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Fragmented Fibre (Including Microplastic) Pollution from Textiles

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1. Introduction

The threat of microplastics to the environment and human health has been highlighted in the past decade. It is estimated that 5 trillion plastic particles corresponding to 270,000 tons are floating in the world's oceans. From which nearly 93% are considered to be at micro-scale (Zambrano et al., 2019). Microplastics have been pointed out as potential hazards to living organism's health by ingestion, inhalation, entanglement and dermal contact and/or by adsorption and transport of hazardous chemicals (e.g. dyes, pigments, mordants) (Dris et al., 2016; Cesa et al., 2017; Catarino et al., 2018). Textiles have been proven to be a well-known source of fragmented fibres (FF) when domestic laundry is carried out (Rios Mendoza et al., 2018). Washing of synthetic fibres has been scored as 7/10 degree priority of microplastics source (Cesa et al., 2017). It is estimated that more than one-third of the microplastics found in the world's oceans are from textile origins (Boucher and Friot, 2017).

Furthermore, a significant mass of FF has been found in atmospheric fall-out that could be generated from textile abrasion of apparel, wear and tear, or travelled from fertilised crops with sewage sludge containing microplastics (Dris et al., 2016). The Ellen MacArthur Foundation estimated that 0.5 of the 57 million tonnes (MT) fibres for clothing end up as MP ocean pollution through the textile washing (Ellen MacArthur Foundation, 2017), which is expected to accumulate at current rates to 22 MT by 2050. The nomenclature of microplastics in terms of size and morphology will be discussed, and a definition relevant to fibrous microplastics will be proposed. The introduction will briefly cover the categorisation of microplastics and the textile origin of FF from textile washing and abrasion. Potential environmental impacts and health hazards of FF will be addressed in this section as well. The objectives of this Textile Progress are:

a. To summarise current methods for measurement and characterisation of FF released from textiles. This will include a critical discussion of published protocols/methods to inform future experimental work. This is particularly important as certain limitations in methods are likely to under/overestimate the amount of FF.
b. To critically review existing studies on fragmented fibres from textiles. Current published literature contains clear evidence on the widespread of fragmented fibres from textile washing to aquatic sources and terrestrial environments by sewage sludge fertiliser with MP. However, this issue will also include the analysis of FF released into the indoor and outdoor environment through dry abrasion of textiles.
c. To explore the impact of textile materials, structure, use and service conditions on the release of FF. This will include a critical comparison of experimental results from published studies.

d. To review mitigation strategies to curb the release of FF. This includes, but is not limited to, textile finishes and structural changes, retrofit washing devices and washing effluent filters.
e. To provide a summary of risks associated with FF from textiles on living organisms, including humans.

1.1 Scope of work

The topic of MP is broad and includes a vast body of research across various disciplines. This publication is focused on Fragmented Fibres (FF) generated and released from textiles during wet (predominantly washing) and dry exposure conditions. The scope of this work is to critically review the definition of MP and, in particular, reference to FF. The work includes a summary and critical review of existing methods used to generate, collect, characterise, and quantify FF released from textile laundry. Furthermore, the impact of textile material and structural parameters on the release of FF has been discussed. In the absence of a standard methodology, any direct comparisons are not possible. The review analysed data from different studies to prepare an estimated comparison. Finally, it covers a comprehensive overview of mitigation strategies to limit the release of FF from textile sources during laundry. A broad description of the collection of FF from the environment (atmospheric deposition, aquatic, and terrestrial environments) and their impact on the environment and human health are provided, but this is not the key focus of the review. The remediation of FF from wastewater treatment plants, aquatic, and terrestrial environments is out of the scope of the present work. However, the appropriate sections have signposted key literature for the relevant areas.

1.2 Microplastics definition

To achieve a better understanding of addressing the problems of microplastic (MP) pollution, there is a need to ensure a common terminology. Until now, an international definition for microplastics or fragmented fibre pollution has not been established; hence the terminology remains ambiguous (Hartmann et al., 2019). Although creating a common nomenclature could limit the scientific freedom to study the field, as Hartmann et al. (2019) mentioned, it is essential to tackle the environmental problems caused by microplastics. The definition for microplastics followed throughout this document is described below. Three main points regarding MP definition that have not been agreed upon are discussed throughout this section, including the classification by source, the size limits, and the materials.

Microplastics (MP)

Polymer-based material particles in size range between <5mm to 100nm, comprising thermoplastics as both virgin plastic resin pellets and resins blended with any additive to enhance the material performance (including plasticisers, colourants, processing aids, stabilisers and fillers); additionally, thermoset polymer materials and polymers that cannot be melt-processed (either natural or synthetic) are also considered within the plastic definition (Sundt et al., 2014; GESAMP, 2015; Lassen et al., 2015; Andrady, 2017; Hann et al., 2018).

Microplastics were first reported in 1972 in a series of two publications (Andrady, 2011; Ryan, 2015; Hartmann et al., 2019; Laskar and Kumar, 2019). The first article studied plastic fragments and pellets collected in the western Sargasso Sea with a size ranging from 0.25 to 0.5 millimetres in diameter (Carpenter and Smith, 1972). A second publication reported an investigation of two different polystyrene spherules with a mean size of 0.5mm in coastal waters off southern New England (Carpenter et al., 1972). Even though these short publications do not include the term microplastics, they highlight the problem of granular plastic debris as a future concern and alert of environmental consequences. Additionally, it is suggested that these particles hold attachment sites for bacteria and polychlorinated biphenyls (PCB's), act as a source of toxic compounds, and generate intestinal blockage of marine individuals (Carpenter and Smith, 1972; Carpenter et al., 1972; Ryan, 2015).

Despite the age of these publications, it was not until 2004 that the term microplastics was first coined by Thompson et al. (Thompson et al., 2004; Seltenrich, 2015; Hartmann et al., 2019). While the term is not defined, it describes microscopic fragments and fibres with typical diameters down to ~20µm. The authors suggested the degradation of more oversized plastic items, clothing, and rope as the source of these particles, but intentional use of MP in cleaning agents was also mentioned.

1.2.1 Primary and secondary microplastics

The source of MP generation would determine their categorisation into primary and secondary microplastics. As well as the definition, this distinction has not been formally established, and variations among authors exist (GESAMP, 2015; Lassen et al., 2015). Two different approaches suggested in the current literature are raised below. Even though MP pollution sources remain the same, the assortment is different.

a) Primary microplastics are defined as particles initially produced in microscopic size, typically added to products, or used as raw materials, directly released into the environment. In comparison, the fragmentation of larger plastic debris creates secondary microplastics by degradation, use, weathering or waste management (Thompson et al., 2004; Ryan et al., 2009; Cole et al., 2011; Lassen et al., 2015; GESAMP, 2015; Thompson, 2015).

b) The second classification considers primary MP as intentionally produced or voluntarily added to the products and includes those created by degradation during the product lifecycle involving human activity. This includes manufacturing, use and maintenance. In contrast, secondary MP are only generated when degradation is caused by environmental processes such as weathering. (Sundt et al., 2014; Boucher and Friot, 2017).

For this document, the first classification is considered as it appears to be the most common within published literature. Therefore, some examples of documented primary MP sources are plastic pellets used as supply materials in the manufacture of larger plastic items and accidental spillage into the environment (Browne et al., 2011; Thompson, 2015), smaller MP particles used as “scrubbers” in cosmetic products such as exfoliating cleansers and facial scrubs (Gregory, 1996; Derraik, 2002; Fendall and Sewell, 2009; Thompson, 2015), or employed in air-blast cleaning media and synthetic sand-blasting media (Zitko and Hanlon, 1991; Gregory, 1996; Cole et al., 2011; Sundt et al., 2014).

Secondary microplastics are generated from the degradation or fragmentation of larger plastic items (Lassen et al., 2015). The former may be caused by one or more environmental factors (biodegradation, photodegradation, thermo-oxidative degradation, thermal degradation, and hydrolysis; see (Barnes et al., 2009; Andrady, 2011; Cole et al., 2011)). While fragmentation is a consequence of the use, maintenance, and cleaning of the product (Thompson, 2015; GESAMP, 2015). Synthetic fragmented fibres would fit into the secondary microplastics category (Andrady, 2017).

There are ongoing efforts to reduce the use of microplastics in the industry and ban them in the cosmetics (Thompson, 2015). A recent publication by ECHA recommended a complete ban of MP as infill materials by 2026 (ECHA, 2020). In this situation, secondary microplastics raise a greater concern since their generation cannot be stopped as it is not intentional. Additionally, they present higher annual emissions to surface waters (Hann et al., 2018), as seen in Figure 1. Thompson (2015) and GESAMP (2015); warn that even with the immediate cease of larger plastic items discharged into the sea, the ongoing degradation of legacy items would continue to increase microplastics generation for several more years. The source and fate of MP in the EU are presented in Figure 2. Tyres, road markings, plastic pellets, and synthetic textiles are placed as more significant sources of MP release among the rest of identified origins (Hann et al., 2018).

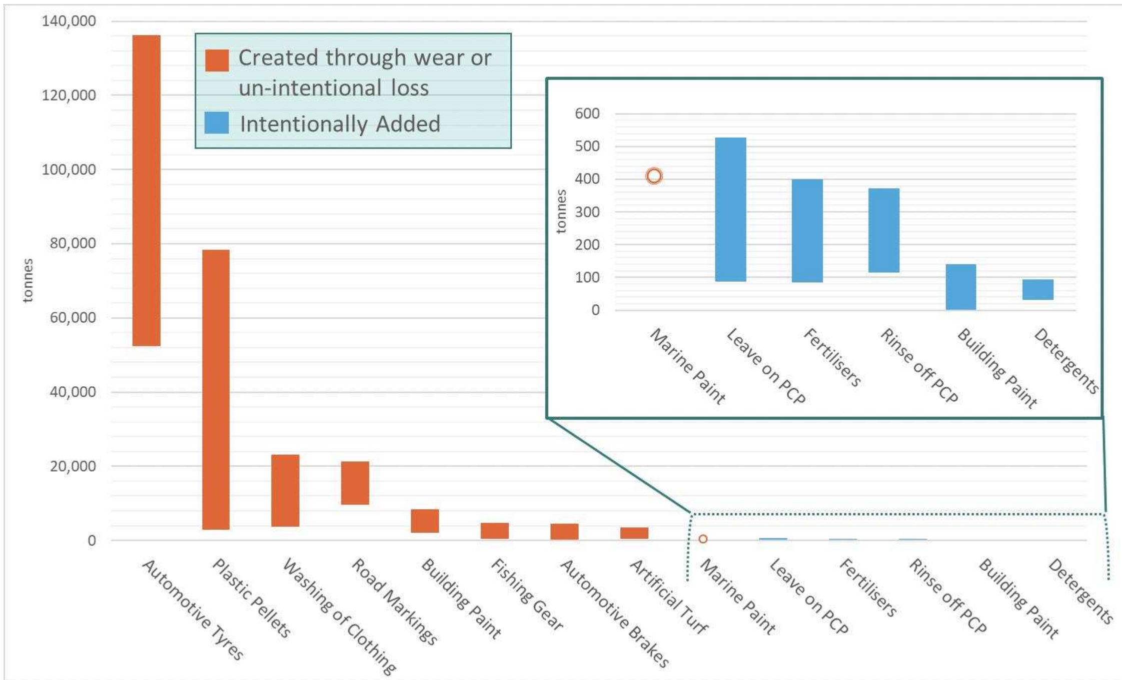


Figure 1 Annual emissions of microplastics to surface water (upper and lower ranges).
PCP – Personal care products
Reprinted with permission from Hann S., et al., 2018. Investigating options for reducing releases in the aquatic environment of microplastics emitted by (but not intentionally added in) products. Final Report, p.iii. Copyright 2018, Eunomia Research & Consulting and ICF (https://www.eunomia.co.uk/reports-tools/investigating-options-for-reducing-releases-in-the-aquatic-environment-of-microplastics-emitted-by-products/)

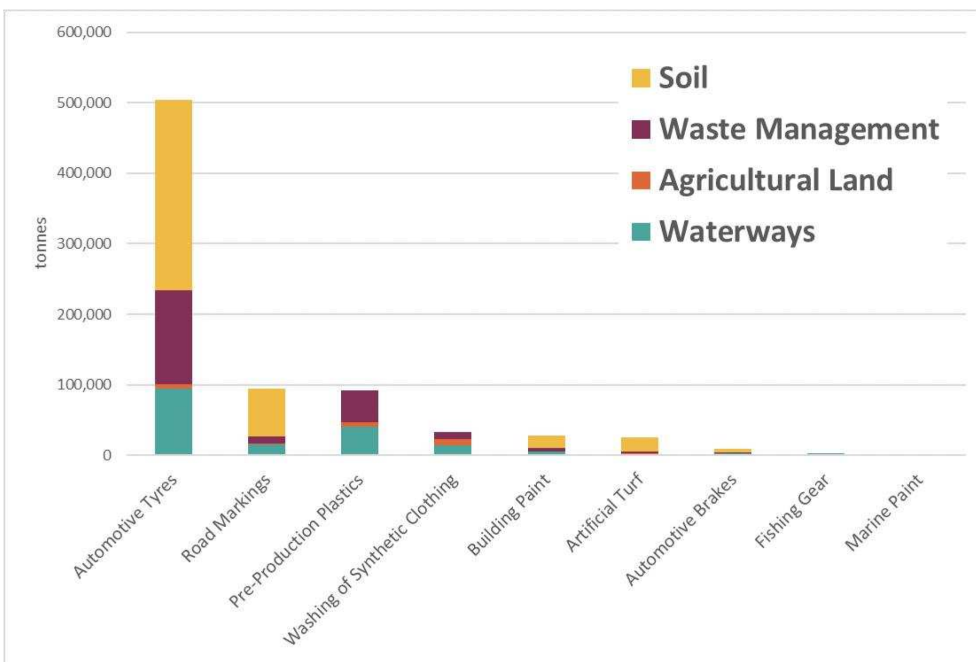


Figure 2 Source generation and fate of microplastics from wear and tear in the EU (midpoint).
Reprinted with permission from Hann S., et al., 2018. Investigating options for reducing releases in the aquatic environment of microplastics emitted by (but not intentionally added in) products. Final Report, p.iii. Copyright 2018, Eunomia Research & Consulting and ICF (https://www.eunomia.co.uk/reports-tools/investigating-options-for-reducing-releases-in-the-aquatic-environment-of-microplastics-emitted-by-products/)

1.2.2 Microplastic size limits

The lack of a formal definition represents the absence of upper and lower size limits of MP particles. Three different proposals to address this issue are discussed as follow:

a) The term "micro-litter" was defined in 2003 as *"inconspicuous, fine plastic detritus with a size range of very fine sand to coarse silt usually found in the marine sediment. On washing it passes through a 500µm sieve but is retained on one at 63µm"* ((Gregory and Andrady, 2003) p.381), and cited again by Andrady, (2011) as the *"barely visible particles that pass through a 500µm sieve but retained by a 67µm sieve (~0.06–0.5mm in diameter)"* ((Andrady, 2011) p.1597). Despite the slight variation in dimensions described by the author between publications, both definitions determine the size limits of the particles. However, subjective terms used to describe microplastics may be arguable and ambiguous.

b) A more common agreement was established in 2008 after the first conference on microplastics pollution held by National Oceanic and Atmospheric Administration (NOAA) in the U.S. Attendees from 6 different countries agreed to define microplastics as plastic particles smaller than 5mm in diameter (Betts, 2008; Thompson, 2015). This definition, also published in the memorandum from the conference workshop, established an upper limit based on the particle size considered more likely to be ingested by marine individuals (Arthur et al., 2009; Hartmann et al., 2019). At the same time, the lower limit of 333µm is grounded on the Neuston nets used for sampling (Arthur et al., 2009; Andrady, 2011)). Even though this definition was the first agreement on the size, it is a pragmatic description that lacks scientific robustness (Lassen et al., 2015; Hartmann et al., 2019). There is a variation of this description that considers a lower limit of 100nm in accordance with the definition of nanomaterials (Hartmann et al., 2019) and supported by the European Chemicals Agency (ECHA) (ECHA, 2020). Any plastic debris below this size could be considered a nano-plastic (Laskar and Kumar, 2019).

c) Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) proposed a classification model based on plastic debris sizes, summarised by Andrady, (2017) in Table 1, compared to size ranges recommended by the European Commission. Despite the suggested assortment, GESAMP has used the most common definition of MP previously discussed (<5mm) with a lower limit of 1nm for its environmental assessments (GESAMP, 2015).

Table 1 Classification of plastic debris in the environment.
Reprinted with permission from Andrady, A.L. 2017. The plastic in microplastics: A review. Marine Pollution Bulletin. 119(1), p.15. Copyright 2017, Elsevier Ltd. DOI: 10.1016/j.marpolbul.2017.01.082.

Class	***Size ranges GESAMP***	***Visualization***	***Technique***	***Size ranges (MSFD GES)****
Macroplastics	100 – 2.5cm	Naked eye	Visual counting	> 2.5cm
Mesoplastics	2.5cm – 0.1cm (1000µm)	Naked eye or optical microscope	Neuston nets or sieving	0.5cm – 2.5cm
Microplastics	0.1cm (1000µm) to 1µm	Optical microscope	Microfilters < 1µm separation	0.5cm (5000µm) to 1µm
Nanoplastics	< 1µm	Electron microscope	Nanofilters	< 1µm

* MSFD GES Technical Subgroup on Marine Litter (2013) Monitoring Guidance for Marine Litter in European Seas. Draft Report of European Commission. Brussels. (Van Cauwenberghe, L., et al., Microplastics in sediments: A review of techniques, occurrence and effects, Marine Environmental Research (2015), http://dx.doi.org/10.1016/j.marenvres.2015.06.007).

The lack of consistency throughout the definition of MP has resulted in communication and regulation challenges and incompatibility of results among publications (Hartmann et al., 2019), as shown in Figure 3. This publication follows the most accepted upper size limits ≤5 millimetres and lower limits of 0.1µm (100nm).

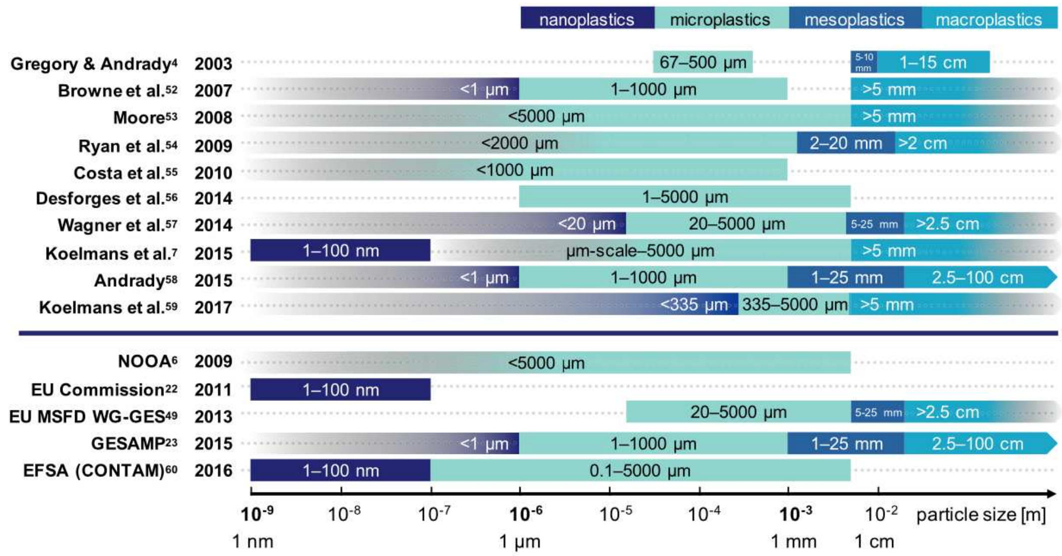


Figure 3 Examples of differences in the categorisation of plastic debris according to size as applied (and/or defined) in scientific literature and in institutional reports. It should be noted that this does not represent an exhaustive overview of all used size classes. Reprinted with permission from Hartmann et al., 2019. Are We Speaking the Same Language? Recommendations for a Definition and Categorization Framework for Plastic Debris. Environmental Science & Technology. 53(3), p.1040. Copyright 2019, American Chemical Society. DOI: 10.1021/acs.est.8b05297.

