

Review

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# Microplastics in the atmosphere: a review

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## Abstract

Microplastics (MPs) are ubiquitous environmental contaminants of considerable persistence and have been a global concern for the past decade. Recently, atmospheric MPs have gained attention. The presence of MPs in the air has been reported from different regions and in air masses over water bodies, demonstrating MPs' capability of long-range transport and wide spatial distribution away from their source of origin. This review of atmospheric MPs raises questions about the validity and legitimacy of approaches adopted for assessing MP in indoor and outdoor aerosols. The review also provides insight into active and passive sampling techniques and draws attention to the use of the data produced. MP abundance in the atmosphere varies widely among studies due to the disparities in methods employed and the heterogeneity in reporting, making comparisons across spatio-temporal domains infructuous. This review also highlights the paucity of data on atmospheric MPs, and the eminent need to harmonize the methodology for generating a useful comparable dataset that can be used for human health risk assessments.

**Keywords:** Airborne microplastic, active sampling, passive sampling, indoor, outdoor aerosol, atmospheric fallout, dust

## INTRODUCTION

Plastic waste has become a menace on a global scale. The environmental concern emanating from this waste



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stream has generated a great deal of scientific interest. With 8.3 billion metric tons of plastics produced between 1950 and 2015, 75.9% was dumped as waste<sup>[1]</sup>. By 2050, it is predicted that about 12,000 million metric tons of plastic debris will be present in landfills<sup>[2]</sup>. Approximately 8.75 million metric tons of plastic enter our ocean each year from land-based sources<sup>[3]</sup> and comprise 61%–87% of marine litter<sup>[4,5]</sup>. The problem is further exacerbated when plastic breaks down into smaller microplastics (MPs) (1  $\mu\text{m}$  to 5 mm) and nanoplastics (< 1  $\mu\text{m}$ ) due to biotic and abiotic degradation<sup>[6,7]</sup>. Reports suggest that > 5.25 trillion plastic particles sized between 0.33 and 4.75 mm are currently floating in the world's oceans<sup>[8]</sup>. A more recent assessment by Uddin *et al.*<sup>[9]</sup> indicates that globally treated effluent from wastewater treatment plants contributes  $1.47 \times 10^{15}$  MPs per annum, while discharge of untreated wastewater disposal adds another  $3.85 \times 10^{16}$  MPs annually to the aquatic environments. Microplastics are found in all environmental matrices of the aquatic and terrestrial environments. The observations are well documented in 6551 published articles up to 3 May 2021.

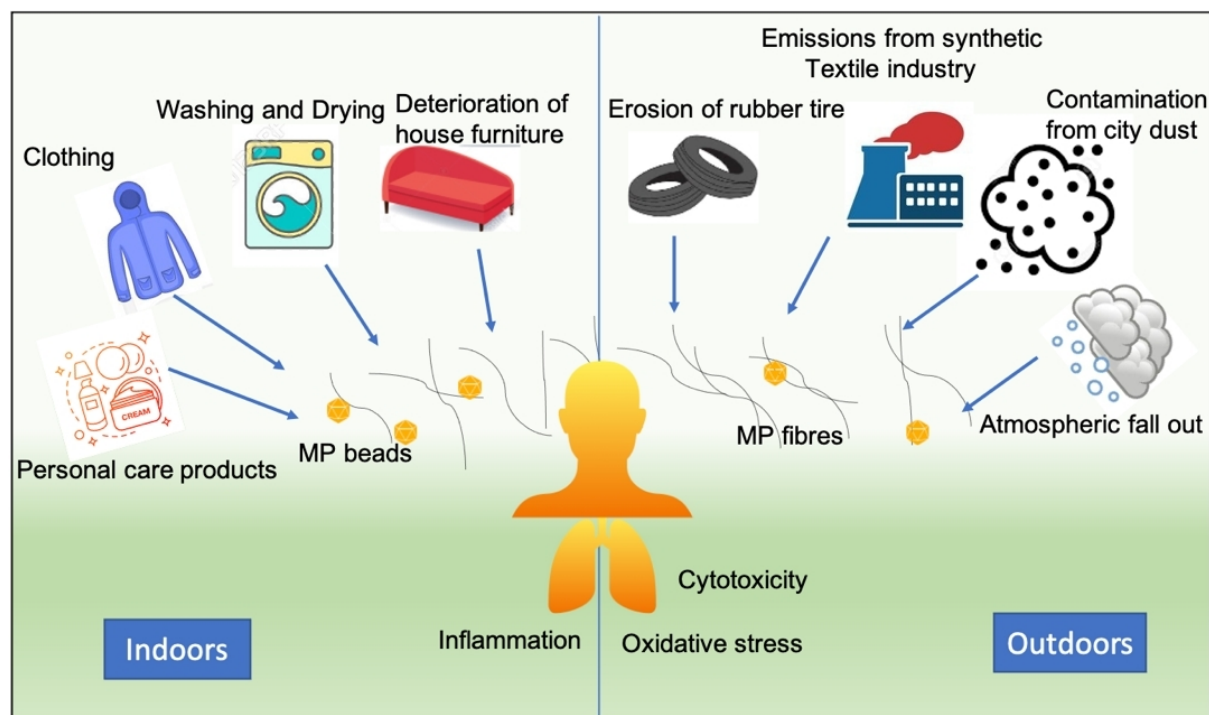
A few studies, just over four dozen thus far, have looked at MPs in atmospheric aerosols. Aerosols are being identified as a significant pathway for inhalation of MPs by humans<sup>[10–14]</sup>. A recent study suggested that MPs can be transferred up to a distance of some 95 km<sup>[15]</sup>. There are 48 studies (as of November 2021) reporting MPs in dust, aerosols, wet and dry atmospheric deposition, snow, *etc.*<sup>[11,15–26]</sup>. MPs in the air are released from wear and tear of clothing material<sup>[27–29]</sup>, by washing and drying<sup>[30]</sup>, erosion of synthetic rubber tires, deterioration of house furniture, emissions from the synthetic textile industry, emissions from vinyl chloride and polyvinyl chloride (PVC) industries, and contamination from city dust<sup>[18–21]</sup> [Figure 1]. In addition, substantial quantities of plastics are being burned in open landfills on a daily basis, which results in the volatilization of various harmful compounds that inevitably integrate into the atmospheric aerosol<sup>[31,32]</sup>.

Considering the finite information available on this vital issue of MPs in aerosol, it has only limited usefulness. Although these studies suggest the omnipresence of MPs in aerosol, they have often used non-comparable techniques for the sample collection, processing, and identification of microplastics in indoor or outdoor aerosol, dust, and snow. The need to standardize these methodologies for making data comparable across spatio-temporal domains seems eminent for improved understanding of the human health implications in relation to inhalation exposure to MPs<sup>[26]</sup>. This review intends to highlight these issues. It also discusses the lack of information on the inhalable and respirable fraction of MPs in most reported literature.

## HEALTH IMPLICATIONS OF MICROPLASTICS

Studies have reported localized inflammation and genotoxicity among humans due to inhalation of MPs<sup>[11,24,33]</sup>. Contrary to popular wisdom, where we expect smaller MPs to affect the human pulmonary system, in one study, fibers up to 250  $\mu\text{m}$  were detected in the human lung<sup>[34]</sup>. Other studies reported fibrous MPs could avert clearance mechanisms and persist in the lungs<sup>[34,35]</sup>. Some others have observed reduced lung capacity, coughing, and breathlessness<sup>[36–38]</sup>. Pauly *et al.*<sup>[34]</sup> found fibers in 99 of 114 malignant lung specimens. On the other hand, a recent study indicated that MP deposition is more likely to occur in the upper airway tract (i.e., nose, mouth, and throat) when inhaled and would reach the gut if ingested<sup>[7,25]</sup>. Fine MPs are believed to translocate to the circulatory system and other organs<sup>[33,39]</sup>.

