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Understanding the atmospheric cycle of microplastics: from emission to deposition

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Atmospheric microplastic (AMP) has recently emerged as a potential vector in the global carbon cycle, and a novel factor influencing the climate system. Previous studies on microplastics primarily focused on terrestrial and aquatic ecosystems, while research on AMP remains insufficient. Given the unique mobility of AMP and its function as a vector linking land, ocean, and atmosphere, this review provides a comprehensive synthesis of current knowledge on AMPs, with a focus on their sources, spatiotemporal distribution, transport and deposition processes, and detection and quantification techniques. Previous literature indicates that AMP originates from diverse sources, and its atmospheric distribution exhibits pronounced spatial heterogeneity, with transmission and deposition processes driven by meteorological conditions, particle properties, and surface–atmosphere interactions. However, substantial knowledge gaps remain regarding standardized monitoring methods, long-range transport mechanisms, and the implications of AMPs for atmospheric processes and climate forcing. In particular, limitations in detection and quantification techniques hinder accurate assessment of AMP in the atmosphere. Here, this review provides theoretical support for a comprehensive understanding of the global cycle of AMP, which offers a scientific basis for relevant environmental risk assessment and climate management decisions.

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Environmental significance

Atmospheric microplastics represent a critical yet poorly understood pathway within the global plastic cycle. This review systematically summarizes the sources, global transport, deposition, and occurrence characteristics of atmospheric microplastics, as well as recent advances in their sampling and analytical methods, emphasizing the urgent need for a better understanding and effective management of atmospheric microplastic pollution.

1. Introduction

Microplastics (MPs), defined as plastic particles < 5 mm, have become ubiquitous contaminants in terrestrial, marine, freshwater, and atmospheric ecosystems.[1] Previous MP research has primarily focused on marine and terrestrial environments.[2] Since the first detection of MPs in the atmosphere in 2015,[3] atmospheric microplastics (AMPs) have emerged as an increasingly critical issue in the climate system. AMPs have been increasingly identified in atmospheric samples from diverse geographical regions, including remote areas previously considered pristine.[4–6] Moreover, indoor AMPs have been detected in various microenvironments (*e.g.*, homes, offices, and hospitals) across multiple countries including France, USA, and China, with concentration levels varying significantly (5–1583 particles per m^3), and typically measuring 100–500 μm in length.[7–9] Previous studies indicate that AMPs originate from diverse and complex sources.[10–13] In addition to direct anthropogenic emissions, oceanic MPs can be aerosolized into the atmosphere through wave-breaking and wind-driven processes.[14,15] The marine-derived AMPs are mainly transferred from seawater to air *via* bubble bursting and sea spray formation under windy conditions. Previous studies estimate that approximately 12.5% of the total MPs released into the continental Asian environment ultimately enter the atmosphere. This phenomenon warrants further investigation, as AMPs constitute a significant yet understudied exposure pathway for both humans and wildlife.[16,17] These persistent AMPs can be directly inhaled and deposited in the respiratory tract, potentially inducing adverse health effects.[18]

Once airborne, these particles undergo intricate dynamic processes including horizontal transport, vertical migration, and aging degradation, exerting profound impacts on regional and even global climate.[14,19–21] The atmosphere serves as a crucial reservoir and transmission medium, facilitating the long-range dispersal of AMPs across continents and oceans.[22] MPs with small particles (<10 μm) can remain airborne for long

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periods of time, and different types of AMPs have already been detected in remote areas, including the Antarctic, Arctic, and Alpine regions.[17,23–27] The life cycle of AMPs, encompassing source emission, atmospheric circulation, and final environmental sinks, is profoundly influenced by complex interactions between meteorological parameters and anthropogenic activities. A thorough understanding of these processes is crucial for developing accurate global plastic budget models, predicting environmental persistence, and conducting comprehensive ecological and health risk assessments.

Based on published literature retrieved from the Web of Science Core Collection, studies related to AMPs were identified by searching for "microplastic*" in the title, combined with "atmosphere*", "airborne", "air", "source", or "model" in the topic, covering the period from 2015 to 2025. The retrieved records were initially screened based on titles and abstracts to exclude irrelevant studies, and the literature database was further supplemented through backward reference screening of the selected articles. Building on this body of literature, this review synthesizes current knowledge of AMPs, with a focus on their sources, sampling and pretreatment methods, deposition processes, spatial distribution, and environmental and health impacts. By integrating these perspectives across the environmental lifecycle of AMPs, this review emphasizes the atmosphere as an important and previously underexplored pathway for MP transport, while highlighting key uncertainties and critical research gaps that warrant further investigation.

2. Sources of atmospheric microplastics

2.1 Primary sources of atmospheric microplastics

AMPs originate from diverse sources that can be broadly categorized into primary and secondary types. Primary sources are largely associated with anthropogenic activities, including urban traffic emissions, domestic activities, waste management, industrial processes, and agricultural practices.[10–13,28]

Synthetic textiles constitute a dominant anthropogenic source of fibrous AMPs in urban areas.[7,29,30] Microfiber release occurs *via* multiple pathways, such as mechanical abrasion during daily wear, hydrodynamic stresses in laundry, and environmental weathering of fiber-containing materials. Studies indicate that laundry processes release >1900 fibers per garment, with ~35% potentially becoming airborne during natural sunlight drying or mechanical agitation.[30–32] It has been estimated that a single person could directly release up to one billion microfibers/year from polyester garments into the atmosphere.[33] Regions characterized by high population density, dense transportation networks, and textile or plastic manufacturing hubs tend to emit substantially higher levels of AMPs. Indoor environments also contribute significantly, with MPs being released through the wear and tear of synthetic furniture and textiles.[28,30]

Traffic-related emissions represent a significant source of AMPs.[34] Tire wear particles (TWPs) represent a predominant source of AMPs in urban environments, accounting for approximately 10–20% of total tire mass loss during their lifetime. A regional atmospheric model was developed for MPs, and the results have shown that Asian land areas alone emit about 310 Gg of AMPs for 2018, largely attributed to tire dust.[16] Moreover, emissions from tire wear are substantially higher than those of other sources of MPs, accounting for approximately 3–7% of $PM_{2.5}$ and 0.1–10% of PM_{10} in urban air.[35,36]

Industrial processes, including waste glass recycling, electronic waste dismantling, and textile manufacturing, have been quantitatively identified as important contributors to AMP emissions.[37] Among these, the textile industry represents a particularly major contributor, due to the production and processing of synthetic fibers such as polyester (PET), polyolefins, polyamide (nylon), polyurethane (PU), polymethyl methacrylate (PMMA), and polyvinyl chloride (PVC).[38] The manufacturing processes, such as cutting, grinding, and weaving, generate a substantial number of microscale synthetic fibers that can be transported into the atmosphere by wind.[39,40]

