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Review

Airborne Microplastics: Challenges, Prospects, and Experimental Approaches

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Abstract: Airborne microplastics are emerging pollutants originating from disposable tableware, packaging materials, textiles, and other consumer goods. Microplastics vary in shape and size and exposed to external factors break down into even smaller fractions. Airborne microplastics are abundant in both urban and natural environments, including water bodies and glaciers, as particles can travel long distances. The potential toxicity of airborne microplastics cannot be underestimated. Microparticles, especially those < 10 μm, entering the human body through inhalation or ingestion have been shown to cause serious adverse health effects, such as chronic inflammation, oxidation stress, physical damage to tissues, etc. Microplastics adsorb toxic chemicals and biopolymers, forming a polymer corona on their surface, affecting their overall toxicity. In addition, microplastics can also affect carbon dynamics in ecosystems and have a serious impact on biochemical cycles. The approaches to improve sampling techniques and develop standardized methods to assess airborne microplastics are still far from being perfect. The mechanisms of microplastic intracellular and tissue transport are still not clear, and the impact of airborne microplastics on human health is not understood well. Reduced consumption followed by collection, reuse, and recycling of microplastics can contribute to solving the microplastic problem. Combinations of different filtration techniques and membrane bioreactors can be used to optimize the removal of microplastic contaminants from wastewater. In this review we critically summarize the existing body of literature on airborne microplastics, including their distribution, identification, and safety assessment.

Keywords: atmospheric microplastics; dry and wet deposition; particle atmospheric transport; detection and identification methods; health impacts; environmental impacts

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1. Introduction

Plastics pervade modern life; various synthetic polymers have become an integral part of the environment, spaces, appliances, and interiors [1–3]. Plastic products offer a range of benefits to consumers: they are versatile, flexible, and durable [4], cheap and easy to manufacture, lightweight and water resistant, which allows for their wide use in industry throughout the world [5]. Synthetic polymers are produced commercially on an extremely large scale; in 2021, world plastics production was estimated at 390.7 million tons, of which 90% was fossil based [6]. China is the world's largest plastic producer, accounting for nearly one third of global plastics production. In 2021, the largest world plastics markets included packaging and construction materials, textiles, transportation, electrical engineering, and industrial equipment [6]; the demand for plastics is predicted to continuously grow. This will apparently increase the abundance of plastic waste; out of 1.9 Gt of municipal solid waste generated per year [7], plastic waste accounts for 9–11% of the total volume [8]. The excessive use of plastic products, inefficient waste management, and limited reuse of

plastic waste lead to plastics accumulation in the environment, where they can persist from decades to millennia, depending on conditions [9–12]. Plastics can degrade under certain conditions, which is facilitated by several factors, such as temperature, ultraviolet irradiation, mechanical action of wind and waves, etc. [13], yet they do not degrade completely. Plastic products fragment into microplastics (MPs), which do not exceed 5 mm in size along the maximum axis, and nanoplastics (<1 μm) [14,15]. Polymer microparticles are ubiquitous; they can be found in surface sea and fresh waters [16–18], soil [19], atmosphere [20], groundwater [21], hydrobionts [22,23], animals [24], and humans [25].

Depending on their origin, MPs can be classified into two categories: primary or secondary MPs. Primary MPs are intentionally produced as micro-sized particles for specific purposes [5]. They are used to produce personal care products (toothpastes, shower gels), cosmetics (sunscreens, nail polishes, hair dyes, scrubs, peels, shaving creams), and abrasives for air blasting (including blast cleaning based on acrylic, melamine, or polyester plastic scrubbers for cars, engines, and boat hulls to remove rust and paint). Small-sized granules are raw materials to manufacture plastic products [13,26–32]. When applied, a face cream containing microspheres can release tens of thousands of particles into the atmosphere as the product dries when exposed to wind [33]. Primary MPs can enter the environment through spills or any other incidental release during production or transportation of the material [34]. Secondary MPs originate from degradation, aging, and fragmentation of large particles, typically caused by physical abrasion and/or exposure to ultraviolet radiation and temperature variation [5,26,35]. Secondary MPs make up the majority of MPs circulating in the environment [36].

MPs may adversely affect living organisms [37], including their reproduction, nutrition, behavior, and habitat degradation [38,39], and pose a threat to biodiversity. MPs can travel from abiotic to biotic environments and vice versa [40]. MP accumulation in the organs and tissues of invertebrate animals [41] may potentially interfere with pharyngeal pumping. The impact of MPs on the human body is being studied [37]; MPs were found to reduce the viability of human skin fibroblasts at higher concentrations [42].

Development of physical and physicochemical methods for MP retention and/or deposition to prevent their transport to water bodies is underway [43,44]. Techniques for catalytical and biocatalytical MP degradation are being actively introduced [45]. Unfortunately, the complete removal of MPs from the environment using currently available technologies is not feasible [46]. Global pollution ('plastisphere') [47] has encouraged regulations that govern plastics production and disposal [48]. Since MPs are a relatively new anthropogenic pollutant, they can be used as an ideal marker of the Anthropocene epoch for future geologists [47,49].

Environmental pollution from MPs in air is a matter of growing concern. Airborne microplastics (AMPs) have not been sufficiently explored yet, since the current studies mainly focus on the analysis of distribution and behavior of MPs in water bodies [17,27,50–53]. Atmospheric–marine exchange of MPs is of great relevance; the annual transport of suspended AMPs to the marine environment is reported to be 1.2 tons [54]. Moreover, almost all plastics transported in the atmosphere are micro-sized [55]. Inhalation of AMPs can adversely affect human health [20,56–59]. AMPs can affect the environment and organisms located far from the MP source [60,61]. Another concern is increased AMP concentrations in the indoor environment where people spend 70–90% of their time [62]. When studying AMPs, researchers meet several challenges:

- The distribution of AMPs and their concentrations vary in different parts of the world, depending on the climate, geographical location of the sampling point, sampling and analytical methods used, etc. Of particular relevance are the factors affecting transport and deposition of AMPs. Intuitively, AMP deposition is supposed to be dependent on precipitation [63], wind speed, and wind direction [40]. AMP deposition can cause further transport of plastics to different environments, including groundwater [21]. The global distribution of AMPs and their seasonal and temporal variations have not yet been identified [64]; therefore, long-term monitoring across multiple seasons is

required [65]. Mobile sources of AMPs should be particularly monitored, as their monitoring is more complicated compared with stationary sources [66];

- Plastic particles of different sizes (macro-, micro-, nano-sized particles) exhibit different properties, and the patterns of their environmental effect are different. For example, nanoplastics are highly reactive and prone to heteroaggregation with natural solids and organic substances [67], and they can act as condensation nuclei in the atmosphere [68]. Most AMPs are very small (microscale/nanoscale) and are hard to detect [20]. The majority of studies on AMPs focus on particles >10 μm in size [69,70], while data on smaller plastics are meager [20];
- The study of AMPs should include the analysis of the plastic composition, since it can affect living organisms; in addition, the toxicity of AMPs may not be linearly related to their concentration [71–73]. The degradation rate of the same type of plastics may vary depending on the study area; their susceptibility to mechanical and biological effects or environmental temperature and UV exposure in different areas may differ as well [74];
- The lack of universal analytical tools leads to gaps in data on the smallest AMPs, particularly nano-sized ones [75,76]. Novel methods for analysis of AMPs are being introduced, and spectroscopy and thermal methods have replaced visual analysis in recent decades, yet there is no unified methodology for studying AMPs [77]. Researchers employ different sampling techniques, reagents, and methods to remove organics and mineral impurities and various analytical methods, which often leads to different interpretation and comparison of quantitative results; a large number of the published studies on AMPs are incommensurable [46]. Along with other challenges, limitations, quality control, and quality assurance of different methodologies should be discussed [78] to develop a unified methodology for studying AMPs.

The aim of this review was to systematize literature data on the causes of AMP formation and the sources of AMP release into the atmosphere, signs of global atmospheric transport of AMPs and factors affecting particle transport, and methodological challenges faced in the study of AMPs. We also focused on the current state of research into this issue, identified the most/least studied areas, assessed the occurrence of AMPs in urban and remote areas, and estimated potential risks for organisms and ecosystems in general.

2. AMPs: The Current State of Research

2.1. Morphology and Polymer Composition of AMPs

MPs exhibit different morphology and can be largely divided into several categories by shape [36,79]:

- Microspheres—spherical plastic particles, e.g., granules;
- Fibers—thread-like particles with a length-to-width ratio of not less than 3:1 and equal thickness along the entire particle length;
- Films—thin and flat particles with a large surface area;
- Fragments—irregularly shaped particles with a length-to-width ratio equal to a unit as the upper limit and a height-to-width ratio equal to the width-to-length ratio as the lower limit;
- Foams—microparticles sometimes classified as fragments; in some studies, they are grouped in a separate category.

Figure 1 shows examples of differently shaped MP particles.

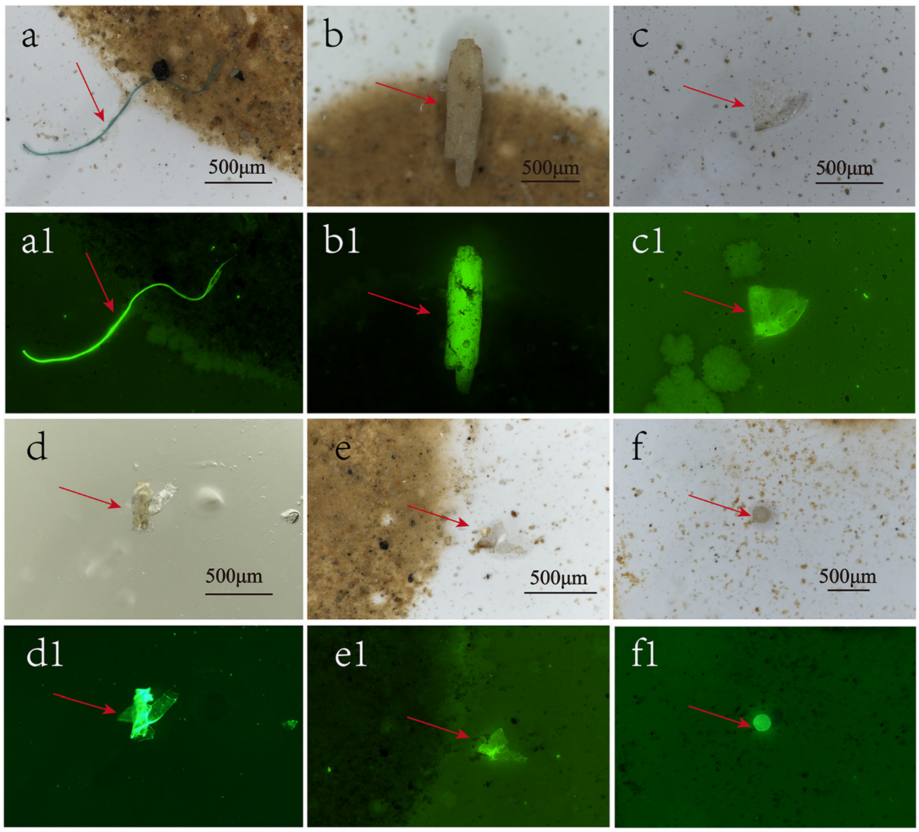

Figure 1. Bright field microscopy images of different types of MPs: (**a**) fiber, (**b**) foam, (**c**,**d**) film, (**e**) fragment, (**f**) microsphere. (**a1**,**b1**,**c1**,**d1**,**e1**,**f1**) are the corresponding fluorescence micrographs [80].

Particle morphology is helpful to identify a potential source of AMPs as they degrade into the forms from which the plastic product was originally made, for example, synthetic textiles degrade into fibers [81] and polystyrene packaging into foams [82], etc. Scanning electron microscopy (SEM) can reveal the high material porosity typical of PP, PA, and PTFE [83]. Sources of microfibers are synthetic products of the textile industry, such as clothing, home textiles, awnings, sails, ropes [34], carpets [84], and medical masks [44]. Synthetic fibers are widespread due to their elasticity, abrasion resistance, and tactile properties, which are critical for textile applications [85]. Jarosz et al. [86] detected aluminum salts on plastic fibers; its source may be antiperspirants/deodorants, which supports the hypothesis of the release of MPs from clothing. However, there are fibers of non-textile origin, such as glass fibers, rubber fibers [87], and cigarettes (a cigarette filter is a bundle of >15 thousand acetate fibers) [44]. Fibers are the predominant morphotype in urban environments [26], both indoors [88] and outdoors [34], and in remote areas [89–91]. Films are formed mainly as a result of fragmentation of plastic bags [92,93] and packaging materials [34]. Foamed MPs are formed from polystyrene products used mainly for packaging materials [93] and containers [94] and that are widely used as thermal insulation and packaging in the fast food industry [95].

As mentioned above, MPs can fragment and enter the atmosphere as a result of weathering, exposure to UV radiation, which promotes oxidation of the polymer matrix and causes destruction of covalent bonds [27,29], biological degradation, physical wear, chemical oxidation [96], and increased temperature [97]. UV irradiation facilitates chemical modifications of polymers, for example, ultraviolet radiation with wavelengths of 253.7, 300, and 350 nm caused bond breakage and the formation of hydroxyl groups in polyallyldiglycol carbonate [98].

Degradation processes affect physical and chemical properties of plastics, such as color, surface morphology, crystallinity, particle size, density, reactivity, surface functionality, and hydrophobicity [52]. In the atmosphere, these processes are faster compared with

the marine environment due to the increased oxygen concentration and UV radiation intensity [99]. Ozone depletion [100] promotes MP degradation, and low temperatures of glaciers, which contribute to long-term persistence of MPs, decrease the degradation rate [101]. Weathering mechanisms are assumed to depend on the particle size, since small particles are less susceptible to physical impact; most weathering damage is caused by UV radiation and atmospheric oxidation [102]. AMP 'aging' is estimated by microscopic methods that identify dents, cracks, breaks, and fiber bifurcation caused by mechanical erosion (for example, from collisions with sand grains in the air) [97,103,104]. Fading due to chemical oxidation may indicate long-term exposure of MPs to air, whereas smooth edges and bright color indicate the recent formation of MPs [97,105].

Along with the particle shape, the polymer composition helps to identify the source of AMPs. Polypropylene (PP) (Figure 2) and polytetrafluoroethylene (PTFE) are used to make fleece, tents, rainwear, and climbing ropes [106]. PP is also used to make disposable plastic cups, food packaging bags [33], and medical supplies [94] and produce fibers in the textile industry [51,107]. Polystyrene (PS) (Figure 3) is widely used for thermal insulation and food packaging [95], and polyethylene (PE) is used as packaging material [108]. Polyethylene terephthalate (PET) is durable [109] even at 120 °C [94], resistant to disintegration [110], and possesses stable physical and mechanical properties. PET is one of the most common plastics used to manufacture textiles and plastic containers such as bottles [94]. Most of the global PET production (over 60%) is for the purpose of synthetic fibers, whereas bottle production surprisingly accounts for only 30% of the global demand [111]. Polyamide (PA) is 4–5-fold stronger than wool; its wear resistance is 10-fold higher than that of cotton, which urges its use in fabrics [94], pharmaceuticals, beverage products, furniture, household appliances, transport [112,113], sails, toothbrushes, packaging, carpets, vehicle parts, and fishing equipment [114]. Polyacrylonitrile (PAN) is used to fabricate knitted clothing and outdoor textiles (awnings, sails, etc. since it offers excellent UV protection); it is widely used to strengthen the mechanical properties of cement mortar materials [115]. Polyvinyl fluoride (PVF) possesses high corrosion resistance and impact resistance and therefore is used in anticorrosion coating, container surface coating, wall finishing material, etc. [94]. Low-density polyethylene (LDPE) exhibits flexibility, lightweight properties, excellent resistance to chemicals, and low cost; it is often used to produce packaging and trash bags. Polyurethane (PUR) is a high-density material used to produce construction foams, yet it quickly loses its properties when exposed to UV radiation [86]. The most likely sources of polyvinyl chloride (PVC) are the construction industry, automobile tires, and textile fibers; hypromellose (HPMC) added to polymer enhances its flexibility and water-holding capacity and improves the quality of cement mortars. Polybutylene (PB) is used in synthetic rubber, adhesives, and sealants [116].

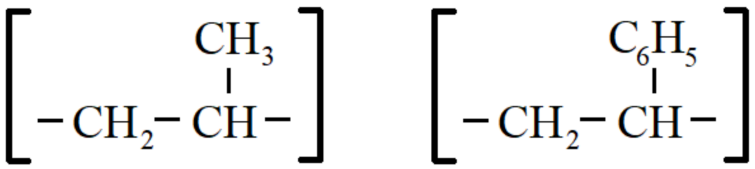

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Figure 2. Chemical structure of common MP components: polypropylene (**1**) and polystyrene (**2**).

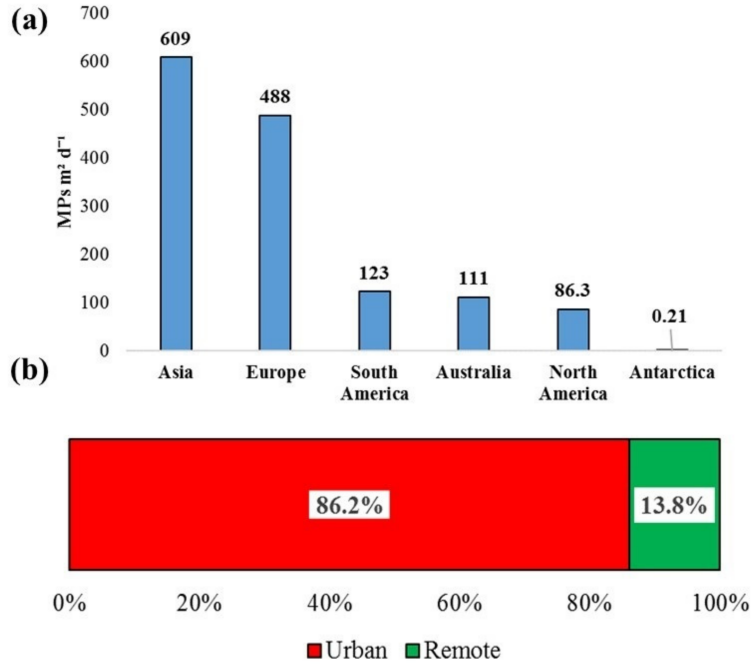

Figure 3. Mean daily AMP deposition (m^{-2} d^{-1}) by region (based on the published data, Table S1)—(**a**); proportion of DAMPs in urban areas and in remote areas, %—(**b**).

2.2. Sources and Transport of AMPs

2.2.1. Major Sources of AMPs

Industrial emissions are caused by recycling, fine grinding of plastics, and plastic waste incineration [117–120]. Industrial sanding, shredding, and cutting of synthetic textiles can also result in the release of large amounts of microfibers. Incomplete combustion byproducts (MPs) from waste incineration are released into the atmosphere [85]. The MP composition depends on the nature of an industrial enterprise; for example, the chemical composition of MPs in Shanghai was similar to that sampled from rivers in Shanghai, yet it significantly differed from the composition of MPs found in Dongguan [85]. In northern Iran, black microspheres were found near a thermal power plant that suggests incomplete destruction of combustion products through peroxidation [103]. In remote areas and waters, industrial plastics found include fragments of hoses, belts, and seals used in the oil industry and fragments of polymer varnishes applied to protect the surfaces of ships, wind turbines, and buildings [114].

Plastic household waste is a significant source of air pollution. AMPs are formed through degradation of plastic waste from open landfills or its abrasion during transportation and recycling [34] of disposable items (packaging material) [97]. Thus, high AMP concentrations could be observed in one of the districts of Mexico City with an active landfill located in this area, which may be the source of AMPs [97].

Sources of AMPs in rural areas are of agricultural origin and imply, for example, the use of plastic mulching (polymer films/sheets) and organic fertilizer derivatives from sewage sludge [121,122]. MPs in sewage sludge persist since incomplete removal during treatment processes of wastewater leads to the discharge of large amounts of MPs [122–125], and plastic mulch is used in crop cultivation [126]. For example, in arid, semi-arid, and cold regions of China, plastic films are used to increase soil temperature and reduce soil moisture evaporation [127]. Plastic mulch used in agriculture has become a significant source of AMPs [128].

Road traffic plays a particular role in AMP pollution. Road polymers include tire wear particles (TWPs), road wear particles (RWPs) [129], brake wear particles (BWPs) [130], and road dust, which may include particles of polymer products—hoses, shoes, electrical insulation, etc. [131]. Layers of asphalt concrete pavement increase the abrasive wear

resistance 3-fold compared to stone mastic asphalt [129]. Studs cause more road wear and increase particle release from the road surface. Sanding can also increase the road surface and tire wear, thus increasing the concentration of TWPs and RWPs. Road maintenance, such as snow removal, can be a source of pollution. Vehicle movement causes friction between the tire and the road surface, which induces high temperatures in the tire tread and increases tire abrasion. Higher speeds increase the concentration of airborne particles, splashes, and spillage effect, which result in fine particle deposition farther from the road. Mean and median concentrations of SBR + BR + SBS polymers in melt water and under mass traffic loads are higher at a 1 m distance compared to a 0 m distance on highways [129]. Off-road vehicles, such as trucks and tractors and construction, forestry, and military equipment are particularly significant contributors to AMP pollution; they are used in more severe conditions and undergo greater loads that promote tire and brake wear [130]. This is confirmed by studies which report that rubber, a tire wear product, is the predominant type of AMP (up to 92%) in the territory of mining enterprises and factories [132]. Generalized schemes of AMP sources are illustrated well in [133,134].

The atmosphere is more mobile than the aquatic environment; therefore, MPs can travel over thousands of kilometers [94], and the extent of such widespread distribution is shown in [135]. MPs can be found in remote and inaccessible areas such as the Arctic [114], Antarctica [91], Ecuadorian Andes [89], Pyrenees [107], Tibetan Plateau [136], protected areas of the USA such as the Grand Canyon and Rocky Mountains [35], Alps [90], and even in the atmosphere over the Pacific Ocean [104]. Researchers distinguish global (long-range) and regional transport.

Global transport involves small-sized MPs. The plastic density is lower compared with soil, which allows MPs to travel over longer distances in contrast to dust particles [137]. The transport of <1 μm nanoplastics is much more efficient than that of MPs, since lighter particles can travel over longer distances from their sources [90]. MPs of smaller size and lower density are more likely to persist in the air longer and, hence, can travel over longer distances [138]. Films are more easily transported in the air than fragments of a similar mass [107], since their surface area is larger [107]. The same goes for fibers; a larger surface-area-to-volume ratio increases drag forces and reduces deposition rates [35].

