



Greenspaces can reduce the level of airborne microplastic contamination in urban environments: Evidence from a lichen biomonitoring study

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ABSTRACT

Microplastics (MPs) have been found across a variety of environments, nonetheless few studies have evaluated atmospheric MPs. In this study, airborne MP contamination was investigated using transplants of the fruticose lichen *Evernia prunastri* in urban sites. Lichen transplants were exposed for seven weeks (April to June 2023) in parking lots ($n = 9$) and urban parks ($n = 9$) in the city of Pisa (Tuscany, Central Italy); in parallel, native samples from rural areas ($n = 4$) were also investigated. The overall aim was the characterization of MPs in terms of number, shape, size and polymer composition under different environmental conditions. Further, the positive role of green urban areas in buffering atmospheric MPs was assessed. We found MPs, including fragments, fibres and tyre wear particles, across all sites. The average number of MPs (per gram dry weight of lichen) significantly increased from rural areas (2 ± 0.4 MP/g dw) to urban parks (7 ± 1.1 MP/g dw) and parking lots (16 ± 4.1 MP/g dw). Average daily MP deposition rates across sites in urban areas was in the range of 12–143 MP/m²/d, suggesting that inhabitants are exposed to varying levels of airborne MPs. There was no difference in the length of the fibres between parking lots and urban parks; however, longer fragments and shorter tyre wear particles were found in parking lots. Polyethylene terephthalate was the dominant polymer detected across sites. The transplants maintained their overall vitality after the exposure (assessed by chlorophyll *a* fluorescence emission analysis), similar to native samples from rural areas, suggesting that the exposure had a negligible effect on lichen metabolism. Overall, our results suggest that lichen transplants are effective biomonitors of atmospheric MPs in urban areas, and that the presence of greenspaces (parks) in urban environments can significantly buffer the level of atmospheric MPs.

1. Introduction

Microplastics (MPs) and their associated chemicals are an emerging global concern due to their potential impacts on human health and their widespread presence, even in remote environments. They are defined as plastic particles smaller than 5 mm (0.1 μm –5 mm; Van Emmerik and Schwarz, 2020), that can enter the environment at every stage of the plastic life cycle (production, use, and disposal) with the potential to infiltrate the food web. Humans can be exposed to MPs through inhalation, ingestion, and dermal contact, with potential consequences such

as lung inflammation, immune and metabolic issues, DNA damage, and oxidative stress, causing issues in the respiratory, digestive, immune, reproductive, and nervous systems (Prata, 2018). As a result of the increase in plastic production and the consequent increase in plastic waste, MPs have become ubiquitous in the environment and have been found in vertebrates, invertebrates, and plants (Chen et al., 2020; Pirsahab et al., 2020).

The atmosphere is a key compartment through which suspended material can be transported over long distances, and it is now evident that even remote areas are contaminated by atmospheric MPs (Allen

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et al., 2019; Bergmann et al., 2019). The behaviour of MPs, their transport, atmospheric concentration and deposition are influenced by various factors, including meteorological conditions, e.g., precipitation, temperature, humidity, wind speed, and wind direction (Allen et al., 2019; Dris et al., 2016; Liu et al. 2019a, 2019b). Furthermore, high levels of atmospheric MPs are closely associated with population density and human activity (Dris et al., 2017; Wright et al., 2020). There is evidence that traffic and industrial emissions, textiles, waste disposal, and construction are among the main sources of MPs and other hazardous pollutants in urban areas (e.g., see Jahandari, 2023, and references therein). Polypropylene (PP), polyethylene (PE), polystyrene (PS) and polyethylene terephthalate (PET) are reportedly the dominant polymers of MPs in atmospheric deposition (Chen et al., 2020). However, there are still comparatively few observations of atmospheric MPs, as traditional sampling networks require specialized infrastructure and are generally labour-intensive, leading to spatially limited observations.

Measurement of air pollutants via plants, i.e., biomonitoring, is an economic, convenient, and reliable method. Lichens depend on the atmosphere for their metabolism; hence they are particularly effective biomonitors of airborne pollutants due to their widespread presence and ability to thrive in diverse environments. They are perennial slow-growing organisms that do not shed thallus parts as readily as vascular plants. They lack a waxy cuticle and stomata to regulate the absorption of contaminants, which can occur across the entire surface of the thallus (Hale, 1983). Hence, they accumulate pollutants from the atmospheric environment to levels far beyond their physiological needs. It is well established that lichens are effective biomonitors of trace contaminant deposition patterns both in space and time, as in the case of potentially toxic elements (Bačkor and Loppi, 2009). Recently, it has been shown that the deposition of airborne MPs can be detected using moss (Roblin and Aherne, 2020; Capozzi et al., 2023; Jafarova et al., 2023; Bertrim and Aherne, 2023) and lichens (Loppi et al., 2021; Jafarova et al., 2022, 2023; Çobanoğlu and Özen, 2024; Taurozzi et al., 2024).

