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Critical Review

A Review of Microplastics in Table Salt, Drinking Water, and Air: Direct Human Exposure

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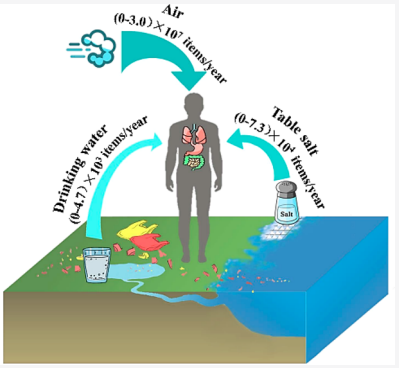
ABSTRACT: The ubiquity of microplastics in aquatic and terrestrial environments and related ecological impacts have gained global attention. Microplastics have been detected in table salt, drinking water, and air, posing inevitable human exposure risk. However, rigorous analytical methods for detection and characterization of microplastics remain scarce. Knowledge about the potential adverse effects on human health via dietary and respiratory exposures is also limited. To address these issues, we reviewed 46 publications concerning abundances, potential sources, and analytical methods of microplastics in table salt, drinking water, and air. We also summarized probable translocation and accumulation pathways of microplastics within human body. Human body burdens of microplastics through table salt, drinking water, and inhalation were estimated to be $(0-7.3)\times10^4$, $(0-4.7)\times10^3$, and $(0-3.0)\times10^7$ items per person per year, respectively. The intake of microplastics via inhalation, especially via indoor air, was much higher than those via other exposure routes. Moreover, microplastics in the air impose threats to both respiratory and digestive systems through breathing and ingestion. Given the lifetime inevitable exposure to microplastics, we urgently call for a better understanding of the potential hazards of microplastics to human health.



INTRODUCTION

Plastic production and use have been growing rapidly since the 1950s, due to the superb properties of plastics such as low cost, versatility, and durability. The widespread use of plastic products has generated large amounts of plastic wastes. Recent modeling results predicted that global plastic waste will triple to 270 million tons from 2015 to 2060.[1] Plastic wastes have undoubtedly aggravated environmental pollution.[2,3] Upon entering the environment, plastic wastes will continuously break down to small fragments and particles.[4] Our current knowledge on environmental behavior and ecological impacts of small plastic fragments and particles is limited, which further complicates the issue of plastic pollution. For example, elimination of micro- and nanosized plastics from the environment is more challenging than bigger plastic debris.

Since the concept of "microplastic" was introduced in 2004,[5] microplastics (MPs) have been found in various environmental compartments and organisms globally.[6−10] Until now, more than 690 marine species have been reported to be contaminated by MPs.[11,12] Numerous experiments have demonstrated toxic effects of MPs, such as growth inhibition, oxidative damage, and immune stress.[13,14] A recent study suggested that high concentration of MPs may have caused direct life history responses in algae and *Daphnia* populations.[15] Microplastic particles can also accumulate in marine organisms and transfer through the food chain to higher trophic levels including humans.[9]

More recently, potential threats of MPs to human health have attracted intense attention because of the widespread detection of MPs in human-related food and environments, such as honey,[16] milk,[17] beer,[18] seafood,[19] table salt,[20,21] drinking water,[22] and air.[23] Consumption of some food products such as seafood, honey, and beer can be intentionally minimized or avoided, but exposure to MP-contaminated table salt, drinking water, and air is inevitable.[24] Despite the small daily intake of salt compared with the other exposure routes presented, salt MP contamination is significant in some regions, for example, in Croatia $(1.4-2.0)\times10^4$ items·kg^{-1} salt and Italy $(1.6-8.2)\times10^3$ items·kg^{-1} salt.[25] Besides, the actual salt intake can be much higher (e.g., 10 g·d^{-1} worldwide and 18 g·d^{-1} in Turkey) than the recommended intake threshold of 5 g·d^{-1} by the World Health Organization.[26,27] Microplastics in table salt and drinking water can enter human body through the digestive tract, whereas MPs in the air can cause exposure of both digestive and respiratory systems. Suspended MPs can be inhaled and deposited MPs can be ingested through hand-to-mouth contact, especially for children.[23,28] Although based on a relatively small sample size, the first evidence of MPs found in human stools suggests that humans are being exposed to MPs.[29]

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Table 1. Summary of Microplastics in Table Salt

country	extraction	separation	pore size (μm)	abundances (item·kg^{-1})			size (μm)	references
				sea salt	lake salt	rock salt		
Australia	UW[a]	NaI	149	0−9	-[b]	-	160−980	37
	17% H_2O_2	-	2.7	46	-	-	100−5000	36
Belarus	17% H_2O_2	-	2.7	-	-	8	100−5000	36
Brazil	17% H_2O_2	-	2.7	2.0×10^2	-	-	100−5000	36
Bulgaria	17% H_2O_2	-	2.7	12	-	-	100−4000	36
China	30% H_2O_2	-	5	$(5.5-6.8)\times10^2$	43−364	7−204	45−4300	20
	17% H_2O_2	-	2.7	$0-1.7 \times 10^3$	28	0−14	100−4000	36
	UW	-	5	9.8	-	-	1−1500	34
Croatia	UW	-	0.45	$(1.4-2.0)\times10^4$	-	-	15−4628	25
	UW	-	0.2	$(0.7-2)\times10^2$	-	-	10−150	33
	17% H_2O_2	-	2.7	58	-	-	100−5000	36
France	UW	NaI	149	0−2	-	-	160−980	37
	17% H_2O_2	-	2.7	0	-	-	-	36
Germany	17% H_2O_2	-	2.7	-	-	2	100	36
Hungary	17% H_2O_2	-	2.7	-	-	12	100−4000	36
India	30% H_2O_2	-	0.45	$(0.6-1.0)\times10^2$	-	-	500−2000	30
	17% H_2O_2	-	2.7	$(0.3-3.7)\times10^2$	-	-	1000−5000	36
Indonesia	17% H_2O_2	-	2.7	1.4×10^4	-	-	-	36
	UW	-	0.45	6.7−53.5	-	-	390−9360	35
Iran	UW	NaI	149	-	1	-	160−980	37
Italy	UW	-	0.45	$(1.6-8.2)\times10^3$	-	-	4−2100	25
	17% H_2O_2	-	2.7	4−30	-	80	100−5000	36
	UW	-	0.2	$(1.7-3.2)\times10^2$	-	-	10−150	33
Japan	UW	NaI	149	0	-	-	-	37
Korea	17% H_2O_2	-	2.7	$(1.0-2.3)\times10^2$	-	-	100−3000	36
Malaysia	UW	NaI	149	1	-	-	160−980	37
New Zealand	UW	NaI	149	0−1	-	-	160−980	37
Pakistan	17% H_2O_2	-	2.7	-	-	100	100−5000	32
Philippines	17% H_2O_2	-	2.7	-	-	120	100−5000	36
Portugal	UW	NaI	149	0−10	-	-	160−980	37
Senegal	17% H_2O_2	-	2.7	48	800	-	100−3000	36
South Africa	UW	NaI	149	1−3	-	-	160−980	37
	UW	NaI	0.2	-	-	-	0−2000	39
Spain	UW	-	5	$(0.5-2.8)\times10^2$	-	115−185	30−3500	32
Thailand	17% H_2O_2	-	2.7	$(0.7-4.0)\times10^2$	-	-	100−5000	36
Turkey	30% H_2O_2	NaI	0.2	16−84	8−102	9−16	20−5000	31
U.S.A.	UW	-	11	$(0.5-8.0)\times10^2$	-	113−367	100−5000	18
	17% H_2O_2	-	2.7	32	-	5	100−1000	36
U.K.	17% H_2O_2	-	2.7	1.4×10^2	-	-	100−2000	36
Vietnam	17% H_2O_2	-	2.7	76−88	-	-	100−5000	36

[a]Ultrapure Water. [b]No data.

Assessing human health risk of MPs remains in its infancy with limited information on exposure routes, biological fates, and health effects. This review aims to survey our current knowledge on direct human exposure to MPs via the three main exposure pathways: table salt, drinking water, and air. It also provides an overview of potential health effects associated with different potential exposure routes. Data from peer-reviewed papers, books, and reports related to MPs in table salt, drinking water, and air published by the end of January 2020 were collected and summarized. The keywords used in iterative literature search were microplastics, table salt, drinking water, air, atmospheric, dust, ingestion, intake, toxicology, risk, and human health. The searched resources included Science Direct, Web of Science, Directory of Open Access Journals (DOAJ), EBSCOhost, Spring Link, Wiley Online Library, BioMed Central, and PubMed Central. In total, 46 publications focusing on the occurrence of MPs in table salt, drinking water, and air were analyzed. The abundances and analytical methods of MPs were summarized and classified in tables and figures. All raw data extracted from the literature were presented in mean values or range values and expressed in unified units. Other literatures concerning ecological hazards, health risk, toxicology, and seafood were also selected and discussed after the initial screening.

