than GDP forecasts. [489]

e) All sources apart from fishing gear and marine paint have a proportion that may move through WWT plants. The calculations take into account projected improvement in WWT in line with the UWWT Directive. This indicates a 12% increase in tertiary treatment and a 9% increase in Secondary treatment. This changes the capture rates from 54-85% to 54.5-93% expected to be implemented evenly over the time period.

---

[486] Tetraplan A/S (2009) *Traffic flow: Scenario, Traffic Forecast and Analysis of Traffic on the TEN-T, Taking into Consideration the External Dimension of the Union.*, Report for European Commission, December 2009

[487] OECD, Data extracted on 14 Jun 2017 10:49 UTC (GMT) from OECD.Stat, http://stats.oecd.org/index.aspx?DatasetCode=POP_PROJ

[488] ESTO (2016) *Market Report Vision 2020*, 2016

[489] OECD (2017), GDP long-term forecast (indicator). doi: 10.1787/d927bc18-en (Accessed on 14 June 2017)

A.5.0 Screening of Policy Options

Tool #14 of the Better Regulation Toolkit identifies the following key criteria for screening options:

- Legal feasibility
    - Options must represent the principle of conferral. They should also respect any obligation arising from the EU Treaties (and relevant international agreements) and ensure respect of fundamental rights. Legal obligations incorporated in existing primary or secondary EU legislation may also rule out certain options
- Technical feasibility
    - Technological and technical constraints may not allow for the implementation, monitoring and/or enforcement of theoretical options
- Previous policy choices
    - Certain options may be ruled out by previous Commission policy choices or mandates by EU institutions
- Coherence with other EU policy objectives
    - Certain options may be ruled out early due to poor coherence with other general EU policy objectives
- Effectiveness and efficiency
    - It may already be possible to show that some options would uncontrovertibly achieve a worse cost-benefit balance than some alternatives
- Proportionality
    - Some options may clearly restrict the scope for national decision making over and above what is needed to achieve the objectives satisfactorily
- Political feasibility
    - Options that would clearly fail to garner the necessary political support for legislative adoption and/or implementation could also be discarded
- Relevance
    - When it can be shown that two options are not likely to differ materially in terms of their significant impacts or their distribution, only one should be retained

In the sections below the initial screening of the policy options for the microplastics sources of relevance is presented.

Where there is one, or a small number of criteria, that clearly preclude an option from being feasible, we note these, without then addressing the other criteria. For options that are identified as feasible, such identification is made after the option has been screened against all criteria.

A.5.1 Selected Options

The following section describes the selected options taken forward for further analysis and discussion with stakeholders. After this process, the final options for assessment were agreed and their impacts assessed (See Section 7.0 of the main report).

A.5.1.1 Automotive Tyre Wear

Table 96: Selected Measures for Automotive Tyre Wear

Type of Measure	Description of Measure
BAU	No change - baseline scenario
Development of a Standard Measure of Tyre Tread Abrasion Rate	Develop a standard EU measure of tread abrasion rate for tyres under the Type Approval Regulation for Tyres and the Tyre Labelling Regulation
Development of a Standard Measure of Tyre Tread Abrasion Rate/Tyre Label	Once a standard test for tread abrasion rate has been developed, include a rating for tread abrasion rate on the Tyre Label
Development of a Standard Measure of Tyre Tread Abrasion Rate / Amend Existing regulation	Once a standard test for tread abrasion rate has been developed, include minimum tread abrasion rate requirements under the Type Approval Regulation for Tyres
Development of a Standard Measure of Tyre Tread Abrasion Rate / Awareness Raising	Once a standard test for tread abrasion rate has been developed, if fuel efficient tyres are show to be associated with reduced rates of treadwear, instead of adding tread abrasion rate to the Tyre Label, raise awareness of this further reason to buy the most fuel efficient tyres under the Tyre Label.
Development of a Standard Measure of Tyre Tread Abrasion Rate /Green Public Procurement	Once a standard test for tread abrasion rate has been developed, and a rating for tread abrasion included in the Tyre Label, include the highest Tyre Label tread abrasion requirements, alongside current fuel efficiency requirements, under the Central Government Green Public Procurement requirements of Annex III of the Energy Efficiency Directive

A.5.1.2 Pre-production Plastics

Table 97: Selected Measures for Pre-production Plastics

Type of Measure	Description of Measure
BAU	No change – baseline scenario
Using existing legislation	Include best practice measures for preventing loss of pre-production plastics as BAT in the Polymer production BREF, either as part of the general update of the BREF, or as a focused addendum to the current version
Voluntary initiative	Implement a procurement based approach whereby business end-users of plastic material such as users of packaging require their suppliers to demonstrate adherence to best practice measures to prevent loss of pre-production plastics, and for those higher up the chain to likewise prove adherence to best practice
New legislation	Implement new legislation at the EU-level specifically to address the matter of pre-production pellet loss

A.5.1.3 Synthetic Clothing

Table 98: Selected Measures for Synthetic Clothing

Type of Measure	Description of Measure
BAU	No change - baseline scenario
Industry awareness raising	EU to support the exchange and roll-out of best practice with the clothing supply chain, building on recommendations from the MERMAIDS project
Industry-led accreditation scheme	Development of an industry-led accreditation scheme for supply chains that adhere to best practice (with best practice being revised and updated as further research is undertaken)
Consumer awareness raising	An information campaign to increase awareness amongst consumers of the measures they can take to reduce microfibre loss when washing. This could be publicly funded, industry funded, or jointly funded
Consumer awareness raising / Regulation	Require labels on synthetic clothing to include guidance on best practice for avoiding microfibre loss during washing
Regulation	Revise the Textiles BREF to incorporate recommendations from the MERMAIDS project, including prewashing of garments prior to being placed on the market

Regulation	Require all synthetic clothing sold in the EU to be pre-washed, with microfibres adequately captured, prior to being placed on the market
Measurement Protocol	Develop a measurement protocol for the rate of loss of synthetic microfibres from specific items of clothing. This could be achieved through industry working with CEN
Regulation	Once the measurement protocol is developed, require labelling to show whether the garment will exhibit a high, medium or low rate of synthetic microfibre loss, or indeed no loss
Ecolabelling	Once the measurement protocol is developed amend the EU Ecolabel criteria for Textile Products to include only synthetic textiles that exhibit the lowest rate of loss
Green Public Procurement	Once the measurement protocol is developed amend Green Public Procurement criteria for Textile Products and Services to include only synthetic textiles that exhibit the lowest rate of loss
Regulation	Once the measurement protocol is developed, ban the manufacture and sale within the EU of clothing that exhibits the highest rate of loss of synthetic microfibres
Research Funding	EU to provide funding for research and development of washing machine filters
Regulation	Require washing machine manufacturers to develop filters to be included in new machines
Regulation	Require professional laundries to install filters
Research Funding	EU to fund testing of the effectiveness and consumer acceptance of using microfibre capture devices such as a Guppy Friend washing bag and a Cora Ball

A.5.1.4 Road Markings

Table 99: Selected Measures for Road Markings

Measure	Description of Measure
BAU	No change - baseline scenario
Research Funding	EU and industry to provide joint funding of research to better understand the relative rates of loss of microplastics from different types of road markings

A.5.1.5 Building Paint

Table 100: Selected Measures for Building Paint

Measure	Description of Measure
BAU	No change - baseline scenario

Research Funding	EU or industry funded research into the development of alternatives to polymers as binders in building paint

A.5.1.6 Artificial Turf

Table 101: Selected Measures for Artificial Turf

Measure	Description of Measure
BAU	No change - baseline scenario
Development of Guidance	Best practice guidance on prevention of infill loss, and the potential for use of natural infill materials, to be developed by FIFA and World Rugby
Industry-led Accreditation	Best practice guidance to be incorporated by FIFA and World Rugby as part of their accreditation scheme for pitches and thus a requirement for those who wish to be (or remain) accredited
Green Public Procurement	Require any public body that owns or manages an artificial sports pitch to adhere to the best practice guidance on preventing infill loss
Regulation	Require any organisation that owns or manages an artificial sports pitch to adhere to the best practice guidance on preventing infill loss

A.5.1.7 Capture of Microplastics on Roads

Table 102: Selected Measures for Capture of Microplastics on Roads

Measure	Description of Measure
Research	The European Commission to fund research into the appropriate measures, and combination of measures, to treat road runoff in order to improve water quality, accounting for a wide range of pollutants, including microplastics from tyre wear and road markings. Such research could then inform a guidance document.
Guidance	Once research into treating road runoff has been completed, guidance to be distributed to National Road Administrations (NRAs) in EU Member States as to the priority locations and circumstances for interventions and the relative cost and effectiveness of such interventions
Research	The European Commission to fund research into the appropriate measures, and combination of measures, in respect of street cleansing that both improve air quality and improve water quality, accounting for a wide range of pollutants, including microplastic particles from tyre wear and road markings. Such research could then inform a guidance document.

Guidance	Once research has been completed into street cleansing, guidance to be distributed to municipalities and National Road Administrations (NRAs) in EU Member States as to the priority locations and circumstances for interventions and the relative cost and effectiveness of such interventions

A.5.1.8 Capture of Microplastics in Wastewater Treatment

Table 103: Selected Measures for Capture of Microplastics in Wastewater Treatment

Measure	Description of Measure
BAU	No change - baseline scenario
Research Funding	EU and/or industry funding to develop a standard approach to measuring the capture rate of microplastics in wastewater treatment facilities
Research funding	EU and/or industry funding for research into techniques that prevent microplastics from being contained within sewage sludge

A.5.2 Rejected Options

The following options were rejected for further analysis.

A.5.2.1 Automotive Tyre Wear

For automotive tyre wear, the following options have been assessed as **not being feasible**. For each, the criteria against which it was judged not to be feasible, and an explanation for this decision, are provided.

- Publicly or industry funded awareness raising campaign as to the factors that affect the rate of tyre wear such as speed, driving style and vehicle weight, and what can be done to reduce tyre wear including driving less and switching to other modes of transport
    - **Effectiveness and efficiency -** Such behaviour is already acknowledged to be best practice in terms of fuel efficient driving. Therefore there would seem to be no merit in a separate campaign to raise awareness of the issue of microplastics from tyre wear. Instead, the issue of avoiding tyre wear (and associated generation of microplastics) could be added in to current eco-driving and sustainable transport campaigns.
- Provide EU grants to manufacturers to undertake research into the development of tyres that abrade at lower rates while maintaining or improving performance in terms of other attributes such as wet grip and external noise
    - **Political feasibility -** While possible, it would be hard to justify the use of public money to fund such research when the manufacturers should be adequately incentivised, through the market, to undertake such research and development themselves

- Once a standard test for tread abrasion rate has been developed, implement a tax on the tyres that abrade at the highest rates (the level of the tax varying based on the rate of abrasion)
    - **Legal feasibility -** Likely to be challenged as Member States have responsibility for most matters in relation to tax
    - **Political feasibility** - Likely to be strongly opposed
    - **Relevance** - Likely to be unnecessary if a rating for tread abrasion is included in the Tyre Label, and the tyres that abrade at the highest rates are removed from the market under the Type Approval Regulations for Tyres
- Once a standard test for tread abrasion rate has been developed, require implementation of Extended Producer Responsibility for vehicle tyres in all Member States, with minimum requirements including modulation of fees based on factors such as tread abrasion rate, noise and fuel efficiency.
    - **Effectiveness and efficiency** - Would be less effective than other options such as a rating for tread abrasion being included in the Tyre Label, and tyres that abrade at the highest rate being removed from the market under the Type Approval Regulations for Tyres
    - **Political feasibility** – Likely to be strongly opposed
    - **Relevance** - Likely to be unnecessary for the purposes of restricting tyre wear if a rating for tread abrasion is included in the Tyre Label, and tyres that abrade at the highest rates are removed from the market under the Type Approval Regulations for Tyres
- Once EPR is implemented, require a financial incentive to move towards the provision of a rental model for tyres, which would have the effect of increasing the incentive for tyres to abrade at lower rates, through much lower EPR fees for tyres that are rented.
    - **Effectiveness and efficiency** - Would be harder to implement, and no more effective than other options such as a rating for tread abrasion being included in the Tyre Label, and tyres that abrade at the highest rates being removed from the market under the Type Approval Regulations for Tyres
    - **Relevance** - Likely to be unnecessary for the purposes of restricting tyre wear if a rating for tread abrasion is included in the Tyre Label, and tyres that abrade at the highest rates are removed from the market under the Type Approval Regulations for Tyres
- A financial incentive to move towards the provision of a rental model for tyres, which would have the effect of increasing the incentive for tyres to abrade at lower rates, through a high rate of tax on the sale of tyres to end consumers
    - **Legal feasibility -** Likely to be challenged as Member States have responsibility for most matters in relation to tax
    - **Effectiveness and efficiency** - Would be harder to implement, and no more effective than other options such as a rating for tread abrasion being included in the Tyre Label, and tyres that abrade at the highest rates being removed from the market under the Type Approval Regulations for Tyres
- A financial incentive to move towards the provision of a rental model for tyres, which would have the effect of increasing the incentive for tyres to abrade at lower rates, through a tax on tyres that cannot be retreaded
    - **Effectiveness and efficiency** - Would be harder to implement, and less effective than other options such as a rating for tread abrasion being included in the Tyre Label, and

  - tyres that abrade at the highest rates being removed from the market under the Type Approval Regulations for Tyres
  - **Political feasibility** – Likely to be strongly opposed

A.5.2.2 Pre-production Pellets

For pre-production plastics, the following options have been assessed as **not being feasible**. For each, the criteria against which it was judged not to be feasible, and an explanation for this decision, are provided.

- Reformulation of pre-production plastics and associated delivery mechanisms in order to significantly reduce the risk of spills and facilitate clean-up (e.g. pre-production plastics being transported in liquid form or as pellets of a much larger size)
  - **Technical feasibility** - This is not technically feasible. The entire supply chain is set up on the basis of pre-production plastics in their current form, and to change this would not be practical, and would be excessively costly and disruptive
  - **Political feasibility** - This would not be politically feasible. There would be huge resistance from the plastics industry
- EU-funded campaign to increase awareness among the public of the problem and of the measures that can be taken by firms to address it
  - **Effectiveness and efficiency** - NGOs are already engaged in significant efforts to raise awareness of the problem, such as the Great European Nurdle Hunt (2nd to 5th June). While there may be merit in a greater level of public awareness *per se*, it's a very different issue from, for example, microbeads in cosmetics where consumers engage directly with brands that incorporate microplastics as ingredients. With pellets, the problem is further up the supply chain, and there is no obvious stakeholder for consumers to target apart from retailers. Accordingly this would be far less effective in bringing about change than has awareness raising on cosmetic microplastics. Engagement with retailers to date suggests they are keen to adopt the procurement-based approach outlined below, as they know this will be well received by customers. Accordingly there seems no need for increased public awareness
- Industry funded campaign to increase awareness among the public of the problem and of the measures that can be taken by firms to address it
  - The same rationale in respect of **effectiveness and efficiency** of an EU-funded campaign holds for an industry funded campaign.
  - **Political Feasibility** - This would not be politically feasible. The obvious question (which would be posed both by industry and the public) is why, rather than simply address the problem, is industry raising public awareness of it. There would not be enthusiasm among industry stakeholders for such a campaign
- The Commission to develop a guidance note for Competent Authorities in Member States on the best practice measures that firms handling pre-production pellets can apply, to encourage such advice to be given in the course of routine inspections of facilities that happen to handle pellets
  - **Effectiveness and efficiency** - This would not be an effective approach. The vast majority of plastics converters are not inspected by regulatory authorities, and there would be no driver for firms to adopt the recommendations

- Plastics industry to obtain independent verification that firms that have signed up to Operation Clean Sweep have actually put all best practice measures in place and that these are effective in preventing pellet loss
    - **Effectiveness and Efficiency** - This would enhance confidence that the reported growth in the number of signatories to the voluntary Operation Clean Sweep programme is leading to a related uptake in best practice measures to prevent pellet loss. However, of itself, it is unlikely to lead to widespread uptake of best practice techniques to prevent loss of pre-production plastics. It would be far less effective than the procurement-based approach, for example.
- Waste Framework Directive - The Commission to issue guidance to Competent Authorities to make clear that spilled pellets are a waste, and that waste producers and handlers must therefore ensure appropriate containment
    - **Effectiveness and Efficiency** - While technically, spilled pellets are a waste (as plastics converters do not reincorporate them into the manufacturing process for fear of contamination), the effect of such a guidance note would be extremely limited, as it would rely on a relatively high likelihood of spills being detected and enforcement action being taken by competent authorities. Given the large number of (especially smaller) plastics converters that are not subject to regular inspections, this is unlikely to be a sufficiently strong incentive to bring about any meaningful change in practices unless fines were very high and given due publicity
- Water Framework Directive - The Commission to issue guidance to Member States on measures needed to prevent the loss of pre-production pellets for the next round of Member State updates of their WFD Programme of Measures (due in 2021)
    - **Effectiveness and Efficiency** - Apart from the time delay inherent in such an approach, using the Water Framework Directive in this way may lead Member States, if they chose to act, to impose differing mechanisms on those in the pellet supply chain within their territory, which may interfere with the operation of the single market. Furthermore, such an approach would not deal with extra-EU emissions of pre-production pellets, which is something that alternative options would achieve.
- Marine Strategy Framework Directive - The Commission to issue guidance to Member States on measures needed to prevent the loss of pre-production pellets for the next round of possible Member State updates of their MSFD Programme of Measures (due in 2022)
    - **Effectiveness and Efficiency -** Apart from the time delay inherent in such an approach, using the Marine Strategy Framework Directive in this way may lead Member States, if they chose to act, to impose differing mechanisms on those in the pellet supply chain within their territory, which may interfere with the operation of the single market. Furthermore, such an approach would not deal with extra-EU emissions of pre-production pellets, which is something that alternative options would achieve.
- Implement a unit-based tax on pre-production plastics in order to increase the financial incentive to avoid spills and to collect spilled material
    - **Effectiveness and Efficiency -** The level of the tax -relative to the cost of pre-production plastics - that would be required to stimulate widespread adoption of best practice measures to prevent the loss of pre-production plastics would have to be extremely high indeed.
    - **Political Feasibility -** Such a tax would be opposed by industry as it would give other materials that compete with plastics an advantage

A.5.2.3 Synthetic Clothing

- Ban the manufacture and sale of synthetic clothing within the EU
    - **Legal feasibility -** This would not be feasible from a legal perspective
    - **Coherence with other EU policy objectives -** The resultant increase in consumption of natural fibres would likely lead to adverse impacts on water quality from expanded production of cotton
    - **Proportionality -** This would be a disproportionate response to the issue
    - **Political Feasibility –** Such a proposal would incur considerable political resistance
- Once a measurement protocol is developed, implement EPR for clothing with fees modulated based on life cycle impacts, including the rate of loss of synthetic microfibres
    - **Legal feasibility -** A fee modulation approach that tried to account for all impacts would likely be subject to legal challenge
    - **Technical feasibility -** There would be significant technical challenges in establishing a defensible basis for fee modulation across the different fibre types
    - **Proportionality -** This would be a disproportionate response to the issue of tackling the release of synthetic microfibres from clothing
    - **Political Feasibility –** There would be significant political opposition to the fee modulation proposed under such an approach
- Implement a tax on synthetic clothing to incentivise a shift towards natural fibres
    - **Legal feasibility** - This would be subject to legal challenge for discriminating against all synthetic fibres
    - **Coherence with other EU policy objectives -** The resultant increase in consumption of natural fibres would likely lead to adverse impacts on water quality from expanded production of cotton
    - **Proportionality -** This would be a disproportionate response to the issue of tackling the release of synthetic microfibres from clothing, especially given the range of alternative options available
    - **Political Feasibility –** There would be significant political opposition to such a proposed tax
- Dependent on the outcome of testing, provide public funding to ensure each household and professional laundry is given a Guppy Friend washing bag and/or Cora Ball
    - **Effectiveness and efficiency -** It is not possible at present to model the possible impacts of this option
- Dependent on the outcome of testing, require synthetic clothing industry to fund the supply of a Guppy Friend washing bag and/or Cora Ball to each household and professional laundry
    - **Effectiveness and efficiency -** It is not possible at present to model the possible impacts of this option
- Implement a tax on all clothing to incentivise consumers to use existing clothes for longer, and to buy second hand clothes
    - **Legal feasibility -** There would be a legal challenge to such a proposal
    - **Proportionality -** This would be a disproportionate response to the issue of synthetic microfibre loss from clothing
    - **Political feasibility –** This would not be feasible from a political perspective
- A publicly funded awareness raising campaign of the benefits of using clothes for longer and of reusing clothes (i.e. buying second hand clothes)
    - **Effectiveness and efficiency -** Given the existence of public awareness campaigns and initiatives to encourage reuse of clothing, it would not be effective to introduce a

campaign solely on the basis of avoiding loss of synthetic microfibres. Instead, this additional rationale should be incorporated into the messaging of ongoing campaigns

A.5.2.4 Road Markings

- Once research is completed, an EU-funded or industry funded campaign to increase awareness among road markings users of the problem and of any way in which selection of different types of road markings may lose microplastics at different rates
  - **Effectiveness and Feasibility** - It is not possible to assess how effective such an approach would be as it would depend firstly upon whether the research identified any variation in rates of loss of microplastics across different types of road markings, and secondly the response of road markings users to the awareness raising
- Once research is completed, implement a tax on road markings that varies according to the rate of loss of microplastics in order to encourage a shift towards those less prone to generate emissions of microplastics
  - **Effectiveness and Feasibility** - It is not possible to assess how effective such an approach would be as it would depend firstly upon whether the research identified any variation in rates of loss of microplastics across different types of road markings, and secondly the response of road markings users to the tax
  - **Political feasibility** - Implementing such a tax would be liable to significant political and industry opposition, including, potentially on grounds of safety
- Once research is completed, implement a ban on the types of road markings that exhibit the greatest rate of loss of microplastics less prone to generate emissions of microplastics
  - **Effectiveness and Feasibility** - It is not possible to assess how effective such an approach would be as it would depend upon whether the research identified any variation in rates of loss of microplastics across different types of road markings, the extent of that variation, and the point at which a restriction would be applied
- An EU-funded campaign to increase awareness among road authorities of alternatives to road markings
  - **Effectiveness and Feasibility** - The development of such approaches to date has primarily been driven by reasons of safety (to slow speeds), aesthetic considerations, and to encourage modal shift. Encouraging the further development of such approaches in order to address microplastics would be a disproportionate and inefficient response. It would make more sense for advocates of such approaches to include microplastics as a further reason to implement such schemes, rather than for a new awareness raising campaign to be developed that focuses solely on microplastics

A.5.2.5 Building Paint

- EU or industry funded awareness raising campaign to encourage contractors and the general public of the problem of microplastics from removing old paint and of mitigating actions they can take
  - **Proportionality -** Given that the transmission of microplastics from building paints to the marine environment is a theoretical pathway (i.e. they have not been found) and the relatively small contribution they are thought to make, along with the potential

for road drainage to capture such microplastics, such a campaign would seem to be a disproportionate response to an issue that's not yet fully characterised

A.5.2.6 Artificial Turf

- Awareness raising among pitch specifiers and operators, and users and the general public, of the issue of infill loss and of best practice measures that can be applied to minimise loss
  - **Effectiveness and efficiency** - Awareness raising alone is likely to be far less effective than other measures
  - **Relevance** - There would be no point in pursuing this approach if other measures, such as the industry-led accreditation, are pursued. The industry-led accreditation will, of itself, raise awareness
- A tax on tyre-derived (and other plastic) infill in order to seek to internalise the external costs associated with loss. In making the infill more expensive this could encourage a greater emphasis among pitch operators on preventing loss. It could also encourage a shift towards alternative infill material such as cork
  - **Legal feasibility** - Implementing such a tax at an EU-level would likely be subject to legal challenge
  - **Effectiveness and efficiency** - This would be less effective in tackling infill loss than other approaches
  - **Proportionality** - This would be a disproportionate response to tackling the loss of infill from artificial sports pitches, which can adequately be addressed through other measures
  - **Political feasibility** - There would be significant opposition to such a tax
- Mandatory EPR for artificial sports pitches whereby the installers are responsible for the entire life cycle management of the pitch
  - **Proportionality** - While desirable of itself, this would be a disproportionate response to tackling the loss of infill from artificial sports pitches, which can adequately be addressed through other measures
- An EU-wide ban on the use of tyre-derived (and other plastic) infill in artificial sports pitches
  - **Effectiveness and efficiency** - While this would be effective in tackling the issue, it would be far less efficient than other options, given the expense that would be incurred by a range of stakeholders
  - **Proportionality** - This could reasonably be argued to be a disproportionate response to the issue, especially when compared to the other options that are likely to achieve a significant reduction in infill loss while giving pitch operators flexibility in how this is achieved
  - **Political feasibility** - There would be significant opposition to such a move from key stakeholders, and there would be real political challenges in achieving such a ban

A.5.2.7 Capture of Microplastics on Roads

- A requirement for all new roads to use porous surfaces which capture vehicle tyre dust particles
  - **Effectiveness and efficiency** - It's not yet clear that such a requirement would lead to an overall benefit in respect of reducing emissions of microplastics from tyre wear

into the aquatic environment. Porous surfaces can also lead to a higher rate of tyre and road wear. Further research is required.