Agricultural activities serve as a significant emission source of AMPs through the environmental degradation of plastic films, particularly low-density polyethylene (PE) mulch films that are widely used in modern farming practices. Due to their thin structure and the lack of plastic recycling facilities, residual plastic fragments can degrade into continuous macroplastics, microplastics, and nanoplastics.[41] Under open-field conditions characterized by high temperatures, solar radiation, and oxidative processes, these particles can become suspended in the atmosphere and transported *via* air currents, eventually depositing on leaf surfaces.[12,42] Additional sources include the shredding of rubber granules used in artificial turf infill, building materials, and billboards, which release large amounts of MPs into the atmosphere. Table 1 summarizes the major primary sources of AMPs reported in the literature. It is observed that urban environments such as London, São Paulo, Dongguan, and Espírito Santo consistently report fibers from clothing and textiles, tyre wear, road dust, and packaging waste as dominant contributors. These reflect intense human activity and traffic-related emissions. Coastal and beach environments, such as Gdynia, highlight fishing gear, pellets, and marine litter as prominent sources. Overall, the table shows that although sources differ by location, textile fibers, tyre wear, and road emissions are the most common global sources of AMPs.

2.2 Secondary sources of atmospheric microplastics

Secondary sources of AMPs primarily involve the resuspension of marine and terrestrial MPs through environmental processes. In marine environments, wind-driven processes such as wave breaking and bubble bursting facilitate the transfer of MPs from ocean surfaces back into the troposphere, completing a dynamic exchange cycle between environmental and climate compartments.[14,15,53] A landmark study published in 2020 first quantified oceanic MP emissions through sea breeze transport along the French coast, revealing a global onshore flux of ~0.14 megatons per year for submicron particles.[54] *In situ* aerosol measurements show that marine-derived MPs have significant potential to be transferred to the atmosphere through the

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Table 1 Sources of atmospheric microplastics summarized from different regions of published literature

No.	Country	Region	Environment type	Period	MPs sources	Ref.
1	The United Kingdom	Central London	Urban	2018	Textiles, packaging, expanded polystyrene (EPS), tyre wear, and road paint	43
2	Brazil	São Paulo	Urban	2019–2020	Synthetic fabrics, tyre wear, road dust, and polyester fibers	44
3	China	S. China Agric. Univ.	Urban	2018–2019	Textile ultraviolet (UV) degradation, alkyd resins, and microbeads	45
4	Germany	Weser river	Mixed	2018	Industrial emissions, tyre wear, and textile shedding	1
5	Canada	Whitehorse	Suburban	2020–2021	Recycling/landfill and construction degradation	46
6	Poland	Gdynia	Coastal	2017–2018	Textiles, packaging, fishing gear, and urban litter	47
7	China	Dongguan	Urban	2016	Fibers (clothes and textiles); films (disposable plastic bags); fragments (thicker plastic products that might be recycled); foams (expanded polystyrene products)	48
8	Thailand	Pathum Thani	Peri-urban	2021–2022	Fibers (the synthetic textile (clothing, upholstery, or carpet)); textiles (fiber abrasion and shedding); fragments (the exposure of larger plastic items to strain, fatigue, or ultraviolet lights)	49
9	Iran	Asaluyeh	Industrial	2017	Industrial emissions, textile shedding, and tyre wear	12
10	China	N. Taiwan coast	Beach	2015	Plastic degradation, pellets, and textiles	50
11	Germany	Hamburg	Mixed	2017–2018	Road dust, tyre abrasion, and polyester (PET) fibers	51
12	Brazil	Espírito Santo	Urban	2023	Polymer recycling, traffic, and textile industry	52

aerosolization of sea spray.[55] Regional modeling suggests that marine contributions may account for 11% of AMP loading in the western United States.[14,53]

On land, MPs deposited on terrestrial surfaces, including agricultural lands, landfills, and urban soils, can become resuspended into the atmosphere during high-wind events, particularly pronounced in arid and semi-arid regions. Sparse vegetation cover and dry surface conditions facilitate repeated deposition–resuspension cycles, while desert dust storms and sand movement can further mobilize MPs.[56,57] Such dynamic processes significantly prolong the atmospheric residence time of AMPs in these environments.[7,58] Soil MPs beneath tree canopies may follow distinct resuspension pathways, leading to redeposition on leaves. Polymer analyses reveal clear compositional differences between soil- and leaf-deposited MPs, with soils largely dominated by PE and rayon, whereas leaf surfaces contain a more diverse polymer spectrum (*e.g.*, PS, PP, and PVC).[59] These distinctions highlight compartment-specific exchange dynamics and the contrasting environmental behaviors of MPs. Nevertheless, current estimates of resuspension fluxes from both marine and terrestrial environments vary widely—from negligible to dominant contributions—reflecting substantial uncertainties and the limitations of existing technologies for detecting and quantifying these processes.[60–62] Overall, the resuspension of MPs is governed by multiple environmental factors, including wind velocity, soil moisture, vegetation coverage, and surface roughness. Current research on the mechanistic understanding of AMP resuspension remains limited, with significant knowledge gaps persisting in the development of quantitative models across different climatic zones.

3. Sampling and pretreatment of atmospheric microplastics

3.1 Sampling and pretreatment methods

The analysis of AMPs typically involves a multi-step process comprising sampling, pretreatment, and characterization. Sampling provides representative airborne particles from different environmental compartments, while pretreatment procedures aim to isolate suspected MPs from complex matrices by removing interfering organic and inorganic substances, and characterization identifies polymer types, morphology, and abundance.

Sampling strategies for AMPs are broadly categorized into active and passive methods, depending on whether the target is suspended or depositional particles. Active sampling collects suspended aerosols by drawing air through filter membranes using pump-driven devices (*e.g.*, vacuum pumps, water pumps, or total suspended particulate samplers), capturing particles larger than the filter pore size.[63–66] This method, widely used in atmospheric pollutant monitoring, enables efficient short-term collection and is particularly suitable for mobile monitoring,

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although it requires an external power source. Concentrations from active sampling are typically expressed as N m^{-3}. In contrast, passive sampling captures depositional particles, including those settled *via* precipitation (rain, snow, and ice) or dry deposition, using gravity-driven collectors.[67,68] Devices include rain samplers,[69] funnel-based bottles,[70] brushes,[71] vacuum cleaners,[72] and specialized instruments such as the Aerochem Metrics wet/dry collector, which separates wet and dry inputs *via* precipitation sensors.[73] Passive sampling has been widely accepted as a standard approach for quantifying total (wet + dry) atmospheric deposition of MPs. Recently, a standardized metallic/glass system has been designed by the Norwegian Institute of Air Research, and this system has since been widely used for passive sampling of atmospheric deposition.[68,74] Passive sampling integrates material over longer periods without external energy input, but accurate flux calculation (N $perm^2$ per day) requires careful consideration of the deposition area.

