



Mini-review on current studies of airborne microplastics: Analytical methods, occurrence, sources, fate and potential risk to human beings

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ABSTRACT

Microplastics (MPs) are emerging pollutants and have potential adverse effects to organisms. Airborne MPs are of great concern, because they are an important contributor to MPs in other environmental compartments such as water and soils and may pose potential risk to human health via inhalation or dust ingestion. Analytical methods, abundance, pollution characteristics, potential sources and risk to human beings of airborne MPs in suspended particulates, atmospheric fallout and deposited dusts were summarized in the present review. Research gaps and suggestions for future works on airborne MPs were also given. Digestion of samples was suggested to avoid the interference of organic materials to airborne MPs. New techniques such as hyperspectral imaging technique, pyrolysis and thermal desorption gas chromatography mass spectrometry were expected to be used for the analysis of airborne MPs. Furthermore, more studies were required to fully understand the pollution status and potential risk of airborne MPs.

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1. Introduction

Microplastics (MPs) are plastic particles with dimension < 5 mm, which originate from primary sources such as synthetic textiles, cosmetics, etc [1], or from secondary sources such as fragmentation and abrasion of large plastic debris [2]. Being considered as emerging pollutants, MPs have been detected in different types of environmental compartments worldwide, such as water [3,4], sediments [5–8], soils [9,10] and atmosphere [11–13]. MPs have also been detected in freshwater and marine organisms due to ingestion of MPs via organisms [3,8,14,15]. Because of the potential negative effects of MPs to aquatic organisms, numerous studies have been carried out in aquatic ecosystems [3,4,16–18], with fiber, granule, pellet, and film being the main shapes of MPs observed [3,4]. With respect to chemical composition, polypropylene (PP), polyethylene

(PE), polyethylene terephthalate (PET), polyvinyl chloride (PVC), polystyrene (PS), and polyamide (PA) were polymer types frequently detected in the aquatic environments and organisms [4,19].

Studies on airborne MPs were relatively scarce and have gained increasing concerns recently. Airborne MPs are considered to be an important contributor to MPs in the aquatic environments and the soils, as airborne MPs would enter these environments via dry or wet deposition [20,21]. Moreover, human beings would be exposed to airborne MPs via inhalation and dust ingestion, causing potential adverse effects to human health [22]. Airborne MPs ingested by human beings would cause inflammation and secondary genotoxicity [23]. Furthermore, airborne MPs may contain various harmful chemicals, such as the unreacted monomers, additives of plastics, and other harmful contaminants absorbed from the environments (such as polycyclic aromatic hydrocarbons, persistent organic pollutants, heavy metals and microorganisms) [22,23]. These harmful chemicals may enhance the toxicity of airborne MPs to human beings [22,23].

The purposes of this review are to 1) review recent methods that were used for sampling, pretreatment, and analysis of airborne MPs; 2) summarize the occurrence and characteristics of airborne MPs in suspended particles, atmospheric fallout and deposited

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dusts; 3) discuss the potential sources, dispersion and deposition of airborne MPs; 4) assess the potential risk of airborne MPs to human beings; and 5) identify the research gaps and give suggestions for future works.

2. Techniques used for airborne MP detection

Generally, detection of airborne MPs included sample collection, pretreatment and instrumental analysis.

2.1. Sample collection

Airborne MPs in different types of environment media were collected using different methods. As shown in Table 1, to analyze suspended atmospheric microplastics (SAMPs), particulate matter (PM) in the air were collected on quartz fiber filters (pore size: 1.6 μm , diameter: 47 mm) [11], glass microfiber filters (pore size: 1.6 μm , diameter: 90 mm) [13] or polytetrafluoroethylene (PTFE) filters (pore size: 2 μm , diameter: 46.2 mm) [24] using suspended particulate samplers. Generally, total suspended particulates (TSP) were collected to analyze SAMPs [11,13], whereas, in addition to TSP, PM_{2.5} and PM₁₀ were also collected in Asaluyeh County, Iran [24]. The air volumes of the samples varied from 2 to 23.93 m³ [11,13,24].

Total atmospheric fallout (including dry and wet atmospheric fallout) were collected using passive samplers [20,21,25–27]. Glass bottles were the most commonly used collectors [20,21,25,26]. The atmospheric fallout samples were collected monthly [26,27] or quarterly [21]. In some studies, samples were collected at various frequencies depending on cumulative of rainfall [20,25]. To collect MPs in dry atmospheric fallout (dust fall) in indoor environments, quartz fibers [11] or double-sided adhesive plastic pads [28] have been used as passive samplers.

Dust samples were collected using vacuum cleaner [11], hog bristle brush [29], or wooden brush [24,30], and then transferred to low-density polyethylene bags [24] or paper bags [29]. To avoid cross contamination, the brushes used to collect dust should to be washed with filtered distilled water [24] or rinsed with ethanol [29].

2.2. Separation and digestion

MPs in environmental matrices usually coexist with organic materials. To obtain accurate characterization and quantification of MPs, it is important to remove the organic materials in the samples. Till now, various methods have been explored to destruct natural organic materials before instrumental analysis of MPs in water, sediment, biota, and soil [3,4,9,18,31]. The most widely used method is to digest samples with 30% (v/v) H₂O₂ solution, and the digestion process was accompanied with heating to speed up the reaction in some cases [3,31]. Solutions such as mixture of 30% H₂O₂ and H₂SO₄ [32], Fenton's reagent (30% H₂O₂ with an iron catalyst) [31,33], HNO₃ [34], 10% KOH [3], and 10 M NaOH [35] have also been used to digest samples. Furthermore, enzymatic digestion of samples using sodium dodecyl sulfate in combination with enzymes has also been reported in several studies [35–37].

To remove the interference of organic matter to SAMPs, SAMPs collected on filters were first washed to glass beaker with deionized water and further treated using 30% H₂O₂ for 8 days [24]. After digestion, MPs were separated using NaI solution with a density of 1.6 g/mL [24]. Density separation is a frequently used method for MP isolation after sample digestion [18]. Solutions such as NaCl (1.2 g/mL), ZnCl₂ (1.5 g/mL or 1.7 g/mL), NaI (1.6 g/mL) and potassium formate (1.5 g/mL) were often used in previous studies [3],

whereas, it was suggested that solution with a density range of 1.6–1.8 g/mL would be optimal for density separation [5].