- Once guidance documents on street cleansing and reducing road runoff have been distributed, implement a regulation at the EU level to require Member States to implement best practices where feasible
    - **Effectiveness and efficiency** - While potentially feasible and desirable, the scale, cost and potential effectiveness of such a measure is unknowable until research has been undertaken to inform the guidance documents. It therefore cannot be taken forward for further consideration at present.
- Once guidance documents on street cleansing and reducing road runoff have been distributed, creation of a dedicated fund for improving road drainage infrastructure to capture microplastics from vehicle tyres, paid for by a fee on tyres. Such a fee could vary based on the tread abrasion rate of the tyre.
    - **Effectiveness and efficiency -** While potentially feasible and desirable, the scale, cost and potential effectiveness of such a measure is unknowable until research has been undertaken to inform the guidance documents. It therefore cannot be taken forward for further consideration at present

A.5.2.8 Capture of Microplastics in Wastewater Treatment

- Once research is completed, and if a new technique has been developed that removes microplastics from sludge, require its implementation, starting with the WWT facilities that currently contribute the greatest amount, taking into account cost-effectiveness
    - **Effectiveness and efficiency -** Without knowing whether such a technique can be developed, nor what it would cost, it is not possible at present to determine the effectiveness and efficiency
- EU to require the monitoring of CSOs in order to understand the frequency of spillages and the amount of untreated wastewater that is spilled. Such information should subsequently be made publically available, and can be used to prioritise investment in addressing such spills
    - **Proportionality** - This would be a disproportionate response to the issue of microplastics alone. However there are wider reasons for wanting such data to be gathered and made publicly available

A.6.0 Assessment of the Impact of Options

A.6.1 Automotive Tyre Wear

The options taken forwards for detailed analysis are as follows:

- **Development of a standard measure of tyre tread abrasion**
  - Such a test will be used to determine the rate at which different tyres abrade (mg/km) under standard conditions. While factors *external* to the tyre such as vehicle weight, driving style, road conditions and level of inflation all have a bearing on real world rates of abrasion, such a test will provide details on the factors that are within the control of tyre manufacturers.
  - Such a test will of itself not lead to any reduction in microplastic emissions from vehicle tyres, but it will be the basis for the subsequent measures for tyres detailed below:
- **Inclusion of tyre tread abrasion rates in the EU Tyre Label Regulation (EC/1222/2009)** (once a standard measure of tyre tread abrasion has been developed)
  - Using the standard A-G rating, this *demand-side* measure would ensure that consumers are adequately informed about the likely rate of tyre tread abrasion for each tyre placed on the market.
- **Using the Type Approval Regulation (EC/661/2009) to restrict the worst performing tyres (in respect of tyre tread abrasion) from the market** (once a standard measure of tyre tread abrasion has been developed)
  - Similar to the approach used in respect of rolling resistance, this *supply-side* measure would restrict access to the European market to those tyres that meet and exceed this threshold for tread abrasion.

In the sections below:

- We note the strong consumer demand for information about tyre tread abrasion rates;
- Identify the relevant impacts associated with the selected measures; and
- Seek to quantify the costs and the benefits.

A.6.1.1 Consumer Demand for Information about Tyre Tread Abrasion Rates

A report for DG ENER of the European Commission, published in March 2016, provides useful insights into the importance placed on the durability of tyres (which relates closely to the tyre tread abrasion rate) by consumers.[490] As part of the research, surveys (for C1 end-users) and interviews were undertaken with different actors in the tyre supply chain, including tyre suppliers

[490] Viegand Maagøe A/S (2016) Review Study on the Regulation (EC) No 1222/2009 on the Labelling of Tyres, Final Report to DG ENER of the European Commission, March 2016. Available at http://www.labellingtyres.eu/downloads/Final_report-Review_study_on_labelling_of_tyres.pdf

(manufacturers and importers), tyre dealers, vehicle suppliers and distributors and end users. End users were broken down into the following categories:

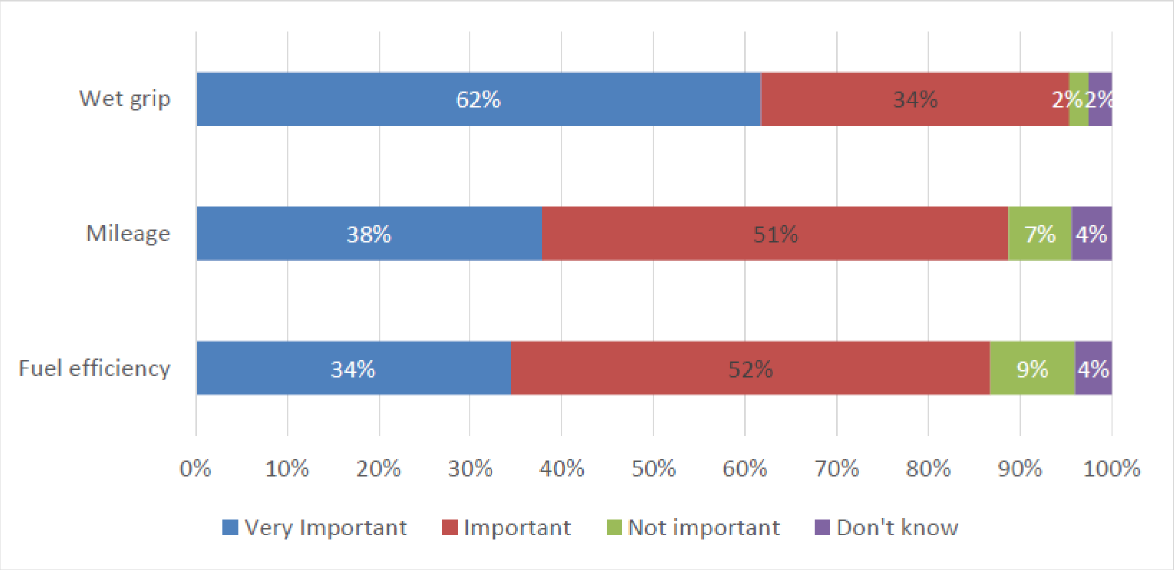
- C1 end-users, including consumers defined as private persons buying tyres for their own private cars, as well as leasing companies buying tyres for their lease cars;
- C2 end-users, defined as the purchasers and users of C2 tyres, used for light duty vehicles (LDVs); and
- C3 end-users – primarily truck fleet owners and operators

The report notes that:[491]

> *According to the dealer associations, the end-users focus primarily on safety aspects, followed by price, durability and to some extent fuel efficiency. This is in accordance with the findings from the C1 end user survey.*

The relative importance of wet grip, fuel efficiency, and 'mileage' (reflecting the durability of the tyre) as indicated in the C1 end-user survey are shown in Figure 32.[492]

Figure 32: C1 End-user Rating of Fuel Efficiency, Mileage and Wet Grip Importance



Source: Viegand Maagøe A/S, 2016

As can be seen, 38% of car owners rate mileage as being "very important" when choosing tyres, with 89% seeing it as either "important" or "very important". This is lower than for wet grip, but higher than for fuel efficiency. The authors of the study note that this indicates that:

> *Including mileage in the Tyre Labelling Regulation would add value to customers, and that it is a parameter that might affect C1 users' choice of tyres.*

---

[491] Viegand Maagøe A/S (2016) Review Study on the Regulation (EC) No 1222/2009 on the Labelling of Tyres, Final Report to DG ENER of the European Commission, March 2016. Available at http://www.labellingtyres.eu/downloads/Final_report-Review_study_on_labelling_of_tyres.pdf

[492] The authors note that 'mileage is a common parameter used to express the durability of tyres as a distance in miles or kilometres', and that the mileage of a tyre is directly correlated to the tyre wear factor (amount of tread lost per kilometre). The authors also note that abrasion (i.e. the removal of materials from the tyre when it interacts with the road surface) 'is related to tyre mileage, since both are linked closely to the tyre wear.'

In respect of C2 end-users, the report states that:

> *Their number one priority is what they in general refer to as tyre "quality", closely followed by the price. The quality, from their point of view, meaning tyre mileage/tread wear and the tyre brand.*

The views of C3 end-users on the Tyre Labelling criteria, were reported as follows:

> *66% thought fuel efficiency was the most important, and 30% that it was wet grip. Most of the respondents wanted some kind of wear-related information on the label (86%), while 20% wanted retreadability to be indicated on the label.*

The Commission's own Impact Assessment from 2008 on the labelling of tyres also notes the importance that consumers place on this aspect, stating that:[493]

> *Market surveys in addition show that wear (i.e. "long lasting tyre") is the most important criteria in consumers purchasing decision.*

However, at the time, inclusion of tread wear (described in the IA as the life duration of a tyre, which is slightly different, but clearly related to the tyre tread abrasion rate) was not taken any further for the following two reasons:[494]

- *The life duration (tread wear) of tyres is a parameter consumers can take clear notice of when they change their tyres. Market surveys show that "long-lasting tyres" constitute the most important criterion in consumers' purchasing decisions. The market is therefore considered to be self-regulating: no established tyre manufacturer would take the risk of decreasing the lifetime of a tyre, knowing that consumers will notice it (even if a few years after the purchasing decision) and lose confidence in the brand*
- *There are no reliable testing methods for tread wear and testing would be very costly (due to the necessity to test tyres over 10,000km)*

The IA continued by making the point that:

> *Aware of the necessity to avoid the potential adverse impact on the environment of reduced tread wear (which would increase tyre waste, hence compensate for the gains due to fuel efficiency), tyre experts agreed that it may be necessary, some time after the implementation of a labelling scheme, to reassess the situation and its impact on tread wear*

Of course our understanding has moved on, and we now know that the rate at which tyre treads wear, i.e. the tyre tread abrasion rate, is of environmental concern not only because of the potential effect on tyre lifetimes, and therefore waste, but also due to the rate at which microplastic particles may be generated. These of course can lead to adverse impacts both on air quality, and also as microplastics that may enter the soil and water.

It's worth challenging the assertion that 'the market is self-regulating' in respect of tyre tread abrasion rates. While some consumers may notice that the last set of tyres purchased may have

[493] European Commission (2008) Commission Staff Working Document: Accompanying document to the Proposal for a Directive of the European Parliament and of the Council on Labelling of Tyres with Respect to Fuel Efficiency and Other Essential Parameters – Impact Assessment, 13.11.2008

[494] European Commission (2008) Commission Staff Working Document: Accompanying document to the Proposal for a Directive of the European Parliament and of the Council on Labelling of Tyres with Respect to Fuel Efficiency and Other Essential Parameters – Impact Assessment, 13.11.2008

worn quicker than the previous set, and thus revert to the former type, this does not allow them to understand whether alternative tyres may offer improve performance in respect of tyre tread abrasion rates. It is certainly no substitute for a clear labelling system that will enable tyre tread abrasion rates to be compared across all tyre types prior to purchase.

Accordingly, in the absence of labelling of tyre tread abrasion rates, there is a market failure arising from the lack of information for end-users. This market failure affects:

- **End-users** (consumers, companies or municipalities owning small or larger fleets such as leasing companies, and road transport operators) who do not benefit from the savings they could obtain from the use of tyre with lower tread abrasion rates;
- **Tyre producers** who have more difficulties in obtaining a return on their investments in R&D to reduce the tyre tread abrasion rate; and
- **Society as a whole** resulting from a reduced rate of tyre tread abrasion which is expected to reduce the rate at which particles are generated from the usage of tyres, with associated benefits in terms of air quality and marine microplastics.

Labelling of tyre tread abrasion rates could also improve competition between tyre producers while providing a level playing field for all. Producers may both have incentives to provide better-performing tyres on the market and benefit from reduced barriers to entry as brand reputation may lose its importance compared to objective tyre performance characteristics. New entrants will be able to demonstrate that they produce well-performing tyres in respect of tread abrasion rates.

A.6.1.2 Development of a Standard Measure of Tyre Tread Abrasion Rate

The development of a standard measure of tyre tread abrasion is an essential pre-condition for the successful introduction of either of the proposed subsequent measures, namely:

- Inclusion of tyre tread abrasion rates in the EU Tyre Label Regulation (EC/1222/2009); and
- Using the Type Approval Regulation (EC/661/2009) to restrict the worst performing tyres (in respect of tyre tread abrasion) from the market.

Accordingly, we identify the costs associated with the development of a standard measure of tyre tread abrasion rate, and to whom these costs might accrue, but we do not consider any benefits. These will be considered in respect of the application of the standard measure of tyre tread abrasion rate in the two subsequent measures.

Is it Possible to Test for Abrasion?

Several laboratory-based tests exist to estimate the abrasion potential of rubber material (DIN, Taber, NBS, PICO Abrasion tests).[495] The most common test used in Europe to measure tyre tread abrasion rates is the Rotary Drum Abrasion (DIN Abrasion) method, as this is the method that is certified under the ISO 4649 2010 standard for determining abrasion resistance of vulcanized or

[495] *Abrasion | Rubber and Elastomer Testing - Physical Properties | Material Testing | Smithers Rapra*, accessed 27 September 2017, http://www.smithersrapra.com/testing-services/by-material/rubber-and-elastomer-physical-testing/abrasion

thermoplastic rubber. This method consists of rubbing a test piece of rubber, such as from a vehicle tyre, over an abrasive sheet of specified grade laid over a cylindrical curved surface which rotates. The test piece may be rotating or non-rotating. The loss in mass of the test piece is determined at the end of the test, and a volume loss can be calculated using the density of the material used for the test piece. The volume loss is then compared against that of a reference compound tested under the same conditions.[496][497]

However, the results of laboratory abrasion tests (LATs) are not always indicative of actual rates of tyre abrasion in real world conditions. Discussion with a number of experts at the Tun Abdul Razak Research Centre (TARRC) in the UK confirms that what happens on the road is much more complicated than the typical tests performed in the laboratory, and that a compound that performs well in a LAT may not perform well on the road. The real challenge is reported to be in developing a LAT that accurately replicates the same kind of tyre abrasion observed on the road.[498] The focus at present is thus on seeking to more fully understand the mechanism of tyre abrasion on the road, and replicate this in LATs. It was noted that all the major brands already undertake such testing, albeit in slightly different ways, with one brand (at least) reported to have developed a test that in the view of the TARRC experts, replicated reasonably well tyre tread abrasion seen in on the road conditions.

An alternative to a LAT is to undertake a road test of tyres, which would typically take place over a period of 3 to 6 months and cover a distance of 3,000 miles.[499] Under this approach, the tyre is removed, deflated and cleaned every day, and then weighed, in order to determine the loss of tread before being re-inflated and placed back on the vehicle.[500] Using such a test, it was felt, would enable a reasonable level of precision in determining the tyre tread abrasion rate, albeit, as always, additional replications would help to increase the precision.[501]

Such a road based test would cost between €5,000 and €10,000 per tyre, while the cost of a LAT would be closer to €1,000 to €2,000 per tyre.

Continental provided indicative figures of its own testing procedures for passenger vehicles and light trucks, which are undertaken on a set on-road route, incorporating urban roads, and stretches of autobahn. This typically involves 4-6 vehicles travelling in convoy, with drivers and wheels being changed to different vehicles on a regular basis in order that any characteristics specific to the driver or vehicle do not skew the test results. Following the pre-defined route, the vehicles are driven for

---

[496] Koike et al (2001) *A new type of tyre tester. Evaluation of tread wear resistance by laboratory testing.*, 2001, http://www.polymerjournals.com/pdfdownload/854489.pdf

[497] ISO (2010) *ISO 4649 2010: Rubber, vulcanized or thermoplastic - determination of abrasion resistance using a rotating cylindrical drum device*, 2010, http://msrpco.ir/wp-content/uploads/2017/07/ISO-4649-2010.pdf

[498] Personal communication with Dr Stuart Cook (Director of Research), Paul Brown (Head, Advanced Materials & Product Development), Dr Andy Chapman (Senior Research Fellow), and Dr Pamela Martin (Advanced Materials and Product Development), Tun Abdul Razak Research Centre, October 2017

[499] It was also noted that accelerated road tests, simulating driving 120-150 thousand kilometres, can deliver reliable results within a few weeks. These use tyres attached to a trailer, with changes being made at regular intervals to the angle of the tyre in order to simulate normal wear in a shorter time than usual.

[500] Such an approach tends to be used for cars and other light vehicles. For HGVs, given the difficulties in removing, deflating, cleaning and replacing and re-inflating tyres, the tyre is left *in situ* and 24 measurements of tread depth are taken each day.

[501] Personal communication with Dr Stuart Cook (Director of Research), Paul Brown (Head, Advanced Materials & Product Development), Dr Andy Chapman (Senior Research Fellow), and Dr Pamela Martin (Advanced Materials and Product Development), Tun Abdul Razak Research Centre, October 2017

eight hours a day, with pre-determined breaks being taken. Overall, the test covers 20,000 km, at a total cost of €40,000 per tyre being tested.[502]

Tread depth is measured in 12 positions on each tyre on a daily basis, and the tyres are weighed regularly, but not daily. It was stated that testing could be undertaken over a shorter distance (5,000km was presented as an example), but that this would not provide a useful basis for extrapolation.[503]

Clearly, while there are tests available, the challenge is to improve the way in which the LATs reflect real 'on the road' abrasion rates in order to reduce the costs, over time, of testing. This is an area of active research, and with the increased public interest and regulatory attention being paid to this issue, it would seem reasonable to assume that future research into this area will intensify.

Is it Possible to Develop a Standardised Test?

While testing for tyre tread abrasion already takes place, in order to enable the development of a tyre label rating for tyre tread abrasion, a standardised test procedure would have to be undertaken.

The ETRMA notes that there are numerous challenges in developing a standardised test:[504]

- Standardising the multitude of different driving conditions (weather, road…) in a single test;
- The results of the test method under standardised conditions [may not fully] replicate the broad variation in real life driving conditions (risking to repeat the same event well known as "Diesel Gate"); and
- Even if a repeatable, reproducible and cost-efficient method could be developed, enforcement by authorities remains to be demonstrated.

The European Commission's 2008 Impact Assessment noted that in order to be properly estimated, industry representatives identified that on-the-road testing would be required with an estimate presented that over 10,000 km would need to be travelled to get significant results.[505]

However, a bigger challenge in developing a standardised test, put forward in discussions with experts at TARRC was in getting the major brands to agree to a standardised test procedure. They will all have invested over time (perhaps to differing degrees) in developing their own approach to determining the likely rate of tyre tread abrasion for their own tyres, and may be reluctant to switch over to a different method. In large part such resistance may be because they would prefer to use their own approach in order to have continuity of data with that previously gathered. There may also be a concern that in divulging their own preferred test method, which presumably will be close to their own current approach, they may be giving away sensitive data that they feel gives them a competitive advantage at present.

Accordingly, it would seem possible to develop a standardised test given that:

- Road based testing of abrasion can already be delivered;

---

[502] Personal communication with Joerg Burfien, Head of Global Standards and Regulations, Continental AG

[503] Personal communication with Joerg Burfien, Head of Global Standards and Regulations, Continental AG

[504] ETRMA Response to the Public Consultation Investigating Options for Reducing Releases to the Environment of Microplastics (12/9/2017)

[505] European Commission (2008) Commission Staff Working Document: Accompanying document to the Proposal for a Directive of the European Parliament and of the Council on Labelling of Tyres with Respect to Fuel Efficiency and Other Essential Parameters – Impact Assessment, 13.11.2008

- Laboratory abrasion tests are being refined so as to more closely reflect the real world conditions of tyres on the road; and
- Major brands already undertake their own tests to determine tyre abrasion.

The real challenge will be one of agreeing which approach to take. Furthermore, while initially road based tests at €5,000 to €10,000, or perhaps up to €40,000 per tyre may be required, it can be anticipated that the requirement for a standardised test will lead to technical innovation with respect to LATs, meaning that the cost may reduce to be closer to €1,000 to €2,000 per tyre over time.

Given that there will be time and resource implications associated with the need to agree a standardised test method, a high level estimate of €500,000 is made to account for the costs, which, we anticipate would fall largely on industry.

How Much Would Testing Add to the Cost of a Tyre?

The cost to test one tyre can be between €5,000 and €10,000 or up to €40,000 for a road based test, depending on the distance covered, while the cost of a LAT would be closer to €1,000 to €2,000 per tyre.

The recent review for DG ENER on tyre labelling forecasts light vehicle (C1+C2) tyre sales of 280 million by 2020.[506] The study also notes that no complete database exists at the EU level for the tyre market, but references two large databases:

10) The German Tyres Online (TOL) database. Germany is the largest EU-28 country and the location of many tyre producers and importers. The data contains the 29 largest tyre brands in all sizes for the years 2012 to 2015, and in total there is data on almost 30,000 tyres.
11) The database from the Dutch Tyre and Wheel Trade Association (VACO). A large part of European tyre trade goes through the Netherlands, and most of the tyres in the database are sold in other European countries. The data from VACO (used in the Viegand Maagøe A/S study for DG ENER) is for the years 2013 to 2015 and includes the top seven brands (Michelin, Continental, Bridgestone, Goodyear, Dunlop, Pirelli, Hankook, Vredenstein) in the seven most sold sizes. In total the VACO dataset contains data of around 2,500 tyre models.

This data can be used in order to determine the average additional cost per tyre of testing.

If we assume that the 280 million tyres placed on the market in 2020 are divided equally between the 30,000 tyre models listed in the TOL database, this means there will be 9,333 tyres of each model placed on the market in that year. If we further assume, arguably conservatively, that each *model* of tyre is available on the market only for 3 years, this would mean 28,000 tyres of each model produced. Assuming €10,000 for each test, the additional costs would be circa €0.36 per individual tyre placed on the market. At €40,000 for each test, the additional costs would be circa €1.43 per individual tyre placed on the market

For more popular models of tyre, the cost per tyre will be lower, while for less popular tyres the cost per tyre will be higher.

[506] Viegand Maagøe A/S (2016) Review Study on the Regulation (EC) No 1222/2009 on the Labelling of Tyres, Final Report to DG ENER of the European Commission, March 2016. Available at http://www.labellingtyres.eu/downloads/Final_report-Review_study_on_labelling_of_tyres.pdf

If we apply the same approach to the VACO data, and assume that the 280 million tyres placed on the market in 2020 are divided equally between the 2,500 tyre models listed, this means there will be 112,000 tyres of each model placed on the market in that year. If we further assume, arguably conservatively, that each *model* of tyre is available on the market only for 3 years, this would mean 336,000 tyres of each model produced. Assuming €10,000 for each test, the additional costs would be circa €0.03 per individual tyre placed on the market. Assuming €40,000 for each test, the additional costs would be circa €0.12 per individual tyre placed on the market.

Accordingly, the average cost of testing will be between €0.03 and €1.43 per individual tyre placed on the market, but higher for some less popular models, and lower for the most popular models.

Potential for Incorporating Revised Wet Grip, Rolling Resistance and External Noise Tests

Testing for rolling resistance, wet grip and external noise is currently undertaken on new tyres.[507] That is to say that there is no testing for the performance against any of these criteria over the lifespan of a tyre, meaning the consumer does not have any indication as to the extent to which these properties (of wet grip, rolling resistance and external noise) will vary over the tyre's lifetime. However, there is an argument that such information is of importance to the consumer. Indeed, the UK's Automobile Association notes, in its advice to drivers, that:[508]

> *Wet grip in particular gets worse as the tread on your tyres wears*

Given that safety (and rolling resistance and external noise) is of concern to motorists throughout the lifetime of the tyre, and not just when it is new, there is arguably a case for the wet grip and other tests to be revised and incorporated within a new test for tyre tread abrasion rates.

For example, if the test were to be conducted over 20,000km, wet grip, rolling resistance and external noise could each be tested at the outset, then at 10,000kms and at 20,000kms. Such testing would appear likely to give a better indication of the lifetime performance of the tyre.

Revising the tests in this way, to operate alongside a new test for tyre abrasion, would seem to offer a number of possible co-benefits.

