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Review

Microplastics in Lichen Thalli: A Photo or a Movie of Local Atmospheric Deposition?

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Abstract

Airborne microplastics (MPs) are a global issue, and there is an urgent need to prevent their spread in the environment. Sensitive and reliable methods are also needed to assess their deposition and effectively evaluate risk in terrestrial ecosystems. Current automated monitoring devices are expensive and do not enable large-scale mapping of MP deposition. As with other persistent atmospheric contaminants, developing accurate, cost-effective and easily applicable biomonitoring methods would therefore be highly beneficial. Cryptogams are among the most suitable biomonitors of airborne contaminants, and preliminary surveys show that epiphytic lichens accumulate higher concentrations of MPs in urban areas and near landfills than in control sites. However, the interaction between lichen thalli and MPs is weak and, as discussed in this review, the anthropogenic fibres and plastic fragments intercepted and retained by lichens probably do not reflect the levels in bulk atmospheric deposition. While emphasizing the need for studies evaluating the effectiveness of cryptogams in accumulating different types of airborne MPs under various meteorological conditions, this review also suggests directing future research efforts toward mosses, which seem to accumulate much higher concentrations of MPs than lichens in both active and passive biomonitoring surveys.

Keywords: airborne microplastics; biomonitoring; epiphytic lichens; MP interception; MP retention; future research

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1. Introduction

In the mid-19th century, Nylander's [1] pioneering study in Paris showed that epiphytic lichens growing on chestnut tree trunks were sensitive to atmospheric pollutants. In the following decades, suitable methodologies were developed to evaluate air quality on a large scale using the biodiversity of epiphytic lichens [2,3]. In the 1960s, analytical methods were discovered and developed for determining trace elements and persistent organic pollutants (POPs) in environmental matrices. Some widespread species of lichens that are less sensitive to phytotoxic contaminants became the most commonly used semi-quantitative biomonitors of atmospheric contaminant deposition [4]. In subsequent years, methods were established for collecting, preparing, and analyzing suitable lichen species that grow naturally in the study area (passive biomonitoring) or that are exposed for a defined period in urban and industrial environments (active biomonitoring) [5,6]. The wide geographical distribution of lichens, their lack of roots and dependence on atmospheric deposition for water and nutrients, and the absence of a cuticle to prevent the ad/absorption of airborne (gaseous, particulate or soluble) contaminants make them ideal accumulative biomonitors

of air pollutants. Moreover, lichen thalli have a long life cycle and a slow growth rate. They are photosynthetically active throughout the year and accumulate in their thalli high and easily detectable concentrations of elements or compounds which occur in atmospheric deposition at very low and variable concentrations [4]. Hundreds of surveys performed in recent decades demonstrate that lichen biomonitoring is a cost-effective approach that does not require frequent sampling and provides valuable information on the sources and deposition patterns of atmospheric contaminants at regional and national scales. This approach plays a fundamental role in remote regions [7–10]. However, while the results of passive and active biomonitoring indicate that concentrations of persistent contaminants in lichen thalli can reflect changes in atmospheric deposition over time and space, quantitative bioaccumulation data alone cannot accurately estimate contaminant concentrations in bulk atmospheric deposition. The physico-chemical and biological processes involved in the interception, uptake and accumulation of airborne contaminants by lichen thalli differ from those that take place in automatic samplers of wet, dry or bulk atmospheric deposition. Furthermore, due to the continuous interaction and exchange between epiphytic lichens and their environment, persistent atmospheric contaminants are not necessarily accumulated in a definitive and integrated manner according to exposure time [4]. Therefore, lichen biomonitoring is a complementary approach and cannot be considered a substitute for conventional instrumental monitoring of airborne contaminants [11].

The first reports of plastic debris in the marine environment date back to the 1970s [12], and over the past two decades, hundreds of papers have reported the occurrence of microplastics (MPs, particles measuring ≤ 5 mm) in aquatic and terrestrial ecosystems. MPs can carry harmful chemical additives and also act as vectors for other pollutants adsorbed from the surrounding environment. Several studies have evaluated their bioavailability and potential transfer along food webs, and the reader can refer to some recent reviews, such as that of Thompson et al. [13] (and references therein), for the knowledge acquired so far on these topics. In general, the lack of standardized methodologies and the non-comparability of results between studies are highlighted, as are the limitations of current methods in identifying the tiniest, highly bioavailable particles (nanoplastics, NPs, typically ≤ 1 µm) in environmental matrices. These limitations make it impossible to reliably evaluate the potential threat posed by MPs and NPs to human and ecosystem health [13].

Although most studies on MPs have focused on the marine environment, a recent estimate based on the most reliable studies of MPs' sources and pollution suggests that over 8.9 million tons of pellets, paints and tyres are released to land every year, compared to 3.8 million tons that leak into the oceans [14]. Moreover, through long-range atmospheric transport, smaller MPs, especially microfibers (MFs), can reach the Earth's most remote regions, such as Mt. Everest [15] and Antarctica [16]. Similar to what is being carried out with airborne metals, POPs, and artificial radionuclides, it would be appropriate to develop biomonitoring procedures for large-scale, long-term surveys of MP deposition in terrestrial ecosystems. Lichens, together with mosses, lacustrine sediments, peat, snow and ice cores, are among the most suitable environmental matrices for these surveys. Vaid et al. [17] reviewed and discussed the challenges of MP biomonitoring in terrestrial ecosystems using lichens, mosses, plant leaves, honeybees, and other organisms. They concluded that, although cryptogams could be suitable biomonitors, the results obtained so far are highly variable and cannot be compared due to the lack of harmonized and standardized methodologies.

