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There's something in the air: A review of sources, prevalence and behaviour of microplastics in the atmosphere

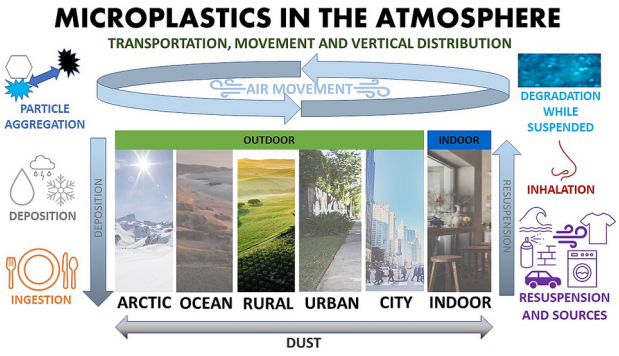
Stacey O'Brien [a,*], Cassandra Rauert [a], Francisca Ribeiro [a,d], Elvis D. Okoffo [a], Stephen D. Burrows [a,d], Jake W. O'Brien [a], Xianyu Wang [a], Stephanie L. Wright [b,c], Kevin V. Thomas [a]

[a] Queensland Alliance of Environmental Health Sciences, The University of Queensland, 20 Cornwall Street, Woolloongabba, Queensland 4102, Australia
[b] MRC Centre for Environment and Health, Imperial College London, London SE1 9NH, UK
[c] National Institute for Health Research Health Protection Research Unit (NIHR HPRU) in Environmental Exposures and Health, Imperial College London, London SW7 2AZ, UK
[d] College of Life and Environmental Sciences, University of Exeter, Geoffrey Pope Building, EX4 4QD, Stocker Road, Exeter, UK

HIGHLIGHTS

- Outdoor concentrations varied between <1 and >1000 microplastics/m^3 (air), 0.5 and 1357 microplastics/m^2/day (deposition).
- Dust concentrations varied between 2 and 477 microplastics/g (road dust), 18 and 225 microplastics/g (outdoor dust).
- Indoor air concentrations varied between <1 microplastic/m^3 and 1583 ± 1181 (mean) microplastics/m^3.
- Indoor dustfall contained between 475 and 19,600 microplastics/m^2/day and indoor dust between 10 and 67,000 microplastics/g.
- Composition was; Air: PET, PE, PP; Deposition: PET, PE, PP; Road dust: PE, PVC PP; Outdoor dust: PET, PP, PA; Indoor dust: PET, PE, PP.

GRAPHICAL ABSTRACT



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ABSTRACT

Literature regarding microplastics in the atmosphere has advanced in recent years. However, studies have been undertaken in isolation with minimal collaboration and exploration of the relationships between air, deposition and dust. This review collates concentrations (particle count and mass-based), shape, size and polymetric characteristics for microplastics in ambient air (m3), deposition (m2/day), dust (microplastics/g) and snow (microplastics/L) from 124 peer-reviewed articles to provide a holistic overview and analysis of our current knowledge.

In summary, ambient air featured concentrations between <1 to >1000 microplastics/m^3 (outdoor) and <1 microplastic/m^3 to 1583 ± 1181 (mean) microplastics/m^3 (indoor), consisting of polyethylene terephthalate, polyethylene, polypropylene. No difference ($p > 0.05$) was observed between indoor and outdoor concentrations or the minimum size of microplastics ($p > 0.5$). Maximum microplastic sizes were larger indoors ($p < 0.05$).

Deposition concentrations ranged between 0.5 and 1357 microplastics/m^2/day (outdoor) and 475 to 19,600 microplastics/m^2/day (indoor), including polyethylene, polystyrene, polypropylene, polyethylene terephthalate. Concentrations varied between indoor and outdoor deposition ($p < 0.05$), being more abundant indoors, potentially closer to sources/sinks. No difference was observed between the minimum or maximum reported microplastic sizes within indoor and outdoor deposition ($p > 0.05$).

* Corresponding author at: University of Queensland, Australia.
E-mail address: stacey.obrien@uq.edu.au (S. O'Brien).

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Road dust concentrations varied between 2 ± 2 and 477 microplastics/g (mean), consisting of polyvinyl chloride, polyethylene, polypropylene. Mean outdoor dust concentrations ranged from <1 microplastic/g (remote desert) to between 18 and 225 microplastics/g, comprised of polyethylene terephthalate, polyamide, polypropylene.
Snow concentrations varied between 0.1 and 30,000 microplastics/L, containing polyethylene, polyamide, polypropylene. Concentrations within indoor dust varied between 10 and 67,000 microplastics/g, including polyethylene terephthalate, polyethylene, polypropylene. No difference was observed between indoor and outdoor concentrations (microplastics/g) or maximum size ($p > 0.05$). The minimum size of microplastics were smaller within outdoor dust ($p > 0.05$).
Although comparability is hindered by differing sampling methods, analytical techniques, polymers investigated, spectral libraries and inconsistent terminology, this review provides a synopsis of knowledge to date regarding atmospheric microplastics.

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1. Introduction

'Plastics' is an all-encompassing term used to reference a suite of organic synthetic and semi-synthetic polymers (Europe, 2017) manufactured into various thermoplastics such as polystyrene (PS), polypropylene (PP), polyethylene terephthalate (PET), polyethylene (PE), polyvinyl chloride (PVC), poly methyl methacrylate (PMMA) and thermoset plastics such as polyurethane (PU), epoxy resins, silicone, vinyl and melamine (Europe, 2019). Microplastics are pieces of plastic between 1 μm to 5 mm along their longest dimension (Arthur et al., 2009; Andrady, 2011a), either purposefully created as primary microplastics (Cole et al., 2011) or become secondary microplastics (Cole et al., 2011) degraded/abraded/fragmented

(Andrady, 2011b) from larger plastic products. Microplastics include particles of differing morphology including shape (fibre, fragment, sphere, film, foam (Lusher et al., 2017)), colour and surface porosity variability (Wang et al., 2018). The polymetric composition of secondary microplastics is also altered during weathering, primarily driven by photooxidation (T. Toapanta, et al., 1879), affecting the concentration of additives and other organic pollutants such as polycyclic aromatic hydrocarbons (PAHs), dichlorodiphenyltrichloroethane (DDT) and polychlorinated biphenyl (PCBs), dictated by sorption behaviours which vary among different plastics (Wang et al., 2018), and thus represent potentially differing toxicities (Rubin and Zucker, 2022). Microplastics are a class of pollutant (Cole et al., 2011) which have been detected in all environmental matrices (Barnes et al., 2009). Research to date has primarily focused on microplastics in the marine environment (Andrady, 2011b), however knowledge regarding the hydrosphere, geosphere and atmosphere remains comparatively limited (Allen et al., 2020).

Taking a holistic approach to the definition of the atmospheric environment, the atmosphere includes ambient air, deposition and dust which is capable of resuspension into the air (Droge, 2022). Specifically, ambient air includes suspended particulates, including total suspended particulates (TSP), particles with an aerodynamic diameter of <10 μm (PM_{10}) and fine particulates with an aerodynamic diameter <2.5 μm ($PM_{2.5}$). Depending on the size, airborne particulates are susceptible to horizontal and transboundary movement and vertical distribution at higher altitudes. (Further information regarding atmospheric behaviour is contained in the Supporting Information (SI)). They also pose different health outcomes as particulates <100 μm are inhaled and move into the nasopharyngeal airways. Smaller PM_{10} particles move into the thoracic region and $PM_{2.5}$ are able to penetrate into the deeper alveolar areas of the lung (Levermore et al., 2020) and are associated with various non-communicable diseases.

Deposition examines the number of particles that fall from the air and settle onto an Earth bound surface, over a known duration of time and surface area (Zhou et al., 2017; Dris et al., 2016a) and can be extended to include the mass of particles suspended in the air (Liu et al., 2019a). Deposition depends on the terminal velocity of the particle (Wright et al., 2020), influenced by its aerodynamic properties, weight and particle size (Dris et al., 2017), and may be susceptible to resuspension (Zhang et al., 2020a). Deposition contributes to dust load, and outdoor deposition can be a source of microplastics to terrestrial environments such as agricultural soil and the ocean (Liu et al., 2019b).

Dust can be a sink for various contaminants (Salthammer, 2003), accumulating compounds of interest which have deposited over an unknown timeframe. 'Fresh dust' is under 1 week old and 'old dust' represents non-episodic emissions (Salthammer, 2003). Dust can be sampled as outdoor dust (from locations such as windowsills), road dust (from the road surface or adjoining gutter) and indoor dust (floor and/or dust accumulated on a raised indoor surface), and is heterogeneous in composition between households and socioeconomic conditions (Salthammer, 2003).

Recently, attention has been directed towards understanding microplastics in the atmosphere, centred around identification in remote locations (Bergmann, 2019; Cincinelli et al., 2017), above the planetary boundary (González-Pleiter et al., 2021) and concerns regarding human health exposure and outcomes (Gasperi et al., 2018; Wright et al., 2021; Prata, 2018; Wright and Kelly, 2017), especially as a constituent of particulate matter in the smaller particle size ranges (Levermore et al., 2020). In an occupational setting (flocking), suspended microplastic fibres have demonstrated adverse health outcomes (Prata, 2018). Microplastics have been detected within human lung samples (Pauly et al., 1998) and ground glass nodules, a form of tumorous growth (Chen et al., 2022a).

Post the initial identification of microplastics in deposition from France in 2015 (Dris et al., 2015), knowledge regarding microplastics in the atmosphere has grown and escalated in interest over recent years (Liu et al., 2020a). Analytical techniques for identifying or quantifying microplastics in atmospheric samples have varied in sophistication and evolved over time from visual identification using microscopy (such as binocular (Dris et al., 2017), fluorescence (Tunahan Kaya et al., 2018; Catarino et al., 2018; Klein and Fischer, 2019) and Scanning Electron Microscopy (SEM) techniques (Dris et al., 2015; Cai et al., 2017)) to visual analysis accompanied by staining with Nile Red, rose Bengal (Prata, 2020; Roblin and Aherne, 2020) or hot needling (Shim et al., 2017; Loppi et al., 2021) to improve identification accuracy (Wright et al., 2020; Tunahan Kaya et al., 2018; Catarino et al., 2018; Klein and Fischer, 2019) and minimise false identification of natural materials as plastics (Dris et al., 2016b). Mass based quantification of microplastics in the atmosphere has been reported using Liquid Chromatography-Mass Spectrometry (LC-MS) (Liu et al., 2019c; Zhang et al., 2020b) (for polyethylene terephthalate and polycarbonate (Liu et al., 2019c; Wang et al., 2017a)), Thermo-Gravimetric Analysis Mass Spectrometry (TGA-MS) for polystyrene (Materić et al., 2020; Peñalver et al., 2021) and Proton Transfer Reaction Mass Spectrometry (TD-PTR-MS) for polystyrene, polyethylene, polypropylene, polyvinyl chloride and polyethylene terephthalate (Materić et al., 2020; Allen et al., 2022). (Further information regarding analytical methods is provided in the Supplementary Information (SI).

Despite analytical differences, this literature review provided an umbrella summarisation and comparison of the research published to date. Research covering the environmental sources of microplastics into the atmosphere (including the air/water interface and wind abrasion) and anthropogenic sources (including human particle and industrial emissions) was initially presented to understand the pathways of plastics into the atmosphere. Literature quantifying microplastics in the various atmospheric compartments of ambient air, deposition and dust was then indiscriminately collated by publication year, synthesised and statistically analysed. The differing sampling and analytical methods, concentrations, morphological information of identified plastics (including shape and size) and the various polymer compositions of microplastics in each matrix were analysed to observe macro trends or similarities. Furthermore, knowledge gaps were subsequently identified, and future research directions presented to assist the acquisition of knowledge in this emerging field and understand the entire plastic exposome.

2. Materials and method

2.1. Identification of literature

Peer-reviewed published literature, industry reports and conference abstracts covering microplastics in all facets of the atmosphere encompassing ambient air, deposition and dust, were identified using traditional search tools (including Scopus, PubMeb, Google Scholar) and employing search terms such as 'microplastic', 'plastic', 'air' and 'atmosphere'. Identified articles canvassing microplastics within the atmospheric compartment were read in their entirety and additional articles were identified and collated via deeper examination of content and reference lists within those peer-reviewed publications. Literature covering waterways, the ocean and soil were not included, unless they examined the atmospheric environment in parallel.

2.2. Normalisation and segregation of results

To facilitate this literature review and comparability between the reported results from various publications, research was normalised for microplastic concentrations and segregated into the compartments of 1) suspended particles in both outdoor and indoor air (particles/m^3), 2) atmospheric deposition and indoor dustfall (particles/m^2/day) and dust (indoor and outdoor) (particles or concentration/g). These segregations were selected based on the different sampling techniques (Fig. 1), analytical methods, reporting units and end use of the research. Dust and deposition concentrations inform ingestion exposure (Liu et al., 2019a; Allen et al., 2019) and ambient air enlightens respiratory and inhalation exposure assessments (Gasperi et al., 2018; Wright et al., 2021; Prata, 2018).

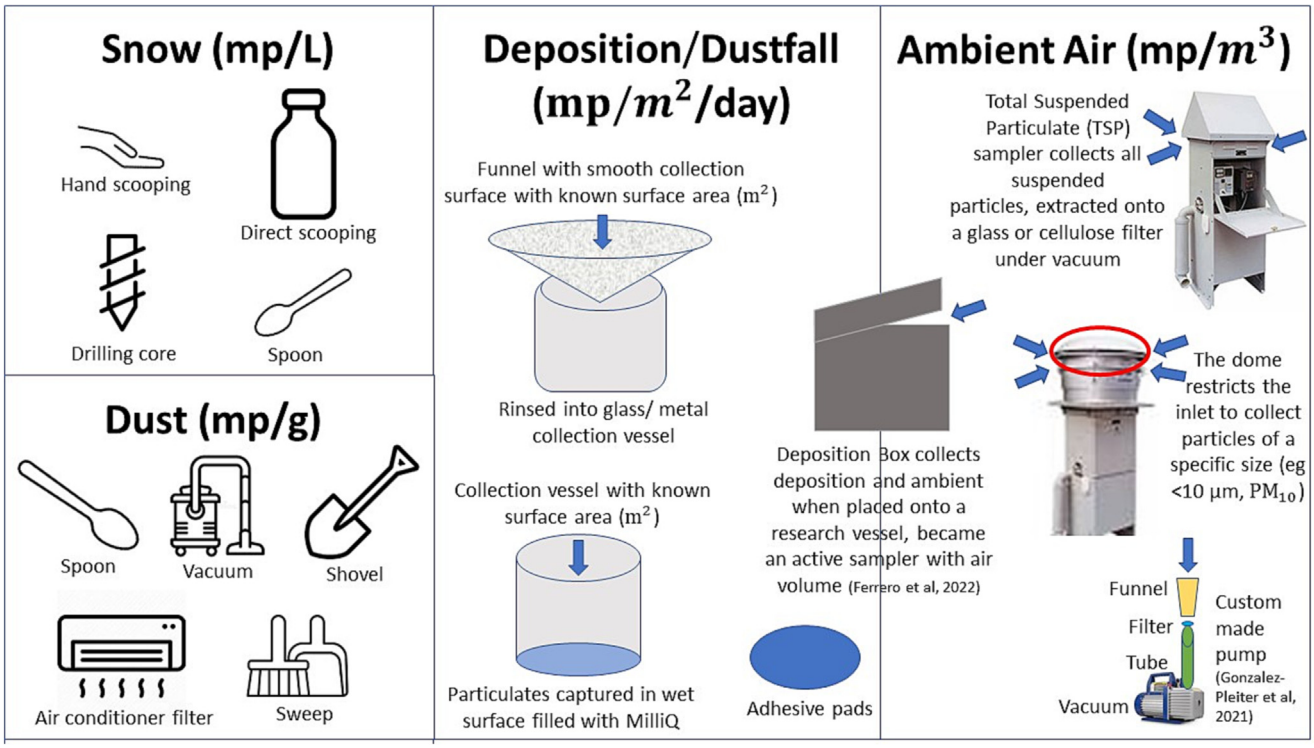


Fig. 1. Visual representation of the various sampling methods for atmospheric microplastics within ambient air, deposition/dustfall, snow and dust.
* 'mp' refers to the number (particle count) of microplastics identified within the matrix.
** Dust is often reported as the number of microplastics per gram. However, some studies have reported the number of microplastics per milligram, which have subsequently been adjusted by the authors to align with other studies referencing per gram.
*** The blue arrows indicate the flow of air into the inlet of the sample collection method.

2.3. Data analysis

All data analysis and graphs were prepared using Microsoft Excel and GraphPad Prism (Version 9).

Bibliometric analysis was completed using VOSviewer software. All literature included within this review and supplementary information (excluding industry reports and two publications which were not able to be imported) and matrix specific research for ambient air, deposition and dust was analysed for authorship prevalence and networks. Analysis involved identifying all authors who had published on atmospheric microplastics, followed by a deeper exploration of the prevalence and networks of authors who have published ≥2 or more publications.

Bibliometric key word analysis was also undertaken to identify the usage, popularity and trends over time of differing key terms in research used in ≥3 publications.

3. Outcomes of the literature search

3.1. The number of articles

At the conclusion of all search activities, 124 peer-reviewed articles and industry papers published between 2015 (Dris et al., 2015) and 30 July 2022 were identified to present research covering microplastics in the atmosphere, consisting of 97 peer-reviewed articles specifically quantifying microplastics in the atmosphere and 22 review articles (SI: Table S4). One article was not published in English and therefore could not be read by the authors, however the abstract was published in English and thus the pertinent results were also included. All identified articles were systematically and indiscriminately collated (as opposed to reviewing quality and application (Wright et al., 2021)) to provide a holistic understanding and review of information to date. (Fig. 2, Supplementary Information (SI): Figs. S1 – S3, Table S4).

3.2. Bibliometric analysis of all literature contained within the article and supplementary information (SI) information

3.2.1. Bibliometric analysis - all literature contained within the article and SI for authorship prevalence and networks

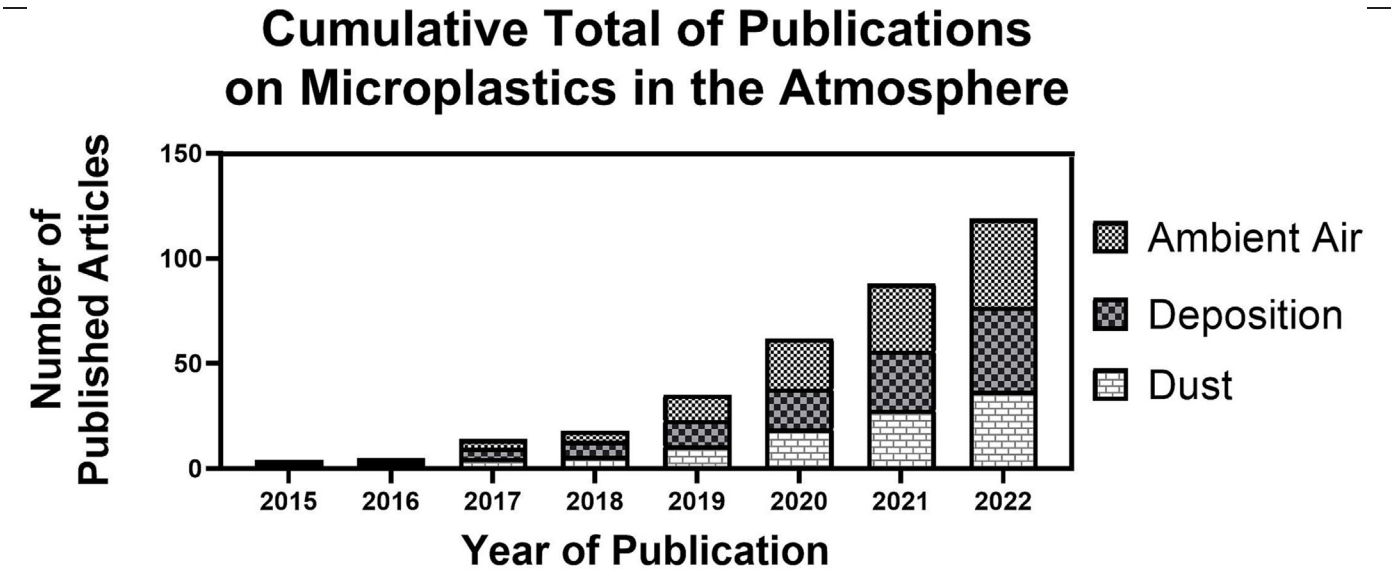
Bibliometric analysis of all literature contained within this review and Supplementary Information identified 824 individual authors who have published material relevant to atmospheric microplastics, which were segregated into 741 separate clusters (SI: Fig. S5). Of the 824 authors, 114 (14 %) had published more than two articles, which were differentiated into 30 different clusters (Fig. 3, SI: Table S6). Importantly, most of the work in this field has been undertaken in silos with minimal collaboration.

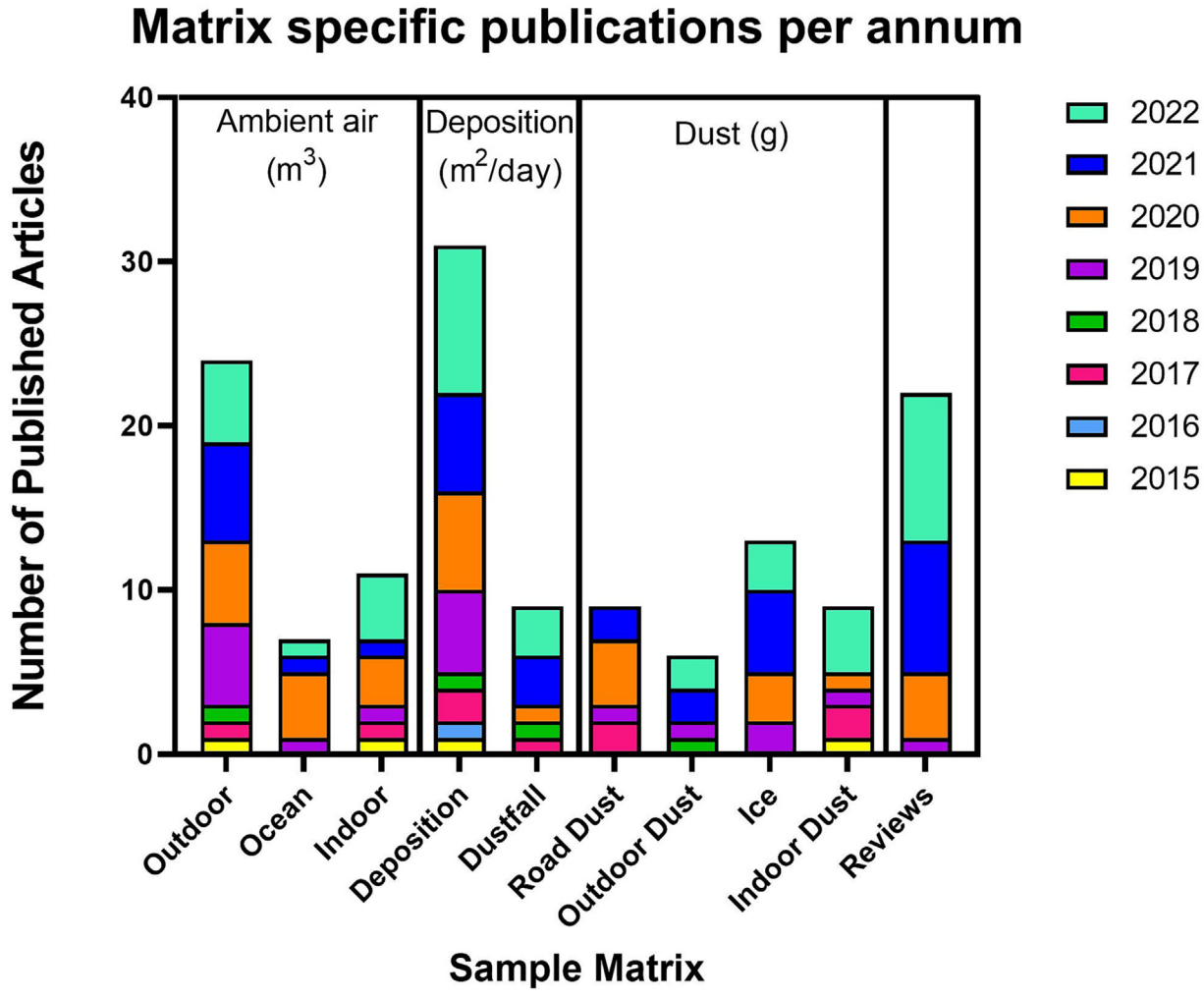
3.2.2. Bibliometric analysis - all articles included within the article and SI regarding key word analysis

Across all articles, 1332 key words were identified and 221 key words (17 %) featured a co-occurrence across three or more publications (Fig. 4), divided into 82 clusters. In addition to emerging authorship trends, transition of the key terms used for literature regarding microplastics in the atmosphere has similarly evolved since 2019. Recent terminology in 2021/2022 moved towards the inclusion of terms focused towards risk assessment, meteorological variables and remote locations such as glaciers, suggesting that it is a rapidly evolving field.