The pretreatment process aims to remove impurities, concentrate MPs, and prepare them for subsequent analysis. The standard workflow typically involves sieving, digestion, density separation, filtration, and drying. Initially, bulk impurities are removed and samples are fractionated using stainless steel sieves (5 mm to 150 μm).[75] Digestion is commonly performed at controlled temperatures using reagents such as H_2O_2, Fenton's reagent, or NaOH to selectively remove organic matter, minimizing damage to AMPs. Density separation is carried out with solutions such as NaCl, NaI, or $ZnCl_2$ to distinguish components of different densities, after which the supernatant is collected on a 0.45 μm membrane.[76] As no universal pretreatment protocol exists, laboratories often adapt the sequence (*e.g.*, density separation before digestion) based on matrix composition.[77,78] In some studies, visual inspection or microscopic examination has been employed to distinguish plastic from non-plastic particles, allowing researchers to bypass density separation.[79–81] Glass microfiber membranes are commonly employed for efficient enrichment, and drying at low temperatures preserves particle morphology.

Characterization generally combines physical observation with chemical identification. Stereomicroscopy provides an initial screening based on morphology (shape, color, size, luster, and surface features),[12] providing a preliminary screening of potential particles. Subsequently, polymer composition is commonly determined using Fourier transform infrared (FTIR) spectroscopy or Raman spectroscopy. FTIR is a mature, non-destructive technique with extensive spectral libraries, but it often struggles to effectively detect particles <10 μm, whereas Raman spectroscopy offers higher spatial resolution (~1 μm) but may induce thermal damage due to a high-energy laser.[6,82] Given time and instrumental constraints, subsampling strategies are often performed. For example, as fibers represent the predominant suspected MP morphology in atmospheric deposition samples, only a random subset (*e.g.*, 20%) of fibers is selected for Raman analysis, while all other suspected particle types are subjected to complete identification.[48] While this practice improves analytical efficiency, it may introduce uncertainty through false-positive visual identification, morphology- or size-dependent selection bias, and extrapolation errors when inferring polymer composition or abundance for the entire particle population. Existing studies have implemented partial confirmation using different strategies, including random subsampling of suspected particles, stratification by particle morphology or size class, and targeted confirmation of representative particles. These approaches can influence reported polymer distributions and concentration estimates if not transparently documented. To improve comparability and uncertainty assessment, best-practice recommendations emerging from the literature emphasize transparent reporting of confirmation strategies and confirmation proportions, preferential use of randomized or stratified subsampling schemes, and reporting of false-positive or misidentification rates where available. Explicit discussion of the limitations associated with partial spectral confirmation is essential for robust interpretation of AMP data and for guiding future methodological standardization. Moreover, to address the challenge of detecting small particles, emerging tools such as surface-enhanced Raman spectroscopy and liquid chromatography-tandem mass spectrometry are being applied. Scanning electron microscopy is often used for detailed examination of surface morphology and degradation features. All spectroscopic analyses require pretreatment steps such as density separation and chemical digestion to reduce matrix interference and the risk of misidentification. A systematic comparison of the advantages and limitations of the various analytical methods is provided in Table S4.

3.2 Quality assurance and quality control

To minimize exogenous contamination, AMP studies have generally implemented multiple layers of quality assurance and quality control (QA/QC) throughout sampling, pretreatment, transport, and analysis. Across the reviewed literature, non-plastic materials (*e.g.*, glass, stainless steel, aluminum, and cellulose filters) were preferentially used for sampling devices, storage containers, and pretreatment tools to reduce contamination from experimental equipment.[63] Samples were typically sealed in glassware and wrapped in aluminum foil immediately after collection to limit exposure to airborne particles during transport and storage. Different types of blanks were employed to identify and quantify contamination sources at each stage of the analytical workflow. Field blanks were commonly generated by exposing clean filters or sampling devices to the ambient environment without active air collection, allowing assessment of contamination from surrounding air, sampling platforms, or nearby infrastructure. Procedural blanks were processed in parallel with real samples and subjected to the complete pretreatment sequence, including sieving, rinsing, chemical digestion, density separation, filtration, and drying, in order to evaluate contamination introduced by reagents, laboratory ware, and handling steps.[83,84] In addition, transport blanks were used in some studies to assess contamination during sample storage and transfer.

During microscopic observation and spectroscopic identification, blank filters that had not contacted samples were

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analyzed under identical conditions to those applied to real samples, including the same optical settings, laser parameters, spectral ranges, and library matching criteria.[47] This approach enabled consistent identification of background contamination originating from filters, laboratory air, or analytical instruments. Reported studies varied in how blank corrections were applied, with some subtracting blank-derived particle counts or polymer types from sample results, while others reported blanks qualitatively or did not explicitly describe correction procedures, contributing to inter-study variability and uncertainty.

4. Transportation process of atmospheric microplastics

Similar to atmospheric particulate matter, AMPs suspended in the atmosphere undergo horizontal advection and vertical turbulent transport, enabling long-range transport and cross-layer migration, as shown in Fig. 1(a). Reported atmospheric transport durations of AMPs range from 1 hour to 6.5 days,[85–87] while a residence time of 9.47–22.85 days has been estimated using natural radionuclides, underscoring their potential for long-distance dispersal before deposition.[88] It is estimated that approximately 1.21 tons of terrestrial MPs are delivered to the oceans annually *via* atmospheric pathways, with air masses capable of carrying MPs over distances exceeding 1000 km.[22,29,89] Field studies in the French Pyrenees detected MPs up to 95 km from potential sources, while UK trajectory analyses demonstrated cross-regional transport.[86,87] Most investigations rely on the Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model—often in combination with backward trajectories, settling calculations, or footprint analyses—to trace potential source regions. The identified sources span adjacent continents, deserts such as the Sahara and Taklamakan, marine environments, and densely populated urban or industrial areas. Notably, even pristine or high-altitude sites receive AMPs transported from distant anthropogenic regions, underscoring the role of atmospheric circulation in redistributing MPs globally. This highlights both the persistence and mobility of AMPs, as well as the need for improved modeling frameworks to better quantify their transport pathways and deposition processes.