Studies looking at MP inhalation by humans are rare<sup>[12,34]</sup>, and equally scarce are the inventories to make reliable human health risk assessments. A study conducted in Australia revealed mean MP inhalation rate ( $0.2 \pm 0.07$  mg/kg-B<sub>w</sub>/year and  $12,891 \pm 4472$  fibers/year) to be highest ( $0.31$  mg/kg-B<sub>w</sub>/year) for the below 0.5 year age group<sup>[24]</sup>.



**Figure 1.** Sources of microplastics in the atmosphere and their health implications.

The primary questions are as follows: what are the reasons for the pulmonary diseases from MP inhalation? Is it the presence of MPs (polymers themselves) or the many additives, dyes, and pigments which are part of plastic production? Conversely, is it several factors together? Could these pulmonary diseases be more than inflammation, which is documented<sup>[13,23]</sup>? More than 4000 compounds are used in plastic manufacturing, many of which are not firmly bound to the polymer and make them susceptible to being released into the surrounding environment<sup>[13]</sup>. Could these additives lead to other health effects such as reproductive toxicity, carcinogenicity, and mutagenicity<sup>[13,24,40,41]</sup>? Therefore, the ability for sorption and toxicity of MP in other environmental matrices has attracted the attention of researchers across the world<sup>[4,10,19,42-59]</sup>. Airborne MPs may act as a vector for transport of the particle reactive pollutants, including polycyclic aromatic hydrocarbons, metals released from traffic emissions, and microorganisms to human lungs<sup>[11]</sup>. The inhalation of these MPs thereby poses a potential threat to human health<sup>[60]</sup> and may cause chronic symptoms<sup>[61]</sup>. A suite of contaminants such as plasticizers, flame retardants, antimicrobial agents, and Bisphenol A (BPA) can be released during MP degradation<sup>[62,63]</sup>. A recent study presented the occurrence of BPA in the urban atmosphere of the city of Córdoba, Argentina<sup>[64]</sup>, with a concentration gradient of: industrial area > urban area > university campus. The results of this study are comparable to those reported in other cities of the world<sup>[65]</sup>. Other contaminants such as phthalates are also present in aerosol<sup>[66]</sup>. However, the magnitude of harm these MPs may cause to human health has not yet been well established<sup>[17,41,66]</sup>. Correlation can be drawn with other published studies where the fibers deposited in terminal bronchioles, alveolar ducts, and alveoli resulted in chronic inflammation, granulomas, or fibrosis<sup>[41]</sup>. In another study, fibers of asbestos, silica, and diesel exhaust were related to secondary genotoxicity due to continuous formation of reactive oxygen species<sup>[67]</sup>. Increased incidence of interstitial lung disease was associated with inhalation of MPs; however, these are based on assays where non-environmental concentrations were used<sup>[16]</sup>, raising concerns of whether we are establishing realistic correlations.

## Sample collection

Studies on atmospheric MPs have used different techniques for sample collection, i.e., active and passive sample collection techniques. The sampling techniques used for MP collection in indoor and outdoor aerosols are summarized in [Supplementary Table 1](#). Out of the 38 outdoor studies, 21 have used passive sampling and 17 have collected samples using an active sampler. In indoor studies, active samplers were used in six studies and passive in five.

### *Active sampling*

The active collectors operate on drawing the aerosol sample using some device, commonly a vacuum pump. These pumps operate with different capacities, some drawing hundreds of liters of air per minute (usually regarded as high volume air samplers), others drawing a couple of liters per minute (low volume air samplers)<sup>[14,68]</sup>. The advantages of the active sampling technique are a known volume of aerosol is drawn, and, usually, a large volume of sample can be collected relatively quickly. It will produce comparable data across the spatio-temporal domain. The technique also enables to work out concentrations per unit volume. Active sampling is a well-established and accepted technique that has been adopted for years to monitor atmospheric pollutants, particularly microbes, carbon, lead, and mercury<sup>[69,70]</sup>. The active samplers require a power source, some basic skills to operate them, and are more expensive to procure and operate compared to passive samplers. Most of the active samplers deployed use a media/collection substrate; few authors have mentioned use of quartz, glass-fiber, or cellulose filter. The question remains on retrieval of an MP sample, as procedural blanks are not always done. Our experience shows significant cross-contamination and loss of sample by using a filter. In our experiments, we tried using a six-stage aluminum impactor for collecting size-fractionated aerosol sample, which gives the most realistic results we have obtained thus far.

### *Passive sampling*

Passive sampling usually provides a time-integrated sample collected via dry and wet atmospheric deposition at a specific location such as deposition over a surface (MP settling). These samplers come in various forms such as simple funnels, rain gauges, bottles, and planar surfaces that allow atmospheric deposition [[Figure 2](#)]. While these samplers cannot be used to assess the concentrations per unit volume of aerosol, they can give an idea of MPs per unit area. Passive samplers are cheaper and easier to deploy but are highly influenced by local weather phenomena, elevation, and angle of deployment. Besides, they cannot be used for a very short period of deployment from minutes to a few hours. Hence, the choice of sampler depends on the scientific objective(s), whether one is interested in long-term mass accumulation rates of MPs or MP concentration in aerosol for health risk assessments.

## Sample type

### *Outdoor air*

Out of the 48 studies on MPs in atmospheric aerosol, 37 targeted the outdoor aerosol [[Figure 3](#)]. Twenty-one used various passive sampling devices that included funnel with a collector bottle, rain gauge for aerosol collection, broom and spoon for collecting dust samples, and glass jar for collecting snow<sup>[15,18,19,71-75]</sup>. Four outdoor studies collected dust samples from street and apartment buildings using a broom and pan<sup>[16,76-78]</sup>. Such samples are usually not pure aerosol samples. Seventeen studies used active samplers where a known quantity of air was passed through the filters or capturing devices<sup>[16,20,22,79-82]</sup>. Most of these samplers were low volume samplers drawing between  $2.07 \times 10^2$  mL min<sup>-1</sup> and  $3 \times 10^5$  mL min<sup>-1</sup> air.

### *Indoor air*

There have only been 11 studies on MP in indoor air up until October 2021. A summary is included in [Figure 3](#) [[Supplementary Table 1](#)]. Six studies sampled indoor air using active samplers<sup>[12,20,22]</sup>, and four collected indoor dust from non-carpeted apartment floors<sup>[78,83]</sup>, while another study collected atmospheric