Using terrestrial cryptogams to evaluate airborne MP deposition is in its infancy, and there is an urgent need for a better understanding of all factors affecting the variability and reliability of results. This includes assessing the capability of different species of foliose and fruticose lichens to intercept and retain MPs of different shapes and sizes in different mete-

orological conditions. Based on literature data on the occurrence of MPs in epiphytic lichen thalli, this review will discuss the physico-chemical and biological mechanisms involved in their interception and retention. Quantifying and characterizing MPs in environmental matrices poses more technical challenges than analyzing atmospheric deposition [17]. Thus, this review will suggest research priorities to improve the available methodologies.

2. Atmospheric Microplastics and Their Deposition

MPs are highly dynamic atmospheric contaminants with extremely variable concentrations in space and time, primarily depending on meteorological conditions, population density, and human activities [18–20]. On a global scale, the main source of airborne MPs is likely the formation of spray at the air/water interface in oceans and other bodies of water [21]. Moreover, it has been estimated that approximately 136,000 tons per year of MPs can be transferred from aquatic to terrestrial environments through bubble burst ejection and wave action [22]. On land, the main sources of MPs are waste disposal, wind abrasion, and industrial, construction and other human activities such as wearing or drying synthetic clothing and using vehicles, which favours the resuspension of tyre wear and other MPs from the ground. Furthermore, plastic fragments in terrestrial environments are exposed to greater temperature fluctuations and UV radiation than those in water and therefore degrade more rapidly into small particles [23]. Once suspended in the air, MP particles with smaller density and size can be transported by the wind over long distances. After temporary deposition, they can be re-suspended and moved elsewhere. Through repetition of these cycles, MPs can reach the most remote regions of Earth via a "grasshopper-like" transport mode [18–20].

Although indoor or outdoor airborne MPs with an aerodynamic diameter of less than 10 μm can be inhaled, and those <2.5 μm can penetrate deeper into the alveolar areas of the lungs, posing potential health risks, most studies have focused on MPs deposited on roads or other surfaces rather than those suspended in the air [24]. In general, available data on atmospheric concentrations of MPs are highly variable and dependent on sampling methods and analytical techniques used. For instance, studies using microscopy report an average of 2.5 MPs per m^3 of air, whereas studies using Fourier Transform Infrared (FT-IR) Spectroscopy or Raman Spectroscopy report an average of more than 350 items per m^3 of air [20]. As a rule, available data show that atmospheric concentrations of MPs over oceans are much lower than over inland areas, with values much higher in urban environments. Even in samples collected by aircraft above the planetary boundary layer, values are higher in urban areas (13.9 items per m^3) than in rural areas (1.5 items per m^3) [25]. Airborne MPs are mainly deposited through rain and snow (wet deposition) or through gravity and other physicochemical processes that determine their settling onto exposed surfaces (dry deposition). Furthermore, since MP deposition is favoured by atmospheric humidity [18], fog, clouds, mist, dew and other often overlooked occult deposition processes must play a significant role in transferring MPs to the ground and reducing their possible resuspension, at least temporarily. Thermal inversion phenomena, in which the air temperature is higher at altitude than at ground level, should also reduce their transfer and transport in the air [26]. Nevertheless, our understanding of how different and variable meteorological conditions affect the transport and deposition of MPs is limited. It is generally believed that raindrops, especially snowflakes, are effective at washing away airborne MPs. It is also demonstrated that their deposition is positively correlated with precipitation intensity and volume [18]. Assessing MPs deposited in the absence of precipitation (dry deposition) is challenging because it results from gravity (sedimentation, impaction), as well as surface electrostatics and diffusive Brownian motion. This process is also influenced by the physico-chemical characteristics of the settling particles and the exposed surfaces.

In general, the sampling of atmospheric airborne MPs is performed using active or passive methods. The latter are easy to use, low-cost and do not require electricity. They essentially involve collecting airborne particles in a glass container via a funnel (wet deposition) or in Petri dishes or glass slides with adhesive surface (dry deposition). These approaches are suitable for long-term, large-scale outdoor monitoring, but they have several drawbacks. Insects, airborne plant propagules, and other materials can contaminate samples, and adhesive substrates can chemically contaminate samples and prevent embedded particles from being seen with an electron microscope [27]. Moreover, to reliably evaluate the results, it is necessary to record the weather conditions during the sampling period.

Active methods for collecting airborne MPs consist of pumping and filtering known volumes of air, similar to the routine monitoring of atmospheric particulate matter (PM_{10} or $PM_{2.5}$). This approach enables the assessment of MP numbers or mass per m^3 of air in indoor and outdoor environments.

Each sampling method has its own advantages and limitations and is usually chosen according to the research objectives and the characteristics of the study area. Unfortunately, there are currently no standardized methodologies for sampling, identifying, and classifying MPs, which prevents reliable comparisons between the results of different surveys. However, Liu et al. [18] provide an updated, comprehensive report on the available worldwide data on the wet and dry deposition of MPs.

3. The Interception of Persistent Atmospheric Contaminants by Epiphytic Lichens

Most lichen and moss biomonitoring surveys conducted thus far have assumed that MPs accumulate almost linearly, similar to the bioaccumulation of POPs or trace metals. However, this assumption has not been proven and seems unlikely. Airborne POPs and natural or anthropogenic particles containing trace metals have different shapes and physico-chemical characteristics than MPs [27]. For example, atmospheric polychlorinated biphenyls (PCBs) can interact with the wax/lipid components on the surface of plant leaves and cryptogam thalli, which have a high cation exchange capacity and can bind inorganic elements through passive processes [28]. Conversely, due to their mostly fibrous shape and largely hydrophobic nature, MPs likely interact very weakly with lichen, moss or leaf surfaces. Their entrapment and accumulation depend largely on their shape and size, meteorological conditions, and the species-specific micro-morphological and chemical features of biological surfaces [27]. While extracellular protonated exchange sites in the cell wall and the outer surface of the plasma membrane of lichens contribute to the ad/absorption of major and trace elements in airborne soil dust and anthropogenic inorganic particles [28], the entrapment of MPs probably depends primarily on the morphology (branching, wrinkling, or roughness) of the lichen thalli, and the retention rate is likely more prone to change under the action of wind and rain. The absorption of cations through passive physicochemical processes, as well as the accumulation of airborne particles in the intercellular spaces of the lichen medulla, increases the concentration of chemical elements in the older parts of the thallus [28]. However, laboratory and field experiments show that extracellular cation binding is a reversible process, wherein cations with lower binding affinity can be displaced by those with higher affinity [28]. Thus, the dynamic equilibrium between the inputs and outputs of elements in lichen thalli is one reason why metal concentrations in these biomonitors do not reliably reflect metal concentrations in bulk atmospheric deposition [4,11]. A linear relationship between MP bioaccumulation and exposure time has yet to be demonstrated in lichens, and it is even more unlikely that