Measuring Airborne PM Emissions from Tyres

A recent study by the JRC estimated that 0.1 – 10% of tyre wear is emitted as airborne PM_{10}. In this study, we calculated the PM_{10} emissions from tyres for each European country, separated by vehicle and road type.[509] Assuming PM_{10} represents on average 5% of all tyre wear, we conclude that **25,179 tonnes of PM_{10} are emitted in Europe per year** (EU Member States (minus Bulgaria and Cyprus, for which data was not available) + Norway).

Applying the country specific damage costs from a recent study by the EEA to the country specific PM_{10} emissions from tyres gives an indication of the damage costs. [510] The total damage costs in

[507] See https://www.blackcircles.com/general/tyre-labelling/tyre-testing

[508] See https://www.theaa.com/driving-advice/safety/tyre-life-and-age

[509] European Commission (2014) Non-exhaust Traffic Related Emission: Brake and Tyre Wear PM, JRC Science and Policy Report, available at http://publications.jrc.ec.europa.eu/repository/bitstream/JRC89231/jrc89231-online%20final%20version%202.pdf

[510] European Environment Agency (2014) Revealing the Costs of Air Pollution from Industrial Facilities in Europe, EEA Technical Report No 15/2011, Table A1.9, using 2020 'Low' and 'High' Estimates in uninflated 2005 prices available at https://www.eea.europa.eu/publications/cost-of-air-pollution

Europe per year from airborne PM_{10} from vehicle tyres are thus estimated to range from 540 million EUR to 1.5 billion EUR. It is important to note that the damage costs in the EEA study are for industrial facilities, which tend to be remote from urban centres and release particulate matter high into the atmosphere. Average damage costs for transport emissions are almost double those for industrial emissions and damage costs in dense urban areas, such as in central London are almost nine times greater (see Table 104).[511] Hence the damage costs from PM emissions from tyres are likely to be more than double than those presented above, meaning that they would range from 1 to 3 billion EUR per year.

Table 104: DEFRA estimates of PM10 Damage Costs by location and source (£/tonne, 2015 prices)

	Central	Central sensitivities	
		Low	High
Transport average	**58,125**	**45,510**	**66,052**
Transport central London	265,637	207,981	301,859
Transport inner London	273,193	213,898	310,447
Transport inner London	178,447	139,717	202,781
Transport inner London	141,248	110,590	160,507
Transport outer conurbation	87,770	68,722	99,739
Transport urban big	104,627	81,918	118,895
Transport urban large	84,283	65,989	95,776
Transport urban medium	66,264	51,881	75,300
Transport urban small	41,850	32,768	47,557
Transport rural	18,020	14,108	20,476
Industry	**30,225**	**23,665**	**34,347**
Domestic	**33,713**	**26,396**	**38,311**

Source: DEFRA (2015), Air Quality Economic Analysis - Damage Costs by Location and Source

Furthermore, these estimates are based on damage costs for PM_{10} emissions. However, PM_{10} emissions include a $PM_{2.5}$ fraction. The damage costs associated with $PM_{2.5}$ are much higher, as these can be inhaled into the lungs and travel through the bloodstream to the brain. Hence applying PM_{10} damage costs to the total emissions provides a conservative estimate, disregarding the higher damage costs of the $PM_{2.5}$ fraction.

It can thus be seen that if measures to encourage reduced levels of tyre abrasion also lead to a reduction in airborne PM emissions, there could be an accompanying air quality benefit. However, while it is would seem reasonable to expect that this might occur, and that the mass of airborne PM might well decline, this might be achieved through a shift away from larger coarse particles (PM_{10} larger than $PM_{2.5}$) and towards $PM_{2.5}$. The distribution of airborne particle sizes from tyres is currently not fully understood. A number of different approaches to testing are used, but there is

511 DEFRA (2015) *Air Quality Economic Analysis - Damage Costs by Location and Source*, December 2015, https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/460398/air-quality-econanalysis-damagecost.pdf

not yet a standardised approach to measuring the amount and distribution of airborne PM from vehicle tyres.[512]

However, there may be merit in seeking to establish a test for measuring the amount and distribution of airborne PM from vehicle tyres to be run alongside the test for tyre abrasion rates. Otherwise, there is a risk that reducing abrasion rates could lead to an increase in airborne $PM_{2.5}$ emissions from tyre wear, with negative associated air quality impacts.

A.6.1.3 Include Tyre Tread Abrasion Rates on the Tyre Label

Including tyre tread abrasion rates on the tyre label is a demand-side measure, which will inform consumer choice. (By contrast, restricting the availability of tyres with the highest rates of abrasion under the type-approval legislation is a supply-side measure).

As the costs of display of the labelling scheme, as identified in the 2008 Impact Assessment are considered to be marginal on a per tyre basis, it is anticipated that the *additional* costs associated with inclusion of tyre tread abrasion rates will be similarly marginal.[513]

Given that no standardised test is currently available to illustrate the distribution of tyre tread abrasion rates across the current stock of tyres on the EU market, it is not possible to fully characterise the range in abrasion rates across all tyre models. However, it is possible to obtain an indicative estimate of the distribution using data from the Uniform Tire Quality Grading (UTQG) test used to measure tread wear in the United States. The UTQG tread wear test provides a numerical index of how well a tyre wears in comparison to a reference tyre, which is graded 100. If a candidate tyre is graded 100, the tread wears with the same rate as the reference tyre, while if it is graded 200 the tread wears at half the rate.

However, under the UTQG, each brand is able to supply its own reference tyre. This means that the UTQG cannot be used to compare tyres *between* brands, but does give an indication of tyre tread abrasion rates *within* brands.

The US Government has a searchable database on tyre ratings.[514] This database was used to obtain data on two major brands that have a presence in both the US and EU markets – Continental and Bridgestone.

Of the 249 Continental tyres listed in the database, the number in each UTQG rating 'band' is shown in Figure 33. The lowest rating is 240 and the highest is 640.

---

[512] See European Commission (2014) Non-exhaust Traffic Related Emission: Brake and Tyre Wear PM, JRC Science and Policy Report, available at http://publications.jrc.ec.europa.eu/repository/bitstream/JRC89231/jrc89231-online%20final%20version%202.pdf

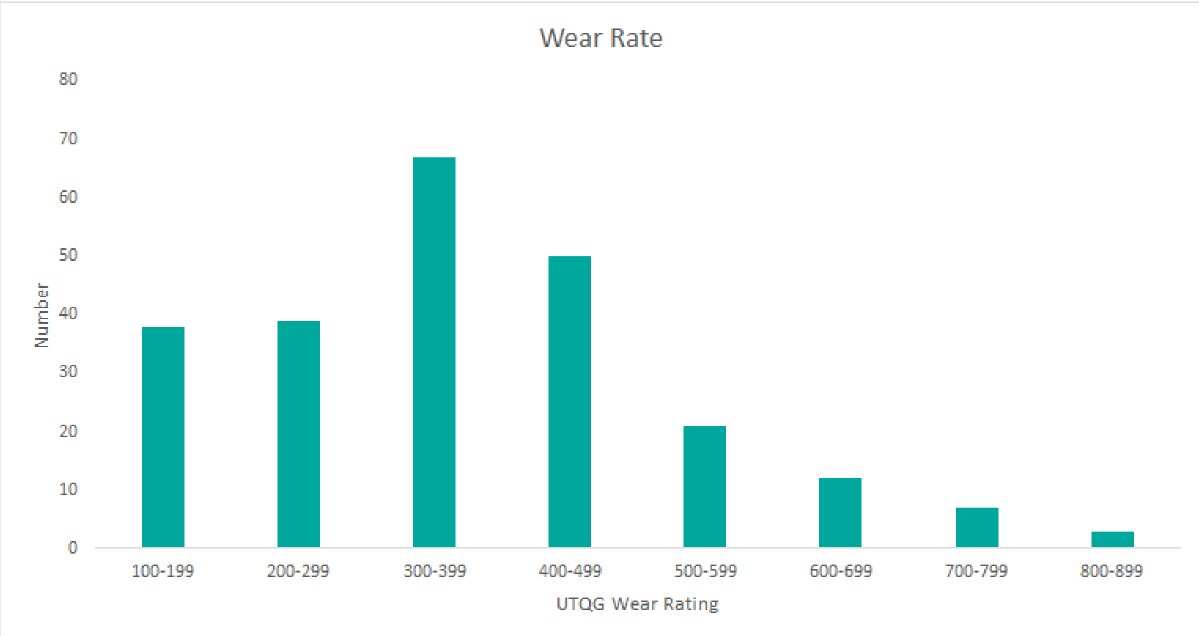
[513] European Commission (2008) Commission Staff Working Document: Accompanying document to the Proposal for a Directive of the European Parliament and of the Council on Labelling of Tyres with Respect to Fuel Efficiency and Other Essential Parameters – Impact Assessment, 13.11.2008

[514] See https://www.safercar.gov/Vehicle-Shoppers/Tires-Rating/Search?brand=Bridgestone&traction=any&temperature=any&wear=any

Figure 33: UTQG Ratings of 249 Continental Tyres on the US Market

Of the 234 Bridgestone tyres listed in the database, the number in each UTQG rating 'band' is shown in Figure 34. The lowest rating is 120 and the highest is 800. We understand that this distribution, observed in the US market for Bridgestone tyres, is likely to be similar to that in the EU market.[515]

Figure 34: UTQG Ratings of 234 Bridgestone Tyres on the US Market



An aspect of the UTQG rating system is that a higher tread wear rating can be achieved through having a deeper tread (without necessarily having a reduced abrasion rate). Normalising, by

[515] Personal communication with Alessandro Cascini, Head of Public Affairs, Bridgestone Europe NV/SA, 24th July 2017

adjusting for tread depth on a sample of these tyres, reduces the differential between the best and worst performing in respect of tread wear.

We therefore assume, arguably conservatively, that the worst performing tyres exhibit double the abrasion rate (mg/km) of the best performing tyres.

Possible Extent of Market Transformation

The existence of a rating for abrasion rate on the Tyre Label could lead to two effects:

12) A move by consumers towards existing tyre models that exhibit a lower abrasion rate; and
13) A move on the part of producers towards the manufacture of tyres that have a lower abrasion rate than current models.

The combined effect of these will be an overall reduction in the average rate of tyre abrasion. However, the speed of the market transformation will depend upon a number of factors including:

- The number of tyre models exhibiting higher than average performance in respect of abrasion rates (i.e. lower abrasion rates than the average);
- The cost of these models relative to other models on the market; and
- The performance of these models in respect of the other attributes detailed in the Tyre Label (i.e. their performance in terms of wet grip, rolling resistance, and external noise).

As shown in Section A.6.1.1, the tyre tread abrasion rate is important to consumers, given that this is a contributory factor to the durability of a tyre. However, as there is (as yet) no standard test for abrasion rate, and it is not yet included in the Tyre Label, it is not possible to identify and account for the factors noted above, and see, for example, how many tyres with lower abrasion rates also perform well on the other attributes, and how much they cost relative to other tyres that have a higher abrasion rate.

It's worth noting that the study for DG ENER in support of a review of the Tyre Labelling Regulation identified rates of market transformation over the period 2013 – 2015 in respect of fuel efficiency (rolling resistance coefficient) and wet grip, as shown in Table 105 and Table 106 respectively.[516]

Table 105: Annual Percentage Changes in Fuel Efficiency (RRC) of Tyres on the EU Market (2013-2015)

Class	Based on TOL Database	Based on VACO Database
C1	-0.4%	-1.1%
C2	-1.2%	-1.1%
C3	-0.5%	-1.3%

Source: Viegand Maagøe A/S, 2016

[516] Viegand Maagøe A/S (2016) Review Study on the Regulation (EC) No 1222/2009 on the Labelling of Tyres, Final Report to DG ENER of the European Commission, March 2016. Available at http://www.labellingtyres.eu/downloads/Final_report-Review_study_on_labelling_of_tyres.pdf

Table 106: Annual Percentage Changes in Wet Grip Values of Tyres on the EU Market (2013-2015)

Class	Based on TOL Database	Based on VACO Database
C1	0.6%	1.0%
C2	0.5%	0.6%
C3	0.51%	1.3%

Source: Viegand Maagøe A/S, 2016

A.6.1.4 Using the Type Approval Regulation to restrict the worst performing tyres (in respect of tyre tread abrasion) from the market

Restricting the availability of tyres with the highest rates of abrasion under the type-approval legislation would be a supply-side measure. (By contrast, including tyre tread abrasion rates on the tyre label is a demand-side measure).

In the absence of a standardised test, and resulting performance data in respect of tyre abrasion, and the distribution of tyre abrasion rates across the EU market, it is not, at present, possible to identify where a 'reasonable' threshold might be for limiting access to the EU market.

Determining such a point, and providing supporting justification for it, would have to take into account the other performance criteria of relevance alongside tyre abrasion rates.

Accordingly, for illustrative purposes, we provide an indication of the effect of using the Type Approval Regulation to prevent, in 2020, the worst performing tyres from being placed on the market, such that the effect is a 10% drop in the tonnage of tyre wear abraded at source. We further illustrate the effect of a similar incremental restriction coming into place in 2025.

A.6.2 Pre-Production Plastics

The options taken forwards for detailed analysis are as follows:

- **Amending the Polymer Production BREF to include best practice pellet loss prevention measures as BAT**
  - MS to address prevention and control of releases of micro-plastics to water as part of the revision of permits of plastics production plants required by the IED to align industry practices with Best Available Techniques; Include in future work on BREFs regarding water pollution from the plastics industry the identification of BAT for prevention and control of releases of micro-plastics to water.
  - This approach could incorporate the best practice measures as described in Operation Clean Sweep - and subsequently verified and enhanced through an expert group – into the Polymer Production Best Available Technique (BAT) Reference Document (BREF) under the Industrial Emissions Directive (IED).

    - All polymer producers in Europe would thus be required to implement BAT in respect of preventing pellet loss, and would be subject to regulation and potential enforcement action as per other aspects of their Environmental Permit
- **Regulation on the Transport of Pellets**
    - This would be a new regulation specifically covering the transport of pellets from and to facilities. All operators undertaking such transportation would be required to implement best practice approaches, again derived from expert knowledge, and further developing the approaches already pioneered by industry via Operation Clean Sweep.
- **Regulation on Plastic Converters**
    - The circa 50,000 plastics converters in the EU are mostly SMEs to whom the polymer production BREF does not apply. This new regulation would thus require all plastic converters in the EU to implement best practice measures to prevent pellet loss. Environmental regulators in each Member State would be required to ensure adherence to the Regulation
- **Regulation Requiring Supply Chain Accreditation of Adherence to Best Practice**
    - This regulatory measure would require those placing plastics on the market (large businesses in the first instance) to ensure their entire supply chain demonstrates best practice in the prevention of pellet loss.
    - Akin to the way in which the Timber Regulations operates, adherence to best practice can be demonstrated through the use of accreditation bodies that certify adherence to best practice criteria.
    - This measure would include anyone directly placing plastics products on the market that were manufactured outside the EU, thus ensuring a level-playing field between the EU plastics producers and converters, and those outside of the EU wishing to sell in.

In the sections below:

- We consider the structure of the pre-production plastics supply chain in the EU, and what this means in respect of possible measures;
- Identify the relevant impacts associated with the selected measures; and
- Seek to quantify the costs and the benefits.

A.6.2.1 Structure of the Plastics Industry

Polymer producers only represent a small proportion of the companies within the European plastics industry. The Polymer Production Best Available Technique Reference Document (BREF) under the Industrial Emissions Directive (IED) is focused on polymer producers. As stated in the BREF:[517]

> *Polymer companies produce a variety of basic products, which range from commodities to high added-value materials and are produced in both batch and continuous processes covering installations with a capacity of some 10000 tonnes per year up to some 300000 tonnes per year.*

[517] European Commission (2007) Reference Document on Best Available Techniques in the Production of Polymers, August 2007, available at http://eippcb.jrc.ec.europa.eu/reference/BREF/pol_bref_0807.pdf

The basic polymers are sold to processing companies, serving an immense range of end-user markets.

Plastics Europe notes that the European plastics industry comprises 60,000 companies, ***mainly small and medium enterprises in the converting sector***. Polymer producers are represented by Plastics Europe, converters are represented by European Plastics Converters (EuPC) and machine manufacturers are represented by EUROMAP.[518]

Plastics Europe notes that its members are among the most important polymer producers in the world, and indicates that 54 companies are members.[519] EUROMAP represents around 1,000 companies.[520]

EuPC represents close to 50,000 companies, and states that:[521]

Plastics converters (sometimes called "Processors") are the heart of the plastics industry. They manufacture plastics semi-finished and finished products for an extremely wide range of industrial and consumer markets - the automotive electrical and electronic, packaging, construction and healthcare industries, to name but a few.

Plastics Converters buy in raw material in granular or powder form, subject it to a process involving pressure, heat and/or chemistry and apply design expertise to manufacture their products. They often undertake additional finishing operations such as printing and assembly work to add further value to their activities

Accordingly, the vast majority of the companies in the sector are small and medium sized enterprises (SMEs).

Given the European Commission's desire to minimize regulatory burden on SMEs, this presents some interesting challenges when considering the appropriate policy response(s).[522] It is informative, in this regard, to understand the views of plastic converters about the current regulatory environment. A recent report on the competitiveness of the European plastics converting industry, produced for EuPC, offers some useful insights.[523]

The authors of the report undertook a survey of a representative sample of 326 EU plastics converters from 19 European countries and more than 20 expert interviews with mostly senior company representatives. The authors, in presenting their findings note the view that:

The bureaucratic and regulatory framework conditions within the EU are assessed as mostly stable for plastics converters. Nevertheless, cost burdens from direct taxes or necessary effort

---

[518] Plastics Europe (2017) The European Plastics Industry, available at http://www.plasticseurope.org/plastics-industry.aspx

[519] Plastics Europe (2017) Our Members, available at http://www.plasticseurope.org/plastics-industry/our-members.aspx

[520] EUROMAP (2017) About EUROMAP, available at http://www.euromap.org/about-us/about-euromap

[521] EuPC (2017) EuPC homepage, available at http://www.plasticsconverters.eu/

[522] See European Commission (2011) Minimizing Regulatory Burden for SMEs: Report from the Commission to the Council and the Council and the European Parliament, 23.11.2011, available at http://ec.europa.eu/smart-regulation/better_regulation/documents/minimizing_burden_sme_en.pdf

[523] Dr. Wieselhuber & Partner GmbH (2016) Competitiveness of the European Plastics Converting Industry: A European Industry Study. Report to EuPC, June 2016, available at https://www.agoria.be/www1.wsc/webextra/prg/nwAttach?appl=enewsv6&enewsdetid=189927&attach=Attach110523001.pdf

to comply with domestic and EU-driven regulations and requirements have worsened substantially compared to previous years. This development poses a massive threat to the competitiveness of EU plastics converters. Still, most converters expect a further worsening of the situation.

This is not a surprising view to be expressed, given the desire of any industry to avoid further regulation. However, of greater relevance to the question of preventing pellet loss, the authors go on to report the view of plastic converters that:

The level of fragmentation from domestic legislation, regulations and bylaws, driving the framework conditions, is assessed as too high and still far from a perfectly harmonized European single market. The root cause for this fragmentation can be found within the member states. EU legislation, by nature aiming at a legislative level playing field, is slowly or sometimes even not adopted to national law by the member states. Other factors further pushing the level of fragmentation are different domestic bylaws and authorities charged with the enforcement of legislation. Thus, companies need to adjust to these differences within the EU market with additional administrative effort. Key drivers for this fragmentation on a national level are different requirements for consumer safety, the use of raw materials, for processing technologies and approvals to sell different plastic products.

This strongly suggests that any policy measures that seek to reduce the loss of pellets from converters should be consistently applied in order to safeguard the functioning of the European single market, in order to minimise the impact on SMEs.

Loss Rates at Different Stages of the Supply Chain

While it is not possible to identify a specific figure, we suspect that percentage losses at plastics producers are likely to be towards the lower end of the 0.01-0.04% range (as identified in Section 2.2.3), while those at converters (including intermediary facilities) may well be towards the higher end of this range. This is for the following three reasons:

- Current level of regulatory attention;
- Level of public scrutiny; and
- Engagement with Operation Clean Sweep

On the first of these, polymer producers, which tend to be large in size, are already regulated under the Industrial Emissions Directive. While pellet loss prevention measures are not specifically included within the Polymer Production BREF, one might reasonably expect that facilities that are more closely regulated may well have better operational procedures. This is not something that can be demonstrated, but discussions with some stakeholders indicate a view that this is the case.

Secondly, to the extent that the pre-production plastics supply chain is visible to the public, it is the larger producers that will be more well-known, and have in place more stringent Corporate Social Responsibility (CSR) reporting. Again, this suggests a greater effort may be expended on maintaining reputations.

Thirdly, and related to the last point, engagement with Operation Clean Sweep varies. Specifically, Plastics Europe's promotion of Operation Clean Sweep has been considerable in recent years and months. It is understood that 50 percent of Plastics Europe members to whom OCS is applicable have signed the pledge. By volume, this accounts for the majority of plastics production in Europe and the target is for 100% coverage by 2017. By contrast, we understand from some NGOs that EuPC (European Plastics Converters), the trade association for the circa 50,000 plastic converters in the EU, has not been open to engagement with them to the same extent as has Plastics Europe.

This view was supported by stakeholders during the workshop on preventing the loss of pre-production plastics (pellets, powders and flakes) held in Brussels on 27th September 2017.

A.6.2.2 Costs of Preventing Pellet Loss

The physical changes and improvements to management practices required to prevent pellet spills in the first place, and ensure that any spills are promptly cleaned up or captured, are well characterised. A number of specialist companies already provide services to assist in identifying where pellet management practices are sub-standard, and how pellet loss may be most cost-effectively prevented.

A provider of such services estimated that in his experience it would cost circa €10,000 to 'seal' a small facility that undertakes plastic converting and thus effectively prevent all pellet loss. A small plant in his terminology would be one that handles up to 10,000 tonnes per annum. Therefore the capital cost of the measures needed to prevent pellet loss would be around €1/tonne of annual capacity for such facilities.

This would include ensuring that all delivery systems are correctly specified, and that the handling of incoming pellets is properly undertaken, along with installing measures such as screen drains, which are standard products but customised to each site.

It is reported that there would be economies of scale, meaning that the capital costs of the measures to prevent pellet loss from larger facilities would be lower than the €1/tonne of annual capacity noted for the small plant (that handles up to 10 kt/annum). While a larger plant would be expected to have more points of spillage, it would reportedly be easier to organise the plant actions to prevent pellet loss.[524]

Sticking with the higher cost estimate (for smaller facilities) of around €1/tonne of annual capacity, on the basis that the loss rate is between 0.01% and 0.04% at producers and converters, this scales up to a one-off capital cost of between €2,500 and €10,000 to prevent the loss of a tonne of pellets, not only in the first year, but in each future year of operation of the facility (assuming the measures remain in place). Therefore, on a per tonne basis, the upfront capital cost per tonne of pellet loss prevented, considered over a ten year period, would be between €250 and €1,000.

Taking a mid-point of €6,250 per tonne prevented, and sharing this over the 16 year period from 2020 to 2035, it equates to a cost of €390 per year per tonne prevented.

It is estimated that the costs per tonne handled would be lower for the transport elements as this simply requires a change to containment methods, better procedures, and kit for cleaning up any spillages *en route*.[525] This is consistent with the approach taken by transport firms that have already implemented best practice measures.[526]

If pellet loss is to be prevented, such investments will have to be made by all relevant facilities. As discussed, on the basis that 50 percent of Plastics Europe members to whom Operation Clean Sweep is applicable have signed the pledge, and that by volume this accounts for the majority of

---

[524] Personal communication with Edward Kosior, Managing Director of Nextek, September 2017
[525] Personal communication with Edward Kosior, Managing Director of Nextek, September 2017
[526] Personal communication with Iain Mitchell, Managing Director, John Mitchell Haulage & Warehousing, October 2017

plastics production in Europe, and in the expectation that the target is for 100% coverage by 2017, many producers may already have put in place these best practice measures.

However, as also discussed, it seems that uptake of best practice measures among the circa 50,000 plastic converters in the EU, will be at a far lower level.

The unit costs of physically implementing the best practices to prevent pellet loss are likely to be similar, regardless of the way in which firms are encouraged or required to implement them. However, there are clear differences in the impacts on the European plastics sector depending on the specific measures selected to lead to uptake of best practice. These are discussed in the following sections.