From a process-oriented perspective, available field observations and trajectory analyses consistently suggest that the atmospheric transport potential of AMPs is governed by a limited set of key variables that are directly relevant for regional and global modeling. These include the effective aerodynamic diameter (or equivalent diameter), particle density, and morphology (*e.g.*, fibers *versus* fragments), which together determine settling velocity and long-range transport efficiency. Meteorological controls such as boundary layer height, wind speed, and atmospheric stability further regulate vertical mixing and residence time, while wet scavenging (in-cloud and below-cloud processes) and dry deposition determine removal rates and surface loading,[14,32,53,90] facilitating their widespread environmental redistribution,[61,85,89] as shown in Fig. 1(a). To fully characterize the environmental cycling of AMPs, it is thus necessary to examine the mechanisms of their deposition, with wet and dry deposition representing the two dominant pathways.

4.1 Wet deposition process of atmospheric microplastics

Currently, most AMPs are collected through bulk deposition, while only a limited number of studies have separately examined dry and wet deposition. Wet deposition, however, is the predominant removal mechanism for AMPs, occurring primarily through rainfall and snowfall (Fig. 1(a)). Rainfall efficiently washes AMPs from the atmosphere, with deposition fluxes often exceeding those of dry deposition within the same sampling period.[30] For instance, in Pathum Thani, Thailand, wet deposition fluxes reached 285 particles per m^2 per day, surpassing dry deposition rates.[49] In Paris, lower fibrous MP content was observed during periods of reduced rainfall.[30] The process also exhibits clear precipitation dependence, and wet deposition fluxes decline sharply within the first 30 minutes of rainfall, indicating rapid washout,[91] and continuous rainfall both enhances deposition and suppresses resuspension of surface MPs.[76,92] However, the correlation between rainfall amount and AMP abundance remains inconsistent. Some

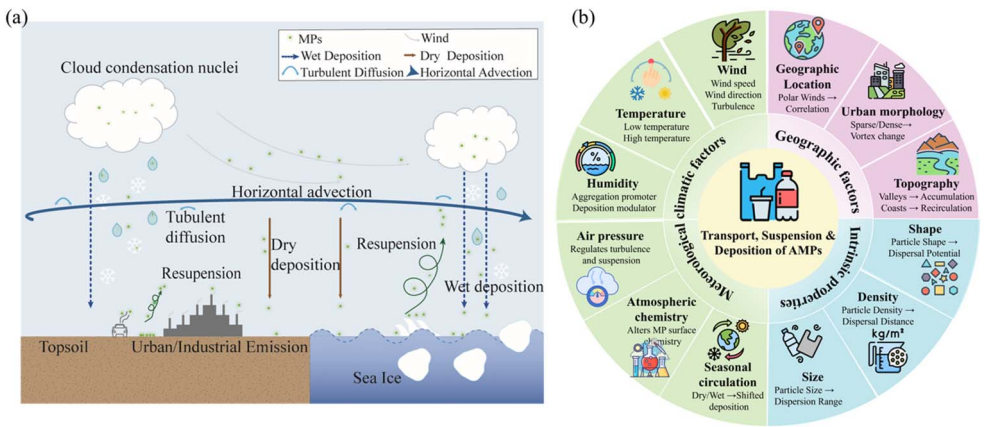

Fig. 1 Schematic of atmospheric microplastic transport (a) and its influencing factors (b).

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studies found no significant association between atmospheric deposition flux of AMPs and cumulative rainfall,[92] suggesting that precipitation intensity, frequency, and timing may play more critical roles than total rainfall.[76] Interestingly, fibrous MPs have been observed to peak during periods of low rainfall and fewer rainy days, indicating that additional meteorological factors regulate AMP deposition. Snowfall, in contrast, is particularly effective at capturing AMPs due to its larger surface area and slower terminal velocity compared with raindrops, allowing it to scavenge a broader diversity of contaminants.[93,94] Together, rainfall and snowfall not only remove AMPs from the atmosphere but also determine where these particles accumulate and their subsequent environmental mobility, influencing terrestrial and aquatic sinks as well as long-term transport patterns.

4.2 Dry deposition process of atmospheric microplastics

Dry deposition of AMPs is regulated by both particle-specific properties and environmental conditions. Gravitational settling predominantly drives the deposition of larger or denser particles, whereas smaller MPs can remain airborne for extended periods.[84] Finer particulates are more effectively removed through turbulent scavenging within the turbulence-dominated atmospheric boundary layer,[95] while Brownian diffusion governs the deposition of ultrafine particles (<0.1 μm).[96] Meteorological factors further influence deposition dynamics. During dry periods, wind speed is negatively correlated with MP deposition rates, as strong wind inhibits particle settling and promote resuspension.[97] In addition, atmospheric chemistry plays an important role, as ozone concentrations have been reported to be negatively correlated with MP deposition rates, dust loads, and polymer diversity, likely due to ozone-induced chemical modification of MP surfaces that reduces their detectability.[98]

In general, dry deposition fluxes are lower than those of wet deposition. For instance, in central Germany, wet deposition samples contained 1.63 ± 0.92 MPs per sample compared to 0.88 ± 1.1 MPs per sample for dry deposition.[99] Similar patterns were reported in Guangzhou, China, where AMP fluxes were significantly higher during the wet season (84.00 ± 6.95 particles per m^2 per day) than in the dry season (47.88 ± 8.35 particles per m^2 per day), and AMP abundance in the atmosphere was likewise greater during wet periods (0.19 ± 0.01 particles per m^3) than during dry periods (0.15 ± 0.02 particles per m^3).[100] Similar trends have also been reported in Jakarta and Shanghai, where wet deposition fluxes were consistently greater than dry deposition fluxes.[101,102] However, in semi-arid regions, the opposite pattern is observed, where deposition is greater during the dry season, with both dust and MP settling exceeding that in the wet season. This is likely because precipitation suppresses the resuspension of ground dust and MPs, leading to a gradual reduction in deposition on consecutive rainy days. These findings suggest that regional geographic and climatic conditions significantly influence the distribution characteristics of dry and wet deposition.[103] These observations also highlight resuspension as a key but poorly constrained process, which can substantially alter near-surface concentrations and deposition fluxes and remains a major source of uncertainty in both observational interpretation and transport modeling of AMPs.

4.3 Factors influencing atmospheric microplastics transport

The transport, suspension, and deposition of AMPs are governed by a combination of meteorological factors (*e.g.* wind speed, wind direction, boundary layer height, temperature, humidity and pollution concentration gradient), geographic, and anthropogenic factors, as well as their own physical–chemical properties (as shown in Fig. 1(b)). This multi-factor dependence results in highly nonlinear transport dynamics, where small perturbations in any parameter (*e.g.*, an increase in wind speed) can disproportionately alter AMP spatial distribution and deposition fluxes.