4. Sources of microplastics into air

Quantified sources of airborne microplastics remain largely unknown (Allen et al., 2020), however include the air/water interface and sea spray (Allen et al., 2020), wind abrasion (Rezaei et al., 2019), resuspension of tyre wear, nail salons (Chen et al., 2022b), vented mechanical drying of synthetic material (Kapp and Miller, 2020; O'Brien, 2020), areas of landfill


Cumulative Total of Publications on Microplastics in the Atmosphere
Number of Published Articles
150
100
50
0
2015
2016
2017
2018
2019
2020
2021
2022
Year of Publication
Ambient Air
Deposition
Dust


Matrix specific publications per annum
Number of Published Articles
40
30
20
10
0
Ambient air (m3)
Deposition (m2/day)
Dust (g)
2022
2021
2020
2019
2018
2017
2016
2015
Outdoor
Ocean
Indoor
Deposition
Dustfall
Road Dust
Outdoor Dust
Ice
Indoor Dust
Reviews
Sample Matrix

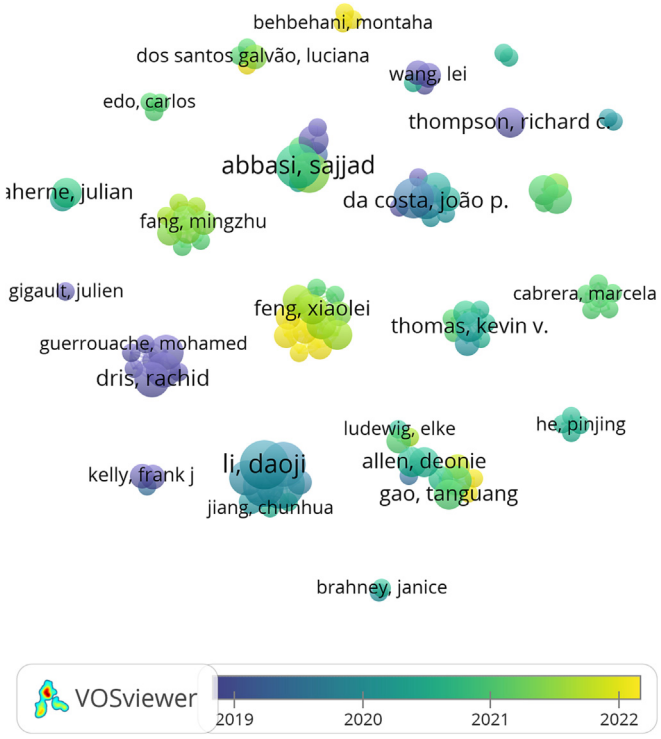


Fig. 3. Bibliometric analysis of authorship prevalence and networks for 114 authors who had published ≥2 articles from all articles included within this literature review and Supplementary Information.
* The colour and year scale represent the year of publication.
** The differing sizes of the circular markings (dots) represent the varying strength of the actors in the field.
*** Included publications were indiscriminately collated based on the topic and did not undergo screening for quality.
**** Industry reports without an associated RIS file were not included in the bibliometric analysis.
***** Two publications were omitted due to lack of associated RIS/Endnote file information capable of importation into the software which inhibited inclusion.

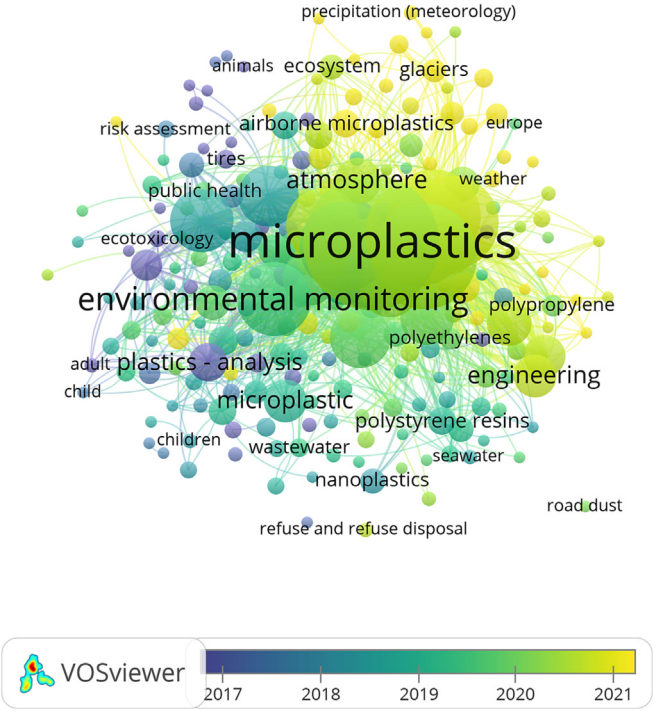


Fig. 4. Bibliometric analysis of 221 key terms used across all literature included within this literature review and Supplementary Information, where the term co-occurred across >3 publications.
* The colour and year scale represent the year of publication.
** The differing sizes of the circular markings (dots) represents the varying frequency of the key term in the field.
*** Included publications were indiscriminately collated based on the topic and did not undergo screening for quality.
**** Industry reports without an associated RIS file were not included in the bibliometric analysis.
***** Two publications were omitted due to lack of associated RIS/Endnote file information capable of importation into the software which inhibited inclusion.

(Dris et al., 2016b) and human movement when wearing synthetic clothing (Yoon and Brimblecombe, 2000).

4.1. Sea spray/air water interface

Examining the oceanic environment, >30 % of atmospheric particles above the ocean were identified to be plastic (Ferrero et al., 2022). Observed consistency between the polymer composition (Trainic et al., 2020), concentration and morphology (colour, shape) of microplastics in the atmosphere and surface sea water (Ferrero et al., 2022) lead to the hypothesis that the ocean could be a source of aerosolised microplastics (Trainic et al., 2020).

From the surface of the ocean, microplastics, 20 ± 13 μm in size were proven to be aerosolised, potentially through bubble burst or ejection in sea spray (Allen et al., 2020), which could represent up to 11 % of point source emissions of microplastics into air (Brahney, 2021). Once suspended into the air from the surface water, microplastics could become available for wind transportation, contribute to suspended particle aggregation and cloud or droplet formation due to altered hydrophobicity (Allen et al., 2020). At the ocean surface, forces such as deposition, atmospheric transportation and aerosolization from the ocean surface affect this interface, creating a 'grasshopper' transport method where microplastics are suspended, transported and then deposited onto the ocean surface in a cyclical manner (Ferrero et al., 2022). Whilst it was previously considered

Fig. 2. (a): Graphical representation of the cumulative number of published articles covering microplastics in the ambient air, deposition and dust according to publication year.
(b): Graphical depiction of the total number of published articles, separated by matrix, in ambient air (microplastics/m^3), deposition (microplastics/m^2/day), dust (microplastics/g) and review articles, coloured according to publication year.
*Ambient air matrices include: 'Outdoor' meaning outdoor ambient air, 'Ocean' meaning air sampled from above an open ocean and 'Indoor' meaning ambient air within an indoor environment.
** Deposition matrices include 'Deposition' meaning the plastics that fall onto a surface over a known duration of time, 'Dustfall' being particles that settle onto an indoor surface over a known duration of time.
***Dust matrices include 'Road dust', being dust collected from the surface of a road, 'Outdoor dust' included the settled particulates on an outdoor surface such as a windowsill over an unknown duration of time, 'Ice' meaning ice and snow, 'Indoor dust' included the settled particulates collected from an indoor surface over an unknown duration of time.
**** Publications covering two compartments/matrices have been added to each compartment respectively.

that marine microplastics were generated from terrestrial activities (Duis and Coors, 2016), Allen et al. (2020) calculated that globally, approximately 136 k tonnes of marine plastics blow ashore annually, providing recirculation into the terrestrial environment.

Within laboratory experiments, the air water interface was shown to transfer smaller nano plastics (majority <350 nm) into the surrounding atmosphere through bubble bursting mechanisms. Aerosolization was influenced by the surface characteristics of the plastics, including hydrophobicity. Plastic particles agglomerate at the ocean surface and are therefore less likely to be contained in the bubble burst. However, UV degradation, plastic age (for polyethylene) and smaller bubble size (with more 'jet' type drops in the bubble burst mechanism) increase aerosolization efficiency for plastics (Masry et al., 2021). Polystyrene can transfer into the surrounding air as individual particles 600 nm in diameter or as groups of up to 3 polystyrene particles, coated in a sulphur atom (Masry et al., 2021). Beyond proving a concept in a laboratory environment, transfer efficiency has not been explored for the differing polymer types and nano plastics in environmental air samples have not been confirmed, therefore understanding the air and water interface requires further exploration.

Beyond the ocean, consistencies have been observed between the concentrations and composition of microplastics within the terrestrial aquatic environment and atmospheric deposition, suggesting that other aquatic environments could also be sources of microplastics into the air through entrainment at moving waterways (Huang et al., 2022).

4.2. Wind abrasion

Environmental terrestrial microplastics degrade into particulates capable of atmospheric suspension faster than their marine counterparts based on increased UV exposure and temperature fluctuations (Horton et al., 2017). Using a simulated wind tunnel, wind erosion was able to suspend surface based microplastics into the air at a rate of 0.08 to 1.48 mg/m^2/min, being an important source of microplastics into the atmosphere (Rezaei et al., 2019).

Outdoors, wind abrasion abrades microplastics from textiles and resuspends them into the air from landfill (Rillig, 2012). Atmospheric emissions from wind abrasion at landfill sites was calculated to be 0.16 microplastics/t of waste/h (Hu et al., 2022), and the polymer composition was associated with the composition of the waste at the time (Hu et al., 2022).

From soils in Iran, approximately 2×10^{12} microplastics could be suspended into the air from a dust storm, depositing 1.5×10^{12} microplastic particles within the region. Approximately 4×10^{13} microplastics could be suspended globally from soil (Abbasi et al., 2022), equating to approximately 5 % of atmospheric microplastics (Brahney, 2021).

4.3. Tyre wear

Re-suspension of microplastics on roads could contribute up to 84 % of plastics in the air (Brahney, 2021). Tyre tread can be abraded or thermally degraded (Evangeliou et al., 2020) into tyre wear particles of both fragment and fibre shapes (Abbasi, 2019), in various sizes ranging from <2.5 μm – 350 μm, with a density of 1.7–2.1 g/cm^3 (Vogelsang et al., 2019). Synthetic rubber in tyre wear, such as styrene-butadiene copolymers and polymers used in bitumen formulations and road marking paints are known to contribute to microplastics in settled road dust (Abbasi, 2019; Vogelsang et al., 2019; Yukioka, 2020; Kitahara and Nakata, 2020) and are therefore hypothesised to suspend into ambient air (Goßmann et al., 2022). In the air, 28 % of fine particles (<10 μm) are resuspended road dust (Manoli et al., 2002) and approximately 5 % of tyre wear emissions become airborne (Wik and Dave, 2009). In high traffic areas, airborne tyre wear concentrations are typically 2–5 μg/m^3 (Panko et al., 2013). In the USA, it has been suggested that 1 % of all PM_{10} originates from road emissions, 15 % of which is derived directly from tyres (Panko et al., 2013).

Global volumes of tyre wear emissions could be influenced by the tyre surface and/or vehicular weight (Verschoor et al., n.d.), therefore influencing microplastic generation and emissions (Wang et al., 2017b; Kim and Lee, 2018). By aggregating literature which examines tyre wear emissions in various countries (Netherlands (Verschoor et al., n.d.), Sweden (Verschoor et al., n.d.), Norway (Sundt et al., 2014; Kole et al., 2017), Denmark (Kole et al., 2017), Germany (Kole et al., 2017), UK (Kole et al., 2017), Italy (Kole et al., 2017; Milani et al., 2004), Japan (Kole et al., 2017), China (Kole et al., 2017), India (Kole et al., 2017), Australia (Kole et al., 2017; Milani et al., 2004), USA (Kole et al., 2017) and Brazil (Kole et al., 2017), and applying a factor of 5 % (proportion which becomes airborne), a calculation of approximately 146 k tonnes of microplastics in tyre wear per annum could be aerosolised from the countries listed prior (SI: Table S7). Evangeliou et al. (2020) calculated airborne emissions of tyre wear to be 2907 k tonnes globally per annum, including 288 k tonnes of PM_{10} and 29 k tonnes of $PM_{2.5}$ (Evangeliou et al., 2020). Regardless of the different estimations, airborne emissions of tyre wear and resuspended road dust particles as a source of microplastics into the atmospheric environment is significant.

4.4. Nail salons

Nail salons are considered to be an emission source of microplastics into the surrounding indoor air (~18 microplastics/m^3), consisting of acrylic (27 %), rubber (21 %) and polyurethane (13 %) composition in fragment shapes <50 μm (Chen et al., 2022b). It remains unknown whether the type of treatment, number of clients per day or indoor ventilation efficiency affect airborne concentrations of microplastics.

4.5. Quantification of synthetic clothing

4.5.1. Microplastic fibres

Microplastic fibres from textiles are mobile and have been detected in outdoor dust (Abbasi, 2019), atmospheric samples from the open ocean (Liu et al., 2019b), in remote locations (Brahney et al., 2020) and arguably contribute to microplastic fibre counts within indoor environments (Liao et al., 2021). Although not specific to clothing, Liu et al. (2020b) estimated that 0–36.64 g/km^2 or a total mass of 8–34 t (Liu et al., 2020b) of airborne microplastic fibres would be generated globally from textiles in 2018.

4.5.2. Mechanical dryers

Mechanical drying of synthetic textiles was found to emit microplastic fibres into the surrounding indoor air at a rate of >1 microplastic fibre/m^3 when drying a blanket for 20 min (vented indoors) (O'Brien, 2020; Brahney et al., 2020) and between 4344 ± 2066 and 12,570 ± 6514 microplastic fibres/m^2 per use into the adjoining outdoor environment (vented outdoors), which settle >9 m from the vent (Kapp and Miller, 2020).

4.5.3. Human particle emissions

'Human particle emissions' (Licina et al., 2017) are also abraded from, captured by and transported on synthetic clothing during normal and everyday movement (Yoon and Brimblecombe, 2000). Associated with fabric type (De Falco et al., 2020) and particle size, particulate emissions generated from the material itself were correlated to the intensity of activity (You et al., 2013), such as dressing and folding clothing (Licina et al., 2019), and were influenced further by the fabric loading, bonding, wear of the clothing item (Licina et al., 2019) and fibre construction (being of woven or knit construction) (De Falco et al., 2020). Licina et al. (2017) estimated that approximately 8 million particulates (1–10 μm, PM_{10}) including skin, particulates associated with the material and particulates adhered to the material were emitted per person, per hour, equating to an exposure of 1.6–13 μg/m^3. The proportion of plastics within this exposure remains unknown, however in another study, 1×10^9 microplastic fibres were calculated to be emitted into air per person, per annum by wearing 1 kg of synthetic clothing for 8 h a day, ranging in size from means of 610–796 μm in

length and diameters ranging between means of 15–19 μm (De Falco et al., 2020).

Although not polymetrically confirmed, exercising whilst wearing polyester sportswear delivered emissions of ~1000–10,000 (1–2 μm), ~600–10,000 (2–5 μm), 10–1000 (5–10 μm) and 10 - >600 (>10 μm) microplastics per sample (You et al., 2013). In another study, microplastic fibres, in the predominant size ranges of 0–5 μm and 100–500 μm, became airborne at concentrations of 0.7–5 mg/g of garment when hitting, shaking and dancing (Yoon and Brimblecombe, 2000). Subjected to an artificial fan (5 m/s) or 'air washing', synthetic garments emitted 1.9–2.1 mg of microplastics per article into the surrounding air (Yoon and Brimblecombe, 2000), suggesting that wind speed could influence microplastic fibre abrasion from clothing. Within schools, clothing type did not influence microplastic concentrations in dust, however wind and wind direction did (Nematollahi et al., 2022), potentially confirming wind abrasion from clothing.

Abrasion of fleece, which represents usual pilling through wear, releases 2.1 ± 0.3 × 10^{11} nano plastics per gram of fabric between 190 ± 90 nm and 188 ± 68 nm in size (depending on instrumentation), alongside 14,100 ± 1500 small fibres (93 ± 70 μm) and 5.3 ± 0.5 × 10^{5} larger fibres (304 ± 270 μm) per gram of fabric (Yang et al., 2021a).

Importantly, clothing based emissions could be a significant source of microplastics into the atmosphere, and remain in the convective plume surrounding the body (Thomas, 2008), within an individual's personal cloud (Thomas et al., 1996), thus increasing inhalation exposure to ambient microplastics when seated (Thomas, 2008) and clothing mediated exposure of chemical additives and particulates (Licina et al., 2019).

5. Ambient air (microplastics/m^3)

5.1. Bibliometric analysis of literature quantifying microplastics in ambient air

5.1.1. Bibliometric analysis of literature quantifying microplastics in ambient air for authorship prevalence and networks

Bibliometric analysis which included all publications quantifying microplastics in ambient air samples identified 220 contributing authors, divided into 27 clusters (Fig. 5). Of the 220 identified authors, 22 authors (10 %) had published two or more articles covering microplastics in ambient air (Fig. 6), separated into 5 clusters (SI: Table S8).

5.1.2. Bibliometric analysis of literature quantifying microplastics in ambient air for key word analysis

Bibliometric analysis of the key words used within research which quantified microplastics in ambient air identified 371 individual terms, 48 of which were consistent across three or more publications, divided into 11 clusters (Fig. 10, SI: Table S9). During 2021, terminology moved to include the terms raman and inhalation, suggesting a shift in focus (Fig. 7).

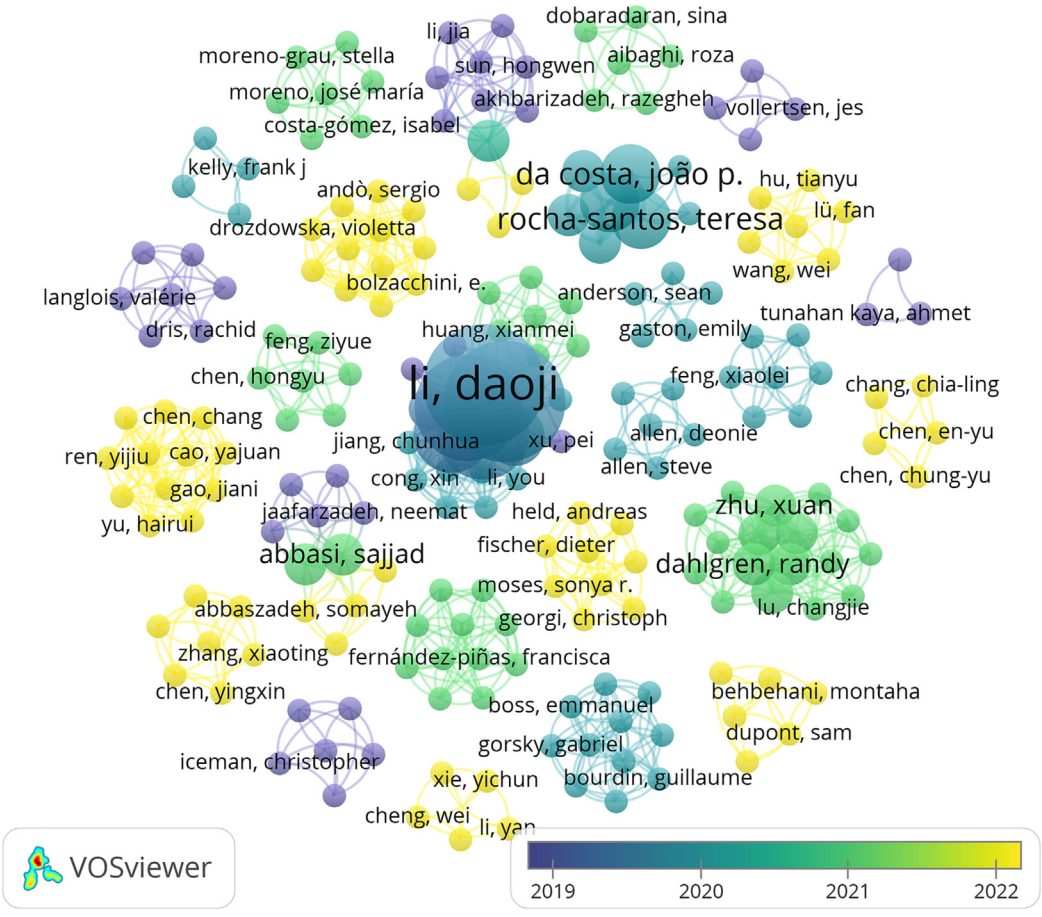


Fig. 5. Bibliometric Analysis of authorship prevalence and networks for all 220 authors included within this literature review and Supplementary Information who had publications covering the quantification of plastics in ambient air.

* The colour and year scale represent the year of publication.

** The differing sizes of the circular markings (dots) represent the varying strength of the actors in the field.

*** Included publications were indiscriminately collated based on the topic and did not undergo screening for quality.

**** Industry reports without an associated RIS file were not included in the bibliometric analysis.

***** Two publications were omitted due to lack of associated RIS/Endnote file information capable of importation into the software which inhibited inclusion.

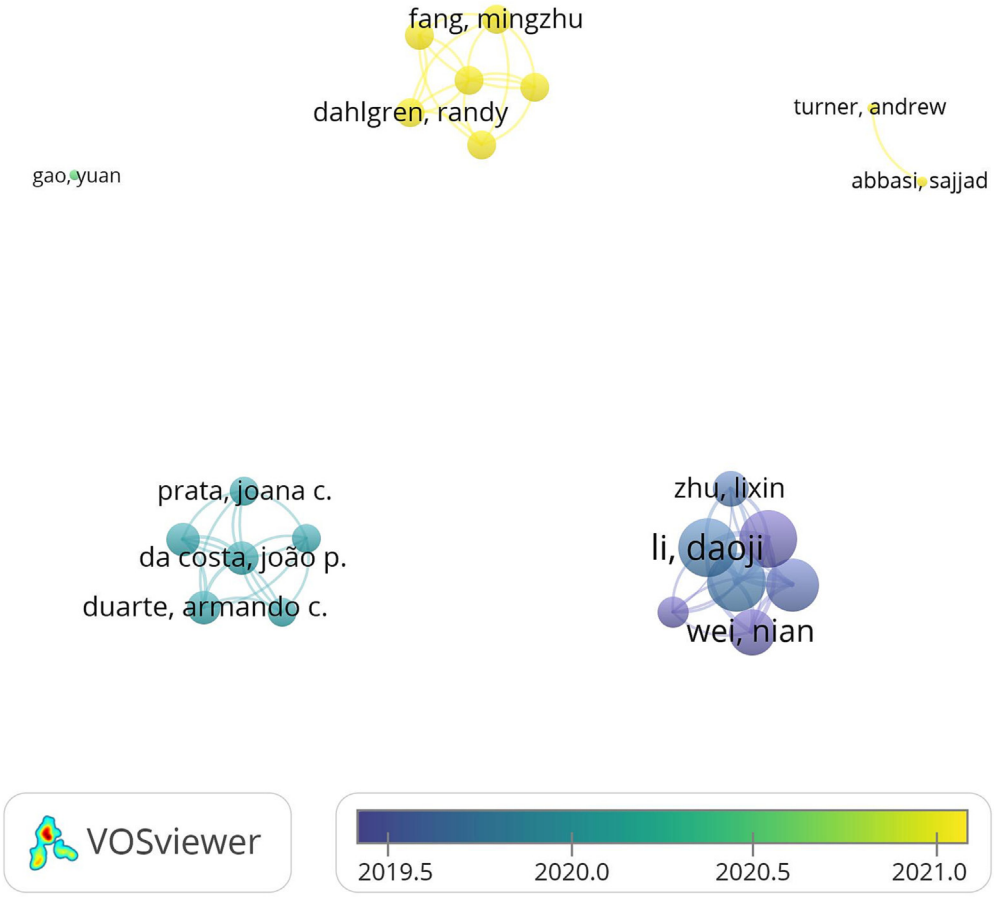


Fig. 6. Bibliometric analysis of the 22 authors included within this literature review and Supplementary Information who had published ≥ 2 publications quantifying microplastics in ambient air.
* The colour and year scale represent the year of publication.
** The differing sizes of the circular markings (dots) represent the varying strength of the actors in the field.
*** Included publications were indisciminately collated based on the topic and did not undergo screening for quality.
**** Industry reports without an associated RIS file were not included in the bibliometric analysis.
***** Two publications were omitted due to lack of associated RIS/Endnote file information capable of importation into the software which inhibited inclusion.

5.2. Ambient air sampling methods and geographical locations

Suspended particulates are sampled using active air samplers, operated using vacuum extraction to pump air through a chamber and collect particulates onto a filter, at low or high volumes (which measure the sampled air volume in L/m^3) (Levermore et al., 2020; Dris et al., 2017) (Fig. 1).