MICROPLASTICS IN TABLE SALT

Occurrence and Abundance. Microplastics have been widely detected in table salt of > 100 brands all over the world (Table 1).[30−35] The abundances of MPs in table salt varied widely. The highest abundance was reported in Croatia (1.4×10^4−2.0×10^4 items·kg^{-1}),[25] followed by Indonesia (1.4×10^4 items·kg^{-1}),[36] Italy (1.6×10^3−8.2×10^3 items·kg^{-1}),[25] the United States (0.5×10^2−8.0×10^2 items·kg^{-1}),[18] and China (5.5×10^2−6.8×10^2 items·kg^{-1}).[20] However, MP pollution in

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Table 2. Number of Different Analytical Methods Applied for Microplastics Analysis in Table Salt, Drinking Water, and Air

	table salt	drinking water			air		
analytical method	sea/lake/well salt	DWTP[a] water	tap water	bottled water	wet and dry deposition	dust	air sampler
dissolution	12[b]	0	0	0	0	0	0
digestion	5	3	0	0	3	6	0
flotation							
NaCl[c]	9	0	0	0	0	0	0
NaI	3	0	0	0	0	0	0
$ZnCl_2$	0	1	0	0	1	3	0
filtration	(pore size)						
<1 μm	6	2	1	1	3	2	2
1−5 μm	3	2	2	2	4	3	7
>5 μm	3	0	0	0	1	2	1
identification	Methods						
μ-FTIR	9	3	2	1	5	5	8
μ-Raman	2	2	0	2	2	0	0
others[d]	1	1	1	2	0	3	2

[a]Indicates drinking water treatment plant. [b]Indicates the number of studies that have applied the corresponding analytical processes. [c]Indicates no additional flotation agent (i.e., filtration of supernatant alone, filtration of all salty solution including deposited sediment, or filtration of supernatant and deposited sediment separately). [d]Includes dyeing, SEM-EDX, and fluorescence.

different regions cannot be directly compared with each other due to different analytical methods used. A recent study compared the MP abundances in table salts collected from different regions, using sea salt as a seawater MP pollution indicator, and found a significantly higher MP abundance in Asia than in other continents.[36] Relatively low abundances of MPs were reported in table salts from Australia, France, Iran, Japan, Malaysia, Zealand, Portugal, and Africa.[37] This was probably caused by the usage of filters with a large pore size (149 μm), allowing smaller-sized MPs to escape in the filtration process and resulting in underestimated MP abundances.[37]

Source Diagnostics. Table salts can be sourced from seas, rocks, or salt lakes. Several studies found that the abundance of MPs was higher in sea salt than in rock salt or lake salt,[20] which could be explained by higher level of MP pollution in coastal zones. However, such a source-specific difference was not found by Iniguez et al.[32] The presence of MPs in rock/well salts suggests that MPs may be introduced during collection, transportation, drying, or packaging processes.[20] Therefore, the general public should pay particular attention to food production, because other commercial foods may also be produced and packed in a similar manner as that for table salt.[38] In contrast, another study found that the origin of MPs in table salt was irrelevant to the packaging or grinding process,[32] implicating for other potential sources of MP contamination during concentration, crystallization, or refinement, such as airborne MPs.

Analytical Methods. The various analytical methods used for MPs in table salt, drinking water, and air are summarized in Table 2. The common analytical method for determining MPs in table salt includes sample collection, dilution, extraction, observation, and identification. However, the differences in experimental instrument, extraction reagent, and filter pore size lead to low comparability of the results among different studies, which urgently call for a standard analytical method. The first step of establishing a standard analytical method is to consider sample quantity as well as brand or type of salts. Three types of salts (sea, lake, and rock salts) and three or more brands are recommended so as to prevent either overestimation or underestimation of MP abundances in salt from a region. Sufficient amount of salt is needed to achieve reasonable detection sensitivity. Based on the results of our group[20] and other groups,[25,32] 100−250 g of salts per sample are suggested. It should be noted that the sample amount is empirical. Reducing the salt quantity would reduce the detection frequency. Conversely, the filter membrane is likely to be clogged by excessive impurities such as soil and organic matter with larger salt sample amount.[31,37] The recommended sample amount is expected to be decreased with the development of identification technologies in future. H_2O_2 has been used to digest organic matter in 40% of the studies,[36] while some investigators believe such digestion is not necessary due to small amounts of organic matter in table salts.[25] Additional flotation agent is commonly excluded, and only three studies used saturated NaI solution as flotation agent to isolate MPs (Table 2).[31,37,39] Although NaI saturated solution (1.8 $g \cdot cm^{-3}$) can enhance MP separation, its use is not recommended for the following reasons: (1) The color of NaI would interfere with MP identification; (2) NaI solution reacts with H_2O_2; and (3) NaI is also an environmental pollutant. Generally, the number of other particles and impurities in table salt is relatively low. Thus, the priority option is to filter all solutions after salt sample dissolution. In the case of large numbers of impurities in table salt, saturated NaCl solution is suggested to be used as a flotation agent. NaCl solution has been proven efficient for separating MPs, including PS, PA, PP, PVA, and PE with recovery rates of 85−95%.[40,41] Other flotation agents (e.g., $ZnBr_2$, $ZnCl_2$, and NaBr) were reported to produce high recovery rates, but all factors including cost, practicability, and environmental friendliness must be taken into consideration.[40,42,43] Filtration is a critical step for MP extraction. The use of different filter membranes with diverse pore sizes ranging from 0.2 to 149 μm impedes the standardization of analytical methods. A 5 μm pore size is recommended for filtration, followed by identification using μ-

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Table 3. Summary of Microplastics in Drinking Water

sampling and locations	pore size (μm)	abundances (item·L^{-1})	size (μm)	references
Drinking Water Treatment Plants				
Germany	3	0−7 × 10^{-3} (raw water)	50−150	45
	3	7 × 10^{-4} (drinking water)	50−150	45
Czech	0.2	(1.5−3.6)×10^{3} (raw water)	1−10	46
	0.2	(3.4−6.3)×10^{2} (drinking water)	1−10	46
Norway	1.2	0	-[b]	47
China	0.22	6.7 × 10^{3} (raw water)	1−100	48
	0.22	9.3 × 10^{2} (drinking water)	1−100	48
Tap Water				
U.K.	2.5	7.7 (3.7−13.0)	100−5000	18
Germany	2.5	0.9 (0−1.8)	100−5000	18
Ireland	2.5	1.8	100−5000	18
Italy	2.5	0	100−5000	18
Slovakia	2.5	3.8 (0−10.9)	100−5000	18
Switzerland	2.5	2.7 (0−5.5)	100−5000	18
U.S.A.	2.5	9.2 (0−60.9)	100−5000	18
Denmark	0.2	0	-	50
India	2.5	6.2 (0−20)	100−5000	18
Indonesia	2.5	3.2 (0−10.8)	100−5000	18
Lebanon	2.5	6.6 (0−23.3)	100−5000	18
Uganda	2.5	3.9 (0−12.7)	100−5000	18
Cuba	2.5	7.2	100−5000	18
Tap Water				
Ecuador	2.5	4.0 (0−9.0)	100−5000	18
Ecuador	2.5	-	-	49
Bottled Water				
Germany	0.4	2.6 × 10^{3} (PET[a] bottle)	0−5	51
	0.4	4.9 × 10^{3} (reusable PET bottle)	0−5	51
	0.4	6.3 × 10^{3} (glass bottle)	0−10	51
Germany	3	11 (beverage carton)	5−100	52
	3	50 (glass bottle)	5−100	52
	3	118 (returnable plastic bottle)	5−100	52
	3	14 (single-use plastic bottle)	5−100	52
Italy	-	5.4 × 10^{7}	0−10	53
U.S.A.	1.5	58−1.4 × 10^{3}	6.5−5000	22
Mexico	1.5	(0.2−6.9)×10^{2}	6.5−5000	22
Brazil	1.5	(0.1−1.5)×10^{2}	6.5−5000	22
Lebanon	1.5	49.3	6.5−5000	22
Thailand	1.5	4.7 × 10^{2}	6.5−5000	22
China	1.5	(0.7−1.6)×10^{2}	6.5−5000	22
Indonesia	1.5	(0.4−7.1)×10^{2}	6.5−5000	22
India	1.5	0−39	6.5−5000	22
Kenya	1.5	74.6	6.5−5000	22

[a]PET = Polyethylene terephthalate. [b]No data.

FTIR, which is a reliable approach for determining the chemical composition of MPs.[44]

MICROPLASTICS IN DRINKING WATER

Occurrence and Abundance. Only 10 studies have investigated on MP contamination in drinking water (Table 3), covering raw and treated water from drinking water treatment plants (DWTPs),[45−48] tap water,[18,49,50] and bottled water[22,51−53] from 22 countries. These data suggested that particles larger than 50 μm can be removed from raw water by traditional drinking water treatments with removal rates in the range of 25−90%, depending on local treatment technologies.[46] The lowest abundance of MPs in tap water was observed in Italy and Denmark (0 items·L^{-1}), while the highest abundance was found in the United States (9.2 items·L^{-1}).[20] The abundance of MPs in bottled water varied from 0 to 5.4 × 10^{7} items·L^{-1}.[22,50−52] Water in returnable-used plastic bottles contained significantly more MPs compared with that in single-used bottles.[51] Similar to table salt, a direct comparison of MP abundances in drinking water samples from different studies is difficult due to the use of filter membranes with different pore sizes and different identification methods.