1.2.3 Materials

By definition, not all polymers are considered plastics regardless of their natural or synthetic origin (Andrady, 2017; Hartmann et al., 2019). Polymers are long chain-like structures of repeated molecules or chemical units (monomers) (Andrady, 2017). Within the polymers, plastics are those derived from oil or gas monomeric polymerisation (Cole et al., 2011) that can be melt-processed into products. Naturals and synthetic polymers that do not allow melt-processing are excluded from the plastic definition (Andrady, 2017). Consequently, fragmented particles of these items are not considered microplastics. In comparison, the European Parliament and Council defined plastics in 2006 as polymeric materials that may be enhanced with additives, excluding naturally occurring polymers in the environment. However, this definition has been adapted by the Directive of 2019 to include plastics manufactured with natural polymers that were modified or those manufactured from bio-based, synthetic, or fossil sources. This adapted definition also comprises polymer-based rubber, bio-based and biodegradable plastics regardless of their derivation source (even biomass) or biodegradation rate (European Parliament and Council, 2019). Therefore, not only the physicochemical properties should be examined to enlist in the MP categorisation, but also the environmental hazard and biodegradability that could make them contribute as MP pollutants (GESAMP, 2015; Andrady, 2017). In this publication, the polymers that cannot be melt-processed and thermoset polymers would also be considered MP apart from thermoplastics. To further explore the definition of plastic, it is suggested to consult (GESAMP, 2015; GESAMP, 2016; Andrady, 2017).

1.3 Fragmented fibres: textile generated pollution

The first report of textile fibre debris in the ocean was published after filtering water samples from the English Channel in 1954. Natural materials such as manila, coir and jute from rope or twine were found (Atkins et al., 1954). The first regenerated fibre fragments from cellulose fibrous material were detected in microscopic examinations of North East Pacific Ocean water samples six years later (McAllister et al., 1960). In comparison, the first report of synthetic fibres in the sea comes from 1971. Plankton samples in the Northumberland coast presented synthetic fibres impurities in alarming proportions, attributed to textile origins (Buchanan, 1971). The author suggested an increase in this material in the last decade due to synthetic marine cordage and netting. Even though sizes are not determined in the research, Buchanan (1971) alarms on the proportions and future growth of this debris at sea. These publications are vital to understanding textile fibres' background and evolution as marine pollutants and their leap between natural and synthetic materials.

Synthetic apparel has been widely proven to be a source of microplastics during the lifecycle of the products in dry and wet state (Browne et al., 2011; Pirc et al., 2016; Hernandez et al., 2017; Rios Mendoza et al., 2018; Zambrano et al., 2019; De Falco et al., 2020; Cai et al., 2020). Browne et al., (2011) first identified laundry as a source of microplastic fibre pollution (Pirc et al., 2016) created by mechanical and, or chemical actions in wet conditions and thermo- and photo-oxidation of the polymers (Singh et al., 2020). Adhering to the conventional description of plastics, only fibres of synthetic origin could be considered MP. However, not only synthetic textiles release fibres that could endanger the environment and potentially human health (Ladewig et al., 2015). Besides, it is essential to note that fibres possess a high ratio of length to thickness (Tasneem, 2018), and the proposed definition of MP could mislead the categorisation of fibrous debris as the mean diameter of the cross-section for most textile fibres is around 10 to 20μm (Cesa et al., 2017). This is highly relevant as chemical effects are aggravated by smaller particle sizes (Stanton et al., 2019; Singh et al., 2020).

A more inclusive term is required to describe this micro-debris from synthetic or other polymeric textile materials. Published literature refers to these specific particles as “microfibres” owing to their textile origin (Napper and Thompson, 2016; Hartline et al., 2016; Pirc et al., 2016; Cesa et al., 2017; Carney Almroth et al., 2018; Hann et al., 2018; De Falco et al., 2020). However, the textile industry commonly defines microfibres as fibre material finer than one denier or decitex and with a cross-section smaller than 10μm (Zambrano et al., 2019). So, for this study, the term Fragmented Fibres (FF) is suggested to define all fibrous masses released from any textile material, and it is described below.

Fragmented Fibres (FF), a sub-class of microplastics, can be defined as polymer-based materials created during the manufacturing, and/or use, and/or service of textile materials/products. The materials include any extruded polymer (synthetic or regenerated) and naturally occurring textile fibres. The size range for these particles fits the common classification of MP as all common textile fibres (the building block of textile fabrics) are ≤100μm (more commonly 10-20 μm) in diameter. The fragments with any dimension less than 100nm would be categorised as Fragmented Nanofibres.

Textile debris comprises a majority of anthropogenic particles found in the environment (Wright and Kelly, 2017; Gago et al., 2018; Athey et al., 2020) and is considered to be one of the most pervasive and enduring pollutants globally (Singh et al., 2020). Natural fibres (processed from plants and animal fibres) and regenerated or semi-synthetic fibres (extruded from dissolved cellulose and derivates) (Stanton et al., 2019) are polymers, if not plastics, that act as environmental and human health hazards (Ladewig et al., 2015). Natural and extruded (synthetic and regenerated) fibre production involves dangerous processes that modify the materials with chemical treatments and additives (Lacasse and Baumann, 2004; Ladewig et al., 2015; Stanton et al., 2019; Athey et al., 2020), which may be released into the environment as they are not chemically bonded to the polymer matrix (Wright and Kelly, 2017).

Even though natural FF have been placed as dominant (93.8%) in freshwater and airborne samples, their environmental consequences are just beginning to be studied (Stanton et al., 2019). Cotton, one of the most popular fibres in clothing applications (Ellen MacArthur Foundation, 2017), has been reported to be ubiquitously found (De Falco et al., 2020). A recent study on indigo denim cotton FF found these cellulose fibres in remote regions sediments. This is an indicator of the long-lasting polymer duration, enough to reach long-range transportation and endanger the biota (Athey et al., 2020). Cotton biodegradation in aquatic environments has not been wholly established (Ladewig et al., 2015). While some studies suggest slow decomposition in hostile environment conditions (Chen and Jakes, 2001; Andrady, 2017), others hypothesise a similar degradation rate compared to aerobic degradation standards (Ladewig et al., 2015; Zambrano et al., 2019). In either case, this could have a negative environmental impact acting as a carrier for hazardous chemicals attached/embedded in the fibre. The ecological effects of FF, mainly focused on synthetics, are further discussed in section 6 of the document.

2. Current Methods for quantification and characterisation of FF from wet and dry routes

The reported methodologies to generate, filter, collect and measure FF in wet and dry states lack consistency owing to the absence of standardised methods. The methods employed in the published peer-reviewed studies have been critically evaluated to act as a guide for future work on the development of methodology and best practices for experimental work. This section describes the protocols proposed in publications focused mainly on FF released from laundry processes. It has been divided into six subsections covering the most critical aspects of the experimental laundry methodology. This includes textile sampling, washing devices, filtration processes, quantification, measurement, and estimated number of FF. Also, a final subsection is dedicated to studying FF collected in dry conditions.

2.1 Textile sampling

2.1.1 Sampling source

Textiles tested in published literature can be mainly categorised into two groups according to their source. Firstly, commercially available textiles, including garments (shirts, sweaters, jeans, and jackets), linen (sheets and blankets), and fabrics, acquired directly from the market or obtained through a textile company for performance analysis. Secondly, bespoke textile samples that have been manufactured by a research institution or a commercial textile company, following the specifics for examination. Nearly 80% of analysed literature employed commercial textile samples. It is pertinent to mention that limitations in this category include the lack of detailed material, history and textile structure properties (Browne et al., 2011; Folkö, 2015; Napper and Thompson, 2016; Pirc et al., 2016; Jönsson et al., 2018), insufficient information to provide a detailed technical description of textiles (Hartline et al., 2016; Sillanpää and Sainio, 2017; Volgare et al., 2021), direct comparison of samples regardless differences in textile properties such as woven and knitted textiles created with different yarn structures (Sillanpää and Sainio, 2017; De Falco, Gullo, et al., 2018; Belzagui et al., 2019; Cai et al., 2020), comparison of staple and filament yarns from different materials (Cesa et al., 2020), and evaluation of similar fabric and yarn structure from different fibre type (Zambrano et al., 2019). Some of the studies have provided a detailed description of textile structures and yarn specifications (Hernandez et al., 2017; Carney Almroth et al., 2018; De Falco, Gullo, et al., 2018; Yang et al., 2019; Cesa et al., 2020; De Falco et al., 2020; Cai et al., 2020).

The bespoke sampling category includes comparison of the same textile structure but different materials (Carney Almroth et al., 2018; Özkan and Gündoğdu, 2021; Palacios-Marín et al., 2022), or different yarn structures (Zambrano et al., 2019; Palacios-Marín et al., 2022). The limitations in this category may include an inaccurate representation of the actual manufacturing procedures employed in the textile industry and consumer applications. Other than these two main categories, some studies collect FF from fieldwork sampling obtained through dry and wet routes. The first may be done by collecting FF from atmospheric depositions (Dris et al., 2016; Stanton et al., 2019) or within indoor environments (De Falco et al., 2020; Q. Zhang et al., 2020). While the latter is acquired from reference sites (surface water and sediments) of aquatic ecosystems such as lakes and rivers, in addition to wastewater treatment plants (WWTP) effluent (Browne et al., 2011; Stanton et al., 2019; Athey et al., 2020), or consumed by aquatic fauna (Athey et al., 2020; Parton et al., 2020). For robust experimental studies and inter-study comparisons, there is a dire need to provide explicit material and structural properties and the processing history of samples. Furthermore, it is essential to test materials and structures which are representative of the global fibre mix and common applications.

2.1.2 Sampling size

Sampling sizes have been selected depending on the chosen washing equipment and the aims for individual research studies. The simulated laundry experiments work with reduced sample sizes that fit into the washing devices regardless of the source or material of the textile sample, (Hernandez et al., 2017; Carney Almroth et al., 2018; De Falco, Gullo, et al., 2018; Jönsson et al., 2018; Zambrano et al., 2019; Haap et al., 2019; Kelly et al., 2019; Cai et al., 2020; Özkan and Gündoğdu, 2021; Palacios-Marín et al., 2022). Since a standard procedure for this experiment has not been defined, some publications select the sampling size following commercial laundering or colour fastness standards with minor variations (Carney Almroth et al., 2018; De Falco, Gullo, et al., 2018; Zambrano et al., 2019; Haap et al., 2019; Cai et al., 2020). Correspondingly, most full-size washing machine testing use garments in their as-bought size (Browne et al., 2011; Folkö, 2015; Hartline et al., 2016; Pirc et al., 2016; Sillanpää and Sainio, 2017; Belzagui et al., 2019; Zambrano et al., 2019; Kelly et al., 2019; Fontana, 2020; Cesa et al., 2020; De Falco et al., 2020; Galvão et al., 2020; Athey et al., 2020; Volgare et al., 2021). However, results from different sizes may not always be comparable since they are presented in different units, as discussed further in section 3.1.1.

2.1.3 Sampling edges

The securing of sample edges is particularly important for simulated washing as samples are small and cut edges can significantly increase the amount of FF release. The process selected for cutting the sample edge impact the amount of FF released when laundering (Cai et al., 2020). Industrial techniques that resemble commercial garments manufacturing such as overlocker stitched, simple or double sewn, hemmed fabric, or a combination of these methods may be preferred (Napper and Thompson, 2016; Hernandez et al., 2017; De Falco, Gullo, et al., 2018; Zambrano et al., 2019; Yang et al., 2019). Furthermore, other approaches have been applied to minimise the FF release from the cut edges, include laser welding (Carney Almroth et al., 2018; Kelly et al., 2019), ultrasonic welding (Jönsson et al., 2018), and glue (Haap et al., 2019). Laser welding and ultrasonic welding are only limited to thermoplastic materials.

2.1.4 Textile material and structure

The textile structure can be studied in a hierarchical approach (Figure 4). On a macro level, the fabric is affected by its architecture and composition of subsequent levels, the yarn and fibre. The yarn is affected by fibre packing fraction, fibre orientation and fibre architecture, including inter-fibre friction. Finally, fibre is affected on a micro-level by persistent dislocating movements that provoke small fractures (Militky and Ibrahim, 2009). This understanding is essential to elucidate the effect of textile structure in the release of FF. Even though some authors have provided exhaustive information regarding textile structures tested (Cesa et al., 2020; De Falco et al., 2020; Cai et al., 2020; Özkan and Gündoğdu, 2021; Palacios-Marín et al., 2022), fibre and yarn level details are not always included (Browne et al., 2011; Folkö, 2015; Napper and Thompson, 2016; Pirc et al., 2016). It is vital to provide fibre, yarn, and fabric details (structure, chemical processing, finishing) to allow inter-study comparison of results.

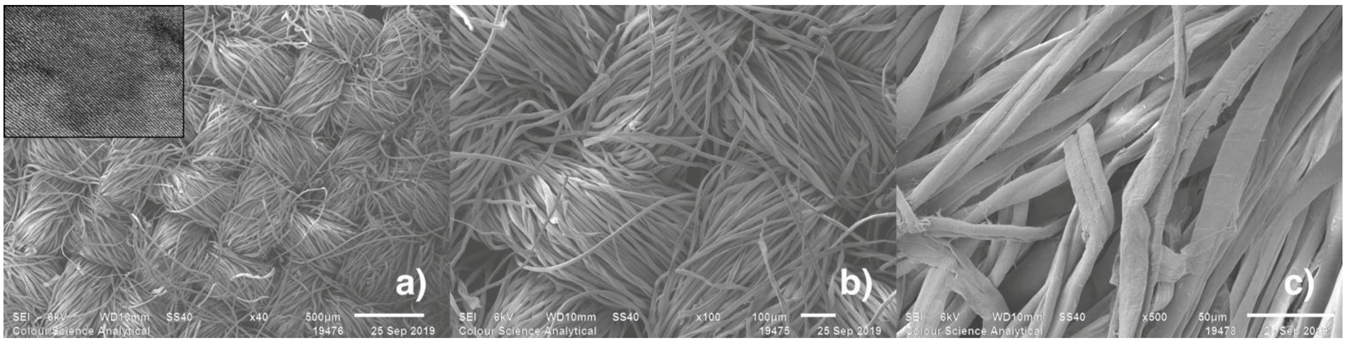


Figure 4 Hierarchical approach of textile structure. (Author).
a) Macro-level: Fabric constructed by the intertwining of yarns.
b) Meso-level: Yarns formed by fibre twisting.
c) Micro-level: individual fibres.

The majority of published literature reported Fourier-transform infrared spectroscopy (FTIR) along with a spectral database to identify the polymer material of recovered FF (Browne et al., 2011; Dris et al., 2016; Napper and Thompson, 2016; De Falco, Gullo, et al., 2018; Stanton et al., 2019; Yang et al., 2019; Fontana, 2020; Cesa et al., 2020; De Falco et al., 2020; Cai et al., 2020; Q. Zhang et al., 2020). Besides, scanning electron microscope (SEM) images are employed to differentiate between natural and manufactured fibres (Haap et al., 2019; Athey et al., 2020). Since the study of FF pollution is derived from the microplastic investigation, synthetic fibres are mainly studied. As a result, there is a paucity of studies on natural and regenerated textile materials (Napper and Thompson, 2016; Sillanpää and Sainio, 2017; Stanton et al., 2019; Zambrano et al., 2019; Yang et al., 2019; Cesa et al., 2020; De Falco et al., 2020; Galvão et al., 2020).

2.2 Washing equipment

Washing equipment employed in laundering experiments may be sorted into two groups: simulated washing devices and commercial laundry machines. The former equipment (Figure 5) is constructed to mimic household washing conditions (Carney Almroth et al., 2018; James Heal, 2021). The benefits of this method include full controlled and standardised washing conditions such as temperature, water volume, cycle duration, washing rate, and mechanical stress. In contrast, shortcomings involve limited sample sizing, unrealistic washing conditions, and the unfeasibility of running a complete washing cycle that includes rinsing.

Figure 5 GyroWash device, an example of a simulated washing machine.
Copyright 2021, James Heal. [Online image] Retrieved from https://www.james-heal.co.uk (Accessed 21 Feb. 2022).

The consumer laundry devices include front-load and top-load washing machines. The main differences between the equipment are the agitator's position (vertical for top-load and horizontal for front-load) and the water volume demanded. Even though both devices present similar procedures for the washing cycle, they are compared in published literature (Hartline et al., 2016). Some other devices such as Pulsator, Platen (Yang et al., 2019) and Vortex (Athey et al., 2020) washing machines are also studied. The advantages of this method include realistic washing conditions, full garments sampling and improved analysis of the laundry cycle followed by consumers. In contrast, disadvantages comprise a challenging filtration method due to the high effluent volume and uncertainty on the specific washing conditions. Only few literature correlates the results between laboratory-scale and consumer laundry devices regarding FF (De Falco, Gullo, et al., 2018).

2.3 Filtration devices and procedures

Most studies employ a filtration procedure to collect FF. There is a wide variety of filtration setups and techniques used within the published methodology. The selection of the filtration device depends upon the washing equipment and the water volume to be filtered. For full-scale laundry machines, entire effluent released after washing was filtered and studied (Browne et al., 2011; Napper and Thompson, 2016; Fontana, 2020), or collected FF are redispersed in beakers for further analysis (Zambrano et al., 2019). Another approach consists of collecting wastewater in barrels, and subsamples are filtered to analyse a representative portion of the effluent (Folkö, 2015; Hartline et al., 2016; Sillanpää and Sainio, 2017). Custom-built filtration assemblies have been proposed as an alternative to the FF collection (Pirc et al., 2016; Hernandez et al., 2017), for instance, a multi-step filtration device (De Falco et al., 2020; Volgare et al., 2021). Finally, reduced volume filtration samples, including most of the simulated washing experimental, and subsamples of commercial laundry effluent employ a vacuum pump filtration assembly consisting of a filtration holder with a glass collection beaker attached to a vacuum pump to accelerate the process

(Carney Almroth et al., 2018; Jönsson et al., 2018; Galvão et al., 2020; Özkan and Gündoğdu, 2021).

A filter or sieve is employed for FF collection, and in the reported studies, the filter pore size varies from 0.2µm (Haap et al., 2019) to 500µm (Cesa et al., 2020) in published literature. As discussed in section 1.3, the predominant FF diameter cross-section range is 10-20µm (Cesa et al., 2017). Therefore, fibres may longitudinally escape through the filter where pores are bigger than FF diameter (Yang et al., 2019), and filters may not collect the totality of released FF and underestimate the FF release. The filter includes paper filters (Browne et al., 2011), nylon mesh (Folkö, 2015; Hartline et al., 2016), stainless steel filter (Pirc et al., 2016), glass microfibre filter (Zambrano et al., 2019) and nitrocellulose membrane filters (Galvão et al., 2020). An alternative to filtration procedures is the analysis of particles while still in the effluent. Literature suggests the Dynamic Image Analysis (DIA) (Haap et al., 2019), nanoparticle tracking (Hernandez et al., 2019) and fibre quality analyser (Zambrano et al., 2019) tools. The use of filters has limitations as MP (and FF) are predominantly present in air and water, which can impact the results. The use of blank runs and established protocols is essential to achieve reliable estimates. This is particularly more important in using simulated washing equipment where only a small mass of the material is tested.