Depending on the model, active air samplers are capable of extracting Total Suspended Particles (TSP), particulates associated with vapour droplets (cloud samplers) (Allen et al., 2020) or can be fitted with restricted inlets to sample specific sizes such as particulate matter with an aerodynamic diameter exclusively ≤10 μm (PM_{10}) and ≤2.5 μm ($PM_{2.5}$). For microplastic analysis, active air samplers for TSP, PM_{10} and cloud samplers (Allen et al., 2020) have been deployed. Novel air samplers have been explored that use custom built equipment, such as the vacuum apparatus designed for aeroplane sampling (González-Pleiter et al., 2021), or the sampler which was designed to deposit collected particulate matter into Eppendorf tubes (Levermore et al., 2020). Biomonitors, such as spiderwebs (Goßmann et al., 2022) and honeybees, have also been explored as particulate air samplers, through the examination of particles to bee bodies (Edo et al., 2021) or within webs for microplastics in ambient air.

Outdoor air has been sampled with active air samplers to collect microplastics in TSP from a number of countries, (Allen et al., 2020; Liu et al., 2019a; Dris et al., 2017; González-Pleiter et al., 2021; Tunahan Kaya et al., 2018; Abbasi, 2019; Gasperi, 2015; Li et al., 2020; Gaston et al., 2020a), over open oceans (Liu et al., 2019b; Liu et al., 2020b; Wang et al., 2020a; Wang et al., 2020b) and above the planetary boundary (González-Pleiter et al., 2021). The specific size fraction of plastics within PM_{10} has been researched in Portugal (Prata, 2020), Spain (Peñalver et al., 2021) and both PM_{10} and $PM_{2.5}$ in Iran (Prata, 2020; Prata et al., 2020a; Akhbarizadeh et al., 2021) and USA (Yao et al., 2022). These studies have extracted samples from locations with varying land uses (Klein and Fischer, 2019; Abbasi, 2019), population densities and elevation (Liu et al., 2019a; Li et al., 2020).

5.3. Outdoor air

5.3.1. Outdoor air - reported concentrations (microplastics/m^3)

5.3.1.1. Outdoor air - terrestrial environment. Concentrations of microplastics over land bodies have been analysed using microscopy (20 %), FTIR (50 %), Raman (25 %) and automated Raman Spectral Imaging technology (5 %) for particle based quantification, with detection limits ranging from 0.8 (Li et al., 2020) to 50 μm (Dris et al., 2015). Concentrations (number of particles/m^3) vary internationally and according to analytical technique, where studies relying on microscopy reported on average 2.5 microplastics/m^3, 353 microplastics/m^3 for FTIR and 438 microplastics/m^3 for Raman techniques. Approximately 43 % of studies feature quantified concentrations of ≤2 microplastics/m^3 (Liu et al., 2019a; Dris et al., 2015;

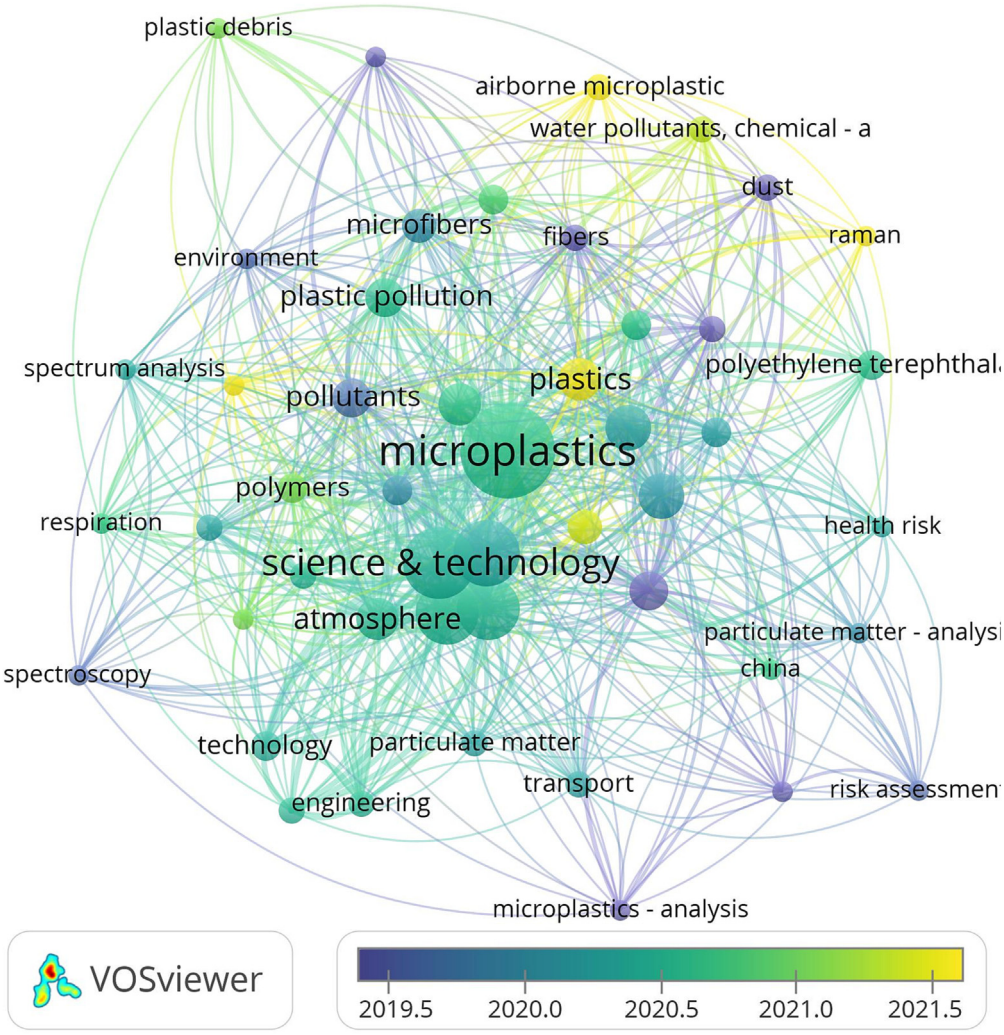


Fig. 7. Bibliometric analysis of the 48 most utilised key terms used across publications included within this literature review and Supplementary Information which quantified microplastics in ambient air, where the term co-occurred across ≥3 publications.
* The colour and year scale represent the year of publication.
** The differing sizes of the circular markings (dots) represents the varying frequency of the key term in the field.
*** Included publications were indisciminately collated based on the topic and did not undergo screening for quality.
**** Industry reports without an associated RIS file were not included in the bibliometric analysis.
***** Two publications were omitted due to lack of associated RIS/Endnote file information capable of importation into the software which inhibited inclusion.

Abbasi, 2019; Gaston et al., 2020a; Prata et al., 2020a; Akhbarizadeh et al., 2021), 33 % between 5 and 100 microplastics/m^3 (Li et al., 2020; Xie et al., 2022; Chen et al., 2022c; Kernchen et al., 2022), 10 % from 100 to 200 microplastics/m^3 (González-Pleiter et al., 2021; Liao et al., 2021; Zhu et al., 2021; Syafei et al., 2019) and 10 % of studies quantified >1000 microplastics/m^3 within TSP. These higher reported concentrations were sampled from a university campus and busy bus station in Turkey (Tunahan Kaya et al., 2018) and within PM_{10} sampled from the United Kingdom (Levermore et al., 2020). Above the planetary boundary (0.5–1.7 km above sea level), concentrations varied from 1.5 to 13.9 microplastics/m^3 and corresponded to the population density below (González-Pleiter et al., 2021). A negative relationship has been observed from urban sampling in China to a coastal town (Liu et al., 2019a), suggesting that the concentration of atmospheric microplastics could be associated with human activity (refer to Fig. 8, SI: Table S10).

Within PM_{10} specifically, quantifications have either not been reported (Prata et al., 2020a; Yao et al., 2022), or reported with great inconsistency, largely the result of analytical difficulties for plastics <10 μm (Prata et al., 2020a). Concentrations have ranged from 0.0065 microplastics/m^3 in Iran (Abbasi, 2023), 0.3–1.1 plastic fibres/m^3 in Iran (Abbasi, 2019), <1 microplastic/m^3 in Portugal (Prata et al., 2020a), 5.2 microplastics/m^3 in Iran (increasing on dusty days) (Akhbarizadeh et al., 2021) to 2502 microplastics/m^3 in the United Kingdom, analysed using Raman Spectral Imaging. Whether or not the higher numbers of microplastic particles within the PM_{10} fraction is related to a higher number of smaller particles remains hypothesised but unconfirmed. Mass based quantification of polystyrene has been undertaken using TGA-MS, calculating means of 36 ng/m^3 from PM_{10} air within Spain (Peñalver et al., 2021) (SI: Table S11).

5.3.1.2. Outdoor air - oceanic environment. Regardless of analytical technique, atmospheric microplastics have been quantified over the open ocean in concentrations featuring much consistency across the globe, being: <LOD – 0.077 microplastics/m^3 (Indian Ocean and South China Sea) (Wang et al., 2020a), 0.01–0.06 microplastics/m^3 (South China Sea) (Ding et al., 2021), 0.1–0–1.37 microplastics/m^3 (Pacific Ocean) (Liu et al., 2019b), and 0.1–0.5 microplastics/m^3 (North Atlantic Ocean) (Trainic et al., 2020). Atmospheric microplastics have also been quantified on the French shore line at 7.7 ± 6.1 microplastics/m^3 (Allen et al., 2020),

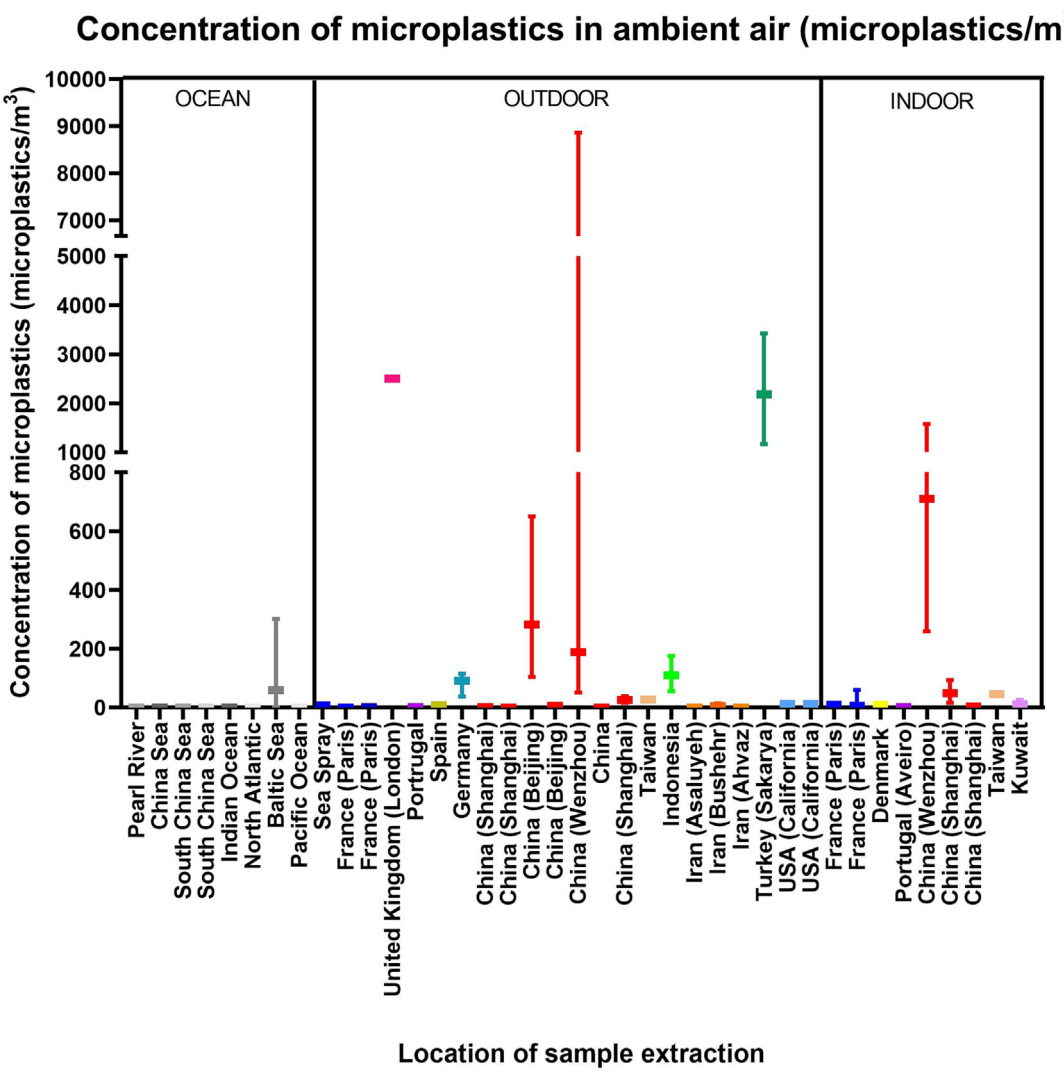


Fig. 8. Concentrations of microplastics reported in ambient air (microplastics/m^3) separated into indoor and outdoor environments and country based on location of sample extraction.
* 'Indoor' refers to ambient air samples extracted within an indoor environment.
** 'Outdoor' refers to ambient air samples extracted from an outdoor environment.
*** 'Ocean' refers to ambient air samples extracted from over an open ocean.
**** Plotted according to mean reported values.
***** Error bars represent minimum and maximum reported values.
****** The same colour is used where multiple values of different sites have been separately reported in the same publication or samples have been taken from the same country.

with abundance negatively correlated to distance from the shoreline (Liu et al., 2019b). Atmospheric microplastics over the open ocean are an order of magnitude lower than the concentrations in the atmosphere of megacities, potentially due to dilution effect (Liu et al., 2019a) or reduced proximity to direct sources, and are variable (possibly episodic) in nature (Wang et al., 2020a) (Fig. 8).

5.3.2. Outdoor air – shape

Within the 21 studies which reported morphology of plastics in outdoor ambient air, 55 % reported fibres as the most abundant shape of the microplastics detected. However, this prevalence is greatly influenced by the exclusivity of reporting only microplastic fibres (Li et al., 2020), analytical method used (where analysis using solely microscopy are more likely to report fibres as the dominant shape) and microplastic size (where fragments dominate the smaller plastic size ranges (Zhu et al., 2021)), rather than geographic location. Studies undertaken more recently, and those which were able to report smaller sized microplastics, observed higher concentrations of fragment shapes (Levermore et al., 2020; Liu et al., 2019a; Liao et al., 2021; Gaston et al., 2020a; Akhbarizadeh et al., 2021; Zhu et al., 2021).

Over the open ocean, fibres constituted 75–80 % (Wang et al., 2020b) of the samples from the Pearl River estuary, South China Sea and Indian Ocean and 60 % over the Pacific Ocean (Liu et al., 2019b). The proportion of fragments detected over these oceans varied between 20 %, 25 % and 30 % of the total sample (Liu et al., 2019b; Wang et al., 2020b). Granules and beads varied between 58 % of the total sample (Liu et al., 2019b) (SI: Table S11, Figs. S17, S18).

5.3.3. Outdoor air – size

Reported size ranges of microplastics in ambient air are influenced by the sampling technique used (e.g. TSP or PM_{10} restrictive inlets which can sample fibres <300 μm (Prata et al., 2020a)). Maximum microplastic

fibre lengths in outdoor air have been observed up to 1475 μm (China) (Zhu et al., 2021), 1650 μm (France) (Dris et al., 2017), 1928 μm (Portugal) (Prata, 2020), 2061 μm (USA) (Gaston et al., 2020a), 2181 μm (China) (Xie et al., 2022), 2191 μm (China) (Liu et al., 2019a) and 8961 μm (USA) (Gaston et al., 2020b), exceeding the microplastic size definition. Within different studies, the most abundant length of fibres in overland outdoor air has been reported as between 520 μm (Liu et al., 2019a), 2–100 μm (Abbasi, 2019), 5–200 μm (Li et al., 2020), 100–500 μm (Gasperi, 2015), 500–1000 μm (Syafei et al., 2019) and <1000 μm (Iran) (Abbasi, 2023). Mean measurements have been reported as 299 and 330 μm in (Portugal) (Prata, 2020; Prata et al., 2020a), 333 ± 306 μm (China), 429 μm (China) (Liu et al., 2019a), 510 ± 283 μm (China) (Yang et al., 2021b) and 616 ± 537 μm (USA) (Gaston et al., 2020b). Ocean based fibres feature a similar upper size limit of <2000 μm (Liu et al., 2019b), 2251 μm (Wang et al., 2020a) and 2986 μm (Liu et al., 2020b). Only one study reported a mean fibre size over the ocean, being comparatively larger than reported means in outdoor air, at 936 ± 557 μm (Wang et al., 2020a). Overall, microplastic fibres have been identified towards the larger end of the size definition of microplastics within outdoor ambient air (Yang et al., 2021b), potentially due to their aerodynamic properties of light weight and drag force which affect settling behaviour.

Terrestrial atmospheric microplastics featuring a fragment shaped morphology have been detected and measured with lengths of up to 59 to 286 μm (ocean) (Wang et al., 2020b), 181 μm (Yang et al., 2021b), 408 μm (Gaston et al., 2020b), <500 μm (Liu et al., 2019b) and 850 μm (Gaston et al., 2020b). Mean lengths have been reported as 77 ± 51 μm (Yang et al., 2021b), 104 ± 69 μm (Gaston et al., 2020b) and 121 μm (Liu et al., 2019d). Up to 67 % of all microplastics quantified were <10 μm (Kernchen et al., 2022). Where smaller PM_{10} particulates have been examined, plastic lengths were a mean of 2–3 μm (Yao et al., 2022). With many studies facing detection limits >10 μm, it is possible that smaller plastics remain unquantified, yet constitute a significant proportion of all plastics present in the outdoor air, in a respirable size. Based on mass quantification, between 0.020.12 % of $PM_{2.5}$ and under was identified to be of microplastic origin (Kernchen et al., 2022).

Within studies that did not differentiate based on shape classifications, the average sizes of microplastics in ambient outdoor air varied between 13 ± 4 μm (Allen et al., 2020), 245 μm (Liu et al., 2019d), 333 ± 306 μm (Yang et al., 2021b) and 358 μm (Liu et al., 2020b) (Fig. 9).

5.3.4. Outdoor air – composition

Compared to natural materials (such as cotton) within ambient air samples, microplastic fibres constituted approximately 4 % of all fibres, however one study included cotton within the reported plastic concentrations (Xie et al., 2022). Clarification of the term microplastic fibre is required, to determine whether fibres of natural origin (such as bamboo, silk, linen or cotton) which are altered with dyes or additives are included. For particles with a fragment shape, 21 % (Wang et al., 2020a), 30 % (Yang et al., 2021b) and 77 % (Liu et al., 2019d) were confirmed to be of plastic origin.

The polymer composition of plastics reported in ambient air samples have included predominantly polyethylene terephthalate (Liu et al., 2019a; Gaston et al., 2020a; Syafei et al., 2019; Abbasi, 2023; Yang et al.,

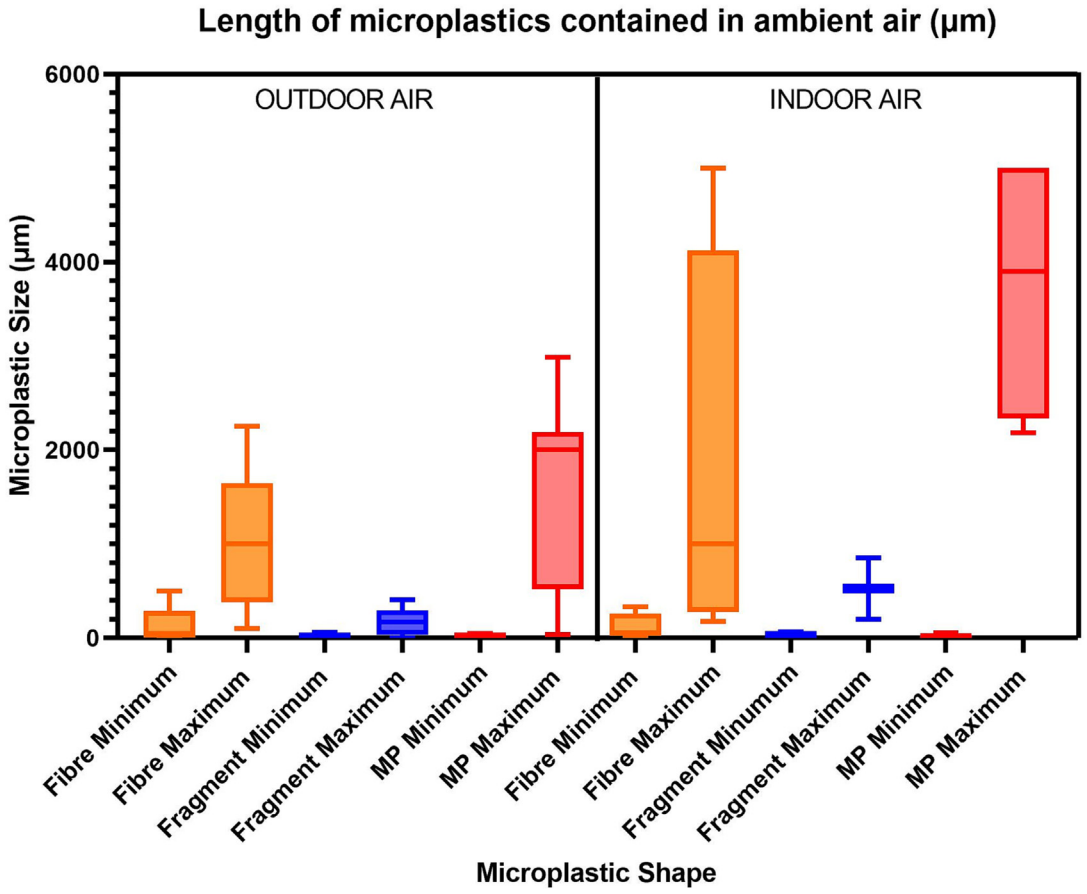


Fig. 9. Mean lengths of microplastics contained in indoor and outdoor ambient air, differentiated by shape.
* Plotted according to mean reported lengths.
** Error bars represent minimum and maximum reported lengths.
*** 'Indoor air' refers to ambient air samples collected within an indoor environment.
**** 'Outdoor air' refers to ambient air samples collected from an outdoor environment.

2021b; Liu et al., 2019d) and polyethylene (Liu et al., 2019a; Tunahan Kaya et al., 2018; Gaston et al., 2020a; Yao et al., 2022; Kernchen et al., 2022; Yang et al., 2021b; Liu et al., 2019d), followed by lesser concentrations of polystyrene (Gaston et al., 2020a; Yang et al., 2021b; Liu et al., 2019d), polypropylene (Allen et al., 2020; Akhbarizadeh et al., 2021; Liu et al., 2019d), polyamide (González-Pleiter et al., 2021; Tunahan Kaya et al., 2018; Liu et al., 2019d), polyvinyl chloride (Gaston et al., 2020a; Gaston et al., 2020b), polyurethane (González-Pleiter et al., 2021; Xie et al., 2022), polymethyl methacrylate (Yang et al., 2021b), cellophane (Syafei et al., 2019), polyacrylonitrile (Liu et al., 2019a), ethylene vinyl acetate (Liu et al., 2019a; Yang et al., 2021b), epoxy resin (Liu et al., 2019a), alkyd resin (Liu et al., 2019d) and polybutadiene (Yang et al., 2021b). Fibre shaped microplastics have been reported to consist predominantly of polyester, polyamide and polypropylene (Zhu et al., 2021). Over the ocean, plastics identified included polyethylene terephthalate (Allen et al., 2020; Liu et al., 2019b) (Wang et al., 2020a), polyethylene (Allen et al., 2020; Liu et al., 2019b; Wang et al., 2020a), polypropylene (Allen et al., 2020; Liu et al., 2019b), polystyrene (Allen et al., 2020; Liu et al., 2019b), polyvinyl chloride (Allen et al., 2020; Liu et al., 2019b), polymethyl methacrylate (Wang et al., 2020a), polyacrylonitrile (Liu et al., 2019b), ethylene vinyl acetate (Wang et al., 2020a), polyvinyl alcohol (Liu et al., 2019b), poly(ethylene-covinyl-acetate) (Wang et al., 2020a) and poly(ethylene-co-propylene) (Wang et al., 2020a).