A.6.2.3 Amending the Polymer Production BREF

- **Amending the Polymer Production BREF to include best practice pellet loss prevention measures as BAT**
    - MS to address prevention and control of releases of micro-plastics to water as part of the revision of permits of plastics production plants required by the IED to align industry practices with Best Available Techniques ; Include in future work on BREFs regarding water pollution from the plastics industry the identification of BAT for prevention and control of releases of micro-plastics to water
    - This approach could incorporate the best practice measures as described in Operation Clean Sweep - and subsequently verified and enhanced through an expert group – into the Polymer Production Best Available Technique (BAT) Reference Document (BREF) under the Industrial Emissions Directive (IED).
    - All polymer producers in Europe would thus be required to implement BAT in respect of preventing pellet loss, and would be subject to regulation and potential enforcement action as per other aspects of their Environmental Permit

Amending the Polymer Production BREF in this way would only affect EU polymer producers, many of whom may already have taken action to address pellet loss, and for whom the loss rate as a proportion of pellets handled is thought to be lower than for plastics converters.[527] On its own, amending the Polymer Production BREF would thus be expected to result in a smaller overall reduction in pellet emissions than would a regulation on converters (See Section A.6.2.5).

However, even if implemented in parallel with other horizontal measures (i.e. a regulation on the transport of pellets (see Section A.6.2.4) and a regulation on converters (see Section A.6.2.5) there is a risk of practices not being harmonious at the point of loading (and indeed unloading) of pellets. That is to say, a polymer producer could be following what it understands to be best practice, and a haulier could also be following what it understands to be best practice, but they may not be adopting procedures that are compatible in reality.[528]

While the setting of Best Available Techniques (BAT) as described in BREFs is a well-established approach, there is a question as to whether it would be appropriate as a method for addressing

[527] It is understood that 50 percent of Plastics Europe members to whom Operation Clean Sweep is applicable have signed the pledge. By volume, this accounts for the majority of plastics production in Europe and the target is for 100% coverage by 2017.

[528] Personal communication with Iain Mitchell, Managing Director, John Mitchell Haulage & Warehousing, October 2017

pellet loss, as preferred solutions were thought by stakeholders to vary considerably based on the specific facility in question. It was felt that in reality, expert judgement would be a much better way of determining what specific investments, or changes in practices, would be needed at the producers' facilities in order to achieve best practice. Accordingly, there was a concern that innovation in pellet loss prevention best practice would be limited, and facilities forced to select from a pre-determined list of practices, rather than seeking new and more cost-effective ways of delivering pellet-loss prevention. Reflecting the view that innovation would occur at a more rapid pace than the updating of BREFs it would seem likely that pellet-loss prevention measures in a BREF would rapidly become outdated.

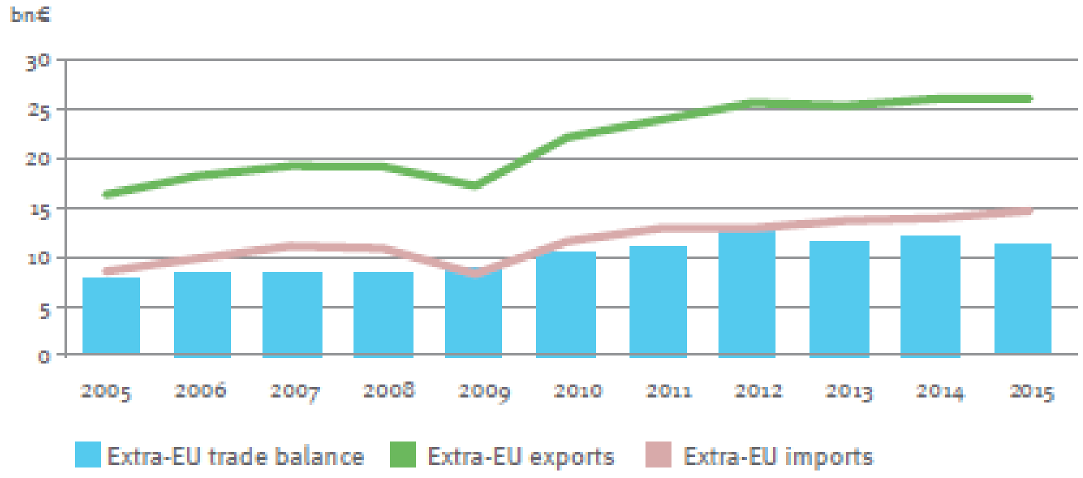
Under an amendment to the BREF, the cost of regulation and enforcement would fall to the national regulators, albeit they could then recover costs through the imposition of fees on regulated industry. As noted above there is concern among stakeholders, as reported in the workshop, that regulators may not have the industry expertise to identify what is best practice in a specific situation, and would thus choose from a pre-determined list of options. It was felt that there may be a tendency for those inspecting sites to simply go through a checklist and identify where things have, or have not, been done, and would not necessarily be able to offer guidance on how best to achieve best practice. It was also felt this meant an increased risk of being fined or given other punitive sanctions.[529]

Finally, this option, in common with other horizontal measures (i.e. a regulation on the transport of pellets (see Section A.6.2.4) and a regulation on converters (see Section A.6.2.5), would focus solely on facilities based in the EU. There would be no requirement for those importing pellets to the EU to have implemented best practice to prevent pellet loss at their own facilities. This was felt by stakeholders to be a significant disadvantage of this option, as it would place a financial burden on EU industry that would not be experienced by those operating outside of the EU. As shown in Figure 35, while there is a positive trade balance for polymer production of more than 11 billion euros in 2015, there is still a considerable amount of imports of plastic pellets, which will be destined for converters in the EU.[530]

[529] As noted in the workshop, one would ideally need at least ten years of plastics industry experience in order to able to effectively advise on what best practice investments and changes in practices would be most appropriate at a specific site

[530] Plastics Europe (2016) Plastic - The Facts 2016: An Analysis of European Plastics Production, Demand and Waste Data, available at http://www.plasticseurope.org/documents/document/20161014113313-plastics_the_facts_2016_final_version.pdf

Figure 35: Plastics Manufacturing Extra-EU Trade Balance



Source: Plastics Europe

A.6.2.4 Regulation on the Transport of Pellets

- **Regulation on the Transport of Pellets**
  - This would be a new regulation specifically covering the transport of pellets from and to facilities. All operators undertaking such transportation would be required to implement best practice approaches, again derived from expert knowledge, and further developing the approaches already pioneered by industry via Operation Clean Sweep.

Introducing this measure in a proportionate manner will be difficult. While some hauliers may specialise in transporting pellets (and powders and flakes), others may only transport them very infrequently, and in very small numbers. Accordingly, identifying those to include in the regulations (if a *de minimus* threshold is to be applied) will not be straightforward.

More significant, however, is the challenge of making sure that a regulation on the transport of pellets, and the approaches adopted by hauliers, particularly in respect of loading and unloading, are compatible with the approaches taken by the polymer producers from whom they collect, and the converters to whom they deliver. As noted by a haulier who has been closely involved in the roll-out of Operation Clean Sweep best practices, "Pellet-loss prevention only works if there's total co-operation up and down the supply chain."[531] Supply chain practices run vertically, and integration between these stages is key, specifically on reaching agreement as to the process for co-operation when things go wrong, i.e. how to clean-up spillages quickly and effectively.

Even if implemented in parallel with other horizontal measures (i.e. amending the polymer production BREF (see Section A.6.2.3) and a regulation on converters (see Section A.6.2.5) there is a risk of practices not being harmonious at the point of loading (and indeed unloading) of pellets. That is to say, a polymer producer could be following what it understands to be best practice, and a

[531] Personal communication with Iain Mitchell, Managing Director, John Mitchell Haulage & Warehousing, October 2017

haulier could also be following what it understands to be best practice, but they may not be adopting procedures that are compatible in reality.[532] Furthermore, without genuine co-operation up and down the supply chain, if both sides feel they have followed best practice, it is less likely that spillages will be adequately addressed, as instead of co-operation, there is more likely to be an attempt to apportion blame rather than focus on clean-up.

Finally, this option, in common with other horizontal measures would focus solely on facilities based in the EU. There would be no requirement for those transporting pellets outside of the EU that may end up being imported to EU converters, or indeed made into finished goods and imported into the EU, to have implemented best practice to prevent pellet loss during transportation. This would place a financial burden on EU industry that would not be experienced by those operating outside of the EU.

A.6.2.5 Regulation on Plastic Converters

- **Regulation on Plastic Converters**
    - The circa 50,000 plastics converters in the EU are mostly SMEs to whom the polymer production BREF does not apply. This new regulation would thus require all plastic converters in the EU to implement best practice measures to prevent pellet loss. Environmental regulators in each Member State would be required to ensure adherence to the Regulation

Introducing such a regulation on EU plastics converters (in isolation) would be likely to lead to a greater reduction in pellet loss than amending the polymer production BREF (in isolation), given that the majority of plastic converters are thought to have only taken minimal action, if any, to address pellet loss.

Under such a regulation, the cost of regulation and enforcement would fall to the national regulators, albeit they could then recover costs through the imposition of fees on regulated industry. As reported in the workshop, there is concern among stakeholders that regulators may not have the industry expertise to identify what might be best practice in specific circumstances, and would thus choose from a pre-determined list of options. It was felt that there may be a tendency for those inspecting plastics converters to simply go through a checklist and identify where things have, or have not, been done, and would not necessarily be able to offer guidance on how best to achieve best practice. It was also felt this meant an increased risk of being fined or given other punitive sanctions.[533]

However, even if implemented in parallel with other horizontal measures (i.e. amending the polymer production BREF (see Section A.6.2.3) and a regulation on the transport of pellets (see Section A.6.2.4) and there is a risk of practices not being harmonious at the point of loading (and indeed unloading) of pellets. That is to say, a plastics converter could be following what it

[532] Personal communication with Iain Mitchell, Managing Director, John Mitchell Haulage & Warehousing, October 2017

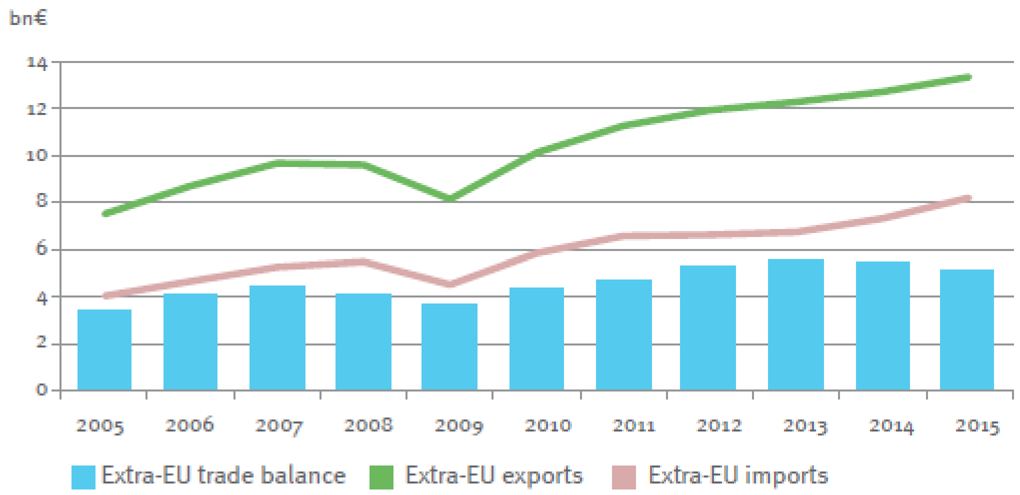
[533] As noted in the workshop, one would ideally need at least ten years of plastics industry experience in order to able to effectively advise on what best practice investments and changes in practices would be most appropriate at a specific site

understands to be best practice, and a haulier could also be following what it understands to be best practice, but they may not be adopting procedures that are compatible in reality.[534]

As previously outlined in respect of the other horizontal measures, ensuring appropriate procedures at the interface between the different stages in the supply chain is important in preventing pellet loss.

Finally, this option, in common with other horizontal measures would focus solely on facilities based in the EU. There would be no requirement for plastics converters outside of the EU whose finished goods may be imported into the EU, to have implemented best practice to prevent pellet loss. This would place a financial burden on EU industry that would not be experienced by those operating outside of the EU. This was felt by stakeholders to be a significant disadvantage of this option, as it would place a financial burden on EU industry that would not be experienced by those operating outside of the EU. As shown in Figure 36, while there is a positive trade balance for plastics processing of more than 5 billion euros in 2015, there is still a considerable amount of imports of plastic pellets, which will be destined for converters in the EU.[535]

Figure 36: Plastics Processing (Conversion) Extra-EU Trade Balance



A.6.2.6 Regulation Requiring Supply Chain Accreditation

- **Regulation Requiring Supply Chain Accreditation of Adherence to Best Practice**
  - This regulatory measure would require those placing plastics on the market (large businesses in the first instance) to ensure their entire supply chain demonstrates best practice in the prevention of pellet loss.
  - Akin to the way in which the Timber Regulations operates, adherence to best practice can be demonstrated through the use of accreditation bodies that certify adherence to best practice criteria.

[534] Personal communication with Iain Mitchell, Managing Director, John Mitchell Haulage & Warehousing, October 2017
[535] Plastics Europe (2016) Plastic - The Facts 2016: An Analysis of European Plastics Production, Demand and Waste Data, available at http://www.plasticseurope.org/documents/document/20161014113313-plastics_the_facts_2016_final_version.pdf

- This measure would include anyone directly placing plastics products on the market that were manufactured outside the EU, thus ensuring a level-playing field between the EU plastics producers and converters, and those outside of the EU wishing to sell in.

This is also a regulatory measure, but one that tackles the entire supply chain through working up from the end users of the plastic items (e.g. companies who place plastic on the market such as Danone, Coca Cola etc.), all the way up to the top of the supply chain. It would be applied in a way that will be familiar to businesses in the supply chain, who already have to commit to certain standards (relating for example to product quality control) as demanded by their customers.

A Regulation on preventing the loss of pre-production plastics would require those placing plastics on the market (i.e. the brand owners) to ensure that their entire plastics supply chain, including all logistics operations, has implemented best practice measures to prevent pellet loss. These best practice measures would build on those developed in Operation Clean Sweep guidance, with an improved emphasis on the safe transport of pre-production plastics. Measures identified as ‘best practice’ for the purposes of the Regulation would be agreed and endorsed by an expert group (comprising representatives of industry, NGOs, regulators and the European Commission – perhaps hosted by the JRC).

Significantly, such an approach would ensure the vertical integration in pellet management practices at the interface between the different stages, such as producers to transporters, and from transporters to converters. As is clear from discussion with stakeholders, for pellet loss prevention measures to be successful, vertical integration of best practices is essential.

The brand owners would be able to demonstrate their compliance with this best practice through the use of one of a number of accredited, independent, privately operated certification organisations, with independent audit, repeated annually, ensuring continued compliance.

The use of accredited privately operated organisations certifying best practice, and accredited auditors, means that a market for these services is created, with competition among such firms leading to reduced costs of verification. It also means that the role of national environmental regulators is one where they would simply check a sample of the audits undertaken on behalf of the certifying organisations, to ensure they are performing their roles adequately. The regulators do not, themselves, have to visit every facility and transport provider that handles pellets in order to ensure best practice is being implemented. This means that the burden on regulators is reduced.

The expert group could host a website where the results of all audits are held centrally, meaning that they are potentially available for view by members of the public and NGOs in order to ensure transparency and demonstrate the actions being taken. This website, and its’ operation, could be funded by fees charged as part of the certification and/or audit process.

Which End-users Would be Covered?

In theory, it would be desirable for anyone who places plastics on the market in the EU to have to demonstrate supply chain compliance with best practice. However, in the first instance it would seem sensible for the regulation to apply only to large businesses (i.e. those employing over 250 staff) that place 5,000 tonnes or more of plastic items on the market each year. This would avoid concern about placing undue burden on small and medium sized end users of plastics. Furthermore, it was estimated that firms that place more than 10,000 tonnes on the market annually account for approximately two thirds of plastics placed on the market, so such a threshold of 5,000 tonnes could be expected to cover a significant proportion of the market.[536]

However, it would still mean that any SMEs within the plastics supply chain serving the large end users would be required to adopt best practices. The difference is that this would be in response to a requirement from their end-user customer rather than a direct instruction from the regulator.

In due course, the threshold for participation should be lowered, such that within a certain number of years, all large companies placing any plastics on the EU market are required to demonstrate that their supply chains adhere to best practice.[537]

If the loss of pre-production plastics is to be addressed, it is inevitable that SMEs will have to make changes. Such changes, and their associated cost, are explained in Section A.6.2.2. However, this approach based on the familiar approach of having to adopt supply chain standards would arguably be, and be perceived as being, much more business friendly than direct enforcement of requirements by regulators. Indeed, this was a key finding from the workshop with pellets stakeholders held in Brussels in September 2017.

As can be seen in Figure 37 the key end-user sectors are:

- Packaging;
- Building & Construction;
- Automotive;
- Electrical and Electronic; and
- Agriculture.

Plastics converters will often supply many end user customers, so once these converters (and their upstream supply chain) are accredited at the request of their large customers, the items placed on the market by their small and medium sized customers will also benefit from adherence to best practice pellet loss prevention measures. This benefit extends across end-user markets. Plastic converters that supply plastic packaging, will often supply other sectors as well, such as automotive, building and construction, electrical & electronic, and agricultural end user markets.[538]

[536] Personal communication with Edward Kosior, Managing Director of Nextek, September 2017

[537] In principle it could subsequently be extended to SMEs placing more than a certain amount of plastic on the market, with the threshold gradually reducing, but the extent to which this is deemed necessary should be assessed in terms of the additional preventative effect that would be achieved.

[538] British Plastics Federation (2017) members Directory 2017, available at http://www.bpf.co.uk/members/members-directory.aspx

Figure 37: Plastic Materials Demand by Main Market Sectors

Once coverage of all large end users has been achieved, it would thus make sense to take stock of the progress made to determine what further action need be taken.

This approach also allows for ongoing learning and reflection as to which approaches might be deemed to be best practice. The more auditing that is undertaken, the more experience is collectively gained as to the relative effectiveness of different measures, and in due course new techniques might be identified as best practice. Such learning should be fed back to the expert group (comprising representatives of industry, NGOs, regulators and the European Commission) and shared on the website, to enable best practice to be updated as necessary. This could be more frequently updated than, for example, an amendment to the polymer production BREF.

Such an approach would involve private firms auditing supply chains, making recommendations, and then subsequently verifying that best practice measures had been implemented and were being adhered to. This was felt by stakeholders to be a key strength of this approach, as the ability to offer expert guidance as to what should be undertaken in specific circumstances was considered to be very important. It was noted that such auditors should have at least 10 years of plastics industry experience in order to be sufficiently familiar with the different processes, and be able to thus interpret what constitutes best practice in each case. By contrast, there was concern that regulators would not have the specific expertise to be able to identify, let alone recommend, best practice in every situation.

Costs

An indication of the likely costs can be given from a firm that already conducts such audits with a view to advising firms on how to prevent loss of pellets.[539] The costs noted below are to:

1) Undertake an initial visit to a facility;
2) Write up a report based on the visit with recommendations as to the measures that need to be taken to prevent pellet loss;
3) Undertake a follow-up visit to review the measures implemented; and
4) Write a report detailing the findings of the follow up visit, and identifying whether the recommended measures and practices have now been implemented.

The costs to undertake these visits and reports will vary based on the size of the facility, and are as follows:

- Small-sized companies (handling ≤10 kt/annum) - €1,850
- Medium-sized companies (handling >10 kt to 30 kt/annum) - €2,970
- Large-sized companies (handling >30 kt/annum) - €4,100

Nextek also charges an annual fee for follow up visits to ensure that best practice measures continue to be implemented, and to establish whether any further interventions are required. The annual fee would be approximately one third of the cost of the initial fee, meaning the fees would be as follows:

- Small-sized companies (handling ≤10 kt/annum) - €620
- Medium-sized companies (handling >10 kt to 30 kt/annum) - €990
- Large-sized companies (handling >30 kt/annum) - €1,370

Assuming that small, medium and large companies handle 5,000, 20,000 and 40,000 tonnes respectively per annum, the upfront costs equate to €0.37, €0.15 and €0.1 per tonne handled respectively. The annual costs would be €0.12, €0.05 and €0.03 per tonne handled respectively.

On the basis that the loss rate is between 0.01% and 0.04% at producers and converters, this leads to average upfront costs (relating to inspection and recommendations) to prevent the subsequent loss of a tonne of pellets not only in the first year, but in each future year of operation of the facility (assuming the measures remain in place), of between €500 and €2,000.

Again, on the basis that the current loss rate is between 0.01% and 0.04%, the average annual costs, relating to ongoing visits, per tonne of pellet loss prevented, lies between €175 and €700.

Under a regulation requiring supply chain accreditation, there would be greatly increased demand for such services, and a corresponding increase in the supply of such services. Accordingly, with greater competition in the market, it is likely that these costs will decline over time.

A.6.3 Synthetic Clothing

The options taken forwards for detailed analysis are as follows:

[539] Discussion with Edward Kosior, Managing Director of Nextek, September 2017

- **Development of a test standard to determine in a consistent manner the rate of fibre release from clothing during washing (and tumble drying)**
    - Such a test standard would likely be carried out on small samples under laboratory conditions rather than on whole garments in standard washing machines.
    - Part of the development of this standard would be to identify which factors affect release of different fibres, and the relative influence of each factor.
    - Such a test will of itself not lead to any reduction in microplastic emissions from clothing, but it will be the basis for subsequent measures detailed below:

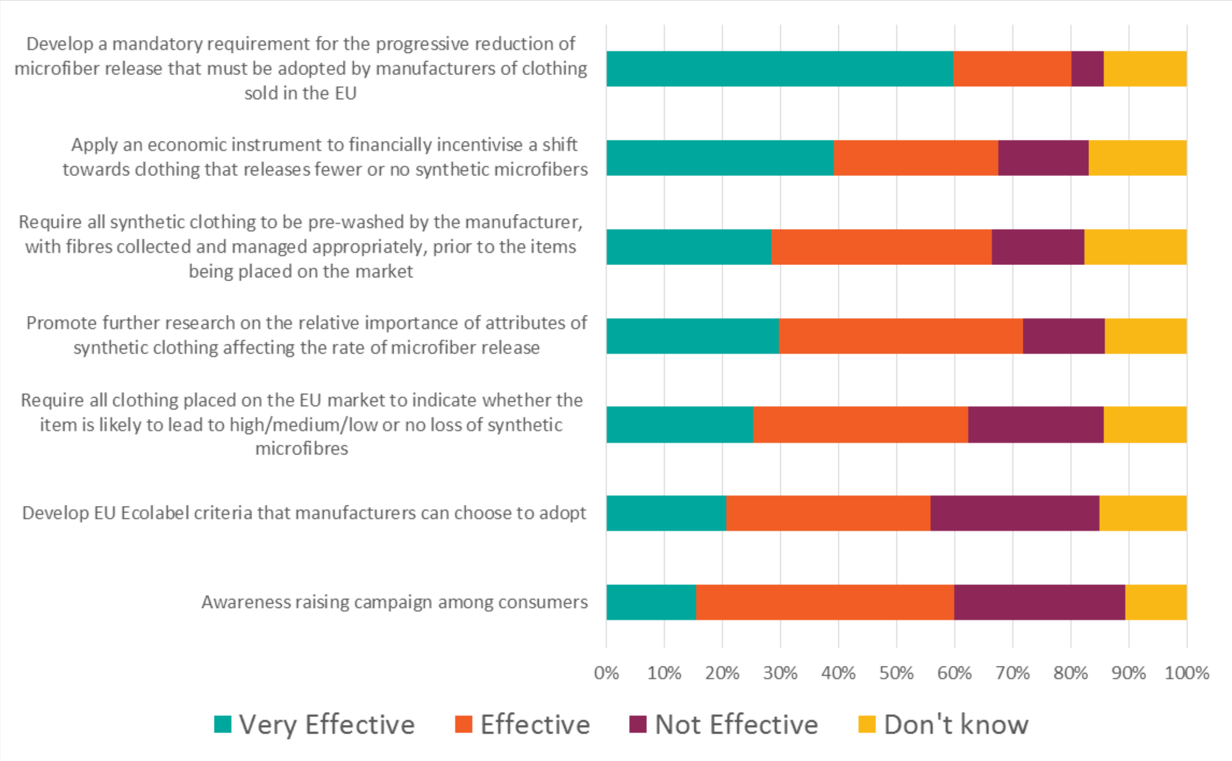
- **Setting a Maximum Threshold for Fibre Release, possibly with a new Regulation in line with the Ecodesign Directive (2009/125/EC)**
    - On development of a test standard, manufacturers of clothing would be required to submit samples of the fabrics used for testing before placing on sale in the EU.
    - The samples must be below a maximum threshold of fibre release in order for the clothing to be placed on the EU market.
    - The threshold will be developed based on the testing of a wide range of fabrics that are available on the market.
- **Development of a label for fibre release from washing of clothing to be included under the Regulation for labelling and marking of the fibre composition of textile products (EU/1007/2011).**
    - On development of a test standard, manufacturers of clothing would be required to include a label attached to the product indicating the relative level of fibre release during washing.
    - Using the standard A-G rating, this *demand-side* measure would ensure that consumers are adequately informed about the relative rate of fibre release for clothing placed on the market.