2.4 Characterisation techniques

2.4.1 Material identification

The material characterisation of recovered FF after filtration processes correlates these particles to their textile origin. Most of the literature that analyses FF after washing processes prefer infrared spectroscopy (IR) and infrared spectroscopy microscopy as leading techniques for MP and FF identification (PerkinElmer, 2021). Fourier Transform Infrared Spectroscopy (FT-IR) is a technique that obtains the infrared spectrum of emission or absorption of the material, and it is also helpful to detect material contaminants such as additives and organic or biological coatings adhered to the surface of the particles (Deena Titus et al., 2019; PerkinElmer, 2021). The results of the material infrared spectrum can be compared against spectral libraries or databases of polymers and textile polymers for identification (PerkinElmer, 2021).

FT-IR technique and micro FT-IR are widely used in literature to analyse FF collected after washing and contamination on samples (see (Browne et al., 2011; Napper and Thompson, 2016; Pirc et al., 2016; De Falco, Gullo, et al., 2018; Stanton et al., 2019; Yang et al., 2019; Hernandez et al., 2019; Fontana, 2020; Cesa et al., 2020; De Falco et al., 2020; Cai et al., 2020)). FF generated in washing procedures in laboratory-controlled conditions usually do not require any extra cleaning step before characterisation. However, those FF and MP collected in wastewater, sediments, or ingested by animals may need extensive cleaning processes for particle isolation (PerkinElmer, 2021).

FF recovered in filters after laundry can be analysed without removing them from the collection filter. Nevertheless, the filter specifications affect the IR analysis. For instance, the material can mask the absorption of the particle because it absorbs part of the spectrum as well. The size will decide the sample capacity and duration of the assessment, and the filter pore size will determine the particle size retained (PerkinElmer, 2021). Several filters are currently available for filtration and later IR analysis, including glass fibre filters, gold and silver coated filters, silicon filters, and others. The compatibility of the filter with the FT-IR analysis mode and the particle size can be consulted in (PerkinElmer, 2021), where an in-depth assessment of filter type was performed for optimum results of MP characterisation.

2.4.2 Quantification methods

The published literature has proposed different approaches to quantify the amount of released FF. The most common is a gravimetric approach, where an analytical balance or a microbalance is employed to measure the increase in filter mass. The results may be presented in simple units such as milligrams (Hartline et al., 2016; Napper and Thompson, 2016; Sillanpää and Sainio, 2017; Kelly et al., 2019; Cesa et al., 2020; Palacios-Marín et al., 2022), or they may be expressed as a percentage of the weight they represent over the initial sample weight (Pirc et al., 2016; Zambrano et al., 2019), per kilogram of square metre of textile (Yang et al., 2019), or litre of effluent (Folkö, 2015). In another approach, the mass was calculated with a mathematical formula that considers the number of released fibres, the fibre average diameter or volume, and the mean density of the polymer (Hernandez et al., 2017; De Falco, Gullo, et al., 2018; Yang et al., 2019; De Falco et al., 2020; Cai et al., 2020; Volgare et al., 2021) or average FF length and yarn count (Belzagui et al., 2019). The average measurements are obtained by running laboratory tests or published standard values. A different methodology calculates recovered FF mass by colour assessment. Effluent with known mass values of FF was subjected to image colour analysis to create a comparative scale. The colour measurement change depending on the fibre concentration. Then, the colour of the recovered laundry effluent is analysed to estimate the mass of fragmented fibres in the solution (Kelly et al., 2019).

2.5 Estimated number of FF released per laundry cycle

A high number of methods has been proposed within published literature to quantify the release of FF from dry and wet routes. The most representative systems are listed below.

1) Individual counting of entire recovered FF collected in the filter by image analysis (Dris et al., 2015; Dris et al., 2016; Dris et al., 2017; Hernandez et al., 2017; Cai et al., 2017; Jönsson et al., 2018; Stanton et al., 2019; Allen et al., 2019; Klein and Fischer, 2019; Cai et al., 2020; Athey et al., 2020; Nematollahi et al., 2022; Abbasi et al., 2022).
2) Quantify FF within a representative segment of the filter area to obtain an average result. The mean value is then multiplied by the total segment number for estimation of the entire FF released (Sillanpää and Sainio, 2017; Carney Almroth et al., 2018; De Falco, Gullo, et al., 2018; Abbasi et al., 2019; Yang et al., 2019; Q. Zhang et al., 2020; Galvão et al., 2020; Özkan and Gündoğdu, 2021). A variation of this method considers a subsample of the effluent and multiplies it by the total volume of the wastewater (Hernandez et al., 2019).
3) The estimated FF number is formulated by dividing the total volume of FF collected from shedding between the mean volume of one fragmented fibre (Napper and Thompson, 2016; Cesa et al., 2020).
4) The number of FF particles is calculated from the gravimetric results. Mass value is divided between yarn count and average FF length (Pirc et al., 2016; Belzagui et al., 2019; Kelly et al., 2019; De Falco et al., 2020; Volgare et al., 2021; Palacios-Marín et al., 2022).

5) Automatic image quantification of FF suspended in the effluent with support of a specialised image analysis device and software (Zambrano et al., 2019; Haap et al., 2019).

2.6 Length distribution profile

Length distribution profile may be obtained by following an automated (Jönsson et al., 2018), semi-automated (Cai et al., 2020), or manual approach (Hernandez et al., 2017). The three methods require imaging recovered FF on a microscope or an image analysis device for further assessment with graphic software. The difference between the methods is the FF tracking procedure. The manual approach involves manual counting of FF and drawing on top of the fibres, following their shape to analyse their length (Hernandez et al., 2017; Palacios-Marín, 2019; Palacios-Marín et al., 2022) (see Figure 6). While the semi-automated manner only requires the selection of the starting and endpoints of each fibre for automatic counting and length measurement (Cai et al., 2020). Finally, the automated analysis does not need manual intervention for fibre counting and fibre length distribution (Jönsson et al., 2018).

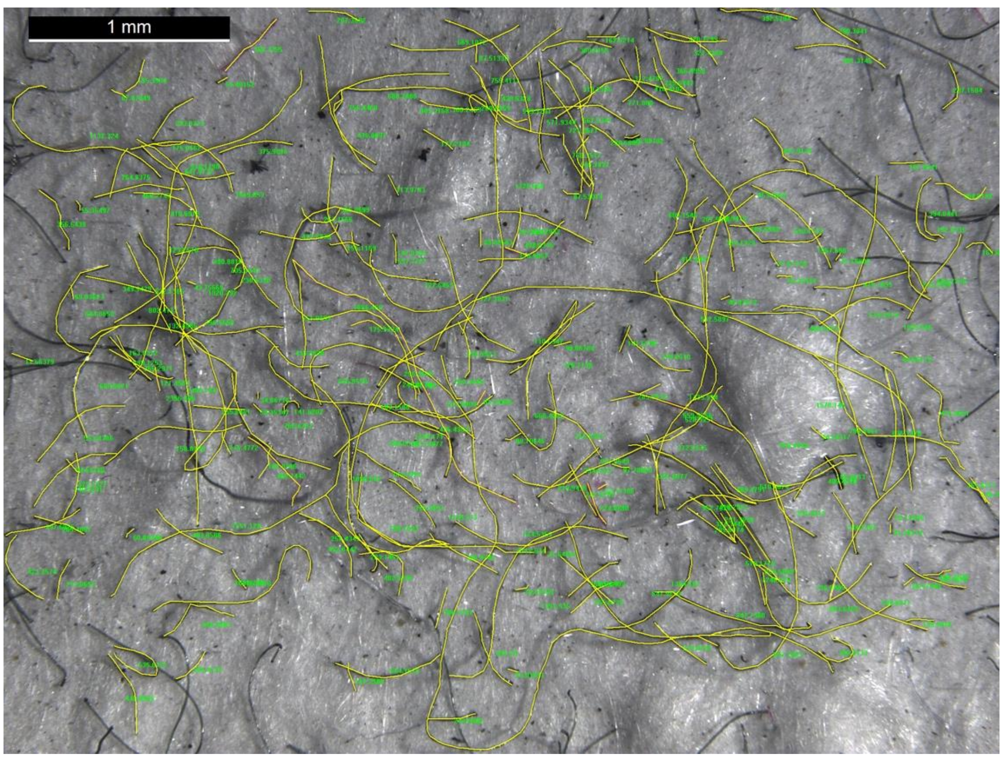


Figure 6 Image of recovered FF, drawn on top, for a manual approach tracking procedure. (Palacios-Marín, 2019).

Devices for these procedures include light microscope (Napper and Thompson, 2016; Carney Almroth et al., 2018; Yang et al., 2019), stereo microscope (Folkö, 2015; Galvão et al., 2020; Athey et al., 2020), digital cameras (Hartline et al., 2016; Cai et al., 2020), scanning electron microscope (De Falco, Gullo, et al., 2018; Hernandez et al., 2019; Fontana, 2020), and dynamic image analysis (Haap et al., 2019). Measurements are done with the support of graphic analytical software such as Image J, (Hartline et al., 2016; Napper and Thompson, 2016; Hernandez et al., 2017; De Falco, Gullo, et al., 2018; Haap et al., 2019; Hernandez et al., 2019; Fontana, 2020;

Özkan and Gündoğdu, 2021). Either totality of FF released is measured (Hernandez et al., 2017), or an average is calculated after analysing a representative sample (De Falco, Gullo, et al., 2018; Belzagui et al., 2019; Cesa et al., 2020).

2.7 Methods for FF collection in dry conditions

Fewer studies to investigate the generation and collection of FF through dry routes are conducted. The MP (and FF) studies can be classified into two groups. The collection of MP/FF from atmospheric fallout in open places (Dris et al., 2015; Dris et al., 2016; Dris et al., 2017; Cai et al., 2017; Abbasi et al., 2019; Allen et al., 2019; Stanton et al., 2019; Liu et al., 2019; Klein and Fischer, 2019; Abbasi et al., 2022) and studies of MP/FF from indoor environments (Dris et al., 2017; Q. Zhang et al., 2020; Nematollahi et al., 2022). Most of the mentioned studies comprise an extensive investigation of MP in any shape, such as spheres and films, and they include the analysis of fibres (Dris et al., 2015; Cai et al., 2017; Abbasi et al., 2019; Allen et al., 2019; Liu et al., 2019; Klein and Fischer, 2019; Nematollahi et al., 2022; Abbasi et al., 2022). A minor number of studies are focused merely on FF from any material in atmospheric or indoor deposition (Dris et al., 2016; Dris et al., 2017; Stanton et al., 2019).

Both groups present different methodologies to capture particles. The most common of them is the air deposition of debris in containers. For instance, the use of a funnel attached to a container installed in a selected open site to collect fallout depositions for a set time duration. Then, the funnel is rinsed with distilled water, and the effluent is collected and filtered for further analysis (Dris et al., 2015; Dris et al., 2016; Cai et al., 2017; Allen et al., 2019; Stanton et al., 2019; Klein and Fischer, 2019). A variation of this method for indoor spaces was presented by (Q. Zhang et al., 2020), using a stainless steel basin to collect FF for further filtration and analysis. Another employed methodologies comprise the retrieval of settled dust particles from vehicles (Abbasi et al., 2022) or horizontal flat surfaces in classrooms (Nematollahi et al., 2022), the direct filtration of air assisted by a device, such as an air-pump (Dris et al., 2017), an ambient filter sampler (Abbasi et al., 2019), and a total suspended particle sampler (Liu et al., 2019). The height of the air filtration device employed to collect MP/FF will affect the type and amount of collected particles (Liu et al., 2019) and are recorded in each investigation.

In addition, (De Falco et al., 2020) conducted experimental research to generate and collect FF from the wear and tear of garments. Volunteers wearing sample garments performed selected movements for 20 minutes in a closed and cleaned room. The floor was covered with cardboard, and Petri dishes with dampened filters were placed in the room to collect released FF. Table 4 present a more detailed comparison of MP/FF indoor and outdoor studies and their results.

Image analysis was performed as part of the studies with the support of a stereomicroscope (Dris et al., 2015; Dris et al., 2016; Dris et al., 2017; Allen et al., 2019; Stanton et al., 2019; Liu et al., 2019; Q. Zhang et al., 2020), a light microscope (Abbasi et al., 2019; De Falco et al., 2020; Nematollahi et al., 2022), a fluorescence microscope (Abbasi et al., 2019; Klein and Fischer, 2019), or a scanning electron microscope (Nematollahi et al., 2022; Abbasi et al., 2022). Finally, particle identification was performed with μ-FTIR (Dris et al., 2016; Dris et al., 2017; Cai et al., 2017; Stanton et al., 2019; Liu et al., 2019; De Falco et al., 2020; Q. Zhang et al., 2020) or μ-

Raman spectroscopy (Allen et al., 2019; Klein and Fischer, 2019; Nematollahi et al., 2022; Abbasi et al., 2022). Fibre quantification followed the described methodology in sections 2.3 to 2.6.

To further understand the role of microplastics and fragmented fibres as a source of atmospheric contamination and their potential impact on the environment and human health, please consult (Dris et al., 2016; Gasperi et al., 2018; Y. Zhang et al., 2020; Q. Zhang et al., 2020; Akanyange et al., 2021; Abbasi et al., 2022).

3. Critical analysis of findings from quantification and characterisation of FF

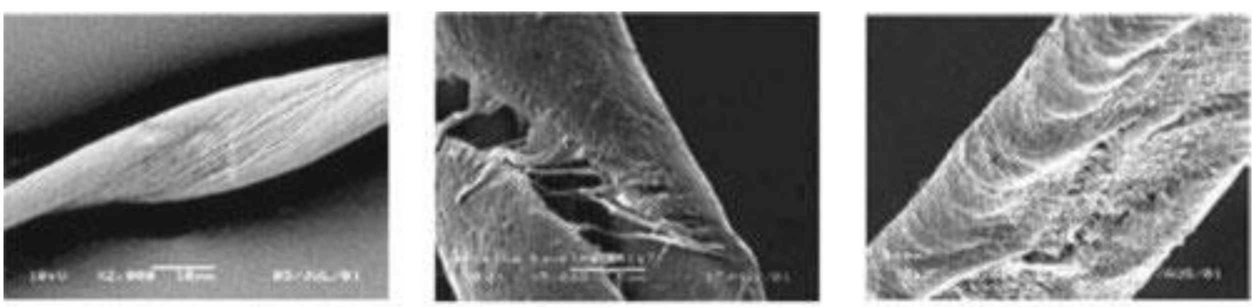
Microplastic pollution into aquatic environments from textiles has been widely reported in published literature. A critical review of existing experimental studies has been included in this section. Additionally, FF release to the atmosphere after clothing wearing is also considered, which apparently is underestimated due to the small number of published studies.

3.1 Release of FF from wet routes (textile laundering)

Regardless of the differences in methodology used in publications, the release of FF in washing effluent is well established. A study reports a correlation between FF in sewage effluent from WWTPs and those found in sediments, indicating laundry processes as a pathway for FF to reach the environment (Browne et al., 2011). In the absence of a standard approach to quantify MP (and FF), understandably, there are variations in the obtained results, but there are specific common findings. Generally, a higher release of FF is reported in the first washing cycle (Hernandez et al., 2017; Sillanpää and Sainio, 2017; Carney Almroth et al., 2018; Kelly et al., 2019; Özkan and Gündoğdu, 2021; Palacios-Marín et al., 2022) or a significant proportion of FF are collected up to the third washing cycle (Folkö, 2015; Cesa et al., 2020). These findings are associated to FF generated due to existing fibre damage during manufacturing processes (Belzagui et al., 2019) and present in the garments before washing (Cai et al., 2020). Besides a decrease in FF shedding for subsequent cycles until results reach a plateau was reported (Folkö, 2015; Napper and Thompson, 2016; Pirc et al., 2016; Belzagui et al., 2019; Zambrano et al., 2019; Cesa et al., 2020; Cai et al., 2020; Özkan and Gündoğdu, 2021). On the other hand, some studies report no significant difference in FF collected over repeated washing cycles (Hartline et al., 2016; Hernandez et al., 2017; Fontana, 2020). This trend could be associated with a pre-washing process (Hernandez et al., 2017; Fontana, 2020), where existing FF (presumably from manufacturing) could already have been removed. The introduction of a pre-wash or cleaning process before washing cycles decreases FF shedding (Napper and Thompson, 2016; Zambrano et al., 2019; Özkan and Gündoğdu, 2021). This is further discussed in section 4.2.1.

The published literature correlates FF shedding to several variables, including physicochemical properties of the material, fibre, yarn and fabric structure (Hernandez et al., 2017), and mechanical and chemical stresses created during washing (De Falco, Gullo, et al., 2018). The higher increase reported in the first wash may be associated with the mechanical and chemical stress that fibres carry through manufacturing processes (Hernandez et al., 2017; Belzagui et al., 2019; Cai et al., 2020). Fibres and yarns are exposed to fibre-fibre and fibre-metal friction when manufactured (Militky and Ibrahim, 2009; Özkan and Gündoğdu, 2021). A single fibre may experience over 10 million contacts with metallic parts during yarn spinning (Elmogahzy, 2008) and get damaged. Figure 7 illustrates an example of fibre damage after rotor spinning.

Fibre damage is produced as a result of the loading and unloading effect during the circulation flow of the yarn spinning (Militky and Ibrahim, 2009), which generate short fibres that will be embedded in the yarn during spinning and released through laundry processes (Hernandez et al., 2017). The short fibres move through their neighbouring fibres by fibre migration, which is promoted by the high tensioned forces and a cyclic interchange of position that fibres complete through the twist insertion process during yarn manufacturing (Klein, 1987; Tyagi, 2010; Özkan and Gündoğdu, 2021). Fibre migration is affected by the spinning method selected to manufacture the yarns (Tyagi, 2010). Generally, short, coarse, and stiffer fibres move from the core to the surface of the yarn, while long, fine, and flexible fibres move from the surface to the core (Tyagi, 2010; Özkan and Gündoğdu, 2021). The different migration behaviour between short and long fibres may explain the washing results reported in the literature. For instance, the higher release of FF from the first washing cycle in comparison with the rest of the cycles reported by (Hernandez et al., 2017; Sillanpää and Sainio, 2017; Carney Almroth et al., 2018; Kelly et al., 2019; Özkan and Gündoğdu, 2021; Palacios-Marín et al., 2022), may correspond to the detachment of the shortest surface fibres (Cai et al., 2020). While the steady discharge results reported in subsequent washing cycles (Folkö, 2015; Napper and Thompson, 2016; Pirc et al., 2016; Belzagui et al., 2019; Zambrano et al., 2019; Cesa et al., 2020; Cai et al., 2020; Özkan and Gündoğdu, 2021) can be a consequence of the slow-release dynamic of the longer fibres that are immersed in the core of the yarn. It is essential to consider that mass from initial washing cycles may include the remains of chemical substances used for manufacturing processes that involve dyes or finishes (Cesa et al., 2020) and may not be eliminated even with a pre-wash process.



(a) (b) (c)

Figure 7 Image damage after rotor spinning.
SEM images: a) raw cotton, b) crack due to rotor spinning c) folds due to rotor spinning
From Effect of textile processing on fatigue (p. 152), by Militky, J. and Ibrahim, S., 2009. In: M. Miraftab, ed. Fatigue failure of textiles fibres. Technical University of LIBEREC, Czech Republic: Woodhead Publishing Limited.
DOI: 10.1533/9781845695729.2.133.