Regional transport involves MPs of larger size and higher density; MPs may be of local origin [90,94,103]. They can typically travel hundreds of kilometers but not more [107].

MP type and chemical composition can help to identify the potential source of plastic particles. For example, polydimethylsiloxane (PDMS) was found in seawater; it is virtually insoluble in water, and its density is lower than that of water, so it can accumulate on the surface of the water column as a microlayer, and it can be aerosolized [139]. PDMS was also detected in several US cities [140].

2.2.2. AMP Distribution

The sources of global transport can be both populated areas (typically, there is a correlation with the population density) and the marine environment. According to [114], a lower MP abundance was found in the Arctic than in European cities, which is related to the population density; the comparison of two remote places such as the Arctic and Antarctica [91] shows a greater abundance of MPs in the Arctic, since air masses pass through areas of higher population density. In [101], MPs deposited on the volcano Vatnajökull could originate from the nearest villages (65 km away), large cities (270 km), and the sea (45 km). AMPs are involved in the global MP cycle; they can enter soil, water, oceans, lakes, streams, and biological chains [137]. Moreover, some studies report that AMPs can be a source of oceanic MPs and vice versa. For example, large amounts of AMPs were found in the pelagic zone of the northwestern Pacific Ocean [104]. AMPs deposited on sea ice can enter the ocean when ice melts in warm waters [114]. At the same time, the ocean is reported to be one of the sources of AMPs [139,141]. AMPs can be resuspended from the ground to the atmosphere [142], followed by aerosolized transport from the sea surface to the atmosphere [143,144]. Strong winds can blow MPs off the ocean surface for secondary

particle transport [130]. A mechanism for MP transport from the ocean through splashes and rising air bubbles, which then explode and release MPs, is reported in [143]. The hypothesis of MP transfer from the ocean to the atmosphere has been confirmed by laboratory experiments [145]. The global transfer of MPs up to 100 μm from the ocean surface to the atmosphere could reach 0.01 megatons/year [146]. MP transport from the ocean to the atmosphere is most effective for MPs up to 2.5 μm in size and nanoplastics [48,139]. However, the efficiency of MP transport can be influenced not only by particle size but also by the properties of natural waters, the type of polymer, and its concentration in the water [147]. Apparently, the exchange of MPs between air and seawater is a complex dynamic cycle of MP migration [104].

Meteorological factors, such as wind, precipitation, humidity, temperature, cyclones, climate, etc., play an essential role in AMP transport and deposition [148]. There is a positive correlation between wind speed >1 m s^{-1} and the amount of AMPs [94,107]. The highest AMP deposition was observed in wet weather [26,88], but the amount of suspended AMPs decreased in wet weather. According to [54], relatively low levels of MPs in the air at night are due to plastics deposition under wet conditions. The surface area of films and granules is larger compared with fibers, which enhances their binding to moisture and solid particles. As a result, precipitation increases the deposition of films and granules [94]. Snowfall 50-fold more effectively binds plastic particles compared with rain [149], since snowflakes exhibit a larger size, lower density, and slower deposition rate compared with raindrops, thereby trapping a greater variety of particles. Due to this, snow is regarded a more effective and simpler synoptic indicator of AMP pollution than rain and may reduce any confounding effects on deposition fluxes related to concurrent or intermittent dry deposition [103]. The duration of precipitation did not affect AMP deposition, since all suspended MPs were washed out of the atmosphere by precipitation; wet ground inhibited local and regional suspension of dust and MPs [144].

Glaciers in cryosphere regions provide an ideal environment for pollutant accumulation from the atmosphere as a result of dry deposition or snowfall. Low temperature and distant location from urban areas made snow effective for the calculation of fluxes caused by atmospheric precipitation based on data on pollutants recorded in snow [65]; ice cores can be used as stratigraphic indicators of AMP deposition and provide valuable data on historical pollution [89].

MP deposition depends on the relief features; micro- and nano-sized plastic particles most often accumulate in depressions, on ridges, or on barriers under wind effects [150]; AMPs deposit when the speed of air masses crossing mountain ranges decreases [35]. Peninsulas and volcanos can diffuse wind, which caused MP accumulation in the air of one of the districts of Mexico City [97].

The amount and composition of MPs indoors and outdoors are different. In [88], the indoors-to-outdoors ratio of MPs was found to be approximately 1:3. Polyester (PET) fibers and PE and PP particles were shown to be predominant indoors, while PE and PET predominated outdoors. However, the amount of MPs indoors can depend on various factors: the type of human activity, air flow turbulence, the number of people, the quantity and quality of textiles [132], the presence of carpets, different household and building materials, the method of drying laundry and cleaning rooms [151], availability of air conditioning [132], and window airing [19]. The main indoor sources of MPs are fibers from clothing and carpets [84]. Low rates of indoor air renewal can cause high MP concentrations [70]. Indoor MP concentrations may depend on the season, with higher levels of MPs indoors in autumn and winter, apparently due to increased use of textiles such as sweaters, wool coats, and blankets [34,151]. Bare floors produce more polyethylene microfibers [152]. The concentration of MPs also depends on the day of the week; in Chinese dormitories, the concentration on weekends exceeded 3-fold that on weekdays, whereas on weekends, the concentrations of MPs in offices were 2-fold lower compared with weekdays [153]. Structural arrangement of buildings that contributes to MP accumulation or dilution is particularly relevant as it

may provide important guidance in future building design [154]. Outdoor microplastic fibers tend to be longer and larger than indoor ones [84].

Indoor environments generally demonstrate higher concentrations of MPs than outdoor environments [88], yet urban spaces are highly susceptible to AMP contamination [94]. The main sources of MPs in urban air are synthetic fabrics, eroded synthetic rubber tires, urban dust [116], construction materials, poorly managed landfills, waste incineration, industrial emissions, sewage sludge, synthetic particles used for horticultural soils, dryer exhaust, outer surfaces of car seats, and plastic recycling processes [77]. The amount and type of MPs released from synthetic fabrics may depend on the season [155,156] and even on fashion trends [157]. Drying laundry in the wind and in the sun can largely contribute to fiber air pollution [35,84,97,151]. In general, MP concentrations can depend on the population density, the level of industrial development, and the occurrence and condition of landfills. According to studies conducted in London [34], the majority of MPs were PAN fibers that is consistent with their textile origin; the highest MP concentration could be observed under wind flows from a commercial area with high foot traffic during the day. In Shanghai, AMP contamination levels were 4–5-fold higher compared with London, which positively correlates with the population density and plastic product consumption. The concentration in the central part of the metropolis 10-fold exceeded that in the suburban area, yet the concentration of MPs was higher in industrial areas of the city, despite a smaller number of people living in these areas [94]. Studies conducted in Krakow reported the greatest AMP deposition in summer, the season with the highest intensity of tourist traffic; in addition, active vegetation and intensive construction and renovation works are often performed during these months [86]. Higher average AMP concentrations were found in areas with high daily traffic, such as the Merced neighborhood in Mexico City, which has a large market and is the commercial gateway to Mexico City [97]. Increased tourist traffic increases MP concentrations in areas with lower population density [34]. Therefore, population migration can change MP distribution in the atmosphere [158].

2.3. Current State of Research on AMPs

To assess the current state of research into AMP distribution, we chose 100 research papers published in the past 5 years (2019–2023). The papers were retrieved from academic research databases Scopus, ScienceDirect, and Google Scholar. The selection of articles was performed according to the PRISMA approach [159]. The keywords for retrieving were as follows: 'microplastic', 'atmosphere', 'airborne', 'dry deposition', 'wet deposition', 'contamination', 'rain', 'snow'. The number of research articles matching the keywords was about 2000. During the selection of articles, publications that were not relevant to the topic of AMPs, such as articles on marine or river MPs, were excluded. As a result, 125 articles were included in the selection. However, emphasis was placed on the geographical diversity of the studies, so some articles with duplicate regions were excluded. This resulted in a final selection of 100 articles. The retrieved papers address not only the content of suspended and deposited AMPs in certain territories and water areas but also the AMP content in moss, lichen, dust, snow, etc., since these are MP reservoirs that can be used to assess AMP concentrations in the study area. For convenience, information was divided into the following categories: year, country, city, urban/remote, outdoor/indoor, SAMP/DAMP, dry/wet deposition, methods, micro/nano, value.

The year category shows the year of publication to evaluate the progress dynamics in the study of AMPs and identify the chronology of discoveries of certain distribution patterns, which consistently interconnects the publications. The country category denotes the country where the study was conducted, yet the reviewed studies include those focusing on territories and water areas that belong to several countries [114] and those outside state jurisdiction, for example, the open ocean [160] or Antarctica [161]. Therefore, this category includes continents and parts of oceans and seas. The city category includes the city where the study was conducted, or another name of the territory, including the name of a geographical object where the study covered a large area [162] or was conducted

far from populated areas [163]. This allows more accurate study area localization and helps to assess the significance of local effects on the concentration of AMPs. The lack of data in this category indicates unavailability of more precise reference to the territory or water area [164]. The urban/remote category provides the type of study area, as the concentration of AMPs in urban and remote areas differs significantly [144]. The types of study area include urban, suburban, industrial, rural, and remote ones. In this category, we highlighted the features of some territories, for example, remote desert [165]. The studies are often conducted in similar areas [166] but in a comprehensive manner, including densely populated parts of a city, natural areas, and industrial and agricultural centers to compare the concentration of AMPs [167]. The outdoor/indoor category distinguishes the studies with regard to the concentration of AMPs outdoors [168], indoors (homes, offices, classrooms, etc.) [152], and comprehensive studies [88]. The SAMP/DAMP category groups the papers by the type of studied AMPs. This helps to choose an appropriate active or passive type of sampling and allows comparison between different studies, as the dispersion in values for suspended and deposited AMPs is wide [85,118,169]. The same applies to the dry/wet deposition category that deals with the deposition type or the sample type. This factor is essential, since the concentration of AMPs in dry and wet deposition differs [170]; different meteorological conditions, such as typhoons [171] and monsoons [172], are also considered in this category. The methods category describes the method used to analyze plastic samples, techniques to identify polymer particles among mineral and organic particles, and techniques to determine the polymer composition of the particles. Visual inspection may lead to overestimation of the number of plastic particles due to misidentification. Therefore, researchers use auxiliary tools, such as FT-IR or Raman spectroscopy [173,174], which are also employed to improve the results of their studies [175], since some methods are limited, for example, in the size of the particles to be detected. The micro/nano category determines the particle size, since not only micro- but also nano- [176] and meso-sized plastic particles [161] are subject to atmospheric transport and deposition. The last category, value, indicates the average MP content reported in the studies. Different goals set and study methods used by the authors caused discrepancies in concentration values and measurement units. The studies may include data on the AMP deposition rate per day or per year [35], the concentration of suspended AMPs [177], and the amount of plastics per gram of dry matter [178]. It should be noted that some researchers count plastic particles [179], and some of them measure MP concentrations in mass [165].

Published data are summarized in Table S1. According to the articles studied [34,35,40,64,88,94,107,144,156,161,162,169,170,173,175,180–191], we can identify regions with the highest and lowest average MP deposition per day per square meter. To summarize the information in the figure, we considered only outdoor DAMP identification methods: visual, hot needle, FT-IR, and Raman spectroscopy, since these indicators provide the largest number of matches in units of measurement. The main difficulty in comparing the data is the lack of a unified study methodology and, as a consequence, obtaining results in different units of measurement. The results are shown in Figure 3.

The analysis of the collected data yields the following conclusions:

- Over the past 5 years, the territory of China has become most studied in terms of the distribution and maintenance of AMPs in a single country. Of the 100 papers, 23 papers address studies of AMPs in China. Few studies have explored individual European countries over the past 5 years, yet in total, they exceed the number of studies conducted in China (36 papers out of 100). In the Asian region (excluding China), Iran stands out (8 papers out of 100). Studies of AMPs in oceans and seas are particularly specific (9 papers out of 100). Antarctica stands out among the most inaccessible and remote territories (3 papers). Very few studies focus on South America, Africa, and the Russian Federation. However, the lack of studies does not mean a lack of pollution in these countries;

- The number of studies that focus on the abundance and transport of AMPs in urban, industrial, and agricultural areas is greater compared with remote, inaccessible, or tourist areas. Research in urban areas is much easier to conduct, since samplers can be installed on the roof of the research facility or equipment sometimes requires stable energy to operate properly [192];
- The number of studies on outdoor AMPs many-fold exceeds the number of studies conducted inside residential, office, and industrial premises. Researchers should focus on this issue, since people spend most of their time indoors (home, work, study, and transport), and hence the risk of AMPs entering the human body is greater;
- The number of studies reporting the results for deposited AMPs exceeds that considering suspended AMPs. This discrepancy is due to the relative simplicity and low cost of passive sampling techniques;
- Researchers prefer to analyze only dry deposition or compare dry and wet deposition in the same study. Few studies have addressed wet deposition alone over the past 5 years, although wet precipitation is one of the main and most effective 'transmitters' of AMPs to other environments, for example, the soil or ocean.

The compiled table demonstrates the main trends in AMP transport. The concentration of AMPs in the urban air environment is higher [84,184,193] than that in remote areas [54,104,160] in both suspended and deposited states [180]. The concentration of AMPs and particle transport [62] in buildings [152,194] is higher than that outside [86,183,187]. AMP deposition is more intense during wet seasons compared with dry seasons [189]. Lichens and mosses and other plants used as study objects effectively illustrate the regional distribution and concentration of AMPs [173,178]; therefore, biomonitoring of AMPs is advisable. Research teams can improve the accuracy and objectivity of studies using instrumental assessment of the quantitative and qualitative composition of AMPs, since visual or hot needle methods can yield erroneous results, as confirmed by a number of studies [187–189,195,196].

Thus, by analyzing the original publications presented in Table S1, the main unexplored issues were identified:

- AMP transport on the African continent, in South America, in the Russian Federation, and in remote and hard-to-reach territories and water areas;
- Assessment of the concentration of AMPs in indoor air, study of AMPs suspended in the air;
- Study of the scale and mechanisms of wet deposition of synthetic microparticles;
- Gradual switch to advanced analytical methods and expanded studies in the field of nanoplastics.

3. Human Health Risks and Ecosystem Effects of AMPs

AMP pollution has detrimental effects on ecosystems as a whole, biota, and human health. This issue has been poorly studied; yet the worldwide production of plastics and, hence, AMP generation are growing. The ubiquitous nature of MPs in every environmental matrix, including the atmosphere, poses a threat to living organisms.

3.1. Physiological Effects of AMPs and Human Health Risks

Figure 4 schematically shows the mechanisms of interaction between AMPs and the human body. AMPs enter the human body via three main routes: inhalation, ingestion, and skin contact [197]. It has been estimated that a person inhales about 20 MPs per day outdoors [85] and up to 180 MPs indoors [196]. Infants are more susceptible to this effect, as they inhale MPs more often [172] spending more time on the floor, in contrast to upright adults [198]. Occupation has a particular effect on MP inhalation. People involved in the textile, flocking, and vinyl chloride/polyvinyl chloride industries (where the concentration of suspended AMPs can be up to 40 mg m^{-3} or >100 particles m^{-3}) are exposed to AMP-associated diseases [199,200]. Smokers are also at high risk for pathologies, since the concentration of MPs in the lung fluid is high; the largest fiber was found in a 75-year-old

smoking shoemaker who suffered from pulmonary parenchymal pathology [25]. Moreover, cigarette filters minimize exposure to more than 300 toxic substances, but complex and toxic compounds adsorbed on the cigarette surface in the form of microparticles can penetrate into human lungs as a result of filter degradation [44]. The COVID-19 pandemic particularly affected MP absorption; during the pandemic, masks were indispensable in society, which resulted in deposition of viscose and polyester fibers in human lungs [25]. People spent more time indoors during that period, which increased the risk of AMP inhalation [201]. Workers exposed to air contaminated with micro- and nano-sized plastic particles for a long time are more susceptible to respiratory diseases [153] and cardiovascular diseases [158]. Chronic inflammatory reactions occur because the rate of MP accumulation in the human body exceeds the rate of their elimination [69]. Most MPs can be subjected to sneezing, mucociliar clearance, phagocytosis of macrophages, and lymphatic transport [202]. However, the remaining particles can cause inflammatory reactions, particularly in people with impaired defense mechanisms [20,59]. Chronic inflammation can cause pulmonary fibrosis and increase a cancer risk [203]. Inflammatory reactions at sites of MP exposure can reduce the epithelial integrity and activate macrophages, thus increasing MPs' mobility in the body [204–206].

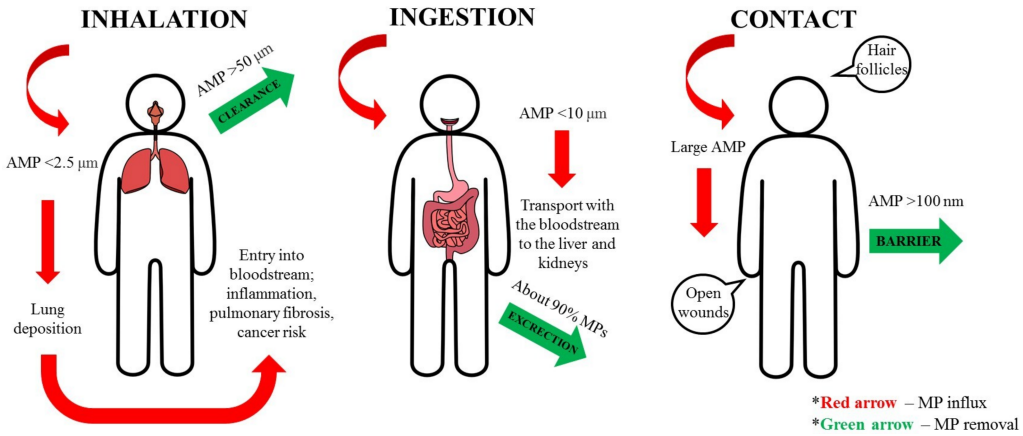

Figure 4. Interaction between AMPs and the human body.

AMPs can enter the gastrointestinal tract with contaminated food through the food chain [207,208]. MPs were found in table salt, sugar, honey, milk, and beer [209,210]. Dust and suspended MPs that come from packaging abrasion can be a source of MPs in food products. For example, the amount of tiny plastic particles released into bottled water increases by repeatedly opening and closing the cap of the bottle, while the concentration of MPs in tap water is lower [211]. The entry of MPs into the human digestive tract is evidenced by their detection in feces [212]. In humans, more than 90% of ingested micro- and nanoplastics are reported to be excreted through the excretory system [59,213]. Of least concern is dermal exposure to AMPs via contact with cosmetics, fabric–skin contact, and deposited dust particles [214].

The size and density of plastic particles are a great concern since MPs of smaller size and lower density enter the human body more easily [202]. Particles larger than 50 µm can be effectively filtered in the human body [44], but particles smaller than 2.5 µm can deposit deep in the lungs, penetrate the respiratory tract, and reach the circulatory system [69]. Particles up to 20 µm in size cannot be removed from the body by the lungs [215]. Fournier et al. [216] described the transport of polymeric nanoparticles inside the human body on the example of maternal–fetal transfer of polystyrene through the placenta that connects the maternal and fetal circulatory systems. After the woman's exposure to MPs through inhalation, particles were found in the liver, heart, brain, lungs, and kidneys of the fetus. Twenty-four hours after maternal lung exposure to nanosized polystyrene particles, the fetal and placental weight decreased, and the number of reabsorption sites increased.

Particle accumulation during fetal development can affect the health of the baby after birth and into adulthood [216].

Larger particles can enter the body through the gastrointestinal tract [44,118]. Although the skin is a barrier to MPs, particles smaller than 100 nm can penetrate through the epithelial barrier. Hair follicles, sweat glands, and open wounds are also potential routes for larger particles' entry through the skin. Particles smaller than 10 μm are reported to cross the mucosal barrier of the gastrointestinal tract and travel through the bloodstream, accumulating in other organs, such as the liver or kidneys [217]. MPs can interact with hemoglobin and interfere with oxygen transport in biological organisms [59]. Particle shape plays an important role in MP transport, since smaller angular particles pass through membrane barriers more easily than those with regular surfaces or longer edges [72]. The toxicity of non-spherical MPs (fragments, fibers, films, foams) with a size exceeding 10 μm is higher than that of spherical MPs with a size of less than 10 μm. Comparison of spherical and non-spherical particles of the same composition with a size of less than 10 μm revealed higher ecotoxicity of non-spherical MPs [197]. The adverse effect of fibers may be due to their bending rigidity, which can potentially damage tissue [218]. In vitro studies of human epithelial and cerebral cells showed the potential cytotoxic effects of micro- (<10 μm) and nano-sized (40–250 nm) plastic particles on human cells in terms of oxidative stress [219–221].

3.2. Adverse Effects of AMPs on Ecosystems

AMP degradation due to UV radiation, physical wear, and other abiotic factors causes particle fragmentation and reduction. As particle sizes decrease, the effect on biota is hypothesized to become more apparent and switch from physical to chemical mechanisms [222]. When exposed to UV radiation, PS and phenol–formaldehyde resin (PFR) MPs generate large amounts of persistent free radicals and reactive oxygen free radicals, which can be easily absorbed [223]. Photooxidative degradation is a chain mechanism of free radical generation that occurs when a polymer is exposed to UV radiation in the presence of oxygen.

Micro- and nano-sized plastic particles pose lower risks to people living in high mountain areas compared to those living in sea-level areas with increased anthropogenic activities [224]. The environmental risk of the open ocean is low compared to nearby coastline areas [224]. Moreover, the environmental risk of air pollution may depend on the season of the year. Air in the port of Bushehr (Iran) was more saturated with AMPs in winter compared to other seasons, which was apparently due to heat sources, ventilation systems, and different atmospheric conditions. Bushehr residents are also at greatest risk of exposure to AMPs and polycyclic aromatic hydrocarbons (PAHs) on dusty days. Dust of petrogenic origin is transported from Iraq and Saudi Arabia [221].