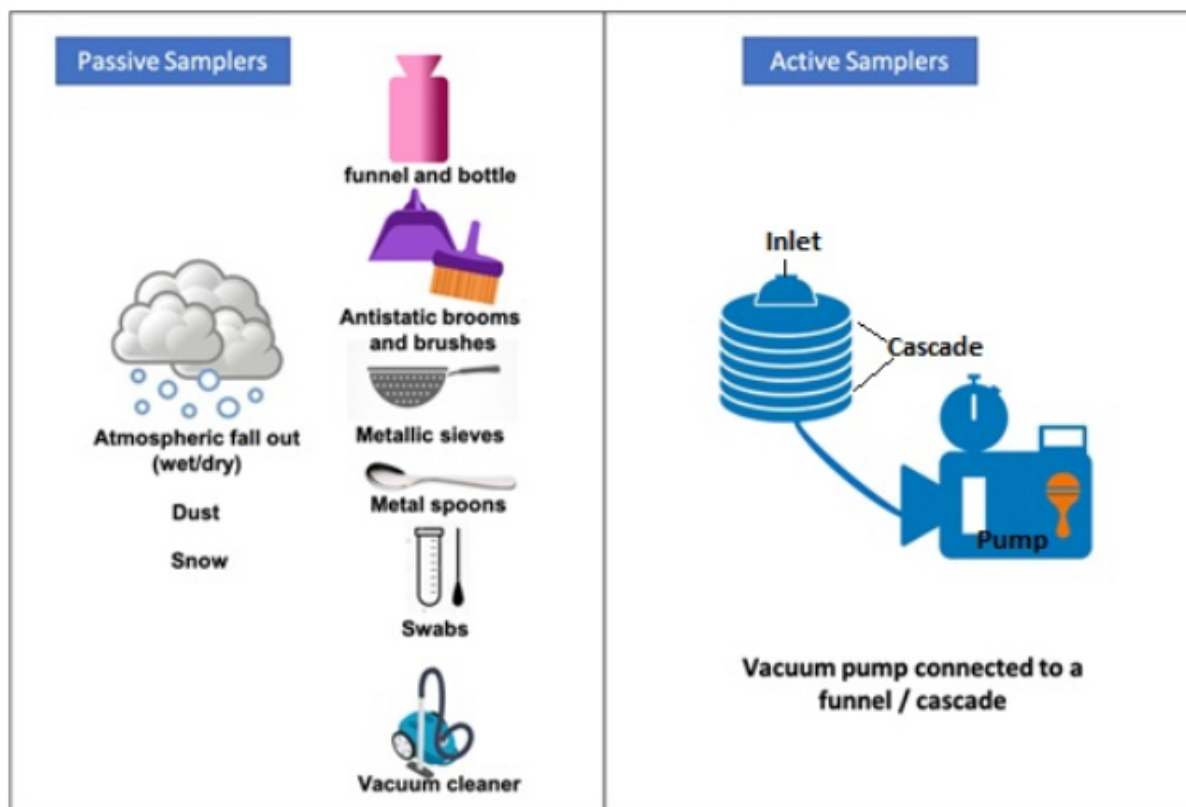


Figure 2. Modes and means of active and passive sample collection.

fallout passively at head height from households in the UK<sup>[84]</sup>. Different types of active samplers have been used for collecting indoor air samples. Dris *et al.*<sup>[20]</sup> used a sampler that drew aerosol @ 8 L min<sup>-1</sup> from two apartments and an office sampling a total volume of 2 and 5 m<sup>3</sup>. In a Danish study, an extremely complicated active sampling was used that involved a manikin simulating the human lungs<sup>[12]</sup>. Although this was an excellent approach, it is still not a standard method employed by people undertaking indoor air quality assessments, and thus cannot be compared to other assessments.

### Sample preparation

Sample preparation is a critical step for segregating MP from environmental matrices. It becomes more important when a filter/substrate is used for aerosol collection. In a recent procedure developed by our group, we started collection of size-fractionated aerosol samples using an aluminum six-stage cascade impactor, without any substrate. We found that this process minimizes the processing step to microscopic examination of fibers and removal using a static charge. The aerosol is a complex mixture of inorganic (mineral), organic matter (pollen, fungi, microbes, soot, *etc.*), and MPs. The removal of MPs from this complex mixture often entails chemical processing that includes digestion of organic matter using potassium hydroxide (KOH), sodium hydroxide (NaOH), hydrogen peroxide (H<sub>2</sub>O<sub>2</sub>), enzymatic methods, or sodium hypochlorite (NaClO)<sup>[15,19,85-89]</sup>. Some researchers have done acid digestions using hydrochloric acid (HCl), nitric acid (HNO<sub>3</sub>), and sulfuric acid (H<sub>2</sub>SO<sub>4</sub>). Different studies have used various processing steps, namely filtration of sample on a glass or quartz fiber filters<sup>[18,19,24,73,78,81]</sup> and digestion with 30%-35% H<sub>2</sub>O<sub>2</sub> for days<sup>[15,16,22,23,75-77,79]</sup>. Some researchers have indicated that the use of 30% H<sub>2</sub>O<sub>2</sub> can lead to significant deterioration of MPs by decolorization, making them further difficult to distinguish from some natural fibers<sup>[15,88,90,91]</sup>. A standardized protocol that can be effective for MP segregation is summarized in Figure 4.



**Figure 3.** The numbers and types of samplings used in different indoor and outdoor studies.

This review of published studies highlights that authors have used different solutions such as NaCl, sodium iodide (NaI), and zinc chloride ( $ZnCl_2$ ) for the density separation of MPs in the aerosol. Synthetic clothes, carpets, and upholstery are the most likely source of MPs in the air<sup>[29]</sup>; the likelihood of polyester/polyamide fiber domination in air samples is high. However, using NaCl for density separation will only float a fraction of the MP load in aerosols, while a fraction of polyamide (nylon) may remain non-buoyant. **Figure 5** presents the polymer densities with respect to the commonly used solutions for density separation.

### Identification and characterization of microplastics in aerosol samples

#### Identification of airborne MPs

Several methods of microplastic identification are reported. Some studies used a scanning electron microscope with an energy dispersive X-ray analyzer at a voltage of 3.0 kV to identify the morphology of potential MPs. A very cost-effective test is the hot needle technique, in which particles can be tested to

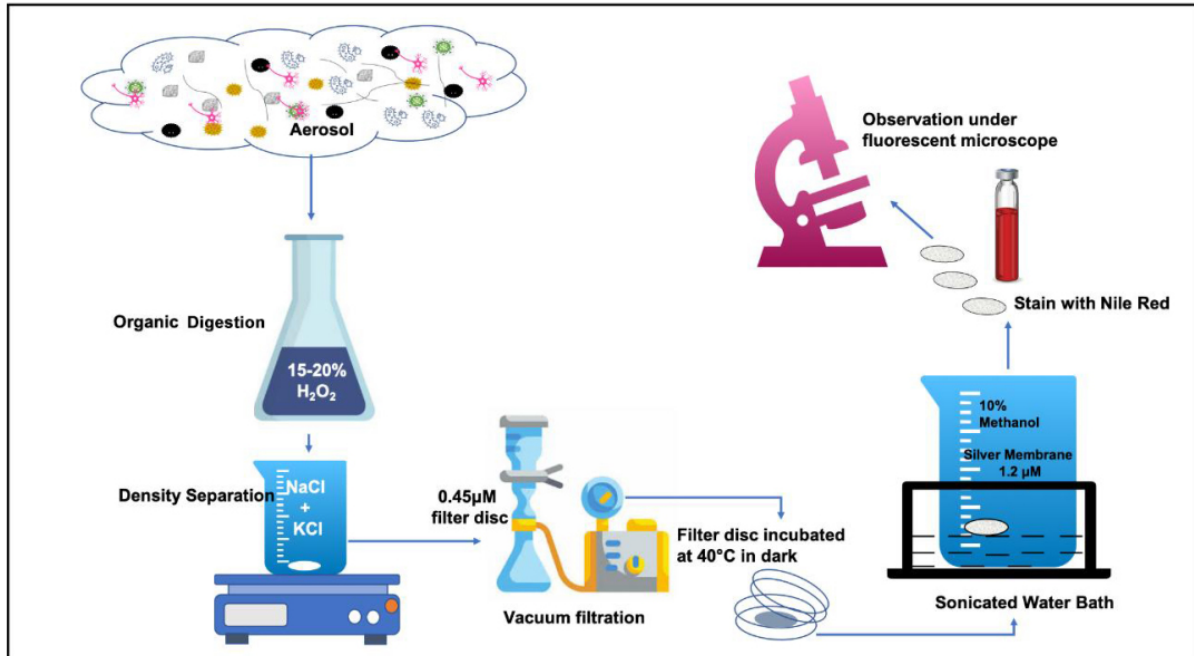


Figure 4. Pictorial representation of the steps involved in aerosol sample processing and microplastic identification.