Atmospheric fallout collected by the passive collectors were first filtered on filters such as glass fibre filters [20,26], quartz fiber filters [25], nitrocellulose filters [21] and PTFE filters [27]. To remove the moss in samples collected in spring from Yantai, China, particles on the filters were rinsed off and digested with 30% H₂O₂ for 6 h at 70°C before filtration [21]. Allen et al. [27] performed a more complicated procedure of H₂O₂ digestion. In brief, particles rinsed from the filters were first digested with 10 mL of H₂O₂ at 55 °C for 7 days, and on day 8, 5 mL of H₂O₂ was added to digest the samples for another 7 days.

Similar to sediments and soils, dusts are also solid samples with high content of organic matter [30]. Dust samples were usually sieved through stainless steel sieves (mesh: 2 mm, 2.5 mm or 5 mm) to remove large debris [11,24,29,30]. After sieving, MPs were density separated from the dust using ZnCl₂ solution (1.6 g/mL or 1.78 g/mL) [11,29,30] or NaI solution (1.6 g/mL) [24]. To remove organic matter in dust, street dust samples were treated with 30% H₂O₂ for 8 days before separation [24,30].

As mentioned above, digesting samples with 30% H₂O₂ was the only method reported for airborne MPs [21,24,27,30], while other methods such as alkaline digestion or enzymatic digestion were not reported for this type of samples. It should be noted that, in some studies, samples were not digested before instrumental analysis of airborne MPs [11,13,20,21,25,26,29], which might lead to over-estimation of MP numbers, as natural materials such as cotton or wool may also been included during enumeration [11,13,25,26,29]. For example, the contribution of natural materials to total fibers was as high as 67% in indoor suspended atmospheric particulate samples from Paris, France, and only 33% of the observed fibers were identified as airborne MPs [11].

2.3. Characterization and quantification

Similar to analysis of MPs in other environment media, instrumental analysis needs to be conducted to obtain physical characteristics and chemical characteristics of airborne MPs [4,38].

To obtain physical characteristics such as shape, size, color and number, airborne MPs were observed and counted under microscopes (Fig. 1). As shown in Table 1, stereomicroscope was the most commonly used device for physical characterization of airborne MPs [11,13,20,21,25,27]. Digital microscope [26] and fluorescence microscopy [24,30] has also been used for visual observation of MPs. Nevertheless, fluorescence microscopy was found to be ineffective for the detection and count of MPs, thus it was used in combination with an optical microscope [24,30]. Scanning electron microscopy (SEM) was used to confirm the morphological characteristics of representative airborne MPs and illustrate the degradation pattern of airborne MPs [26]. Furthermore, SEM could be used in combination with an energy dispersive X-ray spectroscopy (EDS) detector to analyze the surface morphology and elemental composition of airborne MPs [24,30]. However, the SEM technique is time-consuming and expensive, and is only suitable for detecting typical airborne MPs selected [30].

After physical characterization, airborne MPs was further identified for their chemical compositions. As shown in Table 1, Fourier transform infrared micro-spectroscopy (μ -FTIR) was the most widely used device [11,13,21,25,26,29]. Although FTIR was also used for identification of MPs in other environmental samples such as water [39,40], sediment [41] and soil [42], it was not used for airborne MPs. This might be due to that FTIR has lower sensitivity than μ -FTIR [18], and are not efficient for the small-sized airborne MPs. Micro-Raman spectroscopy (μ -Raman) was also used to identify the polymer composition of airborne MPs in atmospheric

Table 1
Analytical methods for airborne microplastics worldwide.

Study area	Environment matrix	Sampler	Separation and digestion	Observation size range (μm)	Characterization	Reference
Paris, France	Total suspended particulates (indoors)	Suspended particulate sampler	–	50–5000	Counted by stereomicroscope and identified by $\mu\text{-FTIR}^{\text{a}}$	[11]
Paris, France	Total suspended particulates (outdoors)	Suspended particulate sampler	–	50–5000	Counted by stereomicroscope and identified by $\mu\text{-FTIR}$	[11]
Shanghai, China	Total suspended particulates	Suspended particulate sampler	–	> 1.6	Enumerated by stereomicroscope and identified by $\mu\text{-FTIR}$	[13]
Asaluyeh County, Iran-industrial station	Total suspended particulates	Suspended particulate sampler	Digested with 30% H_2O_2 for 8 d and then density separated using NaI solution (1.6 g/mL)	2–5000	Visually examined using binocular microscopy, polarized light microscopy and fluorescence microscopy, and determined by SEM ^b /EDS ^c	[24]
Asaluyeh County, Iran-urban station	Total suspended particulates	Passive sampler	Digested with 30% H_2O_2 for 8 d and then density separated using NaI solution (1.6 g/mL)	2–5000	Visually examined using binocular microscopy, polarized light microscopy and fluorescence microscopy, and determined by SEM/EDS	[24]
Paris, France -urban area	Dry and wet atmospheric fallout	Passive sampler	Filtered on glass fibre filters (1.6 μm)	100–5000	Observed by stereomicroscope	[20]
Paris, France -urban area	Dry and wet atmospheric fallout	Passive sampler	Filtered on quartz fiber filters (1.6 μm)	50–5000	Observed by stereomicroscope and identified by $\mu\text{-FTIR}$	[25]
Paris, France -suburban area	dry and wet atmospheric fallout	Passive sampler	Filtered on quartz fiber filters (1.6 μm)	50–5000	Observed by stereomicroscope and identified by $\mu\text{-FTIR}$	[25]
Dongguan, China	Dry and wet atmospheric fallout	Passive sampler	Filtered on glass microfibre filters (1.0 μm)	1–5000	Observed by digital microscope and SEM and identified by $\mu\text{-FTIR}$	[26]
Yantai, China	Dry and wet atmospheric fallout	Passive sampler	Filtered on nitrocellulose filters (5 μm) Sample collected in spring was digested with 30% H_2O_2 at 70°C for 6 h	50–3000	Observed by stereomicroscope and identified by ATR-FTIR ^d and $\mu\text{-FTIR}$	[21]
Pyrenees mountains, France	Dry and wet atmospheric fallout	Passive sampler	Filtered on polytetrafluoroethylene filters (0.45 μm) Digested with 30% H_2O_2 at 55°C for 7 d and then digested for another 7 d with a further 5 mL 30% H_2O_2 added Density separated using ZnCl_2 solution (1.6 g/mL)	0.45–2000	Counted by stereomicroscope and identified by Micro-Raman spectroscopy	[27]
Tehran, Iran	Street dust	Wooden brush	Digested with 30% H_2O_2 for 8 d and then density separated using ZnCl_2 solution (1.78 g/mL)	2–5000	Visually examined using binocular microscopy and fluorescence microscopy, and determined by SEM/EDS	[30]
Asaluyeh County, Iran-urban station	Street dust	Wooden brush	Digested with 30% H_2O_2 for 8 d and then density separated using NaI solution (1.6 g/mL)	2–5000	Visually examined using binocular microscopy, polarized light microscopy and fluorescence microscopy, and determined by SEM/EDS	[24]
China	Indoor dust	Hog bristle brush	Density separated twice using ZnCl_2 solution (1.6 g/mL).	50–2000	Counted under a light microscope and chemically identified by $\mu\text{-FTIR}$; the mass concentrations of specific MPs were analyzed by LC-MS/MS ^e	[29]