the number of MPs accumulated per g or cm^2 of lichen thalli can accurately reflect spatial and temporal variations in atmospheric deposition.

Nevertheless, using epiphytic lichens for passive or active biomonitoring of metals and other persistent contaminants is a very useful approach for obtaining information on deposition patterns, and it is also worth thoroughly investigating whether these organisms can be used for biomonitoring the deposition patterns of airborne MPs. Since passive and, in particular, active sampling devices are usually placed in a limited number of fixed locations, any additional information obtained using lichens or mosses, which are widespread in almost all terrestrial environments, would be extremely valuable, particularly in remote regions.

4. What Do the Preliminary Results of MP Accumulation in Lichens Tell Us?

To the best of our knowledge, Roblin and Aherne's study [29] was the first to evaluate the accumulation of anthropogenic microfibers in cryptogams. In May 2018, they collected the moss *Hylocomium splendens* in three background areas in Ireland. After wet peroxide digestion, the samples were filtered onto glass-fibre filter papers and dyed with Rose Bengal (a biological stain that binds to natural fibres). The filter papers were examined under a stereomicroscope using a five-criteria visual method, and fibres that were not stained by Rose Bengal and met two of the criteria were classified as anthropogenic MFs. The number of those that met 3, 4 or 5 criteria were also recorded and some of them were also analysed with Raman spectroscopy. Anthropogenic MFs occurred in all samples with a mean concentration of 24 items per g dry wt of moss, and a mean length of 1.02 mm. Based on the five-criteria visual assessment, from 13 to 27% of anthropogenic MFs were plastic. Despite possible local differences in meteorological conditions and moss growth rates among the three background areas, and the unknown actual capability of *H. splendens* to intercept and retain MPs, an annual average deposition of about 12,000 plastic MFs per m^2 was estimated by assuming 2 kg of dry wt of moss per m^2 of ground surface [29].

Using the same methodology (detailed further in Annex 5 of the ICP Vegetation protocol [30]), a subsequent survey was conducted collecting samples of the epiphytic foliose lichen *Flavoparmelia caperata* at increasing distances from a landfill site in central Italy [31]. MFs and fragments that met at least two criteria of the visual method were considered anthropogenic, and their number decreased at increasing distances from the landfill (Table 1). The percentage of petrochemical-based polymers (about 40% across all sites) was determined by pressing a hot needle against the tip of randomly selected fibres or fragments. Raman spectroscopy was used to determine the polymer composition, which showed a prevalence (68%) of polyester or polyethylene terephthalate (PET), followed by polyethylene (26%), and polystyrene (5%) [31]. The number of MFs accumulated per g of lichen thalli at the control site (Table 1) was comparable to the number detected in mosses from background areas in Ireland. The authors analysed only the outermost part of the lichen thalli (roughly one year of growth), while in Roblin and Aherne's study moss samples were 2–3 years old. Thus, assuming that lichens accumulate MFs linearly with exposure time, they estimated that the number of MFs accumulated by epiphytic lichens around the landfill could be up to 6 times higher than in mosses from background areas of Ireland. Moreover, since the specific surface area of *F. caperata* was evaluated to be 3 times greater than that of *H. splendens*, they estimated that lichens around the landfill accumulated up to 20 times more MPs than moss [31].

Table 1. Concentrations (items g^{-1} dry wt) of anthropogenic microfibres (MFs), fragments (F), tyre wear particles (TWPs) and microplastics (MPs) in thalli of different species of lichens from passive and active biomonitoring surveys in background and urban areas.