An initial list of early stage measures was presented in the open public consultation (OPC). Figure 38 presents the respondents' opinion on these measures for synthetic clothing. The most effective measure was thought to be a mandatory requirement for a progressive reduction in fibre release. This measure has been adopted for further investigation. Although the development of a product labelling scheme that provides information to consumers about fibre release was not specifically mentioned in the OPC, it largely encompasses elements from two of the suggested measures; development of Ecolabel criteria and consumer awareness raising. Both of these score the lowest on effectiveness although in both cases more than 50% of respondents believe it would be effective. As the current research shows, there is an element of consumer behaviour which may reduce fibre release during washing – using lower spin speeds or different kinds of detergents have both been suggested, but not verified as potential influences. The introduction of a label to provide information to consumers is therefore investigated further as well.

Since the opening of the consultation and through subsequent stakeholder consultation it has become apparent that the precursor to reduction measures is the development of a harmonised standard for measuring fibre release from textiles or garments. Although this will not reduce the occurrence of fibres in the environment in itself it is generally supported by textiles industry stakeholders as the first step necessary to fully understand and quantify fibre release. It is also considered as the precursor to the measures investigated in this report.

Figure 38 – Open Public Consultation Response to Question:

Please express your opinion on whether you believe that the following possible approaches to reduce microplastic (synthetic fibre) emissions from clothing and textiles to the marine environment would be effective (487 responses)



A.6.3.1 Development of a Standard Measurement for Fibre Release

As more textiles samples are subjected to varying tests for fibre release during washing it has become apparent that there is a large need for standardisation in this regard. Researchers are beginning to ascertain which factors are most import in designing a test and therefore obvious methodological improvements can be made. It is, however, problematic to compare studies and develop European level release estimates based on current findings. The many different ways in which samples can be tested and the factors which affect fibre release mean that the observed ranges are currently very large.

It is clear, however, that fibre release during washing has been identified as taking place and therefore testing needs to move on to the next step which is identifying the aggravating factors for fibre release and developing a standardised test.

A number of issues would need to be taken into account in the design of a test. As described in Section 3.3, current tests either use a whole garment or a sample of fabric in either a domestic washing machine or a simulated wash based on ISO 105-C06:2010 colour fastness tests. Both washing approaches have their benefits and limitations.

Washing a whole garment means that the construction of the garment and not just the fabric is taken into account. Issues such as how the edges are finished are thought to contribute to loss rates. This may be more representative of a ‘real-life’ situation (although most such tests undertaken to date only washed a single garment which is less representative), but it is then difficult to isolate the factors which contribute to fibre loss.

Washing using a fabric sample may be a more reproducible method of creating a standard test. Greater control is possible and samples can be directly compared. Small changes can be made (to the way in which edges are finished, for example) which will lead to isolation of the best practices that reduce fibre loss. These tests may be less useful for the calculation of the *absolute* fibre release (on an EU scale), but potentially more useful for comparison between samples in order to set a standard.

A standardised comparative test requires different methodologies to one that is designed to capture and characterise *all* fibres released. For example, for ease of undertaking the tests, a large filter mesh size (~100um) could be used if previous test work has shown that comparison can be accurately made between fabrics using this size filter—i.e. if using a smaller filter is likely to yield the same comparative results. For comparative tests, absolute fibre release count is less important.

Testing Costs

Costs for such a test are difficult to estimate at this point. However, a significant amount of work needs to be carried out in order to develop such a standardised test procedure. The Mermaids project cost over €1 million - albeit its focus was not on developing a standard – and it is expected that a similar amount would need to be spent on developing and agreeing a standard test, but with wider textiles industry support and engagement during the process. There is already an ad-hoc working group chaired by DG GROW where current progress on this issue is being shared. No formal project proposals have been shared as yet, but a tentative voluntary agreement by a cross-sectoral group of textile stakeholders is reportedly being formulated. It is understood that this agreement does not currently include any proposals or commitment to actions that would lead to a reduction in fibre release during washing. This being the case, it is therefore important to investigate some of the potential options that could be adopted; whether they be voluntary or mandatory.

The costs for an individual test are not known at present, but example cost of between €1,000 and €5,000 are used to provide a reasonable scenario for the potential costs to the textiles industry.

A.6.3.2 Setting a Maximum Threshold

After the creation of standardised test method it will be possible to compare fabrics placed on the market for their tendency to release fibres during washing. On this basis it would therefore also be possible to determine a fibre release range and create a threshold that removes the worst performing products from sale.

This threshold could either be adopted as a new Regulation (as it is important that this is harmonised across Europe, and would also apply to all items placed on the market, including imports) or as part of a voluntary agreement.

During discussions with stakeholders it appears that there are a number of barriers that may stand in the way of introducing such a threshold. One of the largest issues is the diffuse nature of the textiles industry. Indeed, the trade associations themselves are not able to say exactly how many articles of clothing are put on the EU market each year or where they come from. This was cited as a significant barrier to identifying the cost to the industry of some form of ‘gate to market’.

It was also suggested that the nature of the clothing industry makes creating a threshold difficult and costly as the industry makes regular changes to move with fashion trends. This was contextualised by one stakeholder who gave the example of personal protective equipment (PPE) clothing that requires CE certification, but is not subject to regular updates in the same way the fashion industry is. This means that certification is required infrequently. With decisions made around nine months before sale, fashion clothing would require regular, short lead-time testing if all garments are required to be certified. This is less of an issue if the clothing brand buys the fabric from within a supplier's standard range which could already be certified.

Similarly, the plastics industry (see Section 2.2.3 on pellet loss) also claims to encompass a complex and difficult to manage supply chain. However, the top-down approach that is presented in this report for the enforcement of Operation Clean Sweep has gained acceptance and will likely work in a similar way for the textiles industry. If the requirement is placed on clothing manufacturers/resellers, they can ensure their supply chain is complying.

The same problems were highlighted before the recent import restrictions on nonylphenol ethoxylates (NPE)—a chemical that was hitherto ubiquitous in clothing and posed a risk to aquatic life through washing. 'Complex supply chains[540]' were cited as potential issue for industry, but not in the context of technical feasibility, but in regard to the timescales chosen. The chemical is now on the REACH authorization list for restriction by 2020. In this case a five-year notification period was chosen to allow sufficient time for the information to filter down the supply chain and effective changes to be implemented. It should be noted that the five-year period was chosen for the NPE restriction despite the chemical being a known issue for a number of years and a ban already in place for textiles manufactured in the EU. In the case of microplastics from textiles, the industry is only beginning to become aware of the issue and its implications. The Committee for Socio-economic Analysis (SEAC) that recommended the five year period also did so with very little information available and therefore mostly relied upon testimony of the textiles industry. The feasibility of a faster or slower implementation period was not investigated and it is suspected there will be even less information available to justify a specific timescale for a microplastics release threshold.

The proposal in this instance is different to the restriction in the use of a chemical in production as it sets a standard for the product itself. In some ways this is easier to implement compared with the NPE restriction which was said to be ubiquitous in textile manufacturing outside of Europe. In that case appropriate alternatives were needed. In the present case there may effectively be a ban on whole ranges of textile types or constructions—the scope would only be finalised after the test standard is developed and the factors which affect fibre emission rates identified.

The administrative burden could be further reduced by the use of a self-certification rather than an individual test for each fabric or clothing article. In this case, certain fibre types or constructions could be restricted and the certification is merely required to confirm compliance with these restrictions. Exceptions could be made—in the case of new innovations— if the standard test is applied and it is shown that fibre release is below the threshold.

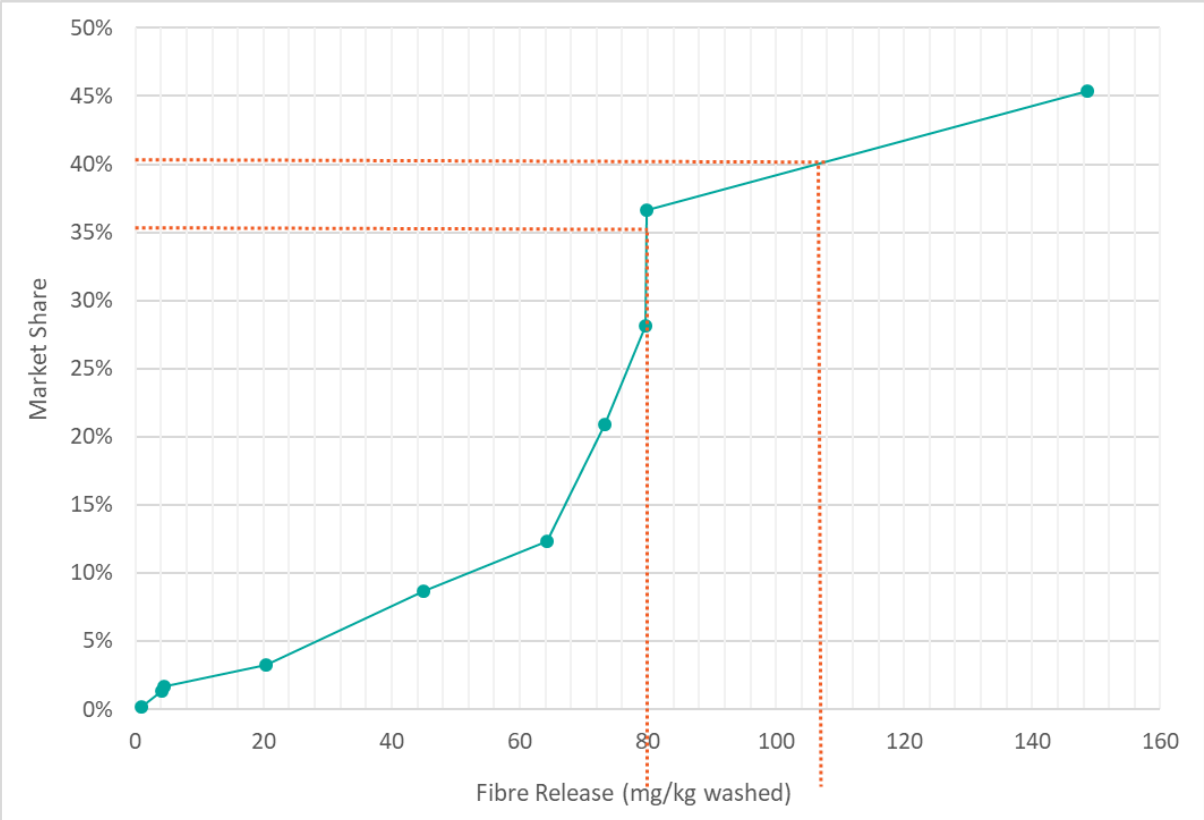
Although there is not enough information available to reliably highlight all of the factors in the fabric manufacture and construction that would influence fibre release we can propose an approach to creating the maximum threshold. Figure 39 shows a summary of the data used to create the lower estimate from the Mermaids project results (fibre numbers from laboratory simulated washing converted to mass). As these experiments looked at a variety of textile types and constructions and measured their fibre release, it is possible to show which ones perform the worst. In this case we

---

[540] https://echa.europa.eu/documents/10162/8a54914c-1407-4352-b8d6-e1321dff8858

take the worst performing 10 and 20 per cent of the man-made textiles market (4.5 and 9 per cent of the whole textiles market). This gives a fibre release threshold of 107 and 80 mg per kg washed, respectively. This would essentially remove from the market any combination of fabrics and construction that are likely to emit fibres above either of these two limits. Using the current data, a 10% limit would reduce overall fibre loss by around 30%. However, it must be stressed that this is an ***illustrative example*** only and there is not enough corroborative data to identify which types could be restricted at present—this can only be confirmed through a harmonised test standard. It does however, show that it may be possible to have a large impact by restricting the worst performing fibre types/constructions.

Figure 39 - Maximum Threshold Scenario



Certification Requirements

Two levels of certification are currently used to comply with requirements for PPE;

- Category 1 PPE only requires a declaration or conformity from the manufacturer, importer or distributor.
- Category 2 PPE requires independent testing by a notified body

Taking these as examples, clearly the latter is more costly and therefore will be more of a burden, however non-compliance with the former would not be identified unless some form of random spot testing is also applied within Member States. In the case of PPE the category 2 testing is primarily aimed at maintaining high quality standards that make the product effective. It would therefore appear to be unnecessary to impose such strict procedures if a set of parameters can be identified that can establish that particular fabrics/constructions should be restricted. A declaration of conformance to these restrictions may be the requirement in a similar way to the administration of the RoHS Directive[541] which primarily affected the use of lead-based solder in electronics—another complex global supply chain. Indeed, it appears that the same supply chain rigour required from the electronic industry may also be required of the textile industry as "…it becomes clear that guaranteeing compliance [with RoHS] will require a well organised and systematic approach involving a dialogue with suppliers, some degree of testing and good record keeping."[542]

Implementation

It is likely that a new Regulation would be required to enforce the maximum threshold. A voluntary agreement is in the very early stages of being discussed, but with no current focus on reduction methods. If voluntary reductions are agreed, it is unclear how effective they would be due to the fragmented nature of the industry with many players overseas in Asia; the very reason that regulation is not supported by the industry also makes a voluntary agreement less effective. It may also create an un-due burden on those that adopt it which creates an un-level playing field and a competitive advantage for those that do not.

The Ecodesign Directive (2009/125/EC) may be the framework in which this and other microplastics reducing legislation could introduced. At present, the Directive is aimed at products that impact energy use therefore a broadening in its scope would be necessary. Following on from this, a supporting Regulation would be required to set the maximum threshold and any other requirements.

Enforcement and Monitoring

Enforcement would likely be at the Member State level with the authorities responsible for enforcement of the restriction either;

- performing random sampling of textile articles and applying the standard test methods to assess the release rate; or
- identifying the fabric and construction to ascertain whether it is on the list of restricted products.

Costs of compiling such information could be limited by conducting them concurrently with the monitoring of existing restrictions under current Market Surveillance[543] activities, such as those on azocolourants, pentachlorophenol (PCP) and NPE in textiles.

This sampling would be conducted alongside the monitoring of microplastics in WWT effluent and in habitats. The former would expect to see a noticeable decline in fibre emissions within a few years of implementation as new clothing replaces old clothing. Concentrations in the environment would be more difficult to assess over time and would be slower to react to changes in source concentrations.

Costs to Industry

The costs to industry for the introduction of the threshold are difficult to assess due to the number of unknowns about the industry, but a possible cost range can be identified to provide an idea of the

---

[541] http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2011:174:0088:0110:EN:PDF

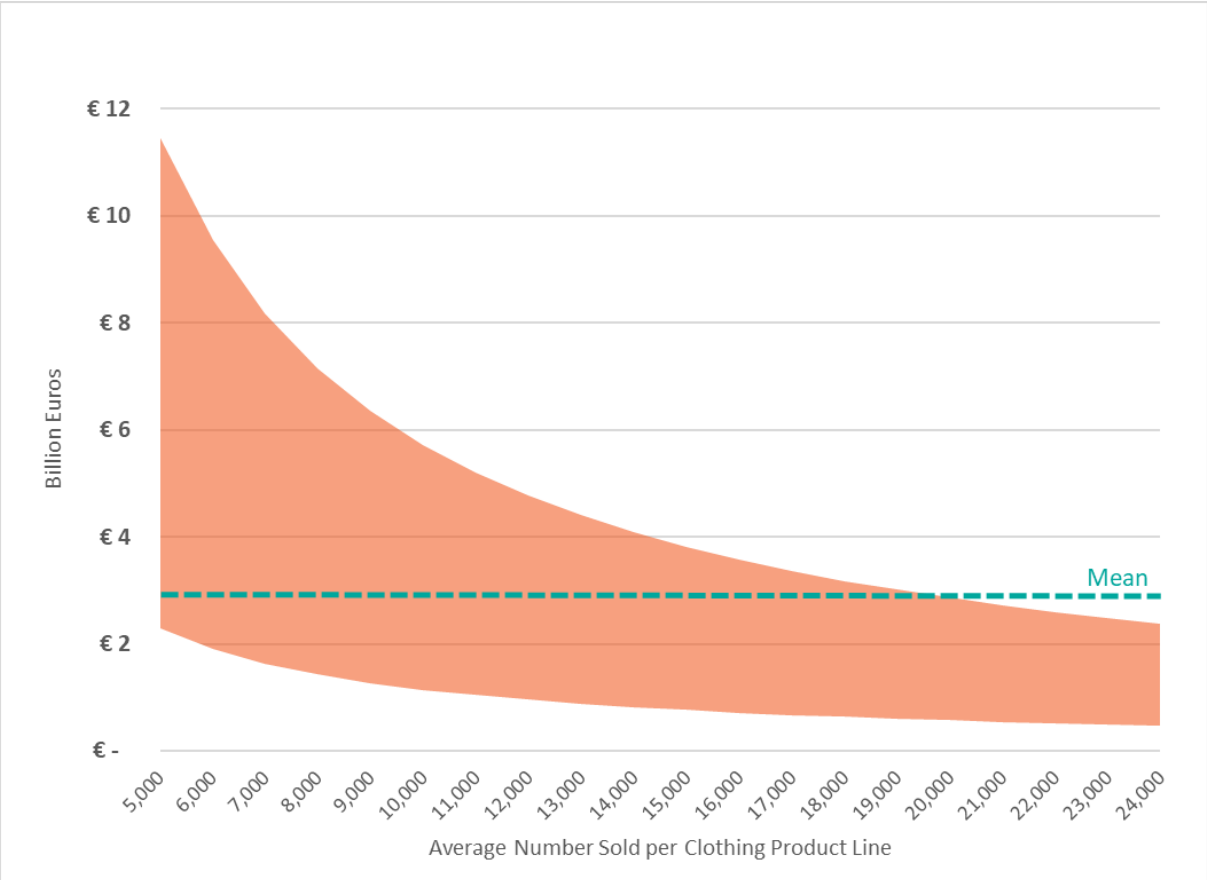
[542] Martin Goosey (2007) Implementation of the RoHS directive and compliance implications for the PCB sector, *Circuit World*, Vol.33, No.1, pp.47–50

[543] https://ec.europa.eu/growth/single-market/goods/building-blocks/market-surveillance_en

potential costs in the worst case i.e. if each garment on the market requires testing and certification. EU sales data by garment is provided by the JRC in their IMPRO[544] report. This data was used to extrapolate the total number of synthetic garments placed on the market which is estimated to be around 11.5 billion. Each one of these garments is not unique, but there is no data available that permits an estimate of the number of product lines that exists.

To provide an example, Figure 40 shows what the potential industry costs for testing would be if each clothing line has sales of between 5 and 24 thousand items (2.3 million to 500,000 unique products). The orange shaded area shows the range of costs that may be incurred if the test costs between €1,000 and €5,000 for each unique product on the market. In this illustrative example, the minimum cost is around €0.5 billion (around 500,000 unique products costing €1,000 each to test), rising up to a maximum of over €11 billion (around 2.3 million unique products costing €5,000 each to test). The mean overall cost is around €3 billion. This assumes that each unique clothing item is required to be tested before being placed on the market and is therefore an extreme example of the costs involved. It also assumes that each product line is entirely new each year if this is an annual cost. This may be true for some products, but equally there will also be some that remain for multiple years within a retailer's core range. Despite this, it becomes clear that the costs to the textiles/clothing industry would be significant; with 3.2 million tonnes of man-made clothing placed on the EU market every year this would equate to an ***additional cost of €0.90 per kg.***

Figure 40 – Indicative Textiles Industry Costs for Garment Fibre Loss Testing



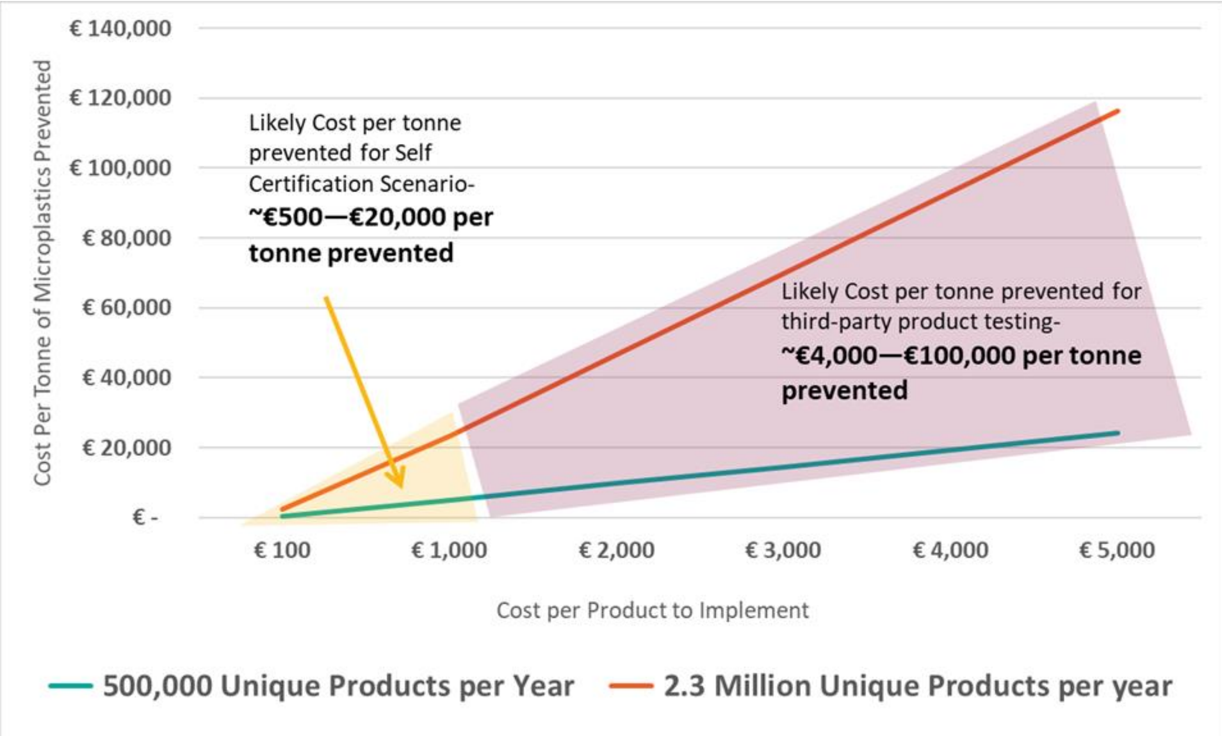
[544] JRC (2014) *Environmental Improvement Potential of Textiles (IMPRO-Textiles)*, Report for European Commission, January 2014

The alternative proposition of requiring the textiles/clothing manufacturers to supply a declaration of conformity (as per the PPE Category 1 example) may therefore be preferable.

The costs to develop a test standard and to test a wide variety of fabrics/constructions that cover the full range of garments sold in the EU are likely to be a great deal lower. This would require a large market research project in collaboration with the textiles/clothing industry to make sure that all product variations are covered. As a minimum it would be expected that the five most prominent man-made fibres would be tested (Viscose, Polyamide, Acrylic, Polypropylene and Polyester) in woven, knitted (including fleece) and non-woven constructions. There are also many subsets and variations of these that should be identified and tested as well as different finishes that can be applied, different fibre cross-sectional shapes, thicknesses (Dtex) and yarn constructions—the permutations are expected to be in the hundreds of variants unless it can be verified early on that certain characteristics have no bearing on fibre release. This is still expected to be considerably less than the number of individual tests that would be required in Figure 40 (0.5—2.2 million tests annually) and it would not be necessary to repeat these tests on an annual basis. The self-certification process would also incur costs for suppliers/manufacturers, but these are expected to be considerably less as this would mostly be comprised of internal admin costs which would also reduce as the companies became more familiar with the process. If these costs were ten-fold lower than direct testing (in the range of €100—500 per product line) the annual cost could be around €300 million.

Figure 15 shows the cost effectiveness per tonne of microplastics prevented could differ between the two certification and testing regimes. The self-certification approach is clearly preferable in this instance. Whether this will function this way in practice will be determined by the development of the test method and whether it lends itself to the self-certification process.