AMP deposition in the marine environment can be detrimental to marine life [139]. In this case, bioavailability of MPs is known to increase as the particle size reduces. More than 95% of AMPs ingested by zooplankton are smaller than 500 μm [225]. Microfibers can have an adverse effect on digestion, reproduction, and development and change the histological structure of the gastrointestinal tract in marine organisms. Marine organisms tend to ingest blue and transparent fibers, which may be due to their resemblance to plankton [226]. Hence, the ingestion of MPs can cause false saturation, which decreases feeding activity and ultimately leads to intestinal blockage [227,228]. In addition, uneven and sharp edges of MPs can cause mechanical damage to the esophagus and the entire digestive system [229–231]. MP ingestion in marine animals was found to cause metabolic disorder, immune system depression, and other dysfunctions (e.g., necrocytosis, inflammation, tissue damage) and sometimes behavioral abnormalities [230,232,233]. MPs can cause gene hyperexpression [234] and oxidative DNA damage [235–237]. In marine organisms, nanoplastics can cause adverse health effects, including inhibited growth, development delays, larval malformations, and subcellular changes such as increased levels of reactive oxygen species (ROS) and lysosome destabilization [238,239]. The formation of an eco-corona (a multilayer

structure formed by encapsulated particles) may enhance the toxic effects of nanoplastics on organisms by inhibiting growth, increasing cellular stress, and altering the uptake, retention, and bioavailability of nanoparticles in marine organisms [102].

Particles with a size of 100 nm can penetrate into the roots of *Vicia faba* L., disrupting the transport of nutrients in the cell membrane [240]. AMPs can enter plants directly through the leaf blade and completely inhibit water circulation [241]. Apparently, plant leaves can accumulate and transport AMPs. Secondary transfer of AMPs occurs more easily from trees than from the Earth's surface, while resuspension occurs at the height of a person's inhalation, which is a risk factor. In this regard, it was proposed to use the amount of AMPs on tree leaves as an indicator of air pollution [242].

MPs can have direct and indirect adverse effects on photosynthesis. The direct effect implies MP attachment and integration into the phytoplankton cell membrane, which decreases chlorophyll synthesis and photosynthetic efficiency; MP adsorption and bioaccumulation on microalgae cells causes a shading effect, thereby affecting photosynthetic efficiency [243,244]. MP deposition on the surface of water bodies has an indirect adverse effect on photosynthesis, since it prevents the penetration of sunlight into the water column, thereby suppressing photosynthesis in microalgae [31,244–246]. Different ages and compositions of particles have different effects on photosynthesis efficiency. Falsini et al. [247] experimentally showed that the photosynthetic activity of leaves of *Tillandsia usneoides* L. decreased under exposure to airborne PVC, but this effect weakened over time. When plants were exposed to other aged MPs, their toxic effect was higher than that posed by aged PVC [247]. The experiment identified the following toxicity levels of aged MPs, which adversely affected and thus decreased plant growth rate: PET > PE > PVC. In this case, unweathered PVC was the most toxic, since it can cause oxidative stress and disrupt the ionic balance, which decreases the growth rate [247]. Among the common types of plastics, PVC requires the largest amount of additives to stabilize the polymer [248]. Blockage of free electrons of the original polymer by newly formed bonds with oxygen atoms decreased the toxicity of PVC over time, resulting in a less hazardous material [247]. The authors showed that aged PET, PE, and PC increase their hydrophilicity when wetted and can oversaturate plants with moisture and nutrients, thus inhibiting plant growth.

The toxic level of MPs can depend on the particle charge. Positively charged particles were found to exhibit higher toxicity and cause higher ROS generation compared to negatively charged ones [249,250]. Particle charge affects both retention and excretion [221] and promotes adhesion of other compounds and pathogenic microorganisms. This can stimulate formation of biofilms for active horizontal gene transfer, which allows microorganisms to adapt to antibiotics [251].

The hydrophobicity of plastic products affects their interaction with biological systems. The water-repellent properties of PVC deposited on the leaves of *Tillandsia usneoides* inhibited plant saturation with moisture and nutrients and thus decreased the growth rate [247]. AMPs can transport hydrophobic contaminants, such as persistent organic contaminants, heavy metals in an organic form [252–254], and microorganisms, including pathogens [251]. Once deposited on the sea surface, AMPs are rapidly colonized by organisms and become new habitats for some marine communities [255], and plastic-associated species indicate colonization zones and spatial distributions of exotic, invasive, and pathogenic organisms [256,257]. AMPs can be associated with metals that contaminate the urban environment, for example, Cd, Zn, Ni, and Pb [253]. Li Y et al. reported [44] that microfibers quite easily adsorb phthalates, organic flame retardants, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DTT), polybrominated diphenyl ethers (PBDEs), hexachlorocyclohexane (HCG), organochlorine pesticides (OCPs), hexabromocyclododecane (HBCD), bisphenol A (BPA), nonylphenol (NP), perfluoroalkyl substances, etc. Adsorption processes are based on different chemical interactions, and the most frequent mechanism of adsorption of organic pollutants by microplastic particles is hydrophobic interaction. There are also other types of bind-

ing: halogen bonds, π–π interactions, hydrogen bonds, electrostatic interaction, and other non-covalent forces [258].

It should be noted that particles of different plastics have different abilities to adsorb toxic substances; for example, PVC and PP [259] readily adsorb heavy metals. In the marine environment, water-soluble contaminants are six times more likely to occur in association with MPs than in the surrounding water [260]. This transfer is likely to cause a Trojan horse effect [214,261], when accumulated toxicants are desorbed from MPs in the body [221]. Larger amounts of PAHs are released from MPs in contrast to their natural deposition during desorption [70].

Plastic additives play a particular role in the release of toxic chemicals. They are added to polymers to impart specific properties, reduce production costs, and improve physical properties or appearance, and their content can range from 20 to 70% of the total mass of the polymer [97,262,263]. For example, Si, Al, and Ca are used as fillers in plastics production, while Ti and Fe are used as pigments [264]. Some additives can improve the durability of plastics, for example, microfibers can persist in the soil for at least 30 years [265]; in the extracellular fluid of the lungs, plastic fibers did not change their surface area over 180 days [266]. A number of plastic particles may taste like food to seabirds, such as albatrosses, and other animals, which increases the risk of MP ingestion [267,268]. Environmental pollution caused by modified natural fibers should also be considered. Studies showed that more than 60% of cellulose-based products can completely biodegrade in the natural environment within 243 days [109], but surface treatment with dyes or flame retardants significantly reduces fiber degradation [269–271]. Therefore, some authors argue that modified natural fibers (most often viscose) should be included in quantitative assessments of pollution along with synthetic fibers [272].

The concentration of MPs in the soil is assumed to several-fold exceed that in the ocean, which poses a threat to the environment [273]. MPs affect not only soil properties but also the structure of microbial communities, thereby increasing the number of anaerobic prokaryotes and the amount of CO_2 and methane emissions [274]. Plastic microfibers reduce soil density, alter water-retaining capacity, and alter soil structure [231], thus reducing resistance to plant root growth. Changes in soil structure can change the rate of microbial processes, which can affect the nitrogen and phosphorus nutrient cycling [275], and increase water evaporation by creating water flow channels. Soil drying and cracking facilitate the migration of contaminants into deeper layers [276]. Changes in the soil structure largely depend on the type of AMPs deposited or introduced. Hence, HDPE reduces soil pH [277], PP increases microbial activity [278], and nano-sized PS inhibits the development of soil microbial communities and their enzymatic activity [279]. The entry of AMPs into agroecosystems can reduce soil fertility, and many agricultural crops can absorb plastic particles, transporting them from roots to shoots, which is detrimental to plants and promotes AMP transfer in food chains [280]. Plastic particles can act as pesticide collectors in the soil and change habitats for soil organisms. The presence of 1–2% MPs in the soil is sufficient to inhibit the growth of earthworms and increase their mortality, which can deteriorate the quality of the soil cover [281].

AMPs can negatively affect not only organisms but also the abiotic components of ecosystems. MPs are deposited in cryosphere zones [114,282,283] and persist there for a long time due to low temperatures [101]. AMP accumulation in cryoconites allows the use of ice cores as stratigraphic indicators of AMP deposition, providing valuable information on historical contamination [89,282]. However, plastic particles can scatter and absorb light, thereby affecting the radiative forcing by up to 0.75 mW m^{-2} [284]. Colored particles, especially those of dark colors, can increase the absorption of UV radiation or reduce the albedo of fresh snow deposition [103]. This may stimulate melting of glaciers [285] and sea ice [286,287], causing even greater release of MPs into the environment [136,288]. For example, melted glaciers on the Tibetan Plateau can add up to 8% to the total amount of MPs entering downstream rivers and lakes [289]. Annual melting and freezing of sea ice can retain MPs in the surface/subsurface waters of the Arctic, which will increase the

volume of MP circulation in the atmosphere and seawater. Together with the new input of MPs from rivers, this effect will only increase with a dramatic decline of sea ice due to global warming [286]. AMPs can act as cloud ice nuclei [68], which is particularly important in areas of the Southern Ocean where clouds are very sensitive to the concentration of ice-nucleating particles [290].

4. Methods for Studying AMPs

The path of MPs from the natural matrix to the analytical laboratory is quite complicated and includes several stages (Figure 5). These stages vary depending on the study objectives, the study area, and the equipment available. This is the reason for the lack of a unified technique for sampling, processing, and analyzing AMPs, which complicates comparison of different studies on the distribution of polymer particles. In general, most research teams employ several methodological approaches. Table S1 summarizes systematized data.

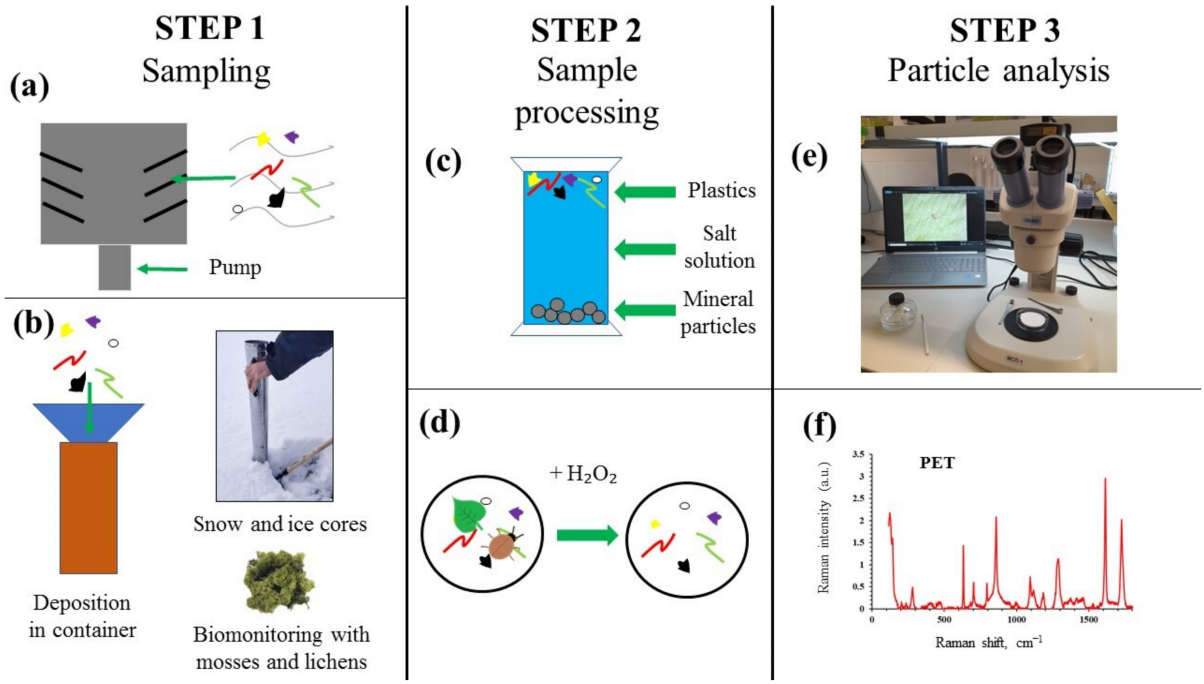

Figure 5. Stages in studying AMPs: Step 1—Sampling by active (**a**) or passive (**b**) methods; Step 2—Laboratory processing with density separation (**c**) and digestion (**d**); Step 3—Particle analysis: quantitative counting, determination of size, shape, color (**e**) and polymer identification (**f**).

4.1. AMP Sampling

The first step includes sampling to examine samples for the presence of MPs. Sampling techniques differ for suspended and deposited MPs. Thus, active sampling is used to sample suspended AMPs. This technique is actively used for various atmospheric pollutants [158]. It typically employs electric pumps, which pump out a certain volume of air from the medium [291] and pass it through filters [158]. Data on the filtration rate and the volume of air samples [291] are used to calculate the concentration of suspended MPs. The volume can reach tens to hundreds of cubic meters [201,221,292]. Glass fiber filters with a pore size of 1.6 μm are the most popular [158]. Yet, some studies employed paper filters with a pore size of 0.45 μm [221], nylon filters with a pore size of 20 μm [62], and steel filters with a pore size of 50 μm [148]. Yuan [158] assumes that filter pore size ranges from 0.22 to 250 μm, but 20 μm filters are most appropriate for field studies since this pore size is a visual physical limit for human recognition [293]. Of particular importance is the height of the sampler, which depends both on physical characteristics (height of the ship [139]) and on study objectives. For example, Liu K et al. [85] installed their sampler at a height of 1.7 m,

which corresponds to the standard breathing zone height. González-Pleiter et al. [292] collected samples on an airplane at altitudes ranging from 701 m a.s.l. to 3496 m a.s.l. A number of authors [62] installed the sampler indoors at a height of 70–75 cm, which corresponded to the height of desks in the homes and offices where sampling was performed; on buses and subway trains, the sampling height corresponded to the height of passenger seats. Abbasi et al. [196] installed filters at a height of 10 m to eliminate anthropogenic effects. AMP sampling on the streets of Krakow was carried out at a height of 35 m at a special site to prevent contamination from the roof [86]. Akhbarizadeh et al. [221] also took into account the presence of high-rise buildings in the area, which may affect samples.

Numerous sampling techniques are used to collect deposited MPs, which depend, among other things, on the area of MP sampling. They are referred to as passive sampling techniques. Passive sampling is much simpler than active sampling since it does not require active intervention and energy costs [158]. Passive samplers used to collect dry and wet deposition [120] include mainly various types of funnels, containers, and tubes made of steel, glass [120,158], and some types of plastic [186]. Petri dishes are used to study deposited AMPs indoors [288]. This technique is cost-effective and easy to use but time consuming. Other passive sampling techniques involve sample removal from the natural environment, for example, [173] describes biomonitoring of mosses and lichens for the presence of AMPs, allowing rapid assessment of the regional AMP distribution [44]. The cryosphere, known for its snow, ice, and low temperatures, often lacks infrastructure. In this regard, it is impossible to conduct long-term experiments, and researchers need to sample snow and ice cores. Ice core sampling provides data on polymer distribution over time, for example, the lower limit of the core reported in the study by Materic et al. [176] was dated to 1966. Core sampling is mainly carried out using hand drills and augers [101,288]. Plastic materials are preferable to steel and glass for sampling mountain glaciers since they are lightweight. However, this can lead to contamination of the samples, which necessitates the use of non-plastic materials and thorough rinsing of sample storage bags; lightweight drills with a small diameter (about 5 cm) can be effectively used for core sampling [89]. Techniques for sampling deposited AMPs include dust collecting with a brush and a metal dustpan [294]. Special focus is on the brush type. In particular, its material should be antistatic [118] and minimize potential contamination from brush fibers [33]; it can be made from natural bristles [294] or from dried plant stems, which can facilitate cleaning and remove any dust particles stuck after sampling [119]. Vacuum cleaners can be used to collect dust with deposited AMPs [118].

4.2. Laboratory Processing of Natural Samples to Account for AMPs

After collection, samples are transported to the laboratory for further processing, cleaning, and removal of excess materials that could interfere with the analysis of MPs. These include various organic and mineral particles, such as sand, glass, and stones [295]. First, the samples are cleaned of organic contaminants, since they float up along with the MPs during separation and can clog vacuum filters, which increases filtration time [296]. Fenton oxidation, 30% hydrogen peroxide, acid digestion, base digestion, and enzymatic digestion, alone or in combination, are used to remove natural organic matter from the surface of MPs [297,298]. These methods have different effects on organic matter and have different advantages and disadvantages. For example, a 30% H_2O_2 solution [221] removes up to 86% of organic matter but damages PA, PP, and PS plastics at 70 °C. Alkali solutions have a negligible effect on MPs (NaOH damaged only PET and PC, and KOH reduced the mass of PC) [297], but they are significantly less effective in removing organic matter, up to 67% and 57%, respectively [298]. At the same time, the efficiency of oxidation of organic substances using H_2O_2 under certain conditions can reach 70–95% [299]. Selection of conditions and concentrations is of high relevance due to a high reactivity of these substances.

Concentrated acids must be used with care as they may damage plastic objects [298], which can subsequently reduce the melting point to 90 °C [121]. HNO_3 and HCl are commonly used [300]; they effectively decompose organic matter up to 98% [301]. Hy-

drochloric acid can remove calcium carbonate, which can potentially cause noise during Raman spectroscopy [201]. Enzymatic digestion does not affect plastic integrity [302], but it is only effective for processing a few samples since this process is time consuming and takes from 24 hours to 30 days [303]. A number of authors report Fenton's reagent as the optimal method for organic matter removal [296,304]. This method virtually does not affect the plastic surface. However, decomposition of organic matter is an exothermic reaction, increasing the temperature of the solution to 89 °C, which can affect MPs; hence, an ice bath should be used to control the temperature [297].

Physical methods are mainly used to remove mineral particles. The most popular method is density separation [295,305] with an optimal solution density of 1.6–1.8 g/cm^3 [306]. Some light polymers (PP and PE) can be easily separated by density sorting in a water medium [295], since their density is <1.0 g/cm^3 [300]; this method is fairly easy and cheap. However, isolation of heavier polymers requires denser solutions and mixtures. The choice of solution depends on the density, cost, filtration ability, and toxicity risks to humans and the environment. Hence, NaCl solutions are recommended for PP and PA isolation [307]; sodium iodide salts (NaI) are used for PET and PVC, but this method is quite expensive [308]. $ZnCl_2$ solutions are effective for separating particles by density [148], but they exhibit pronounced corrosive properties [210]. Mixtures of solutions are feasible to create solutions with a greater density at a lower cost, for example, the density of NaCl solution is about 1.2 g/cm^3 [309], and when sucrose is added, its density reaches 1.38 g/cm^3 [310]. Data on other solutions are also available in the literature: NaBr, $CaCl_2$, $ZnBr_2$ [303], KI [221], etc. The process of separating MPs from mineral particles is time consuming and can take up to 72 h [303]. Centrifugation [221] or sparging [311] are employed to speed up separation, since these methods can reduce the time required for density separation [221]. These procedures must be carried out several times in order to increase the efficiency of MP extraction from solution [312].

After filtering, direct identification of microplastics can be performed using either electron microscopy imaging (supplemented with energy-dispersive X-ray spectroscopy) or using FTIR spectroscopy.

4.3. Polymer Analysis of AMPs

After processing the samples and removing excess materials, MPs are analyzed. There is no unified methodology for this stage yet, so researchers use mostly visual methods, Raman spectroscopy, Fourier transform infrared spectroscopy (FT-IR), and pyrolysis gas chromatography–mass spectrometry. The visual method is the simplest and cheapest one and requires only a microscope, but it is currently criticized as being unreliable due to possible underestimating or overestimating of MP abundance [313,314]. Therefore, this method is used mainly to describe characteristics such as type, shape, color, and stage of plastic decomposition [295]. Visual inspection employs criteria to distinguish MPs from other materials: plastic fibers show a uniform or constant thickness along the entire length; AMPs are of irregular shape and have a worn surface and sharp and broken edges [65]; synthetic fibers have a smooth surface compared to natural fibers; natural fibers are less flat and twisted unlike plastic fibers [315]. To determine the exact size of particles, authors use various types of software to measure particles, for example, ImageJ v. 1.51j8 [85,196] or ImageFocus 4 [292]. The hot needle method is proposed [51] to facilitate the identification of MPs during visual inspection, but this method cannot precisely identify the plastic nature of the particles [65]. Researchers often use scanning electron microscopy (SEM) [36] to obtain data on the morphological characteristics of samples and detect traces of MP degradation—grooves, flakes, pits, cracks, and other adherent particles [316]. To improve the quality of MP identification, the dye Nile Red is often used; its fluorescence occurs only in the hydrophobic environment, which makes it possible to identify even small-sized MPs that could be missed during optical scanning [295,317]. This method is simple and inexpensive [303], employs available equipment, and allows semi-automated sample analysis [207]. The drawback of the Nile Red method is complicated identification of

MPs [318], as the fluorescence intensity varies for different polymers. Thus, PET and weathered PE fluoresce less than PP and unweathered PE [319]. In general, the Nile Red staining of plastics under the microscope ranges from bright yellow to white, while organic matter is stained orange-red [186].

The non-destructive Fourier transform infrared (FT-IR) method is used to determine the composition of plastic particles [312]. This method requires expensive equipment and a skilled operator [36]. FT-IR spectra of an unknown sample are compared with available libraries of reference pure polymers—OMNIC, HR Aldrich Polymers, HR Coatings Technology, HR Hummel Polymer and Additives, and other commercial libraries [55,312,320]. The main disadvantage of FT-IR is the limitation in the size of the analyzed particles. Although it can now analyze particles 5–10 μm in size [321], it is time consuming to analyze plastics smaller than 20 μm [303]. Another non-destructive method for analyzing the polymer composition of plastic is Raman spectroscopy [186,322,323]. The benefit of the method is the ability to analyze particles of 1 μm or smaller [139,322]. In this case, the signal-to-noise ratio for small-sized AMPs (20 μm) is lower than that in Fourier transform infrared spectroscopy [324], and MP detection is 23% more efficient than that performed by FT-IR [325]. The MP composition is determined by comparing the Raman spectrum of the particle with reference spectra (SLOPP [201], KnowItAll® ID Expert from Bio-Rad [221], etc.). The drawbacks of this method are similar to those of the Fourier transform method, namely, it requires a skilled operator and expensive equipment and sample analysis is time consuming.

Another versatile tool extensively used for MP analysis is pyrolysis gas chromatography–mass spectrometry [294]. Since the method is destructive, plastic samples cannot be further analyzed [326], but it allows not only qualitative analysis of small-sized plastics but also their quantitative analysis [327]. In contrast to FT-IR, gas chromatography performs a comprehensive analysis of the particle, not just its surface, thereby minimizing the effect of various additives or pigments [328,329].