(continued on next page)

Table 1 (continued)

Study area	Environment matrix	Sampler	Separation and digestion	Observation size range (μm)	Characterization	Reference
China	Outdoor dust	Hog bristle brush	Density separated twice using ZnCl_2 solution (1.6 g/mL).	50–2000	Counted under a light microscope and chemically identified by $\mu\text{-FTIR}$; the mass concentrations of specific MPs were analyzed by LC-MS/MS. Counted by stereomicroscope and identified by $\mu\text{-FTIR}$.	[29]
Paris, France	Indoor dust	Vacuum cleaner	Density separated using ZnCl_2 solution (1.6 g/mL).	50–5000		[11]

^a Fourier transform infrared micro-spectroscopy.

^b Scanning electron microscopy.

^c Energy dispersive X-ray spectroscopy.

^d Attenuated total reflectance-Fourier transform infrared spectroscopy.

^e Liquid chromatography-tandem mass spectrometry.

fallout [26]. Recently, hyperspectral imaging technique has emerged as a simple and fast method to detect MPs in seawater [43] and soil [44] without digestion. Further studies are suggested to validate whether this method could be used to detect airborne MPs.

In addition to characterization, quantification of specific airborne MP polymers has also been performed. Mass concentrations of two main airborne polymer PET and polycarbonate (PC) were quantified using alkali-assisted thermal depolymerization-liquid chromatography-tandem mass spectrometry (LC-MS/MS) as described by Liu et al. [29]. Techniques such as pyrolysis-gas chromatography mass chromatography (Pyr-GC-MS) [45] and thermal desorption-gas chromatography mass spectrometry (TDS-GC-MS) [46] were also reported to have the ability of quantifying mass concentrations of specific MP polymers in environmental samples and could be explored to quantify airborne MP polymers in future work.

2.4. Quality control and quality assurance

To obtain reliable data of airborne MPs, it is necessary to prevent plastic pollution throughout the processes of sample collection, pretreatment and instrumental analysis. Cotton laboratory coats were suggested to be worn during all experiments [13,20,24–27,30]. The use of plastic materials should be avoided as much as possible [13,20,25,30], if plastic materials were necessary to use, they should be rinsed with ultrapure water at least three times [13]. Furthermore, all the glassware should be properly cleaned and heated at 500°C before use [25], and materials such as glass fibre filters and quartz fiber filters should be heated at 500 °C prior to use. After sampling, it would be better to process the samples in a 'clean room', where working surfaces were thoroughly wiped and all windows and doors were closed [24]. To verify that the samples were not contaminated during the experiments, laboratory blanks such as MilliQ water samples were treated in the same way as were the field samples [20,21,25–27,30]. Moreover, empty petri dishes or dishes full of filtered water were placed as control samples in the laboratory during the experiments to detect whether MPs were added during the extraction and subsequent analysis of airborne MPs [24,30]. In addition, field blanks were also collected and analyzed by Allen et al. [27] to quantify possible MP contamination caused by sampling of atmospheric fallout samples. To ensure the accuracy of data, duplicate samples were often performed [27,30]. For example, duplicate atmospheric fallout samples were collected for each sampling period [27]. Although, duplicate atmospheric fallout samples were not collected by Dris et al. [25], airborne MPs in the samples were counted by three different observers, with differences less than 5%. In addition, airborne MPs in five randomly selected street dust samples were counted twice by Abbasi et al. [24] for replication purpose, with differences ranging from 5 to 17.

3. Occurrence and characteristics of airborne MPs

3.1. MPs in the suspended atmospheric particulates

Abundance and characteristics of airborne MPs in various environmental media were summarized in Table 2.