5.3.5. Relationship between weather conditions and microplastic concentration

Correlations between microplastic particle counts (microplastics/m^3) and total mass of particles collected on the filter have been reported (Liu et al., 2019a; Abbasi, 2019; Akhbarizadeh et al., 2021). Conversely, the relationship between microplastic concentrations in ambient air and humidity remains contested, oscillating between no observed correlation (Abbasi, 2019) (including in $PM_{2.5}$ (Akhbarizadeh et al., 2021)) to a positive correlation (Allen et al., 2020). Microplastics in ambient air have been shown to vary over months, suggesting temporal and seasonal variability (Tunahan Kaya et al., 2018; Abbasi, 2019; Zhu et al., 2021). Preliminary data also suggests the possibility of a diurnal pattern, potentially more prevalent in urban areas, where the concentration of microplastics is lower in the morning, increasing during the midday and early evening period (Zhu et al., 2021) and settle at night (Liu et al., 2019b). However, it remains unknown whether this is an observable pattern and attributable to increased human activity during those times or weather factors such as temperature, humidity or wind speed (Zhu et al., 2021). No relationship has been affirmed between the concentration of microplastics in the air and socioeconomic characteristics such as GDP, moving between a negative relationship and positive relationship which is site specific (Zhu et al., 2021). Further information regarding atmospheric behaviour including precipitation, seasonal variability, temperature and humidity, wind, residence time and in situ degradation, sedimentation behaviour, atmospheric transport, transboundary implications, dilution effect over the ocean, altitude and global loading is contained in the Supplementary Information (SI).

5.3.6. Vertical distribution - microplastic concentrations above ground level

Within the troposphere surrounding China, air analysed for microplastics at various heights from ground level demonstrated consistency in microplastic concentrations between the heights of 1.2 m (5.6 microplastics/m^3) and 18 m (5.7 microplastics/m^3), showing evidence of vertical mixing to this height (Li et al., 2020). In another study, concentrations of plastics at 1.7 m above ground level were reported as 1.5 microplastics/m^3, however microplastics were detected less frequently at heights of 33 m and 80 m above ground level (both averaging ~1 microplastic/m^3) (Liu et al., 2019a), suggesting an altitudinal dilution effect. Observed microplastics were larger towards ground level, where proportionally more fragments and granules were observed (Liu et al., 2019a). As altitude increased to 18 m (Li et al., 2020), 33 m and 80 m respectively, microplastic fibres became proportionally more abundant, potentially due to their lighter densities (Li et al., 2020).

No significant difference in fibre length was observed between ground level and 18 m above ground, with the majority of fibres detected within the respirable range of 5–20 μm and fibres >50 μm rarely observed (Li et al., 2020). Liu et al. (Liu et al., 2019a) found that as altitude increased from ground level to 33 m above ground and further to 80 m above ground, average fibre size and upper size limits of the fibres varied (ground level: mean 598 μm, upper limit 9555 μm; 33 m above ground: mean 176 μm, upper limit 505 μm; 80 m above ground level: mean 830 μm, upper limit 2230 μm) (Liu et al., 2019a). The percentage of microplastic fibres reported in air within China has varied from ~34.6 % of all detected fibres at ~1.5 m to 41.3 % of all fibres being of plastic origin at 80 m height (Li et al., 2020). Polyethylene terephthalate fibres were not detected at 80 m, attributed to its density of 1.45 g/cm^3 (Liu et al., 2019a).

Microplastics have also been detected within the higher troposphere (~2800 m above ground level) in an agricultural town in Spain, at concentrations ranging from 1.5 microplastic/m^3 (mean 1 ± 0 microplastic/m^3) over rural areas to 13.9 microplastics/m^3 (mean 1.71 ± 0.95 microplastic/m^3) above urban areas (González-Pleiter et al., 2021), demonstrating spatial variation at high altitude. Above the planetary boundary, the shape of identified microplastics varied between locations (attributed to differences in local emission sources, the presence of trees, highways and meteorological variability), being predominantly fragments for city and urban samples and fibres in semi-rural and rural samples (González-Pleiter et al., 2021). The most common size of microplastics (both fibres and fragments) was comparatively smaller at the higher altitude, being 30–70 μm, and consisted of polyester, polyamide, polyurethane, polystyrene and undefined polyolefins (González-Pleiter et al., 2021).

5.4. Indoor air

Microplastics within indoor air have been reported in samples collected from France (Dris et al., 2017; Gasperi, 2015), USA (Gaston et al., 2020a), Denmark (Vianello et al., 2019), Portugal (Prata et al., 2020a), China (Hu et al., 2022; Liao et al., 2021; Xie et al., 2022), Taiwan (Chen et al., 2022b) and Kuwait (Uddin et al., 2022), covering microenvironments ranging from residential locations, offices, laboratories, libraries (Gaston et al., 2020a), hospitals, schools (Liao et al., 2021), courtyards and nail salons (Chen et al., 2022b). Indoor air samples have been analysed using microscopy (absent chemical confirmation) (33 %) (Dris et al., 2017; Dris et al., 2015; Prata et al., 2020a; Gasperi, 2015), μ-FTIR (33 %) (Hu et al., 2022; Liao et al., 2021; Gaston et al., 2020a; Chen et al., 2022c), automated μ-FTIR (8 %) (Vianello et al., 2019) and μ-Raman (25 %) (Gaston et al., 2020a; Xie et al., 2022; Uddin et al., 2022). Similar to outdoor air, detection limits ranged in size from 1 μm (Xie et al., 2022) to >50 μm in the earliest works (Dris et al., 2017).

5.4.1. Indoor air - reported concentrations (microplastics/m^3)

Concentrations of microplastics in the TSP of indoor air varied from reported mode concentrations of 5 microplastic fibres/m^3 to mean concentrations of 9 ± 6 microplastics/m^3 (Vianello et al., 2019), 10 ± 6 microplastics/m^3 (Uddin et al., 2022), 46 ± 55 microplastics/m^3 (Chen et al., 2022b) and 48 microplastics/m^3 (Xie et al., 2022) to 1583 ± 1181 microplastics/m^3 (Liao et al., 2021), largely sampled from residential environments. Within the smaller size fraction of PM_{10}, concentrations of <1 microplastic fibre/m^3 (Prata, 2020) and 1.2–11 microplastics/m^3 (mean: 3 microplastics/m^3) (Xie et al., 2022) have been reported. Vertical sampling at 15 m above ground level demonstrated comparatively lower concentrations of 2.3 ± 1.3 microplastics/m^3 at 15 m (Hu et al., 2022) compared to the reported means of other studies with samples extracted from ground level (Fig. 8, SI: Table S10).

Reported concentrations of microplastics in the air increase in number as size detection limits of analytical methods move from 50 μm to <10 μm (Xie et al., 2022). The concentration of microplastics within indoor air depends on ventilation, with a dilution effect if particles move from a more concentrated indoor environment to outdoors (Xie et al., 2022; Chen et al., 2022c). Captured within air conditioner filters, 30 % of all

fibres were identified to be microplastics (1.47–21.4 × 10^2 plastics/cm^2), consisting of polyester (45 %), rayon (28 %) and cellophane (20 %) (Chen et al., 2022c). Air conditioners can act as both a sink (trapping) and a source (emission) of microplastics in indoor air, affecting the concentration of suspended indoor microplastics. Air conditioners are able to trap microplastics when the filter is <30 days from the last clean. However, when the filter is >30 days from the last clean, the filter becomes less porous and releases trapped microplastics into the air, which dilute in concentration as the particle moves further distance from the source air conditioner (Chen et al., 2022c).

5.4.2. Indoor air – shape

In literature (n = 12), fibres were either exclusively reported (Dris et al., 2017; Dris et al., 2015), or constituted the dominant microplastic shape (Gasperi, 2015; Prata et al., 2020a; Xie et al., 2022; Gaston et al., 2020b; Vianello et al., 2019; Prata et al., 2020b). Two studies reported fragments to be the dominant shape of microplastics within indoor air. One study sampled a nail salon and the other examined commercial environments such as an office, school and hospital premises (Chen et al., 2022b; Liao et al., 2021), suggesting that fragment shapes could be associated with commercial locations (SI: Table S10, Figs. S17, S18).

5.4.3. Indoor air – size

The maximum reported length of indoor fibrous microplastics varied from 2181 μm (Xie et al., 2022), 2800 μm (Uddin et al., 2022), 3250 μm (Dris et al., 2017), 3669 μm (Prata, 2020), to >5000 μm (Hu et al., 2022; Gaston et al., 2020b). The reported average length of microplastic fibres ranged between 177 and 237 μm, 250 μm (Prata, 2020), 330, 383 and 641 ± 810 μm (Gaston et al., 2020b), with mode lengths reported as 100–500 μm (Gasperi, 2015) and 101–301 μm (Gaston et al., 2020a). The mean diameter of microplastic fibres has been reported to be 26–30 μm wide (with variability attributed to automatically versus manually calculated measurements) (Vianello et al., 2019).

Microplastic fragment sizes were reported in two studies, being a mean of 68 μm in length and 37 μm in diameter (Vianello et al., 2019) and 58 ± 55 μm (predominantly 76–100 μm) (Gaston et al., 2020a). Another study reported that the majority of plastic fragments within air was <50 μm (Chen et al., 2022b). Importantly, 4 % of all inhalable particles were determined to be of microplastic composition (Vianello et al., 2019). (Fig. 9).

5.4.4. Indoor air – composition

The polymer composition of microplastics in indoor air remains often unreported (Dris et al., 2017; Prata et al., 2020a; Uddin et al., 2022). When reported, identified microplastics have included polyethylene (13 % (Gaston et al., 2020a), 5–28 % (Chen et al., 2022b), 74 % (Xie et al., 2022)), polyvinyl chloride (Xie et al., 2022) (9 % (Chen et al., 2022b), 47 % (Gaston et al., 2020a)), rayon (69 % (Hu et al., 2022)), polypropylene (Hu et al., 2022; Liao et al., 2021; Xie et al., 2022) (0.4–10 % (Vianello et al., 2019)), polyurethane (Hu et al., 2022; Xie et al., 2022) (13 % (Chen et al., 2022b)), polyester (28 % (Liao et al., 2021; Xie et al., 2022), 59–92 % (Vianello et al., 2019)), nylon (0–13 % (Vianello et al., 2019)), rubber (Hu et al., 2022; Xie et al., 2022) (21 % (Chen et al., 2022b)) and acrylic (27 % (Chen et al., 2022b)). Unreported proportions of polyamide (Hu et al., 2022; Liao et al., 2021), resin (Hu et al., 2022; Xie et al., 2022), polystyrene (Gaston et al., 2020a) and polyethylene terephthalate (Hu et al., 2022; Gaston et al., 2020a) have also been identified. Importantly, the polymetric composition appears to be site specific (Hu et al., 2022; Xie et al., 2022).

The composition of plastics in indoor ambient air could be the result of local emissions, and thus potentially vary according to internal space, the volume of human activity and even lifestyle factors such as clothing and textiles. It has been suggested that airborne microplastic concentrations could be site dependant and possibly attributed to different furniture, building materials, cleaning habits and activities between households, more so than seasonality (Dris et al., 2017).

5.5. Indoor v outdoor comparison

Comparing the concentrations of microplastics detected within the ambient air of both indoor and outdoor environments, Dris et al. (2017) and Gaston et al. (2020b) observed a 6 and 6.5 times increase in concentration of microplastics within indoor environments compared to corresponding outdoor environments. Between ambient air sampled from both indoor and outdoor environments, microplastics are smaller outdoors than their indoor counterparts, however indoor environments also demonstrate a higher degree of variability (Fig. 8, SI: Fig. 19).

No statistical difference was observed between the reported minimum size of microplastics in ambient air between all studies ($p > 0.05$), however the mean maximum size of reported microplastics was significantly different between indoor and outdoor air ($p < 0.05$) (Fig. 9, SI: Fig. S20).

6. Deposition and dustfall – (microplastics/m^2/day)

6.1. Bibliometric analysis of literature quantifying microplastics in ambient air

6.1.1. Bibliometric analysis of literature quantifying microplastics in ambient air for authorship prevalence and networks

For the deposition field, 209 individual authors were identified to have worked on quantifying plastics in deposition, globally (Fig. 10), separated into 31 clusters. Of the 209 authors, 18 (9 %) have published two or more publications regarding deposition, clustered into 6 segregated groups (Fig. 11, SI: Table S11).

6.1.2. Bibliometric analysis of literature quantifying microplastics in deposition for key word analysis

Within publications which quantified microplastics in deposition and dustfall, 346 key terms were identified. Of the 346 individual key terms, 53 (15 %) were used across three or more publications, separated into 15 clusters (SI: Table S12).

All key words which were identified across three of more publications originated after 2018, many of which were identified after 2021, suggesting that it is a rapidly evolving field of research. In 2021, key words moved towards terms such as deposition, indoor fallout, atmospheric microplastics, rainfall, weather, precipitation and fibres such as polyamides (Fig. 12).

6.2. Deposition sampling methods and geographical locations

Outdoor deposition is typically sampled as wet (rain, snow), dry or bulk deposition (including both wet and dry weather conditions) over a known duration of time onto a known surface area (cm^2). Deposition samples have been collected using surface wipes or adhesive tape (Salthammer, 2003; Catarino et al., 2018; Thomas et al., 1996). For microplastics analysis, outdoor deposition has been sampled directly into plastic (Klein and Fischer, 2019; Welsh, 2022), metal or glass collection vessels (Dris et al., 2015; Dris et al., 2016b), a wet surface such as wetted filter or water bath (Aurand et al., 1983). Deposition onto biomonitors such as leaves and lichen have also been explored for atmospheric microplastics (Liu et al., 2020a; Loppi et al., 2021). Indoor dustfall has been collected for microplastic analysis using metal (Ouyang et al., 2022) or glass (Soltani et al., 2021) collection vessels and adhesive pads (Catarino et al., 2018) (Fig. 1).

Outdoor deposition has been studied for microplastics in Europe (France (Dris et al., 2016a; Dris et al., 2015; Allen et al., 2019), Germany (Klein and Fischer, 2019; Kernchen et al., 2022), Ireland (Roblin et al., 2020), Poland (Szewc et al., 2021), Spain (González-Pleiter et al., 2021), United Kingdom (Wright et al., 2020; Stanton et al., 2019)), North America (Canada (Welsh, 2022), USA (Brahney et al., 2020; Yao et al., 2022)), South America (Brazil (Amato-Lourenço et al., 2022)), Asia (China (Cai et al., 2017; Tian Yuan et al., 2020), Tibet (Zhang et al., 2021; Dong et al., 2021), Turkey (Tunahan Kaya et al., 2018)), Oceania (Indonesia (Purwiyanto et al., 2022), Vietnam (Thinh, 2020)) and over

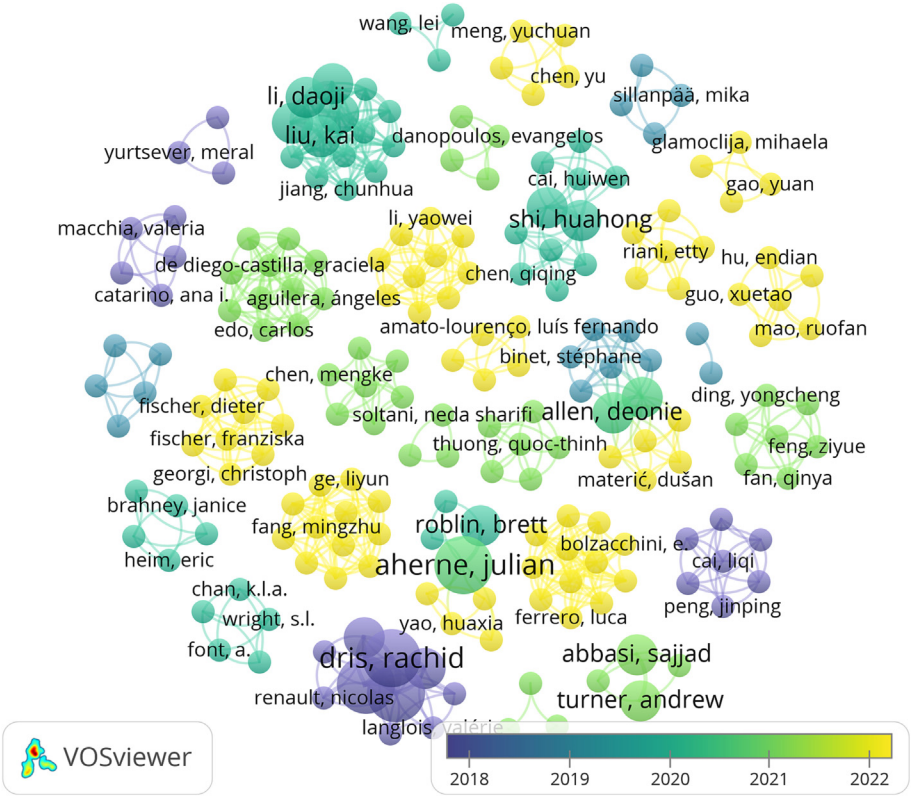


Fig. 10. Bibliometric analysis of authorship prevalence and networks for all 209 authors included within this literature review and Supplementary Information who had publications covering the quantification of plastics in deposition and dustfall.
* The colour and year scale represent the year of publication.
** The differing sizes of the circular markings (dots) represent the varying strength of the actors in the field.
*** Included publications were indiscriminately collated based on the topic and did not undergo screening for quality.
**** Industry reports without an associated RIS file were not included in the bibliometric analysis.
***** Two publications were omitted due to lack of associated RIS/Endnote file information capable of importation into the software which inhibited inclusion.

various oceans (Liu et al., 2019b; Ferrero et al., 2022; Ding et al., 2021). Of the 31 studies examining outdoor deposition, 91 % sampled bulk deposition as the combination of both wet and dry deposition, 6 % examined wet only deposition (particles/L) sampled from glaciers (Zhang et al., 2019) and ice flows (Bergmann, 2019) (formed over a known duration of time) in Tibet (Zhang et al., 2019), from the Arctic and German Alps (Bergmann, 2019) and 3 % examined only dry deposition (Ding et al., 2021).

The majority of research which examined microplastics in deposition collected samples using some form of bottle, box or funnel (65 %), followed by rain gauges (10 %). Approximately 35 % of samples were collected into glass containers, 32 % into metal containers, 3 % into plastic containers and 29 % of sample collection apparatus remain unreported. Only the first published study quantified microplastics in deposition using exclusively microscopy (Dris et al., 2015). Subsequent work has since utilised microscopy in parallel with other complementary techniques, including FTIR (66 %), Raman (23 %) and one study quantified the mass of plastics in deposition using mass spectrometry (3 %) (Allen et al., 2022). Limits of detection for spectroscopic techniques varied from being unreported (Tunahan Kaya et al., 2018) to 1.6 μm (Huang et al., 2022) to 300 μm (Truong et al., 2021).

Indoor deposition/dustfall has been examined from France (Dris et al., 2017), UK (Catarino et al., 2018; Jenner et al., 2021), China (Zhang et al., 2020a; Fang et al., 2022), Australia (Soltani et al., 2021), USA (Yao et al., 2022) and Brazil (Amato-Lourenço et al., 2022). Of the nine studies, 11 % collected samples using adhesive pads, 22 % incorporated metal apparatus, 44 % used glass equipment and 22 % failed to report the medium of sample collection. Regarding analytical techniques, Raman (89 %) and FTIR (9 %) instrumentation was used for analysis. Limits of detection ranged from particle sizes of 2.2 μm (Yao et al., 2022) to <50 μm (Soltani et al., 2021).

6.3. Outdoor deposition

6.3.1. Outdoor deposition – reported concentrations (microplastics/m^2/day)

Concentrations of microplastics in wet, dry, bulk and modelled deposition (microplastics/m^2/day) have varied greatly in literature. This could be attributed to differences between reporting units (which include total fibre counts including those of natural origin and specifically microplastic fibre counts post chemical confirmation), differences in analytical techniques and population densities.

Segregated by continent, mean concentrations of microplastics in deposition from South America have been reported to be 123 ± 47 microplastics/m^2/day (Amato-Lourenço et al., 2022) (Brazil) and 384 microplastics/m^2/day in the Middle East (Turkey) (Tunahan Kaya et al., 2018). Studies which examined microplastics in deposition from samples collected within Northern America reported mean concentrations of 7 microplastics/m^2/day (Canada) (Welsh, 2022) and 329 ± 19 microplastics/m^2/day (USA) (Yao et al., 2022).Within samples collected across Europe, concentrations of microplastics have been reported between 0.5 and 2.9 microplastic fibres/m^2/day (Trent, UK) (Stanton et al., 2019) to 771 ± 167 microplastics/m^2/day) (London, UK) (Wright et al., 2020). In Asia, concentrations of microplastics in deposition were 0.6–1 microplastics/m^2/day (Vietnam) (Truong et al., 2021) and 1357 microplastics/m^2/day (Vietnam landfill) (Thinh, 2020). Oceanic microplastics have been reported with mean concentrations between ~0.6 microplastics/m^2/day (China, coastal) (Liu et al.,

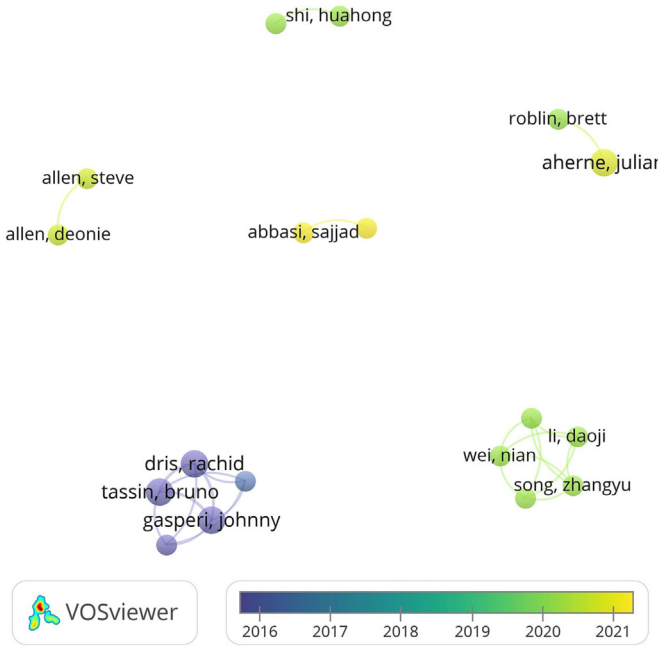


Fig. 11. Bibliometric analysis of the 18 authors included within this literature review and Supplementary Information who had published ≥ 2 publications quantifying microplastics in deposition and dustfall.
* The colour and year scale represent the year of publication.
** The differing sizes of the circular markings (dots) represent the varying strength of the actors in the field.
*** Included publications were indiscriminately collated based on the topic and did not undergo screening for quality.
**** Industry reports without an associated RIS file were not included in the bibliometric analysis.
***** Two publications were omitted due to lack of associated RIS/Endnote file information capable of importation into the software which inhibited inclusion.

2019b; Liu et al., 2020b) to 87 ± 30 microplastics/m^2/day (ocean) (Ferrero et al., 2022) (SI: Table S13, Fig. 13).

Based on known snow volumes and time duration, deposition in the Arctic has been calculated to be 0–2 microplastics/m^2/per annum (Svalbard), 8.8 ± 7.9 microplastics/m^2/per annum (Fram Straight), 66 ± 60 microplastics/m^2/per annum (Alps) (Bergmann, 2019) and 8760 ± 5840 microplastics/m^2/annum in a remote glacier in Tibet (Dong et al., 2021).

Discrepancy exists between reported concentrations of microplastics in deposition. As most of the plastics identified within deposition samples are hypothesised to originate from local sources, surrounding land characteristics are thought to influence both concentration and polymer composition of microplastics in deposition (Purwiyanto et al., 2022). Urban to rural microplastic concentrations in deposition vary significantly in Germany (Kernchen et al., 2022). Separated by land use, Klein and Fischer (2019) observed lower mean concentrations of microplastics over residential locations (137, 245 and 261 microplastics/m^2/day), compared to rural locations (331, 343 and 512 microplastics/m^2/day), demonstrating variability of microplastics between land uses, not necessarily associated with the surrounding population density (Kernchen et al., 2022). Conversely, microplastics in wet deposition on ice floes in Fram Straight (8.8 ± 7.9 microplastics/m^2/day), Svalbard (1.4 ± 0.04 microplastics/m^2/day) and an urban area of the Alps (66 ± 60.1 microplastics/m^2/day) suggest an influence from population (Bergmann, 2019), as does the positive correlation between microplastic concentrations and distance from land in open ocean studies (Liu et al., 2019b). In addition to the macro variables mentioned, variability of reported concentrations could also be attributed to the wide range of limits of detection and sampling at differing heights (1.2 m to 28 m (Purwiyanto et al., 2022)), which can also impact reported plastic sizes.

6.3.2. Outdoor deposition – shape

Within atmospheric deposition samples collected from across the globe, fibres have been the predominant shape of microplastics identified in Ireland (Roblin et al., 2020), Turkey (50 %) (Tunahan Kaya et al., 2018), Poland (60 %) (Szewc et al., 2021), Vietnam (64 % (Thinh, 2020), 98 % (Truong et al., 2021)), USA (66–70 %) (Brahney et al., 2020), Canada (86 %) (Welsh, 2022), France (90 %) (Dris et al., 2015), China (90 % (Cai et al., 2017) (Tian Yuan et al., 2020), 95 % (Zhou et al., 2017)), Iran (95 %) (Abbasi and Turner, 2021a), the UK (92 % (Wright et al., 2020), 93.8 % (Stanton et al., 2019)) and oceanic samples (98 %) (Ferrero et al., 2022) (SI: Table S13, Figs. S17, S18).