4.4. Data Analysis and Study Quality Control

After quantitative and qualitative analysis of sampled AMPs, statistical analysis is conducted to determine MP sources and evaluate dependence of their distribution in the environment on various factors. Statistical analysis software includes IBM SPSS Statistics v. 23.0, v. 25.0 [62,185,330], GraphPad Prism v. 8.0.1 [294], Microsoft Office Excel [331], R v. 1.2 1335 and v. 4.0.2 [34,320], Canoco v. 5.12 [129], and others. These programs comprise Kruskal–Wallis tests [185], paired and unpaired *t*-tests, Pearson correlations [294], ANOVA, Tukey tests [62], Mann–Whitney U tests [332], redundancy analysis (RDA) [129], and many other tools to study AMPs. The statistical analyses have determined the dependence of the amount of suspended AMPs on meteorological factors [34,85]; the significance of differences in the concentration of MPs in samples [320]; the dependence of MP abundance on location [62]; dependence of the concentration of MPs in samples on the average annual traffic, traffic speed, type of road surface, and distance from the road [129]; the effect of population density [90].

MP distribution modeling can help to more effectively determine AMP sources. The Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model is most widely used to build the trajectories of the atmospheric transport and dispersion of pollutants, including MPs [292]. The model is employed to build back trajectories [221] with regard to multiple meteorological inputs, physical processes, and different types of emission sources and provides integrated particle transport, dispersion, and deposition models [55] that simulate sample-derived particles. For this, the equivalent diameter of MPs is calculated [292] and the period of particle distribution and air-mass trajectory [333], the flight altitude of particles [286], etc. are modeled.

Monitoring and assessing the quality of sample analysis are of particular relevance since external contamination can affect the study results. For this purpose, ultrapure conditions [242] and control samples [221] are created. The utensils and tools are cleaned

with acetone [247], ethanol [294], and distilled water [323]. It is recommended to use tweezers, scissors, and other instruments made of non-plastic materials, namely, stainless steel or glass [85,176,320]; the instruments must be heated to 450–500 °C to remove possible adsorbed organic compounds [120,221]. Clothing plays an important role, so it is recommended to wear cotton clothing [323] of bright colors, such as yellow, orange, or purple, in order to exclude fibers of these colors during calculations [292]. Latex gloves [323] are mandatory. Field sampling is carried out at a downwind location [286], without gloves, or in cold conditions with gloves made from natural fibers (wool, leather, cotton) [288]. To avoid external contamination during transportation, samples must be covered or wrapped in aluminum foil [242,294,323] and placed in a refrigerator at 4 °C [242].

5. Problems and Prospects when Studying AMPs

AMP pollution is a global problem; however, studies often report only preliminary assessments, which cannot provide an insight into certain mechanisms of AMP distribution and its negative effects. In this regard, the study of AMPs needs to be addressed at an interdisciplinary transnational level [50]. Yet, the results of different research teams are difficult to compare because they use different sampling techniques and methods of analysis. Moreover, there is an ongoing scientific debate about the definition of MPs [334] and their classification [36], which may lead to conflicting conclusions. Hence, researchers must combine their efforts to advance available sampling techniques and develop standardized methods for studying AMPs [46,64,335]. Intercalibration with standardized units of measurement will contribute to convergence of verification and data comparison. To date, some recommendations on the improvement of the quality of studies have been elaborated. These include simultaneous sampling of suspended and deposited AMPs [48]; FT-IR, Raman spectroscopy, and mass spectrometry used to obtain accurate results [37]; quantitative data in terms of the abundance and mass concentration of AMPs [336]; long-term monitoring [138]. The development of highly efficient methods and tools for effective extraction and rapid and accurate identification of AMPs can provide a faster and better analysis of plastic samples [70].

Some understudied aspects of this issue need to be addressed to gain in-depth knowledge about AMPs:

- Chronic exposure to high MP concentrations can lead to adverse physiological effects, but direct evidence and an adequate model for assessing the human health risk have not been obtained yet [48]. It is important to study the effect of AMPs on sensitive people, such as children and people with respiratory diseases [158]; a high risk of negative physiological changes can be observed among personnel of textile industries [25,58,158]. More comprehensive studies are needed to understand the role of AMPs in pathology [25] and develop effective protective equipment and occupational hygiene rules for personnel. Of relevance is the study of the combined effects of additives [337] and other chemicals adsorbed on plastic particles;
- Another aspect that has not yet been studied is the mechanisms of AMP distribution. This requires more physicochemical and meteorological variables used for monitoring [46] and focus on densely populated but poorly studied areas of Africa, South America, and the Middle East to better identify pollution sources and avoid listing broad categories of potential sources [46]. Further studies are needed to examine the vertical distribution of AMPs and the characteristics and transport of AMPs at high altitude. Modeling the transfer of AMPs is crucial to study MP transport in the atmosphere [65,336] with regard to natural and anthropogenic factors of AMP pollution. Modeling is used to determine and describe the dynamics of MP transfer to the atmosphere, but the available models represent highly simplified estimates [336]. The interaction of AMPs with each other and with other atmospheric particles during their transport must be thoroughly studied. The relationship of AMPs with terrestrial and aquatic polymer synthetic particles [77] and their behavior in the lithosphere and

hydrosphere must be investigated [138]. To develop better models, it is necessary to study the mechanisms and degradation rates of plastic particles [219];

- Of particular importance is the study of nano-sized plastic particles, which is associated with a number of difficulties. Currently, it is impossible to effectively detect particles smaller than 10 μm. There is an increased focus on improving methods for smaller particle identification (<10 μm) [336]. In addition, smaller AMPs are more susceptible to long-range atmospheric transport. Data on atmospheric nanoplastics are particularly relevant, since particles of this fraction better adsorb toxic substances [338], can enter the bloodstream and lymph, and travel to other parts of the body [339].

AMP pollution occurs mainly in urban areas, yet it is necessary to investigate hard-to-reach and remote areas, for example, in mountain ecosystems and the Arctic and Antarctic. The lack of infrastructure and energy and severe environmental conditions hamper studies in these areas [340]. In this regard, there is a need for new portable and rapid techniques of sampling and analysis [322], which must involve volunteers [341], local residents, and advanced technologies, such as drones or autonomous stations [340].

The carbon balance and the impact of AMPs on climate change and biodiversity are the issues to be explored. MPs can affect carbon dynamics and have a significant impact on biochemical cycles [342,343]. Moreover, different MPs are of different colors, including dark ones, which can change the albedo of the Earth's surface. This is critical for the cryosphere, and it causes melting of glaciers and the release of even more 'conserved' MPs [343]. Hard-to-reach areas harbor a variety of endemic species that have limited food webs, so environmental changes caused by AMP pollution can have a significant effect on biodiversity [74]. An ecosystem approach with biota sampling used to study AMPs can provide data on the bioaccumulation, biomagnification, and transport of plastics in the biosphere as a whole [340,344].

However, in addition to the study of AMPs, it is necessary to create methods to reduce their emission and adverse effects. The most obvious opportunities involve elimination of AMP emissions 'at the source' [44]. For example, the concentration of MPs in domestic wastewater can be significantly reduced by the combination of different filtration techniques and membrane bioreactors; MPs can also be efficiently separated using electrocoagulation and agglomeration combined with multistage filtration [43]. Technologies for capturing MPs, which can serve as a material for new products, are of a great interest [345]. The ultimate solution to plastic pollution is its prevention by reducing consumption, as well as by collection and reuse, recycling, and energy recovery [346]. Yet, this problem cannot be solved in the near future, since economic growth is driving up global plastics production. Accordingly, the policy of production and processing of synthetic polymers should be reconsidered. For example, a transition to a climate-neutral circular economy is suggested, which includes the development of new business models for plastics reuse, manufacturing from recycled plastics, as well as for reducing the dependence of materials on fossil oil and gas [6,44,347] and switching to biodegradable plastics [348] and natural materials (particularly textiles) [44]. This comprehensive approach requires changes in materials, design, and recycling practices for plastic waste. However, these are limited by economic feasibility [4] and affect many plastics manufacturers, processors, and even plastics manufacturing and processing equipment manufacturers [6]. It is evident that the legislation must include the requirements for control of the release of primary microplastics, for example, in cosmetics [50,346,349], restrictions on the use of single-use plastic in certain areas [350], and waste sorting as a fairly effective method to reduce anthropogenic environmental load [6].

6. Conclusions

In this review, we have examined the issue of AMPs from a variety of perspectives. The principal sources of atmospheric microplastics are synthetic textiles, tires, industrial processes, and landfills. The combination of polymer particle analysis methods with plastic particle dispersion modeling facilitates more precise identification of the sources of AMP emissions and distribution factors.

The primary objectives in the development of methodology for studying AMPs are the implementation of a unified research protocol, the identification and accounting of microplastics up to 10 μm, and the identification and accounting of nanoplastics. It is precisely these particle sizes that are associated with the risks of transport and accumulation of AMPs in human organs through inhalation and ingestion, causing chronic inflammation and oxidative stress.

The development of a special environmental policy in the field of production, use, and disposal of plastic plays a pivotal role. There should be an open international discussion and decision-making process, as our review demonstrates, as the problem of AMPs is global in scope.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/atmos15111380/s1, Table S1: Results of published studies on AMP quantification. References [10,33–35,40,54,55,62,64,84–86,88–91,94,97,101,103,104,107,114,118,129,132,136,139,141,143,144,152,156,160–197,201,221,242,283,286,292,294,312,320,323,330,332,333,351–366] are cited in the Supplementary Materials.

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References

1. Thompson, R.C.; Swan, S.H.; Moore, C.J.; Vom Saal, F.S. Our plastic age. *Philos. Trans. R. Soc. B Biol. Sci.* **2009**, *364*, 1973–1976. [CrossRef] [PubMed]
2. Dokl, M.; Copot, A.; Krajnc, D.; Fan, Y.V.; Vujanović, A.; Aviso, K.B.; Tan, R.R.; Kravanja, Z.; Čuček, L. Global projections of plastic use, end-of-life fate and potential changes in consumption, reduction, recycling and replacement with bioplastics to 2050. *Sustain. Prod. Consum.* **2024**, *51*, 498–518. [CrossRef]
3. Nayanathara Thathsarani Pilapitiya, P.G.C.; Ratnayake, A.S. The world of plastic waste: A review. *Clean. Mater.* **2024**, *11*, 100220. [CrossRef]
4. SYSTEMIQ. *ReShaping Plastics: Pathways to a Circular, Climate Neutral Plastics System in Europe*; SYSTEMIQ: London, UK, 2022.
5. Geyer, R.; Jambeck, J.R.; Law, K.L. Production, use, and fate of all plastics ever made. *Sci. Adv.* **2017**, *3*, e1700782. [CrossRef]
6. Plastics Europe. *Plastics—The Facts 2022*; Plastics Europe: Brussels, Belgium, 2022.
7. Waste Atlas. 2019. Available online: http://www.atlas.d-waste.com/ (accessed on 31 January 2024).
8. Kumar, A.; Samadder, S.R. A review on technological options of waste to energy for effective management of municipal solid waste. *Waste Manag.* **2017**, *69*, 407–422. [CrossRef]
9. Li, W.C.; Tse, H.F.; Fok, L. Plastic waste in the marine environment: A review of sources, occurrence and effects. *Sci. Total Environ.* **2016**, *566–567*, 333–349. [CrossRef]
10. Mohan, P.M.; Tiwari, S.; Karuvelan, M.; Malairajan, S.; Mageswaran, T.; Sachithanandam, V. A baseline study of meso and microplastic predominance in pristine beach sediment of the Indian tropical island ecosystem. *Mar. Pollut. Bull.* **2022**, *181*, 113825. [CrossRef]
11. Ward, C.P.; Armstrong, C.J.; Walsh, A.N.; Jackson, J.H.; Reddy, C.M. Sunlight Converts Polystyrene to Carbon Dioxide and Dissolved Organic Carbon. *Environ. Sci. Technol. Lett.* **2019**, *6*, 669–674. [CrossRef]
12. Lassen, C.; Hansen, S.F.; Magnusson, K.; Hartmann, N.B.; Rehne Jensen, P.; Nielsen, T.G.; Brinch, A. *Microplastics: Occurrence, Effects and Sources of Releases to the Environment in Denmark*; Danish Environmental Protection Agency: Odense, Denmark, 2015.
13. Cole, M.; Lindeque, P.; Halsband, C.; Galloway, T.S. Microplastics as contaminants in the marine environment: A review. *Mar. Pollut. Bull.* **2011**, *62*, 2588–2597. [CrossRef]

14. Song, Y.K.; Hong, S.H.; Jang, M.; Han, G.M.; Jung, S.W.; Shim, W.J. Combined Effects of UV Exposure Duration and Mechanical Abrasion on Microplastic Fragmentation by Polymer Type. *Environ. Sci. Technol.* **2017**, *51*, 4368–4376. [CrossRef]
15. Da Costa, J.P. Micro- and nanoplastics in the environment: Research and policymaking. *Curr. Opin. Environ. Sci. Health* **2018**, *1*, 12–16. [CrossRef]
16. Wagner, M.; Scherer, C.; Alvarez-Muñoz, D.; Brennholt, N.; Bourrain, X.; Buchinger, S.; Fries, E.; Grosbois, C.; Klasmeier, J.; Marti, T.; et al. Microplastics in freshwater ecosystems: What we know and what we need to know. *Environ. Sci. Eur.* **2014**, *26*, 12. [CrossRef] [PubMed]
17. Coyle, R.; Hardiman, G.; Driscoll, K.O. Microplastics in the marine environment: A review of their sources, distribution processes, uptake and exchange in ecosystems. *Case Stud. Chem. Environ. Eng.* **2020**, *2*, 100010. [CrossRef]
18. Frank, Y.; Ershova, A.; Batasheva, S.; Vorobiev, E.; Rakhmatullina, S.; Vorobiev, D.; Fakhrullin, R. Microplastics in Freshwater: A Focus on the Russian Inland Waters. *Water* **2022**, *14*, 3909. [CrossRef]
19. Liu, C.; Yang, J.; Ji, S.; Lu, Y.; Wu, P.; Chen, C. Influence of natural ventilation rate on indoor PM2.5 deposition. *Build. Environ.* **2018**, *144*, 357–364. [CrossRef]
20. 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. [CrossRef]
21. Koutnik, V.S.; Leonard, J.; Glasman, J.B.; Brar, J.; Koydemir, H.C.; Novoselov, A.; Bertel, R.; Tseng, D.; Ozcan, A.; Ravi, S.; et al. Microplastics retained in stormwater control measures: Where do they come from and where do they go? *Water Res.* **2022**, *210*, 118008. [CrossRef]
22. Thiele, C.J.; Hudson, M.D.; Russell, A.E.; Saluveer, M.; Sidaoui-Haddad, G. Microplastics in fish and fishmeal: An emerging environmental challenge? *Sci. Rep.* **2021**, *11*, 2045. [CrossRef]
23. Ugwu, K.; Herrera, A.; Gómez, M. Microplastics in marine biota: A review. *Mar. Pollut. Bull.* **2021**, *169*, 112540. [CrossRef]
24. Wang, L.; Nabi, G.; Yin, L.; Wang, Y.; Li, S.; Hao, Z.; Li, D. Birds and plastic pollution: Recent advances. *Avian Res.* **2021**, *12*, 59. [CrossRef]
25. Baeza-Martínez, C.; Olmos, S.; González-Pleiter, M.; López-Castellanos, J.; García-Pachón, E.; Masiá-Canuto, M.; Hernández-Blasco, L.; Bayo, J. First evidence of microplastics isolated in European citizens' lower airway. *J. Hazard. Mater.* **2022**, *438*, 129439. [CrossRef] [PubMed]
26. 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. [CrossRef]
27. Auta, H.S.; Emenike, C.U.; Fauziah, S.H. Distribution and importance of microplastics in the marine environment: A review of the sources, fate, effects, and potential solutions. *Environ. Int.* **2017**, *102*, 165–176. [CrossRef]
28. Patel, M.M.; Goyal, B.R.; Bhadada, S.V.; Bhatt, J.S.; Amin, A.F. Getting into the Brain: Approaches to Enhance Brain Drug Delivery. *CNS Drugs* **2009**, *23*, 35–58. [CrossRef]
29. Hopewell, J.; Dvorak, R.; Kosior, E. Plastics recycling: Challenges and opportunities. *Philos. Trans. R. Soc. B Biol. Sci.* **2009**, *364*, 2115–2126. [CrossRef]
30. Fendall, L.S.; Sewell, M.A. Contributing to marine pollution by washing your face: Microplastics in facial cleansers. *Mar. Pollut. Bull.* **2009**, *58*, 1225–1228. [CrossRef]
31. Cole, M.; Lindeque, P.K.; Fileman, E.; Clark, J.; Lewis, C.; Halsband, C.; Galloway, T.S. Microplastics Alter the Properties and Sinking Rates of Zooplankton Faecal Pellets. *Environ. Sci. Technol.* **2016**, *50*, 3239–3246. [CrossRef]
32. Castañeda, R.A.; Avlijas, S.; Simard, M.A.; Ricciardi, A. Microplastic pollution in St. Lawrence River sediments. *Can. J. Fish. Aquat. Sci.* **2014**, *71*, 1767–1771. [CrossRef]
33. Liu, P.; Shao, L.; Li, Y.; Jones, T.; Cao, Y.; Yang, C.-X.; Zhang, M.; Santosh, M.; Feng, X.; BéruBé, K. Microplastic atmospheric dustfall pollution in urban environment: Evidence from the types, distribution, and probable sources in Beijing, China. *Sci. Total Environ.* **2022**, *838*, 155989. [CrossRef]
34. 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. [CrossRef]
35. Brahney, J.; Hallerud, M.; Heim, E.; Hahnenberger, M.; Sukumaran, S. Plastic rain in protected areas of the United States. *Science* **2020**, *368*, 1257–1260. [CrossRef]
36. Priya, A.K.; Jalil, A.A.; Dutta, K.; Rajendran, S.; Vasseghian, Y.; Qin, J.; Soto-Moscoso, M. Microplastics in the environment: Recent developments in characteristic, occurrence, identification and ecological risk. *Chemosphere* **2022**, *298*, 134161. [CrossRef] [PubMed]
37. Akanyange, S.N.; Lyu, X.; Zhao, X.; Li, X.; Zhang, Y.; Crittenden, J.C.; Anning, C.; Chen, T.; Jiang, T.; Zhao, H. Does microplastic really represent a threat? A review of the atmospheric contamination sources and potential impacts. *Sci. Total Environ.* **2021**, *777*, 146020. [CrossRef] [PubMed]
38. 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. [CrossRef] [PubMed]
39. Oehlmann, J.; Schulte-Oehlmann, U.; Kloas, W.; Jagnytsch, O.; Lutz, I.; Kusk, K.O.; Wollenberger, L.; Santos, E.M.; Paull, G.C.; Van Look, K.J.W.; et al. A critical analysis of the biological impacts of plasticizers on wildlife. *Philos. Trans. R. Soc. B Biol. Sci.* **2009**, *364*, 2047–2062. [CrossRef]
40. Huang, Y.; He, T.; Yan, M.; Yang, L.; Gong, H.; Wang, W.; Qing, X.; Wang, J. Atmospheric transport and deposition of microplastics in a subtropical urban environment. *J. Hazard. Mater.* **2021**, *416*, 126168. [CrossRef]