SAMPs have been analyzed in three studies as demonstrated in Table 2. Only fibers were detected in suspended particulates collected in Paris, France [11] and Asaluyeh County, Iran [24]. Fibers in indoor suspended particles from Paris, France were 0.4–59.4 fibers/ m^3 , significantly higher than those in outdoor environments (0.3–1.5 fibers/ m^3), and most fibers fell in the range of 50–250 μm [11]. Similar fiber concentrations were observed in outdoor TSP samples from the industrial (0.76 fibers/ m^3) and urban station (0.63

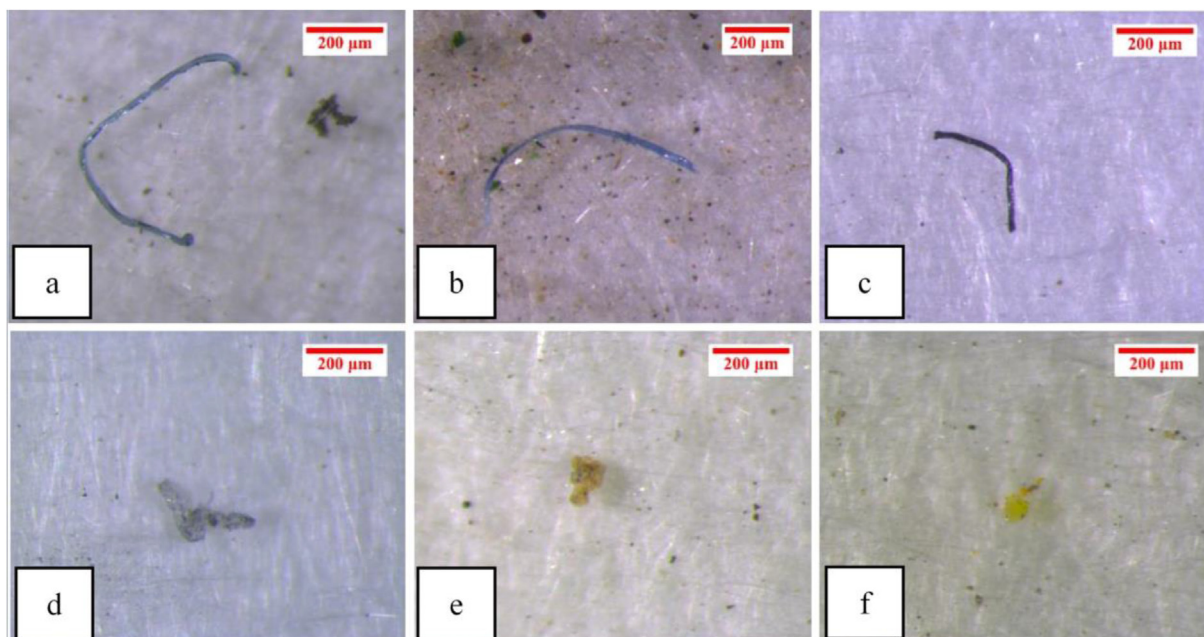


Fig. 1. Photographs of typically observed airborne microplastics. a–c: fibers; d–e: fragments; f: a granule [13].

fibers/m³) of Asaluyeh County, Iran, with white or transparent fibers being the most abundant [24]. Although various shapes of MPs were detected in TSP from Shanghai, China, round and flat fibrous MPs were primarily observed, comprising 67% of the total particles [13]. SAMP concentrations in Shanghai, China fell in the range of 0–4.18 particles/m³ (average: 1.42 ± 1.42 particles/m³), with blue (37%) and black (33%), MPs being the dominant MPs [13]. In terms of polymer type, PP was most abundant for fibers in TSP from Paris [11], whereas, PET was the main polymer type for fibers in TSP from Shanghai (50%) [13]. In addition, PET, PE, and PES comprised nearly half (49%) of the SAMPs observed in Shanghai, followed by PAN (12%) and PAA (9%) [13].

3.2. MPs in the atmospheric fallout

As shown in Table 2, MPs in the total atmospheric fallout samples from Greater Paris fell in the range of 29–280 particles/m²/d, with an average of 18 particles/m²/d [20]. Higher deposition fluxes of MPs were observed in atmospheric fallout samples from Yantai and Dongguan, China, with a range of 130–624 particles/m²/d [21] and 175–313 particles/m²/d, respectively [26]. However, since 77% of fibers and 15.4% of other shapes were identified as non-plastics, the average deposition flux of MPs identified by μ -FTIR in Dongguan was 36 ± 7 particles/m²/d [26]. To discover the extent of the MP atmospheric deposition in remote terrestrial location, MPs in atmospheric fallout from the Pyrenees mountains have been analyzed, and an average deposition flux of 365 ± 69 particles/m²/d was observed [27]. Fibers were dominant in atmospheric fallout collected in Paris [20], Yantai [21] and Dongguan [26], accounting for as much as about 90% of the total MPs. Different from those in cities, fragments were found to be the dominant shape in the atmospheric fallout from the Pyrenees mountains, followed by films and fibers [27]. Foam was observed sometimes in atmospheric fallout, but at very low detection frequencies [21,26]. In terms of size, the predominant fiber length range was 200–600 μ m in the atmospheric fallout from Paris [25]. Fibers in atmospheric fallout from Dongguan were similar to those observed in Paris, and mainly

fell in the range of 200–700 μ m, with a proportion of ~30% [26]. The predominant fiber lengths in atmospheric fallout from the remote catchment of Pyrenees mountains were 100–200 μ m and 200–300 μ m [27], smaller than those in mega cities such as Paris [25] and Dongguan [26]. With respect to fragment and film, the predominant size range was <50 μ m and 50–200 μ m, respectively [27]. Predominant lengths of MPs in the atmospheric fallout from Yantai were less than 500 μ m, accounting more than 50% of total MPs [21]. In addition, PET was the main polymer type for fibers in Paris [25] and Yantai (40%) [21], whereas, PP was the most abundant polymer type for fibers in the Pyrenees mountains [27]. The predominant polymer type for fragments in the Pyrenees Mountains was PS, followed by PE [27]. PE was also dominant for fragments in Yantai [21].