In addition to the release of FF accelerated due to the manufacturing process, laundry procedures can generate new fragmented fibres. In a sequence of washing cycles, the results reach a steady point of release, or slightly increase for the last cycles (Cai et al., 2020), confirming that no fixed amount of FF is shed per garment (Hartline et al., 2016; Carney Almroth et al., 2018). This may be explained by fatigue failure of fibres. When laundry is carried out, fibres are subjected to a combination of tensile, bending and compression forces. The yield of fibre under washing forces arises earlier than the yield in tension when textiles are manufactured (Elmogahzy, 2008), causing fibre rupture. The imaging of the fibres displays fibre fractures and damage after laundering or soaking in hot water (Hernandez et al., 2019; Fontana, 2020).

Different theories for this phenomenon are proposed since FF generation has not been explicitly studied. For instance, pilling is suggested as a mechanism for FF formation (Napper and Thompson, 2016; De Falco, Gullo, et al., 2018; Zambrano et al., 2019). Pilling is defined as the creation of fibre balls (pills) by entangling of protruding fibres from the surface of the fabric and persistence of them (Napper and Thompson, 2016). FF shedding is associated with the ease of fibre break and debris release at the initial stage of the pill formation (Zambrano et al., 2019). Although some FF could be originated by the proposed mechanism, no strong rationale support pilling as a single cause. Since pilling requires protruding fibre ends in the fabric surface, its formation would be prevented by the lack of hairiness of continuous filament fibres. The long filaments' resistance to abrasion and break avoid fibre migration and pill formation (Ukponmwan et al., 1998; Hunter, 2009). Additionally, a high number of FF released after laundry is reported even if no pilling formation is observed (De Falco, Gullo, et al., 2018; Cesa et al., 2020).

3.1.1 Estimated number of FF

Table 2 summarises the number of FF shed from a 6-kilogram laundry load. Differences in results among authors are considerable. For instance, the amount of FF released from a 500gr PET t-shirt could be 11,300 (Pirc et al., 2016), 83,719 (Özkan and Gündoğdu, 2021), 600,000 (De Falco, Gullo, et al., 2018), or 36,000,000 (Cai et al., 2020). As shown (Table 2), results from different authors are not reported in the same units. Therefore, an approach was developed to calculate estimates in comparable units. Given that a standard/harmonised methodology to test the release of FF does not exist, the results can have inherent differences due to the varying parameters of the testing itself. Sixty litres of water were considered as average water-volume consumption per washing load. For studies where FF was reported per unit area (Browne et al., 2011; Carney Almroth et al., 2018; Yang et al., 2019; Athey et al., 2020), the number of FF were estimated following assumptions by Carney Almroth et al., 2018 (1 square metre = one garment) and calculated accounting 1Kg of fabric per square metre. The FF quantification method from reviewed literature was categorised into three different approaches. *The indirect approach* refers to estimating the FF number obtained by dividing the total volume or mass result of collected FF between single fibre volume or mass values. An insight into the methodology is described in numbers 3 and 4 of subsection 2.6. *The scaling approach* indicates that FF from a representative subsample (area or weight) of the collected FF are individually counted and then multiplied by the total number of subsamples comprised within the FF collected sample (see number 2 from subsection 2.6). Finally, if the methodology includes the counting of all particles recovered, it is indicated to follow a *manual*, *semiautomatic,* or *automatic* approach (see numbers 1 and 5 of subsection 2.5 and subsection 2.6 for description).

Results displayed in Table 2 for synthetic and cellulosic fibres show a wide range of FF per load, from 11,400 (Browne et al., 2011) to 432,000,000 (Cai et al., 2020). Such a wide range of variation can be attributed to the differences in filter pore size used (from 0.2μm up to 200μm) and material and textile properties. There is a correlation between filter pore size and the number of FF shed in some cases. The lowest values in Table 2a correspond to the broader pore size (Pirc et al., 2016) and vice versa (Cai et al., 2020). This trend is more evident when cotton fibres results are analysed (Table 2b). However, there are also exceptions. For example, (Özkan and Gündoğdu, 2021) employed the same pore size as (Cai et al., 2020) and did not present similar results. However, this may be explained by the differences in textile properties of samples, pre-wash treatment, use of detergent and washing conditions employed in the two studies. The quantification method selected by the authors does not seem to impact the results or create a recognisable trend. Hence differences in quantification methodology may not influence particle counting. Fibre (polymer) type and yarn structure directly affects FF shedding and are discussed in section 4. The direct comparison of different materials (whether synthetic or not) and fibre length (staple or filament) may result in misleading generic outcomes.

Table 2 Estimated Number of FF released in a 6 kg laundry load.

a) Extruded fragmented fibres reported in current literature.

Publication	Fibre	Filter Pore Size	Estimated No. FF	Unit	Quantification Approach	Estimated No. in 6 Kg load
Browne et al., 2011	PET	Not given	Max - 1,900	Single garment	None specified	11,400
Pirc et al., 2016	PET	200 μm	11,300	500 g textile	Indirect	135,600
Yang et al., 2019	Acetate	5 μm	74,816	FF / 1 m²	Scaling	448,896
Napper and Thompson, 2016	PET	25 μm	496,030	6 kg load	Indirect	496,030
	Acrylic		728,789			728,789
Sillanpää and Sainio, 2017	PET	0.7 μm	223,000	Wash effluent (2.6 kg load)	Scaling	514,615
Carney Almroth et al., 2018	PET	1.2 μm	110,000	Single garment (1 m²)	Scaling	660,000
Belzagui et al., 2019	PET	20 μm	Min – 1,000,000	6 kg load	Indirect	1,000,000
			Max – 6,500,000			6,500,000
Özkan and Gündoğdu, 2020	R-PET	0.45 μm	368,094	FF / Kilogram textile	Scaling	2,208,564
	PET		167,437			1,004,622
De Falco, Gullo, et al., 2018	PET	5 μm	Liquid – 6,000,000	5 kg load	Scaling	7,200,000
			17,000,000 (PD)			20,400,000
Palacios-Marín et al., 2022	PET	1.6 μm	Flat Filament - 10,900,000	6 kg load	Indirect	10,900,000
			Textured Filament - 2,640,000			2,640,000
			Staple - 12,600,000			12,600,000
Galvão et al., 2020	Synthetic	12 μm	18,000,000	6 kg load	Scaling	18,000,000
Volgare et al., 2021	PET	5 μm	0.15kg WL - 4,766,338	N/kg	Scaling	28,598,028
			2.5kg WL - 1,041,736			6,250,416
Cai et al., 2020	PET	0.45 μm	Laser cut – 210	FF / Gram textile	Semiautomatic count of total FF	1,260,000
			Scissor cut – 72,000			432,000,000

b) Cotton and cotton blend fragmented fibres reported in current literature.

Publication	Fibre	Filter Pore Size	Estimated No. FF	Unit	Quantification Approach	Estimated No. in 6 Kg load
Napper and Thompson, 2016	65% PET 35% Cotton	25 μm	137,951	6 kg load	Indirect	137,951
Athey et al., 2020	Cotton	10 μm	56,000	Single garment (denim)	Manually count of total FF	336,000
Haap et al., 2019	50% PET	0.2 μm	51	FF / Gram textile	Automatic count of total FF	306,000
	50% Cotton		317			1,902,000
Sillanpää and Sainio, 2017	Cotton	0.7 μm	973,000	Wash effluent (2.6 kg load)	Scaling	2,245,385
Palacios-Marín et al., 2022	52% PET 48% Cotton	1.6 μm	20,500,000	6 kg load	Indirect	20,500,000
	100% Cotton		24,700,000			24,700,000

Table 3 Average length and diameter of FF reported in current literature.

Publication	Fibre		Filter Pore Size	Average Length (μm)	Average Diameter (μm)	Analysis Method
Pirc et al., 2016	PET		200 μm	5,300	- - -	Image Processing
Napper and Thompson, 2016	PET		25 μm	7,790	12	Image Processing with Image J
	Acrylic			5,440	14	
	Poly-Cotton			4,990	18	
Hernandez et al., 2017	PET		0.45 μm	< 1000	- - -	Image Processing with Image J
Sillanpää and Sainio, 2017	PET		0.7 μm	100-1000	10-20	Image Processing
De Falco, Gullo, et al., 2018	Weave PET		5 μm	340	14	Image Processing with Image J
	Knit PET			478	20	
	Polypropylene			339	19	
Yang et al., 2019	PET		5 μm	499	13	Image Processing
	Polyamide			1056	17	
	Acetate			1128	15	
Palacios-Marín et al., 2022	Flat PET		1.6 μm	550	- - -	Image Processing with Image-Pro 7
	Textured PET			550		
	Staple PET			515		
	Poly-Cotton Blend			673		
	Cotton			379		
Galvão et al., 2020	Synthetic		12 μm	170	- - -	
De Falco et al., 2020	PET Knit Filament		5 μm	610	16	Image Processing with Image J
	PET Weave Fil			760	15	
	PET Knit Staple			796	16	
	Knit	50% PET		420	15	
		50% Cotton		889	19	
Fontana, 2020	PET		40 μm	> 200	- - -	Image Processing with Image J
Cai et al., 2020	PET		0.45 μm	100-1000	- - -	Image Processing with FiberApp
Özkan and Gündoğdu, 2020	Recycled PET		0.45 μm	1,320	- - -	Image Processing with Image J
	PET			1,590		
Volgare et al., 2021	PET		5 μm	549	11.7	Image Processing with Image J

3.1.2 Length distribution profile

The reported results for average FF length and diameter are collated in

Table 3. In most cases, the FF length is reported less than 1000μm (Hernandez et al., 2017; Sillanpää and Sainio, 2017; De Falco, Gullo, et al., 2018; Fontana, 2020; De Falco et al., 2020; Cai et al., 2020; Galvão et al., 2020). Few studies have reported FF between 1000 and 1500μm (Yang et al., 2019; Özkan and Gündoğdu, 2021). Furthermore, two published studies, (Napper and Thompson, 2016; Pirc et al., 2016), have included FF lengths near or above FF upper size limit (5mm). In the case of (Napper and Thompson, 2016), the mean length was obtained from a limited portion of the population since only 10 FF were measured and averaged per fabric type, which represents only 0.001 – 0.002% of ~500,000-700,000 fibres reported. The length distribution from (Pirc et al., 2016) was based on a much bigger filter pore size (200μm) as compared to the usual fibre diameter (10-20μm). This means that the filter cannot capture fibres of smaller length dimensions and longer fibres than the pore size may scape longitudinally. Moreover, an inaccurate fibre length measurement owing to intertwined fibres, which could not be separated, is reported within the results (Pirc et al., 2016).

The filter pore size impacts the fibre length distribution in different ways. For instance, studies using filters with a broader pore size than the average fibre diameter may report a smaller fibre mean length than those employing narrower pore size filters. As an example, it is possible to compare the research from (Fontana, 2020), using a 40μm pore size filter and reporting a FF mean length of ~200μm, with the results from (Özkan and Gündoğdu, 2021) employing a 0.45μm pore size filter and obtaining a FF mean length of 1,590μm. Both studies used the same software for image analysis and fibre material. However, it is known that during filtration, the materials gather to form a cake layer on the filter surface which effectively decreases the mean pore size and results in the filtration of FF smaller than the original pores. Nevertheless, large and short fibres may scape longitudinally from the filter in the case of (Fontana, 2020), which will directly impact the fibre length distribution and fibre quantification. As shown in

Table 3, most studies employed image processing software (commonly Image J) for image analysis. In consistency with previous sections, the lack of a standard approach to test, quantify and measure, and the detailed data of tested sample in terms of employed materials, structure and processing, undermines any comparisons.

Some studies have reported contradictory results after studying the effect of textile ageing on the release of FF and its impact on the length distribution of the released FF. One study reports an increase in length related to washing cycle number is documented (Folkö, 2015; Hartline et al., 2016; Cai et al., 2020). The smaller FF are released during the first cycles, while longer particles have a slower detachment rate. This may be explained by a more entangled position of fibres in the yarn (Cai et al., 2020) (in agreement with the fibre migration mechanism described by (Tyagi, 2010; Özkan and Gündoğdu, 2021)) or by the time needed for fatigue failure to occur and new FF are generated and detached (Folkö, 2015). In contrast, the decrease in the length of aged textiles has also been reported. A reduction of size is related to the washing cycle number (Belzagui et al., 2019), and items previously used by household residents generated more and shorter FF than new ones (Galvão et al., 2020). Finally, no significant differences between length distribution and wash number are reported by (Fontana, 2020). The difference in washing machine type is also documented to affect the length of collected FF. Longer FF released from a front-load washing machine compared to a top-load laundering device were found (Hartline et al., 2016). The household laundry machines were reported to release longer fibre debris compared with lab-scale accelerated laundering devices (Zambrano et al., 2019). However, this could be affected by the additional filtration step carried out in household washing devices (effluent first collected on a 20μm pore size filter) before filtration with a 1.2μm pore size filter, that it is not employed in the accelerated laundering devices filtration process (Zambrano et al., 2019). In contrast, no significant effects on FF length from the same textile structure (acetate, polyamide and polyester) under different washing conditions, including temperature (30ºC, 40ºC and 60ºC), additives (presence or absence of laundry detergent) and laundry machine type (platen or pulsator) were found by (Yang et al., 2019).

Another critical aspect that may affect FF length distribution is textile material properties. First, the fibre morphology as either discrete (staple) or endless (continuous filament) might influence the length of released FF. Longer debris are found after filament samples compared to staple fibres of the same material (Özkan and Gündoğdu, 2021; Palacios-Marín et al., 2022). Additionally, fibre polymer type has a significant impact on FF length. For instance, recycled PET particles break down into shorter debris as compared to PET (Özkan and Gündoğdu, 2021). Moreover, cotton FF are found to be smaller than synthetic fibres (Napper and Thompson, 2016; Napper and Thompson, 2016; Cesa et al., 2020; Galvão et al., 2020; Palacios-Marín et al., 2022). Conversely, (De Falco et al., 2020) described cotton FF longer than PET FF collected after washing a polycotton blend sample. Furthermore, length differences related to the cutting method employed for sample preparation were also reported. Laser-cut showed the shortest FF, and scissors cut the longest (Cai et al., 2020). Finally, no effect on fibre length regarding the level of mechanical stress created by the absence of the addition of 10 and 20 steel balls during laundry is reported (Cai et al., 2020).

3.2 Release of FF from dry routes.

The collection/deposition of MP and FF in dry conditions is a relatively less explored area. This includes the indoor (Dris et al., 2017; Q. Zhang et al., 2020; Nematollahi et al., 2022) and outdoor (Dris et al., 2015; Dris et al., 2016; Dris et al., 2017; Cai et al., 2017; Abbasi et al., 2019; Allen et al., 2019; Stanton et al., 2019; Liu et al., 2019; Klein and Fischer, 2019; Abbasi et al., 2022) environments and wear and tear of garments (De Falco et al., 2020) in dry conditions. All publications found a large amount of collected MP/FF (Table 4). The results show a strong correlation between human presence and particle debris deposition, as it is the case of population density in outdoor environments (Dris et al., 2016; Stanton et al., 2019; Liu et al., 2019) or human activities performed in indoor spaces (Abbasi et al., 2019; Allen et al., 2019; Klein and Fischer, 2019; Q. Zhang et al., 2020). This agrees with previous research on aquatic environments (Browne et al., 2011). Fibre concentration was higher in indoor than outdoor environments as well (Dris et al., 2017). On the contrary, (Klein and Fischer, 2019) reports a higher concentration of particles in rural than urban environments collected after atmospheric deposition. The results are attributed to the comb-out effect of plants (the ability of particle filtration) with solid particles from dry atmospheric deposition. The particles will be retained by plants and trees leaves and washed out after precipitation or wet atmospheric deposition (Klein and Fischer, 2019).

Fragmented fibres are found to be the most common pollutant in the literature studying a broad definition of MP (Cai et al., 2017). Fibres exceed 90% of debris population in studies by (Dris et al., 2015; Nematollahi et al., 2022; Abbasi et al., 2022), or near 70% in other publications (Allen et al., 2019; Liu et al., 2019). The environment plays a decisive role in the shape of the micro-debris collected. For instance, spherules and films MP were reported as the majority in industrial dust with a minimum presence of FF (6%). However, the same study reports over 30% fibres dominance in urban dust (Abbasi et al., 2019). Only FF were collected in atmospheric suspension, while the rest of MP shapes were found in industrial and city dust (Abbasi et al., 2019). Research from (Abbasi et al., 2022) showed a recovery of over 90% of FF, with small contributions from films and fragments, but no spherules were detected on samples recovered after a dust storm in Iran. Population density, industrial units and workshops, geographical environment, and dominant wind direction are decisive factors affecting the MP/FF concentration in indoor environments (Nematollahi et al., 2022). Nevertheless, applied methodology affects the results on debris collection, as it has already been reviewed for FF release/collection through wet routes.

Regarding the polymer type of FF collected from both indoor and outdoor environments, a prevalence of cellulosic fibres was reported (Dris et al., 2016; Stanton et al., 2019; Q. Zhang et al., 2020). Over 90% of the cellulosic FF population is described by (Stanton et al., 2019), while nearly 30% of synthetic FF proportion appears in other publications (Dris et al., 2016; Dris et al., 2017; Cai et al., 2017). The release of cotton fibres from a 50/50 poly-cotton blend sample was documented as higher than PET (De Falco et al., 2020). These findings are in agreement with higher proportions of natural fibres collected in laundry experiments (Zambrano et al., 2019; Haap et al., 2019; Galvão et al., 2020). Deeper insight is needed to analyse the reason for higher natural and cellulosic FF pollution over synthetic materials. Studies investigating solely synthetic FF, report Nylon, PET and polypropylene (PP) as the most abundant materials (Nematollahi et al., 2022; Abbasi et al., 2022). The average amount of particles reported in reviewed literature is displayed in Table 4. The variation in the reported results is not as large as in the case of FF results during laundry (Table 2). This may be attributed to the similarity of filter pore size employed in these studies. However, differences in collection and filtration procedures and material and size criteria are likely to affect the comparison of results.

The MP/FF particles collected in the analysed literature present a similar size distribution. The majority of particles are described to have a length under 1000μm (Dris et al., 2015; Dris et al., 2017; Nematollahi et al., 2022), under 600μm (Dris et al., 2016; Klein and Fischer, 2019), under 100μm (Abbasi et al., 2019; Abbasi et al., 2022), and under 50μm (Allen et al., 2019). Particle length is affected by airflow influence on MP/FF resuspension movements (Q. Zhang et al., 2020). The elevated wind periods (wind events >2ms–1) result in a higher concentration of longer MP (Allen et al., 2019). In addition, the length and mass of the particles define their spatial occurrence. As a result, the longest FF are reported in dust deposition, medium size FF were collected in indoor air suspension while the shortest are described in outdoor atmospheric suspension (Dris et al., 2017). This is in agreement with findings by (Liu et al., 2019), where particles collected at 1.7m height are longer than those collected at 33m height. The same study reports the longest particles collected at 80m height; however, they correspond to materials with lower density than particles gathered at lower altitudes.

The impact of textiles properties such as fibre type (filament or staple), fabric type (woven or knitted), and material (polyester and polycotton blend) on the release of FF can be analysed from the findings of (De Falco et al., 2020) since the source of FF generation can be tracked. Weave design and continuous filament form were found to decrease the amount of FF release. However, knitted samples and staple fibres released a higher amount of FF (De Falco et al., 2020). This can be attributed to lower twist levels for knitted yarns and staple fibres resulting in more fibre ends. In addition, fibres released into the air are longer than those collected from washing cycles (De Falco et al., 2020). This could be explained because textiles, when subjected to laundering processes, experience large hydrodynamic forces that can help release the smallest fibres contained in the core of the yarn (Kelly et al., 2019), while this does not happen in dry conditions.