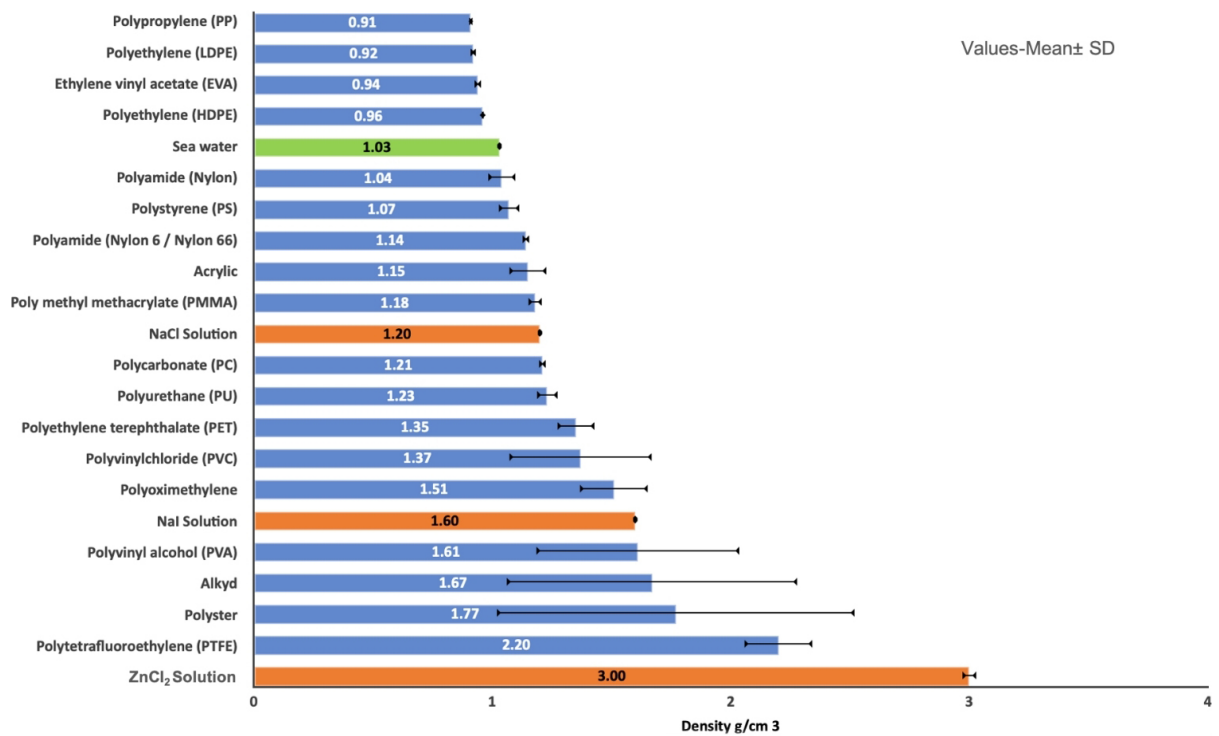


Figure 5. Polymer densities and corresponding solutions (modified after Uddin et al.<sup>[92]</sup>). Blue bars represent the polymer densities, orange bars are for the respective solutions used for density separation, and the green bar shows the density of seawater.

determine whether they are natural fibers or polymers<sup>[93]</sup>. The staining of segregated particles with Nile red and observation under a fluorescence microscope is another method. Several researchers have limited the

observations to microscopic investigation without staining. However, generally, authors have used a two-tier process for MP identification. An initial visual examination using a microscope where the large size MPs ( $> 500 \mu\text{m}$ ) can be picked up based on the absence of cellular structure and thickness consistency along with their length, relatively homogenous coloring, and transparency is followed by a hot needle test and/or sometimes using staining using Nile red and observation under a fluorescence microscope. Sometimes these are followed by polymeric characterization through sophisticated spectroscopic methods such as micro-Raman spectroscopy ( $\mu$ -RS), attenuated total reflectance Fourier transformed infrared (ATR-FTIR), or micro-Fourier transformed infrared spectroscopy ( $\mu$ -FTIR).

#### *Characterization of airborne MPs*

Polymeric characterization of the MPs has been attempted by several researchers using techniques such as  $\mu$ -FTIR, ATR-FTIR,  $\mu$ -RS, high-performance liquid chromatography-tandem mass spectrometry (HPLC-MS-MS), and thermal desorption coupled with thermogravimetric analyses [Figure 6]. Three studies from Iran<sup>[16,76,77]</sup> only used microscopic examination to identify MPs, whereas, in a recent study, polymers were identified using  $\mu$ -RS<sup>[94]</sup>. Some other researchers in Hamburg, Germany<sup>[74]</sup>, Antarctica<sup>[95]</sup>, Ireland<sup>[96]</sup>, the North Atlantic Ocean<sup>[97]</sup>, New Jersey, USA<sup>[98]</sup>, and London, UK<sup>[99]</sup> also used  $\mu$ -RS. A single study used HPLC-MS-MS to identify MPs in 286 indoor house dust samples from 12 countries<sup>[83]</sup>. All other studies reviewed used FTIR for polymeric identification. A single study used thermogravimetry-mass spectrometry to assess the levels of airborne polystyrene from an agricultural area in Spain<sup>[100]</sup>.

It is prudent to mention that these techniques have inherent differences. The ATR-FTIR and  $\mu$ -FTIR spectroscopy can be used for fibers up to 10 and 50  $\mu\text{m}$  particle size, respectively<sup>[12]</sup>.  $\mu$ -RS can be used to identify MPs up to 1  $\mu\text{m}$  in size<sup>[25]</sup>, giving it a technological superiority. However, FTIR has been favored in research due to its high throughput and comprehensive polymer spectral libraries.

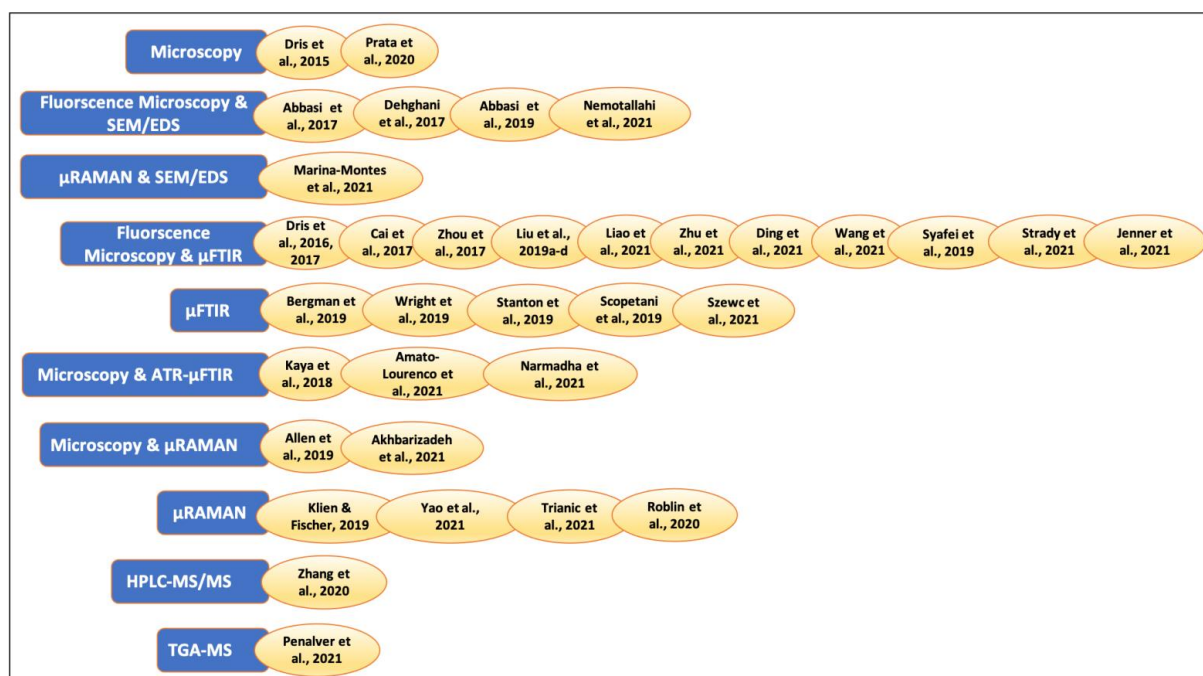
Some researchers have been using polycarbonate, PTFE, and nylon filters, making the application of  $\mu$ -FTIR or  $\mu$ -RS a bit complicated as it interferes with the MP spectra. The preferred filter for  $\mu$ -FTIR and  $\mu$ -RS analysis is an anodisc (aluminum oxide) or a silver filter with a pore size of 0.2  $\mu\text{m}$ <sup>[15,101]</sup>. Py-GCMS, TD, and HPLC-MS-MS are other techniques that can be effectively used to identify MPs but cannot specify the count, shape, and size; instead, they give the concentration of a specific polymer in the sample.