41. Nigamatzyanova, L.; Fakhrullin, R. Dark-field hyperspectral microscopy for label-free microplastics and nanoplastics detection and identification in vivo: A Caenorhabditis elegans study. *Environ. Pollut.* **2021**, *271*, 116337. [CrossRef]
42. Akhatova, F.; Ishmukhametov, I.; Fakhrullina, G.; Fakhrullin, R. Nanomechanical Atomic Force Microscopy to Probe Cellular Microplastics Uptake and Distribution. *Int. J. Mol. Sci.* **2022**, *23*, 806. [CrossRef]
43. Mohammad, R.A.; Qusay, A.-A. Eco-friendly microplastic removal through physical and chemical techniques: A review. *Ann. Adv. Chem.* **2023**, *7*, 0014-0024. [CrossRef]
44. Li, Y.; Lu, Q.; Xing, Y.; Liu, K.; Ling, W.; Yang, J.; Yang, Q.; Wu, T.; Zhang, J.; Pei, Z.; et al. Review of research on migration, distribution, biological effects, and analytical methods of microfibers in the environment. *Sci. Total Environ.* **2023**, *855*, 158922. [CrossRef]
45. Efremenko, E.N.; Lyagin, I.V.; Maslova, O.V.; Senko, O.V.; Stepanov, N.A.; Aslanli, A.G.G. Catalytic degradation of microplastics. *Russ. Chem. Rev.* **2023**, *92*, RCR5069. [CrossRef]
46. Alfonso, M.B.; Arias, A.H.; Ronda, A.C.; Piccolo, M.C. Continental microplastics: Presence, features, and environmental transport pathways. *Sci. Total Environ.* **2021**, *799*, 149447. [CrossRef] [PubMed]
47. Ramkumar, M.; Balasubramani, K.; Santosh, M.; Nagarajan, R. The plastisphere: A morphometric genetic classification of plastic pollutants in the natural environment. *Gondwana Res.* **2022**, *108*, 4–12. [CrossRef]
48. Xu, A.; Shi, M.; Xing, X.; Su, Y.; Li, X.; Liu, W.; Mao, Y.; Hu, T.; Qi, S. Status and prospects of atmospheric microplastics: A review of methods, occurrence, composition, source and health risks. *Environ. Pollut.* **2022**, *303*, 119173. [CrossRef] [PubMed]
49. Corcoran, P. OBSOLETE: Plastics and the Anthropocene. In *Reference Module in Earth Systems and Environmental Sciences*; Elsevier: Amsterdam, The Netherlands, 2018; p. B9780124095489100004. ISBN 978-0-12-409548-9.
50. Batool, I.; Qadir, A.; Levermore, J.M.; Kelly, F.J. Dynamics of airborne microplastics, appraisal and distributional behaviour in atmosphere; a review. *Sci. Total Environ.* **2022**, *806*, 150745. [CrossRef]
51. Hidalgo-Ruz, V.; Gutow, L.; Thompson, R.C.; Thiel, M. Microplastics in the Marine Environment: A Review of the Methods Used for Identification and Quantification. *Environ. Sci. Technol.* **2012**, *46*, 3060–3075. [CrossRef]
52. Guo, X.; Wang, J. The chemical behaviors of microplastics in marine environment: A review. *Mar. Pollut. Bull.* **2019**, *142*, 1–14. [CrossRef]
53. O'Connor, J.D.; Mahon, A.M.; Ramsperger, A.F.R.M.; Trotter, B.; Redondo-Hasselerharm, P.E.; Koelmans, A.A.; Lally, H.T.; Murphy, S. Microplastics in Freshwater Biota: A Critical Review of Isolation, Characterization, and Assessment Methods. *Glob. Chall.* **2020**, *4*, 1800118. [CrossRef]
54. Liu, K.; Wu, T.; Wang, X.; Song, Z.; Zong, C.; Wei, N.; Li, D. Consistent Transport of Terrestrial Microplastics to the Ocean through Atmosphere. *Environ. Sci. Technol.* **2019**, *53*, 10612–10619. [CrossRef]
55. Ding, Y.; Zou, X.; Wang, C.; Feng, Z.; Wang, Y.; Fan, Q.; Chen, H. The abundance and characteristics of atmospheric microplastic deposition in the northwestern South China Sea in the fall. *Atmos. Environ.* **2021**, *253*, 118389. [CrossRef]
56. Wang, Z.; Dong, H.; Wang, Y.; Ren, R.; Qin, X.; Wang, S. Effects of microplastics and their adsorption of cadmium as vectors on the cladoceran Moina monogolica Daday: Implications for plastic-ingesting organisms. *J. Hazard. Mater.* **2020**, *400*, 123239. [CrossRef] [PubMed]
57. Amato-Lourenço, L.F.; Dos Santos Galvão, L.; De Weger, L.A.; Hiemstra, P.S.; Vijver, M.G.; Mauad, T. An emerging class of air pollutants: Potential effects of microplastics to respiratory human health? *Sci. Total Environ.* **2020**, *749*, 141676. [CrossRef] [PubMed]
58. Prata, J.C. Airborne microplastics: Consequences to human health? *Environ. Pollut.* **2018**, *234*, 115–126. [CrossRef]
59. Wright, S.L.; Kelly, F.J. Plastic and Human Health: A Micro Issue? *Environ. Sci. Technol.* **2017**, *51*, 6634–6647. [CrossRef]
60. Kane, I.A.; Clare, M.A.; Miramontes, E.; Wogelius, R.; Rothwell, J.J.; Garreau, P.; Pohl, F. Seafloor microplastic hotspots controlled by deep-sea circulation. *Science* **2020**, *368*, 1140–1145. [CrossRef]
61. Pohl, F.; Eggenhuisen, J.T.; Kane, I.A.; Clare, M.A. Transport and Burial of Microplastics in Deep-Marine Sediments by Turbidity Currents. *Environ. Sci. Technol.* **2020**, *54*, 4180–4189. [CrossRef]
62. Torres-Agullo, A.; Karanasiou, A.; Moreno, T.; Lacorte, S. Airborne microplastic particle concentrations and characterization in indoor urban microenvironments. *Environ. Pollut.* **2022**, *308*, 119707. [CrossRef]
63. Luo, Y.; Shi, H.; Tu, C.; Zhou, Q.; Ji, R.; Pan, X.; Xu, X.; Wu, C.; An, L.; Sun, X.; et al. Research progresses and prospects of microplasticsin the environment. *Chin. Sci. Bull.* **2021**, *66*, 1547–1562. [CrossRef]
64. Knobloch, E.; Ruffell, H.; Aves, A.; Pantos, O.; Gaw, S.; Revell, L.E. Comparison of Deposition Sampling Methods to Collect Airborne Microplastics in Christchurch, New Zealand. *Water Air. Soil Pollut.* **2021**, *232*, 133. [CrossRef]
65. Zhang, Y.; Kang, S.; Allen, S.; Allen, D.; Gao, T.; Sillanpää, M. Atmospheric microplastics: A review on current status and perspectives. *Earth-Sci. Rev.* **2020**, *203*, 103118. [CrossRef]
66. Chen, C.-Y.; Lu, T.-H.; Wang, W.-M.; Liao, C.-M. Assessing regional emissions of vehicle-based tire wear particle from macro- to micro/nano-scales with pandemic lockdowns and electromobility scenarios implications. *Chemosphere* **2023**, *311*, 137209. [CrossRef] [PubMed]
67. Huang, D.; Chen, H.; Shen, M.; Tao, J.; Chen, S.; Yin, L.; Zhou, W.; Wang, X.; Xiao, R.; Li, R. Recent advances on the transport of microplastics/nanoplastics in abiotic and biotic compartments. *J. Hazard. Mater.* **2022**, *438*, 129515. [CrossRef]
68. Ganguly, M.; Ariya, P.A. Ice Nucleation of Model Nanoplastics and Microplastics: A Novel Synthetic Protocol and the Influence of Particle Capping at Diverse Atmospheric Environments. *ACS Earth Space Chem.* **2019**, *3*, 1729–1739. [CrossRef]

69. Enyoh, C.E.; Verla, A.W.; Verla, E.N.; Ibe, F.C.; Amaobi, C.E. Airborne microplastics: A review study on method for analysis, occurrence, movement and risks. *Environ. Monit. Assess.* **2019**, *191*, 668. [CrossRef]
70. Chen, G.; Feng, Q.; Wang, J. Mini-review of microplastics in the atmosphere and their risks to humans. *Sci. Total Environ.* **2020**, *703*, 135504. [CrossRef]
71. Rochman, C.M. The Complex Mixture, Fate and Toxicity of Chemicals Associated with Plastic Debris in the Marine Environment. In *Marine Anthropogenic Litter*; Bergmann, M., Gutow, L., Klages, M., Eds.; Springer International Publishing: Cham, Switzerland, 2015; pp. 117–140, ISBN 978-3-319-16509-7.
72. Rochman, C.M.; Brookson, C.; Bikker, J.; Djuric, N.; Earn, A.; Bucci, K.; Athey, S.; Huntington, A.; McIlwraith, H.; Munno, K.; et al. Rethinking microplastics as a diverse contaminant suite. *Environ. Toxicol. Chem.* **2019**, *38*, 703–711. [CrossRef]
73. Pannetier, P.; Morin, B.; Le Bihanic, F.; Dubreil, L.; Clérandeau, C.; Chouvellon, F.; Van Arkel, K.; Danion, M.; Cachot, J. Environmental samples of microplastics induce significant toxic effects in fish larvae. *Environ. Int.* **2020**, *134*, 105047. [CrossRef]
74. Padha, S.; Kumar, R.; Dhar, A.; Sharma, P. Microplastic pollution in mountain terrains and foothills: A review on source, extraction, and distribution of microplastics in remote areas. *Environ. Res.* **2022**, *207*, 112232. [CrossRef]
75. Mintenig, S.M.; Bäuerlein, P.S.; Koelmans, A.A.; Dekker, S.C.; Van Wezel, A.P. Closing the gap between small and smaller: Towards a framework to analyse nano- and microplastics in aqueous environmental samples. *Environ. Sci. Nano* **2018**, *5*, 1640–1649. [CrossRef]
76. Schwaferts, C.; Niessner, R.; Elsner, M.; Ivleva, N.P. Methods for the analysis of submicrometer- and nanoplastic particles in the environment. *TrAC Trends Anal. Chem.* **2019**, *112*, 52–65. [CrossRef]
77. Jahandari, A. Microplastics in the Urban Atmosphere: Sources, Occurrences, Distribution, and Potential Health Implications. *J. Hazard. Mater. Adv.* **2023**, *12*, 100346. [CrossRef]
78. Chen, G.; Fu, Z.; Yang, H.; Wang, J. An overview of analytical methods for detecting microplastics in the atmosphere. *TrAC Trends Anal. Chem.* **2020**, *130*, 115981. [CrossRef]
79. Frias, J.P.G.L.; Nash, R. Microplastics: Finding a consensus on the definition. *Mar. Pollut. Bull.* **2019**, *138*, 145–147. [CrossRef] [PubMed]
80. Zhao, X.; Liu, Z.; Cai, L.; Han, J. Occurrence and distribution of microplastics in surface sediments of a typical river with a highly eroded catchment, a case of the Yan River, a tributary of the Yellow River. *Sci. Total Environ.* **2023**, *863*, 160932. [CrossRef]
81. Yang, T.; Gao, M.; Nowack, B. Formation of microplastic fibers and fibrils during abrasion of a representative set of 12 polyester textiles. *Sci. Total Environ.* **2023**, *862*, 160758. [CrossRef]
82. Kuroda, M.; Isobe, A.; Uchida, K.; Tokai, T.; Kitakado, T.; Yoshitake, M.; Miyamoto, Y.; Mukai, T.; Imai, K.; Shimizu, K.; et al. Abundance and potential sources of floating polystyrene foam macro- and microplastics around Japan. *Sci. Total Environ.* **2024**, *925*, 171421. [CrossRef]
83. Soliz, D.L.; Paniagua González, G.; Muñoz-Arnanz, J.; Bravo-Yagüe, J.C.; Fernández Hernando, P.; Garcinuño Martínez, R.M. Identification and morphological characterization of different types of plastic microparticles. *Heliyon* **2024**, *10*, e30749. [CrossRef]
84. Syafina, P.R.; Yudison, A.P.; Sembiring, E.; Irsyad, M.; Tomo, H.S. Identification of fibrous suspended atmospheric microplastics in Bandung Metropolitan Area, Indonesia. *Chemosphere* **2022**, *308*, 136194. [CrossRef]
85. 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. [CrossRef]
86. Jarosz, K.; Janus, R.; Wądrzyk, M.; Wilczyńska-Michalik, W.; Natkański, P.; Michalik, M. Airborne Microplastic in the Atmospheric Deposition and How to Identify and Quantify the Threat: Semi-Quantitative Approach Based on Kraków Case Study. *Int. J. Environ. Res. Public Health* **2022**, *19*, 12252. [CrossRef]
87. 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**, *14*, 54–57.
88. Amato-Lourenço, L.F.; Dos Santos Galvão, L.; Wiebeck, H.; Carvalho-Oliveira, R.; Mauad, T. Atmospheric microplastic fallout in outdoor and indoor environments in São Paulo megacity. *Sci. Total Environ.* **2022**, *821*, 153450. [CrossRef] [PubMed]
89. Cabrera, M.; Valencia, B.G.; Lucas-Solis, O.; Calero, J.L.; Maisincho, L.; Conicelli, B.; Massaine Moulatlet, G.; Capparelli, M.V. A new method for microplastic sampling and isolation in mountain glaciers: A case study of one antisana glacier, Ecuadorian Andes. *Case Stud. Chem. Environ. Eng.* **2020**, *2*, 100051. [CrossRef]
90. Materić, D.; Ludewig, E.; Brunner, D.; Röckmann, T.; Holzinger, R. Nanoplastics transport to the remote, high-altitude Alps. *Environ. Pollut.* **2021**, *288*, 117697. [CrossRef]
91. Aves, A.R.; Revell, L.E.; Gaw, S.; Ruffell, H.; Schuddeboom, A.; Wotherspoon, N.E.; LaRue, M.; McDonald, A.J. First evidence of microplastics in Antarctic snow. *Cryosphere* **2022**, *16*, 2127–2145. [CrossRef]
92. Mohamed Nor, N.H.; Obbard, J.P. Microplastics in Singapore's coastal mangrove ecosystems. *Mar. Pollut. Bull.* **2014**, *79*, 278–283. [CrossRef]
93. Di, M.; Liu, X.; Wang, W.; Wang, J. Manuscript prepared for submission to environmental toxicology and pharmacology pollution in drinking water source areas: Microplastics in the Danjiangkou Reservoir, China. *Environ. Toxicol. Pharmacol.* **2019**, *65*, 82–89. [CrossRef]
94. Jia, Q.; Duan, Y.; Han, X.; Sun, X.; Munyaneza, J.; Ma, J.; Xiu, G. Atmospheric deposition of microplastics in the megalopolis (Shanghai) during rainy season: Characteristics, influence factors, and source. *Sci. Total Environ.* **2022**, *847*, 157609. [CrossRef]

95. Song, X.-C.; Lin, Q.-B.; Zhang, Y.-C.; Li, Z.; Zeng, Y.; Chen, Z.-F. Rapid classification of virgin and recycled EPS containers by Fourier transform infrared spectroscopy and chemometrics. *Food Addit. Contam. Part A* **2018**, *35*, 2220–2229. [CrossRef]
96. Jahnke, A.; Arp, H.P.H.; Escher, B.I.; Gewert, B.; Gorokhova, E.; Kühnel, D.; Ogonowski, M.; Potthoff, A.; Rummel, C.; Schmitt-Jansen, M.; et al. Reducing Uncertainty and Confronting Ignorance about the Possible Impacts of Weathering Plastic in the Marine Environment. *Environ. Sci. Technol. Lett.* **2017**, *4*, 85–90. [CrossRef]
97. Shruti, V.C.; Kutralam-Muniasamy, G.; Pérez-Guevara, F.; Roy, P.D.; Martínez, I.E. Occurrence and characteristics of atmospheric microplastics in Mexico City. *Sci. Total Environ.* **2022**, *847*, 157601. [CrossRef] [PubMed]
98. Tse, K.C.C.; Ng, F.M.F.; Yu, K.N. Photo-Degradation of PADC by UV Radiation at Various Wavelengths. *Polym. Degrad. Stab.* **2006**, *91*, 2380–2388. [CrossRef]
99. Mao, R.; Lang, M.; Yu, X.; Wu, R.; Yang, X.; Guo, X. Aging mechanism of microplastics with UV irradiation and its effects on the adsorption of heavy metals. *J. Hazard. Mater.* **2020**, *393*, 122515. [CrossRef] [PubMed]
100. Williamson, C.E.; Neale, P.J.; Hylander, S.; Rose, K.C.; Figueroa, F.L.; Robinson, S.A.; Häder, D.-P.; Wä, S.-Å.; Worrest, R.C. The interactive effects of stratospheric ozone depletion, UV radiation, and climate change on aquatic ecosystems. *Photochem. Photobiol. Sci.* **2019**, *18*, 717–746. [CrossRef]
101. Stefánsson, H.; Peternell, M.; Konrad-Schmolke, M.; Hannesdóttir, H.; Ásbjörnsson, E.J.; Sturkell, E. Microplastics in Glaciers: First Results from the Vatnajökull Ice Cap. *Sustainability* **2021**, *13*, 4183. [CrossRef]
102. Liu, S.; Junaid, M.; Liao, H.; Liu, X.; Wu, Y.; Wang, J. Eco-corona formation and associated ecotoxicological impacts of nanoplastics in the environment. *Sci. Total Environ.* **2022**, *836*, 155703. [CrossRef]
103. Abbasi, S.; Alirezazadeh, M.; Razeghi, N.; Rezaei, M.; Pourmahmood, H.; Dehbandi, R.; Mehr, M.R.; Ashayeri, S.Y.; Oleszczuk, P.; Turner, A. Microplastics captured by snowfall: A study in Northern Iran. *Sci. Total Environ.* **2022**, *822*, 153451. [CrossRef]
104. Ding, J.; Sun, C.; He, C.; Zheng, L.; Dai, D.; Li, F. Atmospheric microplastics in the Northwestern Pacific Ocean: Distribution, source, and deposition. *Sci. Total Environ.* **2022**, *829*, 154337. [CrossRef]
105. Rocha-Santos, T.; Duarte, A.C. (Eds.) *Characterization and Analysis of Microplastics*; Comprehensive Analytical Chemistry; Elsevier: Amsterdam, The Netherlands; Oxford, UK; Cambridge, MA, USA, 2017; ISBN 978-0-444-63898-4.
106. Preston, J.; Man-made fibre. Encycl. Br. 2016. Available online: https://www.britannica.com/technology/man-made-fiber (accessed on 31 January 2024).
107. Allen, S.; Allen, D.; Phoenix, V.R.; Le Roux, G.; Durántez Jiménez, P.; Simonneau, A.; Binet, S.; Galop, D. Atmospheric transport and deposition of microplastics in a remote mountain catchment. *Nat. Geosci.* **2019**, *12*, 339–344. [CrossRef]
108. Andrady, A.L. The plastic in microplastics: A review. *Mar. Pollut. Bull.* **2017**, *119*, 12–22. [CrossRef]
109. Zambrano, M.C.; Pawlak, J.J.; Daystar, J.; Ankeny, M.; Cheng, J.J.; Venditti, R.A. Microfibers generated from the laundering of cotton, rayon and polyester based fabrics and their aquatic biodegradation. *Mar. Pollut. Bull.* **2019**, *142*, 394–407. [CrossRef] [PubMed]
110. Chamas, A.; Moon, H.; Zheng, J.; Qiu, Y.; Tabassum, T.; Jang, J.H.; Abu-Omar, M.; Scott, S.L.; Suh, S. Degradation Rates of Plastics in the Environment. *ACS Sustain. Chem. Eng.* **2020**, *8*, 3494–3511. [CrossRef]
111. Ji, L.N. Study on Preparation Process and Properties of Polyethylene Terephthalate (PET). *Appl. Mech. Mater.* **2013**, *312*, 406–410. [CrossRef]
112. Kasal, A.; Kuşkun, T.; Smardzewski, J. Experimental and Numerical Study on Withdrawal Strength of Different Types of Auxetic Dowels for Furniture Joints. *Materials* **2020**, *13*, 4252. [CrossRef] [PubMed]
113. Welle, F.; Bayer, F.; Franz, R. Quantification of the Sorption Behavior of Polyethylene Terephthalate Polymer versus PET/PA Polymer Blends towards Organic Compounds. *Packag. Technol. Sci.* **2012**, *25*, 341–349. [CrossRef]
114. Bergmann, M.; Mützel, 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*, eaax1157. [CrossRef]
115. Polyacrylonitrile. *Polym. Sci. Learn. Cent.* 2005. Available online: https://pslc.ws/macrog/pan.htm (accessed on 31 January 2024).
116. Wang, C.; Zhao, J.; Xing, B. Environmental source, fate, and toxicity of microplastics. *J. Hazard. Mater.* **2021**, *407*, 124357. [CrossRef]
117. Abbasi, S.; Keshavarzi, B.; Moore, F.; Delshab, H.; Soltani, N.; Sorooshian, A. Investigation of microrubbers, microplastics and heavy metals in street dust: A study in Bushehr city, Iran. *Environ. Earth Sci.* **2017**, *76*, 798. [CrossRef]
118. 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. [CrossRef]
119. Dehghani, S.; Moore, F.; Akhbarizadeh, R. Microplastic pollution in deposited urban dust, Tehran metropolis, Iran. *Environ. Sci. Pollut. Res.* **2017**, *24*, 20360–20371. [CrossRef]
120. 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. [CrossRef]
121. Carr, S.A.; Liu, J.; Tesoro, A.G. Transport and fate of microplastic particles in wastewater treatment plants. *Water Res.* **2016**, *91*, 174–182. [CrossRef] [PubMed]
122. Nizzetto, L.; Langaas, S.; Futter, M. Pollution: Do microplastics spill on to farm soils? *Nature* **2016**, *537*, 488. [CrossRef] [PubMed]
123. Bayo, J.; Olmos, S.; López-Castellanos, J.; Alcolea, A. Microplastics and microfibers in the sludge of a municipal wastewater treatment plant. *Int. J. Sustain. Dev. Plan.* **2016**, *11*, 812–821. [CrossRef]