Sampling Site	Lichen Species	Identification	% of MFs, F and MPs, Mean Size (μm)	Abundance (items g^{-1} dry wt)	Ref.
Near a landfill dumping site (Central Italy) Control site	*Flavoparmelia caperata*	Stereomicroscope, Hot needle test, Raman spectrosc.	MFs 41%, 634 F 59%, 45 MFs 73%, 796 F 27%, 66	60 86 23 8	[31]
Milan (N Italy) City centre Urban parks Semi-periphery Periphery Control site	*Evernia prunastri* Transplants 3-month exp.	Stereomicroscope, Hot needle test	MFs 97 ± 3%, 1076 ± 156 MFs 91 ± 5%, 867 ± 146 MFs 95 ± 2%, 951 ± 143 MFs 93 ± 5%, 885 ± 117 MFs 100%, 616 ± 92	44 ± 1 26 ± 1 48 ± 3 56 ± 5 20 ± 4	[32]
Five remote areas (Central Italy)	*Evernia prunastri*	Stereomicroscope, Hot needle test	MFs 100%, 2286 ± 660 MFs 100%, 1129 ± 357 MFs 100%, 2844 ± 821 MFs 99.2%, 2157 ± 557 MFs 100%, 2025 ± 569	8.6 ± 3.8 7.9 ± 3.5 9.5 ± 4.2 12.5 ± 5.6 10.2 ± 4.6	[33]
Naples (South Italy) Roof (urban building)	*Pseudevernia furfuracea* 6-week exp.	Stereomicroscope, Raman spectrosc.	MFs 90%, 1200 ± 100	90 ± 5	[34]
Rome (Central Italy) Urban site Protected area Background area	*Cladonia sp.* and *Xanthoria sp.*	Stereomicroscope, FT-IR spectrosc.	MFs 94%, 2000 ± 100 MFs 99%, 1600 ± 100 MFs 99%, 1500 ± 200	580 236 39	[35]
Istanbul (Türkiye) Urban areas	*Xanthoria parietina*	Fluorescence microsc. (Nile red staining), FT-IR spectrosc.		22–190 *	[36]
Pisa (Central Italy)					
Parking lots	*E. prunastri* 7-week exp.	Stereomicroscope, Hot needle test, FT-IR spectrosc.	MFs 30%, 1291 ± 87 F 31%, 219 ± 24 TWPs 39%, 101 ± 7	4.66 ± 0.77 5.11 ± 1.47 6.57 ± 2.63	[37]
Urban parks	*E. prunastri* 7-week exp.		MFs 32%, 1288 ± 137 F 31%, 151 ± 18 TWPs 37%, 119 ± 8	1.00 ± 0.21 2.16 ± 0.54 2.63 ± 0.71	
Rural sites	Native		MFs 35%, 1039 ± 414 F 40%, 87 ± 24 TWPs 25%, 82 ± 15	0.78 ± 0.22 0.89 ± 0.26 0.56 ± 0.34	
Control area	Native		MFs n.r., 847 ± 191 F n.r., 199 ± 11 TWPs n.d.	1.67 ± 0.33 0.33 ± 0.33 n.d.	

* n of MPs in 2.5 μm^2 of lichen thalli; n.d. = not detected.

Without better knowledge of the mechanisms involved in the interception and retention of MPs by cryptogams and the impact of different local meteorological conditions, estimating deposition fluxes of MFs as the number of items per square meter of lichen thalli or moss carpets, and making comparisons among studies using different organisms seems unreliable. Furthermore, biomonitoring studies of the atmospheric deposition of other persistent atmospheric contaminants has shown that, due to their different morphology

and specific surface area, mosses and lichens behave differently in intercepting and sequestering contaminants. In general, mosses are more effective than lichens at trapping airborne particulates, such as soil dust containing lithophile elements, or high-molecular-weight Polycyclic Aromatic Hydrocarbons (PAHs), which are mainly associated with airborne particles. Meanwhile, epiphytic lichens are more efficient at accumulating atmophile elements or low-molecular-weight PAHs, which are mainly gaseous [4,38–40]. Thus, as was partly predictable, significantly higher concentrations of synthetic MFs were found in epigeic mosses (12–17 items g^{-1} dry wt) than in epiphytic lichens (8–12 items g^{-1} dry wt) collected in the same background areas of Central Italy [33]. One might object that epiphytic lichens are sheltered by the tree canopy, whereas moss cushions grow and are collected in open areas. However, a survey in which mosses and lichens were exposed for six weeks in cotton bags in Campania (S Italy), in areas with varying levels of human activities, showed a higher mean number of anthropogenic MFs in the moss *Hypnum cupressiforme* (102 ± 24 items g^{-1} dry wt) than in the lichen *Pseudevernia furfuracea* (87 ± 17 items g^{-1} dry wt) [34]. Moreover, the calculated daily deposition flux (items m^{-2}/day) was always positive for mosses, whereas for lichens, it was positive only in a parking area and negative at the other exposure sites [34]. These results are preliminary but seem to indicate that MPs in bulk atmospheric deposition are only partially intercepted and retained in lichen thalli.

In essence, data summarized in Table 1 suggest that the concentration of MPs in different species of lichens (collected in the field or exposed for several weeks) is lower in control sites than in urban areas and near a landfill dumping site. Samples from the latter site showed a higher percentage of fragments, as well as a much higher number of plastic particles among isolated anthropogenic MFs [31]. In most study areas, lichens mainly accumulate synthetic MFs from textile clothing, packaging, rubber tyre abrasion, furniture, and other anthropogenic sources. However, active biomonitoring in the urban area of Pisa reported similar percentages of MFs, F, and TWPs (Table 1). In most background areas in central Italy, the length of fibres was >2 mm (Table 1). However, in sites selected as controls for surveys performed in Milan, Rome, and Pisa, lichens seem to intercept shorter MFs, which are likely transported more easily over long distances [32,35]. Far fewer MPs were detected in lichens exposed for 6–7 weeks or three months in different Italian towns than in lichen samples used as passive samplers in Rome (Table 1). In the latter study, two genera were analysed, and the fruticose genus *Cladonia* accumulated more MPs than the foliose genus *Xanthoria*, though the difference was not statistically significant [35]. Moreover, no correlations were found between thallus size and the number of intercepted MPs. The distribution pattern of MPs corresponded with particulate matter deposition, decreasing from the city centre to the periphery. However, an opposite trend was observed with active biomonitoring in the urban area of Milan (Table 1). In the latter city, the number of MPs intercepted by exposed lichens increased from the city centre to the periphery, and the number in urban parks was in the same range as in the control area (Table 1). Furthermore, this distribution pattern differed from that observed in a previous lichen biomonitoring study of trace element deposition in the same urban area [41].