Some points to ponder are: do we need polymeric identification for establishing the environmental concentration of MPs in aerosol? Will knowing the type of polymer make any difference to human risk assessment? Is the toxicity of each polymer different, and do we consider this when estimating the toxicity due to inhalation? If the answer to many of these questions is no, then is it not worth the time and resources that people spend on these studies.

#### *Types of airborne MPs*

Most of the studies published thus far, except for one study each from Iran, France, and Portugal, have done polymeric characterization. The most widely reported polymers in the aerosol are polyethylene (PE), polyethersulfone (PES), polyacetonitrile (PAN), polyamide (PA), polyethyleneterephthalate (PET), polystyrene (PS), polypropylene (PP), PVC, acrylic (AC), polycarbonate (PC), polyacrylic acid (PAA), ethylene-vinyl acetate (EVA), polytetrafluoroethylene (PTFE), ethylene propylene (EP), ethylene acrylic (EA), ethylene vinyl acetate (EVAC), polymethylacrylate (PMA), polyvinyl acetate (PVA), and polyurethane (PU) [Figure 7]. Their relative abundance is highly variable, in both type and quantity. The variation of sampling methods and identification techniques might have contributed to this variance.





**Figure 6.** Techniques used for the identification and characterization of airborne MPs. The blue boxes represent the techniques, while their applications by different research groups are presented in the yellow circles.

## Microplastic inventories and physical properties

### *Size of airborne MPs*

The length of fibers in aerosol samples have been reported to range 50–600  $\mu\text{m}$  with diameters ranging 7–15  $\mu\text{m}$ <sup>[11]</sup>. Although there is a consensus on the upper and lower size limits of  $\leq 5$  mm and 1  $\mu\text{m}$ , respectively<sup>[102–104]</sup>, for MPs, the smaller MPs are not well characterized. The smaller MPs have greater potential to be transferred via aerosolization<sup>[15]</sup> to the human respiratory system. The sizes of MPs in the reviewed studies are highly variable; for example, the dominant size from Pyrenees Mountains was less than 300  $\mu\text{m}$ , while the majority of fragments from Hamburg were < 63  $\mu\text{m}$  (~60%), followed by 63–300  $\mu\text{m}$  (~30%)<sup>[74]</sup>. The dominant size of fibers in studies conducted in Paris was 200–600  $\mu\text{m}$  (~40%)<sup>[18,19]</sup>. Most of the particles reported from China were < 500  $\mu\text{m}$  (~50%)<sup>[105]</sup>. Snow samples were dominated by fine MPs of < 100  $\mu\text{m}$  (98%), of which 80% were  $\leq 25$   $\mu\text{m}$ <sup>[101]</sup>.

The size of fibers/particles/fragments in indoor samples is smaller compared to outdoor aerosol samples. The study conducted in Aviero, Portugal reported the median fiber size in indoor samples as 250  $\mu\text{m}$  compared to 299  $\mu\text{m}$  in outdoor samples<sup>[23]</sup>. Similarly, in California, the outdoor fibers (104.8  $\mu\text{m}$ ) were double the size of indoor (58.6  $\mu\text{m}$ ) ones<sup>[106]</sup>.

### *Reporting units of airborne MPs*

The inventories of atmospheric MPs are highly variable between studies, in terms of both occurrences and units of reporting [Supplementary Table 1]. For most of the studies where passive samplers were deployed, the results are reported in terms of  $\text{MP m}^{-2} \text{day}^{-1}$ , while others have given MPs per unit mass of dust sample<sup>[16,76–78,83]</sup>. The concentration of MPs in snow samples is reported as MPs per liter<sup>[101,107]</sup>. In contrast, the studies that have utilized active sampling have reported the MP concentration in per unit volume ( $\text{MP m}^{-3}$ ) except a single study that indicated MPs per day<sup>[108]</sup>.