4.3.1 Meteorological factors. Wind-driven uplift is a key driver of long-range transport, particularly for low-density and small-sized MPs.[104] The horizontal transport of MPs is strongly controlled by wind speed and direction. In particular, wind events exceeding 2 m s^{-1} are critical for long-distance transport.[24,105] Shifts in wind direction may also alter MP distribution, often leading to higher abundances in downwind areas; however, this effect becomes less pronounced when sampling sites are geographically close.[32,106] Wind not only facilitates the horizontal diffusion of MPs from sources such as urban and industrial areas but also exerts a strong influence on their deposition processes.[38] Indeed, deposition fluxes of AMPs show a clear dependence on wind dynamics,[32,105,106] with evidence from the Antarctic plateau revealing a significant negative correlation between MP abundance and the east–west wind component.[61] Temperature also plays a critical regulatory role in AMP dynamics. Low temperatures favor the persistence of inversion layers and a shallow boundary layer, which trap MPs near the surface and markedly inhibit vertical dispersion. Such meteorological conditions lead to particle accumulation and increasing concentrations of AMPs in urban regions.[43,45] Additionally, cold conditions weaken atmospheric convection, restricting MP diffusion. In contrast, high temperatures enhance convective activity, facilitating the vertical mixing and dispersion of MPs emitted from the ground. Increased ambient temperatures are also associated with intensified outdoor activities and increased plastic use, indirectly amplifying MP emissions. Other meteorological parameters also influence AMP behavior. Humidity influences dry deposition rates and promotes particle aggregation, while air pressure regulates atmospheric turbulence and suspension. Ozone may alter MP surface chemistry, thereby affecting detectability as well as transport and deposition pathways. Moreover, seasonal transitions between dry and wet periods, such as those driven by the East Asian monsoon, strongly shape transport modes and deposition processes.[38]

4.3.2 Geographic and topographic modulation. Topography strongly influences the transport and deposition of AMPs by constraining air mass exchange and limiting long-range dispersion, thereby promoting local accumulation. For

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example, in Dongguan, China, MPs are primarily deposited through atmospheric fallout and dust cycles,[48] whereas in the valley–basin city of Lanzhou, most MPs originate from surrounding urban emissions.[107] Coastal areas are subject to land–sea breeze systems, which facilitate MP dispersal patterns.[38] Regions downwind from landfills or industrial areas often show increased MP concentrations due to directional airflow.[76]

Cities, as major MP sources, represent the initial stage of AMP transport, where complex urban structures fundamentally reshape dispersion mechanisms. Vertical transport within urban areas is driven by three principal processes: (1) turbulent flux dominates overall particle exchange, (2) dispersive flux becomes significant due to building-induced flow heterogeneity, and (3) upward mean flow enables uplift above low-rise structures, sustaining particle suspension.[108,109] Within the urban canopy sublayer, reduced wind speeds and wake-dominated turbulence promote gravitational settling of particles, thereby significantly reducing their escape fraction compared with open environments. This suppression effect exhibits positive correlations with both building height and plan area fraction.[108] Urban morphology further regulates distinct transport dynamics through the building-height-to-street-width ratio. In sparse building arrays, wake-interference flows dominate, dispersing MPs *via* turbulence generated in building wakes. In contrast, dense urban cores promote skimming flow, where street-canyon vortices trap and recirculate MPs within the lower canopy layer.[109] These flow–topography interactions generate deposition hotspots, particularly in areas with 2.3–3.6% built-up coverage, where the highest deposition rates are observed.[38] Source location within urban areas further facilitates MP transport, particularly for ground-level emissions. These emissions tend to become trapped within street-canyon recirculation layers, where mean vertical velocities are negligible, making turbulent diffusion the sole mechanism for vertical transfer.[110,111] Ultimately, the escape of MPs into the broader atmosphere depends on the interplay between urban form and particle properties, while complex terrain further amplifies spatial heterogeneity in deposition patterns.[112]

4.3.3 Intrinsic properties of microplastics. The atmospheric transport of MPs is strongly governed by their intrinsic properties, particularly shape, density, and size. Shape-dependent transport efficiency is well documented. Fibrous MPs, owing to their low density and reduced settling velocities (*e.g.*, ~0.06 m s^{-1} for fibers *vs.* ~0.32 m s^{-1} for non-fibrous MPs), exhibit greater atmospheric mobility and can be transported over significantly longer distances. For instance, fibers may travel up to ~60 km at wind speeds of 5 m s^{-1}, compared with ~12 km for non-fibrous particles under the same conditions.[43,89] Wind tunnel experiments further confirm that fibers are more easily entrained and carried by wind than spherical MPs.[113] Consequently, fibrous MPs may influence larger areas (640–8700 km^2) compared with non-fibrous ones (186–875 km^2).[43] Particle size also plays a critical role. Smaller MPs disperse more widely than larger ones, as highlighted by global transport simulations showing that fine particles such as $PM_{2.5}$ from road traffic can achieve broader atmospheric spread than coarser fractions like PM_{10}.[23] Together, these findings underscore that MPs with smaller size, lower density, and fibrous morphology are most conducive to long-range atmospheric transport and deposition.

Overall, the transport and deposition of AMPs are jointly regulated by external environmental factors and the intrinsic characteristics of the particles themselves. Meteorological factors such as wind, temperature, humidity, and precipitation determine the dynamic pathways of AMPs by influencing dispersion, vertical mixing, and deposition. Geographic and topographic features—including valleys, basins, coastlines, and complex urban morphologies—further constrain or enhance transport efficiency, shaping localized accumulation and deposition hotspots. In addition, the intrinsic properties of MPs, particularly size, density, and shape, influence their settling velocities and atmospheric residence times. Fibrous, low-density, and small-sized MPs show the highest atmospheric mobility, enabling long-range transport across regional to continental scales. These coupled influences highlight the complex interactions between particle characteristics and environmental conditions, underscoring the need to integrate both aspects when assessing the atmospheric fate of MPs.