5. Are Lichens Bioindicators or Biomonitors of Microplastic Deposition?

Although the terms bioindication and biomonitoring are sometimes used interchangeably, strictly speaking, bioindication involves using organisms to obtain information about the quality of their environment (e.g., lichen biodiversity is an indicator of air quality), whereas biomonitoring involves the continuous detection of semi-quantitative data through bioindicators called biomonitors. As Witting exemplifies [42]: "The difference between bioindication and biomonitoring is the same as that between a photo and a film". Thus, although the data summarized in Table 1 overall indicate higher concentrations of MFs, tyre

wear, and other synthetic particles in lichen thalli from areas impacted by human activities, it is not yet known whether lichens intercept and retain all types of MPs in ways similar to other persistent atmospheric contaminants. Unlike the rapid partitioning of airborne PCBs and metal ions in the fungal hyphae and algal cells of lichens, interactions of MPs with lichens are probably weaker and more easily removed by wind and rain. Exposing lichen thalli to a controlled suspension of microfibres of different sizes for 4 weeks revealed that the number of intercepted MFs correlated with their deposition rate; however, only smaller particles exhibited a higher retention rate [43]. Due to the clear divergence between intercepted and accumulated MPs and the inability to differentiate between them, the authors suggested using the term "retained" to include both loosely and strongly ad-absorbed MPs. Unfortunately, there are no specific studies on the effects of different meteorological conditions on the retention of polymers of different shapes and sizes on lichens.

Interactions between plant leaves and MPs are influenced by waxy cuticles and hydrophobicity, which may increase their retention compared to cryptogams. Moreover, smaller plastic particles could enter the stomata [44]. Nevertheless, significant spatio-temporal fluctuations in the number of MPs have been reported in plant leaves [27]. During a 7-day survey, the abundance of MPs collected by a wide-mouthed glass bottle increased linearly. However, in the leaves of nearby plant species, the abundance of MPs showed strong diurnal fluctuations, and variations within the same leaf type were higher than those among different plant species [27]. Leonard et al. [45] found a number of MPs ranging from 0.1 to 50 items cm^{-2} in different tree leaves from the same spot, and their number peaked within the 0.6–1.2 m height range above the ground. Due to the effects of different wind streams at varying elevations, variations in MP concentrations in leaves in different position in the tree canopy were higher than those among samples of the same species collected in parking lots, commercial areas, and residential areas. Therefore, it was concluded that tree leaves cannot be used as passive samplers of MPs until a sampling protocol is developed that accounts for data variability [45]. However, although methods for sampling of leaves in specific positions within the canopies of suitable tree species could be standardized, local variations in wind and rainfall would still make it difficult to compare results from different sampling sites. Thus, spatial and temporal variations in meteorological conditions could prevent achieving the main goal of biomonitoring, which is defining reliable maps of MP deposition. Nevertheless, tree canopies and bark play an important role in intercepting MPs, and the subsequent removal of these particles by rain and wind creates a series of interception cycles. This makes indigenous evergreen tree species a useful tool for mitigating atmospheric MPs contamination in urban areas [46]. Mosses and lichens associated with trees can also contribute to this ecological service [46], but future research on cryptogams must verify their potential effectiveness as bioindicators or biomonitors of airborne MP deposition. Laboratory experiments and field surveys will be necessary to evaluate the capability of different lichen species to intercept and retain MPs of different shapes, sizes, and densities under different meteorological and environmental conditions. Figure 1 summarizes future research directions for evaluating the potential role of cryptogams in biomonitoring atmospheric MP deposition. An essential prerequisite is developing cost-effective, reliable, and standardized protocols for extracting and identifying MPs. To improve current methodologies, novel approaches have been suggested, such as an environmentally friendly method for separating and identifying MPs in biological and inorganic environmental matrices [47–50]. Once suitable methodologies have been established, it will be necessary to determine if MPs accumulate linearly and if their quantity and types reflect those in samples of bulk atmospheric deposition collected at the same site. Most existing surveys have estimated annual MP deposition fluxes in lichens and mosses and have often found comparable results to those reported in the literature

using conventional active or passive samplers. However, due to the variability in available data on MPs concentrations in atmospheric deposition and uncertainty about cryptogams' ability to retain intercepted MPs, these comparisons are not very reliable.

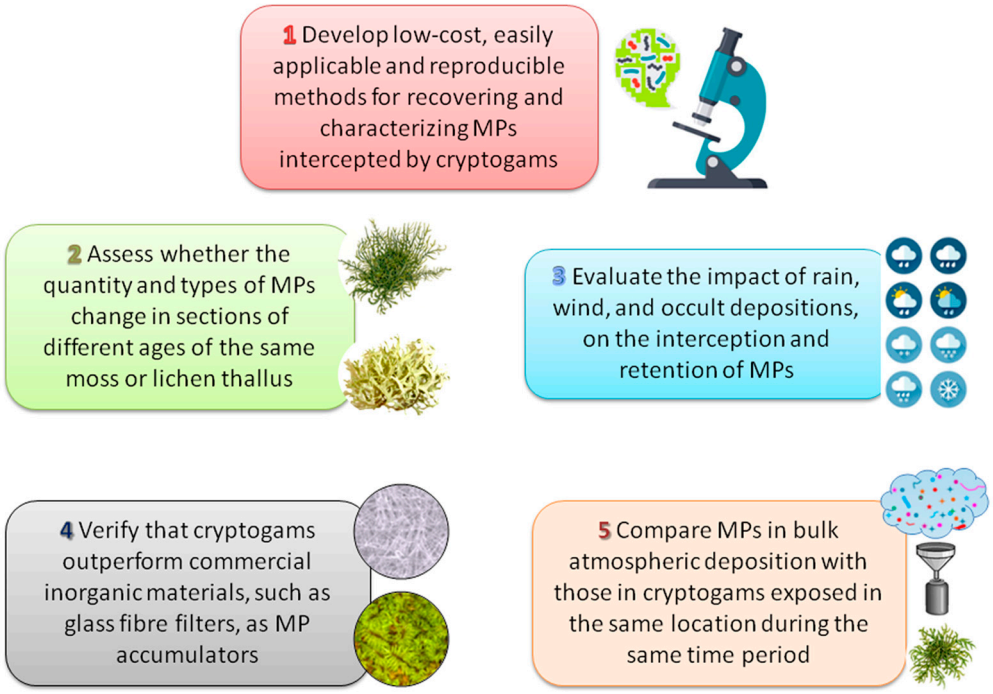

Figure 1. Challenging research directions for evaluating the potential role of cryptogams in monitoring airborne microplastics (MPs).