Source Diagnostics. Surface water and groundwater are important drinking water sources.[54,55] Given that macroplastics and MPs have been widely identified in freshwater bodies,[56] MPs in drinking water are usually believed to originate from polluted freshwater resources, such as lakes, rivers, canals, and groundwater.[45] However, some freshwater bodies are less polluted by MPs compared with tap water and bottled water.[57] As summarized by Koelmans et al.,[57] groundwater (1 × 10^{-2} items·L^{-1}) has the lowest MP abundance in all types of fresh water. Therefore, it is possible that MPs found in drinking water are derived from water supply chain or product packages such as caps and bottle walls.[51] Schymanski et al.[47] showed that the majority types of MPs in bottled water were polyethylene terephthalate and polyester which may be derived from the materials of the bottles. Unexpectedly, large amounts of MPs were also found in glass bottled water (6.3 × 10^{3} ± 1.1 × 10^{4} items·L^{-1}) and the potential source is the abrasion of plastic bottle cap against the glass bottle body.[47] Thus, we consider the packaging process as an important source of MPs for bottled water.

Detection Methods. The analytical methods used for MP detection in drinking water are simple and they share more common steps. Sampling and treatment methods, as well as

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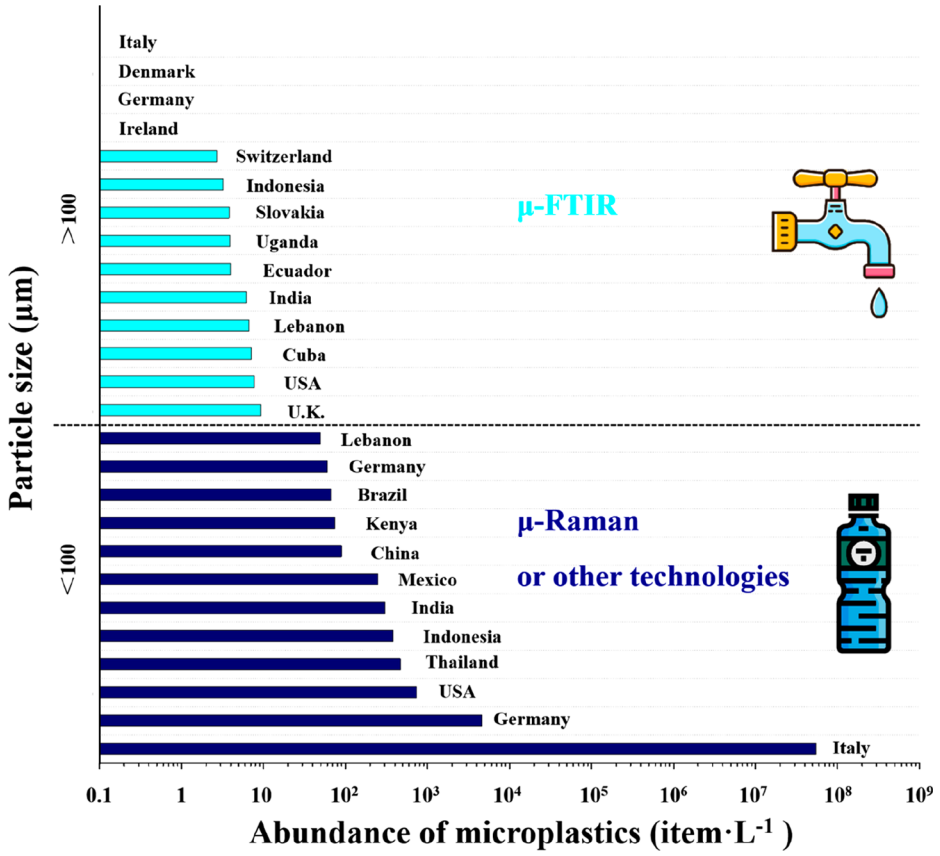

Figure 1. Abundance of microplastics in tap water and bottled water. To show the influence of identification methods on the existing data (size and abundance), the results were classified into two categories according to their identification methods, that is, (1) μ-FTIR and (2) μ-Raman or other technologies (dyeing, SEM-EDX). The microplastics >100 μm with <5% in abundance in bottled water were ignored for clearer comparison.

precautions, have been elaborated by Koelmans et al.[57] The present review only focuses on the methods for filtration and identification of MPs in drinking water. Surprisingly, the abundance of MPs has an up to 11 orders of magnitude difference among samples (Table 3). One of the main factors may be the pore size of filter membrane. For example, MP abundances obtained with 0.4 μm pore size filters[51] were much higher (2.6×10^3–6.3×10^3 items·L^{-1}) than those (0.1×10^2-1.2×10^2 items·L^{-1}) using 3 μm pore size filters,[52] both in bottled water from Germany. As approximately 50% of MPs were smaller than 1.5 μm,[51] most small MPs may be lost if the solution is filtered with 3 μm pore size filters. Therefore, unified membrane pore sizes are necessary for meaningful assessment of MP abundances.

In addition to pore size of filter membrane, the difference in identification methods is another crucial factor affecting the size ranges and abundances of detected MPs. For instance, MPs in tap water were often analyzed by μ-FTIR with a size of > 20 μm captured, whereas MPs in bottled water were normally processed by μ-Raman or other technologies (e.g., dyeing and SEM-EDX) capable of detecting smaller MPs (<10 μm). The results of MPs in drinking water, therefore, can be classified into two groups based on identification methods, that is, 1) μ-FTIR method and 2) μ-Raman or other technologies. Microplastics in tap water are larger in size and lower in abundance, whereas they are smaller in size but higher in abundance in bottled water (Figure 1). It is critical to point out that the reported higher MP abundance in bottled water than that in tap water is likely due to the use of identification method with lower size detection limit. In other words, the reported MP abundance in tap water or other type of samples where only μ-FTIR was used can be underestimated due to the instrumental incapability of detecting MPs smaller than 10 μm. This notion is corroborated by Pivokonsky et al.[46] who obtained high abundances (3.4×10^2 to 3.6×10^3 items·L^{-1}) of small-sized MPs (1–10 μm) in DWTPs using a μ-Raman approach.

MICROPLASTICS IN THE AIR

Occurrence and Abundance. Occurrence of MPs in the air has attracted increasing attention since 2015. Three different sampling methods have been used to collect atmospheric MPs, that is, wet and dry deposition,[23,58–62] atmospheric sampling,[63–69] and dust collection,[43,70–74] which makes a direct comparison of studies employing different sampling approaches not feasible. The size of fibers (the largest dimension of a MP fiber is defined as its size[75]) in the air is in the range of 100–5000 μm,[43,58,59] but much smaller particles can be detected using air samplers.[64,76] The width of MP fibers is small, about a few micrometers to tens of micrometers.[77,78] Some general trends can be found in Table 4. For example, the MP abundance in the air was higher in an urban area than in a suburban area in Paris.[59] Meteorological factors largely determine the dispersion and levels of MPs in the air. For instance, the lowest level of MPs was observed during dry weather periods, whereas the highest level occurred during rainy seasons.[23] In rainy days, rainfalls wash out fibers, inflating the amounts of MPs collected by wet and dry deposition

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Table 4. Summary of Microplastics in the Air

sampling and location	types of samples	pore size (μm)	abundance	size (μm)	references
Wet and Dry Deposition (item·m^{-2}·day^{-1})					
China (Dongguan)	outdoor	1	36	0−5000	58
France (Paris)	outdoor	1.6	1.2×10^2	100−5000	23
	urban	1.6	1.1×10^2	0−5000	59
	suburban	1.6	53	0−5000	59
	indoor	1.6	$(0.2-1.1)\times10^4$	0−5000	63
Germany (Hamburg)	outdoor	5−13	2.8×10^2	0−5000	60
China (Yantai)	outdoor	5	4.0×10^2	50−1000	61
U.K. (London)	outdoor	0.2	7.7×10^2	0−3000	62
France (Pyrenees mountains)	outdoor	0.45	3.7×10^2	0−750	79
Air Sampler (item·m^{-3})					
France (Paris)	indoor	1.6	0.8−6.0 (location 1)	0−3250	63
		1.6	1.3−19.6 (location 2)	0−3250	63
		1.6	0.4−5.4 (location 3)	0−3250	63
Denmark (Aarhus)	indoor	0.8	14.0 ± 2.2 (location 1)	11−105	65
		0.8	10.6 ± 5.9 (location 2)	11−105	65
		0.8	3.4 ± 2.6 (location 3)	11−105	65
France (Paris)	outdoor	1.6	0.01−0.5	0−1650	63
Iran (Asaluyeh)	outdoor	2	1 (0.3−1.1)	2−100	64
China (Shanghai)	outdoor	1.6	0.4 (0−2)	12−2191	66
West Pacific Ocean	outdoor	1.6	0.06 (0−1.4)	16.14−2086	67
Indonesia (Surabaya)	outdoor	1.6	$(1.3-1.8)\times10^4$	0−5000	68
Pearl River Estuary	outdoor	1.6	4.2×10^{-2}	59−2252	69
South China Sea	outdoor	1.6	$(0.8-1.3)\times10^{-2}$	59−2252	69
East Indian Ocean	outdoor	1.6	$(4-6)\times10^{-3}$	59−2252	69
Turkey (Sakarya)	outdoor	50	0.3−12.9	50−500	74
China (Shanghai)	outdoor	1.6	1.42 (0−4.18)	23−9955	76
China (Beijing)	outdoor	0.8	5.7×10^3 (location 1)	5−200	78
		0.8	5.6×10^3 (location 2)	5−200	78
Sweeping Operation[a]					
Iran (Tehran) (item·g^{-1})	outdoor dust	2	2.7−20	0−5000	43
Iran (Asaluyeh) (item·g^{-1})	outdoor dust	2	60 (3.3−67)	1000−5000	64
China (39 cities) (mg·g^{-1})	indoor dust	-[b]	27 (PET); 4.6×10^{-3} (PC)	50−2000	70
	outdoor dust	-	2.8 (PET); 2.0×10^{-3} (PC)	50−2000	70
12 countries (mg·g^{-1})	indoor dust	-	2.9×10^{-2}−1.1×10^2 (PET)	-	71
		-	1.1×10^{-4}−0.8 (PC)	-	71
Forni Glacier (item·g^{-1})	cryoconite	0.45	7.1×10^{-2}	100−5000	72
Japan (item·m^{-2})	outdoor dust	100	2.0 ± 1.6	100−5000	73
Vietnam (item·m^{-2})	outdoor dust	100	19.7 ± 13.7	100−5000	73
Nepal (item·m^{-2})	outdoor dust	100	12.5 ± 10.1	100−5000	73
Turkey (item·g^{-1})	outdoor dust	50	18−29	50−500	74
Arctic Fram Strait (item·L^{-1})	snow	-	$0-1.4 \times 10^4$ (Arctic snow)	11−475	77
		-	1.9×10^2−1.5×10^5 (European snow)	11−475	77