Fragments have been reported as the main microplastic shape in USA (Yao et al., 2022), Brazil (74 %) (Amato-Lourenço et al., 2022), the French Pyrenees Mountains (86 %) (Allen et al., 2019), Indonesia (86 %) (Purwiyanto et al., 2022) and in Germany (88–97 % (Klein and Fischer, 2019), 97 % (Kernchen et al., 2022)). In the Pyrenees Mountains, Allen et al. (2019) noted that, if consistent with other studies examining particles ≥200 μm, fibres would constitute 70 % of the sample. Most fragments were <25 μm (Allen et al., 2019), which is below the detection limit of many studies. In Germany, the percentage of fragment shaped microplastics in deposition increased as samples along a transect moved further from areas of higher population density (Klein and Fischer, 2019). The proportion of fragments and fibres were approximately equal in deposition sampled within Turkey (Tunahan Kaya et al., 2018) and China (Huang et al., 2022; Liu, 2022). Both fragments and fibres have been identified in Tibetan glaciers, their proportions unreported (Zhang et al., 2019).

Critically, microplastics isolated based on visual identification techniques prior to chemical confirmation could underestimate the proportion of fragment shaped microplastics, which are susceptible to analytical difficulties based on their typically smaller size.

6.3.3. Outdoor deposition - size

The length of microplastic fibres in deposition varied based on analytical method and different size groupings used between studies. In Dris et al. (2015), initially 49 % of fibres were measured between 1000 and 5000 μm. (Dris et al., 2015), however in subsequent work, fibres were predominantly between 200-400 μm and 400–600 μm (Dris et al., 2016b). In other studies, mode fibre lengths were reported to be 100–300 μm (China) (Zhou et al., 2017), 400–500 μm (Wright et al., 2020), 500–1000 μm (China) (Ding et al., 2021), 700–1200 μm (China) (Cai et al., 2017) and <900 μm (Poland) (Szewc et al., 2021). Mean microplastic fibre lengths have been reported as 667 to 840 μm (Vietnam) (Truong et al., 2021), <1 mm (Vietnam) (Tian Yuan et al., 2020), 880 μm (Ireland) (Roblin et al., 2020) and 905 ± 641 μm (UK) (Wright et al., 2020), with the mean skewed by fewer fibres of larger sizes (Roblin et al., 2020). The upper size limit of microplastic fibres in deposition samples has been reported as 925 μm (Purwiyanto et al., 2022), 1000 μm (Ding et al., 2021; Tian Yuan et al., 2020), 1945 μm (Kernchen et al., 2022), 2725 μm (Amato-Lourenço et al., 2022), 3000 μm (Allen et al., 2019; Brahney et al., 2020) to exceeding 5000 μm (Huang et al., 2022; Thinh, 2020).

The diameter of fibres has been reported between 7 and 15 μm (France) (Dris et al., 2016b), 24 ± 10 μm (UK) (Wright et al., 2020) and 17 ± 2 μm (ocean) (Ferrero et al., 2022), demonstrating consistency across geographic regions despite analytical differences. Variability could be attributed to fabric and clothing differences between locations, if these are sourced from human particle emissions.

In studies where fragment shapes were reported separately, mean size measurements were smaller than fibres, being <25 μm (Pyrenees mountains, 80 % <50 μm) (Allen et al., 2019), 51 μm (China, (63–75 % <50 μm) (Huang et al., 2022), <63 μm (Germany, 98 % <100 μm) (Klein and Fischer, 2019) and <190 μm (Poland) (Szewc et al., 2021). In the UK, all microplastics (excluding fibres) were <350 μm, with the mode size between 75 and 100 μm (Wright et al., 2020). In the USA non-fibrous

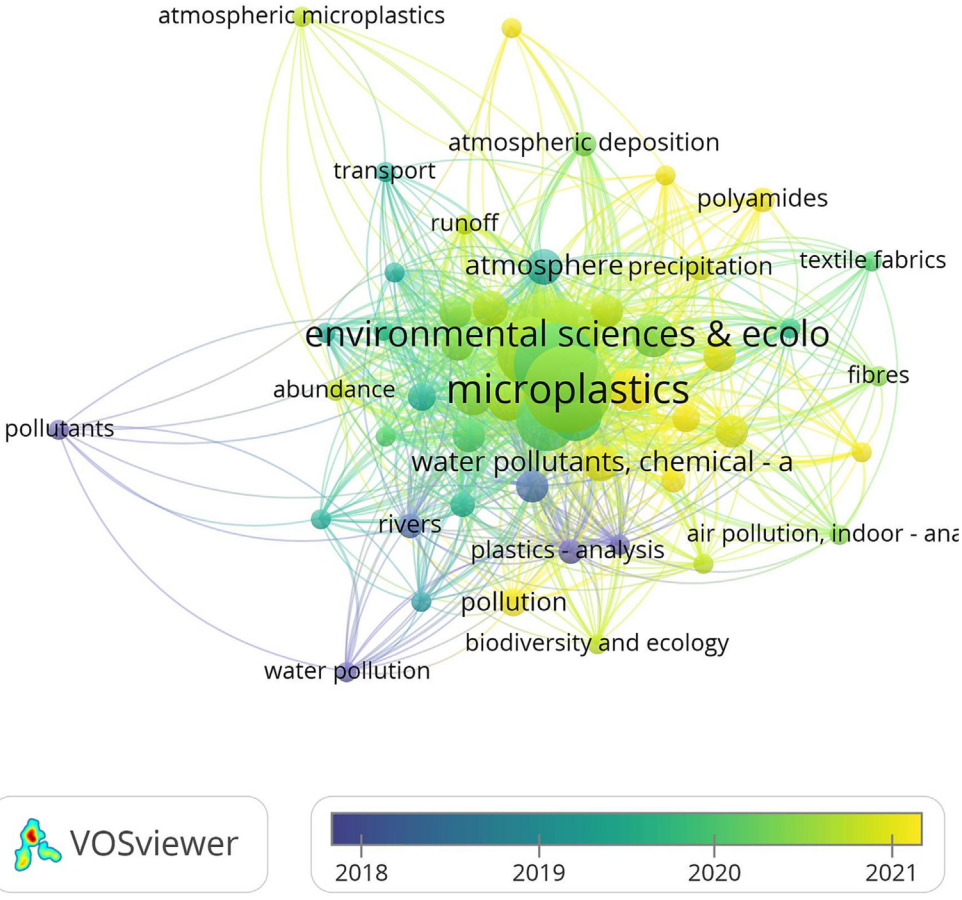


Fig. 12. Bibliometric analysis of the 53 most utilised key terms used across publications included within this literature review and Supplementary Information which quantified microplastics in deposition and dustfall, where the term co-occurred across >3 publications.
* The colour and year scale represent the year of publication.
** The differing sizes of the circular markings (dots) represents the varying frequency of the key term in the field.
*** Included publications were indisciminately collated based on the topic and did not undergo screening for quality.
**** Industry reports without an associated RIS file were not included in the bibliometric analysis.
***** Two publications were omitted due to lack of associated RIS/Endnote file information capable of importation into the software which inhibited inclusion.

microplastics ranged from 4 to 188 μm, being substantially smaller than the fibres measured (20–3000 μm) (Brahney et al., 2020). Spheres have been reported to be <200 μm (Ding et al., 2021; Dong et al., 2021), film shaped plastics from <200 μm (Ding et al., 2021), 390 μm (87 %) (Szewc et al., 2021) to between 10 and 1520 μm (Szewc et al., 2021) and granules as <40 μm (Huang et al., 2022). The upper size limit of non-fibrous plastics has been reported anywhere from 522 μm (Kernchen et al., 2022) to 3480 μm (Welsh, 2022). It is not only possible, but even plausible that suspended fragments/granule/sphere shaped plastics (which have a higher particle density than fibres) will be smaller in size than suspended microplastic fibres.

6.3.4. Outdoor deposition – composition

In reported deposition samples, 15 % (Roblin et al., 2020), 17 % (Wright et al., 2020), 23 % (Cai et al., 2017), 29 % (Dris et al., 2016a) and 33 % (Huang et al., 2022) of selected fibres were confirmed to have petrochemical composition. Chemically characterised microplastic fibres have been identified to include polyethylene terephthalate (Zhou et al., 2017; Dris et al., 2016a; Brahney et al., 2020; Kernchen et al., 2022) (5 % (Ferrero et al., 2022), 9 % (Amato-Lourenço et al., 2022), 19 % (Wright et al., 2020; Welsh, 2022), >30 % (Tian Yuan et al., 2020), 41 % (Szewc et al., 2021), 62 % (Dong et al., 2021), 71 % (Roblin et al., 2020), 82 % (Purwiyanto et al., 2022)), polyethylene (Zhou et al., 2017; Brahney et al., 2020) (1 % (Dong et al., 2021), 8 % (Purwiyanto et al., 2022), 11 % (Roblin et al., 2020), 12 % (Ferrero et al., 2022), 14 % (Cai et al., 2017) and 14 % (Szewc et al., 2021)), polypropylene (Ding et al., 2021) (~2.5 % (Wright et al., 2020), 4 % (Roblin et al., 2020; Dong et al., 2021), 9 % (Cai et al., 2017), 18 % (Szewc et al., 2021) and 91 % (Truong et al., 2021)), polystyrene (Cai et al., 2017) (3 % (Purwiyanto et al., 2022), 4 % (Cai et al., 2017)), polyurethane (Dris et al., 2016a) (~2.5 % (Wright et al., 2020), 5 % (Ferrero et al., 2022)), polyamide (Dris et al., 2016a; Kernchen et al., 2022) (3 % (Ferrero et al., 2022), 9 % (Wright et al., 2020), 24 % (Welsh, 2022), polyvinyl chloride (14 % (Szewc et al., 2021) (9 % (Amato-Lourenço et al., 2022)), polyacrylonitrile (artificial wool (Liu et al., 2019a; Kernchen et al., 2022))(2 % (Dong et al., 2021), 6 % (Amato-Lourenço et al., 2022), 11 % (Roblin et al., 2020) and 67 % (Wright et al., 2020)), polyester (Ding et al., 2021) (40 % (Ferrero et al., 2022), 76 %, (Amato-Lourenço et al., 2022)), rayon (Ding et al., 2021), nylon (30 %) (Dong et al., 2021), polycarbonate (36 % (Ferrero et al., 2022)), polybutadiene (8 % (Purwiyanto et al., 2022)) and polytetrafluoroethylene (PTFE) (possibly from travel products in the USA) (Brahney et al., 2020).

Fragment shaped microplastics contained in deposition samples have been identified as polyethylene (Yao et al., 2022)(4–24 % (Kernchen et al., 2022), 48 % (Klein and Fischer, 2019), 67 % (Amato-Lourenço et al., 2022)), ethyl vinyl acetate copolymer (22 % (Klein and Fischer, 2019)), polystyrene (2–16 % (Kernchen et al., 2022), 8 % (Amato-Lourenço et al., 2022), 19 % (Wright et al., 2020)), polypropylene

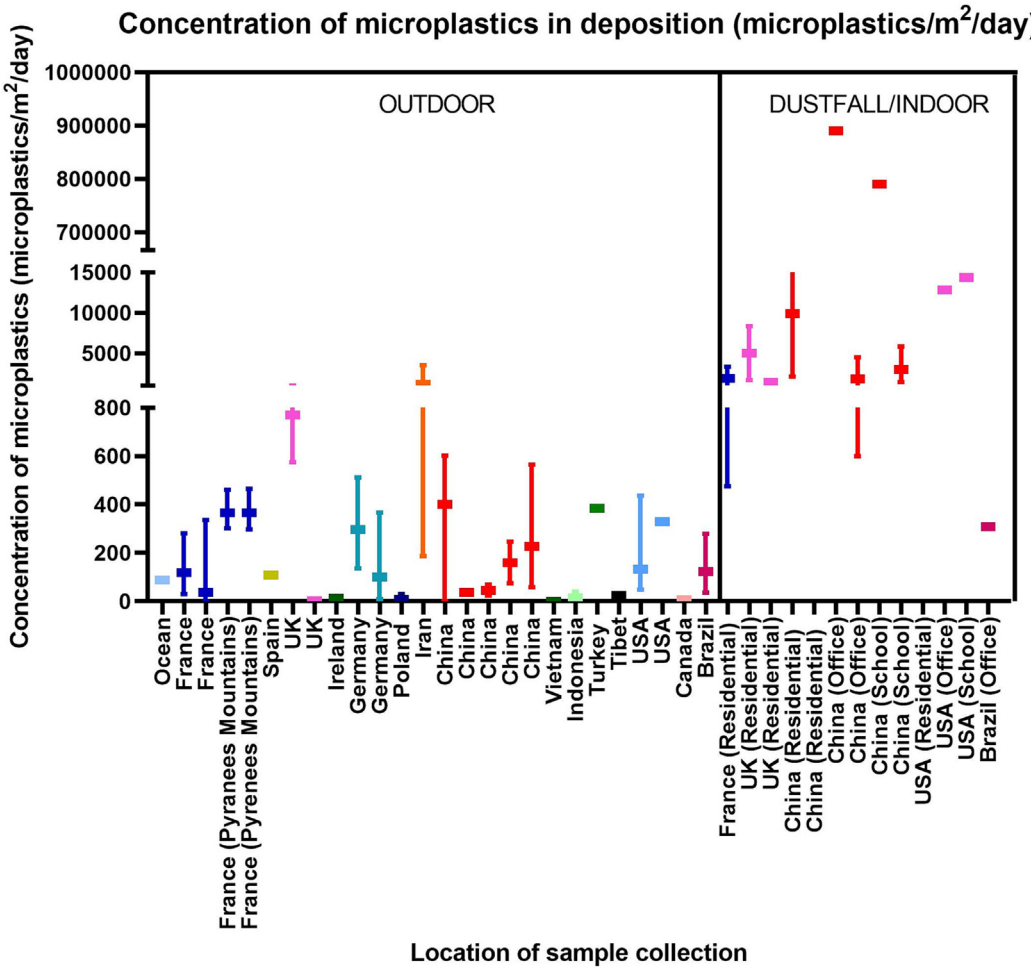


Fig. 13. Concentrations of microplastics reported in deposition (microplastics/m^2/day) separated based on location of sample collection within an indoor or outdoor environment.
* 'Indoor' refers to deposition samples collected within an indoor environment.
** 'Outdoor' refers to deposition samples collected from an outdoor environment.
*** Plotted according to mean reported values.
**** Single data points are the reported mean concentrations, absent reported minimum and maximum concentrations.
***** Error bars represent minimum and maximum reported values.
****** The same colour is used where multiple values of different sites have been separately reported in the same publication.

(Wright et al., 2020; Szewc et al., 2021) (35–65 % (Kernchen et al., 2022), polyethylene terephthalate (Wright et al., 2020; Szewc et al., 2021; Amato-Lourenço et al., 2022) (0–41 % (Kernchen et al., 2022), 25 % (Amato-Lourenço et al., 2022)), polyurethane (Wright et al., 2020), polyvinyl chloride (Wright et al., 2020; Amato-Lourenço et al., 2022) (2–11 % (Kernchen et al., 2022), 67 % (Yao et al., 2022)), polymethyl methacrylate (Szewc et al., 2021), polyvinyl acetate, ethylene acrylic copolymer (Brahney et al., 2020). Beads have been reported usually of polymethyl methacrylate composition (Brahney et al., 2020), whilst film shaped microplastics have been reported as cellophane, polyethylene and polypropylene in composition (Szewc et al., 2021).

6.3.5. Outdoor deposition - mass of deposition

Applying a density of 1 g/cm^3 for polyamide and 1.45 g/cm^3 for polyethylene terephthalate, both widely used in textiles, it has been estimated that between 3 and 10 t of synthetic fibres could be deposited from the atmosphere annually onto the 2500 km^2 area of greater Paris (Dris et al., 2016a). In Germany, deposition has been calculated to be 0.06 to 50 kg/km^2/per annum (Kernchen et al., 2022). In China along the Yantai coastline (100 km) and 1.6 km inland, 0.9 to 1.4 t of microplastics were calculated to be deposited annually (Zhou et al., 2017) and >1000 t of microplastics deposited per annum onto the national parks within the western regions of the USA (Brahney et al., 2020). Based on assumptions of size and plastic density proportions, total suspended microplastics in Shanghai were calculated to weigh approximately 121 kg in the air (Liu et al., 2019a). Calculated from the concentration of 11.9 microplastics/m^3 in the atmosphere, modeling showed that approximately 100–117 microplastics would be deposited per m^2/day, demonstrating relationships between suspended and deposited microplastics (González-Pleiter et al., 2021).

Recently, tree coverage was demonstrated to be a mitigating factor affecting the concentration of microplastics contained in deposition. Trees collected 1 g/km^2/day of plastic fragments and 5 g/km^2/day of plastic fibres (Huang et al., 2022), reducing deposition concentrations detected beneath the canopy.

6.4. Indoor deposition/dustfall

6.4.1. Indoor dustfall - reported concentrations (microplastics/m^2/day)

Indoor deposition/dustfall has been examined in nine studies. Samples have been analysed from different geographical locations including Europe (France (Dris et al., 2017), UK (Catarino et al., 2018; Jenner et al., 2021)), North America (USA (Yao et al., 2022)), South America (Brazil (Amato-

Lourenço et al., 2022)), Asia (China (Zhang et al., 2020a; Ouyang et al., 2022; Fang et al., 2022)) and Oceania (Australia (Soltani et al., 2021)), and collected from different indoor environments including residential locations (Dris et al., 2017; Zhang et al., 2020a; Catarino et al., 2018; Yao et al., 2022; Soltani et al., 2021; Jenner et al., 2021; Fang et al., 2022), schools (Yao et al., 2022; Ouyang et al., 2022; Fang et al., 2022) and office environments (Dris et al., 2017; Yao et al., 2022; Fang et al., 2022).

Within indoor environments, microplastic fibres in indoor deposition have been reported with concentrations ranging between 475-3339 microplastic fibres/m^2/day (Dris et al., 2017) and 22–6169 microplastic fibres/m^2/day (Soltani et al., 2021), or the mean concentration of 19,600 ± 900 microplastic fibres/m^2/day (Yao et al., 2022) in residential locations. Film shaped plastics have also been reported in indoor deposition at concentrations of 5990 ± 2020 microplastic fibres/m^2/day (Yao et al., 2022). Undifferentiated by shape, total concentrations of microplastics in deposition within indoor residential environments have been reported as 1666–8333 microplastics/m^2/day (Catarino et al., 2018) and 940,000 ± 500,000 microplastics/m^2/day (Fang et al., 2022) (Fig. 13).

Within office environments, the concentration of microplastic fibres in deposition has been reported to vary between 307 ± 215 microplastic fibres/m^2/day (Amato-Lourenço et al., 2022) and 6430 ± 2540 microplastic fibres/m^2/day (Yao et al., 2022). Fragment shaped microplastics in deposition have also been reported at concentrations of 2.2 ± 2 microplastics/m^2/day (Amato-Lourenço et al., 2022) and 6340 ± 2540 microplastics/m^2/day (Zhang et al., 2020a; Yao et al., 2022). Undifferentiated by shape, the concentration of microplastics in deposition within office locations has been reported as 890,000 ± 340,000 microplastics/m^2/day (Fang et al., 2022).

School environments have featured microplastic fibre concentrations of 1404–5844 microplastic fibres/m^2/day (Ouyang et al., 2022) and 6200 ± 570 microplastic fibres/m^2/day (Yao et al., 2022). Film shaped microplastics have also been reported in schools at the concentration of 8130 ± 2170 microplastics/m^2/day (Yao et al., 2022). Total microplastic concentrations within school based deposition (irrespective of shape) has been reported to be 790,000 ± 350,000 microplastics/m^2/day (Fang et al., 2022).

Concentrations of microplastics within indoor dustfall were higher during activities such as cooking (Catarino et al., 2018) and increased human activity, such as the activities associated with a cafeteria environment (Zhang et al., 2022). A relationship has also been observed regarding human occupation and activity, where the abundance of microplastics within a residential dormitory over the weekend (compared to weekdays) was higher and the opposite noted for the office environment (Zhang et al., 2020a).

6.4.2. Indoor dustfall – shape

Microplastics in dustfall studies have often omitted microplastic shape classifications (Dris et al., 2017; Catarino et al., 2018) or reported microplastic fibres exclusively (Zhang et al., 2020a). Where the shape of microplastics in dustfall has been reported, microplastic fibres constituted 54 % (Yao et al., 2022), 90 % (Jenner et al., 2021) and 99 % (Ouyang et al., 2022; Soltani et al., 2021) of all microplastics examined. The proportion of fragment shaped microplastics has varied in literature, ranging from <1 % (Ouyang et al., 2022; Soltani et al., 2021), 8 % (Jenner et al., 2021), 64 % (Amato-Lourenço et al., 2022) to 67–87 % (Yao et al., 2022). Spherical shaped microplastics have been identified in deposition (1 % (Jenner et al., 2021)). Other microplastic shapes have also been identified in dustfall, including film shapes (0.2 % (Soltani et al., 2021)), foams (23 %) and granules (4 %) (Amato-Lourenço et al., 2022) (SI: Table S13, Figs. S17, S18).

6.4.3. Indoor dustfall – size

Within dustfall, microplastic fibres have typically occurred in the larger sizes (>50 μm (Soltani et al., 2021)), ranging between 50 and 510 μm (Amato-Lourenço et al., 2022), 50–2000 μm (Zhang et al., 2020a) and 4650–4850 μm (Dris et al., 2017). Microplastic fibre diameters have been reported to be 11–20 μm (63 %) (Jenner et al., 2021) and 19 ± 6 μm, larger compared to their natural counterparts at 17 ± 4 μm (Soltani et al., 2021).

Fragments of microplastic have been detected between 50-200 μm (Soltani et al., 2021), however can range up to 50–989 μm (Amato-Lourenço et al., 2022). Proportionally for each shape, the majority of plastics in the 5–250 μm range were film (88 %), fragment (99 %) and sphere (98 %) (Jenner et al., 2021). The overarching trend in deposition appears to be that the highest proportion of microplastics examined, occur in the smaller size ranges analysed (Jenner et al., 2021; Fang et al., 2022) (SI: Table S13).

6.4.4. Indoor dustfall – composition

In dustfall, microplastics have been identified to include polyethylene terephthalate (Soltani et al., 2021) (26 % (Yao et al., 2022), 49 % (Ouyang et al., 2022), 62 % (Jenner et al., 2021), >90 % (Zhang et al., 2020a)), polypropylene (Zhang et al., 2020a) (4 % (Jenner et al., 2021), 7 % (Yao et al., 2022), 13 % (Ouyang et al., 2022), 14 % (Fang et al., 2022)), polystyrene (Zhang et al., 2020a) (17 % (Fang et al., 2022), 20 % (Yao et al., 2022)), polyacryl nitrile (3 % (Jenner et al., 2021), 23 % (Ouyang et al., 2022)), polyethylene (3 % (Jenner et al., 2021), 25 % (Soltani et al., 2021), 27 % (Fang et al., 2022), 40 % (Yao et al., 2022)), polyester (17 % (Soltani et al., 2021)), polymethyl methacrylate (3 % (Jenner et al., 2021)), polyvinyl chloride (2 % (Ouyang et al., 2022), 7 % (Yao et al., 2022), 15 % (Soltani et al., 2021)) and polyamide (Zhang et al., 2020a) (6 % (Jenner et al., 2021), 14 % (Ouyang et al., 2022; Soltani et al., 2021)) polymer compositions.

Approximately 29, 33 % (Dris et al., 2017; Dris et al., 2016b), 33 % (Dris et al., 2017), 33 % (Jenner et al., 2021), 35–37 % (Zhang et al., 2020a) and 37–43 % (Ouyang et al., 2022) of isolated fibres were analysed to be of petrochemical origin. Microplastic fibres were identified to be polyethylene terephthalate (45 % (Fang et al., 2022), >90 % (Zhang et al., 2020a)), polypropylene (Zhang et al., 2020a), polystyrene (Zhang et al., 2020a), polyamide (Zhang et al., 2020a) (20 % (Fang et al., 2022)), polyethylene (12 % (Fang et al., 2022)) and polyester (Fang et al., 2022).

6.5. Indoor v outdoor comparison

Between compartments, a significant difference ($p < 0.05$) was identified between indoor and outdoor concentrations of microplastics (microplastics/m^2/day) (Fig. 13, SI: Fig. S19). No significant difference was observed between the minimum nor maximum reported sizes of microplastics in outdoor deposition and dustfall ($p > 0.05$) (Fig. 14, SI: Fig. S20).

7. DUST (microplastic particles/g)

7.1. Bibliometric analysis of literature quantifying microplastics in dust

7.1.1. Bibliometric analysis of literature quantifying microplastics in dust for authorship prevalence and networks

Quantification of microplastics in dust has been explored by 224 authors, segregated into 24 clusters (Fig. 15). Of all 224 identified authors, 24 (11 %) have published two or more articles regarding microplastics in dust, separated into 7 distinct clusters (Fig. 16, SI: Table S14). Much of the work covering microplastics in dust was published between 2019 and 2021, with little produced in 2022.