Finally, it would be appropriate to compare the performance of cryptogams as MP accumulators with commercial inorganic materials such as glass fibre filters, which allow for much easier MP recovery.

6. Conclusions

Airborne MPs pose potential risks to human health and ecosystems, so policymakers and environmental managers need adequate tools to address this challenge. As with other persistent atmospheric contaminants, especially in urban and industrial areas, it is more practical to use standardized protocols with automated monitoring devices that can provide real-time information on MP concentrations and, if necessary, trigger mitigation measures. However, instrumental monitoring requires electricity, is expensive, and can only be implemented in a limited number of stations. Thus, biomonitoring with cryptogams would be a very important complementary tool to achieve information on the distribution of airborne MPs at different spatial scales, especially in remote regions, once it is demonstrated that the MPs they intercept and retain somehow reflect those in bulk atmospheric deposition. Preliminary surveys indicate higher concentrations of MPs in lichen samples collected in urban areas than in control sites. However, the available results are generally characterized by high variability and inconsistencies. Moreover, epigeic mosses accumulate greater quantities of MPs than epiphytic lichens in both active and passive biomonitoring surveys. Therefore, future research efforts should focus on standardizing sampling and analytical procedures for mosses. Unlike lichens, epigeic mosses are directly exposed to wet, dry and occult atmospheric deposition. In some widespread species, such as *Hylocomium splendens*, each step along the stem represents a year of growth. Analyzing samples of different ages could be useful for evaluating temporal trends in MP accumulation and their annual accumulation rate.

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Abbreviations

The following abbreviations are used in this manuscript:

F	Fragments
FT-IR	Fourier Transform Infrared Spectroscopy
MFs	Microfibres
MPs	Microplastics
NPs	Nanoplastics
$PM_{2.5}$	Particulate Matter with aerodynamic diameter <2.5 μm
PM_{10}	Particulate Matter with aerodynamic diameter <10 μm
POPs	Persistent Organic Pollutants
PCBs	Polychlorinated Biphenyls
PAHs	Polycyclic Aromatic Hydrocarbons
PET	Polyethylene Terephthalate
TWPs	Tyre Wear Particles

References

1. Nylander, M.W. Les lichens du Jardin du Luxembourg. *Bull. Soc. Bot. France* **1866**, *13*, 364–371. [CrossRef]
2. Hawksworth, D.L. Lichens as litmus for air pollution: A historical review. *Intern. J. Environ. Stud.* **1971**, *1*, 281–296. [CrossRef]
3. Anderson, F.K.; Treshow, M. Responses of lichens to atmospheric pollution. In *Air Pollution and Plant Life*; Treshow, M., Ed.; John Wiley & Sons: Chichester, UK, 1984; pp. 259–289.
4. Bargagli, R. *Trace Elements in Terrestrial Plants: An Ecophysiological Approach to Biomonitoring and Biorecovery*; Springer: Berlin/Heidelberg, Germany, 1998; 324p.
5. Bargagli, R.; Nimis, P.L. Guidelines for the use of epiphytic lichens as biomonitors of atmospheric deposition of trace elements. In *Monitoring with Lichens—Monitoring Lichens*; Nimis, P.L., Scheidegger, C., Wolseley, P.A., Eds.; NATO Science Series; Springer: Dordrecht, The Netherlands, 2002; Volume 7, pp. 295–299. [CrossRef]
6. Tretiach, M.; Adamo, P.; Bargagli, R.; Baruffo, L.; Carletti, L.; Crisafulli, P.; Giordano, S.; Modenesi, P.; Orlando, S.; Pittao, E. Lichen and moss bags as monitoring devices in urban areas. Part I: Influence of exposure on sample vitality. *Environ. Pollut.* **2006**, *146*, 380–391. [CrossRef]
7. Bargagli, R.; Nimis, P.L.; Monaci, F. Lichen biomonitoring of trace element deposition in urban, industrial and reference areas of Italy. *J. Trace Elem. Med. Biol.* **1997**, *11*, 173–175. [CrossRef]
8. Bargagli, R.; Sanchez-Hernandez, J.C.; Monaci, F. Baseline concentrations of elements in the Antarctic macrolichen *Umbilicaria decussata*. *Chemosphere* **1999**, *38*, 475–487. [CrossRef]
9. Bergamaschi, L.; Rizzio, E.; Valcuvia, M.G.; Verza, G.; Profumo, A.; Gallorini, M. Determination of trace elements and evaluation of their enrichment factors in Himalayan lichens. *Environ. Pollut.* **2002**, *120*, 137–144. [CrossRef]
10. Yang, J.; Oh, S.-O.; Hur, J.-S. Lichen as bioindicators: Assessing their response to heavy metal pollution in their native ecosystems. *Mycobiology* **2023**, *51*, 343–353. [CrossRef]
11. Bargagli, R. Moss and lichen biomonitoring of atmospheric mercury: A review. *Sci. Total Environ.* **2016**, *572*, 216–231. [CrossRef]
12. Carpenter, E.J.; Smith, K.L. Plastics on the Sargasso Sea surface. *Science* **1972**, *175*, 1240–1241. [CrossRef]
13. Thompson, R.C.; Courtene-Jones, W.; Boucher, J.; Pahl, S.; Raubenheimer, K.; Koelmans, A.A. Twenty years of microplastic pollution research—What have we learned? *Science* **2024**, *386*, 6720. [CrossRef]