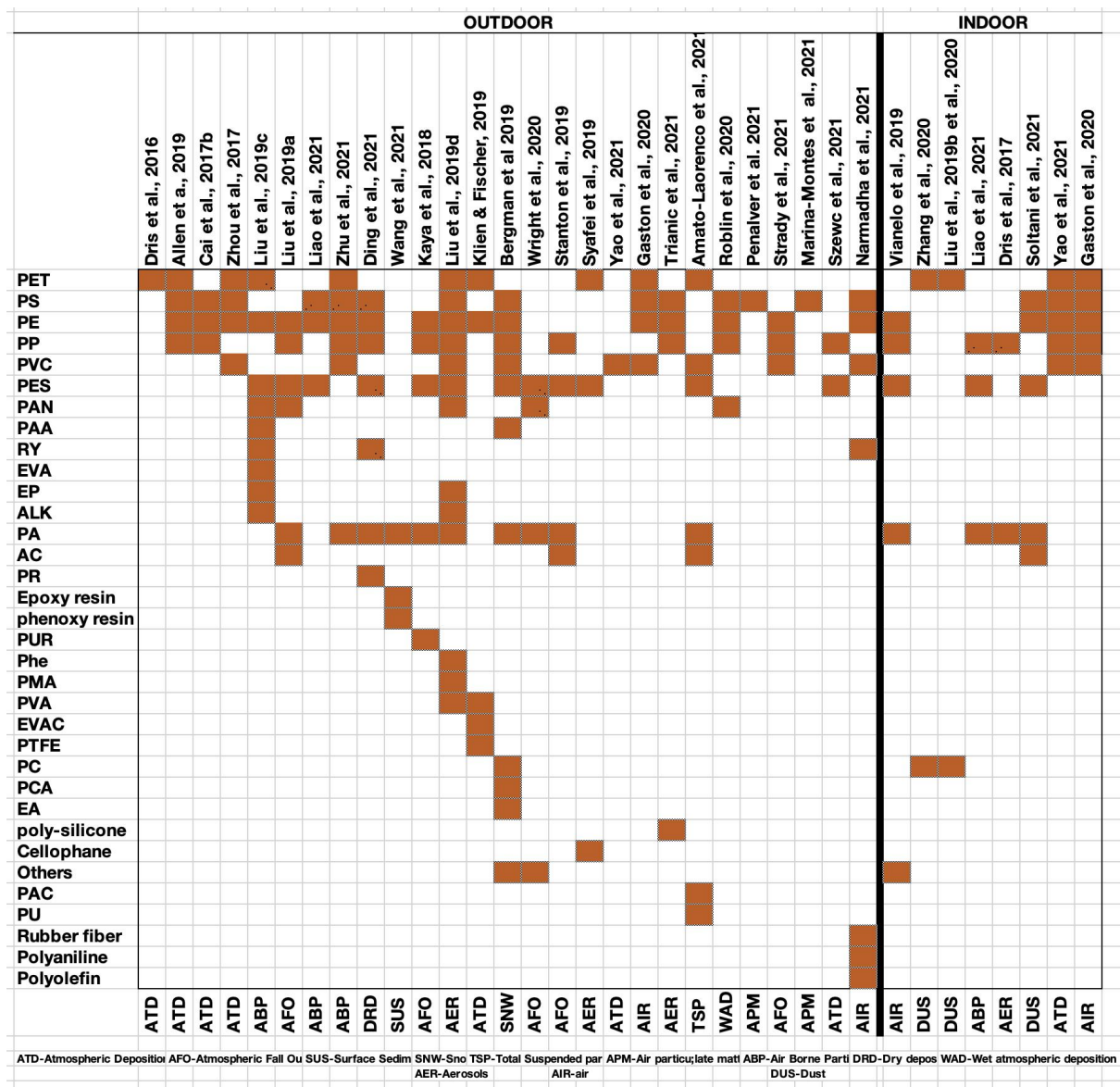
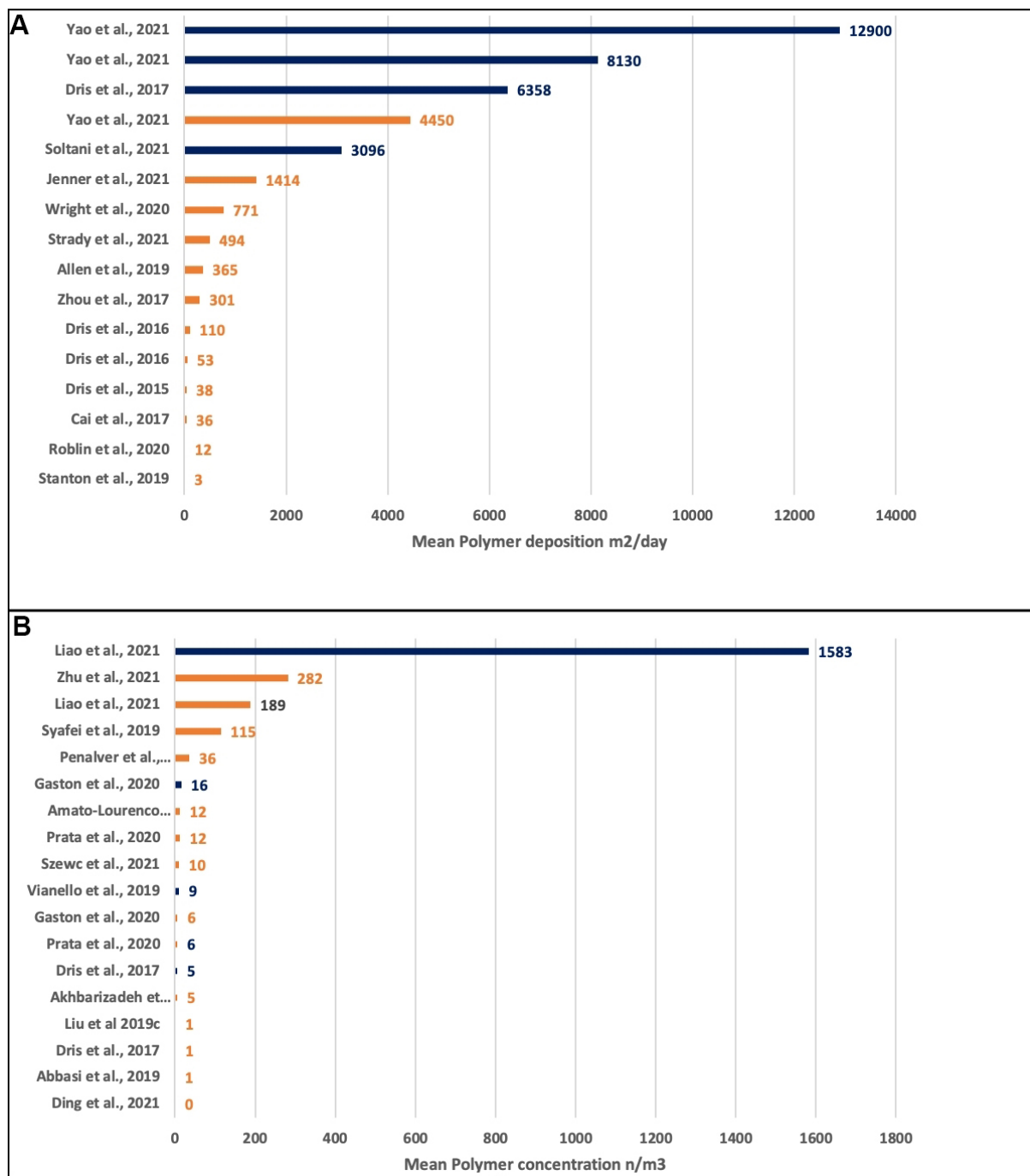


Figure 7. Types of polymers identified in the indoor and outdoor air samples. The types of polymers are listed on the left side of the y-axis. The top bar presents the studies conducted by corresponding research groups in indoor and outdoor aerosols. The types of samples are mentioned on the x-axis. The brown boxes indicate the presence of polymers in the respective sample.

### The concentration of airborne MPs

There is a huge variation in the spatio-temporal concentration of MPs in aerosol [Figure 8]. The average MP concentration measured in outdoor samples in an urban and suburban area of Paris was between 53 and 110 MP m<sup>-2</sup> day<sup>-1</sup>[18,19]. In comparison, it was 275 MP m<sup>-2</sup> day<sup>-1</sup> in Hamburg<sup>[74]</sup>, and the average MP concentration in the Pyrenees was 365 MP m<sup>-2</sup> day<sup>-1</sup>[15]. Microplastic concentrations determined using an active sampler in outdoor samples in Paris were between 0.3 and 1.5 MP m<sup>-3</sup>[20]. In megacities of China, fibers were the most dominant MPs and ranged between 175 and 602 particles m<sup>-2</sup> day<sup>-1</sup>[72,73]. In the eastern city of Wenzhou (China), MP concentration varied significantly between urban and rural sites, ranging from 51 MP m<sup>-3</sup> in the mountains to 8865 MP m<sup>-3</sup> in the city's apartments<sup>[22]</sup>. In Shanghai, the concentrations were between 0 and 4.18 MP m<sup>-3</sup>[80]. In Indonesia, the West Pacific Ocean, Denmark, Ireland,



**Figure 8.** MP polymer (A) deposition rate and (B) concentration. The y-axis represents the research groups. Blue bar represents the indoor samples, whereas the orange bar shows the outdoor samples. The MP concentration/deposition values are written next to the bars.

Brazil, and California the concentrations were 132.75-174.97, 0-1.37, 1.67-16.2, 0-12, 0-24, and 0.6-5.6 MP m<sup>-3</sup>, respectively<sup>[12,66,81,82,96]</sup>. In the UK, the concentrations in Nottingham and Central London were 2.9 and 712 ± 162 particle m<sup>-2</sup> day<sup>-1</sup>, respectively<sup>[71,75]</sup>. In the indoor setting of the Humber region, UK, the concentrations were 1414 MP m<sup>-2</sup> day<sup>-1</sup><sup>[84]</sup>. In Vietnam, the dry and wet atmospheric deposition ranged between 71 and 917 MPs m<sup>-2</sup> day<sup>-1</sup><sup>[109]</sup>. The concentration reported for the Alps and Arctic varied between a

large range of 0.02 and  $154 \times 10^3$  MP L<sup>-1</sup><sup>[101]</sup>, while the average concentration in Finland was  $117.1 \pm 18.4$  MP L<sup>-1</sup><sup>[107]</sup>. This concentration of MP in the snow is several orders of magnitude higher than wet deposition and aerosols, even when reported from urban centers. In Sydney, the MP concentrations were 12 MP m<sup>-3</sup><sup>[24]</sup>. In Gdynia, Poland on the Baltic Sea, the inventory of atmospheric MPs varied from 0 to 30 MP m<sup>-2</sup> day<sup>-1</sup><sup>[110]</sup>. In Badain desert, China MP concentrations were  $0.7 \pm 1.5$  to  $11.7 \pm 15.5$  MP kg<sup>-1</sup><sup>[111]</sup> (dust), and in the South China Sea were 0.013-0.063 MP m<sup>-3</sup><sup>[112]</sup>.