The levels of anthropogenic fibrous pollution generated from textiles and their prevalence in the environment are of concern. Contrary to FF generated through textile washing and collected in WWTP, FF created after wear and tear or in dry processes does not follow an established effluent pathway that allows, even if partially, a collection process (Q. Zhang et al., 2020). The atmospheric fallout may act as a transport mechanism for FF to reach aquatic environments, agronomic systems and even the most remote locations (Dris et al., 2016; Stanton et al., 2019; Q. Zhang et al., 2020). The research carried out by (Q. Zhang et al., 2020) on indoor spaces shows how airflow affects the fluctuation, migration and resuspension of particles. Conclusions are similar for outdoor environments where air acts as a vector for FF dispersion and transport owing to their low settling velocities before dry or wet deposition (Abbasi et al., 2022). The release of FF during textile use and its presence in aquatic and terrestrial environments and their migration between ecosystems is well established. Still, the dispersion pathways between the sources and sinks are not fully understood.

Table 4 Characteristics of indoor and outdoor MP/FF deposition from previous studies.

Publication	Particle type*	Study Area	Collection method	Filter pore size	Abundance (average)
Dris et al., 2015	Microplastic	Atmospheric	Funnel + container	1.6μm	118 particles/m²/day
Dris et al., 2016	Synthetic FF	Atmospheric	Funnel + container	1.6μm	110 fibres/m²/day (urban) 53 fibres/m²/day (sub-urban)
Dris et al., 2017	FF	Atmospheric / indoor	Air-pump filtration device	1.6μm	5.4 fibres/m³ (indoor) 0.9 fibres/m³ (outdoor)
Cai et al., 2017	Microplastic	Atmospheric	Funnel + container	1.0μm	228 particles/m²/day (MP + FF) 36 particles/m²/day (MP + synthetic FF)
Abbasi et al., 2019	Microplastic / Microrubbers	Atmospheric	Ambient filter sampler	2.0μm	1 particle/m³
Allen et al., 2019	Microplastic	Atmospheric	Funnel + container	0.45μm	365 particles/m²/day (MP) 36 fibres/m²/day (Synthetic FF)
Klein and Fischer, 2019	Microplastic	Atmospheric	Funnel + container	5-13μm	215 particles/m²/day (urban) 395 particles/m²/day (rural)
Liu et al., 2019	Microplastic	Atmospheric	Total suspended particle sampler	1.6μm	1.42 particle/m³
Stanton et al., 2019	FF	Atmospheric	Funnel + container	38μm sieve / 0.45μm filter	73 fibres/m²/day (Natural) 1.7 fibres/m²/day (Extruded)
De Falco et al., 2020	Polyester and PET/cotton FF	Indoor generated FF	Petri dishes with dampened filter	N/A	1-347 FF/gram fabric (Polyester) 403 FF/gram fabric (Polycotton)
Zhang et al., 2020	FF	Indoor	Stainless steel basins.	5μm	19,300 fibres/m²/day (FF) 7,117 fibres/m²/day (Synthetic FF)
(Abbasi et al., 2022)	Microplastic	Atmospheric	Retrieval of settled dust particles from vehicles.	2μm	0.44 MP/gram of dust
(Nematollahi et al., 2022)	Microplastic	Indoor	Retrieval of settled dust particles from flat surfaces.	2μm	195.1 MP/gram of dust From which 99.7% where FF.

*The term Microplastics (MP) in particle type within this table include synthetic FF as part of their study, but it does not consider any fragmented fibre from natural or regenerated origin. While Fragmented fibres (FF) include all types of textile fibre materials.

4. Impact of textile materials and structure on the release of FF

Textile materials and structural configurations have been reported to affect the release of FF during laundering. In this section, the key material and structural variables are reviewed to develop insights to understand the role of the variables. There is a lack of systematic studies to elucidate the impact of different manufacturing steps on the fibre properties and eventually propensity to release FF.

4.1 Textile structure

4.1.1 Polymer type

Given that PET is the most common textile polymer, most of the published literature has focussed on PET fibres (Browne et al., 2011; Pirc et al., 2016; Hernandez et al., 2017; Jönsson et al., 2018; Kelly et al., 2019; Fontana, 2020; Cai et al., 2020; Özkan and Gündoğdu, 2021). However, there are studies on other synthetic materials such as polyamide (nylon), polyacrylic (acrylic) and polypropylene (Folkö, 2015; Hartline et al., 2016; Carney Almroth et al., 2018; De Falco, Gullo, et al., 2018; Belzagui et al., 2019; Hernandez et al., 2019; Berruezo et al., 2021; Volgare et al., 2021). Natural fibres, specifically cotton, have also been studied either as a single component (Sillanpää and Sainio, 2017; Stanton et al., 2019; Cesa et al., 2020; Athey et al., 2020) or in a blend with synthetics (Napper and Thompson, 2016; Haap et al., 2019; De Falco et al., 2020). Furthermore, regenerated materials, including acetate (Yang et al., 2019), rayon (Zambrano et al., 2019), and viscose (Galvão et al., 2020), have also been explored and compared with natural and synthetic fibres.

The comparison of synthetic and natural fibre textiles has exhibited significant differences in FF shedding behaviour. A PET-cotton blend (65-35%) was found to shed consistently fewer fibres than synthetic materials (PET and acrylic) (Napper and Thompson, 2016). While other authors reported significantly more FF shed from cotton and cellulosic materials than synthetic samples (Zambrano et al., 2019; Haap et al., 2019; Galvão et al., 2020). In the absence of a common approach and detailed materials and structural information, the studies cannot be directly compared; however, the results show a noticeable difference between shed fibres from different materials. This may be due to the physicochemical properties of the fibre materials (De Falco et al., 2020). Regarding PET, as a high breaking strength fibre, the pills resulting from textile abrasion in the fabric surface tend to stay attached to the fabric (Ukponmwan et al., 1998) and may release fewer fibres. While cellulosic fibres are hydrophilic, which influence fibre wettability and along with an increase in temperature, it causes the fibre to swell, promoting fibre migration from the core to the yarn surface (deGruy et al., 1962; Zambrano et al., 2019; De Falco et al., 2020). Additionally, mechanical abrasion of cotton in a wet state causes fibrillation of the fibre (Okubayashi and Bechtold, 2005; De Falco et al., 2020). Conversely, a lower release of cotton fibres than PET and acrylic (Napper and Thompson, 2016) may be associated with cotton hygroscopicity, which enhances hydrogen bonding in a wet state and increases fibre strength (Lau and Fan, 2009).

Regarding synthetic FF, plastics are formed by highly parallel oriented segments of polymer chains to produce a partly crystalline morphology (Andrady, 2017; Hernandez et al., 2017), embedded in a randomly oriented amorphous matrix (Andrady, 2017) (see Figure 8). This morphology makes the plastic tougher, but a higher crystallinity (which is the case for synthetic textile fibres) may produce a more brittle material (Andrady, 2017). Crystallinity is affected by polymer chemistry, thermal treatment, and processing history, and it directly impacts the fibre strength, density and weatherability, hence FF shedding. Also, most synthetic fibres are hydrophobic, so they do not swell on water, leading to a reduced release of FF. These statements agree with published findings. Different percentages of PET (14%) and cotton (86%) FF were collected from a 50-50 poly-cotton blend sample (Haap et al., 2019). At the same time, a difference in the release of PET and cotton between the proportion of washed fibres and recovered fibres was found (Galvão et al., 2020). Moreover, natural FF are more abundant than synthetic in environmental samples (Dris et al., 2016; Stanton et al., 2019; Q. Zhang et al., 2020; Athey et al., 2020; Parton et al., 2020). It is essential to mention that there is a need for a broader study to compare different materials with comparable structural configurations and explore conditions to understand the effect of polymer type on the amount of shed FF.

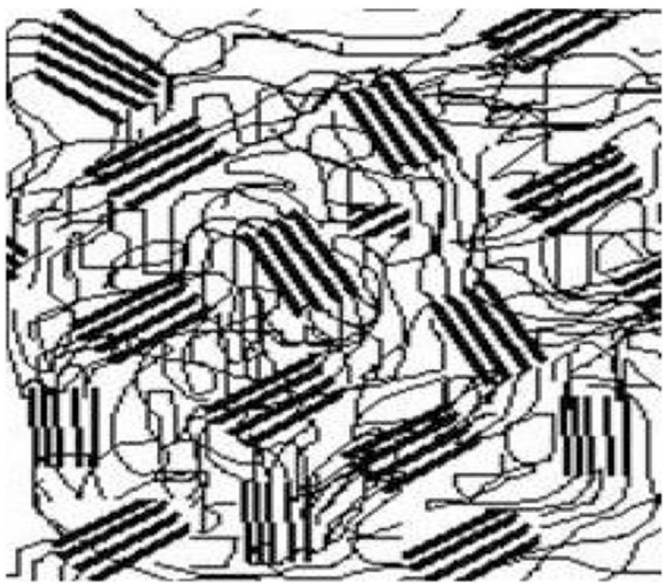


Figure 8.Schematic representation of a semi-crystalline plastic showing ordered segments (thick lines) of polymer chains that have crystal-like properties embedded in an amorphous (thin lines) polymer matrix. Reprinted with permission from Andrady, A.L., 2017. The plastic in microplastics: A review. Marine Pollution Bulletin. 119(1), p.17. Copyright 2017, Elsevier Ltd., DOI: 10.1016/j.marpolbul.2017.01.082.

4.1.2 Fibre form

Fibre form (staple or filament) is known to significantly impact the release of FF when laundering (Cesa et al., 2020; De Falco et al., 2020; Özkan and Gündoğdu, 2021; Palacios-Marín et al., 2022). Staple yarns are predominantly used in clothing and are known to shed more FF compared to continuous filament yarns (De Falco, Gullo, et al., 2018; De Falco et al., 2020; Özkan and Gündoğdu, 2021; Palacios-Marín et al., 2022). The staple length of the fibre may impact shedding since shorter fibres mean more fibre ends protruding from the yarn surface per length (as seen in Figure 9) (Cesa et al., 2020) and may have a greater tendency to escape (De Falco, Gullo, et al., 2018). This is the case with cotton, which presents a broader length distribution than staple extruded fibres owing to its natural origin, with a higher amount of short fibres (Wakeham, 1955; Cesa et al., 2020) that affect FF release (Cesa et al., 2020; Palacios-Marín et al., 2022).

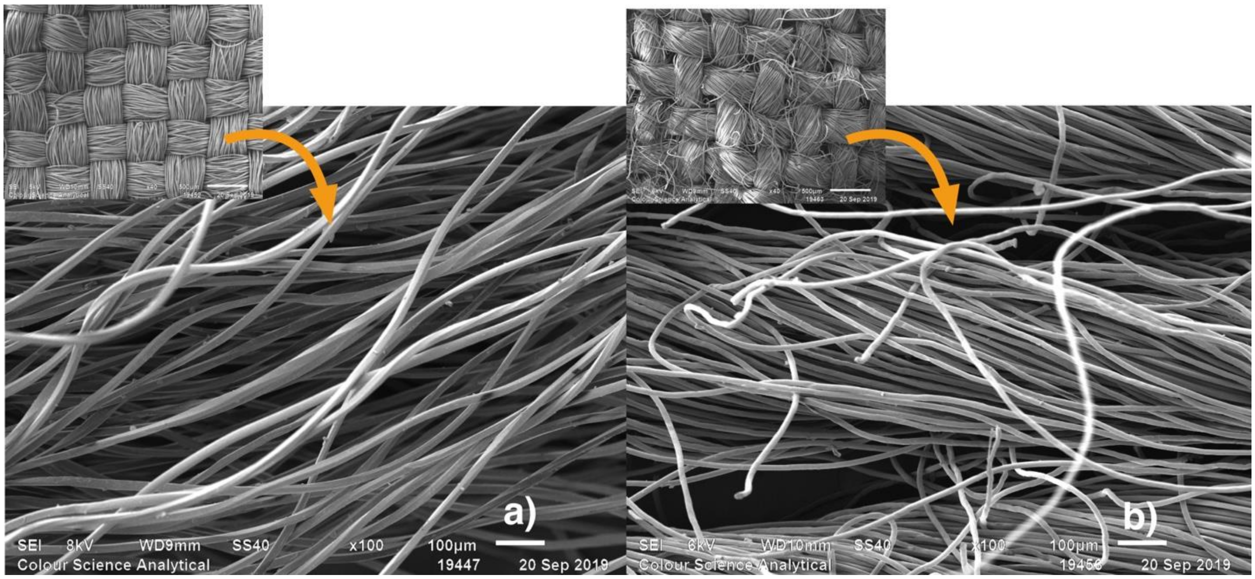


Figure 9 Surface detail of a) Filament yarn and b) Staple yarn. (Author).

4.1.3 Yarn structure

Yarns are defined as a bundle of natural and, or extruded staple fibres and, or filaments. They represent the rudimentary element of any geometrically structured fabric (Tausif et al., 2018). Yarns are produced by twisting a parallel fibrous strand, increasing the fibrous assembly's lateral force. The mass per unit length of a yarn is known as yarn count or number (Tausif et al., 2018). Both the yarn twist and count impact the FF release (Carney Almroth et al., 2018; Yang et al., 2019; De Falco et al., 2020; Palacios-Marín et al., 2022). A study reported that thicker staple yarns shed more fibres (Carney Almroth et al., 2018), and a positive relation between yarn count and FF release is defined owing to the higher amount of fibres per cross-section in the yarn (Yang et al., 2019). On the contrary, no impact was found on collected FF by the number of filaments in the yarn cross-section (Özkan and Gündoğdu, 2021). As stated before, differences in results may be attributed to their nature as continuous filament or staple fibres yarns. Results from (Palacios-Marín et al., 2022) from a comparative analysis of five different yarn structures show that fibre material and yarn structure impact directly in the release of FF.

The higher the twist, the tighter the fibres are bound together, and inter-fibre / inter-filament spaces are reduced (Tausif et al., 2018; De Falco et al., 2020). The higher twist levels may make it difficult for smaller fibres to escape from the yarn core, so the twist level in the yarn impacts FF release (Carney Almroth et al., 2018; De Falco et al., 2020; Palacios-Marín et al., 2022). Yarn twist is related to yarn hairiness and evenness. The first indicates the quantity and length of fibre ends protruding from the yarn surface, while the second refers to the variation level of the yarn diameter along its length (Zambrano et al., 2019). Both properties have a positive impact on the release of FF since a high level of hairiness mean a greater number of fibres per unit area exposed (Carney Almroth et al., 2018; Zambrano et al., 2019; Özkan and Gündoğdu, 2021) and an uneven yarn leads to poor abrasion resistance (Zambrano et al., 2019). Consequently, yarn twist is likely to be a key factor for FF shedding and warrants a detailed study. As this level of structural detail is not included in all the reported studies, this limits the direct comparison of different materials and inter-study comparisons.

4.1.4 Fabric structure

The fabric represents the sum of individual elements (fibres and yarns), their properties and mechanical characteristics (Militky and Ibrahim, 2009), which affect FF release. The impact of fabric structure on the release of FF is reported with differing results. For instance, (Carney Almroth et al., 2018) describes a clear impact of fabric structure affecting FF shedding, while (Hernandez et al., 2017) findings show no significant difference relating fabric type with fibre collection. Furthermore, textile samples manufactured with a tighter structure (higher number of yarns per unit length) are described to release a greater fibre loss owing to the higher availability of fibres per area (Carney Almroth et al., 2018) and a reduction in FF collection due to the difficulty of fibres to escape between narrower inter-yarn spaces from a tight structure (Yang et al., 2019). This is in agreement with a recent study evaluating the influence of weave patterns in FF release. Results show that the weave interlacing coefficient (IC) affects the release of FF. The higher the weft density, the lower FF release reported. This is attributed to the high fibre compactness levels restraining FF from leaving the yarns. The research showed no effect of the three different weave patterns (plain, satin, and twill) studied on the extent of the FF release (Berruezo et al., 2021). Nevertheless, results may be highly affected by the combination of polymer type, fibre form and yarn structure of samples, since they were manufactured of a textured filament PET warp and a staple acrylic weft.

Fleece fabrics are considered to release the highest level of FF in the literature (Browne et al., 2011; Folkö, 2015; Hartline et al., 2016; Pirc et al., 2016; Carney Almroth et al., 2018; Belzagui et al., 2019). The use of filament yarns in fleece fabric involves napping and shearing of fibres on the fabric surface, as seen in Figure 10, which cause fleece fabrics to shed more FF compared to other common fabrics (Browne et al., 2011; Carney Almroth et al., 2018), and shedding reduces after repeated washing (Pirc et al., 2016). The surface density of the fabric and the volume of fibre ends protruding from the surface (Folkö, 2015; Belzagui et al., 2019) due to mechanical finishing results in higher FF release.

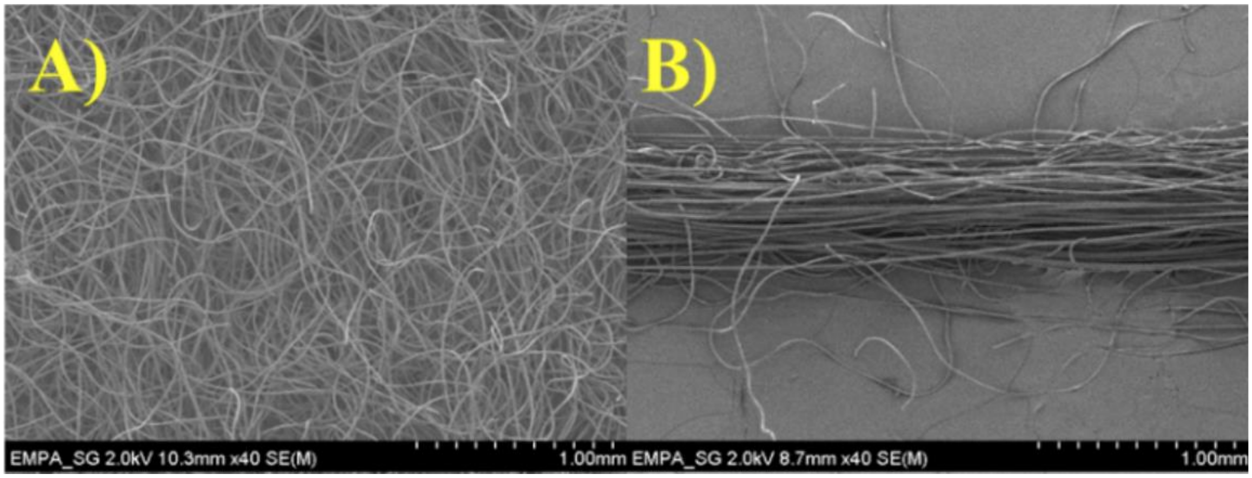


Figure 10 SEM images for the fleece textiles: A) surface and B) single yarns from fleece.
Reprinted with permission from (Cai et al., 2021). Formation of Fiber Fragments during Abrasion of Polyester Textiles. Environmental Science & Technology. 55(12), Figure S1, p. S2, Supplementary Information.
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4.1.5 Textile colouration and finishing

Colouration and finishing can modify the physical and chemical properties of textile materials. This includes adding chemicals and the physical manipulation of the fabric with mechanical devices (Roy Choudhury, 2017). Colouration technology involves numerous steps specific to fibre chemistry and the properties of dyes or pigments. The processing involves the use of high water volume for cleaning, dyeing and rinsing the fabrics (Drumond Chequer et al., 2013). The hydrodynamic forces related to this procedure may impact the generation and release of FF. There is a paucity of reported literature on the influence of finishing on synthetic FF shedding. Polyester textiles released significantly more fibres after their surface was mechanically processed (Cai et al., 2020), as is the case for fleece fabrics (Folkö, 2015; Carney Almroth et al., 2018; Belzagui et al., 2019). In addition, a study found PET fabrics to have a higher release of FF compared to PET-cotton blended samples. The authors suggest that PET textiles are often modified with finishes to improve their surface appearance, which may be responsible for reaching fibre fatigue more quickly (Napper and Thompson, 2016). However, it warrants further research to see the impacts of different finishes applied to textiles and FF release. Regarding natural fibres, a recent publication investigated the effect of colouration and some of the most common textile finishes over cotton. The results show the promotion of FF shedding with all treatments. From higher to lower release of FF, finishes are listed as durable press > softener > water repellent > dyeing. However, only water repellent and softener finish treatment presented a significant difference over the rest of the treatments. The finishing of cotton also influenced FF length and abrasion resistance (Zambrano et al., 2021b). There is a need to study other common fibre types and finishing treatments.