124. Li, X.; Chen, L.; Mei, Q.; Dong, B.; Dai, X.; Ding, G.; Zeng, E.Y. Microplastics in sewage sludge from the wastewater treatment plants in China. *Water Res.* **2018**, *142*, 75–85. [CrossRef]
125. Magni, S.; Binelli, A.; Pittura, L.; Avio, C.G.; Della Torre, C.; Parenti, C.C.; Gorbi, S.; Regoli, F. The fate of microplastics in an Italian Wastewater Treatment Plant. *Sci. Total Environ.* **2019**, *652*, 602–610. [CrossRef]
126. Ekebafe, L.; Ogbeifun, D.E.; Okieimen, F. Polymer Applications in Agriculture. *Biokemistri* **2011**, *23*, 81–89.
127. Changrong, Y.; Wenqing, H.; Neil, C. Plastic-film mulch in Chinese agriculture: Importance and problems. *World Agric.* **2014**, *4*, 32–36. Available online: https://www.world-agriculture.net/article/plastic-film-mulch-in-chinese-agriculture-importance-and-problems (accessed on 12 November 2024).
128. Liu, M.; Lu, S.; Song, Y.; Lei, L.; Hu, J.; Lv, W.; Zhou, W.; Cao, C.; Shi, H.; Yang, X.; et al. Microplastic and mesoplastic pollution in farmland soils in suburbs of Shanghai, China. *Environ. Pollut.* **2018**, *242*, 855–862. [CrossRef]
129. Rødland, E.S.; Lind, O.C.; Reid, M.J.; Heier, L.S.; Okoffo, E.D.; Rauert, C.; Thomas, K.V.; Meland, S. Occurrence of tire and road wear particles in urban and peri-urban snowbanks, and their potential environmental implications. *Sci. Total Environ.* **2022**, *824*, 153785. [CrossRef]
130. Evangeliou, N.; Grythe, H.; Klimont, Z.; Heyes, C.; Eckhardt, S.; Lopez-Aparicio, S.; Stohl, A. Atmospheric transport is a major pathway of microplastics to remote regions. *Nat. Commun.* **2020**, *11*, 3381. [CrossRef]
131. Rhodes, C.J. Plastic Pollution and Potential Solutions. *Sci. Prog.* **2018**, *101*, 207–260. [CrossRef] [PubMed]
132. Zhang, Q.; Wang, R.; Shen, Y.; Zhan, L.; Xu, Z. Characteristics of unorganized emissions of microplastics from road fugitive dust in urban mining bases. *Sci. Total Environ.* **2022**, *827*, 154355. [CrossRef] [PubMed]
133. Zheng, K.; Wang, P.; Lou, X.; Zhou, Z.; Zhou, L.; Hu, Y.; Luan, Y.; Quan, C.; Fang, J.; Zou, H.; et al. A review of airborne micro- and nano-plastics: Sampling methods, analytical techniques, and exposure risks. *Environ Pollut.* **2024**, *363*, 125074. [CrossRef]
134. Fox, S.; Stefánsson, H.; Peternell, M.; Zlotskiy, E.; Ásbjörnsson, E.J.; Sturkell, E.; Wanner, P.; Konrad-Schmolke, M. Physical characteristics of microplastic particles and potential for global atmospheric transport: A meta-analysis. *Environ. Pollut.* **2024**, *342*, 122938. [CrossRef]
135. Nafea, T.H.; Chan, F.K.S.; Xu, Y.; Wang, C.; Wang, X.; Zhao, W.; Ji, D.; Xiao, H.; He, J. Microplastics Aloft: A comprehensive exploration of sources, transport, variations, interactions and their implications on human health in the atmospheric realm. *Earth-Sci. Rev.* **2024**, *255*, 104864. [CrossRef]
136. Wang, Z.; Zhang, Y.; Kang, S.; Yang, L.; Luo, X.; Chen, P.; Guo, J.; Hu, Z.; Yang, C.; Yang, Z.; et al. Long-range transport of atmospheric microplastics deposited onto glacier in southeast Tibetan Plateau. *Environ. Pollut.* **2022**, *306*, 119415. [CrossRef]
137. Rochman, C.M.; Hoellein, T. The global odyssey of plastic pollution. *Science* **2020**, *368*, 1184–1185. [CrossRef]
138. Shao, L.; Li, Y.; Jones, T.; Santosh, M.; Liu, P.; Zhang, M.; Xu, L.; Li, W.; Lu, J.; Yang, C.-X.; et al. Airborne microplastics: A review of current perspectives and environmental implications. *J. Clean. Prod.* **2022**, *347*, 131048. [CrossRef]
139. Trainic, M.; Flores, J.M.; Pinkas, I.; Pedrotti, M.L.; Lombard, F.; Bourdin, G.; Gorsky, G.; Boss, E.; Rudich, Y.; Vardi, A.; et al. Airborne microplastic particles detected in the remote marine atmosphere. *Commun. Earth Environ.* **2020**, *1*, 64. [CrossRef]
140. ECETOC. *Linear Polydimethylsiloxanes CAS No. 63148-62-9*, 2nd ed.; ECETOC: Brussels, Belgium, 2011.
141. Brahney, J.; Mahowald, N.; Prank, M.; Cornwell, G.; Klimont, Z.; Matsui, H.; Prather, K.A. Constraining the atmospheric limb of the plastic cycle. *Proc. Natl. Acad. Sci. USA* **2021**, *118*, e2020719118. [CrossRef]
142. Zhao, X.; Zhou, Y.; Liang, C.; Song, J.; Yu, S.; Liao, G.; Zou, P.; Tang, K.H.D.; Wu, C. Airborne microplastics: Occurrence, sources, fate, risks and mitigation. *Sci. Total Environ.* **2023**, *858*, 159943. [CrossRef] [PubMed]
143. Allen, S.; Allen, D.; Moss, K.; Le Roux, G.; Phoenix, V.R.; Sonke, J.E. Examination of the ocean as a source for atmospheric microplastics. *PLoS ONE* **2020**, *15*, e0232746. [CrossRef] [PubMed]
144. Abbasi, S.; Turner, A. Dry and wet deposition of microplastics in a semi-arid region (Shiraz, Iran). *Sci. Total Environ.* **2021**, *786*, 147358. [CrossRef]
145. Catarino, A.I.; León, M.C.; Li, Y.; Lambert, S.; Vercauteren, M.; Asselman, J.; Janssen, C.R.; Everaert, G.; De Rijcke, M. Micro- and nanoplastics transfer from seawater to the atmosphere through aerosolization under controlled laboratory conditions. *Mar. Pollut. Bull.* **2023**, *192*, 115015. [CrossRef]
146. Yang, S.; Lu, X.; Wang, X. A Perspective on the Controversy over Global Emission Fluxes of Microplastics from Ocean into the Atmosphere. *Environ. Sci. Technol.* **2024**, *58*, 12304–12312. [CrossRef]
147. Lambert, S.; Vercauteren, M.; Catarino, A.I.; Li, Y.; Van Landuyt, J.; Boon, N.; Everaert, G.; De Rijcke, M.; Janssen, C.R.; Asselman, J. Aerosolization of micro- and nanoplastics via sea spray: Investigating the role of polymer type, size, and concentration, and potential implications for human exposure. *Environ. Pollut.* **2024**, *351*, 124105. [CrossRef]
148. Tunahan Kaya, A.; Yurtsever, M.; Çiftçi Bayraktar, S. Ubiquitous exposure to microfiber pollution in the air. *Eur. Phys. J. Plus* **2018**, *133*, 488. [CrossRef]
149. Zhao, S.; Yu, Y.; He, J.; Yin, D.; Wang, B. Below-cloud scavenging of aerosol particles by precipitation in a typical valley city, northwestern China. *Atmos. Environ.* **2015**, *102*, 70–78. [CrossRef]
150. Lai, Y.; Dong, L.; Li, Q.; Li, P.; Liu, J. Sampling of micro- and nano-plastics in environmental matrixes. *TrAC Trends Anal. Chem.* **2021**, *145*, 116461. [CrossRef]
151. 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. [CrossRef]

152. Soltani, N.S.; Taylor, M.P.; Wilson, S.P. Quantification and exposure assessment of microplastics in Australian indoor house dust. *Environ. Pollut.* **2021**, *283*, 117064. [CrossRef] [PubMed]
153. Zhang, Q.; Zhao, Y.; Du, F.; Cai, H.; Wang, G.; Shi, H. Microplastic Fallout in Different Indoor Environments. *Environ. Sci. Technol.* **2020**, *54*, 6530–6539. [CrossRef]
154. Zhang, Q.; Du, F.; Liang, W.; Chen, Q.; Meng, J.; Shi, H. Microfiber fallout during dining and potential human intake. *J. Hazard. Mater.* **2022**, *430*, 128477. [CrossRef] [PubMed]
155. Zhou, Q.; Tian, C.; Luo, Y. Various forms and deposition fluxes of microplastics identified in the coastal urban atmosphere. *Chin. Sci. Bull.* **2017**, *62*, 3902–3909. [CrossRef]
156. Napper, I.E.; Parker-Jurd, F.N.F.; Wright, S.L.; Thompson, R.C. Examining the release of synthetic microfibres to the environment via two major pathways: Atmospheric deposition and treated wastewater effluent. *Sci. Total Environ.* **2023**, *857*, 159317. [CrossRef]
157. Grieve, M.C.; Biermann, T. The population of coloured textile fibres on outdoor surfaces. *Sci. Justice* **1997**, *37*, 231–239. [CrossRef]
158. Yuan, Z.; Li, H.-X.; Lin, L.; Pan, Y.-F.; Liu, S.; Hou, R.; Xu, X.-R. Occurrence and human exposure risks of atmospheric microplastics: A review. *Gondwana Res.* **2022**, *108*, 200–212. [CrossRef]
159. Page, M.J.; McKenzie, J.E.; Bossuyt, P.M.; Boutron, I.; Hoffmann, T.C.; Mulrow, C.D.; Shamseer, L.; Tetzlaff, J.M.; Akl, E.A.; Brennan, S.E.; et al. The PRISMA 2020 statement: An updated guideline for reporting systematic reviews. *BMJ* **2021**, *372*, n71. [CrossRef]
160. Caracci, E.; Vega-Herrera, A.; Dachs, J.; Berrojalbiz, N.; Buonanno, G.; Abad, E.; Llorca, M.; Moreno, T.; Farré, M. Micro(nano)plastics in the atmosphere of the Atlantic Ocean. *J. Hazard. Mater.* **2023**, *450*, 131036. [CrossRef]
161. González-Pleiter, M.; Lacerot, G.; Edo, C.; Pablo Lozoya, J.; Leganés, F.; Fernández-Piñas, F.; Rosal, R.; Teixeira-de-Mello, F. A pilot study about microplastics and mesoplastics in an Antarctic glacier. *Cryosphere* **2021**, *15*, 2531–2539. [CrossRef]
162. Forster, N.A.; Wilson, S.C.; Tighe, M.K. Microplastic pollution on hiking and running trails in Australian protected environments. *Sci. Total Environ.* **2023**, *874*, 162473. [CrossRef] [PubMed]
163. Pastorino, P.; Pizzul, E.; Bertoli, M.; Anselmi, S.; Kušće, M.; Menconi, V.; Prearo, M.; Renzi, M. First insights into plastic and microplastic occurrence in biotic and abiotic compartments, and snow from a high-mountain lake (Carnic Alps). *Chemosphere* **2021**, *265*, 129121. [CrossRef] [PubMed]
164. Ferrero, L.; Scibetta, L.; Markuszewski, P.; Mazurkiewicz, M.; Drozdowska, V.; Makuch, P.; Jutrzenka-Trzebiatowska, P.; Zaleska-Medynska, A.; Andò, S.; Saliu, F.; et al. Airborne and marine microplastics from an oceanographic survey at the Baltic Sea: An emerging role of air-sea interaction? *Sci. Total Environ.* **2022**, *824*, 153709. [CrossRef]
165. Rezaei, M.; Riksen, M.J.P.M.; Sirjani, E.; Sameni, A.; Geissen, V. Wind erosion as a driver for transport of light density microplastics. *Sci. Total Environ.* **2019**, *669*, 273–281. [CrossRef]
166. Crosta, A.; De Felice, B.; Antonioli, D.; Chiarcos, R.; Perin, E.; Ortenzi, M.A.; Gazzotti, S.; Azzoni, R.S.; Fugazza, D.; Gianotti, V.; et al. Microplastic contamination of supraglacial debris differs among glaciers with different anthropic pressures. *Sci. Total Environ.* **2022**, *851*, 158301. [CrossRef]
167. Perera, K.; Ziajahromi, S.; Bengtson Nash, S.; Manage, P.M.; Leusch, F.D.L. Airborne Microplastics in Indoor and Outdoor Environments of a Developing Country in South Asia: Abundance, Distribution, Morphology, and Possible Sources. *Environ. Sci. Technol.* **2022**, *56*, 16676–16685. [CrossRef]
168. Chandrakanthan, K.; Fraser, M.P.; Herckes, P. Airborne microplastics in a suburban location in the desert southwest: Occurrence and identification challenges. *Atmos. Environ.* **2023**, *298*, 119617. [CrossRef]
169. Beaurepaire, M.; Dris, R.; Tassin, B.; Gasperi, J. Comparison of total atmospheric microplastic deposition on a peri-urban and an agricultural site in the Paris region. In Proceedings of the EGU General Assembly 2022, Vienna, Austria, 23–27 May 2022. [CrossRef]
170. Sun, J.; Peng, Z.; Zhu, Z.-R.; Fu, W.; Dai, X.; Ni, B.-J. The atmospheric microplastics deposition contributes to microplastic pollution in urban waters. *Water Res.* **2022**, *225*, 119116. [CrossRef]
171. Li, C.; Wang, X.; Zhu, L.; Liu, K.; Zong, C.; Wei, N.; Li, D. Enhanced impacts evaluation of Typhoon Sinlaku (2020) on atmospheric microplastics in South China Sea during the East Asian Summer Monsoon. *Sci. Total Environ.* **2022**, *806*, 150767. [CrossRef]
172. Wang, X.; Liu, K.; Zhu, L.; Li, C.; Song, Z.; Li, D. Efficient transport of atmospheric microplastics onto the continent via the East Asian summer monsoon. *J. Hazard. Mater.* **2021**, *414*, 125477. [CrossRef]
173. Roblin, B.; Aherne, J. Moss as a biomonitor for the atmospheric deposition of anthropogenic microfibres. *Sci. Total Environ.* **2020**, *715*, 136973. [CrossRef] [PubMed]
174. Romarate, R.A.; Ancla, S.M.B.; Patilan, D.M.M.; Inocente, S.A.T.; Pacilan, C.J.M.; Sinco, A.L.; Guihawan, J.Q.; Capangpangan, R.Y.; Lubguban, A.A.; Bacosa, H.P. Breathing plastics in Metro Manila, Philippines: Presence of suspended atmospheric microplastics in ambient air. *Environ. Sci. Pollut. Res.* **2023**, *30*, 53662–53673. [CrossRef] [PubMed]
175. Kernchen, S.; Löder, M.G.J.; Fischer, F.; Fischer, D.; Moses, S.R.; Georgi, C.; Nölscher, A.C.; Held, A.; Laforsch, C. Airborne microplastic concentrations and deposition across the Weser River catchment. *Sci. Total Environ.* **2022**, *818*, 151812. [CrossRef]
176. Materić, D.; Kjær, H.A.; Vallelonga, P.; Tison, J.-L.; Röckmann, T.; Holzinger, R. Nanoplastics measurements in Northern and Southern polar ice. *Environ. Res.* **2022**, *208*, 112741. [CrossRef]
177. Zhu, X.; Huang, W.; Fang, M.; Liao, Z.; Wang, Y.; Xu, L.; Mu, Q.; Shi, C.; Lu, C.; Deng, H.; et al. Airborne Microplastic Concentrations in Five Megacities of Northern and Southeast China. *Environ. Sci. Technol.* **2021**, *55*, acs.est.1c03618. [CrossRef]

178. Loppi, S.; Roblin, B.; Paoli, L.; Aherne, J. Accumulation of airborne microplastics in lichens from a landfill dumping site (Italy). *Sci. Rep.* **2021**, *11*, 4564. [CrossRef]
179. Xumiao, L.; Prata, J.C.; Alves, J.R.; Duarte, A.C.; Rocha-Santos, T.; Cerqueira, M. Airborne microplastics and fibers in indoor residential environments in Aveiro, Portugal. *Environ. Adv.* **2021**, *6*, 100134. [CrossRef]
180. Postma, J.V. An inexpensive atmospheric microplastic collector for use in remote areas. *Atmos. Pollut. Res.* **2022**, *13*, 101550. [CrossRef]
181. Yuan, Z.; Pei, C.; Li, H.; Lin, L.; Liu, S.; Hou, R.; Liao, R.; Xu, X. Atmospheric microplastics at a southern China metropolis: Occurrence, deposition flux, exposure risk and washout effect of rainfall. *Sci. Total Environ.* **2023**, *869*, 161839. [CrossRef]
182. Stanton, T.; Johnson, M.; Nathanail, P.; MacNaughtan, W.; Gomes, R.L. Freshwater and airborne textile fibre populations are dominated by 'natural', not microplastic, fibres. *Sci. Total Environ.* **2019**, *666*, 377–389. [CrossRef]
183. Szewc, K.; Graca, B.; Dołęga, A. Atmospheric deposition of microplastics in the coastal zone: Characteristics and relationship with meteorological factors. *Sci. Total Environ.* **2021**, *761*, 143272. [CrossRef] [PubMed]
184. Liu, Z.; Bai, Y.; Ma, T.; Liu, X.; Wei, H.; Meng, H.; Fu, Y.; Ma, Z.; Zhang, L.; Zhao, J. Distribution and possible sources of atmospheric microplastic deposition in a valley basin city (Lanzhou, China). *Ecotoxicol. Environ. Saf.* **2022**, *233*, 113353. [CrossRef] [PubMed]
185. Liu, K.; Wang, X.; Song, Z.; Wei, N.; Ye, H.; Cong, X.; Zhao, L.; Li, Y.; Qu, L.; Zhu, L.; et al. Global inventory of atmospheric fibrous microplastics input into the ocean: An implication from the indoor origin. *J. Hazard. Mater.* **2020**, *400*, 123223. [CrossRef]
186. Klein, M.; Fischer, E.K. Microplastic abundance in atmospheric deposition within the Metropolitan area of Hamburg, Germany. *Sci. Total Environ.* **2019**, *685*, 96–103. [CrossRef]
187. Welsh, B.; Aherne, J.; Paterson, A.M.; Yao, H.; McConnell, C. Atmospheric deposition of anthropogenic particles and microplastics in south-central Ontario, Canada. *Sci. Total Environ.* **2022**, *835*, 155426. [CrossRef]
188. Jafarova, M.; Contardo, T.; Aherne, J.; Loppi, S. Lichen Biomonitoring of Airborne Microplastics in Milan (N Italy). *Biology* **2022**, *11*, 1815. [CrossRef]
189. Purwiyanto, A.I.S.; Prartono, T.; Riani, E.; Naulita, Y.; Cordova, M.R.; Koropitan, A.F. The deposition of atmospheric microplastics in Jakarta-Indonesia: The coastal urban area. *Mar. Pollut. Bull.* **2022**, *174*, 113195. [CrossRef] [PubMed]
190. Jenner, L.C.; Sadofsky, L.R.; Danopoulos, E.; Chapman, E.; White, D.; Jenkins, R.L.; Rotchell, J.M. Outdoor Atmospheric Microplastics within the Humber Region (United Kingdom): Quantification and Chemical Characterisation of Deposited Particles Present. *Atmosphere* **2022**, *13*, 265. [CrossRef]
191. Kernchen, S.; Schmalz, H.; Löder, M.G.J.; Georgi, C.; Einhorn, A.; Greiner, A.; Nölscher, A.C.; Laforsch, C.; Held, A. Wet and Dry Deposition Flux Measurements of Atmospheric Microplastic Particles in Central Germany. *Res. Sq.* **2022**. [CrossRef]
192. Hasnatul, A.H.; Azman, N.F.; NorRuwaida, J.; Dewika, M.; Sara, Y.Y. Relationship of Suspended Atmospheric Microplastics and Meteorological Parameters in Universiti Teknologi Malaysia, Kuala Lumpur. *IOP Conf. Ser. Earth Environ. Sci.* **2023**, *1135*, 012042. [CrossRef]
193. Rosso, B.; Corami, F.; Barbante, C.; Gambaro, A. Quantification and identification of airborne small microplastics (<100 μm) and other microlitter components in atmospheric aerosol via a novel elutriation and oleo-extraction method. *Environ. Pollut.* **2023**, *318*, 120889. [CrossRef]
194. Zhu, J.; Zhang, X.; Liao, K.; Wu, P.; Jin, H. Microplastics in dust from different indoor environments. *Sci. Total Environ.* **2022**, *833*, 155256. [CrossRef] [PubMed]
195. Roblin, B.; Ryan, M.; Vreugdenhil, A.; Aherne, J. Ambient Atmospheric Deposition of Anthropogenic Microfibers and Microplastics on the Western Periphery of Europe (Ireland). *Environ. Sci. Technol.* **2020**, *54*, 11100–11108. [CrossRef] [PubMed]
196. Abbasi, S.; Rezaei, M.; Mina, M.; Sameni, A.; Oleszczuk, P.; Turner, A.; Ritsema, C. Entrainment and horizontal atmospheric transport of microplastics from soil. *Chemosphere* **2023**, *322*, 138150. [CrossRef]
197. Liu, Z.; Huang, Q.; Chen, L.; Li, J.; Jia, H. Is the impact of atmospheric microplastics on human health underestimated? Uncertainty in risk assessment: A case study of urban atmosphere in Xi'an, Northwest China. *Sci. Total Environ.* **2022**, *851*, 158167. [CrossRef] [PubMed]
198. Hyytiäinen, H.K.; Jayaprakash, B.; Kirjavainen, P.V.; Saari, S.E.; Holopainen, R.; Keskinen, J.; Hämeri, K.; Hyvärinen, A.; Boor, B.E.; Täubel, M. Crawling-induced floor dust resuspension affects the microbiota of the infant breathing zone. *Microbiome* **2018**, *6*, 25. [CrossRef] [PubMed]
199. Burkhart, J.; Piacitelli, C.; Schwegler-Berry, D.; Jones, W. Environmental Study of Nylon Flocking Process. *J. Toxicol. Environ. Health A* **1999**, *57*, 1–23. [CrossRef]
200. Bahners, T.; Ehrler, P.; Hengstberger, M. Erste Untersuchungen zur Erfassung und Charakterisierung textiler Feinstaube. *Melliand Text* **1994**, *75*, 24–30.
201. Xie, Y.; Li, Y.; Feng, Y.; Cheng, W.; Wang, Y. Inhalable microplastics prevails in air: Exploring the size detection limit. *Environ. Int.* **2022**, *162*, 107151. [CrossRef]
202. Zhang, Q.; Xu, E.G.; Li, J.; Chen, Q.; Ma, L.; Zeng, E.Y.; Shi, H. A Review of Microplastics in Table Salt, Drinking Water, and Air: Direct Human Exposure. *Environ. Sci. Technol.* **2020**, *54*, 3740–3751. [CrossRef]
203. Duzgoren-Aydin, N.S. Health Effects of Atmospheric Particulates: A Medical Geology Perspective. *J. Environ. Sci. Health Part C* **2008**, *26*, 1–39. [CrossRef] [PubMed]