3.3. MPs in urban deposited dust

Fiber levels in apartment dusts from Paris ranged from 190.0–670.0 fibers/mg [11]. Similar fiber levels have been observed in indoor samples from major cities of China, with a range of 17–620 fibers/mg (average: 342 fibers/mg) [29]. As shown in Table 2, MP concentrations in outdoor dusts from China were significantly lower than those in indoor dusts [29]. MPs in street dusts varied from 83 ± 10 to 605 ± 10 particles/30 g dust in the central area of Tehran, Iran [30], and from 52 to 7748 particles/15 g in Asaluyeh County, Iran [24]. Fibers were dominant in MPs in indoor (88.0%) and outdoor (73.7%) dusts from China [29], whereas, granules (range: 54.5–82.2%; mean: 65.9%) were more abundant than fibers (range: 16.9–44.3%; mean: 33.5%) in street dusts from Tehran, Iran [30]. Spherical (74%) and film-like (14%) particles were the dominant MPs in street dust, from Asaluyeh County, Iran [24]. In terms of size range, MPs ranging from 250 to 500 μ m were the most abundant particles (mean: 33.7%) in street dusts from Tehran, Iran [30]. Much finer MPs dominated in street dusts from Asaluyeh County, Iran, where 77% of MPs were less than 100 μ m, in particular, 98% of spherical MPs were <100 μ m [24]. Colors of MPs were also obtained for street dusts from Iran. Similar to those in TSP samples, white or

Table 2
Occurrence and characteristics of airborne microplastics worldwide.

Study area	Environment matrix	Abundance	Size range (μm)	Shape	Color	Polymer type ^a	Reference
Paris, France	Total suspended particulates (indoors)	Range: 0.4–59.4 particles/m ³ ; median: 5.4 particles/m ³	50–3250	Fiber	–	RY, PA, PE, PP	[11]
Paris, France	Total suspended particulates (outdoors)	Range: 0.3–1.5 particles/m ³ ; median: 0.9 particles/m ³	50–1650	Fiber	–	–	[11]
Shanghai, China	Total suspended particulates	Range: 0–4.18 particles/m ³ ; Mean: 1.42 \pm 1.42 particles/m ³	23–9555	Fiber, fragment, granule	Blue, black, red, transparent, brown, green, yellow, grey	PET, PE, PES, PAN, PAA, RY, EVA, EP, ALK	[13]
Asaluyeh County, Iran-industrial station	Total suspended particulates	0.76 particles/m ³	2–5000	Fiber	White-transparent, red-pink, black-grey	–	[24]
Asaluyeh County, Iran-urban station	Total suspended particulates	0.63 particles/m ³	2–5000	Fiber	White-transparent, yellow-orange, red-pink, black-grey	–	[24]
Paris, France - urban area	Dry and wet atmospheric fallout	Range: 29–280 particles/m ² /d	Fiber: 100–500 (22%); 500–1000 (29%); 1000–5000 (50%)	Fiber, fragment	–	–	[20]
Paris, France -urban area	Dry and wet atmospheric fallout	Range: 2–355 particles/m ² /d; mean: 110 \pm 96 particles/m ² /d	50–3200	Fiber	–	RY, Acetate, PET, PA, PU	[25]
Paris, France - suburban area	Dry and wet atmospheric fallout	Mean: 53 \pm 38 particles/m ² /d	50–3200	Fiber	–	RY, Acetate, PET, PA, PU	[25]
Dongguan, China	Dry and wet atmospheric fallout	Range: 175–313 particles/m ² /d; range: 228 \pm 43 particles/m ² /d	200–2200 (dominant)	Fiber, fragment, film, foam	–	PE, PP, PS	[26]
Yantai, China	Dry and wet atmospheric fallout	Range: 130–624 particles/m ² /d	50–3000; 100–300 (dominant)	Fiber, fragment, film, foam	Fiber: white, black, red, transparent	PET, PVC, PE, PS	[21]
Pyrenees mountains, France	Dry and wet atmospheric fallout	Mean: 365 \pm 69 particles/m ² /d	Fiber: 50–700 (dominant); Fragment: <50 (>80%); Film: 50–200 (>70%)	Fiber, fragment, film	–	PS, PE, PP, PVC, PET	[27]
Tehran, Iran	Street dust	Range: 83 \pm 10–605 \pm 10 particles/30 g	Size range: 2–5000; >100 (dominant)	Fragment, fiber, spherule	Black, yellow, transparent, blue, red, green	–	[30]
Asaluyeh County, Iran-urban station	Street dust	Range: 52–7748 particles/15 g; mean: 900 particles/15 g	Size range: 2–5000; <100 (77%)	Spherule, film, fiber, fragment	White-transparent, yellow-orange, red-pink, blue/green, black-grey	–	[24]
China	Indoor dust	Range of fibers: 17–620 fibers/mg, mean of fibers: 342 fibers/mg; range of granules: 6–184 particles/mg	50–2000	Fiber, granule	–	PET, PAN, PA, PE, PP, PU, PEI, acrylic, alkyd, cellulose, rayon, protein	[29]
China	Outdoor dust	Mean of fibers: 114 fibers/mg; range of granules: 0–100 particles/mg	50–2000	Fiber, granule	–	–	[29]
Paris, France	Indoor dust	Range: 190–670 fibers/mg	50–5000	Fiber	–	–	[11]

^a Polyethylene terephthalate (PET), polyethylene (PE), polyester (PES), polyacrylonitrile (PAN), poly (*N*-methyl acrylamide) (PAA), rayon (RY), ethylene vinyl acetate (EVA), epoxy resin (EP), alkyd resin (ALK), polypropylene (PP), polyamide (PA), polyurethane (PU), polystyrene (PS), polyvinylchloride (PVC), Polyethylenimine (PEI).

transparent MPs were most abundant in street dusts from Asaluyeh County, Iran [24]. Black and yellow particles comprised more than 50% of total MPs in street dusts from Tehran, Iran, with a contribution of 29.9% and 26.4%, respectively, followed by transparent MPs (17.3%), blue MPs (12.6%), and red MPs (10.2%) [30]. Chemical identification was only performed for indoor dusts from China, with PET being the main polymer type [29]. Furthermore, high concentrations of PET have been found in both indoor and outdoor dust from China, with a range of 1550–120,000 mg/kg and 212–9020 mg/kg, respectively [29].