5. Spatial heterogeneity of atmospheric microplastics

Concentrations and spatiotemporal distributions of AMPs vary widely, with significant regional variations observed in deposition flux, polymer composition, morphological characteristics, and size distribution. Tables S1–S3 summarize the distribution of AMPs in different regions extracted from the published literature. Reported deposition fluxes span several orders of magnitude, from a few tens to hundreds. For instance, values in Germany range from 29 ± 26 to 183 ± 124 particles per m^2 per day,[1] while studies in Iran document dry deposition fluxes of 0–46.5 particles per m^2 per day and wet deposition fluxes of up to 129.3 particles per m^2 per day.[82] Certain regions in Colombia exhibit exceptionally high MP deposition fluxes, reaching up to 9472 ± 702 particles per m^2 per day, which are attributed to the presence of multiple plastic manufacturing facilities with annual production capacities of several thousand tons, together with substantial inputs from the degradation of urban plastic waste.[114] Similarly high AMP deposition fluxes have been documented worldwide and are generally associated with intensive anthropogenic activities, strong local emission sources, and favorable environmental conditions. Studies in China have reported deposition rates ranging from 302.31 ± 107.40 particles per m^2 per day to 5094.93 ± 3431.01 particles per m^2 per day,[102,115] and even higher fluxes have been documented in highly polluted environments such as textile manufacturing facilities in Bangladesh, where MP fluxes reach 179 000 particles per m^2 per day.[116] These observations indicate that the high deposition reported in Colombia falls within the upper range of globally observed AMP fluxes. However, these data are derived from different studies (as shown in Tables S1–S3) with variations in sampling regions, sampling periods, and analytical

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methods. Therefore, the abundance data do not represent the current status of MP pollution in the region, but rather reflect the results obtained under specific sampling conditions and analytical methodologies.

In terms of polymer composition, synthetic fibers such as PE, PS, PET, polyester (PES), polyamide (PA), and PP are the most frequently reported, as shown in Fig. 2(a). The data in Fig. 2 were calculated from Tables S1–S3 and represent the percentage of publications that reported each plastic polymer type/morphology. Plastic types also show distinct regional variations. PES and PE dominate samples in Brazil,[44] whereas polyacrylonitrile (PAN), PET, and PA prevail in the United Kingdom.[43] Studies in Iran and China commonly detect PET, PP, PE, and PVC, while some investigations also report natural fibers (cellulose) and regenerated fibers (rayon), underscoring the heterogeneity of atmospheric particulate matter sources.[107,117] Morphologically, fibers and fragments are consistently the dominant forms (Fig. 2(b)). Fibers account for 60–90% of AMPs, as observed in Iran, Turkey, China, and Malaysia.[107,117] In contrast, fragments dominate in certain European regions, such as France, where they account for over 60% of detected particles.[24] Other morphotypes—films, granules/spheres, foams, and microbeads–are occasionally reported but generally constitute a minor fraction. Urban areas tend to exhibit fiber-dominant profiles, while remote or industrially influenced regions show relatively higher proportions of fragments.

Particle-size distributions indicate that most AMPs are concentrated in small size ranges. In many cases, over 70% of particles are <500 μm, with some regions, such as Malaysia, reporting that >90% of particles are ≤50 μm (Tables S1–S3). In China, Iran, and Germany, a large proportion of AMPs are <100 μm. Nevertheless, fibrous particles often extend to the millimeter scale, with fibers up to 20 mm in length documented in Ireland. Overall, AMPs are characterized by small particle sizes with coexisting fibers and fragments, although notable regional differences remain. Overall, these findings suggest that the occurrence state of AMPs is highly heterogeneous and context-dependent.

This heterogeneity arises from the combined effects of human activities, geographic location, temporal scale, proximity to urban areas,[118] and methodological differences such as sampling height, sample pretreatment, and polymer identification techniques.[47] In addition, AMP distributions are further shaped by both the physicochemical properties of the plastics and prevailing meteorological factors, including atmospheric turbulence and synoptic wind patterns. Globally, urban areas exhibit higher AMP concentrations than remote or natural regions due to dense population, economic activity, industrial structure, and plastic use and disposal.[119] Studies in Paris showed urban AMP levels twice those in suburbs,[92] with GDP affecting polymer composition.[38,119] In addition, TWPs represent an important urban source of AMPs with distinct spatio-temporal variability. Their emission depends on traffic volume, tire properties, driving patterns, road conditions, and weather.[120–122] Concentrations of TWPs increase near highways and during rush hours or dry periods.[123] Dry conditions promote higher TWP release due to reduced adhesion and enhanced resuspension.[123] Waste management practices also contribute significantly. Landfills are estimated to contain 21–42% of global plastic waste, acting as long-term sources of AMPs through wind dispersal.[124,125] Inefficient management, including landfilling and incineration, increases the release of MPs into the atmosphere.[126]

6. Impacts of atmospheric microplastics on human health and the environment

6.1 Impacts on human health

Air represents one of the most critical media for continuous human exposure, as airborne pollutants, including MPs, can be readily inhaled during respiration, constituting a long-term exposure pathway.[127,128,136] The detection of MPs in clinical lung samples provides direct evidence of respiratory uptake.[129] On average, individuals may inhale 26–130 MPs per day, with

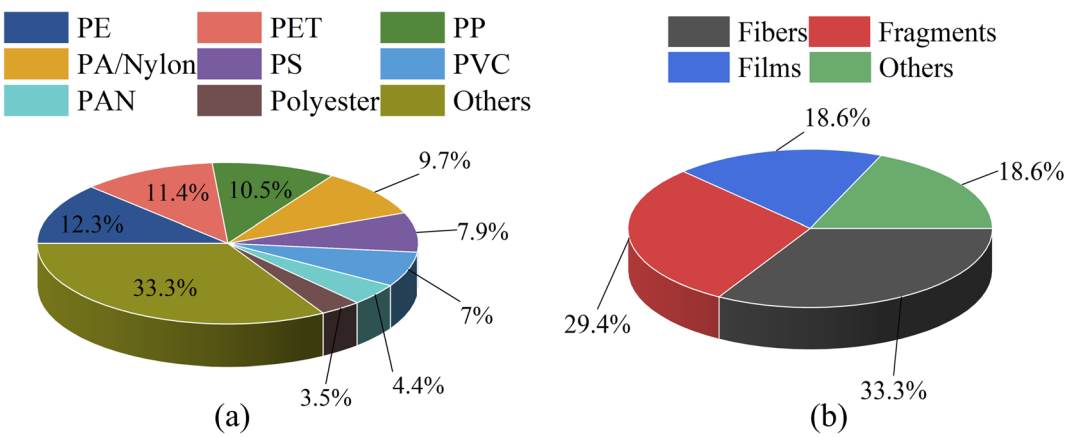

Fig. 2 Distribution of atmospheric microplastic types and morphologies: (a) microplastic types; (b) microplastic morphologies (data in the figure source: Tables S1–S3); PE: polyethylene; PET: polyester; PP: polypropylene; PA: polyamide; PS: polystyrene; PVC: polyvinyl chloride; PAN: polyacrylonitrile.