[a]The data units of sweeping operation were inconsistent due to the differences in identification methods or sample quantification; the data of Iran (Tehran and Asaluyeh) and Turkey were selected for intake calculation. [b]No data.

method.[59] In addition to larger populations, weaker airflows in urban areas also greatly contribute to higher atmospheric levels (and therefore stronger deposition) of MPs compared to rural areas.[59] More suspended MPs have been found in indoor environments than in outdoor environments.[63] Road dusts have also been recognized as an important source of MPs in urban areas.[43,64] Moreover, children may directly ingest large amounts of deposited dust through mouthing toys and hands. Abbasi et al.[64] calculated that more than 900 MP particles may be ingested by a child per year through dust ingestion (200 mg·day^{-1}) in a normal exposure scenario.

Source Diagnostics. Synthetic textiles (e.g., plastic fibers or fragments from clothes), rubber tire erosion, and road dust are considered as the major sources of primary atmospheric MPs, which can be transferred to other environmental compartments by winds.[79] Other sources of MPs in the air may be household furniture products, building materials, rubbish incineration, landfills, industrial discharge, and particulates emitted by vehicles.[23,59,63] In addition, the horticulture field also releases MPs through synthetic particles used in soils as well as sewage sludge used as fertilizers.[80]

Analytical Methods. Different sampling methods can be selected based on specific objectives (Figure 2). Wet and dry

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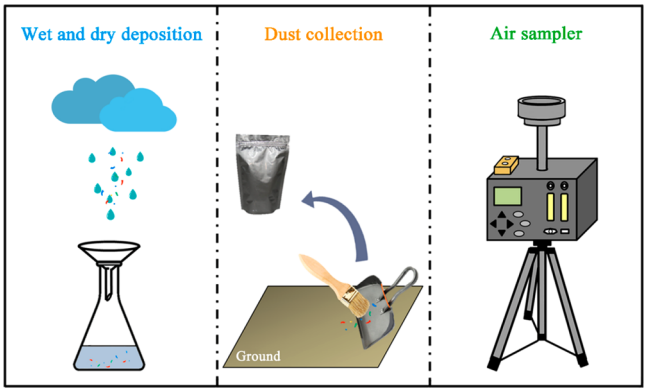

Figure 2. Sampling methods for microplastics in the air.

deposition, for instance, is simple and suitable for monitoring total MPs,[23,58,59] whereas atmospheric sampling within a breathing zone is more appropriate for estimating human inhalation.[64,76] Dust ingestion usually occurs in children or construction workers; therefore, sampling deposited dust is of significance.[70]

Subsequent treatment procedures depend on the type of samples (Table 2). Generally, aqueous samples obtained by wet and dry deposition require filtration.[58] For air samplers, the filters can be detached from inside of the device for direct observation.[64,76] MPs on air sampler filters can also be washed off and filtered again for further analysis.[64,70] Subsequent digestion (e.g., 30% H_2O_2) and flotation (e.g., saturated NaCl solution) are necessary for dust samples.[55] μ-FTIR is commonly used for identifying MPs in air samples, and other methods including SEM-EDX,[43] fluorescence microscopy,[64] and μ-Raman,[79] are also used.

EXPOSURE PATHWAYS AND HUMAN HEALTH RISK OF MICROPLASTICS

Human Body Burden. The most common routes for MPs to penetrate into the human body are ingestion and inhalation. Contaminated food (e.g., table salt, drinking water, and seafood, etc.) and dust containing deposited MPs from air are the sources of gastrointestinal exposure, and suspended MPs in the air may enter the respiratory system. An estimate on the body burdens of MPs was made based on the abundances of MPs detected in table salt, drinking water, and air and the average exposure rate of each route (Figure 3). The abundances of MPs were extracted from the "abundance" column in Tables 1, 3, and 4. Mean value was used if there was one, and maximum and minimum were used when range values were available. Although not the focus of the present review, the ingestion of MPs via seafood consumption was also compared with the other three media amid the availability of numerous data on MPs in seafood (Figure 3). For statistical analysis of the MP intake comparison, normality of the data was tested with Shapiro-Wilk's test. The Kruskal−Wallis test was then used followed by the Mann−Whitney U test using Bonferroni correction to adjust the probability (SPSS 22.0).

We first estimated the intake of MPs through gastrointestinal exposure. The abundance of MPs in table salt ranges widely from 0 to 2.0 × 10^4 items·kg^{-1} (Table 1). When global mean value (10 g·day^{-1}) is selected as salt exposure rate,[81] the intake of MPs from salt is estimated to range from 0 to 200 items per day, equivalent to 0−7.3 × 10^4 items per year (Figure 3). With consumption of 1.4 L water per day,[36,82] the annual MP intake

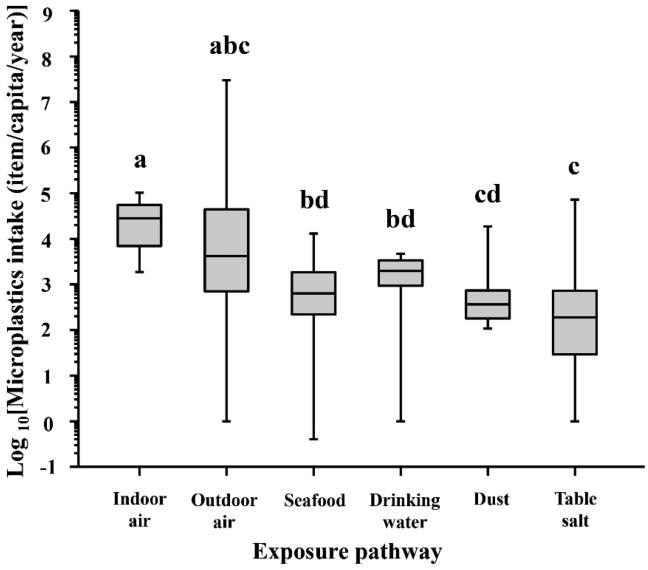

Figure 3. Human intake of microplastics through different exposure pathways. Maximum, minimum, and mean values obtained from literatures were plotted. The upper and lower boundaries of each box represent the 75th and 25th percentiles, respectively. The horizontal line inside the box represents the median value. The whisker represents the maximum or minimum value. The minimum values of MP intake through outdoor air, drinking water, and table salt were zero, which were not suitable for logarithmic representation and were not shown. The general ratio of the soft tissue mass to the total mass of shellfish (0.4) reported in previous surveys was used for microplastic intake calculation via seafood consumption.

through tap water and bottled water, is estimated to be 0−2.8 × 10^{10} items. The worst scenario can be calculated based on the abundance of MPs in bottled water of Italy (5.4 × 10^7 items·L^{-1}).[51] However, no detailed procedures for sample preparation and detection was provided in this study.[48] Because MPs detected in bottled water showed very different size fractions (<10 μm) from those detected in other samples (i.e., tap water, air, and table salt), we only calculated human MP intake through drinking water (0−4.7 × 10^3 items per year; Figure 3) using the tap water data, without considering the bottled water data.

Another gastrointestinal exposure pathway of MPs is dust ingestion, especially through mouthing dirty toys and hands by children.[83] After excluding the data of cryoconite,[72] only three studies have been conducted on MPs in dust with the unit of item·g^{-1}, and the abundance ranges are too limit for calculating human MP intake.[43,74,84] Therefore, we used the original abundance data of each sampling site from these articles. The range of MP intake through dust ingestion is estimated to be 1 × 10^2 to 1.9 × 10^4 items per year for adults, based on the average dust exposure of 100 mg·day^{-1} (Figure 3).[43] It needs to be acknowledged that the actual individual intake may be influenced by "activity", for example, the portions of outdoor and indoor activities. Among various food items, the presence of MPs in seafood has been widely recognized.[85−87] We extracted the MP abundances in seafood from Hantoro et al.[88] and Li et al.[10] as well as some more recent studies (SI Table S1). Only shellfish data but not fish data were used for calculation. It is because MPs in fish are mostly found in the gastrointestinal tract, which is usually discarded. The global shellfish consumption rates reported by the Food and Agriculture Organization (FAO) are 1.79 kg·capita^{-1}·year^{-1}

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for Crustacean and 2.5 kg·capita^{-1}·year^{-1} for Molluscs.[89] It should be noted that the consumption rates were calculated based on whole tissue including shell and soft tissue, while MP abundance of shellfish is often expressed as item·g^{-1} wet soft tissue. Considering the general ratio of the soft tissue mass to the total mass of shellfish reported in previous surveys (0.4),[90−94] MP intake via shellfish consumption ranges from 0 to 1.3 × 10^4 particles per person (Figure 3).