14. Earth Action. Leakage of Microplastics into Oceans and Land. EA's Global Assessment & Benchmark of Literature–2023 Update. Available online: https://www.e-a.earth/wp-content/uploads/2024/05/EA_2024_Update_Primary_Microplastics.pdf (accessed on 3 June 2025).
15. Napper, I.E.; Davies, B.F.R.; Clifford, H.; Elvin, S.; Koldewey, H.J.; Mayewski, P.A.; Miner, K.R.; Potocki, M.; Elmore, A.C.; Gajurel, A.P.; et al. Reaching new heights in plastic pollution—Preliminary findings of microplastics on Mount Everest. *One Earth* **2020**, *3*, 621–630. [CrossRef]
16. Rota, E.; Bergami, E.; Corsi, I.; Bargagli, R. Macro- and microplastics in the Antarctic environment: Ongoing assessment and perspectives. *Environments* **2022**, *9*, 93. [CrossRef]
17. Vaid, M.; Sarma, K.; Gupta, A. Exploiting the potential of terrestrial and freshwater organisms for biomonitoring of microplastics: Challenges and prospects. *TrAC Trends Anal. Chem.* **2024**, *178*, 117854. [CrossRef]
18. Liu, Y.; Nie, Z.; Meng, Y.; Liu, G.; Chen, Y.; Chai, G. Influence of meteorological conditions on atmospheric microplastic transport and deposition. *Environ. Res.* **2025**, *265*, 120460. [CrossRef]
19. Zhang, Y.; Kang, S.; Allen, S.; Allen, D.; Gao, T.; Sillanpää, M. Atmospheric microplastics: A review on current status and perspectives. *Earth Sci. Rev.* **2020**, *203*, 103118. [CrossRef]
20. O'Brien, S.; Rauert, C.; Ribeiro, F.; Okoffo, E.D.; Burrows, S.D.; O'Brien, J.W.; Wang, X.; Wright, S.L.; Thomas, K.V. There's something in the air: A review of sources, prevalence and behaviour of microplastics in the atmosphere. *Sci. Total Environ.* **2023**, *874*, 162193. [CrossRef] [PubMed]
21. Ferrero, L.; Scibetta, L.; Markuszewski, P.; Mazurkiewicz, M.; Drozdowska, V.; Przemyslaw, M.; Jutrzenka-Trzebiatowska, P.; Zaleska-Medynska, A.; Andò, S.; Saliu, F.; et al. Airborne and marine microplastics from an oceanographic survey at the Baltic Sea: An emerging role of air-sea interaction? *Sci. Total Environ.* **2022**, *824*, 153709. [CrossRef] [PubMed]
22. Allen, S.; Allen, D.; Moss, K.; Le Roux, G.; Phoenix, V.R.; Sonke, J.E. Examination of the ocean as a source for atmospheric microplastic. *PLoS ONE* **2020**, *15*, e0232746. [CrossRef]
23. Horton, A.A.; Walton, A.; Spurgeon, D.J.; Lahive, E.; Svendsen, C. Microplastics in freshwater and terrestrial environments: Evaluating the current understanding to identify the knowledge gaps and future research priorities. *Sci. Total Environ.* **2017**, *586*, 127–141. [CrossRef]
24. Romarate, R.A.; Ancla, S.M.B.; Patilan, D.M.M.; Inocente, S.A.T.; Pacilan, C.J.M.; Sinco, A.L.; Guihawan, J.Q.; Capangpangan, R.Y.; Lubguban, A.A.; Bacosa, H.P. Breathing plastics in Metro Manila, Philippines: Presence of suspended microplastics in ambient air. *Environ. Sci. Pollut. Res.* **2023**, *30*, 53662–53673. [CrossRef]
25. González-Pleiter, M.; Edo, C.; Aguilera, A.; Viúdez-Moreiras, D.; Pulido-Reyes, G.; González-Toril, E.; Osuna, S.; de Diego-Castilla, G.; Leganés, F.; Fernández-Piñas, F.; et al. Occurrence and transport of microplastics sampled within and above the planetary boundary layer. *Sci. Total Environ.* **2021**, *761*, 143213. [CrossRef]
26. Shao, L.; Li, Y.; Jones, T.; Santosh, M.; Liu, P.; Zhang, M.; Xu, L.; Li, W.; Lu, J.; Yang, C.-X.; et al. Airborne microplastics: A review of current perspectives and environmental implications. *J. Clean. Prod.* **2022**, *347*, 131048. [CrossRef]
27. Jiao, M.; Wang, Y.; Yang, F.; Zhao, Z.; Wei, Y.; Li, R.; Wang, Y. Dynamic fluctuations in plant leaf interception of airborne microplastics. *Sci. Total Environ.* **2024**, *906*, 167877. [CrossRef]
28. Bargagli, R.; Mikhailova, I. Accumulation of inorganic contaminants. In *Monitoring with Lichens—Monitoring Lichens*; Nimis, P.L., Scheidegger, C., Wolseley, P.A., Eds.; NATO Science Series; Springer: Dordrecht, Germany, 2002; Volume 7, pp. 65–84. [CrossRef]
29. Roblin, B.; Aherne, J. Moss as a biomonitor for the atmospheric deposition of anthropogenic microfibers. *Sci. Total Environ.* **2020**, *715*, 136973. [CrossRef]
30. International Cooperation Programme (ICP) Vegetation. Heavy Metals, Nitrogen and POPs in European Mosses: 2020 Survey, Monitoring Manual. Available online: http://icpvegetation.ceh.ac.uk (accessed on 4 July 2025).
31. Loppi, S.; Roblin, B.; Paoli, L.; Aherne, J. Accumulation of airborne microplastics in lichens from a landfill dumping site (Italy). *Sci. Rep.* **2021**, *11*, 4564. [CrossRef]
32. Jafarova, M.; Contardo, T.; Aherne, J.; Loppi, S. Lichen biomonitoring of airborne microplastics in Milan (N Italy). *Biology* **2022**, *11*, 1815. [CrossRef] [PubMed]
33. Jafarova, M.; Grifoni, L.; Aherne, J.; Loppi, S. Comparison of lichens and mosses as biomonitors of airborne microplastics. *Atmosphere* **2023**, *14*, 1007. [CrossRef]
34. Capozzi, F.; Sorrentino, M.C.; Granata, A.; Vergara, A.; Alberico, M.; Rossi, M.; Spagnuolo, V.; Giordano, S. Optimizing moss and lichen transplants as biomonitors of airborne anthropogenic microfibers. *Biology* **2023**, *12*, 1278. [CrossRef]
35. Taurozzi, D.; Gallitelli, L.; Cesarini, G.; Romano, S.; Orsini, M.; Scalici, M. Passive biomonitoring of airborne microplastics using lichens: A comparison between urban, natural and protected environments. *Environ. Int.* **2024**, *187*, 108707. [CrossRef]
36. Çobanoğlu, G.; Özen, E. Detection of atmospheric microplastics accumulated in *Xanthoria parietina*: A lichen biomonitoring study on the Asian side of Istanbul. *Int. J. Environ. Res.* **2024**, *18*, 65. [CrossRef]