204. Hamoir, J.; Nemmar, A.; Halloy, D.; Wirth, D.; Vincke, G.; Vanderplasschen, A.; Nemery, B.; Gustin, P. Effect of polystyrene particles on lung microvascular permeability in isolated perfused rabbit lungs: Role of size and surface properties. *Toxicol. Appl. Pharmacol.* **2003**, *190*, 278–285. [CrossRef] [PubMed]
205. Yacobi, N.R.; DeMaio, L.; Xie, J.; Hamm-Alvarez, S.F.; Borok, Z.; Kim, K.-J.; Crandall, E.D. Polystyrene nanoparticle trafficking across alveolar epithelium. *Nanomed. Nanotechnol. Biol. Med.* **2008**, *4*, 139–145. [CrossRef] [PubMed]
206. Dong, C.-D.; Chen, C.-W.; Chen, Y.-C.; Chen, H.-H.; Lee, J.-S.; Lin, C.-H. Polystyrene microplastic particles: In vitro pulmonary toxicity assessment. *J. Hazard. Mater.* **2020**, *385*, 121575. [CrossRef]
207. Toussaint, B.; Raffael, B.; Angers-Loustau, A.; Gilliland, D.; Kestens, V.; Petrillo, M.; Rio-Echevarria, I.M.; Van Den Eede, G. Review of micro- and nanoplastic contamination in the food chain. *Food Addit. Contam. Part A* **2019**, *36*, 639–673. [CrossRef]
208. Choy, C.A.; Robison, B.H.; Gagne, T.O.; Erwin, B.; Firl, E.; Halden, R.U.; Hamilton, J.A.; Katija, K.; Lisin, S.E.; Rolsky, C.; et al. The vertical distribution and biological transport of marine microplastics across the epipelagic and mesopelagic water column. *Sci. Rep.* **2019**, *9*, 7843. [CrossRef]
209. 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. [CrossRef]
210. Diaz-Basantes, M.F.; Conesa, J.A.; Fullana, A. Microplastics in Honey, Beer, Milk and Refreshments in Ecuador as Emerging Contaminants. *Sustainability* **2020**, *12*, 5514. [CrossRef]
211. Winkler, A.; Santo, N.; Ortenzi, M.A.; Bolzoni, E.; Bacchetta, R.; Tremolada, P. Does mechanical stress cause microplastic release from plastic water bottles? *Water Res.* **2019**, *166*, 115082. [CrossRef]
212. Yan, Z.; Liu, Y.; Zhang, T.; Zhang, F.; Ren, H.; Zhang, Y. Analysis of Microplastics in Human Feces Reveals a Correlation between Fecal Microplastics and Inflammatory Bowel Disease Status. *Environ. Sci. Technol.* **2022**, *56*, 414–421. [CrossRef]
213. Smith, M.; Love, D.C.; Rochman, C.M.; Neff, R.A. Microplastics in Seafood and the Implications for Human Health. *Curr. Environ. Health Rep.* **2018**, *5*, 375–386. [CrossRef] [PubMed]
214. Ageel, H.K.; Harrad, S.; Abdallah, M.A.-E. Occurrence, human exposure, and risk of microplastics in the indoor environment. *Environ. Sci. Process. Impacts* **2022**, *24*, 17–31. [CrossRef] [PubMed]
215. Swaen, G.G.; Teggeler, O.; Van Amelsvoort, L.G. False positive outcomes and design characteristics in occupational cancer epidemiology studies. *Int. J. Epidemiol.* **2001**, *30*, 948–954. [CrossRef]
216. Fournier, S.B.; D'Errico, J.N.; Adler, D.S.; Kollontzi, S.; Goedken, M.J.; Fabris, L.; Yurkow, E.J.; Stapleton, P.A. Nanopolystyrene translocation and fetal deposition after acute lung exposure during late-stage pregnancy. *Part. Fibre Toxicol.* **2020**, *17*, 55. [CrossRef]
217. Banerjee, A.; Shelver, W.L. Micro- and nanoplastic induced cellular toxicity in mammals: A review. *Sci. Total Environ.* **2021**, *755*, 142518. [CrossRef]
218. Lehmann, S.G.; Toybou, D.; Pradas Del Real, A.-E.; Arndt, D.; Tagmount, A.; Viau, M.; Safi, M.; Pacureanu, A.; Cloetens, P.; Bohic, S.; et al. Crumpling of silver nanowires by endolysosomes strongly reduces toxicity. *Proc. Natl. Acad. Sci. USA* **2019**, *116*, 14893–14898. [CrossRef]
219. Amobonye, A.; Bhagwat, P.; Raveendran, S.; Singh, S.; Pillai, S. Environmental Impacts of Microplastics and Nanoplastics: A Current Overview. *Front. Microbiol.* **2021**, *12*, 768297. [CrossRef]
220. Schirinzi, G.F.; Pérez-Pomeda, I.; Sanchís, J.; Rossini, C.; Farré, M.; Barceló, D. Cytotoxic effects of commonly used nanomaterials and microplastics on cerebral and epithelial human cells. *Environ. Res.* **2017**, *159*, 579–587. [CrossRef]
221. Akhbarizadeh, R.; Dobaradaran, S.; Amouei Torkmahalleh, M.; Saeedi, R.; Aibaghi, R.; Faraji Ghasemi, F. Suspended fine particulate matter (PM2.5), microplastics (MPs), and polycyclic aromatic hydrocarbons (PAHs) in air: Their possible relationships and health implications. *Environ. Res.* **2021**, *192*, 110339. [CrossRef]
222. Yang, J.; Cao, W.; Rui, Y. Interactions between nanoparticles and plants: Phytotoxicity and defense mechanisms. *J. Plant Interact.* **2017**, *12*, 158–169. [CrossRef]
223. Zhu, K.; Jia, H.; Zhao, S.; Xia, T.; Guo, X.; Wang, T.; Zhu, L. Formation of Environmentally Persistent Free Radicals on Microplastics under Light Irradiation. *Environ. Sci. Technol.* **2019**, *53*, 8177–8186. [CrossRef] [PubMed]
224. Zhang, Y.; Li, Y.; Su, F.; Peng, L.; Liu, D. The life cycle of micro-nano plastics in domestic sewage. *Sci. Total Environ.* **2022**, *802*, 149658. [CrossRef] [PubMed]
225. Sun, X.; Liang, J.; Zhu, M.; Zhao, Y.; Zhang, B. Microplastics in seawater and zooplankton from the Yellow Sea. *Environ. Pollut.* **2018**, *242*, 585–595. [CrossRef]
226. Lim, Y.C.; Chen, C.-W.; Cheng, Y.-R.; Chen, C.-F.; Dong, C.-D. Impacts of microplastics on scleractinian corals nearshore Liuqiu Island southwestern Taiwan. *Environ. Pollut.* **2022**, *306*, 119371. [CrossRef]
227. Duis, K.; Coors, A. Microplastics in the aquatic and terrestrial environment: Sources (with a specific focus on personal care products), fate and effects. *Environ. Sci. Eur.* **2016**, *28*, 2. [CrossRef]
228. Setälä, O.; Norkko, J.; Lehtiniemi, M. Feeding type affects microplastic ingestion in a coastal invertebrate community. *Mar. Pollut. Bull.* **2016**, *102*, 95–101. [CrossRef]
229. Song, Y.; Cao, C.; Qiu, R.; Hu, J.; Liu, M.; Lu, S.; Shi, H.; Raley-Susman, K.M.; He, D. Uptake and adverse effects of polyethylene terephthalate microplastics fibers on terrestrial snails (Achatina fulica) after soil exposure. *Environ. Pollut.* **2019**, *250*, 447–455. [CrossRef]

230. Lei, L.; Wu, S.; Lu, S.; Liu, M.; Song, Y.; Fu, Z.; Shi, H.; Raley-Susman, K.M.; He, D. Microplastic particles cause intestinal damage and other adverse effects in zebrafish Danio rerio and nematode Caenorhabditis elegans. *Sci. Total Environ.* **2018**, *619–620*, 1–8. [CrossRef]
231. De Souza Machado, A.A.; Lau, C.W.; Till, J.; Kloas, W.; Lehmann, A.; Becker, R.; Rillig, M.C. Impacts of Microplastics on the Soil Biophysical Environment. *Environ. Sci. Technol.* **2018**, *52*, 9656–9665. [CrossRef]
232. Besseling, E.; Wegner, A.; Foekema, E.M.; Van Den Heuvel-Greve, M.J.; Koelmans, A.A. Effects of Microplastic on Fitness and PCB Bioaccumulation by the Lugworm *Arenicola marina* (L.). *Environ. Sci. Technol.* **2013**, *47*, 593–600. [CrossRef]
233. Peng, L.; Fu, D.; Qi, H.; Lan, C.Q.; Yu, H.; Ge, C. Micro- and nano-plastics in marine environment: Source, distribution and threats—A review. *Sci. Total Environ.* **2020**, *698*, 134254. [CrossRef] [PubMed]
234. Carrasco-Navarro, V.; Muñiz-González, A.-B.; Sorvari, J.; Martínez-Guitarte, J.-L. Altered gene expression in Chironomus riparius (insecta) in response to tire rubber and polystyrene microplastics. *Environ. Pollut.* **2021**, *285*, 117462. [CrossRef] [PubMed]
235. Lippmann, M. Effects of fiber characteristics on lung deposition, retention, and disease. *Environ. Health Perspect.* **1990**, *88*, 311–317. [CrossRef] [PubMed]
236. Theakston, F.; Weltgesundheitsorganisation (Eds.) *Air Quality Guidelines for Europe*, 2nd ed.; WHO Regional Publications European Series; World Health Organization, Regional Office for Europe: Copenhagen, Denmark, 2000; ISBN 978-92-890-1358-1.
237. Cavallo, D.; Campopiano, A.; Cardinali, G.; Casciardi, S.; Simone, P.D.; Kovacs, D.; Perniconi, B.; Spagnoli, G.; Ursini, C.L.; Fanizza, C. Cytotoxic and oxidative effects induced by man-made vitreous fibers (MMVFs) in a human mesothelial cell line. *Toxicology* **2004**, *201*, 219–229. [CrossRef]
238. Ferreira, I.; Venâncio, C.; Lopes, I.; Oliveira, M. Nanoplastics and marine organisms: What has been studied? *Environ. Toxicol. Pharmacol.* **2019**, *67*, 1–7. [CrossRef]
239. Lehner, R.; Weder, C.; Petri-Fink, A.; Rothen-Rutishauser, B. Emergence of Nanoplastic in the Environment and Possible Impact on Human Health. *Environ. Sci. Technol.* **2019**, *53*, 1748–1765. [CrossRef]
240. Jiang, X.; Chen, H.; Liao, Y.; Ye, Z.; Li, M.; Klobučar, G. Ecotoxicity and genotoxicity of polystyrene microplastics on higher plant Vicia faba. *Environ. Pollut.* **2019**, *250*, 831–838. [CrossRef]
241. Huang, D.; Tao, J.; Cheng, M.; Deng, R.; Chen, S.; Yin, L.; Li, R. Microplastics and nanoplastics in the environment: Macroscopic transport and effects on creatures. *J. Hazard. Mater.* **2021**, *407*, 124399. [CrossRef]
242. Liu, K.; Wang, X.; Song, Z.; Wei, N.; Li, D. Terrestrial plants as a potential temporary sink of atmospheric microplastics during transport. *Sci. Total Environ.* **2020**, *742*, 140523. [CrossRef]
243. Zhang, C.; Chen, X.; Wang, J.; Tan, L. Toxic effects of microplastic on marine microalgae Skeletonema costatum: Interactions between microplastic and algae. *Environ. Pollut.* **2017**, *220*, 1282–1288. [CrossRef]
244. Fu, D.; Zhang, Q.; Fan, Z.; Qi, H.; Wang, Z.; Peng, L. Aged microplastics polyvinyl chloride interact with copper and cause oxidative stress towards microalgae Chlorella vulgaris. *Aquat. Toxicol.* **2019**, *216*, 105319. [CrossRef] [PubMed]
245. Bhattacharya, P.; Lin, S.; Turner, J.P.; Ke, P.C. Physical Adsorption of Charged Plastic Nanoparticles Affects Algal Photosynthesis. *J. Phys. Chem. C* **2010**, *114*, 16556–16561. [CrossRef]
246. 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. [CrossRef] [PubMed]
247. Falsini, S.; Colzi, I.; Chelazzi, D.; Dainelli, M.; Schiff, S.; Papini, A.; Coppi, A.; Gonnelli, C.; Ristori, S. Plastic is in the air: Impact of micro-nanoplastics from airborne pollution on *Tillandsia usneoides* (L.) L. (Bromeliaceae) as a possible green sensor. *J. Hazard. Mater.* **2022**, *437*, 129314. [CrossRef]
248. Lithner, D.; Larsson, Å.; Dave, G. Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. *Sci. Total Environ.* **2011**, *409*, 3309–3324. [CrossRef]
249. Sun, X.; Chen, B.; Li, Q.; Liu, N.; Xia, B.; Zhu, L.; Qu, K. Toxicities of polystyrene nano- and microplastics toward marine bacterium Halomonas alkaliphila. *Sci. Total Environ.* **2018**, *642*, 1378–1385. [CrossRef]
250. Miao, L.; Guo, S.; Liu, Z.; Liu, S.; You, G.; Qu, H.; Hou, J. Effects of Nanoplastics on Freshwater Biofilm Microbial Metabolic Functions as Determined by BIOLOG ECO Microplates. *Int. J. Environ. Res. Public Health* **2019**, *16*, 4639. [CrossRef]
251. Mammo, F.K.; Amoah, I.D.; Gani, K.M.; Pillay, L.; Ratha, S.K.; Bux, F.; Kumari, S. Microplastics in the environment: Interactions with microbes and chemical contaminants. *Sci. Total Environ.* **2020**, *743*, 140518. [CrossRef]
252. Bakir, A.; Rowland, S.J.; Thompson, R.C. Enhanced desorption of persistent organic pollutants from microplastics under simulated physiological conditions. *Environ. Pollut.* **2014**, *185*, 16–23. [CrossRef]
253. Brennecke, D.; Duarte, B.; Paiva, F.; Caçador, I.; Canning-Clode, J. Microplastics as vector for heavy metal contamination from the marine environment. *Estuar. Coast. Shelf Sci.* **2016**, *178*, 189–195. [CrossRef]
254. United Nations Environment Programme. *UNEP Year Book 2011: Emerging Issues in Our Global Environment*; United Nations Environment Programme: Brussels, Belgium, 2011.
255. Gorsky, G.; Bourdin, G.; Lombard, F.; Pedrotti, M.L.; Audrain, S.; Bin, N.; Boss, E.; Bowler, C.; Cassar, N.; Caudan, L.; et al. Expanding Tara Oceans Protocols for Underway, Ecosystemic Sampling of the Ocean-Atmosphere Interface During Tara Pacific Expedition (2016–2018). *Front. Mar. Sci.* **2019**, *6*, 750. [CrossRef]
256. De Tender, C.; Schlundt, C.; Devriese, L.I.; Mincer, T.J.; Zettler, E.R.; Amaral-Zettler, L.A. A review of microscopy and comparative molecular-based methods to characterize "Plastisphere" communities. *Anal. Methods* **2017**, *9*, 2132–2143. [CrossRef]

257. Jacquin, J.; Cheng, J.; Odobel, C.; Pandin, C.; Conan, P.; Pujo-Pay, M.; Barbe, V.; Meistertzheim, A.-L.; Ghiglione, J.-F. Microbial Ecotoxicology of Marine Plastic Debris: A Review on Colonization and Biodegradation by the "Plastisphere". *Front. Microbiol.* **2019**, *10*, 865. [CrossRef]
258. Fu, L.; Li, J.; Wang, G.; Luan, Y.; Dai, W. Adsorption Behavior of Organic Pollutants on Microplastics. *Ecotoxicol. Environ. Saf.* **2021**, *217*, 112207. [CrossRef]
259. Gao, F.; Li, J.; Sun, C.; Zhang, L.; Jiang, F.; Cao, W.; Zheng, L. Study on the capability and characteristics of heavy metals enriched on microplastics in marine environment. *Mar. Pollut. Bull.* **2019**, *144*, 61–67. [CrossRef]
260. Wu, P.; Huang, J.; Zheng, Y.; Yang, Y.; Zhang, Y.; He, F.; Chen, H.; Quan, G.; Yan, J.; Li, T.; et al. Environmental occurrences, fate, and impacts of microplastics. *Ecotoxicol. Environ. Saf.* **2019**, *184*, 109612. [CrossRef]
261. Trevisan, R.; Uzochukwu, D.; Di Giulio, R.T. PAH Sorption to Nanoplastics and the Trojan Horse Effect as Drivers of Mitochondrial Toxicity and PAH Localization in Zebrafish. *Front. Environ. Sci.* **2020**, *8*, 78. [CrossRef]
262. Abrams, R.; Ali, M.; Denton, P.; Igualada, J.A.; Groen, M.; Gschwind, E. Colouring plastics: Fundamentals and trends. *Plast. Addit. Compd.* **2001**, *3*, 18–25. [CrossRef]
263. Sridharan, S.; Kumar, M.; Saha, M.; Kirkham, M.B.; Singh, L.; Bolan, N.S. The Polymers and Their Additives in Particulate Plastics: What Makes Them Hazardous to the Fauna? *Sci. Total Environ.* **2022**, *824*, 153828. [CrossRef]
264. Turner, A.; Filella, M. Hazardous metal additives in plastics and their environmental impacts. *Environ. Int.* **2021**, *156*, 106622. [CrossRef]
265. Belzagui, F.; Buscio, V.; Gutiérrez-Bouzán, C.; Vilaseca, M. Cigarette butts as a microfiber source with a microplastic level of concern. *Sci. Total Environ.* **2021**, *762*, 144165. [CrossRef] [PubMed]
266. Law, B.D.; Bunn, W.B.; Hesterberg, T.W. Solubility of Polymeric Organic Fibers and Manmade Vitreous Fibers in Gambles Solution. *Inhal. Toxicol.* **1990**, *2*, 321–339. [CrossRef]
267. Fossi, M.C.; Coppola, D.; Baini, M.; Giannetti, M.; Guerranti, C.; Marsili, L.; Panti, C.; De Sabata, E.; Clò, S. Large filter feeding marine organisms as indicators of microplastic in the pelagic environment: The case studies of the Mediterranean basking shark (*Cetorhinus maximus*) and fin whale (*Balaenoptera physalus*). *Mar. Environ. Res.* **2014**, *100*, 17–24. [CrossRef]
268. Gewert, B.; Plassmann, M.M.; MacLeod, M. Pathways for degradation of plastic polymers floating in the marine environment. *Environ. Sci. Process. Impacts* **2015**, *17*, 1513–1521. [CrossRef]
269. Ladewig, S.M.; Bao, S.; Chow, A.T. Natural Fibers: A Missing Link to Chemical Pollution Dispersion in Aquatic Environments. *Environ. Sci. Technol.* **2015**, *49*, 12609–12610. [CrossRef]
270. Popoola, A.V. Comparative fastness assessment performance of cellulosic fibers dyed using natural colorants. *J. Appl. Polym. Sci.* **2000**, *77*, 752–755. [CrossRef]
271. Subasinghe, A.; Somashekar, A.A.; Bhattacharyya, D. Effects of wool fibre and other additives on the flammability and mechanical performance of polypropylene/kenaf composites. *Compos. Part B Eng.* **2018**, *136*, 168–176. [CrossRef]
272. Finnegan, A.M.D.; Süsserott, R.; Gabbott, S.E.; Gouramanis, C. Man-made natural and regenerated cellulosic fibres greatly outnumber microplastic fibres in the atmosphere. *Environ. Pollut.* **2022**, *310*, 119808. [CrossRef]
273. Wang, J.; Liu, X.; Li, Y.; Powell, T.; Wang, X.; Wang, G.; Zhang, P. Microplastics as contaminants in the soil environment: A mini-review. *Sci. Total Environ.* **2019**, *691*, 848–857. [CrossRef]
274. Ng, E.L.; Lin, S.Y.; Dungan, A.M.; Colwell, J.M.; Ede, S.; Huerta Lwanga, E.; Meng, K.; Geissen, V.; Blackall, L.L.; Chen, D. Microplastic pollution alters forest soil microbiome. *J. Hazard. Mater.* **2021**, *409*, 124606. [CrossRef]
275. Rillig, M.C.; Lehmann, A. Microplastic in terrestrial ecosystems. *Science* **2020**, *368*, 1430–1431. [CrossRef] [PubMed]
276. Rillig, M.C.; Ingraffia, R.; De Souza Machado, A.A. Microplastic Incorporation into Soil in Agroecosystems. *Front. Plant Sci.* **2017**, *8*, 1805. [CrossRef] [PubMed]
277. Boots, B.; Russell, C.W.; Green, D.S. Effects of Microplastics in Soil Ecosystems: Above and Below Ground. *Environ. Sci. Technol.* **2019**, *53*, 11496–11506. [CrossRef]
278. Kim, S.W.; An, Y.-J. Soil microplastics inhibit the movement of springtail species. *Environ. Int.* **2019**, *126*, 699–706. [CrossRef]
279. Awet, T.T.; Kohl, Y.; Meier, F.; Straskraba, S.; Grün, A.-L.; Ruf, T.; Jost, C.; Drexel, R.; Tunc, E.; Emmerling, C. Effects of polystyrene nanoparticles on the microbiota and functional diversity of enzymes in soil. *Environ. Sci. Eur.* **2018**, *30*, 11. [CrossRef]
280. Li, J.; Liu, H.; Paul Chen, J. Microplastics in freshwater systems: A review on occurrence, environmental effects, and methods for microplastics detection. *Water Res.* **2018**, *137*, 362–374. [CrossRef]
281. Cao, D.; Wang, X.; Luo, X.; Liu, G.; Zheng, H. Effects of polystyrene microplastics on the fitness of earthworms in an agricultural soil. *IOP Conf. Ser. Earth Environ. Sci.* **2017**, *61*, 012148. [CrossRef]
282. Turner, S.; Horton, A.A.; Rose, N.L.; Hall, C. A temporal sediment record of microplastics in an urban lake, London, UK. *J. Paleolimnol.* **2019**, *61*, 449–462. [CrossRef]
283. Malygina, N.S.; Biryukov, R.Y.; Kuryatnikova, N.A.; Mitrofanova, E.Y.; Pershin, D.K.; Zolotov, D.V.; Chernykh, D.V. Microplastics in the snow cover of the south of Western Siberia. *IOP Conf. Ser. Earth Environ. Sci.* **2020**, *611*, 012034. [CrossRef]
284. Revell, L.E.; Kuma, P.; Le Ru, E.C.; Somerville, W.R.C.; Gaw, S. Direct radiative effects of airborne microplastics. *Nature* **2021**, *598*, 462–467. [CrossRef]
285. Zhang, Y.-L.; Kang, S.-C.; Gao, T.-G. Microplastics have light-absorbing ability to enhance cryospheric melting. *Adv. Clim. Chang. Res.* **2022**, *13*, 455–458. [CrossRef]