Garment manufacturing and finishing can also impact FF release. A recent study compared FF shedding of "distressed" (manufactured with intentional rips and holes) and non-distressed jeans. Results show a higher release of FF from non-distressed jeans and suggest that the distressing degree is likely to impact the particles shedding (Athey et al., 2020); however, a deeper investigation for garment manufacturing and treatment is recommended.

4.2 Impact of washing conditions

This subsection analyses the impact of washing conditions on the release of fragmented fibres. The release of FF is influenced by differences in washing cycles, including time, temperature, mechanical stress and water volume (Fontana, 2020). However, in the absence of directly comparable conditions, the current literature understandably reports disagreeing findings. For example, mechanical stress in accelerated washing procedures shows no significant impact when the number of steel balls used is increased (Hernandez et al., 2017; Cai et al., 2020). Contrarily, a positive relation between mechanical stress and FF shedding is attributed to a higher level of stress generated on top-load machines compared to front-load devices (Hartline et al., 2016). The differences in findings may be attributed to laundering devices, sampling textile properties and washing conditions. Moreover, results comparing top-load versus front-load appliances may be influenced by the water volume employed in each device (Kelly et al., 2019). The literature suggests that water volume represents the most significant factor affecting FF release during laundry (Kelly et al., 2019). When washing processes are carried out, fibres are subjected to massive viscous forces (fluid frictional forces that oppose the motion of adjacent fluid layers or the resistance to flow displayed by a fluid (Arfken et al., 1984)), and their large surface area to volume-ratio ease the release of FF from textiles. The same hydrodynamic forces could also diminish yarn strength, causing a higher fibre collection (Kelly et al., 2019). In this regard, a recent study evaluated the impact of washing load (WL) (0.15kg, 0.88kg, 1.64kg, and 2.5kg) on the extent of FF release. Results show a progressive decrease of FF release with increasing washing load and a significant impact in FF length. This is attributed to the synergistic effect between the water-volume to fabric ratio and the mechanical stress on the samples caused by the different textile amount used in each test (Volgare et al., 2021).

4.2.1 Sample preparation

a. Pre-washing assessment

Pre-washing processes are performed as part of the methodology to eliminate impurities that may attach to textiles during manufacture and storage (Jönsson et al., 2018). This process may be executed in wet or dry conditions. Wet conditions include a rinse cycle with distilled water (Hernandez et al., 2017), short washing cycles without additives (Carney Almroth et al., 2018; Zambrano et al., 2019; Yang et al., 2019), several washing cycles prior to the experimental, which maintain the conditions established for the experimental laundry (Napper and Thompson, 2016), scoured and dried (Fontana, 2020), and pre-wash cycle with additives (De Falco et al., 2020). Assessments proposed by dry routes involve gentle shake (Hartline et al., 2016), use of lint roller (Haap et al., 2019), and vacuuming of samples (Jönsson et al., 2018; Özkan and Gündoğdu, 2021). As discussed in section 3, the pre-washing assessment may or may not influence FF release. For instance, the higher release of FF after the first washing cycle is reported even though different pre-washing processes are carried out in dry and wet conditions (Napper and Thompson, 2016; Zambrano et al., 2019; Özkan and Gündoğdu, 2021). However, it is not always the case, since other findings show a steady release during laundry cycles after pre-washing assessment

(Hernandez et al., 2017; Fontana, 2020) or without following any pre-washing procedure (Hartline et al., 2016). Therefore, pre-washing of samples may not be the only factor affecting this behaviour, and more importantly, the material processing history is essential to be reported/known.

b. Cutting and hemming techniques

The sample preparation for the FF trails is critical, especially when lab-scale laundry experiments are conducted with smaller specimens rather than full garments. The cutting method was found to impact the release of fragmented fibres significantly. This included a significant difference between the scissor cut and ultrasound cut materials (Jönsson et al., 2018) and up to 21 times higher release with the scissor-cut compared to the laser-cut samples (Cai et al., 2020). The opening of the yarn ends may cause these results, inducing FF shedding (Cai et al., 2020) or new FF generation at the fabric edge (Jönsson et al., 2018). Also, finishes on fabric edge may impact FF shedding since additional fibre damage is generated when cutting or needling, as is the case for overlocked samples with higher levels of FF than samples with a raw edges (Cai et al., 2020). Additionally, laser-cut significantly releases fewer fibres than overlocking and double-folded sewing because the edges of synthetic fibres are melted when applying this technique (Cai et al., 2020).

4.2.2 Use of detergent

Additives including detergent, bio-detergent and fabric conditioner or softener are studied in published literature. A positive correlation between the use of detergent and the release of fragmented fibres is described (Hernandez et al., 2017; Carney Almroth et al., 2018; De Falco, Gullo, et al., 2018; Yang et al., 2019). It may be caused by decreased frictional forces and lubrication of fibre surface, which facilitates FF release between inter-fibre spaces (Hernandez et al., 2017; Yang et al., 2019). However, a higher amount of surfactant is not equal to higher FF shedding since similar levels of FF release are found after increasing the detergent dosage (Hernandez et al., 2017). The type of detergent used (liquid or powder) is documented in the literature, with findings showing no difference between detergent type (Hernandez et al., 2017) or higher FF levels caused by powder detergent (De Falco, Gullo, et al., 2018). The latter results could be explained by the insolubility of some powder detergent compounds in water, which cause friction against fibres and a higher pH of the solution induced by the powder detergent (De Falco, Gullo, et al., 2018). Since softener increase pilling and fabric breaking strength, especially in synthetic fibres, the use of this additive is suggested to influence the FF release (Napper and Thompson, 2016).

The use of detergents and their form (powder, liquid (Hernandez et al., 2017; De Falco, Gullo, et al., 2018)) is essential to simulate practical washing conditions (Hartline et al., 2016; De Falco, Gullo, et al., 2018). The studies on repeated washings of the same textile articles have reported contradictory results of no effect (Hernandez et al., 2017; Volgare et al., 2021), increase (Hartline et al., 2016), or decrease (Pirc et al., 2016; Cesa et al., 2020) on the emission of FF. The latter findings are attributed to a reduction of mechanical action and a decrease in textile damage due to foaming created by the surfactant (Cesa et al., 2020). Also, significant results that do not follow an evident tendency comparing the absence of additives, bio-detergent, non-bio detergent, fabric conditioner and their combination in different textile samples are reported (Napper and Thompson, 2016). Finally, tumble drying aggravates the release of FF (Pirc et al., 2016).

4.2.3 Temperature

The temperature during washing is another crucial factor that could impact FF shedding. Either no impact is reported after an increase in temperature (Hernandez et al., 2017; Kelly et al., 2019), or a significant higher FF release is described (De Falco, Gullo, et al., 2018; Yang et al., 2019). This effect is only found in cotton fibres (Zambrano et al., 2019) due to its hydrophilic nature. Even though synthetic fibre results are not statistically significant (Zambrano et al., 2019), high temperatures in water cause damage in synthetic fibres. Dents and fractures are observed in fibres after steeping textiles at 95ºC (Figure 11) (Hernandez et al., 2019). These findings demonstrate that high temperatures are likely to affect the release of FF from natural and synthetic fibre textiles.

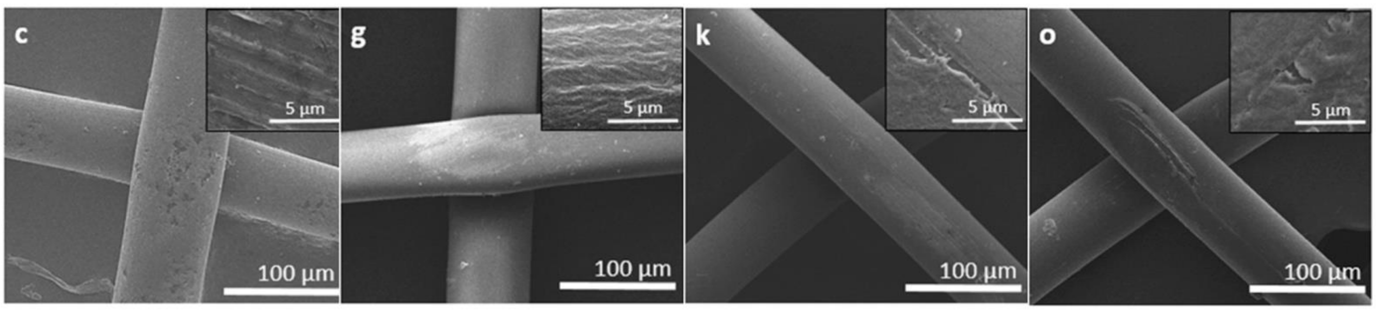

Figure 11 Imaging at 1000× of the teabags after steeping reveals a rougher surface, at higher magnification (30000× insets) dents and fractures are observed. Adapted with permission from Hernandez, L.M., et al., 2019. Plastic Teabags Release Billions of Microparticles and Nanoparticles into Tea. Environmental Science & Technology. 53(21), p. 12301. Copyright 2019, American Chemical Society. DOI: 10.1021/acs.est.9b02540.

5. Mitigation strategies currently implemented

Though the primary focus has been on understanding the level of release of FF from textiles, different approaches have been explored to mitigate the release of FF to the environment. This includes interventions during textile manufacturing and collection of FF in washing effluent, which are discussed in reference to the applicability of these solutions at the industrial and consumer level.

The laundry processes are recognised as a critical source of fragmented fibres (Browne et al., 2011). The wastewater effluent from household laundry devices may be discharged directly into the environment or treated in WWTP prior to reaching aquatic environments (Murphy et al., 2016). A recent study reports retention of 99.9% of MP in WWTP (Ball, 2019). However, sheer water volumes in WWTP (Acharya et al., 2021) and intense rainfall seasons may exceed the WWTP handling limit (Napper et al., 2020) and result in a straight discharge of untreated effluent into aquatic environments (Napper et al., 2020; Acharya et al., 2021). Even if a high filtration of MP and FF is considered, still a substantial number of particles can reach the ecosystem. For instance, an estimation of 65 million MP released daily into the environment was calculated for a 650,000 population with a removal efficiency of 98.4% in WWTP (Murphy et al., 2016). And probably, a larger proportion of the particles escaping WWTP would correspond to FF over other MP forms (Napper et al., 2020; Acharya et al., 2021). Moreover, because anaerobic or aerobic processes do not decompose synthetic FF in WWTP, they accumulate in sewage sludge. FF can reach the environment again when sewage sludge is used for agricultural purposes, dumped into the ocean or at land (Acharya et al., 2021). The mitigation strategies include the manipulation of textiles and their structure and collection devices to be used in laundry machines.

5.1 Textile Finishing

Textile finishing involves the post-treatment of fabrics to impart a specific aesthetic or functional aspect. In literature, different chemical treatments are reported to coat the surface of synthetic fabrics and mitigate fibre fragmentation and release. The choice of the finishing material was aimed to be from eco-friendly sources to prevent the introduction of external contaminants into the environment (De Falco, Gentile, et al., 2018; De Falco et al., 2019). This included a pectin-based bio-coating obtained from natural polysaccharide organic waste. The abundant availability of raw material and low cost can promote this product to be commercially feasible (De Falco, Gentile, et al., 2018). In addition, two biodegradable polymers, Poly (lactic acid) (PLA) and poly (butylene succinate-co-butylene adipate) (PBSA), were tested as coating materials for polyamide fabrics. PLA is a form of polyester derived from renewable sources that offers a high coating resistance during washing cycles. At the same time, PBSA is a low-cost random copolymer with an excellent biodegradation rate in the marine environment (De Falco et al., 2019). The finishes were applied to the fabric samples by padding process for pectin based (De Falco, Gentile, et al., 2018) and ElectroFluido-Dynamic method for PLA and PBSA coatings (De Falco et al., 2019). The simulated washing processes were carried out to assess their efficiency, one cycle for Pectin and five for PLA and PBSA. Laundry effluent was filtered, and the amount of FF was calculated. The fabric samples were 100% polyamide-6,6. Commercially available detergent was employed for all the trials (De Falco, Gentile, et al., 2018; De Falco et al., 2019).

All finishes show a reduction of around 90% of FF shedding compared to untreated polyamide. This is the average value after one cycle for pectin-based bio-coating and five cycles for PLA and PBSA coatings. Also, there is a noticeable increase (PLA and PBSA) and a slight increase (pectin based) in fabric tear strength after treatment. The analysis of the sample surface exhibits the presence of the coating after washing in all cases and after five laundry cycles for PLA and PBSA. In addition, FF released from treated samples have a greater mean FF length and diameter than those detached from untreated polyamide. The fabric hand, assessed by holding coated fabrics, and fibre morphology were not altered compared to untreated samples (De Falco, Gentile, et al., 2018; De Falco et al., 2019).

With the low number of washing cycles assessed (one for Pectin, five for PLA and PBSA) and in the absence of exposure to other environmental variables, the long-term durability of the treatment is not determined. For instance, even if the surface analysis after one (Pectin) and five (PLA, PBSA) washing cycles exhibited the presence of the coating treatment, fibre mitigation effectiveness was evaluated only on the first laundry cycle. The coating treatments could potentially achieve a commercial application due to the simple application process and compatibility with the existing industrial technology (De Falco, Gentile, et al., 2018; De Falco et al., 2019), but require further work. The FF detachment during dry processes is needed along with the treatment and performance of coatings in other synthetic, natural, and regenerated materials. In particular, polyamides are already known to exhibit high abrasion resistance. It is important to note that PLA particles detached from the textile finishing surface may be considered MP because PLA is not fully degraded under natural and aquatic conditions (Lambert and Wagner, 2017; SAPEA, 2019).

5.2 Changes in the textile structure

The modification of textile structures to prevent fibre shedding is an active area of research. One example includes an initiative to reduce FF release during laundry processes by Polartec. In November 2018, the company released the new textile "*Power Air*", which is engineered to mitigate FF by trapping insulating lofted fibres through two surfaces of knitting, creating individual air pockets (see Figure 12). Power Air is created as a mid-layer weight fabric to maintain heat and replace fleece fabrics. This product features an internal grid structure that encapsulates lofted fibres within a continuous yarn, multilayer fabric. The outer side presents a flat surface from the same multilayer fabric. This structure is engineered to offer advanced thermal efficiency. The durable outer surface from both sides has a better pilling resistance, minimising the contact with fibre endings and creating new FF. According to the company performance tests, the fabric releases five times less FF, which are not described in detail (POLARTEC, 2018).

Figure 12 Detail of "Power Air" fabric.
(POLARTEC, 2018) [Online image] Retrieved from https://www.polartec.com/fabrics/insulation/power-air (Accessed 31 Jan. 2022).

5.3 Retrofit devices used in washing machines

The use of retrofit devices to collect FF during the laundry process is discussed in this section. All of them are placed inside the washing machine during laundry and removed when the process is finished. FF have then to be cleaned from the device for its reuse. The description of each device is below, and Figure 13 display the devices:

a) Cora Ball (Cora Ball, VT, USA, Figure 13a): A ball-shaped device formed by stalks with hooks on the end. The design is inspired by the filtration system of coral reefs. This device must be placed inside the washing machine while laundering. It captures FF into visible fuzz to be disposed of by the consumer (Napper et al., 2020; Cora Ball, 2022). The product is commercially available.

b) Guppyfriend Washing bag (Langbrett, Germany, Figure 13b): A polyamide mesh bag where the garments are placed inside and laundered. FF are collected in the corners of the bag to be easily removed and disposed of by the consumer after laundry (Napper et al., 2020; Guppyfriend, 2022). The product is commercially available.

c) Fourth element Washing Bag (Fourth Element, U.K.): This mesh bag follows the same function as the Guppyfriend but was designed to wash thermal undergarments (Napper et al., 2020). Fleece fabrics, which are made of, typically release a higher amount of FF than other fabrics (Folkö, 2015; Hartline et al., 2016). This is a product prototype at the time of the review.

Figure 13 Retrofit devices for washing machines.
a) Coraball. (Cora Ball, 2022) [Online image] Retrieved from https://www.coraball.com. (Accessed 20 Feb 2022).
b) GuppyFriend (Guppyfriend, 2022) [Online image] Retrieved from https://en.guppyfriend.com. (Accessed 20 Feb 2022).

In a study evaluating the efficiency of FF retention from different devices by (Napper et al., 2020), results show a successful reduction percentage of 54 ± 14% for the Guppyfriend, 31 ± 8% for Cora Ball and 21 ± 9% for the Fourth Element prototype. Only the Guppyfriend washing bag has a significant reduction (p-value<0.05) compared to the control sample without any device in use. The vast difference between the two washing bag devices does not depend on their pore size since both have comparable measures (50μm), so it may be related to the shape and design of the bags (Napper et al., 2020). In a similar study, results from Cora Ball report to reduce 26% the number of FF/L of effluent (McIlwraith et al., 2019), comparable to earlier reported results.

The mentioned devices benefit from the practical design and easy use for the consumer. Nevertheless, captured FF are disposed of in the bin. Even if they do not follow the pathway of washing effluent to reach aquatic environments, they could still represent a hazard for terrestrial environments and finally reach aquatic ecosystems. The shortcomings of these devices and disposal options for recovered FF are further discussed compared to external filtration units for washing effluent in the following section. The environmental pathways for FF and their impact can be consulted in section 5.4.

5.4 Collection of FF in washing effluent

The external filtration devices are installed on the pipe carrying the effluent from the washing machine. These devices filter the washing effluent and collect FF to prevent their release to WWTP and/or aquatic environments. Three such devices are discussed in this section and can be consulted in Figure 14.

a) Lint LUV-R (Environmental Enhancements, NS, Canada): This filtration system is mounted under a cabinet, shelf, or wall next to the washing machine. Wastewater effluent is filtered, and the device collects the FF. The efficiency improves after collecting an initial amount of FF on the device walls that will serve as filter lint. The cleaning is recommended after 2-3 laundry loads. Collected FF are disposed into the bin (Environmental Enhancements, 2020; Napper et al., 2020).

b) XFiltra prototype (Xeros Technology Group, U.K.): This filtration device is designed to be integrated into any commercial laundry machine. It includes an integrated pump, a filter, and a de-watering device that capture the FF by separating them from the water of the laundry effluent. The device is designed to support easier disposal and the cleaning is recommended every 30 cycles. The dry filtered mass is then removed and disposed of in the bin (Xeros, 2020; Napper et al., 2020).

c) PlanetCare (PlanetCare Limited, U.K.): This filter device is designed in a layered structure for a better collection and distribution of FF. The reusable cartridges must be replaced after 20 washing cycles. One of the benefits is the closed-loop offered to the customer since cartridges containing recovered FF are returned to the company for an appropriate disposal (PlanetCare, 2020; Napper et al., 2020).

Figure 14 External filtration devices
a) Lint LUV-R (Environmental Enhancements, 2020) [Online image] Retrieved from https://environmentalenhancements.com/store/ (Accessed 20 Feb 2022).
b) XFiltra prototype (Xeros, 2020) [Online image] Retrieved from https://www.xerostech.com/technologies#xfiltra

(Accessed 20 Feb 2022).
c) Planet Care (PlanetCare, 2020) [Online image] Retrieved from https://planetcare.org (Accessed 20 Feb 2022).