The aim of the present study was the characterization of MPs trapped by lichen thalli (*Evernia prunastri*) along a gradient of increasing pollution, ranging from rural areas to urban environments, namely urban parks and parking lots in Pisa, Tuscany. We wanted to assess the appropriateness of the lichen transplantation method, applied to different experimental situations along a gradient in anthropogenic activity following recent studies (Jafarova et al., 2022; Taurozzi et al., 2024), but with a focus on the influence of vegetation on atmospheric MPs. There is growing evidence that the presence of greenspaces in urban environments can be associated with a reduction of particulate matter and other forms of atmospheric pollution (Ai et al., 2023) due to the capacity of vegetation to intercept airborne pollutants. Similarly for atmospheric MPs, higher deposition has been reported along roads and associated areas, compared with greenspaces within the same urban environment (Cakaj et al., 2023). Here, airborne MP contamination was assessed in terms of number, shape, size and polymer composition. We predicted that MP deposition would differ between urban parks and parking lots within the same urban area (Pisa, Tuscany, Central Italy) and that exposed (transplanted) lichens would be significantly enriched in MPs with respect to native lichens from rural sites of the same region, in line with previous studies (Jafarova et al., 2022, 2023; Taurozzi et al., 2024).

2. Materials and methods

2.1. The model species

Evernia prunastri (L.) Ach. was selected because of its wide distribution in rural areas of Tuscany, as well as its presence in the urban area of Pisa. Nonetheless, the urban area did not host enough thalli for in situ biomonitoring, therefore we resorted to lichen transplants. *Evernia prunastri* is one of the most commonly used lichen species in

biomonitoring and ecophysiological studies (Paoli et al., 2010; Munzi et al., 2012). In addition, the results of previous studies highlight its suitability as a biomonitor for MP deposition (Jafarova et al., 2022, 2023). The thallus is sub-foliose to fruticose, attached with a basal holdfast, erect and tufted to sub-pendent, dorsiventral, green to yellowish green above, white beneath, the margins can be soreadate, and the photobiont is a green alga (Nimis, 2024). Its fruticose habitus allows for easy collection and removal of extraneous material such as adhering bark, mosses, other lichen species and soil particles, ensuring better sample homogeneity and higher data accuracy (Wolterbeek and Bode, 1995). The thalli for the transplant were harvested together with their substrate (branches of *Prunus* or *Quercus*) from a control site in Tuscany (43.182090, 11.367308, Murlo, Siena), used in previous studies as a source area for *E. prunastri*, characterized by negligible levels of pollution and the presence of rich lichen communities (e.g., Loppi and Paoli, 2015; Paoli et al., 2014).

2.2. Study sites and experimental design

Atmospheric MP deposition was investigated by means of a mixed design encompassing native and transplanted samples (Supplementary Material – Fig. S1). The transplant experiment accounted for two conditions: parking lots and urban parks in the town of Pisa. Initially, considering the limited number of potentially suitable sites (plots) and our knowledge of the area, a preliminary assessment was conducted to evaluate all possible plots. This preselection process identified approximately fifteen potentially independent plots for each type. Subsequently, based on field verification (availability of suitable trees to support the transplants, the safe feasibility of exposure, and the likelihood to retrieve the transplanted material), some plots (approximately 20 %) were excluded. Among the remaining ones, 9 parking lots and 9 urban parks were randomly selected, independent of each other (Fig. 1). At each sampling plot, lichens were transplanted together with their substrate (three twigs per plot) on three different trees, at ca. 2 m above ground, ensuring similar exposure conditions. Each selected twig generally carried three or more thalli of *E. prunastri* with an average length of 4–5 cm. Overall, 54 twigs were selected for transplant (27 for urban parks and 27 for parking lots, each one representing an analytical sample), another three twigs were used as controls. Similarly, for the characterization of MPs in rural areas of Southern Tuscany, three nature reserves hosting rich lichen communities typical in pristine environments were selected for native (passive) lichen collection (Mt Labbro, Mt Penna, Scansano; Fig. 1), and a fourth remote site (Murlo; Fig. 1) was used as the control area. Lichen transplants were taken from the control area and exposed for 7 weeks (from April 21, 2023 to June 9, 2023) in the urban areas (Supplementary Material – Table S1a–b). Further, native samples were harvested from rural areas during June, corresponding to the end of the transplant period. In all cases (transplants and native samples), once retrieved, the thalli were air dried (residual water <10 %) and stored in aluminium foil at –20 °C for subsequent measurements. During the exposure period, rainfall in Pisa was 142 mm, while in the rural areas it ranged from 150 to 217 mm (during the same period), showing on the whole a comparable pattern.