Figure 41 – Costs per tonne of Microplastics Prevented by Certification Type



A.6.3.3 Development of Product Labelling

Textiles industry stakeholders are not in favour of this measure as they are convinced that the consumer rarely looks at existing labels (those that are mandatory as part of EU Regulation 1007/2011) and in many cases will cut them off, removing useful washing information in the process. It is clear that introducing more information onto the current sewn-in labels may not be the most effective way to increase consumer awareness of the issue, and thus influence their buying decisions, however there is the potential to include a label that is prominent at the point of sale, but is removable later. Research suggests that washing habits can influence fibre release and although the absolute effect is not known, there is enough evidence to begin to make recommendations to consumers. The clothing company Patagonia is beginning to do this, but it's unclear whether this includes a label on the garment itself at this stage.[545]

In conclusion there are therefore two ways of including a label which can be used to achieve different outcomes;

- **A Sewn in label**—containing washing and user guidance which can be referred to on an ongoing basis; and
- **A Removable Label**—containing information that is designed to provide environmental information and influence buying decisions.

There may be scope to include one or either of these labels. As information on the sewn-in label is already mandatory it would be straightforward to amend the current Regulation with extra requirements. The effectiveness of this relative to the increased burden on manufacturers (i.e. the potential for the label to increase in size) is not known.

The inclusion of a specific additional label may be designed in line with the well-recognised and understood labelling systems for energy using products (Energy Labelling Directive 2010/30/EU[546]) and tyres. This may be in the form of a removable card tag attached prominently on the outside of the garment similar to the tags that are often provided to provide information for more technical garments of branding on mid-high range clothing. This would incorporate an A-G rating based upon the expected level of fibre release. It may be argued, however, that a label specifically for fibre release is disproportionate when compared with other environmental impacts associated with clothing and textile manufacture. In the case of energy labelling, the energy use of the product is highlighted to cause the most environmental damage over the life of the product. There is no evidence to suggest this is true for fibre release from textile products and there is currently no way of comparing this with impacts such as greenhouse gas emissions or water use during manufacture. Equally, both the energy label and tyre label incorporate information that affects the product's performance in-use which are important factors in the buying decision. It is currently unclear whether there is any correlation between fibre release and the quality of the garment. If this is found to be the case, a label could be justified due show the increase in longevity that could be expected from better performing (durable) clothing.

Similar to introducing the maximum threshold, there is also the issue of whether the garment is individually tested or is placed on the A—G scale based upon the fabric type and construction method. In either case the burden of introducing such a label is significantly decreased if the

[545] https://www.patagonia.com/blog/2017/02/an-update-on-microfiber-pollution/
[546] http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32010L0030

information is already available through the maximum threshold introduction. For this reason, it is recommended that the threshold be introduced first.

A.6.3.4 Extended Producer Responsibility

The final option that is investigated is the application of EPR so that the textiles industry are obliged to pay for the implementation of recovery measures. This can be achieved in the two key points in which fibres from clothing are known to pass through; the washing machine and the wastewater treatment plant.

Funding of improved WWT is discussed in more detail in A.6.4 which addresses measures specific to the wastewater industry.

In this section, the effectiveness of capture at the washing machine is discussed. This may be a more appropriate place to capture fibres as;

- There will be losses between the washing machine and the WWT plant;
- Expenditure on WWT requires a large capital investment over a time period of decades, whereas washing machines are replaced more regularly.

There are a number of potential systems that have been proposed or developed recently that are designed to work with a washing machine. They broadly come under two categories;

- An in-built washing machine effluent filter; and
- A device placed in the washing machine drum which is independent of the washing machine itself.

There is a third option which the inclusion of a standalone effluent filtration system for which variations have been on the market[547] for some time for those who are not connected to waste water treatment. This is not considered as a viable solution to the problem at an EU level due to the relatively high cost that would prohibit most households from installing one.

It is difficult to determine the effectiveness of devices that are designed to be placed into a washing machine drum by the user. To assess this in terms of the best case for reduction potential, it is assumed that one would be supplied with every purchase of a new washing machine. Similarly, this is compared against the introduction of a compulsory filter which is built into the effluent outlet of all new washing machine.

There are around 21 million washing machines sold in the EU annually[548]. If 90% of households own one, this equals 203 million washing machines currently in use. It also means that within 10 years they will all be replaced (assuming one per household).

The designers of the 'PlanetCare Filter' suggest their prototype product will capture around 80% of fibres. Over the course of its 10 year life, each filter is therefore expected to capture between 0.7

---

547 http://www.septicsafe.com/washing-machine-filter

548 Boyano Larriba et al (2017) *Ecodesign and Energy Label for Household Washing machines and washer dryers*, Report for JRC Scientific and Technical Reports, 2017

and 1.7 kg of fibres[549]. At a cost of €3 per filter (and regular replacements at €1.50 each), this puts the **cost per tonne captured at between €49—125 thousand.** Calculations for this and other devices are shown in Table 107 which demonstrates similar costs per tonne removed (using assumptions around the likelihood the device will be used). The Cora Ball[550] is designed to be placed into the drum with the clothing and is free to move around. The Guppy Friend[551] requires the user to place synthetic clothing with a bag which is designed to prevent the loose fibres from exiting. All three products are new to the market and their potential impact on fibre release is yet to be fully determined. In the case of the Planet Care Filter this is also not commercially available yet and therefore the costs are speculative at this stage. The concept requires regular replacement of the filter by returning it to the manufacturer—this prevents accidental release of the fibre via washing of the filter but incorporates an additional cost and time burden for the consumer. To implement an EPR scheme, the cost of this should be borne by the textiles industry.

Table 107 – Cost Benefit of Fibre Capture in Washing Machines

	PlanetCare Filter	Cora Ball	Guppy Friend
(a) Product Cost	€80.45[1]	€18	€30
Product Life (years)[2]	10	10	10
(b) Fibre Capture Rate	80%	26%	80%
(c) Use Factor[3]	100%	80%	40%
(d) Fibre Release per hhld per Washing Machine (kg)	0.84—2.1		
Cost per tonne Captured a / (d x b x c)	€49,200—€124,600	€40,820—€103,349	€44,222—€111,962

Notes:
1. **Filter cost is €3 and is replaced every 30 washes at a cost of €1.50. At 166 washes per year over 10 years this equals 54 replacements.**
2. **All products assumed to last the life of the washing machine**
3. **Estimated as filter is permanently installed, the Cora ball requires placement in the drum only, whereas the Guppy Friend requires separation of synthetics to be placed in the bag.**

[549] Releases per wash are between 0.28g and 1.32g (a). The average number of washes per household is 166 (b). The lifespan of the machine is 9.7 years (c). Captured fibres = a x b x c.
[550] http://coraball.com/
[551] http://guppyfriend.com/en/

A.6.4 Wastewater Treatment

The options taken forwards for detailed analysis are as follows:

- **Development of a test standard for the quantification (both in mass and number) of the microplastics in the influent, effluent and sludge output of wastewater treatment plants.**
  - Such a test would:
    - take account of possible contamination from microfibres in the atmosphere;
    - use filters as small as practicably possible to make sure the smallest microplastics are measured;
    - sample a high enough proportion of the influent/effluent so as to be confident that the sampling is representative; and
    - be conducted over a time period that would allow accurate estimation of annual microplastic loads.
  - Where possible the test will also be used to identify where the microplastics have come from. This may be impossible for some types, but microplastics from paint, tyres and clothing fibres are likely to be unique enough to be identifiable.
  - Such a test will of itself not lead to any additional capture of microplastics through WWT, but it will be the basis for subsequent measures detailed below or in the context of a future review of the UWWT Directive
- **Development of an EPR Scheme such that the sources responsible for microplastics in WWT cover the respective costs of remedial action**
  - This may be administered and be applied differently from country to country and will rely on sampling at a local level. This will lead to different industries being required to contribute differently within each member state.
  - Similarly, the fee would go towards (and the level of the fee would be determined by) the most appropriate means of mitigation, whether that be, for example:
    - Adding additional (existing or novel) treatments to WWTPs
    - Improving roadside capture; or
    - Increasing road cleaning activity

The OPC found that in most cases the respondents thought that the most effective measure for downstream capture of microplastics is to install technologies that are proven to capture microplastics in WWT to prevent them from entering effluents. The exceptions to this were for fibres from clothing, artificial turf and pellets where capture at the point of emission was seen as equally effective. For tyre wear, sustainable drainage (SuDS) was also seen as an effective method to capture the particles, albeit less so than capture at WWT.

The following sections first discuss the cost implications of increasing capture of microplastics at a WWT plant and in storm water run-off. We then consider how these costs would be apportioned if, under an Extended Producer Responsibility (EPR) approach, the contributing sources of microplastics cover these costs in proportion to their respective contributions.

A.6.4.1 Development of a test standard

The development of a standard test method that can be used to characterise the microplastics entering and leaving a WWT plant is critical to the application of any further measures for microplastic reduction in WWT. Although several studies from across Europe as well as the US and Australia have attempted the measure the occurrence of microplastics in WWT there are often key methodological differences that not only make comparison of result difficult, but also highly speculative when used to scale up emissions beyond the plant in question.

The differences in wastewater effluent data occurs mainly due the differences in sampling volumes, mesh-sizes of the filters and material characterization (visual identification vs. Fourier-transform infrared spectroscopy (FTIR)). These are the most crucial methodological steps that affect the results.

The latest studies, however, (Murphy et al. 2016, Ziajahromi et al. 2017, Mintenig et al. 2017, Talvitie et al. 2017b) are somewhat comparable due the similar methods and the results do not differ hugely. (A summary of this can be found in Table 108). The methods described in Mintenig et al. 2017 are potentially the best methods currently devised for wastewater effluent microplastics measurement due to the large sample volumes (>/=1 m^3), a small mesh-size for filtration (<10 µm) and automated material analyses.

Data from the wastewater influent and sludge is very preliminary and reliable, comparable data is likely to take considerably longer to obtain. The extraction of microplastics from influent and sludge is very difficult (due to filter clogging), leading to very small sample sizes. Having said this, the few studies that have attempted this show that majority (98 - 99%) of the microplastics entering the WWT plant within the influent is retained in sludge (Murphy et al. 2016, Talvitie et al. 2017a). Only one article focusing on sludge treatments and their impact on microplastic abundance has been published (Mahon et al. 2016). It is however very difficult to draw any conclusion from these studies at this time. At the moment we can say that MPs in a sludge should be taken into consideration when designing and implementing sludge treatments and when making decisions about sludge use.

Textile fibres are relatively easy to identify from wastewater samples due to their shape and colours. The material analyses of the fibres are performed with FT-IR microscopy. FT-IR analyses and identify synthetic fibres relatively well, but natural textile fibre materials are more challenging (this includes 'man-made' fibres such as viscose). The provenience of other microplastics is more difficult as their characteristics (size, shape, material type, colour, etc.) may be identified, but it can often be far more challenging to positively trace these back to an emission source.

The identification of specific materials may be more or less important depending upon what sort of measure the results are supporting. There are two main ways that a test standard can be used the support legislative measures;

- To identify whether upstream measures are effective at reducing microplastics; and
- To identify which sources of microplastics are contributing the most to microplastics loads through WWT plants.

The former may not require 100% positive identification as there is no 'penalty' attached to it. The test would be undertaken merely to identify whether policy measures have the desired effect. The latter would require a high degree of certainty around the identification of the source of the microplastics as this would directly link to any EPR payments (the EPR scheme itself is discussed further in the following sections). It is unclear at present, whether this is technically possible or cost effective. The costs of either approaches are also not known at present.

The implication would be that each WWT plant would need to sample their influent, effluent and sludge on a regular basis (unless it can be ascertained that a few plants are representative of the rest of the system). The scope could follow the UWWT Directive by limiting to WWT plants servicing towns and cities over 2,000 inhabitants. According to the European Environment Agency, there were around 19,000 such plants in 2010[552]. If each one of these plants spend €10,000 per year on testing, the annual cost of this for the EU would be €190 million. This is highly speculative without further information on the costs of test regimes which can only be accurately assessed once the testing is scoped and developed. The costs of such testing would have to be incorporated into any of the subsequent reduction measures.

It may even be possible to narrow down the scope further to the 463 'big cities' (> 150,000 inhabitants) covered by the UWWT Directive which represents 46% of the pollution load[553] (WWT pollution load not necessarily being equal to the microplastics load). It is unclear how many plants this would encompass, however.

Table 108 – Microplastic Concentrations Observed in WWT Plants

(Highlighted rows designate similar methodologies)

MP Concentration (L^{-1})		Size (µm)	Sample volume (L)	Material analyses	Effluent type	Study
1		> 1.6	0.75	Visual + FTIR	Tertiary	(Browne et al. 2011)
0.008		> 300	1000	Visual + FTIR	Secondary	(Magnusson and Norén 2014)
35		> 100	0.05	Visual	Secondary	(Dris et al. 2015)

552 https://www.eea.europa.eu/themes/water/water-pollution/uwwtd/waste-water-infrastructure/urban-waste-water-treatment-plants
553 European Commission (2016) *Eighth Report on the Implementation Status and the Programmes for Implementation (as required by Article 17) of Council Directive 91/271/EEC concerning urban waste water treatment*, April 2016

5.9		> 20	10 – 20	Visual	Secondary	(Michielssen et al. 2016)
2.6		> 20	34 – 38	Visual	Tertiary	(Michielssen et al. 2016)
0.0009		> 180	5680	Visual + FTIR	Secondary	(Carr et al. 2016)
0		> 45	189000	Visual + FTIR	Tertiary	(Carr et al. 2016)
0.05		> 125	500 - 41000	Visual	Tertiary	(Mason et al. 2016)
0.25		> 65	50 L	Visual + FTIR	Secondary	(Murphy et al. 2016)
0 – 9		> 20	390 – 1000	FTIR	Tertiary	(Mintenig et al. 2017)
1.54		> 25	16.5 – 100	Visual + FTIR	Primary	(Ziajahromi et al. 2017)
0.48		> 25	150	Visual + FT-IR	Secondary	(Ziajahromi et al. 2017)
0.28		> 25	200	Visual + FT-IR	Tertiary	(Ziajahromi et al. 2017)
0.2 – 2.0		> 20	17 - 1000	Visual + FT-IR	Secondary	Talvitie et al. 2017b
0.02 – 0.3		>20	6 - 1000	Visual + FT-IR	Tertiary	Talvitie et al. 2017b

Source: Adapted from Talvitie, unpublished

A.6.4.2 Improved Wastewater Treatment EPR

Eureau supplied information[554] on the potential cost implications of incorporating one of the following tertiary treatment processes that are expected to increase microplastics removal from effluent (but sequestered in sludge) to close to 100%:

- Disk filter
- Sand filtration
- Membrane micro-filtration
- Membrane bio-reactor

[554] http://www.eumicroplastics.com/eumpwp/wp-content/uploads/eureau.pdf

The costs were estimated by Eureau to be between €0.08-0.20 per cubic metre of wastewater treated per year. This is an annualised cost which includes infrastructure and running costs. Eureau also estimated that between 23 and 38 billion cubic meters of wastewater are treated annually in the EU (although the upper figure is the total treatment capacity). Their calculated costs for the upgrade multiply the cost of the treatment by the total wastewater. However, there are currently existing plants that have tertiary treatment[555] and therefore the costs should be applied to improving those that do not. There are no comprehensive official datasets for the volume of wastewater treated by country, but the overall estimates from Eureau can be split by population equivalents (PE) for each country. The Eurostat data used to model the microplastics retention rates in Section 2.2.8.1 is used to ascertain the capacity gap for those that do not currently have tertiary treatment. This suggests that there are between 10 and 16 million cubic meters of wastewater that could benefit from an upgrade to increase retention. This would cost €0.76—3.14 billion per year assuming the unit costs from Eureau. However, the baseline calculations in Section 4.0 show that the compliance with the Urban Waste Water Treatment Directive will increase tertiary treatment by around 12% by 2035. This means that additional spending is lessened and therefore costs are reduced to €0.6—2.4 billion. With an average cost of €1.49 billion per year the basis of an EPR scheme can be developed.

Table 109 shows how the costs to improve WWT can be applied to the different product groups based on their proportional contribution. In practice this would rely upon accurate identification and characterisation of the particles entering into WWT (or in the effluent if sewage sludge is disposed of through incineration). Presently this example uses the estimated tonnages that are modelled to enter WWT. Costs range from €10 million for building paint to just over €1 billion for textiles.

On a cost per tonne of microplastics captured basis the costs range between €45,000 per tonne for the upper level emissions with a 95% capture rate (current baseline of 60,600 tonnes being released from WWT into surface waters) and €137,000 per tonne for the lower level emission with a 100% capture rate (current baseline of 4,239 tonnes being released from WWT into surface waters).

Table 109 – Example Calculations for Assigning Cost for WWT Improvements to Polluting Industries

Emission Source	Microplastics Entering WWT (tonnes)		Overall Proportion in WWT		WWT Improvement Costs Assigned (Billion €)	
	Upper	Lower	Upper	Lower	Upper	Lower
Washing of Clothing/Textiles[1]	44,329	17,509	34%	65%	€ 0.51	€ 0.97
Pellets	41,801	847	32%	3%	€ 0.48	€ 0.05
Automotive Tyres	33,390	6,678	26%	25%	€ 0.38	€ 0.37

[555] It is recognised that the term 'tertiary' is very broad and is not well defined, however it is assumed that tertiary treatment in whichever form it takes will increase microplastics capture rates.

Artificial Turf	4,351	914	3%	3%	€ 0.05	€ 0.05
Road Markings	3,314	663	3%	2%	€ 0.04	€ 0.04
Building Paint	2,213	268	2%	1%	€ 0.03	€ 0.01
Totals	**129,398**	**26,879**	**100%**	**100%**	**€ 1.49**	**€ 1.49**

Notes:

1. **Washing of clothes/ textiles category also includes cleaning cloths as the fibres are likely to be indistinguishable in a WWT plant.**

One potential method of administering an EPR scheme would be via a fee applied to each product based on the level of contribution to microplastics entering WWT. It would be expected that the fee would vary between Member States as different industries are likely to contribute differently between countries. It is therefore important to have representative microplastic data from WWT plants in each of the countries.

Table 110 shows how this could work for each of the source emissions. For most of the products a fee per kg placed on the market appears to be the most appropriate. This is the case for textiles, pellets and paints. The weight of a tyre itself has no direct link to wear rate, albeit the overall weight of the vehicle does, and therefore it is more appropriate (and administratively simpler) to apply the fee per unit (tyre) sold. Tyres have been split into two groups as the average wear rate per km per tyre is considerably higher for truck tyres—by volume truck tyres are around 5% of the market, but they contribute to 26% of tyre wear emissions (see Appendix A.6.1). They also cost more, therefore a higher fee can be applied while resulting in a similar proportional increase in the product price as for a car tyre.

Table 110 shows that in most cases the fee is less than 1% of the product cost and often as low as 0.1%. The cost per tyre is increased by as much as 7% for the lowest value tyres on the market. It is not clear what proportion of the market is comprised of the lower end, but it is likely the average price of a tyre would be considerably higher. No data is available on the average cost of tyres in the EU.

Artificial turf is different to the others as the infill remains in service for around 10 years and does not inherently become an environmental issue as part of its product life, i.e. unlike tyres which are designed to wear during their life. The application of a fee on the cost of the infill would add around €4,000 to the installation cost of a typical football pitch. This money is likely to be better spent introducing on-site mitigation and containment measures which would reduce the burden on downstream captures mechanisms such as WWT.

Table 110 – Example Fees per Product

Emission Source	**Annual Sales**	**Unit**	**Fee per Unit**		**Retail Cost of Product per unit[6]**	**Product Cost Increase**
			Upper	Lower		

Washing of Clothing	7.5 Billion[1]	kg	€ 0.07	€ 0.13	€20—€200+	0.03—0.65%
Pellets	70 billion[2]	kg	€ 0.01	€ 0.001	€1—€2	0.34—0.1%
Automotive Tyres	Cars - 200[3]	million units	€ 1.38	€ 1.33	€30—€200+	0.7—4.5%
	Trucks - 10[3]		€ 9.88	€ 9.51	€300—€500	2—3%
Artificial Turf	110 million[4]	sqm	€ 0.45	€ 0.46	€5—15	3—9%
Road Markings	258 million[5]	kg	€ 1.27	€ 1.22	€20—€50	2.5—6%
Building Paint	11 Billion[5]	kg	€ 0.02	€ 0.01	€10—€30	0.07—0.13%

Notes:

1. **See calculations in Section A.3.1**
2. **See calculations in Section A.3.6**
3. **ETRMA - car tyres include passenger car, SUVs and light commercial vehicles**
4. **See calculations in Appendix A.3.4**
5. **Data from CEPE.**
6. **Costs are estimates to provide context. In many cases (especially for textile clothing) the cost of the finished product at retail does not have a direct relationship to its weight.**

Conclusion

The cost increases shown in Table 110 demonstrate that the financing of improved WWT is possible in principal as they are reasonably small proportions of the products. It is expected that in the majority of cases this would be a last resort if the associated industries fail to improve their products and reduce emissions accordingly. For tyres and paints this may be the only way to pay for and appropriately capture microplastics emissions before they enter waterways as preventative measures will not solve the entire problem.

A.6.4.3 Improved Storm Water Treatment EPR

Storm water has been identified as a significant pathway for microplastics, in particular tyre wear particles from road run-off. It is also expected that a great deal of those particles will sediment in one or more of the various road-side treatment devices that may be installed.

As well as the more technical solutions that can be added to WWT plants, there are also more natural alternatives that may perform equally well and be used as an end-of-pipe treatment solution for both WWT effluent and storm water run-off. The Coalition Clean Baltic identified that wetlands can offer removal efficiencies of close to 100%[556].

Although wetlands are primarily constructed to remove nutrients and pharmaceuticals before entering receiving water (for which their effectiveness is well documented[557]), a recent Swedish thesis[558] looked at the effectiveness of these for microplastic retention. A large number of microplastic particles—particularly those thought to be tyre particles—were sampled from storm water inlets into wetlands. These were reduced by 100% for microplastics >300 µm and over 90% for particles 20—300 µm.

The Coalition Clean Baltic also cites another Swedish thesis[559] that assessed the effectiveness and costs of various wetlands in Sweden; albeit not from the perspective of microplastics capture. The investment and operating costs are presented in the thesis in 2008 Swedish Krone values. These costs have been annualised over 20 years, converted to Euros and put in 2017 values (see Table 111). Based on this, the average cost per cubic metre treated is calculated to be €0.06 for the six sites in Sweden. This varies depending on size (larger is more cost effective) and whether pumping is required (more expensive). These costs are at the lower end of the costs identified for improved tertiary treatment, however stormwater run-off will be much greater than WWT effluent, therefore not all of it could or should be treated. The costs also does not take into account some of the wider potential benefits of wetlands which can include revenue generated from increased tourism and additional habitats for wildlife.

Table 111 – Costs of Wetlands

	Flow rate m3/day	Investment Cost (MSEK)	Operating Cost (MSEK/Year)	Total Costs Annualised over 20 years	Treated per year (m^3)	Cost per m^3 treated (SEK)	Cost per m3 treated (Euro 2017 Prices)
Ekeby	44,963	23	0.20	1.35	16,411,495	0.08	€ 0.01
Alhagen	5,218	20	0.40	1.40	1,904,570	0.74	€ 0.09
Brannäs	4,396	8	0.10	0.50	1,604,540	0.31	€ 0.04
Magle	12,369	11	0.25	0.80	4,514,685	0.18	€ 0.02

[556] Coalition Clean Baltic (2016) *Concrete ways of reducing microplastics in stormwater and sewage*, Report for HELCOM, April 2016

[557] J. B. Ellis, R.B.E.Shutes and M.D.Revitt (2003) *Constructed Wetlands and Links with Sustainable Drainage Systems*, Report for UK Environment Agency, 2003

[558] Jönsson, R.(2016) *Mikroplast i dagvatten och spillvatten : Avskiljning i dagvattendammar och anlagda våtmarker*,

[559] Linda Flyckt (2016) *Treatment results, operational experiences and cost efficiency in constructed wetlands for waste water treatment in Sweden. (in Swedish)*, dissertation submitted at Linköping University Department of Physics, Chemistry and Biology, 2016

Thong	1,703	12	0.21	0.81	621,595	1.30	€ 0.15
Vagnharad	5,218	7	0.14	0.49	1,904,570	0.26	€ 0.03
Source of costs in MSEK: Linda Flyckt (2016) Treatment results, operational experiences and cost efficiency in constructed wetlands for waste water treatment in Sweden. (in Swedish), dissertation submitted at Linköping University Department of Physics, Chemistry and Biology, 2016							

An example of this is Willen lake, a storm water balancing lake in the UK which has a total surface area of 68 ha[560]. The storm water from the whole of Milton Keynes (population 250,000) is directed here and was made possible due to separate sewage systems being installed from the town's inception in the 1970s. It consists of two interconnected basins; the first receives the storm water and is also used for recreational activities, the second is zoned for nature conservation and is kept at a different level and allows water to pass into it during rain events. This setup reduces pollutants transport to the nature reserve and ultimately to the nearby river for final discharge. The large lake volume also has a buffering effect on any pollutant loads during rain events.