7.1.2. Bibliometric analysis of literature quantifying microplastics in dust for key word analysis

Key word analysis identified 404 individual key words used for publications involving microplastics in dust, 48 of which (12 %) were identified across three of more publications and divided into 11 clusters (Fig. 17, SI: Table S15). Key words within the field moved from dust and exposure towards terms including analysis, polyethylene terephthalate, human exposure, environmental monitoring, snow and glacier subject matter.

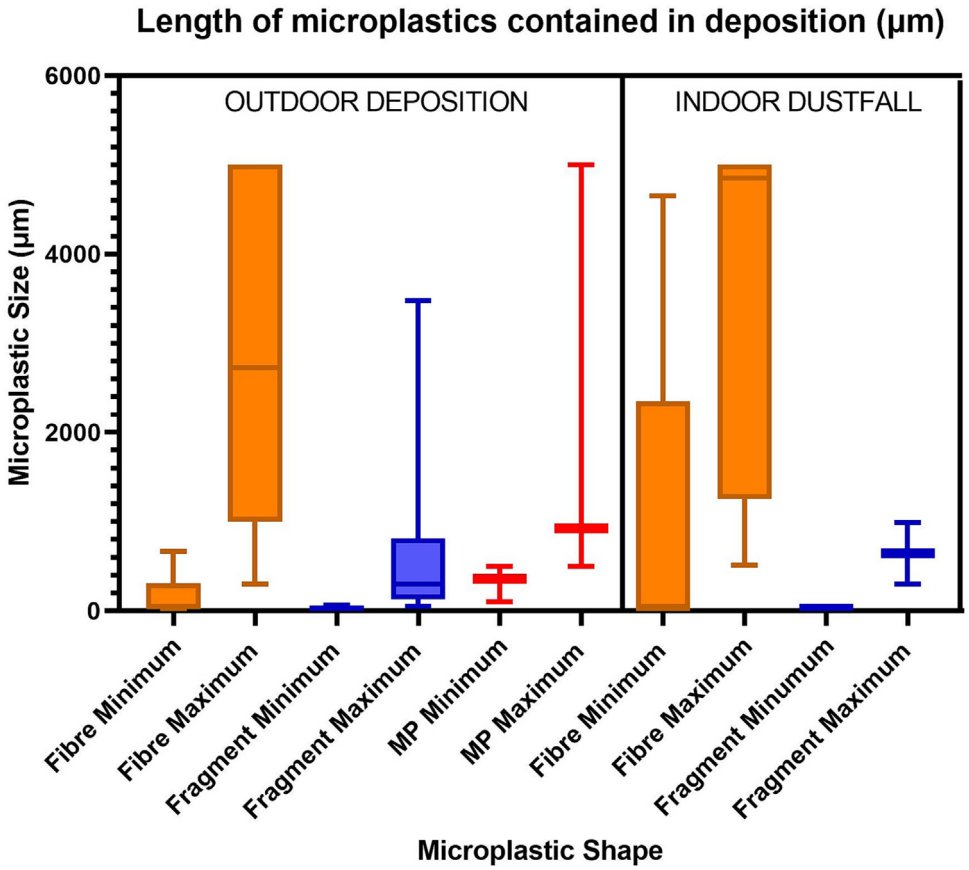


Fig. 14. Mean lengths of microplastics contained in indoor dustfall and outdoor deposition, differentiated by shape.
* Plotted according to mean reported lengths.
** Error bars represent minimum and maximum reported lengths.
*The minimum and maximum size of microplastics reported in various studies are plotted with the error bars representing the mean and interquartile ranges.
*** 'Indoor dustfall' refers to deposition samples collected within an indoor environment.
**** 'Outdoor deposition' refers to deposition samples collected from an outdoor environment.

7.2. Dust sampling methods and geographical locations

Dust can be separated into road dust (collected from the surface of the road and often includes tyre wear), outdoor dust, snow/ice and indoor dust. The characteristic specific to dust is that content is collected over an unknown duration of time and often contains settled particulates.

Road dust has been examined for microplastics in nine studies, with dust collected from Iran (Abbasi, 2019; Dehghani et al., 2017; Abbasi, 2017), Japan (Yukioka, 2020; Kitahara and Nakata, 2020), Nepal, Vietnam (Yukioka, 2020), Australia (Roychand and Pramanik, 2020; O'Brien et al., 2021; Su et al., 2020) and India (Patchaiyappan, 2021). Dust samples have been collected via sweeping (78 %), spoon collection (Su et al., 2020) or using a vacuum (22 % (Yukioka, 2020; Roychand and Pramanik, 2020)). Road dust samples have been analysed for microplastics using microscopy (33 %) (Abbasi, 2019; Dehghani et al., 2017; Abbasi, 2017), Raman (11 %) (Patchaiyappan, 2021), FTIR (67 %) (Yukioka, 2020; Kitahara and Nakata, 2020; Roychand and Pramanik, 2020; Su et al., 2020) and mass spectrometry (11 %) (O'Brien et al., 2021).

Outdoor dust has been examined in six publications, from locations including China (Liu, 2022), Turkey (Tunahan Kaya et al., 2018), Iran (Abbasi et al., 2022) and various desert environments (Wang et al., 2021; Abbasi, 2021). Samples have been collected via sweep (50 % (Liu et al., 2019c; Abbasi et al., 2022; Liu, 2022)) (Patchaiyappan, 2021) and metal shovels (Wang et al., 2021; Abbasi, 2021) and analysed using microscopy (14 % (Liu, 2022)), FTIR (43 %) (Wang et al., 2021), Raman (29 %) (Abbasi et al., 2022; Patchaiyappan, 2021; Abbasi, 2021), Laser Direct Infrared Imaging (LDIR) (14 % (Liu, 2022) and mass spectrometry (14 %) (Liu et al., 2019c).

Snow and ice samples have been analysed in thirteen studies, from Italy (Ambrosini et al., 2019; Pastorino et al., 2021; Parolini, et al., 1660), the Arctic, Alps (Bergmann, 2019), Mount Everest (Napper et al., 2020), Ecuador (Cabrera et al., 2020; Cabrera et al., 2022), Iceland (Stefánsson et al., 2021), Tibet (Zhang et al., 2021; Wang et al., 2022), Austria (Materić et al., 2020; Materić et al., 2021) and Antarctica (Aves et al., 2022). Samples have been collected from snow and ice using a range of methods including direct scooping (Zhang et al., 2021; Aves et al., 2022), spooning (Bergmann, 2019; Napper et al., 2020; Wang et al., 2022), ice cores (Stefánsson et al., 2021) and bare hands (Pastorino et al., 2021). Samples were then melted and analysed using FTIR (64 %) (Zhang et al., 2021; Pastorino et al., 2021; Parolini, et al., 1660; Napper et al., 2020; Cabrera et al., 2020; Cabrera et al., 2022; Aves et al., 2022), Raman (21 %) (Zhang et al., 2021; Stefánsson et al., 2021; Wang et al., 2022) and mass spectrometry (14 %) (Materić et al., 2020; Materić et al., 2021) instrumentation.

Concentrations of microplastics in indoor dust have been reported in nine publications, covering France (Dris et al., 2017; Gasperi, 2015), China (Liu et al., 2019c; Zhang et al., 2020b; Wang et al., 2017a; Chen et al., 2022b; Zhu et al., 2022), Iran (Nematollahi et al., 2022; Abbasi, 2022), Vietnam, South Korea, Saudi Arabia, Romania, Pakistan, Kuwait, Japan, India, Greece and Columbia (Zhang et al., 2020b). Various methods have been used to collect samples of indoor dust, including the detachment of an air conditioning filter (Chen et al., 2022c), sweeping (Liu et al., 2019c;

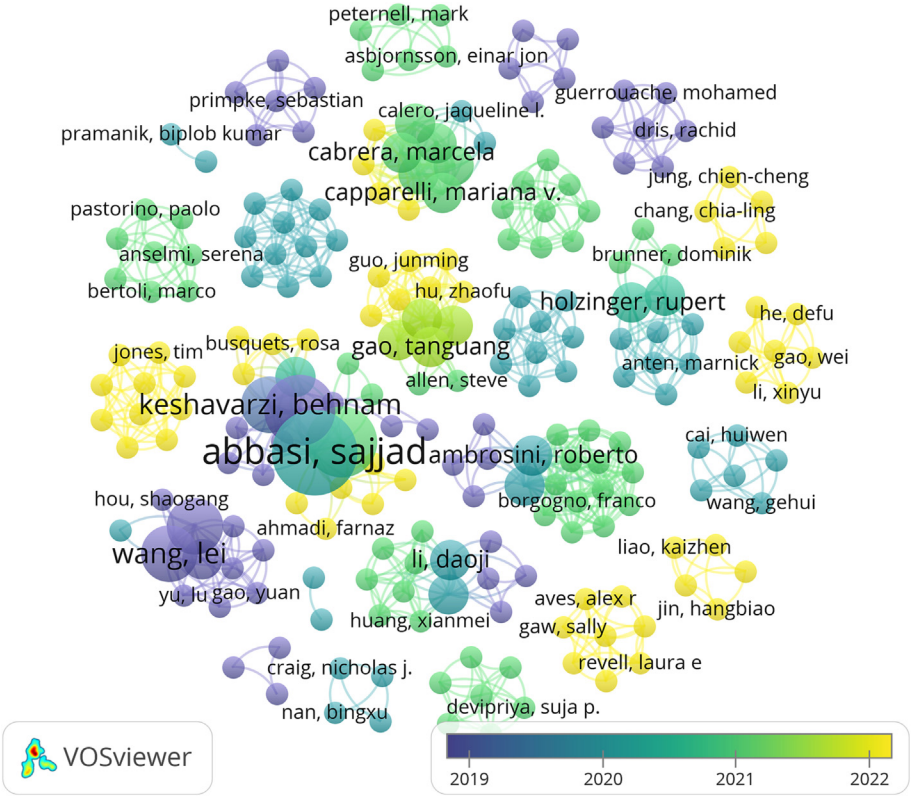


Fig. 15. Bibliometric Analysis of authorship prevalence and networks for all 224 authors included within this literature review and Supplementary Information who had publications covering the quantification of plastics in road dust, outdoor dust, snow and indoor dust.
* The colour and year scale represent the year of publication.
** The differing sizes of the circular markings (dots) represent the varying strength of the actors in the field.
*** Included publications were indiscriminately collated based on the topic and did not undergo screening for quality.
**** Industry reports without an associated RIS file were not included in the bibliometric analysis.
***** Two publications were omitted due to lack of associated RIS/Endnote file information capable of importation into the software which inhibited inclusion.

Zhang et al., 2020b; Wang et al., 2017a; Nematollahi et al., 2022; Abbasi, 2022) (including with nylon brush (Zhang et al., 2020b)) and vacuum collection (Dris et al., 2017; Zhang et al., 2020b; Gasperi, 2015; Zhu et al., 2022). Indoor dust has been collected using personal vacuum cleaners (Dris et al., 2017), which have differing components and suction capacities, and in one instance sorbed the chemicals of interest and affected recoveries (Zhang et al., 2020b). Dust samples have typically been processed using density separation and/or digestion techniques, followed by analysis with microscopy (11 %), FTIR (44 %), Raman (11 %) and mass spectrometry (33 %).

7.3. Road dust

7.3.1. Road dust – reported concentrations (microplastics/g, microplastics/m^2)

Road dust has been reported as the number of microplastics/g of dust (Abbasi, 2019; Yukioka, 2020; Kitahara and Nakata, 2020; Dehghani et al., 2017; Abbasi, 2017; Su et al., 2020; Patchaiyappan, 2021), number of microplastics/m^2 of road surface (Yukioka, 2020) and the mass of microplastics/m^2 (mg/m^2) (O'Brien et al., 2021). Normalised to per gram of dust collected, mean concentrations of microplastics (excluding rubber) have been reported to be 0.1 ± 0.09 microplastics/g (Kitahara and Nakata, 2020), 0.2 microplastics/g (Su et al., 2020), 3 ± 3 microplastics/g (Yukioka, 2020), 4 ± 4 microplastics/g (Yukioka, 2020), 9 microplastics/g (Dehghani et al., 2017), 60 microplastics/g (Abbasi, 2019) and 74 ± 43 microplastics/g (Abbasi, 2017). The mass based quantification of seven microplastics in road dust was a mean of 1.9 ± 2.1 mg/g, with a range of 0.5 to 6 mg/g, depending on location and traffic densities (O'Brien et al., 2021) (SI: Table S16, Fig. S11).

Based on the surface area of the road, microplastic concentrations (≥100 μm, excluding fibres) have been reported to be 2 ± 2 microplastics/m^2 (Japan), 13 ± 11 microplastics/m^2 (Nepal) and 20 ± 14 microplastics/m^2 (Vietnam), varying between countries (Yukioka, 2020). Normalised to the number of microplastics/m^2, reported concentrations have been ~294 microplastics/m^2 (>50 μm) (Iran) (Dehghani et al., 2017) and ~477 microplastics/m^2 (Iran) (Abbasi, 2017) (SI: Table S16).

7.3.2. Road dust – shape

In road dust, fibre shaped microplastics have accounted for 7 % (Yukioka, 2020), 8 % (Yukioka, 2020; Patchaiyappan, 2021), 10 % (Yukioka, 2020), 34 % (Dehghani et al., 2017), 76 % (Abbasi, 2019) and 46–100 % (Su et al., 2020) of all microplastics examined in samples. Interestingly, 35 % and 36 % of micro rubbers analysed were observed to be fibre shaped (Abbasi, 2019; Abbasi, 2017).

In road dust, fragment shaped microplastics have been reported to consist of 18 % (Abbasi, 2017), 0–46 % (Su et al., 2020), 66 % (Dehghani et al., 2017), 68 % (Yukioka, 2020), 72 % (Yukioka, 2020), 81 % (Yukioka, 2020) and 93 % (Patchaiyappan, 2021) of all microplastics examined. Other microplastic shapes including film (Su et al., 2020) (8 % (Yukioka, 2020), 13 % (Yukioka, 2020)), granule (4 %, 7 %, 10 % (Yukioka, 2020)) and pellet (Su et al., 2020) have also been observed in road rust (SI: Table S16, Figs. S17, S18).

7.3.3. Road dust – size

The size of microplastics reported within road dust has varied from a mode size of 100–250 μm (Iran) (Abbasi, 2019), <250 μm (Australia) (O'Brien et al., 2021), 250–500 μm (Iran) (Dehghani et al., 2017),

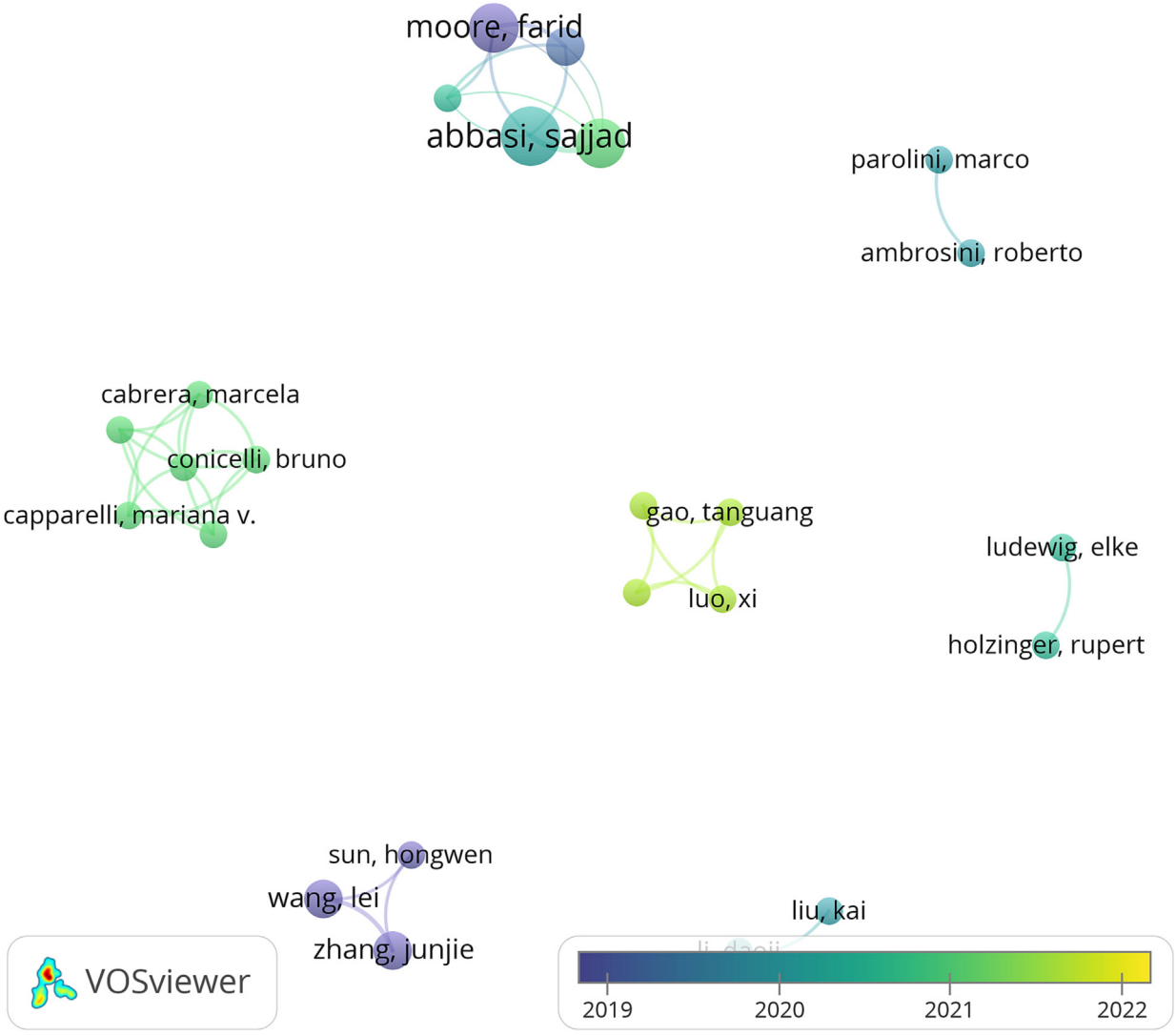


Fig. 16. Bibliometric analysis of the 24 authors included within this literature review and Supplementary Information who had published ≥ 2 publications quantifying microplastics in road dust, outdoor dust, snow and indoor dust.
* The colour and year scale represent the year of publication.
** The differing sizes of the circular markings (dots) represent the varying strength of the actors in the field.
*** Included publications were indisciminately collated based on the topic and did not undergo screening for quality.
**** Industry reports without an associated RIS file were not included in the bibliometric analysis.
***** Two publications were omitted due to lack of associated RIS/Endnote file information capable of importation into the software which inhibited inclusion.

400–500 μm (Vietnam, Nepal) (Yukioka, 2020), 700–800 μm (Japan) (Yukioka, 2020) and <1000 μm (Australia) (Su et al., 2020). Mean microplastic sizes have been reported comparatively higher, as 820 ± 240 μm (Vietnam) (Yukioka, 2020), 930 ± 290 μm (Nepal) (Yukioka, 2020), 1200 μm (Australia) (Su et al., 2020) and 1360 ± 490 μm (Japan) (Yukioka, 2020), suggesting that fewer plastics in the upper size limits (1220 μm, 1520 μm, 2190 μm (Yukioka, 2020) and 4700 μm (Su et al., 2020)) increased the mean size (SI: Table S16, Fig. 19).

7.3.4. Road dust – composition

Road dust has proven to be a heterogenous mix of microplastics, with 9 different plastic compositions identified within samples from India (Patchaiyappan, 2021), 13 plastic types within Japan, 15 in Nepal and 14 in Vietnam (Yukioka, 2020). The predominant plastics detected were polyvinyl chloride (12 % (Kitahara and Nakata, 2020), 24 % (Patchaiyappan, 2021), 29 % (O'Brien et al., 2021), polyethylene (3.2 % (Su et al., 2020), 11 % (Kitahara and Nakata, 2020), 21 % (O'Brien et al., 2021) 32 ± 15 %, 45 ± 19 %, 47 ± 22 % (Yukioka, 2020)), polypropylene (7 ± 13 %, 8 ± 9 % (Yukioka, 2020), 12 %, 18 ± 33 % (Yukioka, 2020), polystyrene (<1 % (Kitahara and Nakata, 2020), 1 % (O'Brien et al., 2021), 2 % (Su et al., 2020), 1 ± 3 %, 1 ± 2 %, 10 ± 29 % (Yukioka, 2020)), polyethylene terephthalate (<1 % (Kitahara and Nakata, 2020), 2 ± 6 %, 4 ± 7 %, 7 ± 8 % (Yukioka, 2020), 15 % (Su et al., 2020), 29 % (O'Brien et al., 2021)), polyacrylate (5 ± 7 %, 11 ± 11 %, 23 ± 20 % (Yukioka, 2020)), polyurethane (1 ± 2 %, 5 ± 8 % (Yukioka, 2020)), polymethyl methacrylate (2 % (Su et al., 2020), 8 % (O'Brien et al., 2021), 9 % (Kitahara and Nakata, 2020)), polycarbonate (<1 % (Su et al., 2020)), polyacrylate, epoxy resin (EPC) (Yukioka, 2020), ethylene/propylene/diene rubber (EPDM) (Yukioka, 2020), polyvinyl stearate (PVS) (Yukioka, 2020), high density polyethylene, p-tetrafluoroethylene, superflex (Patchaiyappan, 2021), polyester (11 %), polydiallyl phthalate (13 %) and poly(alpha-chloroacrolein-*co*-styrene) (4 %) (Kitahara and Nakata, 2020). Styrene Butadiene Rubber (SBR) has been reported to constitute 3 ± 4 %, 4 ± 5 % and 4 ± 13 % of microplastics quantified (Yukioka, 2020) and rubber as <1 % (Su et al., 2020).

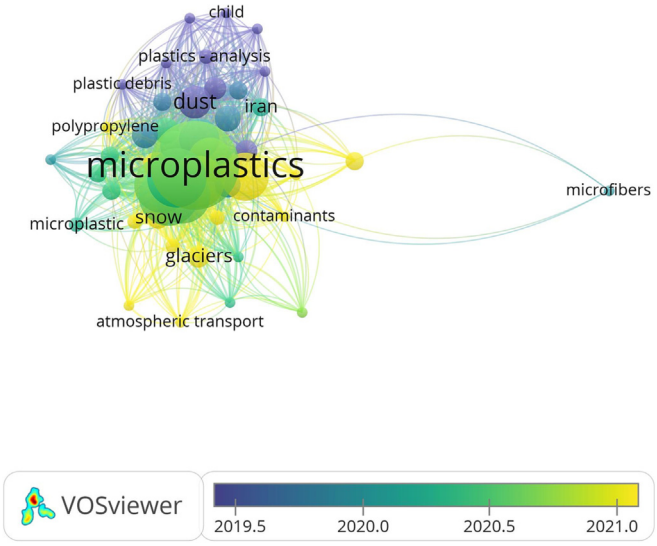


Fig. 17. Bibliometric analysis of the 48 most utilised key terms used across publications included within this literature review and supplementary information which quantified microplastics in road dust, outdoor dust, snow/ice and indoor dust where the term co-occurred across >3 publications.
* the colour and year scale represent the year of publication.
** the differing sizes of the circular markings (dots) represents the varying frequency of the key term in the field.
*** Included publications were indisciminately collated based on the topic and did not undergo screening for quality.
**** Industry reports without an associated RIS file were not included in the bibliometric analysis.
***** Two publications were omitted due to lack of associated RIS/endnote file information capable of importation into the software which inhibited inclusion

7.4. Outdoor dust

Outdoor dust has been collected from windowsills in China (Liu et al., 2019c), car windshields in Iran (Abbasi et al., 2022) and analysed from Turkey (Tunahan Kaya et al., 2018), China (Liu, 2022) and the Badain Jaran, Iran-Lut and Kavir deserts, reported as the number of microplastics per nominated gram weight. The mass concentration of polyethylene terephthalate and polycarbonate has been reported in one study from China (Liu et al., 2019c).

7.4.1. Outdoor dust - reported concentrations (microplastics/g, mg/g)

Based on particle count, concentrations of microplastics in urban outdoor dust have been reported to be 18, 22 and 29 microplastics/g (Tunahan Kaya et al., 2018), 24, 171 and 225 microplastics/g (Liu, 2022). Within desert environments, lower mean concentrations have been reported as 0.006 ± 0.0154 microplastics/g (Wang et al., 2021), 0.2 microplastics/g (Abbasi, 2021) and 0.3 microplastics/g (Abbasi et al., 2022). Mass based quantification of polyethylene terephthalate and polycarbonate exclusively from outdoor dust collected in China featured a mean concentration of 1.65 mg/g (median 2.8 mg/g) and a range of 0.21 to 9.08 mg/g (Liu et al., 2019c), respectively (SI: Table S16).