A study investigating indoor and outdoor fibers in air reported a higher indoor concentration ranging from 1 to 60 MP m<sup>-3</sup> as opposed to a significantly lower outdoor concentration ranging between 0.3 and 1.5 MP m<sup>-3</sup><sup>[20]</sup>. Similar observations of higher MP in the indoor air ( $1583 \pm 1180$  MP m<sup>-3</sup>) compared to outdoor air ( $189 \pm 85$  MP m<sup>-3</sup>) were reported by Liao *et al.*<sup>[22]</sup>. The observations of Gaston *et al.*<sup>[106]</sup> were similar as well; indoor concentrations were  $3.3 \pm 2.9$  fibers m<sup>-3</sup> and  $12.6 \pm 8.0$  fragments m<sup>-3</sup> compared to  $0.6 \pm 0.6$  fibers m<sup>-3</sup> and  $5.6 \pm 3.2$  fragments m<sup>-3</sup> in outdoor air. The deposition rate of MPs in the ambient air on a roof of a building was only about 2%-8% of the indoor deposition rates<sup>[98]</sup>. In contrast, MP concentrations in outdoor (12 MP m<sup>-3</sup>) air were higher than those in indoor (5 m<sup>-3</sup>) air in Aveiro, Portugal<sup>[23]</sup>.

In summary, it is evident that average microplastic concentrations were higher in indoor environments than in outdoor aerosol, probably due to more sources, ventilation, and dilution<sup>[13,20,24]</sup>. It is prudent to mention that most people spend 70%-90% of their time indoors, which may enhance their exposure levels to MPs. Interestingly, microplastics generated indoors can frequently contaminate the environment outdoors, whereas only 30% of particulate matter produced outdoors can penetrate the indoor environment. This elucidates the importance of indoor environments as the main exposure source of airborne microplastics<sup>[113]</sup>.

#### *Shapes of airborne MPs*

Diverse shapes, including fiber, foam, fragment, and film, have been detected in the atmospheric MPs [Figure 9]. In Hamburg, more than 90% of MP detected were fragments and less than 10% were fibers<sup>[74]</sup>. Similarly, fragments (89.6%) were dominant in Wenzhou, China, followed by fibers (10.4%)<sup>[22]</sup>. A similar trend was observed in Badain Jaran Desert, China<sup>[111]</sup>, Busheher port of Iran<sup>[94]</sup>, and central London<sup>[26]</sup>. Approximately 87% of the MPs were fragments in the indoor air samples collected in Denmark<sup>[12]</sup>. On the other hand, MP research in other cities of China reported 67%-80% fibers, < 30% fragments, and < 3% granules. Films were reported in outdoor samples in Pyrenees Mountains<sup>[15]</sup>, Asaluyeh city, Iran<sup>[16]</sup>, Dongguan city, China<sup>[73]</sup>, Yantan city, China<sup>[72]</sup>, South China Sea<sup>[112]</sup>, central London, UK<sup>[71]</sup>, Surabaya, Indonesia<sup>[82]</sup>, New Jersey, USA<sup>[98]</sup>, and the Baltic Sea<sup>[110]</sup>. MP films were reported in the indoor samples from New Jersey, USA<sup>[98]</sup> and Humber region, UK<sup>[84]</sup>. The single study in Asaluyeh city, Iran reported spherules (74%) as the dominating type of MPs followed by films (14%)<sup>[16]</sup>.

#### *Color of airborne MPs*

The microplastics consist of different colors with the most dominant ones being red, orange, yellow, white, grey, blue, green transparent, and black [Figure 10]. The use of color to identify potential sources of plastic debris is sometimes practiced<sup>[104]</sup>; however, this can be quite misleading. Several studies in this review have reported the color of the identified MPs [Supplementary Table 1]. Blue and red MPs were reported from Paris and China<sup>[18,73]</sup>, while black, blue, red, transparent, brown, green, yellow, and grey particles were reported from Shanghai, China<sup>[80]</sup> and Iran<sup>[16,76,77]</sup>. Dark blue, white, transparent, and brown colored MPs were reported in aerosol samples from the West Pacific Ocean<sup>[81]</sup>. The colors were bright yellow to white in atmospheric deposition from Hamburg, Germany<sup>[74]</sup>. Black and grey MP fibers were found in passive fallout samples in Nottingham, UK<sup>[75]</sup>. The wet atmospheric deposits in Ireland were rich in red, blue, and yellow fibers<sup>[96]</sup>. White transparent fibers were detected in the ambient air samples collected from the streets of

		Fibers	Fragments	Films	Granules	Spherules	Foams	Particles	Microbeads	Sheets
<b>OUTDOOR</b>	Dris et al., 2015	100–5000 µm								
	Dris et al., 2016	50–4850 µm								
	Dris et al., 2017	50–1650 µm								
	Allen et al., 2019		<100->2600 µm							
	Akhbarizadeh et al., 2021									
	Abbasi et al., 2017	<100–<5000 µm								
	Dehghani et al., 2017									
	Abbasi et al., 2019				<100->1000 µm					
	Cai et al., 2017b	200–4200 µm								
	Zhou et al., 2017	50–1000 µm								
	Liu et al., 2019c	23–5000 µm								
	Liu et al., 2019a	50–2 mm								
	Liao et al., 2021									
	Zhu et al., 2021	5.9–1475.3								
	Ding et al., 2021	50–2210 µm								
	Wang et al., 2021			50–200 µm						
	Kaya et al., 2018	500 µm	50 µm							
	Liu et al., 2019d									
	Klien and Fischer 2019									
	Bergman et al., 2019								<25–<100 µm	
Wright et al., 2020			400–500 µm							
Stanton et al., 2019	38 µm									
Scopetani et al., 2019										
Syafei et al., 2019	<500–5000 µm									
Bergman et al., 2019	11–475 µm									
Prata et al., 2019	26–1928 µm									
Yao et al., 2021	35–1000 µm			200–5000 µm <sup>1</sup>						
Gaston et al., 2020	104.8 µm									
Trianic et al., 2021								>5 µm		
Amato-Lourenco et al., 2021	50–1579 µm							50–877 µm		
Roblin et al., 2020	<200–4000 µm									
Strady et al., 2021										
Marina-Montes et al., 2021										
Szewc et al., 2021										
Narmadha et al., 2021										
<b>INDOOR</b>	Vianelo et al., 2019		4–398 µm							
	Liu et al., 2019b									
	Liao et al., 2021			5–4665 µm						
	Dris et al., 2017	50–1650 µm								
	Soltani et al., 2021	200–400 µm								
	Prata et al., 2020	17–3669 µm								
	Yao et al., 2021	35–1000 µm			200–5000 µm <sup>2</sup>					
	Gaston et al., 2020	58.6 µm								
	Nematollahi et al., 2021	500–1000 µm								
	Jenner et al., 2021	5–250 µm								