The respiratory tract exposure of MPs was also considered. Only abundance data of atmospheric MPs with unit of item·m^{-3} were used for calculation. Extremely high levels (1.3 × 10^4−1.7 × 10^4 items·m^{-3}) of MPs in outdoor air samples from a heavily trafficked roads[68] and low levels (0−1.37 items·m^{-3}) of atmospheric MPs from oceans[67,69] were excluded because neither site is the main place for human activities. The inhalation rate is 14.3 m^3·day^{-1}.[95] The abundance of inhalable MPs was reported to range from 0 to 5.7 × 10^3 items·m^{-3}.[66,78] As MP abundance in indoor environments is generally higher than that in outdoor settings, human MP intakes through indoor and outdoor air inhalation were calculated separately. An adult is expected to annually inhale 1.9 × 10^3−1.0 × 10^5 and 0−3.0 × 10^7 MPs through indoor and outdoor air, respectively (Figure 3).

Overall, the human MP intakes were calculated based on the abundance of MPs with similar size range, making data comparison more reasonable. Among the different exposure pathways, inhalation of indoor and outdoor air contributes the most to human exposure to MPs (Figure 3), suggesting a long-term monitoring of airborne MPs in the future. In our estimation, the amount of MP inhalation presents the MPs entering human body through nose. It is still unknown for the exact quantity of MPs entering the trachea, bronchus, and lung. Besides, up to date, the abundance of smaller MPs (e.g., <5 μm) and nanoplastics has not been documented due to the limitations of analytical methods. As we know, however, small particles are more likely to enter the lower respiratory system. Therefore, more efforts are highly needed to overcome these difficulties. In the current stage, it is still hard to compare human intake and health risks of MP inhalation with other inhalable pollutants such as PM 2.5. First of all, PM 2.5 is often expressed as μg·m^{-3}, whereas suspended MPs are often expressed as item·m^{-3}. Besides, due to the special characteristics of plastic material and the additives it contains, the toxicological mechanism of MPs may also differ from that of PM 2.5 or other pollutants.

Table salt, drinking water, and air not only represent direct MP exposure routes of humans, but also cause indirect MPs exposure during human food consumption. Salt is used as a preservation agent in many processed food items. Water is also commonly used throughout the entire food consumption process. Air contact is almost inevitable from food acquisition to human ingestion. The indirect consumption routes complicate the assessment of human exposure to MPs through food intake. The quantity of salt or water added into various foods and ingestion rates of processed food vary largely among people, making it difficult to incoporate indirect exposure pathways into estimation of MP intake. In addition, the discovery of more MP exposure routes suggests MPs are entering human body in imperceptible ways, such as the MPs released from tea bags.[38]

In a recent review, Cox et al.[96] estimated the human intake of MPs, with a focus on the recommended intakes for Americans (e.g., salt, honey, sugar, seafood, bottled water, tap water, and alcohol) and air inhalation. The authors indicated that the total intake of MPs ranged from 7.4 × 10^4 to 1.2 × 10^5 items per year, which is within the range reported in the present review. Different to their study, we emphasized on up-to-date global data and reviewed different analytical methods, and more importantly, we provided original and novel insights on the MP intake. Cox et al.[96] suggested that an effective way to reduce MP intake is to abandon bottled water. However, it might be inappropriate to draw that conclusion based on the current knowledge because of the large differences in pore size of the filters used as well as instrumental limitations, which has been often ignored. High concentrations of small-sized MPs (<10 μm) have only been reported in bottled water but not in any other media (tap water, table salt, and air), which may be because μ-Raman used for bottled water has higher particle size sensitivity. Thus, comparison of MP abundance should be conducted within a similar particle size range. Additionally, actual MP intake varies among individuals and is greatly influenced by regional pollution levels. In the future, unified protocols and large-scale surveys will allow for more comparable and accurate estimation of MP intake.

Translocation and Accumulation in Human Body. Upon ingestion or inhalation, MPs are capable of translocating and accumulating in different organs and tissues. Microplastics have been found to be internalized in the gastrointestinal tract, and the unabsorbed portion is excreted with human feces.[29] Some MPs may enter the respiratory tract. The depth of settlement depends on their aerodynamic equivalent diameter, which is used to measure the settling velocities of particles with different densities and shapes.[97] Particles with smaller aerodynamic equivalent diameters are likely to reach the lower airway. Plastic fibers have been detected in lung tissue, confirming that fibers can penetrate into the deep lung.[98] An in vitro study showed that polypropylene and polyethylene fibers exhibited no dissolution and changes after 180 days in synthetic lung fluid, suggesting high potential persistence of MPs in the respiratory tract.[99] Other nanosized plastic particles were shown to penetrate across the blood-brain barrier and placenta, and even cell membranes.[100] However, no direct evidence shows the distribution and accumulation of MPs in human organs. The only mouse-model-based experiment has shown that MPs can accumulate in liver, kidney, and gut.[101]

Human Health Risk. Current knowledge on whether MPs would reach human organs and cause adverse health impacts remains poor. The available animal testing results may have some implications for human health effects of MPs. Ingestion of MPs can cause inflammatory responses in the digestive system in *Mytilus*.[102] The immune system of fish is also the target of MP attack.[103] Exposure of the innate immune system of fathead minnow to nanoplastics significantly increased degranulation of primary granules and neutrophil extracellular trap release.[103] Inflammations including chemokine expression and pulmonary hypertension were induced by intrajugular injection of polystyrene microspheres in rats, probably due to increased blood coagulability or vascular occlusions.[104] *In vivo* experiments showed that polystyrene could be internalized in macrophages, erythrocytes, and rat alveolar epithelial cells, damaging intracellular structures.[105] Moreover, persistent organic pollutants, metals, and pathogenic microorganisms can sorb on MPs, and the leaching of chemical additives can also aggravate the toxic effects of MPs.[106,107] Potential harmful effects of MPs on human health remain debatable. Some researchers emphasized the dangers posed by food chain

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transfer, while others claimed no adverse effect caused by MPs or MP additives.[107] The controversies mostly lie in the uncertainty of MP intake estimates, and therefore more efforts on MP intake measurements and modeling are desirable.

Even though MP toxicology is in its infancy, occupational diseases have been associated with inhalation of MP particles.[97] Flock workers exposed to polypropylene may have an increased risk of 3.6 (odds ratio of 3.6) for respiratory symptoms compared to nonexposed individuals.[97] Gene mutation may also result from chronic inhalation exposure to low concentrations of fine particles.[108] A higher cancer incidence rate was observed in synthetic textile workers after 10−20 years of exposure to polypropylene fibers.[97] Polyvinyl chloride workers suffered increased lung cancer risk, with age, working years, and exposure duration at the factories.[97] More investigations are needed to quantify the atmospheric and tissue concentrations of MPs and understand the mechanical toxicity of MPs.

PERSPECTIVES

Microplastics were first discovered in oceanic water and sediment that are considered as the sinks of plastic debris. Only until recent years have researchers began to recognize the association of MPs with human health through food consumption. This may explain the relatively small number of literature reviewed here when comparing with the marine MP counterpart. More surveys and studies, therefore, are required to assess the occurrence of MPs in human exposure pathways and related health impacts. It is reasonable to consider table salt, drinking water, and air as the three major human exposure pathways of MPs. Among the different pathways leading to human body burdens of MPs, the intake of atmospheric MPs through inhalation is estimated to be the most significant (1.9×10^3 to 1.0×10^5 items·year^{-1} indoor air; 0−3.0×10^7 items·year^{-1} outdoor air).

So far, many important questions on MPs remain unanswered. The exact routes of MP cellular intake, the tissue accumulation of MPs, and the potential adverse effects after long-term MP exposure in human are unknown. The fate and transport of MPs upon entering an organism through absorption and excretion is unclear. The changes at cellular level or even molecular level and specific mechanisms have not been studied. The potential health risks to human body are only speculated by referring to animal testing results, and this knowledge gap needs to be filled. To move forward, high vertebrate human homologue *in vivo* models such as mice complemented with *in vitro* human cell bioassays can be employed to reveal the toxicity mechanisms at molecular, cellular, and individual levels. Researchers can also learn from epidemiology and occupational studies for other environmental particle pollutants. Also, similar to other hazards, generalization in human health risk assessment of MPs should be highly cautious when the research objectives are occupational or vulnerable populations (e.g., the elderly and children).

ASSOCIATED CONTENT

Supporting Information

This information is available free of charge via the Internet at . The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.9b04535.

Data on microplastics in shellfish species (Table S1) and geographical distribution of microplastic pollution in table salt (Figure S1) (PDF)

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Notes

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REFERENCES

(1) Lebreton, L.; Andrady, A. Future scenarios of global plastic waste generation and disposal. *Palgrave Commun.* **2019**, *5*, 6.

(2) Rochman, C. M.; Browne, M. A.; Halpern, B. S.; Hentschel, B. T.; Hoh, E.; Karapanagioti, H. K.; Rios-Mendoza, L. M.; Takada, H.; Teh, S.; Thompson, R. C. Classify plastic waste as hazardous. *Nature* **2013**, *494*, 169−171.

(3) Wilcox, C.; Hardesty, B. D.; Law, K. L. Abundance of floating plastic particles is increasing in the western North Atlantic Ocean. *Environ. Sci. Technol.* **2020**, *54*, 790−796.