7.4.2. Outdoor dust – shape

Microplastics in outdoor dust have been reported to be predominantly fibre shaped (74 % (Liu et al., 2019c), 92 % (Abbasi et al., 2022), 93 % (Abbasi, 2021)), approximately evenly spread between microplastic fibres and fragments (36 % 44 %, 55 % (Tunahan Kaya et al., 2018), 40 % (Wang et al., 2021)) and as mostly fragment shaped microplastics (45 %, 56 %, 64 % (Tunahan Kaya et al., 2018), 60 % (Wang et al., 2021), 71 %) and pellets (20 %) (Liu, 2022). Film shaped microplastics were identified in one study, being 7 % of all microplastics examined (Abbasi et al., 2022) (SI: Table S16, Figs. S17, S18).

7.4.3. Outdoor dust – size

The mean size of microplastics in outdoor dust has been reported to be 58, 59 and 67 μm (Liu, 2022) and 446 ± 709 μm (Wang et al., 2021), and mode size classifications reported as 50–200 μm (Wang et al., 2021), <100 μm (Abbasi, 2021) and 100–1000 μm (Abbasi, 2021) (SI: Table S16).

7.4.4. Outdoor size – composition

Polyethylene terephthalate has been detected in every publication examining outdoor dust (Tunahan Kaya et al., 2018; Liu et al., 2019c; Liu, 2022; Abbasi, 2021). Where the percentage is reported, polyethylene terephthalate has constituted 5 % (Liu, 2022), 21 % (Abbasi et al., 2022) and 25 % (Wang et al., 2021) of all microplastics in outdoor dust. Within outdoor dust, other polymer compositions of identified microplastics include polyamide (Tunahan Kaya et al., 2018; Abbasi, 2021) (9 % (Wang et al., 2021), 23 % (Liu, 2022)), polyurethane (Tunahan Kaya et al., 2018) (4 % (Liu, 2022)), polypropylene (28 % (Abbasi, 2021), 45 % (Liu, 2022)), polystyrene (5 % (Abbasi et al., 2022), 6 % (Liu, 2022)), polyethylene (7 % (Liu, 2022)), polyvinyl chloride (6 % (Liu, 2022)), polycarbonate (0.4 % (Liu, 2022)), nylon (46 % (Abbasi et al., 2022)), epoxy resin (28 %) and phenoxy resin (25 %) (Wang et al., 2021).

7.5. Snow

7.5.1. Snow – reported concentrations (microplastics/L)

Within snow and snowfall, mean microplastic concentrations have varied substantially, being 0.1 ± 0.2 microplastics/L (Italy) (Pastorino et al., 2021), 1 microplastic/L (Iceland) (Stefánsson et al., 2021), 2 ± 1 microplastic/L (Italy) (Parolini, et al., Microplastic Contamination in Snow from Western Italian Alps. LID - 10.3390/ijerph18020768 [doi] LID - 768, 1660), 10 ± 1 microplastics/l (Tibet) (Wang et al., 2022), 29 ± 5 microplastics/L (Antarctica) (Aves et al., 2022), 30 ± 11 microplastics/L (Mount Everest) (Napper et al., 2020), 74 ± 28 microplastics/L (Italy) (Ambrosini et al., 2019), 131 ± 24 microplastics/L (Ecuador) (Cabrera et al., 2022), 359 microplastics/L (Ecuador) (Cabrera et al., 2020), 1760 ± 1580 fragments/L plus 386 microplastic fibres/L (Arctic) and 24,600 ± 18,600 microplastic fragments plus 400 microplastic fibres/L in Europe (Bergmann, 2019).

7.5.2. Snow – shape

Little consistency exists between the proportion of microplastic fibres (11 % (Bergmann, 2019), 33 % (Stefánsson et al., 2021), 39 % (Parolini, et al., 1660), 50 % (Wang et al., 2022), 60 % (Aves et al., 2022), 66 % (Ambrosini et al., 2019), 70 % (Cabrera et al., 2022), 71 % (Cabrera et al., 2020), 95 % (Napper et al., 2020)) and fragments (5 % (Napper et al., 2020), 14 % (Cabrera et al., 2022), 17 % (Wang et al., 2022), 35 % (Ambrosini et al., 2019), 39 % (Aves et al., 2022), 61 % (Parolini, et al., 1660), 67 % (Stefánsson et al., 2021), 89 % (Bergmann, 2019)) in snow samples, as a proportion of all microplastics examined. Film shaped microplastics have also been identified in snow (Zhang et al., 2021) (1 % (Aves et al., 2022), 15 % (Cabrera et al., 2022), 33 % (Wang et al., 2022)), along with spheres in one study (1 %) (Cabrera et al., 2022) (SI: Figs. S17, S18).

7.5.3. Snow – size

Within snow samples, microplastic fibres have been reported with a mode size range of 100–300 μm (Zhang et al., 2021) and mean lengths of 606 μm (Aves et al., 2022), 1036 ± 829 μm (Napper et al., 2020) and 1224 ± 1803 μm (Bergmann, 2019). The upper size limit of microplastic fibres in snow has been reported to be 1910 μm (Parolini, et al., 1660), 2500 μm (Cabrera et al., 2022), 3000 μm (Stefánsson et al., 2021), 3210 μm (Aves et al., 2022), 3800 μm (Napper et al., 2020) to >5000 μm (Bergmann, 2019). Fragment shaped microplastics are typically smaller, having reported means of <50 μm (Zhang et al., 2021), 200 μm (Aves et al., 2022), 220 μm (Pastorino et al., 2021) and a mode size of

11–25 μm (61 %) (Bergmann, 2019). Where shape is unspecified, mean microplastic sizes have been reported between 339 ± 103 μm (Parolini, 1660), 1266 ± 1423 μm (Stefánsson et al., 2021) and a mode size of <200 μm (Wang et al., 2022) (SI: Table S16).

7.5.4. Snow – composition

The composition of microplastics in snow and snowfall is complex. Snow samples have included polyethylene (Ambrosini et al., 2019) (1 % (Materić et al., 2021), 5 % (Aves et al., 2022), 8 % (Bergmann, 2019), 14 % (Zhang et al., 2021), 22 % (Cabrera et al., 2022), 39 % (Parolini, et al., 1660)), polyamide (Ambrosini et al., 2019) (7 % (Aves et al., 2022), 11 % (Cabrera et al., 2020; Cabrera et al., 2022), 13 % (Bergmann, 2019), 17 % (Stefánsson et al., 2021), >25 % (Zhang et al., 2021)) polypropylene (Ambrosini et al., 2019; Aves et al., 2022) (1 % (Zhang et al., 2021), 2 % (Bergmann, 2019), 5 % (Napper et al., 2020), 65 % (Materić et al., 2020; Materić et al., 2021)), polyurethane (5 % (Parolini, et al., 1660), 11 % (Bergmann, 2019), 50 % (Stefánsson et al., 2021), 56 % (Cabrera et al., 2022)), polycarbonate (<1 % (Bergmann, 2019), 4 % (Zhang et al., 2021)), polystyrene (<1 % (Bergmann, 2019; Materić et al., 2021), 5 % (Zhang et al., 2021)), polyester (PES) (Ambrosini et al., 2019) (<1 % (Bergmann, 2019), 11 % (Parolini, et al., 1660), 22 % (Cabrera et al., 2020), 56 % (Napper et al., 2020)), polyvinyl chloride (Materić et al., 2020) (<1 % (found exclusively in arctic) (Bergmann, 2019), 1 % (Zhang et al., 2021), 10 % (Aves et al., 2022), 17 % (Stefánsson et al., 2021), 40 % (Wang et al., 2022)), polyethylene terephthalate (Materić et al., 2020; Pastorino et al., 2021) (8 % (Zhang et al., 2021), 17 % (Parolini, et al., 1660), 33 % (Materić et al., 2021), 41 % (Aves et al., 2022)), high density polyethylene (17 % (Parolini, et al., 1660)), low density polyethylene (5 % (Parolini, et al., 1660)), polymethyl methacrylate (3 % (Zhang et al., 2021), 10 % (Aves et al., 2022)), acrylic (31 % (Napper et al., 2020)), nylon (9 % (Napper et al., 2020)), poly (2,6 dimethyl 1 Poly (23 % (Wang et al., 2022)), hydroxypropyl cellulose (13 % (Wang et al., 2022)), alkyd (5 % (Aves et al., 2022)), cellulose nitrate (5 % (Aves et al., 2022)), MC, PB and PVA (3 % each, respectively (Wang et al., 2022)), polypropylene carbonate (Materić et al., 2020) and polytetrafluoroethylene (PTFE), polyvinylidene, silicone and polymethyl anhydride (Aves et al., 2022). Rubber was omitted from the majority of studies, however was 10 % (Zhang et al., 2021), 17 % (Stefánsson et al., 2021) and 55 % of all microplastics when analysed (Bergmann, 2019).

7.6. Indoor dust

7.6.1. Indoor dust – reported concentrations (microplastics/g, mg/g)

Indoor dust could be sink for microplastics (Nematollahi et al., 2022). Based on particle count, mean concentrations of microplastics in indoor dust have been reported to be 195 microplastics/g (range: 10–635 microplastics/g) (Nematollahi et al., 2022), 3771 microplastics/g (range: 81–55,830 microplastics/g) (Abbasi, 2022) and 62–3861 microplastics/g (Zhu et al., 2022). Microplastic fibres have been reported in concentrations varying from 19,000–67,000 microplastic fibres/g (Dris et al., 2017). Based on surface area, concentrations of microplastics have been reported to be 1.47×10^6 to 21.4×10^6 microplastics/m^2 (Chen et al., 2022c) (SI: Table S16).

Mass based quantification of polyethylene terephthalate and polycarbonate has been undertaken in two studies using LC-MS-MS, examining indoor dust from twelve countries (Zhang et al., 2020b; Wang et al., 2017a). Mean concentrations of polyethylene terephthalate have been reported as 0.43 mg/g in one student dormitory (Wang et al., 2017a), 23 mg/g from 39 cities in China (Liu et al., 2019c) and 2 mg/g (India), 3.9 mg/g (Columbia), 4.4 mg/g (Pakistan), 9.2 mg/g (China), 10 mg/g (Kuwait), 11 mg/g (Vietnam), 14 mg/g (USA and Romania), 15 mg/g (Greece), 20 mg/g (Japan), 20 mg/g (Saudi Arabia) and 27 mg/g (South Korea) (Zhang et al., 2020b). Mean concentrations of polycarbonate were lower, reported as 240 μg/g in the one student dormitory (Wang et al., 2017a), 1.8 μg/g (China) (Liu et al., 2019c) and 6 μg/g (Pakistan), 7 μg/g (Kuwait), 11 μg/g (Columbia), 15 μg/g (Romania), 18 μg/g (China), 20 μg/g (India), 34 μg/g (Greece), 54 μg/g (South Korea), 58 μg/g (Saudi Arabia), 63 μg/g (Japan), 87 μg/g (USA), 120 μg/g (Vietnam) (Zhang et al., 2020b). The mass concentration of other polymers in indoor dust has not been reported to date (Fig. 18).

7.6.2. Indoor dust – shape

The shape of microplastics in indoor dust often remained unreported in literature (Zhang et al., 2020b), or studies reported exclusively fibre shaped microplastics (Dris et al., 2017; Zhang et al., 2020b). In the four studies to report different shapes, fibres remained the dominant shape of microplastic (Chen et al., 2022c), constituting 85 % (Iran) (Abbasi, 2022), 80–92 % (China) (Zhu et al., 2022), 99.7 % (Iran) (Nematollahi et al., 2022) of the microplastics identified. The proportion of fragment shaped microplastics varied from 0.2 % (Nematollahi et al., 2022), 5.3–11 % (Zhu et al., 2022) and 15 % (Abbasi, 2022) of total microplastics examined. Film shaped microplastics have also been observed in indoor dust, being 0.1 % (Nematollahi et al., 2022) and 0–15 % (Zhu et al., 2022) of total microplastics examined (SI: Table S16, Figs. S17, S18).

7.6.3. Indoor dust – size

The size range of microplastics in indoor dust has been reported to be >50–2500 μm (Dris et al., 2017), with mode size ranges between 200 and 1000 μm (45–63 %) (Zhu et al., 2022) and 500–1000 μm (32 % (Nematollahi et al., 2022), 25 % (Abbasi, 2022)). Mean microplastic sizes have been reported as 312 ± 430 μm (living room), 670 ± 918 μm (dormitory) and 526 ± 628 μm (office) (Chen et al., 2022c) (SI: Table S16).

7.6.4. Indoor dust - composition

Suspected microplastics (which have been isolated based on visual detection) have been confirmed to be microplastics for 33 % (France) (Dris et al., 2017), 39 % (Australia) (Soltani et al., 2021) and 48 % of all fibres and 40 % of all fragments (China) (Liu et al., 2019c). Based on the maximum concentration of 120,000 μg/g for polyethylene terephthalate in indoor dust collected from China, polyethylene terephthalate constituted 12 % of the total dust weight (Liu et al., 2019c).

Microplastics in dust are heterogenous regarding polymetric composition (Nematollahi et al., 2022). In addition to the polyethylene terephthalate and polycarbonate mass quantified (Zhang et al., 2020b), the following microplastics and their reported proportions have been detected within indoor dust: polyethylene terephthalate (45 ± 11 % (Chen et al., 2022c), 50 % (Nematollahi et al., 2022)), polyethylene (35 % (Nematollahi et al., 2022)), polyethylene/polypropylene, polyethylene/polypropylene/polyester (60–72 % (Zhu et al., 2022)), polypropylene (Dris et al., 2017; Chen et al., 2022c; Zhu et al., 2022) (33 % (Abbasi, 2022)), polystyrene (10 % (Nematollahi et al., 2022), 13 % (Abbasi, 2022)), nylon (5 % (Nematollahi et al., 2022)), rayon (28 ± 6 % (Chen et al., 2022c)), cellophane (20 ± 9 % (Chen et al., 2022c)), polyurethane, polyacrylic nitrile and plyphthalamide (Chen et al., 2022c).

Within differing indoor environments, polyethylene and polypropylene were identified in classroom environments and polyester within hotels, dormitories and residential environments (Zhu et al., 2022).

Separated by microplastic shape, fragments have been polymetrically identified to be polyester, polyethylene, polyurethane and polyethylene/polypropylene (Liu et al., 2019c). Microplastic fibres have been identified to be polypropylene (Dris et al., 2017), polyester, polyurethane, polyamide, polyethylene, and polyacrylic nitrile (Liu et al., 2019c). (SI: Table S16). Interestingly, the concentrations of microplastics in two rural homes correlated to GDP per capita in the countries investigated (polyethylene terephthalate: $r = 0.77$, polycarbonate: $r = 0.60$) (Zhang et al., 2020b).

7.7. Indoor v outdoor comparison

Between outdoor and indoor dust, no significant difference was identified between microplastic concentrations per gram ($p > 0.05$) across all studies (Fig. 18). Regarding size, a significant difference was observed between the minimum microplastic sizes reported across outdoor and indoor dust ($p < 0.05$), with outdoor dust reporting lower minimum microplastic

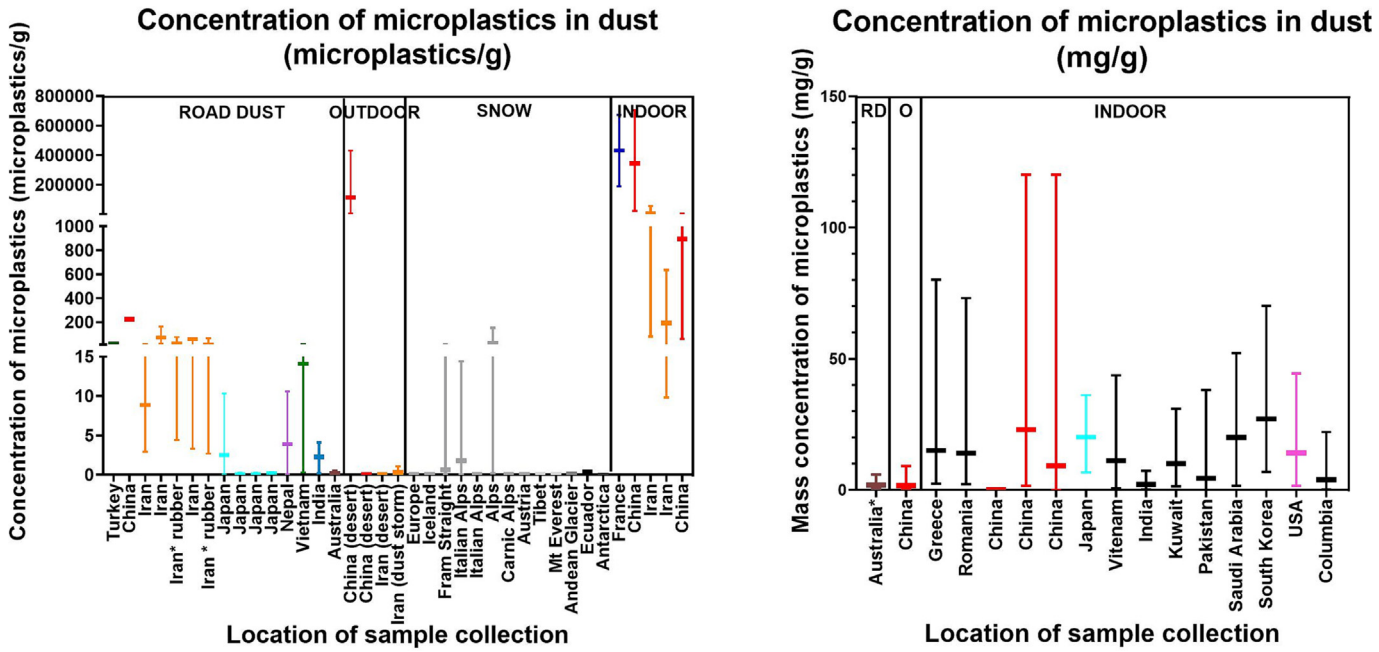


Fig. 18. Concentrations of microplastics in dust (microplastic particles/g and mg/g) in road dust, outdoor dust, snow and indoor dust.
* Single data points are the reported mean concentrations, absent reported minimum and maximum concentrations.
** Error bars represent minimum and maximum reported values.
*** 'Indoor' refers to dust samples collected from within an indoor environment.
**** 'RD' and 'road dust' refer to samples collected from the surface of a road.
***** 'O' and 'outdoor' refers to dust collected from an outdoor environment such as a windowsill.
****** 'Snow' refers to the collection of snow for microplastic analysis.
******* PET and PC concentrations only are provided for the mass quantification in all studies but Australia*, meaning that reported concentrations are exclusive of other microplastic polymer compositions.

sizes. No significant difference was observed between the maximum sized microplastics in indoor and outdoor dust ($p > 0.05$) (Fig. 19).

8. Biomonitors

Atmospheric microplastics have recently been sampled using a variety of biomonitors, including leaf surfaces (Liu et al., 2020a), moss (Roblin and Aherne, 2020) and lichen (Loppi et al., 2021) for deposition and honeybees (Edo et al., 2021) and spider webs (Goßmann et al., 2022) for ambient air.

8.1. Biomonitors - leaf surfaces

Liu et al. (2020a) examined leaves in China using microscopy and μ-FTIR to determine that 28 % of particles adhered to the leaf surface were microplastics shaped as fibres (59 %), fragments (33 %) and beads (8 %). Microplastics were detected at concentrations varying from 0.07 to 0.19 microplastics/cm^2 (mean: 0.13 ± 2.39 microplastics/cm^2), between 15 and 3989 μm (mean 471 ± 681 μm) in size and of polyethylene terephthalate, polyethylene, polypropylene, polyamide and cellophane composition.

8.2. Biomonitors – moss

Roblin and Aherne (2020) examined 5 g of moss in Ireland (estimated to have grown over a 2–3 year period) using microscopy and Raman spectroscopy to determine that 13–27 % of all particles identified were suspected microplastics, ranging in size from 30 to 5000 μm (mean: 760–1020 μm, mode: 40 % <400 μm) at concentrations of 24 microplastic fibres/g (range 15–30 microplastic fibres/g). From the concentration contained within the moss, deposition within remote areas of Ireland (in river and lake catchments) was calculated to range from 84 to 161 microplastics/m^2/day (mean 130 microplastics/m^2/day). Compared to another deposition study from Ireland by the same author, the reported mean deposition concentration of 12 microplastic fibres/m^2/day was lower in the predominantly wet deposition samples collected from remote and rural locations of Ireland used specifically to monitor air quality. The mean microplastic size of 880 μm and mode size range of 200–400 and 400–600 μm (Roblin et al., 2020) were consistent with the moss results.

Ombrotrophic peat has also been examined for temporal trends in atmospheric microplastic deposition (Allen et al., 2021). Taking cored peat samples, microplastic concentrations were highest towards the surface (178 ± 79 microplastics/m^2/day) with only nominal concentrations detected in peat core samples representing the years between 1940 and 1960 (<1 microplastics/m^2/day). The proportion of microplastic fibres increased from ≤20 % in 1960–1980 samples to >30 % of total microplastics within surface samples.

8.3. Biomonitors – lichen

Lichen surrounding landfill ($n = 3$) in Italy was analysed using the hot needle test, followed by μ-Raman to determine that approximately 40 % of identified anthropogenic material were microplastics of polyester polyethylene terephthalate (80 %) or polyethylene (26 %) composition. Mean microplastic concentrations were 7–79 microplastics/g, which ranged in mean size between 45-66 μm (Loppi et al., 2021). Other studies from Italy are not comparable, being from remote alpine regions (Pastorino et al., 2021; Parolini, et al., 1660) and having detection limits (~100 μm (Ambrosini et al., 2019)) above the size reported on lichen.

8.4. Biomonitors – honeybees

Using bees as active ambient air samplers, Edo et al. (2021) isolated microplastics on honeybees from apiaries (n = 19) in Denmark. Microplastics (56 particles) were identified to be fibres (38 %, diameter 64 ± 39 μm) and fragments (52 %, 234 ± 156 μm). Fibres were predominantly polyester

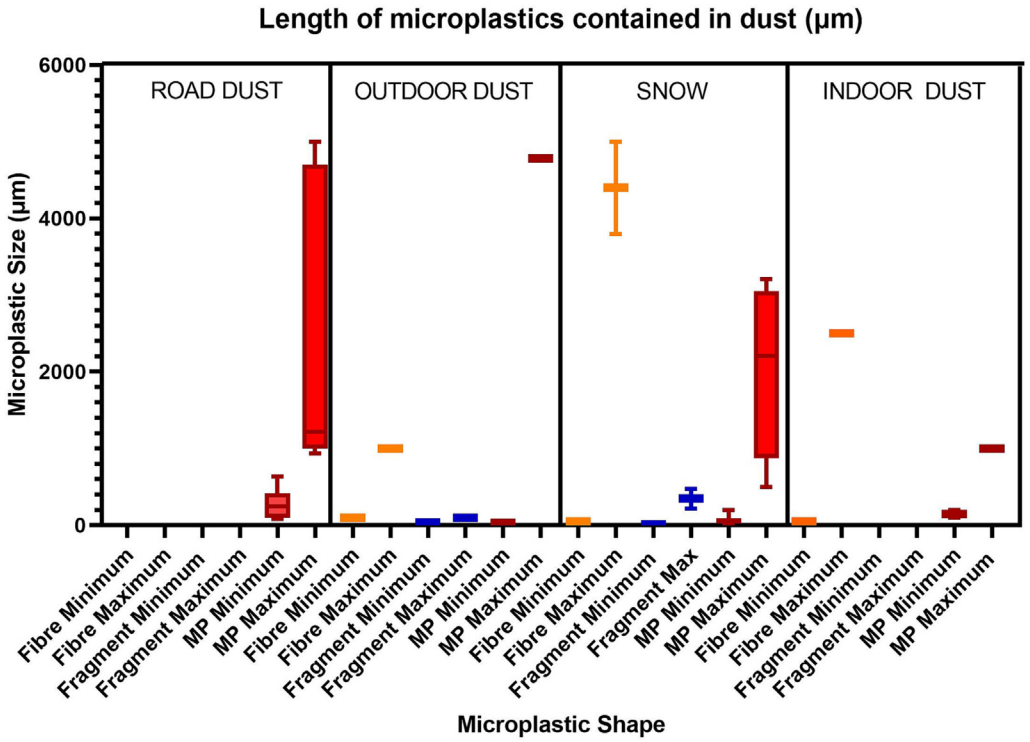


Fig. 19. Mean lengths of microplastics reported in road dust, outdoor dust, snow and indoor dust, differentiated by shape.
* Plotted according to mean reported lengths.
** Error bars represent minimum and maximum reported lengths.
*** 'Indoor dust' refers to dust samples collected from within an indoor environment.
**** 'Road dust' refer to samples collected from the surface of a road.
***** 'Outdoor dust' refers to dust collected from an outdoor environment such as a windowsill.
****** 'Snow' refers to the collection of snow for microplastic analysis.