2.3. Microplastic analysis

Microplastic analysis was carried out at the School of the Environment, Trent University (Ontario, Canada). Lichen samples were cleaned (manually) under a stereomicroscope of extraneous material, the thallus was separated from the supporting branch, wearing nitrile gloves and cotton clothing. Each sample (independent replicate) consisted of 1 g of clean lichen material placed in a glass beaker. The organic component of the samples was digested using a wet peroxide oxidation method to extract accumulated MPs (Masura et al., 2015). The digestion procedure was performed by adding 40 mL of 30 % H₂O₂ to each beaker; the mixture was left in an oven at 50 °C for 24 h. Subsequently, 40 mL of



Fig. 1. Map of the study sites in Tuscany (Central Italy): nine parking lots and nine urban parks selected for the transplant experiment in the town of Pisa; three remote sites in Southern Tuscany (Mt Labbro, Mt Penna, Scansano) for the assessment of MPs in native *Evernia prunastri*, plus a control remote area (Murlo) used as a source of lichen material for the transplant experiment. See text for further details.

0.05 M Fe(II) solution per gram of lichen was added. To stimulate the reaction, the contents of each beaker was heated to 55 °C, an additional aliquot of 20 mL of H₂O₂ was added when the reaction slowed down, especially if organic matter was still visible. This latter procedure was generally repeated at least three times. Once organic matter was digested, the solution was vacuum filtered onto filter paper disks (Fisherbrand Glass Fiber G6 filter paper, diameter 7.0 cm), then placed into petri dishes until microscope analysis. Overall, for each investigated plot, three samples were processed and analysed individually (four samples could not be retrieved in the urban areas).

Filter papers were examined for MPs under a stereomicroscope with a digital camera (AmScope with MU1000 camera). Microplastic identification was performed following five visual identification criteria (Windsor et al., 2019; Hidalgo-Ruz et al., 2012): I) it had an unnatural colour compared to other particles/debris; II) it appeared homogeneous in terms of material and texture, without visible cellular structures or protrusions, and had a constant width throughout its length; III) it remained intact and not fragile when compressed or prodded with tweezers; IV) it had a shiny or glossy appearance; and V) it did not appear frayed and bore no resemblance to natural fibres. Particles that met at least two of the criteria were considered as anthropogenic and photographed (Loppi et al., 2021). In cases of uncertainty, a hot needle test was conducted: if a particle melted in the presence of a hot needle, it was counted as a MP. Microplastic particles were grouped into three categories: fibres, fragments, and tyre wear particles (TWPs). Tyre particles are more difficult to identify since they do not react to a hot needle; therefore, they were classified based on specific criteria: I) dark colour (black); II) elongated or cylindrical shape; III) rough surface texture; and IV) rubbery flexibility when manipulated (Bertrim and Aherne, 2023). A visual inspection under UV light (540 nm) was also conducted at the end of the process to help identify transparent particles (NightSea Fluorescence Adapter). Microplastics were photographed and then measured using the open-source Image processing software ImageJ. To confirm the synthetic nature of the particles, approximately 25 % from each site were selected and analysed by Fourier Transform Infrared (FTIR) spectroscopy (LUMOS II, Bruker). The FTIR determines the composition of a particle (its molecular structure) by examining the sample using an infrared wavelength range (between 300 nm and 3000

nm). The resulting spectrum was cross-referenced against polymer libraries to define particle composition (Cowger et al., 2021). In our study, all samples were analysed by FTIR in attenuated total reflectance (ATR) mode, and the spectra were uploaded to Open Specy (URL: openanalysis.org/openspecy), an open-source digital library to identify spectra (Cowger et al., 2021).

2.4. Vitality of the samples

To evaluate whether the lichen samples degraded during their exposure and whether this potentially influenced the accumulation of MP deposition, we evaluated their vitality: *i*) by visual assessment: thalli were attributed to the categories of ‘damaged’ (1 – presence of diffused discoloration, necrosis), ‘variable’ (2 – thalli with occasional discoloration or necrosis), or ‘healthy’ (3 – thalli with no visual injury); and *ii*) through the analysis of chlorophyll (Chl) *a* fluorescence emission, represented here by the potential quantum yield of primary photochemistry (F_v/F_m). Thalli were kept hydrated (sprayed with mineral water) and dark adapted for at least 10 min (covered with a black velvet cloth) prior to measurement. Then, using a Plant Efficiency Analyzer fluorimeter (Handy PEA, Hansatech Ltd, Norfolk, UK), lichen samples were exposed for 1 s to a saturating light pulse (up to 3000 μmol of photosynthetically active photons m⁻² s⁻¹) and fluorescence emission was recorded. Five measurements were randomly carried out for each sample.