(4) Chen, Q. Q.; Reisser, J.; Cunsolo, S.; Kwadijk, C.; Kotterman, M.; Proietti, M.; Slat, B.; Ferrari, F. F.; Schwarz, A.; Levivier, A.; Yin, D. Q.; Hollert, H.; Koelmans, A. A. Pollutants in plastics within the north Pacific subtropical gyre. *Environ. Sci. Technol.* **2018**, *52*, 446−456.

(5) Thompson, R. C.; Olsen, Y.; Mitchell, R. P.; Davis, A.; Rowland, S. J.; John, A. W. G.; McGonigle, D.; Russell, A. E. Lost at sea: Where is all the plastic? *Science* **2004**, *304*, 838−838.

(6) Alimi, O. S.; Farner Budarz, J.; Hernandez, L. M.; Tufenkji, N. Microplastics and nanoplastics in aquatic environments: aggregation, deposition, and enhanced contaminant transport. *Environ. Sci. Technol.* **2018**, *52*, 1704−1724.
(7) Wright, S. L.; Thompson, R. C.; Galloway, T. S. The physical impacts of microplastics on marine organisms: A review. *Environ. Pollut.* **2013**, *178*, 483−492.
(8) Van Cauwenberghe, L.; Devriese, L.; Galgani, F.; Robbens, J.; Janssen, C. R. Microplastics in sediments: A review of techniques, occurrence and effects. *Mar. Environ. Res.* **2015**, *111*, 5−17.
(9) Van Cauwenberghe, L.; Claessens, M.; Vandegehuchte, M. B.; Janssen, C. R. Microplastics are taken up by mussels (Mytilus edulis) and lugworms (Arenicola marina) living in natural habitats. *Environ. Pollut.* **2015**, *199*, 10−17.
(10) Li, J. N.; Lusher, A. L.; Rotchell, J. M.; Deudero, S.; Turra, A.; Brate, I. L. N.; Sun, C. J.; Hossain, M. S.; Li, Q. P.; Kolandhasamy, P.; Shi, H. H. Using mussel as a global bioindicator of coastal microplastic pollution. *Environ. Pollut.* **2019**, *244*, 522−533.
(11) Carbery, M.; O'Connor, W.; Palanisami, T. Trophic transfer of microplastics and mixed contaminants in the marine food web and implications for human health. *Environ. Int.* **2018**, *115*, 400−409.
(12) Toussaint, B.; Raffael, B.; Angers-Loustau, A.; Gilliland, D.; Kestens, V.; Petrillo, M.; Rio-Echevarria, I. M.; Eede, G. V. D. Review of micro- and nanoplastic contamination in the food chain. *Food Addit. Contam., Part A* **2019**, *36*, 639−673.
(13) Avio, C. G.; Gorbi, S.; Milan, M.; Benedetti, M.; Fattorini, D.; d'Errico, G.; Pauletto, M.; Bargelloni, L.; Regoli, F. Pollutants bioavailability and toxicological risk from microplastics to marine mussels. *Environ. Pollut.* **2015**, *198*, 211−222.
(14) Lu, Y. F.; Zhang, Y.; Deng, Y. F.; Jiang, W.; Zhao, Y. P.; Geng, J. J.; Ding, L. L.; Ren, H. Q. Uptake and Accumulation of polystyrene microplastics in zebrafish (Danio rerio) and toxic effects in liver. *Environ. Sci. Technol.* **2016**, *50*, 4054−4060.
(15) Besseling, E.; Wang, B.; Lürling, M.; Koelmans, A. A. Nanoplastic affects growth of S. obliquus and reproduction of D. magna. *Environ. Sci. Technol.* **2014**, *48*, 12336−12343.
(16) Liebezeit, G.; Liebezeit, E. Non-pollen particulates in honey and sugar. *Food Addit. Contam., Part A* **2013**, *30*, 2136−2140.
(17) Kutralam-Muniasamy, G.; Pérez-Guevara, F.; Elizalde-Martínez, I.; Shruti, V. C. Branded milks − Are they immune from microplastics contamination? *Sci. Total Environ.* **2020**, *714*, 136823.
(18) Kosuth, M.; Mason, S. A.; Wattenberg, E. V. Anthropogenic contamination of tap water, beer, and sea salt. *PLoS One* **2018**, *13*, No. e0194970.
(19) Santillo, D.; Miller, K.; Johnston, P. Microplastics as contaminants in commercially important seafood species. *Integr. Environ. Assess. Manage.* **2017**, *13*, 516−521.
(20) Yang, D.; Shi, H.; Li, L.; Li, J.; Jabeen, K.; Kolandhasamy, P. Microplastic pollution in table salts from China. *Environ. Sci. Technol.* **2015**, *49*, 13622−7.
(21) Peixoto, D.; Pinheiro, C.; Amorim, J.; Oliva-Teles, L.; Guilhermino, L.; Vieira, M. N. Microplastic pollution in commercial salt for human consumption: A review. *Estuarine, Coastal Shelf Sci.* **2019**, *219*, 161−168.
(22) Mason, S. A.; Welch, V. G.; Neratko, J. Synthetic polymer contamination in bottled water. *Front. Chem.* **2018**, *6*, 407.
(23) Dris, R.; Gasperi, J.; Rocher, V.; Saad, M.; Renault, N.; Tassin, B. Microplastic contamination in an urban area: a case study in Greater Paris. *Environ. Chem.* **2015**, *12*, 592−599.
(24) Barboza, L. G. A.; Vethaak, A. D.; Lavorante, B. R. B. O.; Lundebye, A. K.; Guilhermino, L. Marine microplastic debris: An emerging issue for food security, food safety and human health. *Mar. Pollut. Bull.* **2018**, *133*, 336−348.
(25) Renzi, M.; Blaskovic, A. Litter & microplastics features in table salts from marine origin: Italian versus Croatian brands. *Mar. Pollut. Bull.* **2018**, *135*, 62−68.
(26) Erkoyun, E.; Sozmen, K.; Bennett, K.; Unal, B.; Boshuizen, H. C. Predicting the health impact of lowering salt consumption in Turkey using the DYNAMO health impact assessment tool. *Public Health* **2016**, *140*, 228−234.
(27) Mozaffarian, D.; Fahimi, S.; Singh, G. M.; Micha, R.; Khatibzadeh, S.; Engell, R. E.; Lim, S.; Danaei, G.; Ezzati, M.; Powles, J.; Dis, G. B. D. N. C. Global sodium consumption and death from cardiovascular causes. *N. Engl. J. Med.* **2014**, *371*, 624−634.
(28) Gasperi, J.; Wright, S. L.; Dris, R.; Collard, F.; Mandin, C.; Guerrouache, M.; Langlois, V.; Kelly, F. J.; Tassin, B. Microplastics in air: Are we breathing it in? *Curr. Opin. Environ. Sci. Health* **2018**, *1*, 1−5.
(29) Schwabl, P.; Koppel, S.; Konigshofer, P.; Bucsics, T.; Trauner, M.; Reiberger, T.; Liebmann, B. Detection of various microplastics in human stool: A prospective case series. *Ann. Intern. Med.* **2019**, *171*, 45.
(30) Seth, C. K.; Shriwastav, A. Contamination of Indian sea salts with microplastics and a potential prevention strategy. *Environ. Sci. Pollut. Res.* **2018**, *25*, 30122−30131.
(31) Gundogdu, S. Contamination of table salts from Turkey with microplastics. *Food Addit. Contam., Part A* **2018**, *35*, 1006−1014.
(32) Iniguez, M. E.; Conesa, J. A.; Fullana, A. Microplastics in Spanish table salt. *Sci. Rep.* **2017**, *7*, 8620.
(33) Renzi, M.; Grazioli, E.; Bertacchini, E.; Blaskovic, A. Microparticles in Table Salt: Levels and chemical composition of the smallest dimensional fraction. *J. Mar. Sci. Eng.* **2019**, *7*, 310.
(34) Lee, H.; Kunz, A.; Shim, W. J.; Walther, B. A. Microplastic contamination of table salts from Taiwan, including a global review. *Sci. Rep.* **2019**, *9*, 10145.
(35) Tahir, A.; Taba, P.; Samawi, M. F.; Werorilangi, S. Microplastics in water, sediment and salts from traditional salt producing ponds. *Global J. Environ. Sci. Manage* **2019**, *5*, 431−440.
(36) Kim, J. S.; Lee, H. J.; Kim, S. K.; Kim, H. J. Global pattern of microplastics (MPs) in commercial food- grade salts: sea salt as an indicator of seawater MP pollution. *Environ. Sci. Technol.* **2018**, *52*, 12819−12828.
(37) Karami, A.; Golieskardi, A.; Choo, C. K.; Larat, V.; Galloway, T. S.; Salamatinia, B. The presence of microplastics in commercial salts from different countries. *Sci. Rep.* **2017**, *7*, 46173.
(38) Hernandez, L. M.; Xu, E. G.; Larsson, H. C. E.; Tahara, R.; Maisuria, V. B.; Tufenkji, N. Plastic teabags release billions of microparticles and nanoparticles into tea. *Environ. Sci. Technol.* **2019**, *53*, 12300−12310.
(39) Selvam, S.; Manisha, A.; Venkatramanan, S.; Chung, S. Y.; Paramasivam, C. R.; Singaraja, C. Microplastic presence in commercial marine sea salts: A baseline study along Tuticorin Coastal salt pan stations, Gulf of Mannar, South India. *Mar. Pollut. Bull.* **2020**, *150*, 110675.
(40) Miller, M. E.; Kroon, F. J.; Motti, C. A. Recovering microplastics from marine samples: A review of current practices. *Mar. Pollut. Bull.* **2017**, *123*, 6−18.
(41) Quinn, B.; Murphy, F.; Ewins, C. Validation of density separation for the rapid recovery of microplastics from sediment. *Anal. Methods* **2017**, *9*, 1491−1498.
(42) Chen, Y.; Leng, Y.; Liu, X.; Wang, J. Microplastic pollution in vegetable farmlands of suburb Wuhan, central China. *Environ. Pollut.* **2020**, *257*, 113449.
(43) Dehghani, S.; Moore, F.; Akhbarizadeh, R. Microplastic pollution in deposited urban dust, Tehran metropolis, Iran. *Environ. Sci. Pollut. Res.* **2017**, *24*, 20360−20371.
(44) Kappler, A.; Fischer, D.; Oberbeckmann, S.; Schernewski, G.; Labrenz, M.; Eichhorn, K. J.; Voit, B. Analysis of environmental microplastics by vibrational microspectroscopy: FTIR, Raman or both? *Anal. Bioanal. Chem.* **2016**, *408*, 8377−8391.
(45) Mintenig, S. M.; Loder, M. G. J.; Primpke, S.; Gerdts, G. Low numbers of microplastics detected in drinking water from ground water sources. *Sci. Total Environ.* **2019**, *648*, 631−635.
(46) Pivokonsky, M.; Cermakova, L.; Novotna, K.; Peer, P.; Cajthaml, T.; Janda, V. Occurrence of microplastics in raw and treated drinking water. *Sci. Total Environ.* **2018**, *643*, 1644−1651.