(80 %). Other polymer compositions including polyethylene, polyvinyl chloride, polyurethane, epoxy resin, polyvinyl acetate, polyacrylic nitrile, polyoxymethylene, polypropylene, polystyrene, polytetrafluoroethylene and polyamide were also identified. Similarly, Vianello et al. (2019) polymetrically confirmed 81 % of all examined microplastics to be polyester and also observed polyethylene and polypropylene within indoor deposition samples collected in Denmark. Microplastic fragments on bees (234 ± 156 μm) were larger than the fragments identified in indoor air (68 μm) and fibres on bees were a diameter of 64 ± 39 μm, whilst only the length of fibres (177 μm) was reported in indoor air.

8.5. Biomonitors – spiderwebs

Spiderwebs have been collected in Germany with microplastic concentrations mass quantified using Pyr-GC–MS (Goßmann et al., 2022). Within spiderwebs, concentrations of microplastics were 11–108 μg/mg (mean: 44 μg/mg), and the highest concentration equated to 11 % of the total spiderweb mass (Goßmann et al., 2022). Microplastic composition included polyethylene terephthalate (36 %), polyvinyl chloride (12 %), tyre wear (47 %), polyethylene (3 %), polypropylene (<1 %), polystyrene (<1 %), polymethyl methacrylate, polycarbonate and the identification of polyamide6 and polyurethane (Goßmann et al., 2022).

9. Human exposure pathways

9.1. Human exposure pathways – ingestion

Ingestion exposure for microplastics contained within outdoor dust has been calculated (based on mean concentrations of 60 microplastics/g) to be 5–15 microplastics per day for construction workers and 2–7 microplastics per day for children in Iran, both considered to be vulnerable groups (Abbasi, 2019). Calculated from the Environmental Protection Agency (EPA) guidelines which indicate that involuntary ingestion of dust from hand to mouth activities for an adult is 10 mg/day for outdoor settled dust (Assessment, 2017), and the mean concentration of 25 microplastics/g across all studies, an average adult could be exposed to 0.25 microplastics/day via ingestion.

Based on the same EPA guidelines, an average adult ingests 20 mg/day of indoor settled dust (Assessment, 2017). Calculated from the mean reported concentration of microplastics in indoor dust across all studies (1249 microplastics/g), ingestion of microplastics contained within indoor dust could approximate 25 microplastics/day. It is noteworthy that indoor dust concentrations vary across publications, where a more acute exposure could be in the vicinity of 207 microplastics/day based on the mean of the highest reported concentrations. From the mass based concentrations of polyethylene terephthalate and polycarbonate in indoor dust (Zhang et al., 2020b), adults could ingest 0.76 to 2400 mg/day (median 118 mg/day) of polyethylene terephthalate and <1 to 34 mg/day of polycarbonate (median 0.18 mg/day).

Ingestion from deposition (not dust) has been estimated to be 1 million plastics per annum, consisting of fragments <100 μm (Fang et al., 2022).

9.2. Human exposure pathways – inhalation

Inhalation exposure for a general population has been calculated in Shanghai with reported estimations of 21 microplastic particles inhaled per day (range of 0 to 4.18 microplastics/m^3) (Liu et al., 2019a). Calculated from the average reported mean concentration of suspended microplastics in outdoor air across all publications (mean 339 microplastics/m^3) and the Environmental Protection Agency Guidelines of an adult inhaling

between 15.7 m^3 and 16 m^3/day (Agency, 2011), inhalation exposure to suspended microplastics is calculated to be approximately 5300–5400 microplastics/day. However it must be noted that variability exists regarding reported ambient air concentrations, where concentrations have ranged from being below detection and at low concentrations <10 microplastics, to means of <2000 microplastics/m^3 (Levermore et al., 2020; Tunahan Kaya et al., 2018) and maximum reported concentrations >8800 microplastics/m^3 (Liao et al., 2021).

Within an indoor environment, exposure is calculated to be an average of 1470–1500 microplastics per day. Intuitively, if the source of microplastics is from indoors, one would assume that indoor and outdoor air exchange would provide a dilution effect. The exposure calculations for both indoor and outdoor environments is very uncertain, with comparatively few studies undertaken for ambient air quality indoors.

Inhalation exposure can also be calculated from deposition, where the concentration of microplastics per m^2 equates to 1.2 m^3 inhaled (Soltani et al., 2021). Applying this equation to outdoor deposition concentrations, based on a mean concentration of 225 microplastics/m^2/day across all studies, humans could be exposed to 2950–5570 microplastics/day. Based on mean concentrations of microplastics in indoor dust, inhalation exposure is calculated to range between 2.2×10^6 and 5.7×10^6 microplastics per day. Undeniably, exposure to atmospheric microplastics via inhalation is the dominant avenue and requires immediate understanding and clarification, followed by ingestion and dermal exposure and uptake.

10. Future research directions

10.1. Recommendations for the microplastics field as a whole

Comparability between studies is inherently difficult, however ambiguity within the field could be reduced via establishing uniformity between terminology, reporting units, clear segregation of results, consistent size delineations, consistent polymer classifications, reported contamination mitigation measures and reported contamination levels. Bibliographic analysis has indicated that the majority of research is undertaken in isolation. Therefore, authorship and institutional collaboration could facilitate consistency and adopted industry standards could be established in the field.

10.1.1. Consistent reporting and terminology

Collaboration could bring harmonisation and consistency regarding the use of size, shape and polymetric classifications. Consistency among reported microplastic size delineations between studies, particularly as research progresses into the smaller size fractions <50 μm, provides elucidation for fate, behaviour and health implications. The terminology surrounding microplastic shape classifications such as 'granule'/'fragment', 'bead'/'sphere' should be clarified and uniformly adopted. Transparency surrounding the term 'fibre' is also required, where 'plastic fibres' should be reported with a clear segregation from 'anthropogenic fibres' (having undergone processing/dying) and/or 'natural fibres' rather than grouped together. The polymer composition and definition of 'polyester' requires determination regarding whether or not 'polyester' as a term is removed for uncertainty, includes polyethylene terephthalate exclusively or consists of polyethylene terephthalate and other chemical compositions such as poly butylene terephthalate and poly tri methylene (Ding et al., 2021). Similar clarification around the polymetric composition of 'nylon' and 'acrylic' is also required. Furthermore, the percentage of various chemical compositions within studies should not be grouped (for example 10 % polystyrene/polyethylene terephthalate), but rather separated into specific polymetric compositions to provide clarification for source attribution and proportionality.

10.1.2. Use multiple analytical techniques

Reporting particle count and mass based quantification in isolation inhibits comparison (Chen et al., 2020) between studies, however both deliver useful information for differing purposes. Utilisation of multiple analytical techniques that provide both mass concentration and morphological information could facilitate comparisons between research and is critical to ascertain environmental presence and inform behaviour, fate, toxicity and source attribution.

10.2. Recommendations for future atmospheric plastics studies

10.2.1. Improving reliability via mitigating contamination and reporting

Microplastic contamination from the air is inevitable (Prata et al., 2021) during both sampling and laboratory analysis. Efforts such as washing or burning glassware (Song et al., 2021), rigorous cleaning (Prata, 2020), covering samples, minimising the use of plastic in experiments, controlling air flow/using exclusion rooms (Prata et al., 2020a) and wearing cotton clothing are important to ascertain the quality and repeatability of a study, yet often remain unreported (Wright et al., 2021). In future studies, mitigation measures should be reported, alongside contamination levels for field, procedural (Prata, 2020) and laboratory based blanks, adhering to the criteria suggested by Wright et al. (2021) to improve overall quality assurance and provide confidence in the results reported.

10.2.2. Sampling

Consensus regarding the appropriate methods to sample each matrix have not been established, with many of the samples analysed collected on an opportunistic basis. This inconsistency between sampling methods within each matrix inhibits comparability. For example, ambient air has been examined via the extraction of suspended particulates onto a filter, using differing apparatus which deliver various sampling volumes. Consistency of, or at the very least, reporting the pre-treatment or decontamination process of the collection filter, type (preferably non plastic (Trainic et al., 2020)), diameter and pore size should be mandatory. To understand plastics in ambient air more thoroughly, sampling within different geographical locations, surrounding potential point sources (including various distances from the point source) and at different tropospheric levels could provide further understanding regarding atmospheric plastic fate and behaviour.

Likewise, deposition has been collected using a variety of techniques between studies which inhibits comparability. The sample medium (plate, deposition collector, water bath), surface area, exposure duration, season, collection height and other variables should be disclosed. Deposition is also often collected in water or rinsed from the collection vessel using water. As microplastics have also been detected in water, this form of contamination should also be considered and reported in deposition work via the use of sample blanks. If possible, the mass of deposition should also be recorded. Further investigation of deposition at different heights (consistently between studies) or exploring the mitigation effects of trees or other medium also merits further investigation.

Dust has been collected using various techniques which limits comparability between results based on the efficiency of the method used to collect the particulates. For example, sweeping collects only a portion of particles <75–80 μm (Amato et al., 2010; Grottker, 1987). Preferably metal or other non-plastic apparatus should be consistently utilised as opposed to hands, with field blanks taken simultaneously.

10.2.3. Sample storage and processing

Sample storage protocols should eliminate the use of plastic containers (Abbasi, 2019) and bags (Kitahara and Nakata, 2020; Welsh, 2022) in preference for glass or metal storage containers. The effect of refrigeration and freezing samples and their impact on biological and photo degradation requires controlled investigation.

Depending on the analytical technique and sample matrix, various methods of sample treatment such as sieving, density separation and digestion have been explored, including digestion using hydrogen peroxide (H_2O_2), potassium hydroxide (KOH), Fenton's Reagent (Goßmann et al., 2022) or enzymes to remove interferences and biological material from environmental samples (Okoffo, 2019). Each digestion technique is subject to varying degrees of success regarding removing the matrix and minimising

degradation of the polymer for analysis (Okoffo, 2019). Hydrogen Peroxide digestion has the potential to discolour polyethylene terephthalate and polypropylene and can result in other visible changes to polyamide and polycarbonate. If heated, hydrogen peroxide digestion could reduce the size of polypropylene and degrade polyamide, polymethyl methacrylate and polystyrene, influencing spectral signals during Raman spectroscopy (Okoffo, 2019). Fenton's Reagent generates temperatures of 98 °C (if uncontrolled) during the exothermic reaction, which could potentially degrade plastics (Okoffo, 2019). Polymer degradation and the effect each different digestion method has on the plastics could therefore have an impact on the reported plastic concentrations, size fractions and polymetric composition (as a proxy for weathering or degradation/environmental exposure), particularly analysed using Raman spectroscopy techniques. Consistency across protocol for different matrices is necessary, along with undertaking and reporting recovery experiments.

10.2.4. Analysis

Although the volume of research on microplastics in air is growing exponentially, it becomes apparent that current knowledge regarding the polymer composition of these plastics in air, deposition and dust remains unknown. Often polymetric confirmation has not been undertaken for particles or fibres visually identified to be plastic. Where reported, understanding the polymer composition and proportion of plastics has been hindered largely by analytical technique limitations, instrument libraries (Stanton et al., 2019), lack of available standards and an insufficient number of individual plastics being sub sampled for polymer analysis (ranging between 20 (Dehghani et al., 2017), 21 (Abbasi, 2022), 34 (Abbasi and Turner, 2021b), 35 (Truong et al., 2021) particles and 3 % (Cabrera et al., 2022), 10 % (Wright et al., 2020), 15 % (Amato-Lourenço et al., 2022) of identified plastics). The field would benefit from the provision of a variety of different plastic standards to facilitate identification, open access spectral and mass-spectrometry libraries. Although cumbersome, the analysis of more (>100 individual plastic particles, >50 % of the total identified plastic particles or ≥50 individual particles per sample (Koelmans et al., 2019)) isolated plastic particles could improve the accuracy of particle-based extrapolation for understanding the polymetric composition. Furthermore, utilising confocal microscopy to individually measure the x, y and z (height) of microplastic fibres would facilitate an understanding of whether microplastic fibres conform to the 1/3 length to width ratio associated with detrimental health outcomes.

Method development has proven difficult for the lower sized microplastics in the air (Levermore et al., 2020; Wright et al., 2019), with detection limits of 1 μm for μ-Raman spectroscopy (Xu et al., 2019) and 10 μm for μ-FTIR, whilst the very particles remaining of interest for public health are ≤10 and ≤2.5 μm. Methods to analyse and quantify this fraction (<20 μm (Koutnik et al., 2021)) are paramount to provide a more accurate quantification of total plastic content, and this information could inform health and environmental fate assessments.

The majority of atmospheric microplastic studies have reported particle count concentrations with mass-based quantification for ambient air, deposition and dust all remaining understudied (Liu et al., 2019c; O'Brien et al., 2021). Size fractionated samples (<10, <2.5 and <1 μm) analysed using spectrometry techniques (LC-MS, ESI-MS, Pyr-GC–MS, TGA-MS) could provide further understanding regarding concentrations of microplastics in these pertinent size fractions, overcoming previous instrumentation limits of detection. Beyond microplastics, the exploration of nano plastics in environmental matrices (Materić et al., 2021; Wahl, et al., 1879; Ter Halle et al., 2017) and the observed relationship between microplastic and nano plastic concentration and composition (Allen et al., 2022) would be beneficial for atmospheric samples. Understanding the smaller size fractions is particularly important given the compounding influence of legacy plastics for atmospheric microplastics (Brahney, 2021) when degradation fractures plastics into sizes currently below detection limits.

10.3. Outdoor environment

10.3.1. Concentrations of plastics

Only one author has reported the concentration of micro rubbers (Abbasi, 2019; Abbasi, 2017), a plastic by definition but neglected component of microplastic analysis, in outdoor air microplastic concentrations. The further identification and quantification of micro rubbers in ambient air, dust and deposition would provide a better holistic understanding of atmospheric emissions and behaviour of plastics.

10.3.2. Outdoor emission sources

The percentage contribution of different atmospheric emission sources to the total concentration of microplastics in the air remains unknown. In the oceanic environment, a relationship has been observed between the concentration of microplastics in deposition and a surrounding shipping industry (Allen et al., 2020; Kernchen et al., 2022), therefore both the shipping and transportation industries require investigation as potential emission sources of microplastics into air. In the terrestrial environment, atmospheric emissions of microplastics (excluding tyre wear) from outdoor sources include wind erosion from landfill, dried landfill leachate (He et al., 2019) and entrainment from waste compaction (Hu et al., 2022), all of which require further investigation. Although waste water treatment plants have been shown not to emit microplastics into the air (Kernchen et al., 2022), the authors believe that wind abrasion of plastic rich biosolid applied to agricultural land (Brahney, 2021) is worthy of investigation. Within the built environment, fugitive dust from roads, construction and demolition sites contribute to $PM_{2.5}$ and PM_{10} (Ho et al., 2003). If containing plastic material, roads, construction sites and demolition could contribute to microplastic concentrations in the surrounding air, meriting exploration. The influence of ultra violet light and wind on scaffold net degradation and emission into the atmosphere would also prove informative (Mehmood and Peng, 2022). Additionally, the investigation of ultra violet light exposure as a mechanism of polyethylene degradation from synthetic grasses (Lozano and Ferguson, 2021) and resuspension behaviour from playing sport or movement on these materials could be investigated.

As a mitigation strategy, trees can retain atmospheric microplastics (40 microplastics/m^2/day or 16 % of the observed concentrations depending on rain volume (<14 mm) and intensity (<12 mm/day) (Huang et al., 2022)), thus reducing microplastic concentrations beneath the canopy and corresponding human inhalation exposure. The role of tree coverage into the urban environment merits investigation.

10.3.3. Suspended behaviour

Observations and patterns are beginning to emerge regarding atmospheric microplastics and geographic transportation distances, weather (wind, rain and monsoon transportation), dilution over the ocean (Allen et al., 2020), wind mixing downwind from emission sources (Liao et al., 2021), atmospheric degradation/fragmentation processes (Zhang et al., 2019) and vertical distribution (Li et al., 2020). These factors affect both ambient air concentrations and deposition of microplastics, and all require further examination to inform transportation behaviours. Understanding the concentration (microplastics/m^3 and mass/m^3), morphology, polymer characterization, and mass load of environmental microplastics in outdoor air would assist to understand the suspended microplastic load and plastic exposome. Topographical variables, land use, population density, human activity and other macro factors could be explored. Resuspension of settled plastics from road dust and household dust into the air also merits investigation.

The physical behaviour of the microplastic particles when suspended within the atmosphere also requires attention (Allen et al., 2020). This includes analysing the influence of particle characteristics, corona formation, in air particle aggregation (Prata, 2018; Amato-Lourenço et al., 2020), droplet formation, precipitation and snow. Laboratory based experiments to explore the behaviour for each different microplastic shape (fragment, fibre, foam, film) which affect drag forces and terminal velocity should be undertaken. Additionally, the influence of particle size (lower size

thresholds behave differently, influenced by static and particle aggregation), for each plastic composition (polypropylene, polyethylene terephthalate, polyethylene, polystyrene, polyvinyl chloride, polymethyl methacrylate, polycarbonate, each with different densities), differing aspect ratios (low aspect ratio take longer to sediment) (Rong et al., 2008) and fibre curvature should be undertaken in a systematic fashion to understand variables which influence the terminal velocity, residence time and thus transportation of microplastics in the atmosphere. Modeling has been predicated on the assumption that plastics are spherical (Koutnik et al., 2021), and should be improved as further details of suspended plastic characteristics, fate and behaviour become available.

10.4. Indoor environments

10.4.1. Indoor emission sources

Airborne microplastic emissions could potentially be sourced from a variety of avenues (Brahney et al., 2020). Microplastic concentrations have been shown to be higher in indoor dustfall and dust, with increased size variability in indoor air compared to outdoor. The increased concentration of plastics in indoor atmospheric environments could suggest that emission sources may originate from within the indoor environment, or alternatively, the indoor environment (or air conditioners) could act as a sink for atmospheric plastics. Further work towards the influence of air conditioner operation for capturing, emitting and resuspending microplastics requires additional clarification (Zhang et al., 2020a; Chen et al., 2022c), including the examination of variables such as split system unit compared to central plant (Uddin et al., 2022), operation duration, operation function/speed, occupant density, building area, occupant clothing or the influence of indoor air filtration efficiency (including HEPA filter) could shed light on indoor air concentrations and exposure. The influence of behaviour modifications including closing doors and modifying airflow from outdoors into indoor spaces (Chen et al., 2022c) could influence indoor microplastic concentrations and should be investigated. The mitigation effects of odour collection devices, droplet based deodorising systems (Hu et al., 2022) or humidifiers to reduce airborne microplastic inhalation exposure also merits further development.

It is known that human presence and activity influence particulate emissions into the surrounding air. Resuspension of dust or emissions in indoor environments from the floor based on human activity merits further investigation (Gasperi, 2015). Examination of the shedding propensity of various items of synthetic clothing into the air and respiratory cloud should also be considered as priority. Similar to Belzagui et al. (2019), the systematic study of microfibre detachment rates of synthetic clothing (polyester, acrylic and polyamide) during wear, emissions from ultra violet degradation whilst drying under sunlight (Akhbarizadeh et al., 2021) and mechanical drying (including the potential emission of nano plastics (Yang et al., 2021a)) should be undertaken to quantify emissions into the air from clothing.

Additional hypothesised but unquantified sources of atmospheric microplastics in the indoor residential environment include 3D printers (Amato-Lourenço et al., 2020; Zhang et al., 2017), carpets (Soltani et al., 2021; Amato-Lourenço et al., 2020) and synthetic bedding (Chen et al., 2022c; Board on Population Health and Public Health Practice et al., 2016). Baby wipes (Lee et al., 2021), sanitary items (Briain et al., 2020) and cleaning wipes (Lee et al., 2021) remain sources of polyethylene terephthalate, polypropylene and cellulose fibres into waste water treatment plants (Briain et al., 2020), however these woven fibre products are also frequently used in indoor environments. The shedding propensity and fate of these products during and after use remains unexamined yet could potentially be detected and remain in dust and/or capable of resuspension.

10.4.2. Indoor concentrations, characterization, and fate

The fate of indoor microplastics has been compared to that of particulate matter, being susceptible to Brownian diffusion, gravity deposition, lift force and turbulent fluctuation (Chen et al., 2022c). Indoor behaviour of different microplastic sizes, shapes, and polymer compositions merits investigation to better inform ingestion and inhalation exposure calculations where most people spend most of their time.

Indoor environments experienced comparatively higher concentrations of microplastics in deposition and dust comparted to outdoor environments, however lower concentrations of microplastics were observed in indoor ambient air compared to outdoor ambient air (SI: Fig. 8). Although this finding could be attributed to analytical size limitations which struggle in the smaller size fractions, understanding the concentration (microplastics/m^3 and mass/m^3), shape, size, polymer composition and relative proportion of microplastics in the atmospheric environment of differing indoor environments (including residential, office, school, hospital, commercial, retail) would inform indoor exposure assessments. Delving into more detail, specific building uses and variables such as internal space, occupant density, human activity, foot traffic volumes, cleaning frequency, furnishings and indoor/outdoor air exchange (Xie et al., 2022) would ultimately refine inhalation and ingestion exposure calculations for these specific indoor environments. Priority could be given towards understanding prevalence within indoor environments (Amato-Lourenço et al., 2020) where vulnerable populations are located including children, the elderly and occupational exposure settings.

10.5. Human exposure

10.5.1. Concentration for exposure assessment

Exposure to suspended microplastics is unavoidable (Xie et al., 2022). Understanding concentrations in all facets of the atmospheric environment would improve the accuracy of exposure calculations. Morphological information (indulging shape and size) in the differing atmospheric environments could elucidate health outcomes and inform risk assessments, as particulates <5 μm can enter the lung (Jabbal et al., 2017). In the meantime, the back calculation of microplastics detected in deposition/dustfall and dust, although less likely to be inhaled (Xie et al., 2022; Zhang et al., 2020c), could be used for preliminary inhalation and ingestion exposure calculations based on entrainment and resuspension.

10.5.2. Occupational settings

In an occupational setting, workers could be exposed to atmospheric microplastics in various employment situations. More broadly, para occupational exposure to microplastics adhered to clothing and transported to the residential environment could affect others. In a retail environment, occupational exposure for both workers and customers of nail salons requires attention (Chen et al., 2022b). In an industrial setting, plastic fibres may be degraded from larger fibrous items such as textiles (flocking industry, dry cleaning) and building materials (Liu et al., 2019a; Li et al., 2020), affecting construction industries. Preliminary evidence of PTFE and polyethylene fibres and pink polymethyl methacrylate spheres, hypothesised to be generated from industrial applications and paints, have been identified in deposition (Brahney et al., 2020). Therefore, the spray application and degradation of paint (Brahney et al., 2020), industrial coatings and abrasives (Brahney et al., 2020; Gregory, 1996) requires quantification as a source of microplastics into suspended air (Xie et al., 2022) and exploration of potential occupational exposure. Microplastics abraded from personal protective equipment (PPE, such as masks and synthetic uniforms) requires investigation, based on the immediate inhalation risk (Li et al., 2021) and application across industries.

11. Conclusion

Microplastics have been ubiquitously detected in the environment, including within all facets of the atmospheric environment, in remote areas (Zhang et al., 2019) and at high altitudes (Allen et al., 2019). Microplastics have been identified and quantified in both ambient indoor and outdoor air, indoor dustfall, outdoor deposition, indoor dust, outdoor dust, road dust, snow and using various bio monitors. Whilst the number of publications covering this topic have recently exploded, there remains many gaps in knowledge. Specifically, studies have historically reported

microplastic concentrations as a particle count per sample of air (m^3), mass (g) or liquid (L), which without relying on various assumptions that may or may not be applicable, be converted into mass-based concentrations, to become consistent with other containment reporting and facilitate regulation. Where mass-based quantification has occurred, each has been limited to specific polymer types and fail to report morphological details, which are relevant to determining movement and fate throughout the environment. The actual atmospheric fate, behaviour, transformation and degradation whilst airborne for microplastics still needs further investigation (Allen et al., 2020). Above all, this field requires urgent attention to clarify concentrations (particularly in the smaller size fractions) to accurately inform human ingestion and inhalation exposures. Beyond exposure, the systematic review and assessment of toxicities of varying plastic compositions, sizes, shapes, surface characteristics, degradation status and mixtures to this ubiquitous contaminant are required for each human exposure avenue as air, deposition, dust affects all.

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CRediT authorship contribution statement

Stacey O'Brien – conceptualisation, data curation, formal analysis, investigation, methodology, validation, writing original draft, reviews and editing.

Cassandra Rauert- data curation, investigation, methodology, validation, writing - reviews and editing, supervision.

Francisca Ribeiro - writing - reviews and editing.

Elvis Okoffo - writing – review and editing.

Stephen Burrows - writing – review and editing.

Jake O'Brien - conceptualisation, data curation, formal analysis, methodology, writing - reviews and editing, software, supervision.

Xianyu Wang - writing - reviews and editing, supervision.

Stephanie Wright - methodology, validation, writing - reviews and editing, supervision.

Kevin Thomas – conceptualisation, funding acquisition, methodology, formal analysis, resources, supervision, validation, writing – reviewing and editing.

Data availability

Data will be made available on request.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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