For both this example and other wetlands, microplastic pollution is not monitored (as it is currently not considered as a pollutant). It is therefore unclear what effect these will have on a wetland habitat as they are not currently designed with microplastic retention in mind. Creating habitats as a purposeful sink for microplastics may have adverse effects on the wildlife that will inevitably be drawn to the area, however it will prevent the microplastics from dispersing further into rivers and oceans and lead to habitat creation relative to the counterfactual. Single point sinks would be easier to monitor for negative impacts and develop mitigation strategies for, for example, the regular removal of settle debris.

Table 112 shows the estimated impact of improving storm water run-off capture. Between 9—20% of total microplastic emissions are estimated to end up in waterways through storm water. For all of these sources this is the dominant pathway to waterways except for artificial turf. For Automotive tyres the 84% of the emissions to waterways (23% of total tyre emissions) are via storm water run-off.

Table 112 – Estimated Microplastics Emissions from Surface Run-off to Waterways (tonnes)

Source	Emissions		Proportion of Source Emissions	
	Upper	Lower	Upper	Lower
Automotive Tyres	114,001	44,663	23%	9%
Pellets	50,162	1,694	30%	10%

560 Antony Merritt(1994) *Wetlands, Industry & Wildlife: A manual of principles and practices*, The Wildfowl & Wetlands Trust

Road Markings	19,135	8,766	20%	9%
Building Paint	6,918	1,826	20%	9%
Artificial Turf	1,088	91	3%	1%
Total	**191,304**	**57,041**	**21%**	**9%**

The total estimated emissions are 57-191 thousand—this is greater than the amount estimated to go through WWT plants. However, in this case there are huge number of potential points in which these can enter waterways. This is compared with the relatively few WWT plants. It would also certainly not be cost effective to place a wetland at the end of every storm water run-off pipe especially in more dispersed rural areas.

For this reason, it will be very unlikely that full coverage could be achieved via this method. Improvements to storm water capture would be more cost effective in urban areas where the concentration of emissions is highest. Urban areas also account for 40-50% of the total run-off emissions.

Hotspots for emissions would have to be identified. This could initially be carried out by simply looking for the roads which have the most traffic over the course of a year. The run-off from these should be sampled to ascertain the concentrations levels that are present. A key question that would also need to be answered is what level of concentration is deemed high enough to install mitigating measures. This decision is likely to be made at a Member State level based on local sampling. At this point it is too early to be able to estimate the true cost of this or whether it is a cost-effective measure.

A.6.4.4 Innovative Wastewater Technologies

Wasser 3.0[561] is a project led by University of Koblenz-Landau to investigate solutions to purify wastewater to prevent pollutants entering water systems. Below is a summary of the technology from information provided by Dr. Katrin Schuhen.

What is the basic science behind the technology?

- The scientific and innovative part of Wasser 3.0 is the development of new hybrid silica materials and new processing and engineering surrounding for the material application for the elimination of organic in both soluble and solids form.

[561] http://www.wasserdreinull.de/en/home.html

- The technique can be transferred to microplastic elimination. This leads to a single solution for the elimination of soluble organic stressors (e.g. pharmaceuticals, pesticides….) and solid microplastic particles.
- The microplastics bind together to create larger plastic lumps that can be easily separated of filtered from the effluent.

What benefits does your technology have over other methods of capturing microplastics in waste water treatment?

- The technology is the first system which can be implemented on the one hand in the 4th cleaning step of a WWT and on the other hand it could be single used in e.g. industrial application.
- If we can implement in addition to the technical aspects the analytical solution for microplastic/particle visualization/detection for our continuous system, we are able to confirm the 2in1-removal of all kind of organic anthropogenic stressors from every water.

How effective is it?

- After the lab and batch scale the technology is able to remove up to 95% of microplastics without large technical changes at the WWT plant.

What is done with the microplastics after they are captured?

- After thermal decomposition we get SiO_2, CO_2 and water. Beside this known non-toxic decomposition step experiments are currently being run to find more about the captured system. Secondary material recycling without an additional technological step is a possibility. This is future works for 2018.

What stage is the testing at?

- The process and engineering partners are building the plant. We will run the first test in December 2017.

A.7.0 Data Quality Assessment

The data quality process is outlined as follows: The aim is for all data assessed against the below matrix should score 3 or lower in each category. Data or assumptions that score 4 or 5 constitute a data gap. If the mass flow model demonstrates that missing data accounts for less than 10% of the result, it will be deemed low priority. Anything less than 1% will be considered for exclusion from the model. Higher priority data gaps (>10%) will be followed up initially by attempting to gain data from relevant stakeholders. If they refuse or cannot provide data this will be flagged. If the data gap is very high priority (>50%) this will be flagged directly to the Commission for discussion.

Table 113 – Data Quality Matrix

Score	1	2	3	4	5
Reliability	Verified based on measurements	Verified data partly based on assumptions	Non-verified data based on estimates	Qualified estimate (e.g. by expert)	Non-qualified estimate
Completeness	Representative data	Representative for some scenarios.	Representative for few scenarios	Only representative of one scenario	Representativeness unknown
Temporal	<3 years old	<6 years old	<10 years old	<15 years old	Age unknown
Geography	EU level data	Average data from one or more member state	Comparable Member state data	Similar geography conditions	Unknown or very different geography

The overall impact of the data on the calculations is also assessed qualitatively. A score of 1—5 is given to each data source with 1 having a very low impact on the calculated figure for microplastics emissions and 5 having a very high impact. The data quality scores are multiplied with the impact scores to create a final score for that data source from Table 114.

Table 114 – Data source Overall Scoring

1-5	6-10	11-15	16-20	21-25
Data quality high and/or low impact on result	Likely to be good data quality and/or less impact on results	Data quality is likely to be average with an impact on results	Data quality may be low and/or impact on results is high	Unacceptable data quality and a high impact on results
High Certainty		Medium Certainly	Low Certainty	

Table 115 – Data Quality and Impact Assessment for Source Emissions of Microplastics

Data Quality Score 1-5 (1 is highest quality), Data Impact Assessment, 1-5 (5 is highest impact)

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Textiles - Clothing					
Sales data by clothing type, fibre type and fabric construction. **JRC (2014)[562]**	Composition of washing loads by fibre and construction type (knitted/weaved)	2 — EU data from a reliable source but with some implied calculations and assumptions. Specific to clothing but data is over 6 years' old.	4 – Key data source, but unlikely to be very different in reality and is similar to other composition sources.	**8**	Sales data gives an indication of what is being bought but might not be fully representative of what is being placed in a washing machine.

[562] JRC (2014) *Environmental Improvement Potential of Textiles (IMPRO-Textiles)*, Report for European Commission, January 2014

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Data on the number of wash cycles done per household on a country by country basis. **Pakula and Stamminger (2010)[563]**	The total number of washes done per year (EU 28 + Norway and Sweden)	3 – Study gathered data from several regional studies. These figures are used along with several key assumptions to apply to the rest of Europe.	4 – Key data source. Data is likely fairly accurate overall and is similar with that from other recent studies[564].	**12**	Would have been more accurate to have more localised research so that fewer assumptions were necessary. More recent data would be valuable.
The number of households in the EU on a country by country basis. **Eurostat, 2016[565]**	Number of washes per country (EU 28)	1 – Eurostat data likely to be of high quality. Recent, verified data.	4 – Key data source.	**4**	n/a
Data on number of households in Switzerland and Norway. **UNECE, 2014[566]**	Number of washes per country (Norway and Switzerland).	1 – Eurostat data likely to be of high quality. Fairly recent, verified data.	2 – Falls in line with other countries. Low impact.	**2**	More recent data.
Release of microfibres from differing fabrics and fibre types. **De Falco[567]**	Fibre release from different fabrics and construction types.	3 – Very recent and most representative data on fibre release. Still relatively small scale.	5 – Very High impact.	**15**	Thorough research into how release varies between different garments and textile types rather than just woven or knitted.

[563] Christiane Pakula, and Rainer Stamminger (2010) Electricity and water consumption for laundry washing by washing machine worldwide, *Energy Efficiency*, Vol.3, No.4, pp.365–382

[564] AISE (International Association for Soaps, Detergents and Maintenance Products) (2014) AISE Consumers Habits Survey Summary

[565] Eurostat *Eurostat - Data Explorer. Number of private households by household composition, number of children and age of youngest child*, accessed 7 June 2017, http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=lfst_hhnhtych&lang=en

[566] UNECE *Private households by Household Type, Measurement, Country and Year*, accessed 7 June 2017, http://w3.unece.org/PXWeb2015PXWeb2015/pxweb/en/STAT/STAT__30-GE__02-Families_households/08_en_GEFHPrivHouse_r.px/

[567] De Falco, F., Gullo, M.P., Gentile, G., et al. (2017) Evaluation of microplastic release caused by textile washing processes of synthetic fabrics, *Environmental Pollution*

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Mermaids Life, ISE Appliances. 2017[568]	Average capacity of washing machine	3 – Recent data. Can presume it is based on a fair amount of assumptions. Mermaids paper quotes ISE website, source of actual figure not clear.	2 – Medium impact. Figure falls in line with other studies and appears logical.	**6**	Unclear how recent this data is or the scale of research leading to this result.
Fibre masses, weight and length, used to calculate a detex value for various fibre types. **De Falco[569]**	Used to convert fibre numbers given into mass (mg).	2 – Very recent. Fibres have been directly measured, but not all fibres, therefore some extrapolation was necessary.	5 – Very High impact.	**10**	More detailed research is needed into the release of fibres from different clothes as fibre lengths and masses likely vary between garment types. It would be useful to have the necessary conversions for all different types of synthetic materials rather than a few.
Consumer habits survey **AISE, 2014[570]**	% of washing machine that is full	3 – Fairly recent data. Survey included 200 people from each of the 23 European countries included.	2 – Medium impact on results	**6**	More recent data.
Average Score = 8					
Automotive Tyre Wear					

[568] Mermaids (2017) *Report on localization and estimation of laundry microplastics sources and on micro and nanoplastics present in washing wastewater effluents. Deliverable A1.*, May 2017

[569] De Falco, F., Gullo, M.P., Gentile, G., et al. (2017) Evaluation of microplastic release caused by textile washing processes of synthetic fabrics, *Environmental Pollution*

[570] AISE (International Association for Soaps, Detergents and Maintenance Products) (2014) AISE Consumers Habits Survey Summary

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Traffic Activity by Member State, Road Type and Vehicle Type. **Eurostat, 2012[571]**	Averaged across Member States and used as a factor to further disaggregate traffic data, already disaggregated by vehicle types, by road type prior to application to road type-specific wear rates.	2 – Averaged Member State data from a reliable source but missing for many member states.	4 – Key data source – Strong influence on estimated quantity of wear deposited due to impact on the specific wear rates that are applied. Strong influence on subsequent pathways analysis due to impact on distribution of wear across road types.	**8**	Finding such data for more (if not all) Member States so that it needn't be used as a factor to scale other forms of traffic activity data.
Total Traffic Activity by Member State – not disaggregated. **OECD, 2013[572]**	For 19 Member States this is disaggregated using tyre sales data to arrive at European-level traffic activity disaggregated by vehicle type.	2 – Data from a reliable source but missing for 6 member states.	4 – Key data source for many Member States. Strong influence on estimations of emissions at source.	**8**	Finding more recent data
Traffic Activity by Member State and Vehicle Type **Eurostat, 2012[573]**	For 12 Member States this is summed with the traffic activity data collected from the OECD (which is scaled by sales data disaggregated by vehicle type) to arrive at European-level traffic activity data disaggregated by vehicle type.	2 – Data from a reliable source but missing for 6 member states.	4 – Key Data source for many Member States. Strong influence on estimations of emissions at source.	**8**	Finding this data for more countries so that scaling by tyre sales is used for fewer.

[571] Eurostat (2016) Road traffic on national territory by type of vehicle and type of road (million Vkm)
[572] OECD (2013) *"Road traffic, vehicles and networks", in Environment at a Glance 2013: OECD Indicators*, 2013
[573] Eurostat (2016) Road traffic on national territory by type of vehicle and type of road (million Vkm)

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
European Tyre Sales Data by Member State and Vehicle Type **ETRMA, 2016[574]**	Used to disaggregate OECD, and in two cases (Finland and Lithuania) Eurostat, total national traffic activity data by vehicle type.	2 – Reliable data source. Missing for Switzerland.	4 – Key Data source. Applied to 21 Member States. Strong influence on calculated quantity of wear deposited due to impact on the specific wear rates that are applied.	**8**	Finding this data for Switzerland so that total traffic activity can be disaggregated by vehicle type. Alternatively finding traffic activity already disaggregated by vehicle type.
Wear Rates for Passenger Cars and Trucks (lower and upper bounds) **ETRMA, 2017[575]**	Applied to traffic activity data to estimate European deposited tyre wear-derived microplastics by vehicle type (only used to confirm accuracy of alternate approach).	3 – ETRMA have not supplied evidence of how these wear rates have been derived, but as an industry source it is expected to be reasonably reliable.	3 – Influential data in that the total deposited microplastics it contributes to estimating justifies the use of Ten Broeke et al. (2016)'s wear rates (see below).	**n/a**	None
Wear Rates by Vehicle Type (Motorcycle, Passenger Car, Bus, Van and Lorry) and Road Type (Rural, Urban and Highway) **Deltares and TNO (2016)[576]**	Applied to traffic activity data to estimate European deposited tyre wear-derived microplastics by vehicle type and road type.	3 – Non-verified data based in part on estimate/expert opinion	5 – Key Data source. Strong influence on estimated quantity of wear. Strong influence on subsequent pathways analysis due to impact on distribution of wear across road types.	**15**	Finding more robust wear rates disaggregated by both vehicle type and road type.
Average Score = 9					

[574] ETRMA (2016) *European Tyre & Rubber Industry Statistics Edition*, 2016
[575] Personal Communication
[576] Deltares and TNO (2016) Bandenslijtage wegverkeer, on behalf of Rijkswaterstaat - WVL

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Paints - Road					
Data from DOW for road paint sales in 15 EU countries from 2006.	Extrapolated up to all EU countries using road length data and using GDP increases, brought up to 2015.	3 - It is unclear how DOW derived the data, but as an industry source it is expected to be reasonably reliable. The increases to 2015 are less well understood and not all countries were represented.	5- Used as the basis for all following calculations	**15**	Up to date data for all European countries is desirably to improve accuracy.
Road Length data for EU countries from European Union Road Federation (ERF) from 2012.	Used to fill data gaps in paint sales data by using reported road lengths.	3 - Road lengths are reported differently from country to country and the amount of paint used per km may also differ.	3 – Accounts for around 25% of the sales data that is missing.	**9**	As above. This is not needed if sales data is complete
Datasheets from various paint manufacturers which state the solid content of their paints.	Used as a factor for the amount of paint that can be considered a microplastic	1 – Data from industry is considered representative. Where values differ, a range is presented	3 – Is a factor with a high influence, but is unlikely to vary much.	**3**	None
New roads—Data from Eurostat for increase in road lengths and one data point from Germany for road paint use on existing roads.	Used as a factor for the amount of new roads that are built that will not contribute to microplastics emissions.	2 – Eurostat data on roads for 14 years, but is not complete for all countries and road measurement can be different.	2 – Used as a range which the true value is likely to fall into therefore the effect on results may be minimal.	**4**	More data of country examples of the amount of paint that is used on existing roads.
Split between urban, rural and highways roads from Eurostat. Averaged from seven EU countries.	Used to calculate which type of roads the microplastics are emitted on.	3 – Averaged from seven member states which had data available. Cross checked with data from EFR on highway lengths.	3– The road compartment that microplastics land effects the rate of capture but the results are not hugely sensitive to changes.	**9**	More country level data (especially for countries such as Germany, France, Italy etc.) on road types.
Average score = 8					

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Paints - Marine					
Details paint sales and application data provided by CEPE	Used as the basis for the total marine paint used per year	1 — Best available industry data.	4—key data source	**4**	
Emissions report from CEPE provided as written response to interim report	Used for the total solids content	1 — Best available industry data	4—key data source	**4**	
OECD Emissions Scenario Document and estimates provided by CEPE	Used the emissions assumptions provided for wear and removal. These were contested by CEPE although their basis is also still theoretical	3 – Fairly old (OECD) and based on expert opinion	5 – Is the entire basis for emission estimates	**15**	Sampling at source (and in WWT) to determine paint emissions.
Average Score = 12					
Paints – Building Paints					
Details paint sales and application data provided by CEPE	Used as the basis for the total decorative paint used per year	1 — Best available industry data.	4—key data source	**4**	
Emissions report from CEPE provided as written response to interim report	Used for the total solids content	1 — Best available industry data	4—key data source	**4**	

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
OECD Emissions Scenario Document and estimates provided by CEPE	Used the emissions assumptions provided for wear and removal. These were contested by CEPE although their basis is also still theoretical	3 – Fairly old (OECD) and based on expert opinion	5 – Is the entire basis for emission estimates	**15**	Sampling at source (and in WWT) to determine paint emissions.
CEPE sales data	Split of DIY/Trade	1 — Best available industry data	4—key data source	**4**	
Average Score = 7					
Artificial Turf					
ESTO Market Report	Used for installed pitch data	3 - Data provided by local FA's so possibly underrepresenting the true scale. Many countries had to be extrapolated	4—key data source	**12**	
FIFA provided data on 3,000 installed pitches	Used to calculated infill density	1—Only for FIFA certified turf, but from 3,000 installations so considered representative.	4—key data source	**4**	
Industry communication	Infill replacement rates suggested at 3% per year. Range of 1—4% used	4—Estimated replacement rates which may not accurately reflect the true loss rates, however is likely to fall within this range.	5—Key Assumption	**20**	Direct sampling from pitches and a mass flow analysis of infill.
Average Score = 12					

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Pellets and other Pre-Production Plastics					
Material handled by producers.[577]	Used to calculate the quantity of material lost at producers' sites, intermediary facilities and offsite waste management companies.	3 – Non-verified data published by Plastics Europe. Likely to be an informed estimate.	5 – Model results are highly sensitive to this value. Value differs significantly from figure published by Eurostat suggesting high level of uncertainty or large a in one of the sources.	15	Consult Plastics Europe to provide complete transparency on how this figure was calculated.
Material handled by processors.[578]	Used to calculate the quantity of material lost at processor sites and offsite waste management companies.	3 – Non-verified data published by Plastics Europe. Likely to be an informed estimate.	3 – Medium level of sensitivity.	9	Consult Plastics Europe to provide complete transparency on how this figure was calculated.
Extra-EU Trade in Pre-Production Plastics.[579]	Used to calculate the quantity of material lost through shipping, and at intermediary facilities.	2 – Eurostat Data is likely to be accurate for extra-EU trade. However, we use it as a proxy for all shipping of this material into and around the EU.	2 – Not a key input.	4	Source shipping data into and around the EU.
Percentage of material handled that ends up as waste.[580]	Used to calculate the quantity of material lost at offsite waste management companies.	4 – One component in the figure is over 15 years old and measured at a site in the US.	2 – Not a key input.	8	Consult facilities or industry bodies to provide an alternative figure.

[577] Plastics Europe (2016) *Plastics – the Facts 2015: An analysis of European plastics production, demand and waste data*. Includes CH and NO but these countries only account for 2% of plastics demand in Europe.
[578] Plastics Europe (2016) *Plastics – the Facts 2015: An analysis of European plastics production, demand and waste data*. Includes CH and NO but these countries only account for 2% of plastics demand in Europe.
[579] Eurostat Extra-EU trade in 20XX, pre-production plastics product codes.
[580] Eunomia (2016), Report for Fidra on Study to Quantify Pellet Emissions in the UK, March 2016.

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Average number of times material is handled at facilities between the producer and the processor.	Used to calculate the quantity of material lost at intermediary facilities.	5 – There is no public data available to inform this on an EU level.	5 – This one of the most sensitive inputs in the calculations for this source of microplastics.	**25**	Consult industry bodies and companies involved in supply chain to provide a more accurate figure.
Loss rate of pre-production plastics at plastics facilities (not signed up to OCS).[581]	Used to calculate the lower range of the quantity of material lost at producers' sites, intermediary facilities, processors and offsite waste management companies.	4 – Based on survey of OCS signee facilities and assumes non-OCS facilities lose 10 times more material. The basis for using this factor of 10 is not clear and is probably the author's own estimate.	5 – This one of the most sensitive inputs in the calculations for this source of microplastics. As the number is so small and derived from a very rough calculation it is particularly sensitive to inaccuracy.	**20**	Additional primary research of losses at these facilities would be useful but it is ultimately very difficult to establish an accurate EU-wide estimate due to variations from facility to facility and region to region.
Loss rate of pre-production plastics at one Norwegian processor.[582]	Used to calculate the higher range of the quantity of material lost at producers' sites, intermediary facilities, processors and offsite waste management companies.	4 – Based on empirical measurements at a processing facility but it is not known how representative this one facility is of losses from the EU plastics industry as a whole.	5 – This one of the most sensitive inputs in the calculations for this source of microplastics. As the number is so small and derived from a very rough calculation it is particularly sensitive to inaccuracy.	**20**	Additional primary research of losses at these facilities would be useful but it is ultimately very difficult to establish an accurate EU-wide estimate due to variations from facility to facility and region to region.

[581] Carsten Lassen (2015) *Microplastics - Occurrence, effects and sources of releases to the environment in Denmark*, Report for The Danish Environmental Protection Agency, 2015

[582] Mepex (2014) *Sources of microplastic pollution to the marine environment*, Report for Norwegian Environment Agency, April 2014

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Loss rates of pre-production plastics during shipping.[583]	Used to calculate the quantity of material lost through shipping.	2 – Based on container losses in global shipping over periods of 3 and 6 years. Data is up to 9 years old. Although data is for global shipping we would not expect EU shipping to be significantly different.	3 – It is difficult to estimate the level of uncertainty in this value so the impact is unknown.	**6**	None.
		Average Score = 13			
		Automotive Brake Dust			
Traffic Activity by Member State, Road Type and Vehicle Type. **Eurostat, 2012[584]**	Averaged across Member States and used as a factor to further disaggregate traffic data, already disaggregated by vehicle types, by road type prior to application to road type-specific wear rates.	2 – Averaged Member State data from a reliable source but missing for many member states.	4 – Key data source – Strong influence on estimated quantity of wear deposited due to impact on the specific wear rates that are applied. Strong influence on subsequent pathways analysis due to impact on distribution of wear across road types.	**8**	Finding such data for more (if not all) Member States so that it needn't be used as a factor to scale other forms of traffic activity data.