4. Potential sources, dispersion and deposition of airborne MPs

Synthetic textile is an important source for airborne MPs [13,22]. Small fibers used in clothing may be released into the atmosphere during clothing wears or drying [13,23,25]. Fibers used in soft furnishings such as carpets, curtains, etc., may also enter the atmosphere during their usage [24,25]. Furthermore, fibrous MPs may undergo photo-oxidative degradation, wind shear, or abrasion, and eventually be fragmented into fine particles [23]. Coating materials

may also be sources for airborne MPs. For example, epoxy and alkyd resins would gradually turn to fragment MPs in the atmosphere after long-term exposure to UV radiation and physical abrasion [13]. In addition, plastic mulching films, waste incineration, land-filling, sewage sludge used as fertilizer in agriculture, and synthetic particles in horticultural soils would be potential sources of airborne MPs [23,47]. Dust was considered as a sink of airborne MPs, on the other hand, MPs deposited on the ground would also resuspend and enter the air. Thus, dust could be suggested as a secondary source of airborne MPs.

MPs in the atmosphere could transport far away and deposit to the ground as well [20,27]. Source and transport analysis of MPs in the remote area of the Pyrenees Mountains indicated a source area that extended to 95 km from the sampling site [27]. Factors such as pollution concentration gradient, wind speed, wind direction, temperature and humidity would influence the dispersion and deposition of airborne MPs [22,47]. For example, SAMP abundance in Shanghai decreased as the distance to the coastline decreased because of the dilution effect of ocean air and the decreased influence of human activities [13]. Due to the role of dilution, MP levels in outdoor environments were significantly lower than those in indoor environments [11,29]. Precipitation was also indicated to play an important role in the deposition of MPs [22,23]. The lowest MP level was observed in total atmospheric fallout in Paris during a dry-weather period, while the highest MP level was observed during the period when almost daily rainfall occurred [20]. In addition, rain and snow was suggested to be the drivers for MP deposition in the remote mountain catchment of the Pyrenees Mountains [27].

5. Risk on human exposure to airborne MPs

Suspended airborne MPs may enter the respiratory system via inhalation. Assuming that the average air volume consumed by an adult was 15 m³/d, adult exposure to airborne MPs via inhalation by Shanghai residents would be 21 particles/d [13]. However, it should be noted that not all the suspended airborne MPs could reach and finally deposit in the deep lung. For example, fibrous MPs with length-to-diameter ratio greater than 3:1 may stay in the upper airways due to mucociliary clearance [23]. Furthermore, ecological risk from suspended airborne MPs in Shanghai was estimated by calculating the potential ecological hazardous single indices, with values ranging from 0.23 to 6.54, indicating a minor threat to the study area [13].

Dust ingestion is also a significant pathway for human exposure to harmful pollutants, especially for the sensitive population such as infants and toddlers. Previous studies indicated that human beings might be exposed to toxic pollutants such as heavy metals and persistent organic pollutants through dust ingestion [48–50]. Human intake of fibers via ingesting dust fall incidentally during a meal ranged from 13,731 to 68,415 particles/year/person in Scotland, which were much higher than the values via mussel consumption [28]. Based on the MP concentrations in dusts from the main cities of China, estimated daily intakes (EDIs) of fibrous and granule MPs via dust ingestion for different age groups of Chinese were estimated to be in the range of 64.1–889 fibers/kg-bw/d and 8.44–119 particles/kg-bw/d, respectively [29]. Furthermore, human uptake of PET and PC MPs by dust ingestion for the different age groups fell in the range of 6500–89,700 ng/kg-bw/d and 0.53–7.37 ng/kg-bw/d, respectively [29]. Highest EDI values were obtained for infants due to their lowest body weight, and higher dust ingestion rate. Human beings may also be exposed to MPs in street dust. The EDI values of MPs via street dust ingestion of Tehran, Iran ranged from 0.6 to 4.0 for children and from 0.3 to 2.0 particles/d for adults in normal exposure scenario (dust ingestion

rate: 200 mg/d for children; 100 mg/d for adults), and the ranges would increase to 3–20 particles/d and 1.0–6.7 particles/d, respectively, in acute exposure scenario (dust ingestion rate: 1000 mg/d for children; 330 mg/d for adults) [30]. Higher EDI values of MPs via street dust ingestion were obtained for children and adults in Asaluyeh County, Iran, with a range of 0.7–103.3 and 0.3–51.7 particles/d, respectively, in normal scenario, and the values were 3.5–516.5 particles/d and 1.1–170.5 particles/d, respectively, in acute scenario [24].

6. Conclusions

This review summarizes analytical methods, occurrence, potential sources, fate and potential risk of airborne MPs from available literatures. MPs could be detected not only in urban areas but also in remote mountains. Inhalation and dust ingestion were both important pathways for human exposure to airborne MPs. However, there are still some issues to be addressed in future studies on airborne MPs:

- (1) Standardization of analytical methods for airborne MPs lacked. As indicated by the overview, samples have not been digested in some studies, which would lead to over-estimation of airborne MPs due to the interference of organic materials such as cotton or wool. To remove as much of organic materials as possible, digestion of samples is suggested before visual observation.
- (2) It is important to identify the chemical characteristics of airborne MPs, whereas, in many studies, only physical characteristics were obtained. In addition, besides μ -FTIR and μ -Raman, hyperspectral imaging technique is also expected to identify the chemical composition of airborne MPs.
- (3) Mass concentrations of specific polymer types would provide more information for understanding the potential sources and risk of airborne MPs. However, data on mass concentrations of airborne MP polymers is limited. Therefore, quantification of specific MP polymers should be reinforced in future studies of airborne MPs. Furthermore, techniques such as Pys-GC-MS and TDS-GC-MS could be explored to quantify airborne MPs in future work.
- (4) In most studies, the observation size limit of airborne MPs was defined to 50 μ m. Nevertheless, microplastics with size less than 50 μ m are more prone to be inhaled by human beings and may cause greater health risk. Thus, more advanced analytical techniques should be explored to observe airborne MPs of small size.
- (5) Current data on the occurrence of airborne MPs are too limited and are insufficient to assess the pollution status and potential risk of airborne MPs in the global or regional range. More studies of airborne MPs in suspended particulates, atmospheric fallout and dust are required.

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References

- [1] M.A. Browne, P. Crump, S.J. Niven, E. Teuten, A. Tonkin, T. Galloway, R. Thompson, Accumulation of microplastic on shorelines worldwide: sources and sinks, *Environ. Sci. Technol.* 45 (2011) 9175–9179.