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this number rising to ~272 MPs during light physical activity.[58,130] In Shanghai, residents are estimated to be exposed to ~21 MPs per day from outdoor sources alone.[32] Children and workers in the polymer industry might suffer from serious effects of MP pollution. Children face higher exposure risks, while workers in industries such as synthetic textiles, flocking, and PVC manufacturing may be subject to greater occupational risks owing to inadequate protective measures.[127] However, some studies indicate that under acute exposure scenarios, estimated inhalation of MPs can increase by up to 5-fold in adults and 3.3-fold in children, further widening the disparity in exposure levels between these two groups.[131] Beyond inhalation, AMPs can settle on food and water sources, becoming a secondary exposure route through dietary intake. This pathway has been corroborated by the detection of MPs in human feces, facial and hand skin, hair, and saliva.[132,133]

Inhalable MPs deposit in the upper airways, whereas respirable MPs (<10 μm) penetrate deeper into the lungs and may accumulate long term in alveolar epithelial cells.[134,135] Toxicological evidence shows that MPs can trigger oxidative stress, inflammation, and impaired cell junctions, and may even contribute to carcinogenesis.[64] *In vitro* studies demonstrate that 2 μm PS-MPs induce inflammatory responses and oxidative stress, and reduce cell viability in human bronchial cells,[136] while exposure to 1–10 μm PS-MPs inhibits the proliferation of alveolar A549 cells.[137] In addition to respiratory injury, MPs may disrupt gastrointestinal functions. Studies show that MPs stimulate intestinal nerves, enhance upper intestinal secretions, and accelerate gastrointestinal transit, potentially impairing nutrient absorption and leading to deficiencies.[138,139] Furthermore, MPs can migrate within the body, accumulate in organs, and cause inflammation or physical blockage. While such mechanisms are better documented in marine organisms, their potential implications for humans remain a significant concern.

6.2 Phyllosphere-level impacts of atmospheric microplastics

As an important sink of atmospheric deposition, the phyllosphere represents a key interface through which AMPs interact with terrestrial ecosystems. Following dry and wet deposition, AMPs can accumulate on leaf surfaces, where they are either tightly adsorbed or enter the leaf and are subsequently transported to other plant tissues.[140] Surface deposition of AMPs on the phyllosphere can reduce light availability, alter leaf optical properties, and impair photosynthesis. Experimental studies report a 7–12% reduction in chlorophyll-associated photosynthetic rates in terrestrial plants, marine algae, and freshwater algae following MP exposure,[141,142] reducing crop yield and nutritional quality.[146] Following initial interactions at the phyllosphere, MPs may penetrate leaf tissues *via* apoplastic pathways and subsequently be translocated to other plant tissues through the phloem, where they can exert adverse effects on plant growth.[143] For example, studies have shown that MPs can significantly affect lettuce growth, manifested as reduced biomass (including plant height, leaf number, fresh weight of leaves and roots, *etc.*) and increased reactive oxygen species in leaves, leading to enhanced activities of superoxide dismutase, peroxidase, and catalase with increasing MP levels.[144] Moreover, AMPs can serve as carriers for co-existing airborne pollutants (*e.g.*, persistent organic pollutants and heavy metals), thereby amplifying ecological and health risks.[118,119] Upon ingestion, MPs may facilitate the transfer of adsorbed contaminants into biological systems, potentially disrupting physiological functions and adversely affecting health.[145,146] Beyond direct physiological effects on plants, the accumulation of AMPs on the phyllosphere may also perturb leaf-associated microbial communities, which play critical roles in nutrient cycling, pathogen suppression, and trace gas exchange. By altering the physical habitat and chemical microenvironment of the phyllosphere, deposited AMPs may indirectly influence biogeochemical processes such as carbon assimilation, nitrogen transformations, and plant–microbe interactions. These secondary effects highlight the phyllosphere as a sensitive receptor of AMP pollution, with implications extending beyond individual plant health to ecosystem-level functioning.

6.3 Radiative and cloud-mediated climate impacts of atmospheric microplastics

AMPs can influence atmospheric radiative processes through their optical properties, morphology, and vertical distribution. Although studies quantifying these effects remain limited, existing evidence indicates that MPs can both scatter and absorb radiation. Experimental studies on non-pigmented fragments and fibers suggest that MPs efficiently scatter ultraviolet and visible radiation, producing a surface cooling effect, while simultaneously absorbing infrared radiation, particularly in the atmospheric window (8–12 μm), contributing to greenhouse warming.[147] These results provide mechanistic insight into the potential optical behavior of MPs but do not directly constrain their climatic relevance under ambient atmospheric conditions. Model-based estimates of the radiative effects of AMPs further illustrate the strong dependence of outcomes on assumed particle properties and atmospheric distributions. For example, the net direct radiative effect (DRE) of AMPs was first estimated to be 0.044 ± 0.399 mW m^{-2} under the assumption of a uniform surface concentration of 1 particle per m^3 and an exponential decrease with height. When MPs were confined within the atmospheric boundary layer, the net DRE shifted to -0.746 ± 0.553 mW m^{-2}, highlighting the sensitivity of radiative forcing estimates to vertical distribution assumptions.[147] More recently, the global radiative impacts of MPs were further quantified by coupling atmospheric transport and radiative transfer models, yielding a global annual mean shortwave DRE of −9.07 (−40.57 to 2.04) μW m^{-2} and a longwave DRE of +2.73 (0.61 to 12.21) μW m^{-2}.[148] These effects were primarily concentrated in densely populated and arid regions, driven by elevated emission rates and favorable atmospheric conditions. The magnitude and sign of Effective Radiative Forcing (ERF) are influenced by MPs' atmospheric concentration and optical properties, including composition, color, and wavelength-dependent refractive index. ERF accounts for rapid atmospheric adjustments such as changes in temperature, humidity,

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and cloud properties, providing a more realistic estimate of MPs' net climatic impact than DRE alone. Although ERF for MPs has not been fully quantified, these rapid adjustments may amplify or mitigate the direct radiative effects observed in DRE calculations.[23,149] These estimates are based on simplified assumptions regarding MP refractive indices, external mixing state, prescribed size distributions, and idealized vertical profiles, and should therefore be interpreted as order-of-magnitude assessments rather than observationally constrained forcings. In addition, observations indicate that colorful MPs deposited on snow and ice absorb sunlight and reduce surface albedo, generating positive radiative forcing that may accelerate ice melt, especially in climate-sensitive Arctic regions.[150–152] Currently, due to the unclear distribution of MPs in the atmosphere, their precise radiative impacts cannot be accurately quantified.