The research by (Napper et al., 2020) compares the aforementioned devices. The XFiltra prototype is placed as the most successful apparatus in retaining FF from washing effluent. The reduction percentage is 78 ± 5%, compared to 29 ± 15% and 25 ± 20% from Lint LUV-R and PlanetCare, respectively. Also, only the XFiltra retains significantly more FF than the control sample. The success of this device is attributed to the finest mesh pore size (60µm) compared to the other filtration devices and the design structure. In comparison, the research by (McIlwraith et al., 2019) on filtration devices (Lint LUV-R and Coraball) displays an 87% of FF reduction of Lint LUV-R compared to control. The contrast in results can be potentially associated with the methodology differences followed in both research studies. Moreover, the average size of FF captured with Lint LUV-R was longer than those of Cora Ball and control samples (McIlwraith et al., 2019). Accordingly, it warrants further research on retention devices to investigate the size/amount of FF, and any impacts on the environment.

Better performance of external filtration units for washing effluent is reported compared to devices placed inside the washing machine (Napper et al., 2020). However, the downsides of these appliances are the extra cost for the consumer and specialist fitting of the additional devices, except for XFiltra which is designed to be included as part of the laundry machine. Apart from the PlanetCare filter, external and in-drum devices require a cleaning process by the consumer and FF lint can be disposed into the bin (Napper et al., 2020). The disposal of FF may end up in the environment by other means. If filters are washed, the collected mass can be unintentionally released into the environment via drained water. This can include terrestrial environments, such as agricultural soils, and endanger biota and biological natural processes (Henry et al., 2019). Moreover, they could also be transported by airflow and reach aquatic environments or other ecological systems by atmospheric deposition (Singh et al., 2020). An alternative solution to benefit from the lint accumulation in these devices is proposed by a research team from Kaunas University of Technology (KTU) and the Lithuanian Energy Institute. Lint FF collected from drying machines were thermally treated in a pilot pyrolysis plant to successfully extract oil, gas, and char from these debris. A 70% conversion rate into energy products is estimated for lint FF after thermally treated, which represents massive amounts of energy production and carbon footprint reduction (Innovation in Textiles, 2021).

It is crucial to consider any financial implications for the consumer and the space required for the installation of external filtration devices. These factors may restrict consumers to apply any of the technologies to their day-to-day life (Singh et al., 2020). Fragmented fibres are not entirely collected with any of the devices used during laundry; therefore, other measures must be considered to achieve higher collection efficiency. A combination of filtration and collection devices with modifications on textiles discussed throughout this section may result in the solution for capturing FF before reaching the environment.

5.5 Wastewater treatment plants

The role of wastewater treatment plants in releasing FF into the environment has not been completely defined. However, they are considered a representative pathway of MP and FF to reach the aquatic and terrestrial environments (Acharya et al., 2021). Wastewater treatment processes commonly comprise preliminary, primary, and secondary treatment stages. Additionally, a tertiary treatment process is often included based on specific requirements of the plants (Acharya et al., 2021). A representation of WWTP processes can be consulted in (Masiá et al., 2020) and it is shown in Figure 15. Assessment of MP/FF removal from WWTP shows a highly effective recovery system with a reduction of 98-99% of particles after secondary treatment (Murphy et al., 2016; Talvitie, Mikola, Setälä, et al., 2017; Acharya et al., 2021). However, even with advanced treatment processes, the WWTP represent a considerable source of FF and MP (Acharya et al., 2021; Sol et al., 2021). For instance, a study regarding the fate of MP on a highly efficient secondary WWTP for a city of 650,000 inhabitants in Scotland reported a recovery of 98.41% from the water influent and a release of 0.25 (±0.04) MP per litre in the final effluent. This corresponds to 65 million MP released per day (Murphy et al., 2016; Acharya et al., 2021). A similar study conducted on several WWTP across the United States found 0.05 (±0.024) microparticles per litre of effluent. The daily discharged average is over 4 million microparticles, rising to 15 million microparticles a day (Mason et al., 2016; Acharya et al., 2021). Finally, (Talvitie, Mikola, Setälä, et al., 2017) investigated natural and synthetic micro-litters from a WWTP with a 99% removal efficiency rate. Their results estimated 200 – 790 million micro-litter particles released on a daily basis, from which 1.7–140 million particles correspond to MP released per day to the Baltic Sea (Talvitie, Mikola, Setälä, et al., 2017; Acharya et al., 2021).

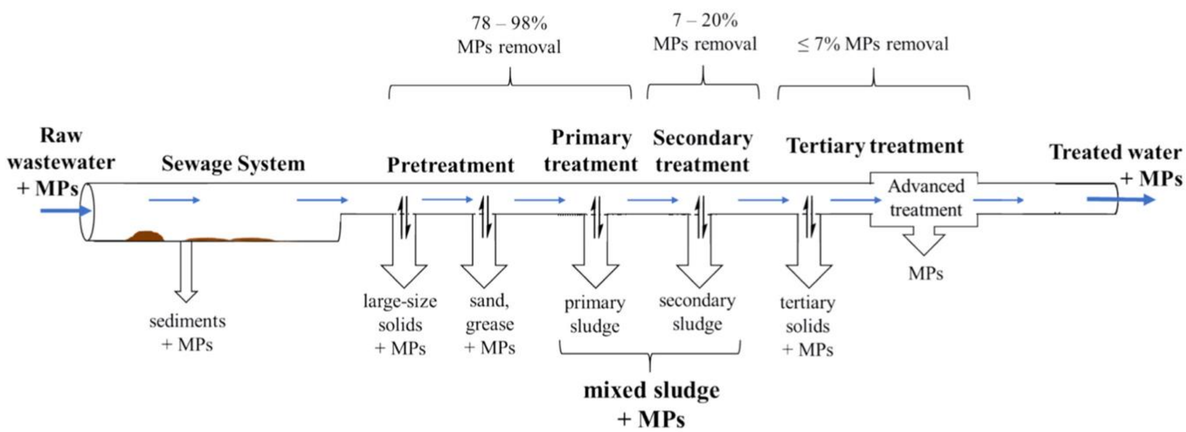

Figure 15 A schematic representation of WWTP processes and percentages of MPs removal during processing. Reprinted with permission from Masiá, P., et al., 2020. Bioremediation as a promising strategy for microplastics removal in wastewater treatment plants. Marine Pollution Bulletin. 156, p. 2. Copyright 2020, Elsevier Ltd., DOI: 10.1016/j.marpolbul.2020.111252.

These results are exacerbated when the textile industrial sewage discharge is analysed. The manufacturing steps that textiles go through when converted to final products increase the amount of FF released. The FF concentration in textile industrial sewage is up to 10,000 times higher than municipal sewages in China, especially on the effluent discharged after dyeing and printing processes. Even with a successful FF removal rate of over 90%, FF discharged into waterbodies from a single mill can rise up to 7.5×10^7 a day (Xu et al., 2021). The analysed studies suggest that millions of microparticles are released from WWTP every day, even with an outstanding recovery rate. Moreover, FF are the predominant form of microparticles released in WWTP (Acharya et al., 2021). For instance, (Mason et al., 2016) found that fibre-shaped particles were released with an occurrence of 59% among microparticles released in the final effluent. At the same time, a study from (Michielssen et al., 2016) shows a fibre form predominance of 83% among microplastic and other micro-anthropogenic litter released in the final stage of WWTP. Although results from (Murphy et al., 2016) do not display the percentage of incidence for fibrous microplastics, the most common materials collected on the final effluent are polyester and polyamide, which most likely correspond to FF released from textiles (Acharya et al., 2021). On the contrary, findings from (Talvitie, Mikola, Setälä, et al., 2017) show an extraordinarily low percentage (14%) of fibres released in the final effluent. The difference between results may be associated with the distinct treatment processes carried out on the WWTP studied. A detailed overview of the incidence of MPs and FF in the influent and the effluent of WWTPs worldwide can be consulted in (Sol et al., 2021).

Microplastics can affect the primary, secondary, and tertiary treatment stages of wastewater. The primary treatment uses precipitation, sedimentation, and other physical methods to remove suspended solids in wastewater. Strong acid, alkali and concentrated toxics are neutralised before the effluent reaches the next stage. The excessive accumulation of MP may cause blockage of the grille bars and a higher dosage demand of reagents. Moreover, MP can adsorb toxic components, which affects the proper pollutant removal (Bakir et al., 2014; Zhang and Chen, 2020; Masiá et al., 2020). The secondary treatment is a biochemical based action for biodegradation of organic matter and the subsequent separation of solid particles from the treated water (Masiá et al., 2020). The presence of microplastics in this stage affects the microbial-mediated process that controls the production and reduction of ammonium and the biological conversion efficiency of inorganic nitrogen. The impact of these processes affects the toxicity levels in water for aquatic organisms (Bendell et al., 2014; Cluzard et al., 2015; Zhang and Chen, 2020). Finally, during the third stage, disinfection of the wastewater is carried out to eliminate active pathogens through chlorination and UV irradiation (Masiá et al., 2020). MP agglomerate due to their size, density, and surface properties. These MP aggregates are different from expected structures in wastewater. The changes of the parameters from the original design may increase chemicals needed for the treatment, the ease of MP to adsorb pollutants, and the impossibility to bring existent contaminants into the water surface (Lai et al., 2014; Perren et al., 2018; Zhang and Chen, 2020).

Fragmented fibres intercepted during the wastewater treatment do not biodegrade easily with aerobic or anaerobic processes and will end as part of the sewage sludge (Acharya et al., 2021). Especially those particles removed in the first and second stages of the treatment (Masiá et al., 2020). Therefore, FF can be released to the environment at some stage (Acharya et al., 2021). The presence of micro and nanoplastics in sewage sludge impacts the sludge digestion rate. Commonly, the anaerobic digestion method is used for sludge stabilisation. In this process, particulate organic matter is converted into soluble substrates. High levels of MP in waste activated sludge (WAS) may negatively affect the hydrolysis of polysaccharides and proteins. Consequently, a reduction in the degradation rate and increase of chemical toxicity are expected (Wei, Huang, et al., 2019; Wei, Zhang, et al., 2019; Zhang and Chen, 2020).

The effective removal of MP from wastewater and sewage sludge is still under investigation. Existing techniques result in inefficient, expensive methods with chemical reagents that pollute and endanger the ecosystem. In addition, filtrations systems may not be adequate since shortcomings include rupture of MP due to mechanical stress, enabling the particles to reach the environment (Zhang and Chen, 2020). Different approaches to MP removal are described below.

A summary of the efficiency of different MP removal methods can be consulted in (Zhang and Chen, 2020) and (Masiá et al., 2020).

a) Sol-gel method: This technique aims to create large particle agglomerates by sol-gel induction. Flocculation of MP is promoted by a pH-induced sol-gel process, allowing the isolation of MP agglomerates by separation systems in WWTP. This sustainable process is suitable for any type, size and pollutant amount of MP particles (Herbort, Sturm and Schuhen, 2018; Herbort, Sturm, Fiedler, et al., 2018; Zhang and Chen, 2020).

b) Electrocoagulation: This technology produces coagulants electrically with metal electrodes. The coagulants produced by ions are created by electrolysis and aimed to form a sludge layer that captures suspended particles such as MP. Finally, the sludge layer is brought into the effluent surface and removed. It is a compatible, cost and energy effective technique that decreases sludge formation and may remove over 90% of MP in the wastewater (Garcia-Segura et al., 2017; Perren et al., 2018; Zhang and Chen, 2020).

c) Dynamic membranes: This filtering technique applies existing pollutants in wastewater to form a filter layer. There is no need to add any external chemicals for this process. Low-density non-degradable particles, such as MP, are trapped in the filter layer for proper removal. This is one of the most effective removal techniques that require low energy and cost (Chu et al., 2012; Li et al., 2018; Zhang and Chen, 2020).

d) Bioremediation: This is a new and promising technique that involves biological species such as higher eukaryotes, sandworms, sea cucumbers, seagrass and macrophytes to eliminate MP from wastewater. Species first collect or digest MP particles, and individuals are transferred to a clean environment where they can eliminate MP safely and return to the WWTP effluent. Further studies are needed to enhance the bioremediation processes (Masiá et al., 2020).

The occurrence, fate, and mitigation strategies of microplastic and fragmented fibre particles in wastewater treatment plants are out of the scope of the present work. For an exhaustive review, it is suggested to study (Michielssen et al., 2016; Mason et al., 2016; Talvitie, Mikola, Setälä, et al., 2017; Talvitie, Mikola, Koistinen, et al., 2017; Gatidou et al., 2019; Masiá et al., 2020; Acharya et al., 2021; Sol et al., 2021).

6. Environmental impact and risks associated with FF

Fragmented fibres are released from textiles during production, use, service, and end-of-life disposal into aquatic and terrestrial environments following different pathways. Among them, washing effluent from clothing is pointed out as a critical source of FF pollution for both environments (Browne et al., 2011; Hartline et al., 2016; Carney Almroth et al., 2018). Wastewater treatment plants are unable to retain all FF, and as a result, particles may escape by effluent to be released to aquatic environments (Napper and Thompson, 2016; Cesa et al., 2017; De Falco, Gentile, et al., 2018; Cai et al., 2020; Acharya et al., 2021). The route for terrestrial ecosystems depends mainly on the sludge from WWTP that is applied to agricultural fields as fertiliser or dumped into the land or the ocean (Browne et al., 2011; Murphy et al., 2016; De Falco, Gullo, et al., 2018; Acharya et al., 2021). Since MP and FF are not monitored as hazardous substances, the recovered particles are directly applied into the soil (Henry et al., 2019) and may remain there even after 15 years of application without any significant change in their fibrous structure (Zubris and Richards, 2005; Henry et al., 2019).

The fate of FF between aquatic and terrestrial environments is interconnected. FF released into the atmosphere by textile wearing and cleaning or accumulated in sludge-treated soils may return directly to aquatic environments by atmospheric deposition. Moreover, WWTP are partially open systems with some of the stages exposed to atmospheric fallout; therefore, FF could travel and return into the final effluent of WWTP reaching aquatic environments (Dris et al., 2016; Stanton et al., 2019; Cai et al., 2020). Fragmented fibre shedding through textile washing (Carney Almroth et al., 2018; Belzagui et al., 2019; Athey et al., 2020; Özkan and Gündoğdu, 2021) and its frequency in the atmosphere (Dris et al., 2016; Stanton et al., 2019; Chen et al., 2020; Q. Zhang et al., 2020) have been thoroughly studied. Despite this extensive information, FF's pathways between sources and sinks are not entirely understood. Figure 16 shows indigo denim FF recovered from different sources. This is a clear example of the wide variety of aquatic sinks for FF. The degradation rates for these particles and their ability to become airborne and carried between terrestrial and aquatic environments still need to be elucidated (Henry et al., 2019). Furthermore, there is insufficient data on alternative pathways for FF to reach the environment, for instance, the discharge of household wet-cleaning effluent into the sewage system (Lassen et al., 2015). Although data availability in some cases is minimal, there is sufficient evidence to prove FF pollution as a threat to the environment and human health (Singh et al., 2020).

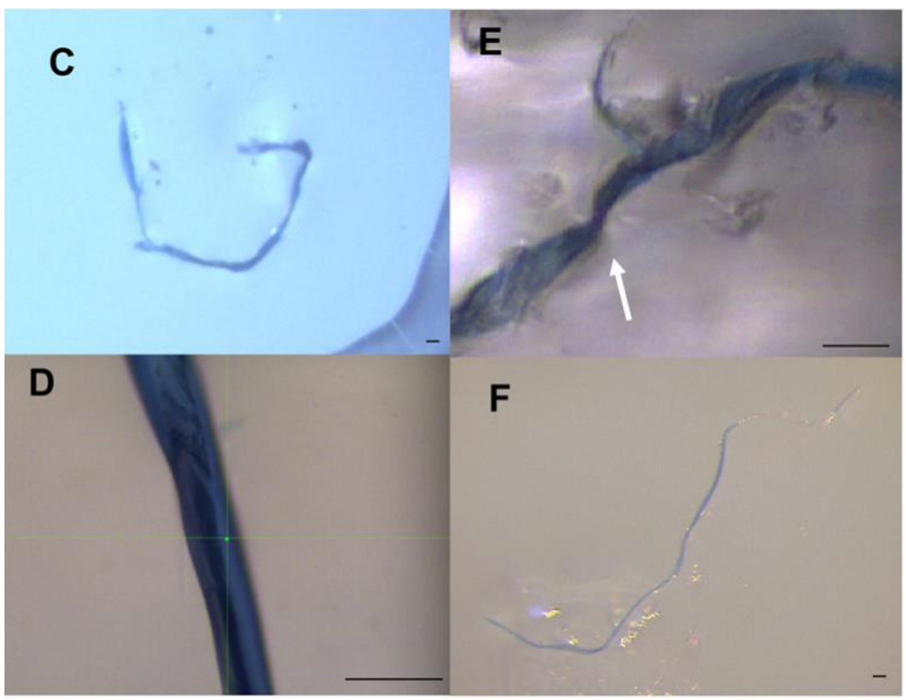

Figure 16 Images of an indigo denim fibre identified as cotton found in (C) Arctic sediments, (D) Great Lakes fish, and (E) WWTP effluent and (F) a denim fibre released from blue jeans collected from wash water effluent. Images acquired using a Leica microscope (80×) (C and F) or a micro-Raman spectrometer (HORIBA Raman Xplora Plus) (D and E) (100× and 500×, respectively). Arrows highlight the unique collapsed shape of the cotton fibre. Scale bars are 10μm.
Reprinted with permission from Athey, S.N., et al., 2020. The Widespread Environmental Footprint of Indigo Denim Microfibers

from Blue Jeans. Environmental Science & Technology Letters. 7(11), p. 841. Copyright 2020, American Chemical Society. DOI: 10.1021/acs.estlett.0c00498.

6.1 Impact on environment

The plastic pollution problem has been redefined after acknowledging that only 6% of plastic pieces entering the ocean are visible on the marine surface and beaches (Eunomia, 2016; Henry et al., 2019). In both terrestrial and aquatic environments, the presence of synthetic and natural FF has been established (Browne et al., 2011; Murphy et al., 2016; Cesa et al., 2017; Carney Almroth et al., 2018; De Falco et al., 2020). FF pollution has reached even the most remote locations globally (Wright and Kelly, 2017; Henry et al., 2019; Singh et al., 2020). Their ubiquitous nature and rising proportions in the environment have placed them as one of the most enduring and pervasive anthropogenic pollutants in the earth atmosphere (Singh et al., 2020). It is estimated that 0.19 MT of fragmented fibres from synthetic textiles enter annually into the ocean (Henry et al., 2019). However, this number does not consider any FF detached from regenerated or natural textile materials.

Fragmented fibres can potentially harm the health of living organisms by fulfilling two essential roles, potential source and transport medium of toxic chemicals (Jönsson et al., 2018). Textile polymers are in contact with catalysts, pesticides, solvents, dyes, and additives such as phthalates and nonylphenol when converted from raw material and manufactured into final garments (Cesa et al., 2017; Rios Mendoza et al., 2018; Stanton et al., 2019; Athey et al., 2020). These compounds may not be chemically attached to the polymer matrix and are susceptible to release (Wright and Kelly, 2017), acting as a source that could cause unknown collateral effects to living organism after being ingested, inhaled or in contact (Cesa et al., 2017; Rios Mendoza et al., 2018). FF regularly release additives in aquatic environments, which have been reported as lethal and carcinogenic (Singh et al., 2020).

As carriers, FF can also be nourished from toxic substances of the surrounding environment such as contaminants from WWTP and landfills, including heavy metals, non-polar persistent organic pollutants (POPs), polycyclic aromatic hydrocarbon (PAH), endocrine-disrupting compounds (EDCs), polychlorinated biphenyls (PCBs), among others (Cesa et al., 2017; Catarino et al., 2018; Gasperi et al., 2018; Singh et al., 2020). FF possess a high surface-area-to-volume which increase the risk of bioavailability of toxic compounds since they could accumulate in animal tissues after ingested, providing a possible pathway for cells to interact with external chemicals (Browne et al., 2011; Cesa et al., 2017; Henry et al., 2019).