- [2] A.L. Andrady, Microplastics in the marine environment, *Mar. Pollut. Bull.* 62 (2011) 1596–1605.
- [3] Z. Fu, J. Wang, Current practices and future perspectives of microplastic pollution in freshwater ecosystems in China, *Sci. Total Environ.* 691 (2019) 697–712.
- [4] K. Zhang, H. Shi, J. Peng, Y. Wang, X. Xiong, C. Wu, P.K.S. Lam, Microplastic pollution in China's inland water systems: a review of findings, methods, characteristics, effects, and management, *Sci. Total Environ.* 630 (2018) 1641–1653.
- [5] L. Van Cauwenberghe, L. Devriese, F. Galgani, J. Robbins, C.R. Janssen, Microplastics in sediments: a review of techniques, occurrence and effects, *Mar. Environ. Res.* 111 (2015) 5–17.
- [6] J. Wang, J. Peng, Z. Tan, Y. Gao, Z. Zhan, Q. Chen, L. Cai, Microplastics in the surface sediments from the Beijing River littoral zone: composition, abundance, surface textures and interaction with heavy metals, *Chemosphere* 171 (2016) 248–258.
- [7] J. Zhao, W. Ran, J. Teng, Y. Liu, H. Liu, X. Yin, R. Cao, Q. Wang, Microplastic pollution in sediments from the Bohai sea and the yellow sea, China, *Sci. Total Environ.* 640–641 (2018) 637–645.
- [8] J. Wang, M. Wang, S. Ru, X. Liu, High levels of microplastic pollution in the sediments and benthic organisms of the South Yellow Sea, China, *Sci. Total Environ.* 651 (2019) 1661–1669.
- [9] D. He, Y. Luo, S. Lu, M. Liu, Y. Song, L. Lei, Microplastics in soils: analytical methods, pollution characteristics and ecological risks, *Trends Anal. Chem.* 109 (2018) 163–172.
- [10] J. Wang, X. Liu, Y. Li, T. Powell, X. Wang, G. Wang, P. Zhang, Microplastics as contaminants in the soil environment: a mini-review, *Sci. Total Environ.* 691 (2019) 848–857.
- [11] R. Dris, J. Gasperi, C. Mirande, C. Mandin, M. Guerrouache, V. Langlois, B. Tassin, A first overview of textile fibers, including microplastics, in indoor and outdoor environments, *Environ. Pollut.* 221 (2017) 453–458.
- [12] A.T. Kaya, M.Y. Yurtsever, S.Ç. Bayraktar, Ubiquitous exposure to microfiber pollution in the air, *Eur. Phys. J. - Plus* 133 (2018) 488.
- [13] K. Liu, X. Wang, T. Fang, P. Xu, L. Zhu, D. Li, Source and potential risk assessment of suspended atmospheric microplastics in Shanghai, *Sci. Total Environ.* 675 (2019) 462–471.
- [14] A. Dehaut, L. Hermabessiere, G. Duflos, Current frontiers and recommendations for the study of microplastics in seafood, *Trends Anal. Chem.* 116 (2019) 346–359.
- [15] E. Guzzetti, A. Sureda, S. Tejada, C. Faggio, Microplastic in marine organism: environmental and toxicological effects, *Environ. Toxicol. Pharmacol.* 64 (2019) 164–171.
- [16] C.G. Alimba, C. Faggio, Microplastics in the marine environment: current trends in environmental pollution and mechanisms of toxicological profile, *Environ. Toxicol. Pharmacol.* 68 (2019) (2019) 61–74.
- [17] H.S. Auta, C.U. Emenike, S.H. Fauziah, Distribution and importance of microplastics in the marine environment: a review of the sources, fate, effects, and potential solutions, *Environ. Int.* 102 (2017) 165–176.
- [18] Y. Pico, A. Alfarhan, D. Barcelo, Nano- and microplastic analysis: focus on their occurrence in freshwater ecosystems and remediation technologies, *Trends Anal. Chem.* 113 (2019) 409–425.
- [19] M. Yan, H. Nie, K. Xu, Y. He, Y. Hu, Y. Huang, J. Wang, Microplastic abundance, distribution and composition in the Pearl River along Guangzhou city and Pearl River estuary, China, *Chemosphere* 217 (2019) 879–886.
- [20] R. Dris, J. Gasperi, V. Rocher, M. Saad, N. Renault, B. Tassin, Microplastic contamination in an urban area: a case study in Greater Paris, *Environ. Chem.* 12 (2015) 592–599.
- [21] Q. Zhou, C. Tian, Y. Luo, Various forms and deposition fluxes of microplastics identified in the coastal urban atmosphere, *Chin. Sci. Bull.* 62 (2017) 3902–3910.
- [22] J.C. Prata, Airborne microplastics: consequences to human health? *Environ. Pollut.* 234 (2018) 115–126.
- [23] J. Gasperi, S.L. Wright, R. Dris, F. Collard, C. Mandin, M. Guerrouache, V. Langlois, F.J. Kelly, B. Tassin, Microplastics in air: are we breathing it in? *Curr. Opin. Environ. Sci. Health* 1 (2018) 1–5.
- [24] S. Abbasi, B. Keshavarzi, F. Moore, A. Turner, F.J. Kelly, A.O. Dominguez, N. Jaafarzadeh, Distribution and potential health impacts of microplastics and microrubbers in air and street dusts from Asaluyeh County, Iran, *Environ. Pollut.* 244 (2019) 153–164.
- [25] R. Dris, J. Gasperi, M. Saad, C. Mirande, B. Tassin, Synthetic fibers in atmospheric fallout: a source of microplastics in the environment? *Mar. Pollut. Bull.* 104 (2016) 290–293.
- [26] L. Cai, J. Wang, J. Peng, Z. Tan, Z. Zhan, X. Tan, Q. Chen, Characteristic of microplastics in the atmospheric fallout from Dongguan city, China: preliminary research and first evidence, *Environ. Sci. Pollut. Res.* 24 (2017) 24928–24935.