286. Kim, S.-K.; Lee, H.-J.; Kim, J.-S.; Kang, S.-H.; Yang, E.-J.; Cho, K.-H.; Tian, Z.; Andrady, A. Importance of seasonal sea ice in the western Arctic ocean to the Arctic and global microplastic budgets. *J. Hazard. Mater.* **2021**, *418*, 125971. [CrossRef] [PubMed]
287. Chubarenko, I. Physical processes behind interactions of microplastic particles with natural ice. *Environ. Res. Commun.* **2022**, *4*, 012001. [CrossRef]
288. Hamilton, B.M.; Jantunen, L.; Bergmann, M.; Vorkamp, K.; Aherne, J.; Magnusson, K.; Herzke, D.; Granberg, M.; Hallanger, I.G.; Gomiero, A.; et al. Monitoring microplastics in the atmosphere and cryosphere in the circumpolar North: A case for multi-compartment monitoring. *Arct. Sci.* **2022**, *8*, 1116–1126. [CrossRef]
289. Dong, H.; Wang, L.; Wang, X.; Xu, L.; Chen, M.; Gong, P.; Wang, C. Microplastics in a Remote Lake Basin of the Tibetan Plateau: Impacts of Atmospheric Transport and Glacial Melting. *Environ. Sci. Technol.* **2021**, *55*, 12951–12960. [CrossRef]
290. Vergara-Temprado, J.; Miltenberger, A.K.; Furtado, K.; Grosvenor, D.P.; Shipway, B.J.; Hill, A.A.; Wilkinson, J.M.; Field, P.R.; Murray, B.J.; Carslaw, K.S. Strong control of Southern Ocean cloud reflectivity by ice-nucleating particles. *Proc. Natl. Acad. Sci. USA* **2018**, *115*, 2687–2692. [CrossRef]
291. Beaurepaire, M.; Dris, R.; Gasperi, J.; Tassin, B. Microplastics in the atmospheric compartment: A comprehensive review on methods, results on their occurrence and determining factors. *Curr. Opin. Food Sci.* **2021**, *41*, 159–168. [CrossRef]
292. González-Pleiter, M.; Edo, C.; Aguilera, Á.; Viúdez-Moreiras, D.; Pulido-Reyes, G.; González-Toril, E.; Osuna, S.; De Diego-Castilla, G.; Leganés, F.; Fernández-Piñas, F.; et al. Occurrence and transport of microplastics sampled within and above the planetary boundary layer. *Sci. Total Environ.* **2021**, *761*, 143213. [CrossRef]
293. Cai, H.; Chen, M.; Chen, Q.; Du, F.; Liu, J.; Shi, H. Microplastic quantification affected by structure and pore size of filters. *Chemosphere* **2020**, *257*, 127198. [CrossRef]
294. O'Brien, S.; Okoffo, E.D.; Rauert, C.; O'Brien, J.W.; Ribeiro, F.; Burrows, S.D.; Toapanta, T.; Wang, X.; Thomas, K.V. Quantification of selected microplastics in Australian urban road dust. *J. Hazard. Mater.* **2021**, *416*, 125811. [CrossRef]
295. Sholokhova, A.; Denafas, G.; Mykhaylenko, V. The organic output from mechanical–biological treatment plants as a source of microplastics: Mini-review on current knowledge, research methodology and future study perspectives. *Waste Manag. Res. J. Sustain. Circ. Econ.* **2021**, *39*, 652–663. [CrossRef] [PubMed]
296. Vermeiren, P.; Muñoz, C.; Ikejima, K. Microplastic identification and quantification from organic rich sediments: A validated laboratory protocol. *Environ. Pollut.* **2020**, *262*, 114298. [CrossRef] [PubMed]
297. Hurley, R.R.; Lusher, A.L.; Olsen, M.; Nizzetto, L. Validation of a Method for Extracting Microplastics from Complex, Organic-Rich, Environmental Matrices. *Environ. Sci. Technol.* **2018**, *52*, 7409–7417. [CrossRef]
298. Al-Azzawi, M.S.M.; Kefer, S.; Weißer, J.; Reichel, J.; Schwaller, C.; Glas, K.; Knoop, O.; Drewes, J.E. Validation of Sample Preparation Methods for Microplastic Analysis in Wastewater Matrices—Reproducibility and Standardization. *Water* **2020**, *12*, 2445. [CrossRef]
299. Cojocariu, A.M.; Mutin, P.H.; Dumitriu, E.; Fajula, F.; Vioux, A.; Hulea, V. Mild Oxidation of Bulky Organic Compounds with Hydrogen Peroxide over Mesoporous TiO2-SiO2 Xerogels Prepared by Non-Hydrolytic Sol–Gel. *Appl. Catal. B Environ.* **2010**, *97*, 407–413. [CrossRef]
300. Martinho, S.D.; Fernandes, V.C.; Figueiredo, S.A.; Delerue-Matos, C. Microplastic Pollution Focused on Sources, Distribution, Contaminant Interactions, Analytical Methods, and Wastewater Removal Strategies: A Review. *Int. J. Environ. Res. Public Health* **2022**, *19*, 5610. [CrossRef]
301. Claessens, M.; Van Cauwenberghe, L.; Vandegehuchte, M.B.; Janssen, C.R. New techniques for the detection of microplastics in sediments and field collected organisms. *Mar. Pollut. Bull.* **2013**, *70*, 227–233. [CrossRef]
302. Löder, M.G.J.; Gerdts, G. Methodology Used for the Detection and Identification of Microplastics—A Critical Appraisal. In *Marine Anthropogenic Litter*; Bergmann, M., Gutow, L., Klages, M., Eds.; Springer International Publishing: Cham, Switzerland, 2015; pp. 201–227, ISBN 978-3-319-16509-7.
303. Park, H.; Park, B. Review of Microplastic Distribution, Toxicity, Analysis Methods, and Removal Technologies. *Water* **2021**, *13*, 2736. [CrossRef]
304. Tagg, A.S.; Harrison, J.P.; Ju-Nam, Y.; Sapp, M.; Bradley, E.L.; Sinclair, C.J.; Ojeda, J.J. Fenton's reagent for the rapid and efficient isolation of microplastics from wastewater. *Chem. Commun.* **2017**, *53*, 372–375. [CrossRef]
305. Lusher, A.L.; Munno, K.; Hermabessiere, L.; Carr, S. Isolation and Extraction of Microplastics from Environmental Samples: An Evaluation of Practical Approaches and Recommendations for Further Harmonization. *Appl. Spectrosc.* **2020**, *74*, 1049–1065. [CrossRef]
306. 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. [CrossRef] [PubMed]
307. Frias, J.P.G.L.; Pagter, E.; Nash, R.; O'Connor, I.; Carretero, O.; Filgueiras, A.; Viñas, L.; Gago, J.; Antunes, J.C.; Bessa, F.; et al. *Standardised Protocol for Monitoring Microplastics in Sediments*; JPI-Oceans BASEMAN Project: Brussels, Belgium, 2018. [CrossRef]
308. Kedzierski, M.; Le Tilly, V.; César, G.; Sire, O.; Bruzaud, S. Efficient microplastics extraction from sand. A cost effective methodology based on sodium iodide recycling. *Mar. Pollut. Bull.* **2017**, *115*, 120–129. [CrossRef] [PubMed]
309. Renner, G.; Nellessen, A.; Schwiers, A.; Wenzel, M.; Schmidt, T.C.; Schram, J. Hydrophobicity–water/air–based enrichment cell for microplastics analysis within environmental samples: A proof of concept. *MethodsX* **2020**, *7*, 100732. [CrossRef] [PubMed]
310. Budke, M.; Vortmann, S.B.M. *The GreenLab_OS-Micro Plastics in Our Environment*; Libreria Universitaria: Florence, Italy, 2018; pp. 559–563.

311. Radford, F.; Zapata-Restrepo, L.M.; Horton, A.A.; Hudson, M.D.; Shaw, P.J.; Williams, I.D. Developing a systematic method for extraction of microplastics in soils. *Anal. Methods* **2021**, *13*, 1695–1705. [CrossRef]
312. Tian, X.; Yang, M.; Guo, Z.; Chang, C.; Li, J.; Guo, Z.; Wang, R.; Li, Q.; Zou, X. Plastic mulch film induced soil microplastic enrichment and its impact on wind-blown sand and dust. *Sci. Total Environ.* **2022**, *813*, 152490. [CrossRef]
313. Prata, J.C.; Da Costa, J.P.; Duarte, A.C.; Rocha-Santos, T. Methods for sampling and detection of microplastics in water and sediment: A critical review. *TrAC Trends Anal. Chem.* **2019**, *110*, 150–159. [CrossRef]
314. Lenz, R.; Enders, K.; Stedmon, C.A.; Mackenzie, D.M.A.; Nielsen, T.G. A critical assessment of visual identification of marine microplastic using Raman spectroscopy for analysis improvement. *Mar. Pollut. Bull.* **2015**, *100*, 82–91. [CrossRef]
315. Prata, J.C.; Castro, J.L.; Da Costa, J.P.; Duarte, A.C.; Rocha-Santos, T.; Cerqueira, M. The importance of contamination control in airborne fibers and microplastic sampling: Experiences from indoor and outdoor air sampling in Aveiro, Portugal. *Mar. Pollut. Bull.* **2020**, *159*, 111522. [CrossRef]
316. 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. [CrossRef]
317. Erni-Cassola, G.; Gibson, M.I.; Thompson, R.C.; Christie-Oleza, J.A. Lost, but Found with Nile Red: A Novel Method for Detecting and Quantifying Small Microplastics (1 mm to 20 μm) in Environmental Samples. *Environ. Sci. Technol.* **2017**, *51*, 13641–13648. [CrossRef]
318. Kang, H.; Park, S.; Lee, B.; Ahn, J.; Kim, S. Modification of a Nile Red Staining Method for Microplastics Analysis: A Nile Red Plate Method. *Water* **2020**, *12*, 3251. [CrossRef]
319. Gaston, E.; Woo, M.; Steele, C.; Sukumaran, S.; Anderson, S. Microplastics Differ Between Indoor and Outdoor Air Masses: Insights from Multiple Microscopy Methodologies. *Appl. Spectrosc.* **2020**, *74*, 1079–1098. [CrossRef] [PubMed]
320. Parolini, M.; Antonioli, D.; Borgogno, F.; Gibellino, M.C.; Fresta, J.; Albonico, C.; De Felice, B.; Canuto, S.; Concedi, D.; Romani, A.; et al. Microplastic Contamination in Snow from Western Italian Alps. *Int. J. Environ. Res. Public Health* **2021**, *18*, 768. [CrossRef]
321. Vianello, A. A Journey into Microplastic Analysis Using FTIR Spectroscopy. Ph.D. Thesis, Aalborg University, Aalborg, Denmark, 2020.
322. Ly, N.H.; Kim, M.-K.; Lee, H.; Lee, C.; Son, S.J.; Zoh, K.-D.; Vasseghian, Y.; Joo, S.-W. Advanced microplastic monitoring using Raman spectroscopy with a combination of nanostructure-based substrates. *J. Nanostructure Chem.* **2022**, *12*, 865–888. [CrossRef] [PubMed]
323. Abbasi, S.; Jaafarzadeh, N.; Zahedi, A.; Ravanbakhsh, M.; Abbaszadeh, S.; Turner, A. Microplastics in the atmosphere of Ahvaz City, Iran. *J. Environ. Sci.* **2023**, *126*, 95–102. [CrossRef] [PubMed]
324. Käppler, 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. [CrossRef]
325. Primpke, S.; Fischer, M.; Lorenz, C.; Gerdts, G.; Scholz-Böttcher, B.M. Comparison of pyrolysis gas chromatography/mass spectrometry and hyperspectral FTIR imaging spectroscopy for the analysis of microplastics. *Anal. Bioanal. Chem.* **2020**, *412*, 8283–8298. [CrossRef]
326. Peters, C.A.; Hendrickson, E.; Minor, E.C.; Schreiner, K.; Halbur, J.; Bratton, S.P. Pyr-GC/MS analysis of microplastics extracted from the stomach content of benthivore fish from the Texas Gulf Coast. *Mar. Pollut. Bull.* **2018**, *137*, 91–95. [CrossRef]
327. Ainali, N.M.; Kalaronis, D.; Kontogiannis, A.; Evgenidou, E.; Kyzas, G.Z.; Yang, X.; Bikiaris, D.N.; Lambropoulou, D.A. Microplastics in the environment: Sampling, pretreatment, analysis and occurrence based on current and newly-exploited chromatographic approaches. *Sci. Total Environ.* **2021**, *794*, 148725. [CrossRef]
328. Hendrickson, E.; Minor, E.C.; Schreiner, K. Microplastic Abundance and Composition in Western Lake Superior As Determined via Microscopy, Pyr-GC/MS, and FTIR. *Environ. Sci. Technol.* **2018**, *52*, 1787–1796. [CrossRef]
329. Cowger, W.; Gray, A.; Christiansen, S.H.; DeFrond, H.; Deshpande, A.D.; Hemabessiere, L.; Lee, E.; Mill, L.; Munno, K.; Ossmann, B.E.; et al. Critical Review of Processing and Classification Techniques for Images and Spectra in Microplastic Research. *Appl. Spectrosc.* **2020**, *74*, 989–1010. [CrossRef] [PubMed]
330. Peñalver, R.; Costa-Gómez, I.; Arroyo-Manzanares, N.; Moreno, J.M.; López-García, I.; Moreno-Grau, S.; Córdoba, M.H. Assessing the level of airborne polystyrene microplastics using thermogravimetry-mass spectrometry: Results for an agricultural area. *Sci. Total Environ.* **2021**, *787*, 147656. [CrossRef]
331. Shiu, R.-F.; Chen, L.-Y.; Lee, H.-J.; Gong, G.-C.; Lee, C. New insights into the role of marine plastic-gels in microplastic transfer from water to the atmosphere via bubble bursting. *Water Res.* **2022**, *222*, 118856. [CrossRef] [PubMed]
332. 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.* **2020**, *389*, 121846. [CrossRef]
333. Yang, Z.; Lü, F.; Zhang, H.; Wang, W.; Xu, X.; Shao, L.; Che, Z.; Lu, B.; Ye, J.; He, P. A neglected transport of plastic debris to cities from farmland in remote arid regions. *Sci. Total Environ.* **2022**, *807*, 150982. [CrossRef]
334. Mbachu, O.; Jenkins, G.; Pratt, C.; Kaparaju, P. A New Contaminant Superhighway? A Review of Sources, Measurement Techniques and Fate of Atmospheric Microplastics. *Water Air. Soil Pollut.* **2020**, *231*, 85. [CrossRef]
335. Dos Santos Galvão, L.; Fernandes, E.M.S.; Ferreira, R.R.; Dos Santos Rosa, D.; Wiebeck, H. Critical steps for microplastics characterization from the atmosphere. *J. Hazard. Mater.* **2022**, *424*, 127668. [CrossRef]
336. Luo, X.; Wang, Z.; Yang, L.; Gao, T.; Zhang, Y. A review of analytical methods and models used in atmospheric microplastic research. *Sci. Total Environ.* **2022**, *828*, 154487. [CrossRef]

337. Revel, M.; Châtel, A.; Mouneyrac, C. Micro(nano)plastics: A threat to human health? *Curr. Opin. Environ. Sci. Health* **2018**, *1*, 17–23. [CrossRef]
338. Shen, M.; Zhang, Y.; Zhu, Y.; Song, B.; Zeng, G.; Hu, D.; Wen, X.; Ren, X. Recent advances in toxicological research of nanoplastics in the environment: A review. *Environ. Pollut.* **2019**, *252*, 511–521. [CrossRef]
339. Shuo, W.; Weiyu, D.; Huanming, L.; Rui, S.; Xu, Z. The Pollution of Atmospheric Microplastics and Their Potential Risks to Humans. *IOP Conf. Ser. Earth Environ. Sci.* **2021**, *793*, 012016. [CrossRef]
340. Bergmann, M.; Collard, F.; Fabres, J.; Gabrielsen, G.W.; Provencher, J.F.; Rochman, C.M.; Van Sebille, E.; Tekman, M.B. Plastic pollution in the Arctic. *Nat. Rev. Earth Environ.* **2022**, *3*, 323–337. [CrossRef]
341. Ershova, A.; Makeeva, I.; Malgina, E.; Sobolev, N.; Smolokurov, A. Combining citizen and conventional science for microplastics monitoring in the White Sea basin (Russian Arctic). *Mar. Pollut. Bull.* **2021**, *173*, 112955. [CrossRef]
342. Ren, X.; Tang, J.; Liu, X.; Liu, Q. Effects of microplastics on greenhouse gas emissions and the microbial community in fertilized soil. *Environ. Pollut.* **2020**, *256*, 113347. [CrossRef]
343. Zhang, Y.; Gao, T.; Kang, S.; Shi, H.; Mai, L.; Allen, D.; Allen, S. Current status and future perspectives of microplastic pollution in typical cryospheric regions. *Earth-Sci. Rev.* **2022**, *226*, 103924. [CrossRef]
344. Browne, M.A.; Niven, S.J.; Galloway, T.S.; Rowland, S.J.; Thompson, R.C. Microplastic Moves Pollutants and Additives to Worms, Reducing Functions Linked to Health and Biodiversity. *Curr. Biol.* **2013**, *23*, 2388–2392. [CrossRef]
345. Sulaeman, A.P.; Gao, Y.; Dugmore, T.; Remón, J.; Matharu, A.S. From unavoidable food waste to advanced biomaterials: Microfibrilated lignocellulose production by microwave-assisted hydrothermal treatment of cassava peel and almond hull. *Cellulose* **2021**, *28*, 7687–7705. [CrossRef]
346. Akanyange, S.N.; Zhang, Y.; Zhao, X.; Adom-Asamoah, G.; Ature, A.-R.A.; Anning, C.; Tianpeng, C.; Zhao, H.; Lyu, X.; Crittenden, J.C. A holistic assessment of microplastic ubiquitousness: Pathway for source identification in the environment. *Sustain. Prod. Consum.* **2022**, *33*, 113–145. [CrossRef]
347. Xu, Y.; Chan, F.K.S.; Stanton, T.; Johnson, M.F.; Kay, P.; He, J.; Wang, J.; Kong, C.; Wang, Z.; Liu, D.; et al. Synthesis of dominant plastic microfibre prevalence and pollution control feasibility in Chinese freshwater environments. *Sci. Total Environ.* **2021**, *783*, 146863. [CrossRef]
348. Pettipas, S.; Bernier, M.; Walker, T.R. A Canadian policy framework to mitigate plastic marine pollution. *Mar. Policy* **2016**, *68*, 117–122. [CrossRef]
349. Jiang, J.-Q. Occurrence of microplastics and its pollution in the environment: A review. *Sustain. Prod. Consum.* **2018**, *13*, 16–23. [CrossRef]
350. Pokharel, S.; Hollingsworth, J. Nepal Is Banning Single-Use Plastic in the Everest Region. 2019. Available online: https://edition.cnn.com/2019/08/22/asia/nepal-mount-everest-plastics-intl-hnk-trnds/index.html (accessed on 12 November 2024).
351. 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. [CrossRef] [PubMed]
352. Cui, J.; Chen, C.; Gan, Q.; Wang, T.; Li, W.; Zeng, W.; Xu, X.; Chen, G.; Wang, L.; Lu, Z.; et al. Indoor Microplastics and Bacteria in the Atmospheric Fallout in Urban Homes. *Sci. Total Environ.* **2022**, *852*, 158233. [CrossRef]
353. Villanova-Solano, C.; Hernández-Sánchez, C.; Díaz-Peña, F.J.; González-Sálamo, J.; González-Pleiter, M.; Hernández-Borges, J. Microplastics in Snow of a High Mountain National Park: El Teide, Tenerife (Canary Islands, Spain). *Sci. Total Environ.* **2023**, *873*, 162276. [CrossRef]
354. Materić, D.; Kasper-Giebl, A.; Kau, D.; Anten, M.; Greilinger, M.; Ludewig, E.; Van Sebille, E.; Röckmann, T.; Holzinger, R. Micro- and Nanoplastics in Alpine Snow: A New Method for Chemical Identification and (Semi)Quantification in the Nanogram Range. *Environ. Sci. Technol.* **2020**, *54*, 2353–2359. [CrossRef]
355. Vijayan, A.; Österlund, H.; Magnusson, K.; Marsalek, J.; Viklander, M. Microplastics (MPs) in Urban Roadside Snowbanks: Quantities, Size Fractions and Dynamics of Release. *Sci. Total Environ.* **2022**, *851*, 158306. [CrossRef]
356. Liu, X.; Lu, J.; He, S.; Tong, Y.; Liu, Z.; Li, W.; Xiayihazi, N. Evaluation of Microplastic Pollution in Shihezi City, China, Using Pine Needles as a Biological Passive Sampler. *Sci. Total Environ.* **2022**, *821*, 153181. [CrossRef]
357. Wang, X.; Wei, N.; Liu, K.; Zhu, L.; Li, C.; Zong, C.; Li, D. Exponential Decrease of Airborne Microplastics: From Megacity to Open Ocean. *Sci. Total Environ.* **2022**, *849*, 157702. [CrossRef]
358. Capozzi, F.; Sorrentino, M.C.; Cascone, E.; Iuliano, M.; De Tommaso, G.; Granata, A.; Giordano, S.; Spagnuolo, V. Biomonitoring of Airborne Microplastic Deposition in Semi-Natural and Rural Sites Using the Moss Hypnum Cupressiforme. *Plants* **2023**, *12*, 977. [CrossRef]
359. Fan, W.; Salmond, J.A.; Dirks, K.N.; Cabedo Sanz, P.; Miskelly, G.M.; Rindelaub, J.D. Evidence and Mass Quantification of Atmospheric Microplastics in a Coastal New Zealand City. *Environ. Sci. Technol.* **2022**, *56*, 17556–17568. [CrossRef]
360. Boakes, L.C.; Patmore, I.R.; Bancone, C.E.P.; Rose, N.L. High Temporal Resolution Records of Outdoor and Indoor Airborne Microplastics. *Environ. Sci. Pollut. Res.* **2023**, *30*, 39246–39257. [CrossRef] [PubMed]
361. Wang, F.; Lai, Z.; Peng, G.; Luo, L.; Liu, K.; Huang, X.; Xu, Y.; Shen, Q.; Li, D. Microplastic Abundance and Distribution in a Central Asian Desert. *Sci. Total Environ.* **2021**, *800*, 149529. [CrossRef] [PubMed]
362. Cabrera, M.; Moulatlet, G.M.; Valencia, B.G.; Maisincho, L.; Rodríguez-Barroso, R.; Albendín, G.; Sakali, A.; Lucas-Solis, O.; Conicelli, B.; Capparelli, M.V. Microplastics in a Tropical Andean Glacier: A Transportation Process across the Amazon Basin? *Sci. Total Environ.* **2022**, *805*, 150334. [CrossRef] [PubMed]

363. Shahraki, M.; Rezaei Kahkha, M.R.; Piri, J.; Sharafi, A.; Kaykhaii, M. Microplastics in Atmospheric Dust Samples of Sistan: Sources and Distribution. *J. Environ. Health Sci. Eng.* **2022**, *20*, 931–936. [CrossRef] [PubMed]
364. Kyriakoudes, G.; Turner, A. Suspended and Deposited Microplastics in the Coastal Atmosphere of Southwest England. *Chemosphere* **2023**, *343*, 140258. [CrossRef] [PubMed]
365. Chernykh, D.V.; Biryukov, R.Y.; Kuryatnikova, N.A.; Malygina, N.S. Microplastics in the Snow Cover of Urban Landscapes: A Case Study of Barnaul. *Geogr. Nat. Resour.* **2022**, *43*, S44–S49. [CrossRef]
366. Napper, I.E.; Davies, B.F.R.; Clifford, H.; Elvin, S.; Koldewey, H.J.; Mayewski, P.A.; Miner, K.R.; Potocki, M.; Elmore, A.C.; Gajurel, A.P.; et al. Reaching New Heights in Plastic Pollution—Preliminary Findings of Microplastics on Mount Everest. *One Earth* **2020**, *3*, 621–630. [CrossRef]