[583] Based on Marine Insight (2014) *Survey: How Many Containers are Lost at Sea?*, http://www.marineinsight.com/shipping-news/survey-how-many-containers-are-lost-at-sea/
[584] Eurostat (2016) Road traffic on national territory by type of vehicle and type of road (million Vkm)

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Total Traffic Activity by Member State – not disaggregated. **OECD, 2013**[585]	For 19 Member States this is disaggregated using tyre sales data to arrive at European-level traffic activity disaggregated by vehicle type.	2 – Data from a reliable source but missing for 6 member states.	4 – Key data source for many Member States. Strong influence on estimations of emissions at source.	**8**	Finding more recent data
Traffic Activity by Member State and Vehicle Type **Eurostat, 2012**[586]	For 12 Member States this is summed with the traffic activity data collected from the OECD (which is scaled by sales data disaggregated by vehicle type) to arrive at European-level traffic activity data disaggregated by vehicle type.	2 – Data from a reliable source but missing for 6 member states.	4 – Key Data source for many Member States. Strong influence on estimations of emissions at source.	**8**	Finding this data for more countries so that scaling by tyre sales is used for fewer.
European Tyre Sales Data by Member State and Vehicle Type **ETRMA, 2016**[587]	Used to disaggregate OECD, and in two cases (Finland and Lithuania) Eurostat, total national traffic activity data by vehicle type.	2 – Reliable data source. Missing for Switzerland.	4 – Key Data source. Applied to 21 Member States. Strong influence on calculated quantity of wear deposited due to impact on the specific wear rates that are applied.	**8**	Finding this data for Switzerland so that total traffic activity can be disaggregated by vehicle type. Alternatively finding traffic activity already disaggregated by vehicle type.
Luhana et al. (2004)	Wear rates	2—appear to be representative	3—Important source	**6**	

[585] OECD (2013) *"Road traffic, vehicles and networks", in Environment at a Glance 2013: OECD Indicators*, 2013
[586] Eurostat (2016) Road traffic on national territory by type of vehicle and type of road (million Vkm)
[587] ETRMA (2016) *European Tyre & Rubber Industry Statistics Edition*, 2016

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Ntziachristos, L., and Boulter, P. (2016)	Course fraction ranges	2—The absolute figure is uncertain, but the true figure is likely to be in this range	3—Important source	**6**	
Ntziachristos, L., and Boulter, P. (2016)	Entrapment of particles at 50%	4—This figure is highly uncertain and affected by a range of localised factors	4—reduces overall emissions by 50%	**16**	
Average Score = 9					
Fishing Gear					
Eurostat (PRODCOM)	Amount of fishing nets used per year from directly reported figures for 2015 EU28. Also figures for Iceland and Norway are extrapolated from catch data.	4—Unclear how representative this is as there is no other data to verify against. It is likely underreporting.	5—Very important	**20**	Verified fishing industry data
Magnusson et al. (2016)	Used estimated loss rate of 1—10%	3—This estimate is not based on data, but is likely to be in this range	5—Very important	**15**	More research is needed to verify the loss rates from nets
Average Score = 18					
Pathways to the Aquatic Environment					
Residential Sewerage					

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Eurostat	Data on primary, secondary and tertiary treatment processes is used to	3—Data is not current for all countries, especially those who continue to improve under the urban wastewater treatment directive	5—The type of treatment across Europe has a very large baring on the results	**15**	
Various studies on the effectiveness of WWT microplastic retention	The upper and lower bounds for each treatment type are applied to the Eurostat data	2—Retention rates can differ greatly and there is no standardised test. However, the true figure is likely to be in these ranges	4—High importance	**8**	More research into retention rates for a range of treatment types and a standardised test protocol.
Various data points on combined sewers	The assumption of 50:50 is used for combined and separate sewers and a 5% CSO release.	3—No firm data is available and is very country specific.	2—The main difference is CSO's so the impact is minimal	**5**	Improved data on CSO's and the country level sewerage systems in place.
Average Score = 9					
Urban Roads (and urban non-road drains without road cleaning)					
Number of rainfall days per year in Europe	Fraction of wear deposited on roads which are swept which is removed by rainfall runoff	3 – Reliable data from National Meteorological Agencies. However, data was averaged for locations in regions of Members States rather than being Member State-level figures.	2 – Has a relatively strong influence on the portion of wear captured on urban roads, but a low influence on the pathway as a whole	**6**	Finding data for more European Member States or a Europe-wide average.
Efficiency of road sweeping technologies in removing dust from the road surface	Averaged to calculate an estimate of the average efficiency for all technologies in capturing tyre wear particles	2 – Data from recent extensive literature reviews. Likely to be accurate.	3 – Has a relatively strong influence on the portion of wear captured on urban roads.	**6**	None

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
Eurostat	Data on primary, secondary and tertiary treatment processes is used to	3—Data is not current for all countries, especially those who continue to improve under the urban wastewater treatment directive	5—The type of treatment across Europe has a very large baring on the results	**15**	
Various studies on the effectiveness of WWT microplastic retention	The upper and lower bounds for each treatment type are applied to the Eurostat data	2—Retention rates can differ greatly and there is no standardised test. However, the true figure is likely to be in these ranges	4—High importance	**8**	More research into retention rates for a range of treatment types and a standardised test protocol.
Various data points on combined sewers	The assumption of 50:50 is used for combined and separate sewers and a 5% CSO release.	3—No firm data is available and is very country specific.	2—The main difference is CSO's so the impact is minimal	**5**	Improved data on CSO's and the country level sewerage systems in place.
Stormtac Database	Data on the capture of particles in sedimentation devices—a range of 40-80%	2—Averaged from a large number of data sources so this range is likely to be representative	5—This is a key capture point on roads	**10**	
TNO (2016)	Estimate derived from but decreased to 30% to soil from 40%	5—lack of data on this subject requires assumptions.	5—Has a large influence on where particles end up	**25**	Better understanding of which compartments particles from roads end up in
Average Score = 11					
Rural Roads					

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
TNO (2016)	Proportion direct to surface water (remainder to soil). Estimate derived from this study with an additional 10% added for the upper range	5—lack of data on this subject requires assumptions.	5—Has a large influence on where particles end up	**25**	Better understanding of which compartments particles from roads end up in
Average Score = 25					
Highways					
Stormtac Database	Data on the capture of particles in sedimentation devices—a range of 40-80%	2—Averaged from a large number of data sources so this range is likely to be representative	5—This is a key capture point on roads	**10**	
Number of rainfall days per year in Europe	Fraction of wear deposited on roads which are swept which is removed by rainfall runoff	3 – Reliable data from National Meteorological Agencies. However, data was averaged for locations in regions of Members States rather than being Member State-level figures.	2 – Has a relatively strong influence on the portion of wear captured on urban roads, but a low influence on the pathway as a whole	**6**	Finding data for more European Member States or a Europe-wide average.
Efficiency of road sweeping technologies in removing dust from the road surface	Averaged to calculate an estimate of the average efficiency for all technologies in capturing tyre wear particles	2 – Data from recent extensive literature reviews. Likely to be accurate.	3 – Has a relatively strong influence on the portion of wear captured on urban roads.	**6**	None
TNO (2016)	Derived from Urban but adjusted to 40% to account for larger soil areas nearby	5—lack of data on this subject requires assumptions.	5—Has a large influence on where particles end up	**25**	Better understanding of which compartments particles from roads end up in

Data Item	Used to Calculate	Data Quality Assessment	Data Impact Assessment	Score	Suggested Improvement
European Asphalt Pavement Association	Sales data extrapolated to estimate market penetration of 5% for highways	3—Data may not be representative	2— Less important	**6**	Country specific data—only Netherlands is available at the moment.
Stormtac Database	Capture rates for porous asphalt	2—Averaged from a large number of data sources so this range is likely to be representative	2—Less important	**4**	
Average Score = 10					

A.8.0 Summary of Calculation Changes

From the writing of the interim report several stakeholder meetings were held as well as significant interaction with the stakeholders outside of the meetings this had led to several methodological changes which are summarised in the following table.

Table 116 -Summary of Calculation Changes

Tyre Wear	No change to source emission figures
Brake Wear	No change to source emission figures
Pre-Production plastics	Upper estimate revised down from 195k to 167k due to reducing the number of times handled from five to four based on discussion with stakeholders.
Textiles	Total revised down from 68-121 ktonne to 18 – 33 ktonne. Discussions with stakeholders and new data became available which has led to a recalculation. Data from Mermaids project reports was found to be too unreliable to draw conclusions from
Artificial Sports Turf	No change to source emission figures
Paints and Coatings	This section is fully revised to take into account data provided by CEPE. Building and marine paints have reduced significantly whilst road markings have reduced, but remain within the same order of magnitude. Marine changed from 5.4 – 9 ktonne to 400 tonnes due to better information on the specific applications of marine paint where it would be subject to wear. Building paint changed from 92 ktonne to 21 – 35 ktonne due to better sales data reflecting a smaller proportion of paint sold for external use.
Fishing Gear	Revised down by around 50% due to a removal of categories deemed not refer to fishing nets in PRODCOM
Pathways Model	CSO release has been revised up to be 10% based on discussions in stakeholder meetings.

A.9.0 Stakeholder Consultation

Three workshops were held as part of the consultation process. The first was a general meeting with stakeholders from varied backgrounds. The two further meetings were dedicated to specific topics that were agreed with the Commission to be key areas that required significant stakeholder engagement and discussion.

A.9.1 Stakeholder Meeting, Brussels, 6th July 2017

The morning session involved a presentation by the project team on the quantification exercise and to gain feedback from stakeholders around this.
The afternoon session focused on presenting and discussing the long list of proposed measures. Separate focused sessions looked at tyres, pellets, textiles and paints. The following key points arose from these sessions;

- **Textiles**
    - **The main agreement was the need for a standardised test for fibre release.** The following points were also made by individual participants, and were uncontested:
        - The industry would be willing to consider implementing best practices, but needs more reliable and representative evidence to understand what these might be.
        - It was suggested that accreditation measures may be hard to implement because of the imperfect evidence base.
        - The textiles BREF isn't set up to address this type of activity. It is supposed to be for pre-treatment and will only affect EU manufacturers – and less than 25% of garments sold in the EU are manufactured in the EU.
        - Industry is ready to support best practice, but needs reliable and representative information
- **Tyres**
    - **Several key points were made which were investigated further after the meeting:**
        - There are no currently standardised tests to determine the full range of abrasion rates
        - Changing a key parameter in the performance of a tyre, such as abrasion rate, may influence other key parameters such as wet grip or rolling resistance.
        - Difficult to develop a standardised test for tread abrasion rate, as it must be repeatable, cost-effective, and reflect real-world conditions.
        - There is a precedent set by the development of a standard test for Fuel Efficiency which has fed into incorporation in the EU Tyre Label and the inclusion of requirements under the Type Approval Regulation for Tyres.
- **Paints**
    - **The main agreed action was for CEPE to look into this issue in more detail and provide a position statement on the current state of knowledge.**
        - It was agreed that the microplastic issue is quite new, and that research on marine paints, abrasion and microplastic release is only now being looked at.

        - There is knowledge about different types and durability in general – but there are specific gaps around how paints degrade vs erode and the implications for release of microplastics vs other.
        - Across all types – road marking, marine, building paints - it was suggested by a number of participants that there should be a move towards more durable paints – those already in existence where available and development of new types. However, it was cautioned that it would likely take decades before any ‘new’ preferred paints are developed.
- **Pellets**
    - **It was mostly agreed that a procurement led approach has the most potential, but needs to be mandatory rather than voluntary. The action was to investigate further.**
        - It was highlighted that there are a huge number of companies which handle pellets at which loss can occur and so targeting just one group such as producers through amendments to the BREF may be ineffective.
        - It was noted by stakeholders that plastics converters are not currently engaged in Operation Clean Sweep (OCS) to the same extent as producers.
        - The efficacy of OCS is currently only measured in terms of the number of member companies and associations engaged
        - There was agreement that demonstrating the impacts of OCS in terms of reduced emissions of microplastics to the environment is challenging, and that there are no ready solutions to this challenge other than communicating the levels of engagement and developing measures for the quality of implementation of OCS.

A.9.2 Microplastics and the Implications for Wastewater Treatment, Brussels, 13th September 2017

A summary of the main topics discussed is as follows:

- **WWT standards**
  Some plants had especially high levels of filtration (usually where the water was needed for drinking) and that the WWTP microplastics removal rate went from the simplest (around 20% efficiency) to the highest rate (98%+). It was advised that the plants with the highest levels of standards have high costs associated with them. When asked about the possibility of bringing other WWTP’s up to the highest standard, Eureau advised that the problem is a lack of money and the long life cycles of WWTP’s.
- **The responsibility of WWTPs vs producer responsibility**
  This subject was a key theme throughout the stakeholder meeting. The presentation given by Eureau acknowledged the issue of microplastics but suggested that producer responsibility was the best way to tackle pollution. Eureau made the argument that if the focus was on improving capture from WWT then there would be little incentive for polluters to stop polluting. This was met with some disagreement and it was argued that Eureau could not disregard their responsibility purely because there was a bigger issue elsewhere.
- **Biobeads**
  Being a new topic for many, there was significant interest around the subject of biobeads. There was a limited response from stakeholders on this subject. Eureau confirmed that their

members' plants were using biobeads (or similar plastic media) but stated that the beads were expensive and were not designed to be lost.

- **Regulations around sewage sludge**
  It was the general opinion from the WWT industry that if microplastics represented a significant issue then this would already have been noticed; the influence of sludge on land and crops had been tested for years on a global scale and no ill effects have been found.
- **Separating microplastics from sewage sludge**
  When asked directly whether it was possible to separate microplastics from sewage sludge Eureau answered that it was hard enough to find the microplastics let alone separate them. The point was raised that if sludge was incinerated anyway then there was no point in removing more microplastics.
- **Microplastic fibres occurrences in drinking water**
  There were concerns amongst participants regarding the reliability of sampling methods and quality assurance when measuring microplastics in drinking water. It transpired that there were currently no rules regarding microplastics in drinking water as the issue had 'popped out of nowhere'.
- **Solutions/ next steps**
  It was agreed that there was an urgent need for a standard testing procedure on a national, and preferably international level.

A.9.3 Stakeholder Workshop on Preventing the Loss of Pre-production Plastics (Pellets, Powders and Flakes), Brussels, 27th September 2017

The meeting focused on discussion of measures to prevent loss of pre-production plastic pellets with the following outcomes-

- **Amending the Polymer Production BREF**
  It was generally concluded that amending the BREF alone would not be sufficient, but could be considered by the Commission as a way of formalising best practice amongst producers.
- **Classifying pellets as waste such that pellet loss could be covered by waste regulation**
  It was generally concluded that this measure was inappropriate.
- **Including pellets under legislation covering the transportation of dangerous goods**
  It was generally concluded that a horizontal measure on transportation could represent an appropriate way of at least ensuring that robust containers are used for the transportation of pellets. However, concern remained about the interface between the different steps.
- **Regulation on Converters**
  It was felt that a horizontal measure on converters might tackle part of the issue, but would lead to problems at the interface between different steps
- **Pre-production plastics regulation which mandates supply chain standards**
  Overall there was consensus that this measure has potential to deliver significant improvements in the management of pellets throughout the supply chain.

A.9.4 Direct Stakeholder Interaction

Following on from the stakeholder meetings and in collaboration with the Commission, the list of measures was developed for further discussion with stakeholders. Although many more individuals and organisations contributed, the following is a summary of the key interactions:

- **Tyres**
    - ETRMA – Several discussions took place with the ETRMA. These mostly revolved around terminology and the ETRMA's instance that the report refer to the wear from tyres as 'tread abrasion rate'. This was duly noted and adopted. Further discussion on specific measures was ought, but not provided.
    - Tun Abdul Razak Research Centre – a meeting with four senior experts of this internationally renowned research institution which focuses on all aspects of vehicle tyre performance and testing, to discuss possible methods to test for abrasion rates
    - Continental Tyres and Bridgestone Tyres – ongoing discussions and exchanges of information, as well as a conference call with key technical staff at Continental to discuss testing procedures
- **Paints**
  As part of their commitment made during the stakeholder meeting, CEPE provided a report which gave a calculation methodology for microplastic paint emissions from building, marine and road paints. These figures were discussed and in the most part adopted for the final report.
- **Textiles**
  The project team took part in one of two meetings that was set up by DG GROW in the context of the forthcoming plastics strategy. The development of a test standard was discussed which everyone agreed was necessary. Using the test to develop a threshold was met with some resistance. Participants agreed to provide further data to support this stance, but none was forthcoming (there are many issues with the availability of data with the textiles industry).
    - Participants in the group were-
        - Euratex
        - FESI
        - CIRFS
        - AISE
        - CECED
        - EDANA
        - Orgalime
    - The project team also attempted to gain further insight into the EU funded 'Mermaids' project, which studied fibre release. The Mermaids team refused to provide any further detail around their methodology after questions were raised in the Stakeholder meeting. The project team were referred to a recent journal article published by the Mermaids team for definitive information, but also did not respond to questions about this article.

A.9.5 Open Public Consultation

The consultation ran from 26th June 2017 to 16th October 2017 and the total number of respondents was 487.

Overall, just under two thirds of the responses came from interested individuals/citizens, and over one third came from stakeholders/experts. The results described in the sections below are based on answers from all respondents**.** The results were also analysed using the separate data from individuals and stakeholders in order to establish whether there were differing views between the two groups. No strong differences were noted, so in the interest of brevity, only the overall results are shown in the figures in the following sections.

There are, however, some general observations around some of differences between individuals' and stakeholders' responses;

- Stakeholders showed a greater level of awareness of the possible sources of microplastics than individuals.
- Stakeholders were also more likely to respond "Don't Know" to questions, showing a greater level of caution for expressing their opinion on matters they were not experts in.
- Individuals showed a higher level of environmental concern than stakeholders and were more likely to response "Very effective" to the proposed measures, which, again reflects the higher level of caution exhibited by stakeholders.
- The top three microplastic sources which were of most environmental concern to individuals were clothing/textiles, cleaning products and cosmetics, whereas those of most concern to stakeholders were clothing and textiles, road tyres and pre-production pellets.
- The sources of least environmental concern for individuals were building paints, road paint and artificial sports turf, whereas for stakeholders, there were agricultural mulch films, industrial abrasives and artificial sports turf.

There was a consensus between individuals and stakeholders that the manufacturers of products concerned should bear the financial responsibility for reducing microplastics emissions to the marine environment, followed by the (public or private) waste and waste water treatment companies (costs potentially included in water price/taxes) and finally governments and tax payers.

A.9.6 Summary of Results for the Sources of Microplastic Emissions

A.9.6.1 Road Tyres

The measure that was thought to be the most effective to **reduce the wear rate of tyres** was legislation requiring producers to increase the durability of their tyres (including phasing out the least durable tyres over time).

The measure that was thought to be the most effective to **increase the capture of tyre particles** was the development and installation of technologies that are proven to capture microplastics in a municipal waste water treatment plant and prevent them from entering effluents (and subsequently surface waters).

The weight of responsibility for reducing tyre microplastic emissions was primarily attributed to the tyre industry.

A.9.6.2 Pre-production Plastic Pellets

The measure that was thought to be the most effective to **prevent supply chain loss through implementation of industry recognised best practice** was legislation at the EU level requiring all companies placing plastics on the EU market to demonstrate that their supply chain adheres to best practice as outlined in Operation Clean Sweep guidance.

The measure that was thought to be the most effective to **increase the capture of plastic pellets** was to mandate the installation of technologies that are proven to capture microplastics on manufacturing locations or sites handling pellets (it should be recognised that this is also part of industry best practice).

The weight of responsibility for reducing pre-production plastic pellets emissions was primarily attributed plastic pellets producers and plastic pellet converters.

A.9.6.3 Clothing and Textiles

The measure that was thought to be the most effective to **reduce the propensity of synthetic textiles to be shed from clothing** was the development of a mandatory requirement for the progressive reduction of microfiber release that must be adopted by manufacturers of clothing sold in the EU.

The measure that was thought to be the most effective to **increase the capture synthetic textiles shed from clothing** was the development and installation of technologies that are proven to capture microfibres in a municipal waste water treatment plant and prevent them from entering effluents (and subsequently surface waters).

The weight of responsibility for reducing synthetic fibre emissions was primarily attributed to textiles/fibres manufacturers and clothing manufacturers.

A.9.6.4 Artificial Sports Turf

The measure that was thought to be the most effective to **bring changes to handling and management of infill** was to mandate the installation of technologies that are proven to capture microplastics on sports turf sites e.g. drain traps or onsite waste water treatment.

The measure that was thought to be the most effective to **bring changes to the nature of the infill** was a ban on the use of polymer based infill as an infill material for artificial sports turf.

The weight of responsibility for reducing emissions from artificial sports turf was primarily attributed to the artificial turf manufacturers and installers.

A.9.7 Written Responses

The following section details the written responses provided by consultees. They were given the opportunity at the end of the survey to provide their own written evidence up to a maximum of 4 pages.

A.9.8 Summary of Responses

Information presented in the additional documents provided by consultation respondents indicated that there is a strong element of commitment to reducing the impact of microplastics on the environment and an appreciation of the importance and severity of the issue overall.

In their entirety industry stakeholders emphasised their **commitment** to reducing the impact of their operations to the environment with common references to the **circular economy** and the constant drive for optimisation of existing inputs, materials, processes and outputs and the impact these have on the amount of microplastics found in the land and aquatic environment.

Among the responses received to the consultation there were **76 stakeholders** who provided additional documentation in support of their response. These documents varied from policy briefs to position statements and summaries of survey results. Table 1 shows the number of responses with additional documents sent by stakeholder category. Responses received originated mainly from private companies, industrial or trade associations and NGOs.

Table 117: Number of 'additional document' responses per stakeholder category

Stakeholder category	Number of responses
Industry stakeholders[588]	36
Non-governmental organisation (NGO)	21
Interested individual/citizen/consume	7
Academic/ scientist / research	5
Local / National authority / State-owned enterprise	3
European Institution / International body	3
Other association	2

A.9.8.1 Major sources of microplastics

In responses received from industry stakeholders and trade associations there was a strong urge to the European Commission to **identify all major sources of microplastics** and consider the **proportionality of contributions** to the issue by different sources and sectors prior to decisions on additional measures or restrictions. A number of other considerations, beyond proportionality, were also identified and are discussed in the context of stakeholders' recommendations and suggested solutions to the European Commission (see Section 4).

Noting that further research and science-based evidence is required, a relatively large number of stakeholder responses **objected to the ranking of major sources as well as the sectoral contributions** to microplastics. As long as uncertainty remains, there was a recommendation that this should be acknowledged and clearly communicated by the European Commission.

[588] Includes 'Private companies' and 'Industrial or Trade associations'

Looking at **sources of microplastics** pollution from a **geographic** point of view, the overwhelming majority of stakeholders referred to the importance of a **harmonised approach across European countries.** This was thought to be crucial to ensure efficiency, coherence and prevent the multiplication of national restrictions which can impede the function of the single European market and constitute a barrier to trade. A smaller number of stakeholders (mainly trade associations) argued that a European approach would not be adequate quoting studies on the relative contribution of non-EU countries showing that a small number of Asian countries[589] account for 2/3 of global inputs to microplastics (Jambeck et al., 2015)[590]. This was found to be particularly the case for stakeholders in the textile and garment industry.

On the issue of **industry specific sources of microplastics**, most stakeholders in their responses referred to a range of sources suggesting that a **cross-sectoral approach** involving actors and stakeholders throughout a product's life cycle and supply chain would be most effective.

In addition to the sources discussed in the consultation a large number of stakeholders across categories referred to **macroplastics** and macroplastic fragmentation as the most important source of microplastics, and questioned why this wasn't included in the survey. Another source that was thought to be missing was **waste water** and untreated discharges from Waste Water Treatment Plants (WWTP). It should be noted that opinions on waste water were slightly contradicting, with a large number of stakeholders noting that WWTPs can capture between 85% - 99%[591] of microplastics by implementing main flow advanced filtration treatment (e.g. membrane ultrafiltration techniques). In relation to waste water, the use of treated **sewage sludge** in agriculture (currently 50% in EU according to responses) was highlighted as a concern. There was consent across stakeholders that research would be necessary to establish sewage sludge concentration in microplastics as agricultural application implies their release and distribution in the environment

A.9.8.2 Key data gaps and research needs

Issues around data gaps and the need for **further research in the sources, release pathways, amounts and impacts of microplastics,** dominated the concerns of stakeholders across all categories. Addressing a number of gaps and shortcomings was highlighted as crucial to the quality of the public debate and the policy response to address the issue. Across industries, from tyres to cosmetics, industrial stakeholders suggested there is insufficient or unsubstantiated scientific evidence as to the impact and of their industry and overstatements of their contribution.

The main gaps, as identified by respondents, included:

- Knowledge gap regarding the **sources of origin of microplastics**. This includes a **lack of baseline data** as this is a new area of research.
- Knowledge gaps in the **volumes** of microplastics currently present in the environment.
- **Lack of a harmonized measurement / calculation models**. Available studies use a range of calculation models which, depending on the underlying assumptions, present huge

[589] Namely China, Indonesia, Philippines, Vietnam, Sri Lanka and Thailand
[590] Jambeck et al. (2015). Plastic waste input from land into the ocean. Science 347(6223), 768-771, February 13th 2015.
[591] Range across responses

discrepancies and inconsistencies in the volume of microplastics released in the environment. That was thought to be particularly acute for the textile sector.

- **Significant scientific uncertainty on the impacts** of microplastics. For example, a number of industrial stakeholders noted that the suggestion that microplastics "*facilitate the adsorption of toxic substances from the natural environment and increase their potential bioavailability to organisms throughout the food-chain*" has not been scientifically validated, and queried the data available to confirm the microplastics desorption in organisms and hence accumulation in the food chain, noting such studies are limited and only offer initial indications.
- Lack of monitoring and evaluation **of existing practices and voluntary initiatives.**

A.9.8.3 Suggested approaches and measures for addressing microplastics

Most of the recommendations formed by stakeholders are applicable across industries and are summarised below.

1) Additional research
2) Clear and agreed definitions
3) Emphasis on measures aimed to control pollution at the source
4) Measures that are proportionate and source-specific: no one-size-fits-all approach
5) Investment in R&D and support in innovation across sectors
6) Multi-level governance and stakeholder involvement
7) EU-wide regulatory measures aimed at limiting the use and release of microplastics
8) Consumer choice measures
9) Voluntary industry initiatives