Exposure to microplastics and fragmented fibres of aquatic organisms, including phytoplankton, zooplankton, mussels, an assortment of fish and bird species, turtles, penguins, and seals, among others, has been studied. A detailed chart with the hazardous effects in different marine animals and humans on exposure to FF can be consulted in (Singh et al., 2020). After being mistakenly ingested, fibres enter the digestive tract and could reach the circulatory system (Henry et al., 2019). It is reported that ingestion and exposure to FF in marine animals cause toxicity in liver and heart, neurotoxicity, negative impact in eating behaviour, stomach damage, obstruction in gills and disruption of the endocrine system, harmful effects on reproduction behaviour, starvation, and malnutrition (UNESCO, 2017; Chatterjee and Sharma, 2019; Singh et al., 2020). It is estimated that around 100,000 marine animals and over a million sea birds die annually due to plastic debris ingestion (UNESCO, 2017). Moreover, it is believed that the number of FF in the sea will overpass that of fish by 2050 (Singh et al., 2020).

The degradation of FF in the marine environment depends upon weathering factors. In contrast to terrestrial environments, the degradation rate is prolonged, particularly for synthetic FF due to the lack of UV-ray, low oxygen concentration, and low temperatures (Singh et al., 2020). Since polyester fibre density is higher than seawater, it is suggested that the FF of this polymer may settle in the seabed. The biological processes that occur in the deep ocean are less investigated than the rest (Barnes et al., 2009; Cesa et al., 2017), even though the presence of FF in oceanic sediment is of vast magnitude (Singh et al., 2020). The limitation of gas exchange and proliferation of non-native species are part of the research limitations still to be studied (Barnes et al., 2009; Cesa et al., 2017).

Recently it has been suggested that FF pollution on terrestrial environments may be over 20 times higher than marine locations (Singh et al., 2020). Research on microplastics in agricultural soil in southwestern China showed that 100% of the samples contained plastic debris, and 92% were fragmented fibres from them. Direct irrigation of soil with washing effluent from laundry is pointed out as a likely source for fibrous contamination (Zhang and Liu, 2018). Microplastics presence in crops and agricultural land has been reported to cause a reduction in the natural soil's sorption (Hüffer et al., 2019) and modify the soil properties, including aeration, moisture content, and aggregate stability (Lozano and Rillig, 2020). FF in soil have an initial interaction with the biota, which may modify the geo-chemically and biophysical atmosphere leading to environmental toxicity (Singh et al., 2020). These particles may be attached or consumed by earthworms in the soil surface and carried out into depth layers, where they can affect biota and organisms of deeper layers and could even reach groundwater (Rillig et al., 2017).

It has been reported that MP modify the relationship mechanisms of water retention, porosity and aggregation that impact the soil biophysical functions. Fragmented fibres shed from textiles may entangle aggregates and mould due to their linear and flexible structure (Forster et al., 2020). These particles immersed in soil reduce the soil bulk density and increase the soil macroporosity, improving aeration, which ultimately facilitates the penetration of the roots in the soil matrix and increases the root biomass. The effects of modification in root morphological traits have not been long-term studied and have different impacts depending on the plant species. FF may affect plant community evenness and performance, which could promote the dominance of invasive species and impact plant community resistance or resilience (Lozano and Rillig, 2020).

The levels of FF reported in the atmosphere are alarming. The magnitude of textile FF pollution deposited on household surfaces is similar as emitted during the laundry processes (Sundt et al., 2014). On the contrary of particles generated through textile washing, FF released to the atmosphere cannot be captured (Q. Zhang et al., 2020) as occur in WWTP. Hence, atmospheric fallout may act as a transport of textile particles to reach aquatic environments, agronomic systems, and even the most isolated locations (Dris et al., 2016; Stanton et al., 2019; Q. Zhang et al., 2020). FF pollution can be transported at a fast pace over long distances from the source to remote environments (Henry et al., 2019). The research carried out in indoor spaces shows how airflow affects the fluctuation, migration and resuspension of particles, which can also affect FF found in outdoor environments since air acts as a FF vector for contamination (Q. Zhang et al., 2020).

The ecological effects of plastic alternatives, including natural and regenerated fibres, are yet to be established. Natural fibres are placed as dominants – 93.8% in freshwater and airborne samples (Stanton et al., 2019) and 33.3% ingested by demersal sharks (Parton et al., 2020), in recent publications. Cotton fragmented fibres are ubiquitous encountered (De Falco et al., 2020); for instance, indigo denim cotton FF were located in remote regions sediments. Hence, the persistence of these fibres is durable enough to undergo long-range transport from sources and awakes biota concern (Athey et al., 2020). The biodegradation of cellulosic natural and extruded fibres in aquatic environments and their chemical sorption behaviour still need to be confirmed (Ladewig et al., 2015). A slow decomposition created by the hostile environmental and weathering conditions discussed previously is suggested (Chen and Jakes, 2001; Andrady, 2017). Whereas other publications hypothesise a standard degradation rate (Ladewig et al., 2015; Zambrano et al., 2019). A low decomposition may involve environmental consequences similar to synthetic FF, while a faster pace could represent a potential route for hazardous chemicals attached to the fibre (Stanton et al., 2019). The presence of certain fibre types could also be linked to the sample preparation protocols as certain polymer types are susceptible to degradation during sample preparation for separating the collected FF/MP from collect samples in the environment.

6.2 Impact on human health

MP and FF can potentially also pose a human health risk. Inhalation, ingestion and dermal contact are the primary routes for MP and FF entering the human body (Acharya et al., 2021). The uptake of synthetic FF by humans represents exposure to the plastic particle and the chemicals associated with them (Acharya et al., 2021). Natural and regenerated FF are also associated with dangerous chemicals and may represent a similar hazard (Ladewig et al., 2015). The human effects for plastic consumption gather three main points: particle toxicity, chemical toxicity, and pathogen-parasite vectors (Acharya et al., 2021).

Fragmented fibres have been found in products sold for human consumption, including drinking water, sea salt, seafood, sugar, and beer, with clothing as a likely source (Liebezeit and Liebezeit, 2014; Singh et al., 2020; Sol et al., 2021). Additionally, these particles may enter into human food supply after being ingested by marine organisms (Hartline et al., 2016; Pirc et al., 2016; Cesa et al., 2017; Rios Mendoza et al., 2018). For instance, shellfish represents the principal source of FF in a dietary pathway to human exposure (Wright and Kelly, 2017). An average of 7,500 synthetic FF per year has been estimated to be ingested by an adult only via tap water (Sol et al., 2021). MP found in human stool evidence the ingestion and passage through the gastrointestinal tract of these particles (Mohamed Nor et al., 2021). Even if humans do not directly consume fibres, the chemicals detached from FF may be transferred to the upper nutrients rank through the food chain. Moreover, FF can also carry parasites and other pathogenic microorganisms (Singh et al., 2020).

The occupational health of labourers from microfibres manufacturing plants has been studied. Chronic exposure to FF is reported to cause lung infection and inflammation, an interstitial lung disease that induces dyspnoea, coughing and reduces lung capacity. Also, it has been related to cause stomach, liver, kidney and brain damage, genotoxicity, cardiovascular problems, oxidative stress, and apoptosis (Bahners et al., 1994; Eschenbacher et al., 1999; Boag et al., 1999; Wright and Kelly, 2017; Prata et al., 2020). Although airborne FF concentration in flocking areas (a textile manufacturing step for a particular type of products) is higher than particles found in the environment (up to 1,000,000 fibres/m^3) (Bahners et al., 1994), this evidence suggests the potential hazard of FF pollution to trigger localised biological responses and endanger human health (Wright and Kelly, 2017).

Cellulosic and synthetic respirable FF deposited in lung tissues and found in malignant human lung specimens is also documented (Pauly et al., 1998; Thompson et al., 2004; Dris et al., 2017; Cesa et al., 2017; Prata et al., 2020; Singh et al., 2020). Moreover, plastic nanoparticles were found in all sampled specimens of human organs, including lungs, spleen, kidneys, and liver (Tangermann, 202AD). Chemicals attached to FF such as phthalates, BPAs, POPs, PCBs, and EDCs may have hazardous effects on human health such as injury in the intestines, liver and kidneys, infection in the blood, breast cancer, and hormonal balance in female reproductive systems (Chatterjee and Sharma, 2019; Singh et al., 2020; Acharya et al., 2021). Furthermore, the size and hydrophobicity of FF facilitate their route through the placenta and blood-brain barrier to reach the gastrointestinal tract and lungs (Smith et al., 2018). The translocation of FF from the digestive system to the circulatory system affects the immune system and deposit on secondary organs (GESAMP, 2016). A detailed diagram of predicted pathways for the uptake of MP in the lung and gastrointestinal tract can be consulted in (Wright and Kelly, 2017).

The repercussions of chronic exposure to fragmented fibres when ingested and inhaled are in an early stage of research and yet need to be established (Singh et al., 2020). A MP exposure assessment and a chemical exposure via MP assessment was performed about dietary and inhalation intake by (Mohamed Nor et al., 2021). The results show that the highest rates come from the air and the lowest from fish. Moreover, it is suggested that smaller-sized MP (1−10μm) may remain throughout a lifetime in the body. Despite the high amount of MP intake, chemical leaching from MP and particles volume did not reflect a substantial impact on human health; hence the proportion of MP seems insufficient to cause a chemical change in the body (Mohamed Nor et al., 2021). However, increased pollution levels and future research could evidence MP and fibrous debris as a clear threat in the human health (SAPEA, 2019).

The impact of microplastics and fragmented fibres in the environment and their potential risks for human health are broad topics other researchers have studied. Since they are not included in the scope of this study, it is suggested to review the work of (GESAMP, 2015; GESAMP, 2016; Wright and Kelly, 2017; Gasperi et al., 2018; Prata, 2018; SAPEA, 2019; Henry et al., 2019; Prata et al., 2020; Singh et al., 2020; Acharya et al., 2021; Mohamed Nor et al., 2021; Palacios-Mateo et al., 2021).

6.3 Aquatic degradation of fragmented fibres

As discussed throughout this document, one of the major concerns of FF released from textiles during washing processes is their fate and persistence in aquatic environments. Available literature regarding the degradation of fragmented fibres in aquatic environments is scarce. The slow degradation of plastics and prolonged persistence time periods of synthetic FF highlight the risks of these contaminants (Gaylarde et al., 2021).

There is no single concept to define "degradation". In the polymers, including MP and FF, it may refer to a chemical change that affects their properties, a breakdown into smaller pieces or mineralization of the material (Andrady, 2017). Aquatic degradation of MP and FF can only be achieved by mineralization of the polymers into small molecules of CO_2 and H_2O (Andrady, 2017). In the same way, the aquatic degradability of materials is not straightforward to measure or certify (Zambrano et al., 2019). This leads to different results from publications reporting the aquatic degradability of FF, including a qualitative analysis of degradation signs in the polymer surface (Zambrano et al., 2021b; Sait et al., 2021), changes in polymer composition with FTIR profiles (Zambrano et al., 2020; Sait et al., 2021), and comparison of oxygen uptake of the material in a closed system against the theoretical oxygen demand (Zambrano et al., 2019; Zambrano et al., 2020; Zambrano et al., 2021a).

Research by Zambrano et al., (Zambrano et al., 2019; Zambrano et al., 2020) compared aerobic aquatic biodegradation between activated sludge at a low concentration from WWTP, lake water and seawater of microcrystalline cellulose (MCC), cotton, rayon, 50-50 polyester-cotton blend and polyester fabrics. The findings of these studies, shown in Figure 17, report broadly that cellulosic fibres were degraded while polyester fibres were not. The biodegradation rate of samples was found to be MCC > Cotton > Rayon > Polycotton > Polyester in all inoculums. However, environmental conditions impact the degradation percentage reached by the fibres after 35 days. For instance, cotton and rayon present a 77% degradation rate in lake water, while only 49% in seawater. Similarly, the polycotton blend degrades 33% in lake water and only 14% in seawater, while polyester did not degrade in any inoculum (Zambrano et al., 2019; Zambrano et al., 2020).

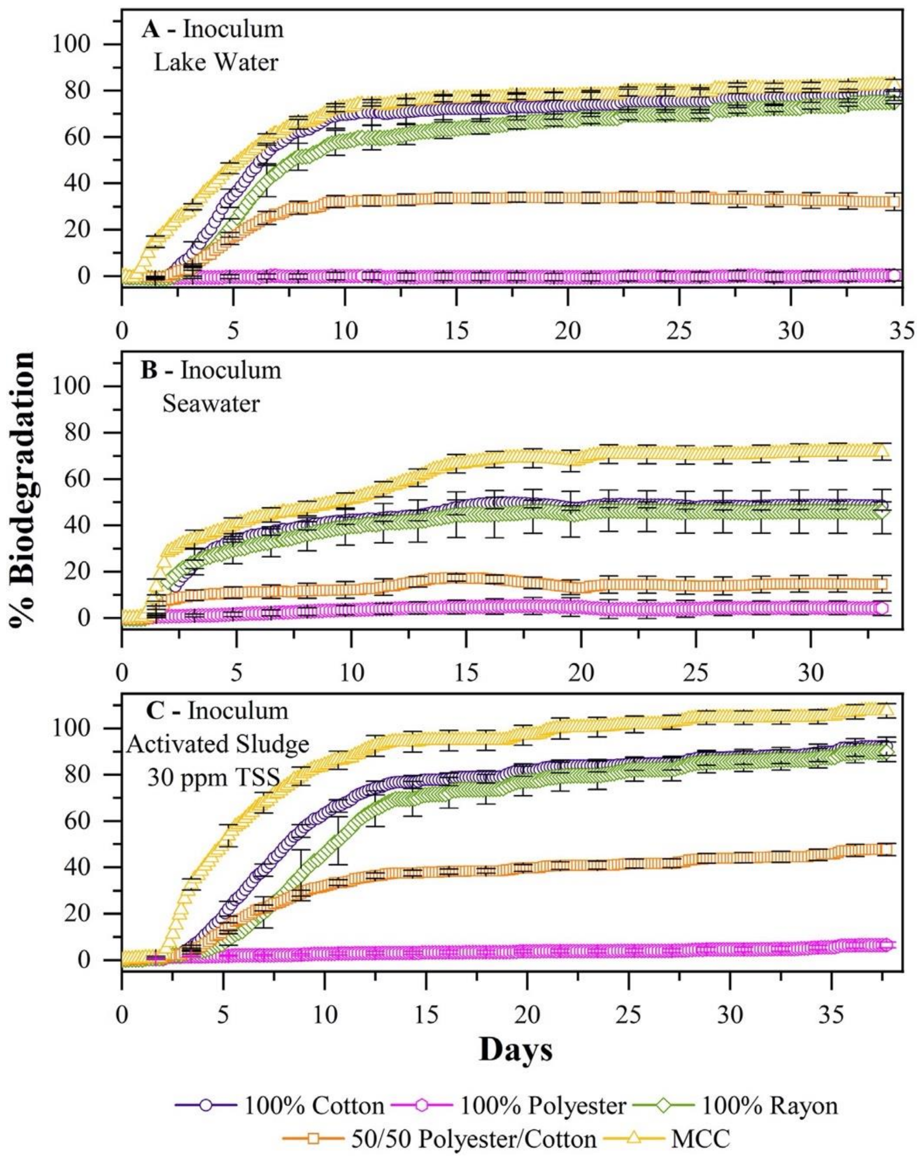

Figure 17 Biodegradation curves of the textile yarns based on the oxygen uptake of the system versus the theoretical oxygen demand calculated for each yarn, corrected for nitrification reactions. A – inoculum: lake water from Lake Raleigh (Raleigh, NC), method ISO 14851 (N = 3). B – inoculum: seawater from Fort Fisher Park (Wilmington, NC), method ASTM D6691 (N = 4). C – inoculum: activated sludge from the Neuse River WWTP, 30 ppm of total suspended solids (TSS), method ISO 14851 (N = 4). The error bars represent the standard error of the mean. (Zambrano et al., 2020, p.5).
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Zambrano, M.C., et al., 2020. Aerobic biodegradation in freshwater and marine environments of textile microfibers generated in clothes laundering: Effects of cellulose and polyester-based microfibers on the microbiome. Marine Pollution Bulletin. 151, p. 5. Copyright 2019, Elsevier Ltd., DOI: 10.1016/j.marpolbul.2019.110826.

On the other hand, photodegradation of synthetic FF in freshwater and seawater has also been investigated (Sait et al., 2021). FF from PET, polyamide (PA), and polyacrylonitrile (PAN) were exposed to UV in freshwater and seawater for ten months. Findings show changes in surface morphology on PET and PA fibres and chemical alterations of PA. However, PAN presents no significant degradation after the same period. Similar results were observed for freshwater and seawater. In this regard, salinity only influences the leaching of chemical additives from the fibres (Sait et al., 2021). Further research of plastics biodegradation is still needed (Gaylarde et al., 2021).

The degradation success of cellulosic over synthetic fragmented fibres does not exempt them from being hazardous to the environment. The work by Zambrano et al., (Zambrano et al., 2019; Zambrano et al., 2020) employed raw yarns without finishing agents. Hence, the samples do not represent commercial textiles used in the fashion industry as colouration and chemical/mechanical finishing processes are generally applied. Recent research analysed the impact on the aquatic biodegradability of the most common treatments applied on cotton textile fibres (Zambrano et al., 2021a). The results show that some of the finishes significantly affect the biodegradability of polymers, especially those increasing the crosslinking and hydrophobicity of the fibre. As a final result, fibres with all tested treatments are expected to biodegrade. However, their degradation rate is significantly affected, increasing the persistence of these pollutants in the environment. Moreover, active compounds of some treatment formulations may not biodegrade and could reach the environment as the ultimate fate (Zambrano et al., 2021a). Hence, the degraded fibres could act as a delivery vehicle for any hazardous chemistry. However, these merits further research in reference to exposed concentration and commonly applied colouration and finishing agents.

7. Summary and Future direction

In recent years, the quantification of fragmented fibres (FF) from textiles during laundering has extensively been studied. In the absence of standard/harmonised methods to quantify the release of FF during dry and wet environments, the extent of release of FF cannot be directly compared from different studies. In addition, the detailed data of fibre/yarn/fabric properties and material history is generally not reported. There is a need for systematic evaluation of the impact of essential material and structural variables on the release of FF during manufacturing, use and service of textiles. The interconnections between sources and sinks of FF are not fully understood, and the dispersal route of the FF needs to be elucidated.

The threat of microplastic (MP) pollution to our ecosystem is well established, and remedial measures are challenging given the ubiquity of MP. Improved plastics with reduced environmental impact, plastic recycling and reduced consumption are vital. Still, the release of fragmented fibres (FF) from textiles remains a challenge since their production is non-intentional. The intervention strategies to produce novel textile structures that stop the release of fragmented fibres from textiles would present at-source solutions. In line with this, our current research (funded by Engineering and Physical Sciences Research Council, UK Research and Innovation) is in progress to fundamentally understand fibre damage leading to fibre fragmentation. In particular, the project will explore physical and chemical strategies to manufacture novel textiles which stops the release of fragmented fibres.

The degradability of FF is another topic of concern and future work. The conditions needed for natural, regenerated, and synthetic fibres to degrade in terrestrial and aquatic environments and the consequences of the chemical additives leaching from the fibre debris demand further investigation. Moreover, the remarked abundance of natural and regenerated FF over the global mix of FF evidence the gap of research on the degradability and persistence of these cellulosic fibres. Therefore, it is recommended to carry out future work for a better understanding and an effective contribution towards textile sustainability.

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