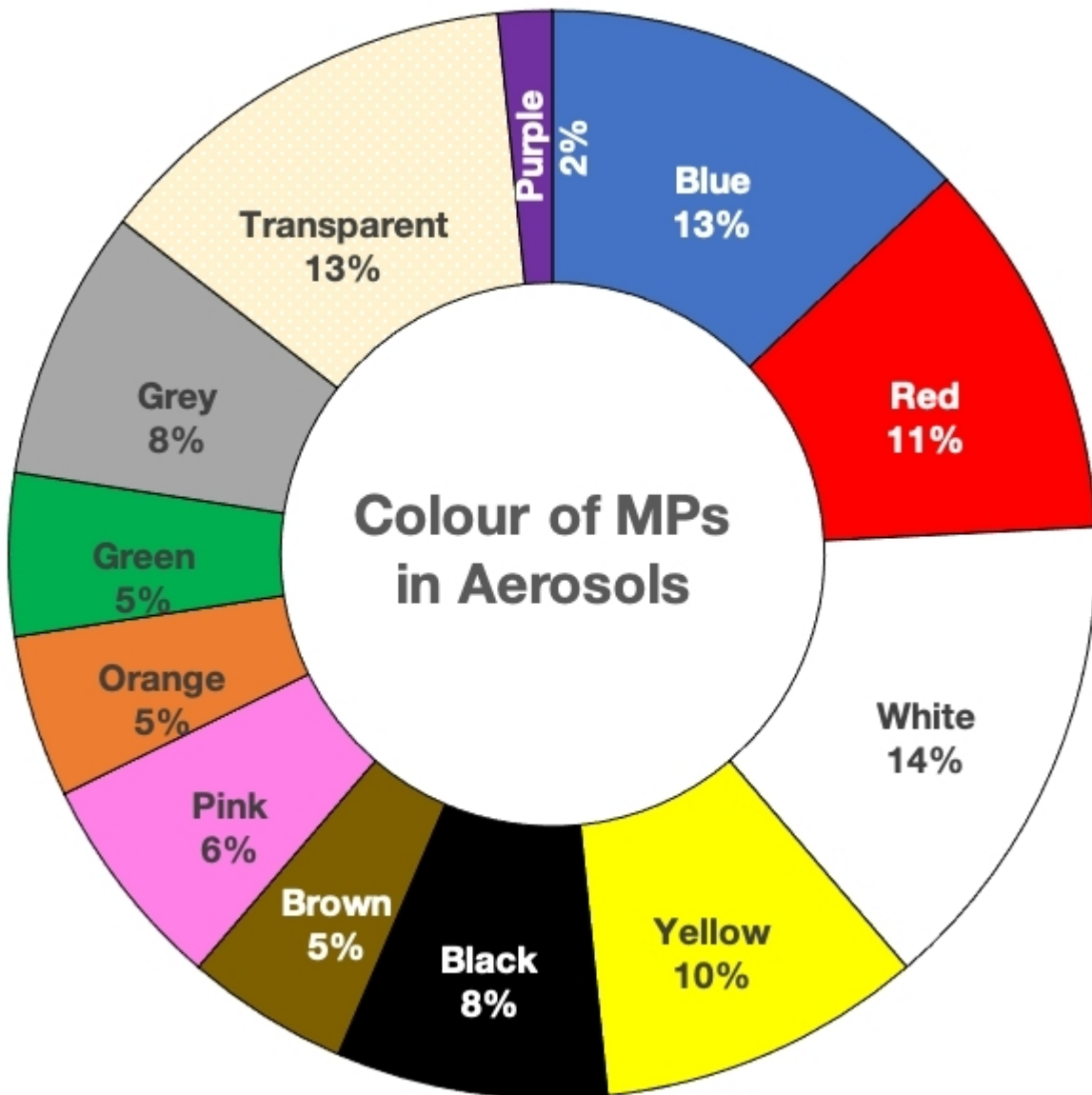
Figure 9. Shape and size of microplastics found in aerosols.

Nagpur, India<sup>[108]</sup>.

It is also prudent to mention that most of these studies have used 30% and 35% H<sub>2</sub>O<sub>2</sub> to digest the organic material, but this treatment can cause significant degradation of polymers in terms of both physical and chemical properties. One of the effects of such treatment can be the discoloration of microplastics. The other point that needs discussion is whether this information can be used in any way; e.g., can color be a reliable indication source?

### CONCLUSIONS

The review highlights the limited information on MP in aerosol from only about four dozen studies. Even within these studies, different approaches have been used for sample collection, sample processing, and reporting units, making any direct comparisons across spatial and temporal domains incoherent. However, despite these limitations, it could be confidently inferred that MPs are more abundant in indoor compared to outdoor aerosol. The observation is not surprising as the sources of MPs are clothing, carpets, dryers, upholstery, etc., and there is less mixing dilution in the indoor environment compared to outdoors. Fibers and fragments are the predominant shapes of MPs in aerosol samples, with white, transparent, blue, red, and yellow being the most prevailing colors. The need of adapting standardized protocols for atmospheric MP research is evident from this review in order to make these assessments comparable, to establish spatio-temporal trends, and for undertaking comprehensive human health risk assessments. The choice of sampling is very important, and, based on the research question, it should be appropriately designed. Use of



**Figure 10.** Color of microplastics (MPs) found in aerosol samples.

passive sampling by atmospheric deposition, taking swipes over a surface, and relating them to inhalation is no better than a first-order approximation.

Some other aspects of atmospheric MP research that need some discussion and consensus are the following: should the active and passive sampling techniques be used in tandem to answer questions on health risk and seasonal MP loading and accumulation rates over a long period? The emphasis on polymeric characterization is significant in current MP research, and it is certainly important to know what are the dominant polymers for source apportionment, but is there much value of this effort for environmental or human health risk assessment? A simpler hot needle technique can provide equally useful information.

It has been observed in MP research that significant issues of degradation, losses, and cross-contamination are introduced due to sample processing, be it the use of impure salts for making solutions, discoloration due to H<sub>2</sub>O<sub>2</sub>, or degradation due to use of acids, the use of inappropriate filters to separate MPs from associated matrices, or heating of sample while processing and cross-contamination during microscopic examination. From our experience and a comparative study (unpublished), we found that the use of an active sampler with cascade impactor and without any filter media provided a cleaner, size-fractionated aerosol sample that can be directly subjected to microscopic examination under stereomicroscope and UV fluorescence microscope after staining with Nile red. The hot needle technique was found to be very effective to differentiate MPs from natural fibers.

This review also highlights the fact that most researchers use a multi-tier approach for MP identification, with the first step using microscopy, followed by polymeric characterization. Another pertinent issue is the variable set of information provided by different researchers; there is no standardization of nomenclature, with some researchers only indicating the use of FTIR without specifying ATR or micro. The nomenclature for shape is also quite variable. Some researchers have only indicated particles, others have been specific about fibers and fragments, some have used the term “spherules”, and others have used “granules”, making it a difficult task to standardize and compare them. Many studies have deployed passive samplers, but with no ancillary data or discussion on wind directions, velocities, dust loadings, *etc.* that prevailed during sample collection. With all this missing information, readers cannot make any realistic inferences on likely health implications, which is the core of this research.

Several researchers have reported color of MPs, but will this knowledge have any implication for risk? Can color itself in the absence of other parameters be of any use? The measurement of length and diameter is done under a microscope and is often a tedious task, so would the use of a size-fractionated cascade impactor be a better approach for size segregation while sampling?

The most important aspect of atmospheric MP research is the human health implications. It is well established that PM<sub>10</sub> and PM<sub>2.5</sub> are extremely important particle sizes for pulmonary risk assessments. Most studies have not addressed these size fractions<sup>[114]</sup>. In addition, the use of ATR-FTIR and  $\mu$ -FTIR cannot help in polymeric identification of < 10  $\mu$ m particles, so will it help in human health risk assessment due to inhalation where PM<sub>2.5</sub> and finer particle sizes are more important to assess the respirable fraction? On the other hand, the  $\mu$ -RS can provide polymer identification of MP up to 1  $\mu$ m that can be more useful, but does polymer type make a difference, or is it the size that matters? Should there be an emphasis on the use of aerosol size fractions or the total number of MPs per cubic meter or liter to generate meaningful information for human health risk?

Does the information from TGA, HPLC-MS-MS, and pyro GC-MS on polymer mass in milligrams per cubic meter have any use without knowing to which size fraction it belongs? Are they inhalable or respirable? Is it reasonable to report number of MP per unit mass of size-fractionated aerosol (MP mg<sup>-1</sup> of the aerosol fraction)? This will be a technological challenge due to their infinitesimal mass in even large sample volumes.

The review overwhelmingly highlights the need to harmonize the methodology for studying MPs in aerosol. There is an urgent need to discuss the importance for using size fractionation, polymeric characterization, sample volumes, and reporting units keeping in mind what is actually useful for the scientific question one is addressing.

## DECLARATIONS

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### Authors' contributions

Conceptualization: Habibi N, Uddin S, Fowler SW

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Software: Habibi N

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Writing - original draft preparation: Habibi N, Uddin S, Behbehani M

Writing - review and editing: Uddin S, Habibi N, Fowler SW

Visualization: Habibi N

All authors have read and agreed to the published version of the manuscript.

### Availability of data and materials

The data presented in this review are available as a supplementary file.

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All authors declared that there are no conflicts of interest.

### Ethical approval and consent to participate

Not applicable.

### Consent for publication

Not applicable.

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