https://dx.doi.org/10.1021/acs.est.9b04535
Environ. Sci. Technol. 2020, 54, 3740−3751

(47) Uhl, W.; Eftekhardadkhah, M.; Svendsen, C. Mapping microplastic in Norwegian drinking water. *Norsk Vann* **2018**, 241.
(48) Wang, Z.; Lin, T.; Chen, W. Occurrence and removal of microplastics in an advanced drinking water treatment plant (ADWTP). *Sci. Total Environ.* **2020**, *700*, 134520.
(49) Paredes, M.; Castillo, T.; Viteri, R.; Fuentes, G.; Bodero, E. Microplastics in the drinking water of the Riobamba city, Ecuador. *Sci. Rev. Eng. Env. Sci.* **2019**, *28*, 653−663.
(50) Strand, J.; Feld, L.; Murphy, F.; Mackevica, A.; Hartmann, N. B., Analysis of microplastic particles in Danish drinking water. DCE-Danish Centre for Environment and Energy, 2018, p 34.
(51) Ossmann, B. E.; Sarau, G.; Holtmannspotter, H.; Pischetsrieder, M.; Christiansen, S. H.; Dicke, W. Small- sized microplastics and pigmented particles in bottled mineral water. *Water Res.* **2018**, *141*, 307−316.
(52) Schymanski, D.; Goldbeck, C.; Humpf, H. U.; Furst, P. Analysis of microplastics in water by micro-Raman spectroscopy: Release of plastic particles from different packaging into mineral water. *Water Res.* **2018**, *129*, 154−162.
(53) Zuccarello, P.; Ferrante, M.; Cristaldi, A.; Copat, C.; Grasso, A.; Sangregorio, D.; Fiore, M.; Conti, G. O. Exposure to microplastics (< 10 μm) associated to plastic bottles mineral water consumption: The first quantitative study. *Water Res.* **2019**, *157*, 365−371.
(54) Teunis, P. F. M.; Medema, G. J.; Kruidenier, L.; Havelaar, A. H. Assessment of the risk of infection by *Cryptosporidium* or *Giardia* in drinking water from a surface water source. *Water Res.* **1997**, *31*, 1333−1346.
(55) Santhi, V. A.; Sakai, N.; Ahmad, E. D.; Mustafa, A. M. Occurrence of bisphenol A in surface water, drinking water and plasma from Malaysia with exposure assessment from consumption of drinking water. *Sci. Total Environ.* **2012**, *427*, 332−338.
(56) Eerkes-Medrano, D.; Thompson, R. C.; Aldridge, D. C. Microplastics in freshwater systems: A review of the emerging threats, identification of knowledge gaps and prioritisation of research needs. *Water Res.* **2015**, *75*, 63−82.
(57) Koelmans, A. A.; Mohamed Nor, N. H.; Hermsen, E.; Kooi, M.; Mintenig, S. M.; De France, J. Microplastics in freshwaters and drinking water: Critical review and assessment of data quality. *Water Res.* **2019**, *155*, 410−422.
(58) Cai, L. Q.; Wang, J. D.; Peng, J. P.; Tan, Z.; Zhan, Z. W.; Tan, X. L.; Chen, Q. Q. Characteristic of microplastics in the atmospheric fallout from Dongguan city, China: preliminary research and first evidence. *Environ. Sci. Pollut. Res.* **2017**, *24*, 24928−24935.
(59) Dris, R.; Gasperi, J.; Saad, M.; Mirande, C.; Tassin, B. Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? *Mar. Pollut. Bull.* **2016**, *104*, 290−293.
(60) Klein, M.; Fischer, E. K. Microplastic abundance in atmospheric deposition within the Metropolitan area of Hamburg. *Sci. Total Environ.* **2019**, *685*, 96−103.
(61) Zhou, Q.; Tian, C. G.; Luo, Y. M. Various forms and deposition fluxes of microplastics identified in the coastal urban atmosphere. *Kexue Tongbao* **2017**, *62*, 3902−3910.
(62) Wright, S. L.; Ulke, J.; Font, A.; Chan, K. L. A.; Kelly, F. J. Atmospheric microplastic deposition in an urban environment and an evaluation of transport. *Environ. Int.* **2020**, *136*, 105411.
(63) Dris, R.; Gasperi, J.; Mirande, C.; Mandin, C.; Guerrouache, M.; Langlois, V.; Tassin, B. A first overview of textile fibers, including microplastics, in indoor and outdoor environments. *Environ. Pollut.* **2017**, *221*, 453−458.
(64) Abbasi, S.; Keshavarzi, B.; Moore, F.; Turner, A.; Kelly, F. J.; Dominguez, A. O.; Jaafarzadeh, N. Distribution and potential health impacts of microplastics and microrubbers in air and street dusts from Asaluyeh County, Iran. *Environ. Pollut.* **2019**, *244*, 153−164.
(65) Vianello, A.; Jensen, R. L.; Liu, L.; Vollertsen, J. Simulating human exposure to indoor airborne microplastics using a Breathing Thermal Manikin. *Sci. Rep.* **2019**, *9*, 8670.
(66) Liu, K.; Wang, X. H.; Wei, N.; Song, Z. Y.; Li, D. J. Accurate quantification and transport estimation of suspended atmospheric microplastics in megacities: Implications for human health. *Environ. Int.* **2019**, *132*, 105127.
(67) Liu, K.; Wu, T. N.; Wang, X. H.; Song, Z. Y.; Zong, C. X.; Wei, N. A.; Li, D. J. Consistent transport of terrestrial microplastics to the ocean through atmosphere. *Environ. Sci. Technol.* **2019**, *53*, 10612−10619.
(68) Asrin, N. R. N.; Dipareza, A. Microplastics in ambient air (case study: Urip Sumoharjo street and Mayjend Sungkono street of Surabaya city, Indonesia). *IAETSD J. Adv. Res. Appl. Sci.* **2019**, 54−57.
(69) Wang, X.; Li, C.; Liu, K.; Zhu, L.; Song, Z.; Li, D. Atmospheric microplastic over the South China Sea and East Indian Ocean: abundance, distribution and source. *J. Hazard. Mater.* **2019**, 121846.
(70) Liu, C.; Li, J.; Zhang, Y.; Wang, L.; Deng, J.; Gao, Y.; Yu, L.; Zhang, J.; Sun, H. Widespread distribution of PET and PC microplastics in dust in urban China and their estimated human exposure. *Environ. Int.* **2019**, *128*, 116−124.
(71) Zhang, J.; Wang, L.; Kannan, K. Microplastics in house dust from 12 countries and associated human exposure. *Environ. Int.* **2020**, *134*, 105314.
(72) Ambrosini, R.; Azzoni, R. S.; Pittino, F.; Diolaiuti, G.; Franzetti, A.; Parolini, M. First evidence of microplastic contamination in the supraglacial debris of an alpine glacier. *Environ. Pollut.* **2019**, *253*, 297−301.
(73) Yukioka, S.; Tanaka, S.; Nabetani, Y.; Suzuki, Y.; Ushijima, T.; Fujii, S.; Takada, H.; Van Tran, Q.; Singh, S. Occurrence and characteristics of microplastics in surface road dust in Kusatsu (Japan), Da Nang (Vietnam), and Kathmandu (Nepal). *Environ. Pollut.* **2020**, *256*, 113447.
(74) Kaya, A. T.; Yurtsever, M.; Bayraktar, S. C. Ubiquitous exposure to microfiber pollution in the air. *Eur. Phys. J. Plus* **2018**, *133*, 488.
(75) Su, L.; Xue, Y. G.; Li, L. Y.; Yang, D. Q.; Kolandhasamy, P.; Li, D. J.; Shi, H. H. Microplastics in taihu lake, China. *Environ. Pollut.* **2016**, *216*, 711−719.
(76) Liu, K.; Wang, X.; Fang, T.; Xu, P.; Zhu, L.; Li, D. Source and potential risk assessment of suspended atmospheric microplastics in Shanghai. *Sci. Total Environ.* **2019**, *675*, 462−471.
(77) Bergmann, M.; Mutzel, S.; Primpke, S.; Tekman, M. B.; Trachsel, J.; Gerdts, G. White and wonderful? Microplastics prevail in snow from the Alps to the Arctic. *Sci. Adv.* **2019**, *5*, No. eaax1157.
(78) Li, Y.; Shao, L.; Wang, W.; Zhang, M.; Feng, X.; Li, W.; Zhang, D. Airborne fiber particles: Types, size and concentration observed in Beijing. *Sci. Total Environ.* **2020**, *705*, 135967.
(79) Allen, S.; Allen, D.; Phoenix, V. R.; Le Roux, G.; Jimenez, P. D.; Simonneau, A.; Binet, S.; Galop, D. Atmospheric transport and deposition of microplastics in a remote mountain catchment. *Nat. Geosci.* **2019**, *12*, 339.
(80) Liebezeit, G.; Liebezeit, E. Origin of synthetic particles in honeys. *Pol. J. Food Nutr. Sci.* **2015**, *65*, 143−147.
(81) Powles, J.; Fahimi, S.; Micha, R.; Khatibzadeh, S.; Shi, P. L.; Ezzati, M.; Engell, R. E.; Lim, S. S.; Danaei, G.; Mozaffarian, D.; Dis, G. B. D. N. C. Global, regional and national sodium intakes in 1990 and 2010: a systematic analysis of 24 h urinary sodium excretion and dietary surveys worldwide. *BMJ Open* **2013**, *3*, No. e003733.
(82) Thompson, J.; Eaglesham, G.; Mueller, J. Concentrations of PFOS, PFOA and other perfluorinated alkyl acids in Australian drinking water. *Chemosphere* **2011**, *83*, 1320−1325.
(83) Ljung, K.; Selinus, O.; Otabbong, E.; Berglund, M. Metal and arsenic distribution in soil particle sizes relevant to soil ingestion by children. *Appl. Geochem.* **2006**, *21*, 1613−1624.
(84) Abbasi, S.; Soltani, N.; Keshavarzi, B.; Moore, F.; Turner, A.; Hassanaghaei, M. Microplastics in different tissues of fish and prawn from the Musa Estuary. *Chemosphere* **2018**, *205*, 80−87.
(85) Li, J. N.; Qu, X. Y.; Su, L.; Zhang, W. W.; Yang, D. Q.; Kolandhasamy, P.; Li, D. J.; Shi, H. H. Microplastics in mussels along the coastal waters of China. *Environ. Pollut.* **2016**, *214*, 177−184.
(86) Li, J. N.; Green, C.; Reynolds, A.; Shi, H. H.; Rotchell, J. M. Microplastics in mussels sampled from coastal waters and supermarkets in the United Kingdom. *Environ. Pollut.* **2018**, *241*, 35−44.