Beyond direct radiative interactions, AMPs may also influence climate indirectly by modifying cloud microphysical processes. In the atmosphere, suspended MPs can be transported to altitudes where low temperatures and high relative humidity favor water uptake on particle surfaces, enabling their potential role as cloud condensation nuclei or ice-nucleating particles. Following water condensation, freezing may occur on MP surfaces at sufficiently low temperatures, leading to heterogeneous ice nucleation. Experimental evidence suggests that MPs can induce heterogeneous ice nucleation predominantly through immersion freezing under atmospherically relevant conditions.[153] In this pathway, ice formation is initiated when supercooled water droplets containing MPs freeze at temperatures warmer than those required for homogeneous freezing. Nanoplastics have been detected in the atmosphere,[154] and their small size and long atmospheric residence time increase the likelihood of interactions with cloud droplets, particularly in mixed-phase cloud regimes. The ice-nucleating activity of MPs is strongly influenced by surface aging processes during atmospheric transport. Sunlight-induced weathering alters the physicochemical properties of MP surfaces by increasing surface roughness and introducing oxygen-containing functional groups (*e.g.*, C=O and C–O bonds), which enhance surface hydrophilicity and modify the structure of surface-bound water molecules.[153] These changes can lower the relative humidity required for water uptake and increase the ice nucleation temperature, thereby facilitating the water-to-ice phase transition. In addition to photochemical aging, prolonged atmospheric exposure promotes the adsorption of hygroscopic salts such as NaCl, further enhancing water affinity and ice-nucleating efficiency. The combined effects of increased surface roughness and salt coatings reduce the energetic barrier for ice embryo formation, suggesting that aged MPs may act as more efficient ice-nucleating particles than pristine particles.

MP–cloud interactions may be particularly relevant in remote and pristine regions such as the Southern Ocean, where cloud properties are highly sensitive to ice-nucleating particle abundance.[155] By influencing cloud phase partitioning, lifetime, albedo, and spatial coverage, AMPs may exert indirect radiative effects and contribute to regional climate modulation. However, large uncertainties remain regarding the ambient concentration, size distribution, and vertical transport of ice-active microplastics, limiting current assessments of their overall climatic significance.

7. Research gaps and future perspectives

Despite rapidly growing interest in AMPs, substantial uncertainties remain regarding their environmental occurrence, transport, and impacts, largely due to methodological and conceptual limitations across the sampling–analysis–modeling chain. In recent years, notable progress has been made in AMP detection and characterization; however, these advances have not yet fully resolved key challenges related to representativeness, quantification, and standardization.

At the sampling and analytical level, recent methodological developments have expanded the detectable size range of airborne plastics, particularly toward environmentally relevant nanoplastics (NPs, <1 μm). Raman-based identification using surface-enhanced substrates (*e.g.*, gold-coated slides) has enabled improved detection of submicron airborne plastic particles, while combined spectroscopic and pyrolysis-based approaches now allow polymer identification across a broad particle-size spectrum. These advances represent important steps toward comprehensive AMP characterization. Nevertheless, major challenges persist in achieving quantitative, size-resolved, and number-based measurements under field conditions, especially for NPs, which are of particular concern due to their potential for deep respiratory uptake. Optical imaging techniques remain constrained by spatial resolution, whereas pyrolysis–GC/MS, although chemically specific, cannot provide particle size or abundance information. As a result, uncertainties in AMP concentration, size distribution, and mass-number conversion continue to limit cross-study comparability. Moreover, the spatiotemporal distribution of AMPs is influenced by diverse environmental factors, yet the underlying mechanisms governing these patterns are still insufficiently understood. Although passive bulk-deposition and dust-fall sampling approaches are well established in atmospheric particulate monitoring and explicitly address issues such as resuspension and long-term representativeness, these frameworks have not yet been systematically adapted or validated for AMPs, particularly with respect to particle size selectivity, polymer-specific identification, and post-depositional transformation. Consequently, accurate assessment of the environmental burden of AMPs is still hindered by the absence of standardized pretreatment protocols, robust quantification methods, and systematic parameterization of transport and deposition mechanisms.

Future research should prioritize an integrated and interdisciplinary framework that combines improved measurement techniques, standardized sampling protocols, and process-based atmospheric modeling. Comprehensive characterization of AMPs should extend beyond abundance to include size distribution, polymer composition, morphology, surface aging features, and optical properties, thereby enabling more reliable assessments of human health risks and climate-relevant effects. Ultimately,

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clarifying the role of AMPs within the global plastic cycle—particularly their interactions with the atmosphere as a transport medium—remains a critical yet challenging goal that will require coordinated efforts across atmospheric science, environmental chemistry, toxicology, and materials science.

8. Conclusion

AMPs have been increasingly recognized as an active component influencing ecosystems, linking terrestrial, oceanic, and atmospheric compartments. Existing observations indicate that reported AMP concentrations span several orders of magnitude across different regions and environments, reflecting strong spatial heterogeneity and the influence of source intensity, meteorological conditions, and surface–atmosphere exchange processes. Deposition fluxes of AMPs also vary substantially, with dry and wet deposition contributing differently depending on particle size, density, and local climate regimes. Current evidence suggests that secondary emissions driven by surface resuspension represent a non-negligible contribution to atmospheric AMP burdens, particularly in urban and agricultural regions. Despite these advances, quantitative understanding of AMPs remains constrained by methodological limitations, as standardized protocols for sampling, identification, and quantification are still lacking. Detection limits, size cutoffs, and reported abundance metrics vary considerably among studies, complicating cross-regional comparisons and large-scale assessments. These uncertainties hinder robust evaluation of AMP transport, long-range dispersion, and their potential implications for atmospheric processes and climate forcing. Overall, further efforts are needed to better constrain AMP sources, transport efficiencies, and deposition pathways, which are essential for integrating AMPs into global material cycles and supporting environmental risk assessment and climate-related policy decisions.

Author contributions

Jiajia Shan: writing – original draft; Bominghan Yu: data curation; Jindong Guo: visualization; Wenliang Chen: visualization; Ying Su: writing – review & editing; Xue Wang: funding acquisition, writing – review & editing; Xin Wang: conceptualization, funding acquisition, writing – review & editing.

Conflicts of interest

There are no conflicts of interest to declare.

Abbreviations

AMPs	Atmospheric microplastics
MPs	Microplastics
TWPs	Tire wear particles
PET	Polyester
PU	Polyurethane
PMMA	Polymethyl methacrylate
PVC	Polyvinyl chloride
PE	Polyethylene
PP	Polypropylene
PS	Polystyrene
EPS	Expanded polystyrene
PES	Polyester
PA	Polyamide
PAN	Polyacrylonitrile
FTIR	Fourier transform infrared
QA/QC	Quality assurance and quality control
HYSPLIT	Hybrid single-particle Lagrangian integrated trajectory
DRE	Direct radiative effect
ERF	Effective radiative forcing

Data availability

The data supporting this article have been included as part of the supplementary information (SI). Supplementary information: summary data of atmospheric microplastic deposition flux and density reported in different literature. See DOI: https://doi.org/10.1039/d5ea00162e.

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