37. Stringa Basile, C.; Jafarova, M.; Ahene, J.; Wyonch, J.; Ruffini Castiglione, M.; Spanò, C.; Fačkovcová, Z.; Loppi, S.; Paoli, L. Greenspaces can reduce the level of airborne microplastic contamination in urban environments: Evidence from a lichen biomonitoring study. *Environ. Res.* **2025**, *285*, 122334. [CrossRef]
38. Bargagli, R.; Monaci, F.; Borghini, F.; Bravi, F.; Agnorelli, C. Mosses and lichens as biomonitors of trace elements. A comparison study on *Hypnum cupressiforme* and *Parmelia caperata* in a former mining district in Italy. *Environ. Pollut.* **2002**, *116*, 279–287. [CrossRef]
39. Adamo, P.; Crisafulli, P.; Giordano, S.; Minganti, V.; Modenesi, P.; Monaci, F.; Pittao, E.; Tretiach, M.; Bargagli, R. Lichen and moss bags as monitoring devices in urban areas. Part II: Trace element content in living and dead biomonitors and comparison with synthetic materials. *Environ. Pollut.* **2007**, *146*, 392–399. [CrossRef] [PubMed]
40. Capozzi, F.; Sorrentino, M.C.; Di Palma, A.; Mele, F.; Arena, C.; Adamo, P.; Spagnuolo, V.; Giordano, S. Implication of vitality, seasonality and specific leaf area on PAH uptake in moss and lichen transplanted in bags. *Ecol. Ind.* **2020**, *108*, 105727. [CrossRef]
41. Contardo, T.; Vannini, A.; Sharma, K.; Giordani, P.; Loppi, S. Disentangling sources of trace element air pollution in complex urban areas by lichen biomonitoring. A case study in Milan (Italy). *Chemosphere* **2020**, *256*, 127155. [CrossRef]
42. Wittig, R. General aspects of biomonitoring heavy metals by plants. In *Plants as Biomonitors: Indicators for Heavy Metals in the Terrestrial Environment*; Markert, B., Ed.; VCH Publishers, Inc.: Weinheim, Germany, 1993; pp. 3–27.
43. Nistal, A.; Coviella, C.; Gomez, J. Methodological predicament: Distinguishing between the accumulated and deposited microplastics in lichen thalli. *Environ. Monit. Contam. Res.* **2025**, *5*, 76–82. [CrossRef]
44. Sun, H.; Lei, C.; Xu, J.; Li, R. Foliar uptake and leaf-to-root translocation of nanoplastics with different coating charge in maize plants. *J. Hazard. Mater.* **2021**, *416*, 125854. [CrossRef]
45. Leonard, J.; Borthakur, A.; Koutnik, V.S.; Brar, J.; Glasman, J.; Cowger, W.; Dittrich, T.M.; Mohanty, S.K. Challenges of using leaves as a biomonitoring system to assess airborne microplastic deposition on urban tree canopies. *Atmos. Pollut. Res.* **2023**, *14*, 101651. [CrossRef]
46. Bhattacharjee, S.; Ghosh, P.K.; Podder, A.; Mandal, B.; Mukherjee, A. Urban trees as natural interceptors for aerial microplastics: Mechanisms, influencing factors, and selection of tree types for mitigation. *Sci. Total Environ.* **2025**, *992*, 179950. [CrossRef]
47. Wenzel, M.; Schoettl, J.; Pruin, L.; Fischer, B.; Wolf, C.; Kube, C.; Renner, G.; Schram, J.; Schmidt, T.C.; Tuerk, J. Determination of atmospherically deposited microplastics in moss: Method development and performance evaluation. *Green Anal. Chem.* **2023**, *7*, 100078. [CrossRef]
48. Hagelskjær, O.; Le Roux, G.; Liu, R.; Dubreuil, B.; Behra, P.; Sonke, J.E. The recovery of aerosol-sized microplastics in highly refractory vegetal matrices for identification by automated Raman microspectroscopy. *Chemopshere* **2023**, *328*, 138487. [CrossRef] [PubMed]
49. Falakdin, P.; Lopez-Rosales, A.; Andrade, J.; Terzaghi, E.; Di Guardo, A.; Muniategui-Lorenzo, S. Comparison of microplastic type, size, and composition in atmospheric and foliage samples in an urban scenario. *Environ. Pollut.* **2024**, *349*, 123911. [CrossRef] [PubMed]
50. Awada, A.; Potter, M.; Aherne, J.; Lavoie-Bernstein, S.; Diamond, M.L.; Helm, P.A.; Jantunen, L.; Welsh, B.; Mutus, B.; Rondeau-Gagné, S. Facile detection of microplastics from a variety of environmental samples with conjugated polymer nanoparticles. *Environ. Sci. Adv.* **2025**, *4*, 270–278. [CrossRef]