- [27] S. Allen, D. Allen, V.R. Phoenix, G.L. Roux, P.D. Jiménez, A. Simonneau, S. Binet, D. Galop, Atmospheric transport and deposition of microplastics in a remote mountain catchment, *Nat. Geosci.* 12 (2019) 339–344.
- [28] A.I. Catarino, V. Macchia, W.G. Sanderson, R.C. Thompson, T.B. Henry, Low levels of microplastics (MP) in wild mussels indicate that MP ingestion by humans is minimal compared to exposure via household fibres fallout during a meal, *Environ. Pollut.* 237 (2018) 675–684.
- [29] C. Liu, J. Li, Y. Zhang, L. Wang, J. Deng, Y. Gao, L. Yu, J. Zhang, H. Sun, Wide-spread distribution of PET and PC microplastics in dust in urban China and their estimated human exposure, *Environ. Int.* 128 (2019) 116–124.
- [30] S. Dehghani, F. Moore, R. Akhbarzadeh, Microplastic pollution in deposited urban dust, Tehran metropolis, Iran, *Environ. Sci. Pollut. Res.* 24 (2017) 20360–20371.
- [31] R.R. Hurley, A.L. Lusher, M. Olsen, L. Nizzetto, Validation of a method for extracting microplastics from complex, organic-rich, environmental matrices, *Environ. Sci. Technol.* 52 (2018) 7409–7417.
- [32] S. Klein, E. Worch, T.P. Knepper, Occurrence and spatial distribution of microplastics in river shore sediments of the rhine-main area in Germany, *Environ. Sci. Technol.* 49 (2015) 6070–6076.
- [33] A.S. Tagg, J.P. Harrison, Y. Ju-Nam, M. Sapp, E.L. Bradley, C.J. Sinclair, J.J. Ojeda, Fenton's reagent for the rapid and efficient isolation of microplastics from wastewater, *Chem. Commun.* 53 (2017) 372–375.
- [34] M. Scheurer, M. Bigalke, Microplastics in Swiss floodplain soils, *Environ. Sci. Technol.* 52 (2018) 3591–3598.
- [35] S.M. Mintenig, I. Int-Veen, M.G.J. Löder, S. Primpke, G. Gerdt, Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging, *Water Res.* 108 (2017) 365–372.
- [36] T. Mani, A. Hauk, U. Walter, P. Burkhardt-Holm, Microplastics profile along the rhine river, *Sci. Rep.* 5 (2015).
- [37] L. Lahens, E. Strady, T.C. Kieu-Le, R. Dris, K. Boukerma, E. Rinnert, J. Gasperi, B. Tassin, Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity, *Environ. Pollut.* 236 (2018) 661–671.
- [38] J. Sun, X. Dai, Q. Wang, M.C.M. van Loosdrecht, B.-J. Ni, Microplastics in wastewater treatment plants: detection, occurrence and removal, *Water Res.* 152 (2019) 21–37.
- [39] W. Wang, A.W. Ndungu, Z. Li, J. Wang, Microplastics pollution in inland freshwaters of China: a case study in urban surface waters of Wuhan, China, *Sci. Total Environ.* 575 (2017) 1369–1374.
- [40] W. Luo, L. Su, N.J. Craig, F. Du, C. Wu, H. Shi, Comparison of microplastic pollution in different water bodies from urban creeks to coastal waters, *Environ. Pollut.* 246 (2019) 174–182.
- [41] K. Zhang, X. Chen, X. Xiong, Y. Ruan, H. Zhou, C. Wu, P.K.S. Lam, The hydro-fluctuation belt of the Three Gorges Reservoir: source or sink of microplastics in the water? *Environ. Pollut.* 248 (2019) 279–285.
- [42] S. Fuller, A. Gautam, A procedure for measuring microplastics using pressurized fluid extraction, *Environ. Sci. Technol.* 50 (2016) 5774–5780.
- [43] J. Shan, J. Zhao, Y. Zhang, L. Liu, F. Wu, X. Wang, Simple and rapid detection of microplastics in seawater using hyperspectral imaging technology, *Anal. Chim. Acta* 1050 (2019) 161–168.
- [44] J. Shan, J. Zhao, L. Liu, Y. Zhang, X. Wang, F. Wu, A novel way to rapidly monitor microplastics in soil by hyperspectral imaging technology and chemometrics, *Environ. Pollut.* 238 (2018) 121–129.
- [45] M. Fischer, B.M. Scholz-Böttcher, Simultaneous trace identification and quantification of common types of microplastics in environmental samples by pyrolysis-gas chromatography–mass spectrometry, *Environ. Sci. Technol.* 51 (2017) 5052–5060.
- [46] E. Dümichen, A.K. Barthel, U. Braun, C.G. Bannick, K. Brand, M. Jekel, R. Senz, Analysis of polyethylene microplastics in environmental samples, using a thermal decomposition method, *Water Res.* 85 (2015) 451–457.
- [47] S. Karbalaei, P. Hanachi, T.R. Walker, M. Cole, Occurrence, sources, human health impacts and mitigation of microplastic pollution, *Environ. Sci. Pollut. Res.* 25 (2018) 36046–36063.
- [48] L. Chen, Y. Huang, Z. Xu, L. Wen, X. Peng, Z. Ye, S. Zhang, X.-Z. Meng, Human exposure to PBDEs via house dust ingestion in Guangzhou, South China, *Arch. Environ. Contam. Toxicol.* 60 (2011) 556–564.
- [49] J. Wang, Y. Wang, Z. Shi, X. Zhou, Z. Sun, Legacy and novel brominated flame retardants in indoor dust from Beijing, China: occurrence, human exposure assessment and evidence for PBDEs replacement, *Sci. Total Environ.* 618 (2018) 48–59.
- [50] L. Zhou, G. Liu, M. Shen, R. Hu, M. Sun, Y. Liu, Characteristics and health risk assessment of heavy metals in indoor dust from different functional areas in Hefei, China, *Environ. Pollut.* 251 (2019) 839–849.