https://dx.doi.org/10.1021/acs.est.9b04535
Environ. Sci. Technol. 2020, 54, 3740−3751

(87) Li, J. N.; Yang, D. Q.; Li, L.; Jabeen, K.; Shi, H. H. Microplastics in commercial bivalves from China. *Environ. Pollut.* **2015**, *207*, 190−195.
(88) Hantoro, I.; Lohr, A. J.; Van Belleghem, F.; Widianarko, B.; Ragas, A. M. J. Microplastics in coastal areas and seafood: implications for food safety. *Food Addit. Contam., Part A* **2019**, *36*, 644−711.
(89) Food and Agriculture Organization. 2017, FAOSTAT [Internet]. http://www.fao.org/faostat/en/#data/CL.
(90) Cartier, S.; Pellerin, J.; Fournier, M.; Tamigneaux, E.; Girault, L.; Lemaire, N. Use of an index based on the blue mussel (Mytilus edulis and Mytilus trossulus) digestive gland weight to assess the nutritional quality of mussel farm sites. *Aquaculture* **2004**, *241*, 633−654.
(91) Benali, I.; Boutiba, Z.; Grandjean, D.; de Alencastro, L. F.; Rouane-Hacene, O.; Chevre, N. Spatial distribution and biological effects of trace metals (Cu, Zn, Pb, Cd) and organic micropollutants (PCBs, PAHs) in mussels Mytilus galloprovincialis along the Algerian west coast. *Mar. Pollut. Bull.* **2017**, *115*, 539−550.
(92) Benali, I.; Boutiba, Z.; Merabet, A.; Chevre, N. Integrated use of biomarkers and condition indices in mussels (Mytilus galloprovincialis) for monitoring pollution and development of biomarker index to assess the potential toxic of coastal sites. *Mar. Pollut. Bull.* **2015**, *95*, 385−94.
(93) Peharda, M.; Župan, I.; Bavčević, L.; Frankić, A.; Klanjšček, T. Growth and condition index of mussel Mytilus galloprovincialis in experimental integrated aquaculture. *Aquacult. Res.* **2007**, *38*, 1714−1720.
(94) Çelik, M. Y.; Karayücel, S.; Karayücel, İ.; Öztürk, R.; Eyüboğlu, B. Meat yield, condition index, and biochemical composition of mussels (Mytilus galloprovincialis Lamarck, 1819) in Sinop, South of the Black Sea. *J. Aquat. Food Prod. Technol.* **2012**, *21*, 198−205.
(95) Kim, S. K.; Shoeib, M.; Kim, K. S.; Park, J. E. Indoor and outdoor poly- and perfluoroalkyl substances (PFASs) in Korea determined by passive air sampler. *Environ. Pollut.* **2012**, *162*, 144−150.
(96) Cox, K. D.; Covernton, G. A.; Davies, H. L.; Dower, J. F.; Juanes, F.; Dudas, S. E. Human consumption of microplastics. *Environ. Sci. Technol.* **2019**, *53*, 7068−7074.
(97) Prata, J. C. Airborne microplastics: Consequences to human health? *Environ. Pollut.* **2018**, *234*, 115−126.
(98) Pauly, J. L.; Stegmeier, S. J.; Allaart, H. A.; Cheney, R. T.; Zhang, P. J.; Mayer, A. G.; Streck, R. J. Inhaled cellulosic and plastic fibers found in human lung tissue. *Cancer Epidemiol., Biomarkers Prev.* **1998**, *7*, 419−428.
(99) Law, B. D.; Bunn, W. B.; Hesterberg, T. W. Solubility of polymeric organic fibers and manmade vitreous fibers in Gambles solution. *Inhalation Toxicol.* **1990**, *2*, 321−339.
(100) Schirinzi, G. F.; Perez-Pomeda, I.; Sanchis, J.; Rossini, C.; Farre, M.; Barcelo, D. Cytotoxic effects of commonly used nanomaterials and microplastics on cerebral and epithelial human cells. *Environ. Res.* **2017**, *159*, 579−587.
(101) Deng, Y.; Zhang, Y.; Lemos, B.; Ren, H. Tissue accumulation of microplastics in mice and biomarker responses suggest widespread health risks of exposure. *Sci. Rep.* **2017**, *7*, 46687.
(102) von Moos, N.; Burkhardt-Holm, P.; Kohler, A. Uptake and effects of microplastics on cells and tissue of the blue mussel mytilus edulis L. after an experimental exposure. *Environ. Sci. Technol.* **2012**, *46*, 11327−11335.
(103) Greven, A. C.; Merk, T.; Karagoz, F.; Mohr, K.; Klapper, M.; Jovanovic, B.; Palic, D. Polycarbonate and polystyrene nanoplastic particles act as stressors to the innate immune system of fathead minnow (Pimephales promelas). *Environ. Toxicol. Chem.* **2016**, *35*, 3093−3100.
(104) Jones, A. E.; Watts, J. A.; Debelak, J. P.; Thornton, L. R.; Younger, J. G.; Kline, J. A. Inhibition of prostaglandin synthesis during polystyrene microsphere-induced pulmonary embolism in the rat. *Am. J. Lung Cell Mol. Physiol.* **2003**, *284*, 1072−81.
(105) Geiser, M.; Rothen-Rutishauser, B.; Kapp, N.; Schurch, S.; Kreyling, W.; Schulz, H.; Semmler, M.; Hof, V. I.; Heyder, J.; Gehr, P. Ultrafine particles cross cellular membranes by nonphagocytic mechanisms in lungs and in cultured cells. *Environ. Health Perspect.* **2005**, *113*, 1555−1560.
(106) Oberbeckmann, S.; Loder, M. G. J.; Labrenz, M. Marine microplastic- associated biofilms - a review. *Environ. Chem.* **2015**, *12*, 551−562.
(107) Bouwmeester, H.; Hollman, P. C. H.; Peters, R. J. B. Potential health impact of environmentally released micro- and nanoplastics in the human food production chain: experiences from nanotoxicology. *Environ. Sci. Technol.* **2015**, *49*, 8932−8947.
(108) Kingsley, S. L.; Deyssenroth, M. A.; Kelsey, K. T.; Abu Awad, Y.; Kloog, I.; Schwartz, J. D.; Lambertini, L.; Chen, J.; Marsit, C. J.; Wellenius, G. A. Maternal residential air pollution and placental imprinted gene expression. *Environ. Int.* **2017**, *108*, 204−211.

https://dx.doi.org/10.1021/acs.est.9b04535
Environ. Sci. Technol. 2020, 54, 3740−3751