2.5. Quality control, data interpretation and statistics

To check for any field or laboratory contamination, open-air blanks and analytical process blanks were run during the transplants and analysis. Open-air blanks were used to determine the amount of potential contamination during the sample handling in the field: in three parking lots and three urban parks, 1 g of lichen material was exposed to the air for as long as it took to perform the transplants and then wrapped in aluminium foil until analysis. Strict quality-control procedures were followed to ensure that contamination was minimized during laboratory analyses. All solutions were vacuum filtered prior to use. Further, all glassware was covered with aluminium foil to prevent airborne contamination and was rinsed with filtered deionised water before use.

Laboratory process blanks and spikes (addition of a known number of pink polyethylene beads ($n = 10$; 212–250 μm) to determine recovery rates) were performed during the digestion and filtration of the samples. However, no blank correction was applied to the sample data as most blanks contained no detectable MPs. Further, the average spike recovery was 85 %, in line with previous findings (Welsh et al., 2024; Walker and Aherne, 2024), and reflects the influence of processing steps on microplastic recovery efficiency (Dimante-Deimantovica et al., 2022).

The number of MPs accumulated on the lichen thallus was converted into MPs deposition rates. The mass/surface ratio for *E. prunastri* of $\sim 160 \text{ g/m}^2$ reported in Jafarova et al. (2022) was used as a reference. Based on the exposure period of 7 weeks, the deposition of MPs in lichens was converted into estimates of average daily MPs deposition, according to the formula:

$$\text{MP deposition (MPs/m}^2\text{/d)} = \text{MP concentration (MPs/g dw lichen)} \times 160 \text{ (g/m}^2\text{)} / 49 \text{ (days)}$$

An exposed-to-control (EC) ratio originally developed for trace element deposition in *E. prunastri* (Frati et al., 2005) was adapted to MPs and calculated in the urban area of Pisa (for parks and parking lots), as the ratio between the number of MPs after the exposure to that of the samples prior to the exposure, separately for fragments, fibres and TWPs. For the latter, the average value of rural sites was used as reference.

Data normality was checked with the Shapiro–Wilk test. Potential differences between site groups were assessed by one-way ANOVA, followed by LSD test for post hoc comparisons (significance level at $p < 0.05$). Differences for EC ratios between parks and parking lots were assessed by the Mann-Whitney *U* test (significance level at $p < 0.05$). The analyses were carried out using the open-source software R (R Core Team, 2024).

3. Results

Microplastics were detected in *E. prunastri* across all urban sites (urban parks and parking lots) and rural sites, including the site of origin of the transplants – control (Table 1, Supplementary Material – Table S2a–c). The average number of MPs (per gram dry weight of lichen) increased from rural areas, including the control (2 MP/g dw), to urban parks (7 MP/g dw) and parking lots (16 MP/g dw).

A comparison between Exposed-to-Control ratios in urban parks and parking lots revealed a large accumulation for fragments, fibres, and TWPs in both urban parks and parking lots, with values more than doubled in parking lots compared with urban parks (Table 2).

Overall, 594 MPs were found in the urban area of Pisa (across 50 lichen transplants retrieved from the 18 plots); among them, 31 % were classified as fragments, 30 % as fibres and 39 % as TWPs, with a similar percentage classification in urban parks (total of 169 MPs) and parking lots (425 MPs). Average daily MP deposition was estimated in the range 12–143 MP/m²/d, with peaks in parking lots (up to 143 MP/m²/d), compared with urban parks (44 MP/m²/d). Rural areas (as well as the

Table 1

Characteristics of microplastics (MPs) detected in *Evernia prunastri* from urban sites (parks and parking lots, Pisa, Tuscany) and rural sites (Tuscany). Number per gram of dry lichen (n/g dw) and length (μm) of MPs (average \pm standard deviation) – fragments, fibres and tyre wear particles (TWPs). Chlorophyll *a* fluorescence emission (F_V/F_M) used as an indicator of the vitality of the samples. n.d. = not detected. Data followed by a different letter are statistically different (LSD test, $p < 0.05$).

	Control (autochthonous)	Rural sites (autochthonous)	Urban parks (transplants)	Parking lots (transplants)
MPs (n/g dw)	2.0 \pm 0.6 c	2.2 \pm 0.4 c	7.0 \pm 1.1 b	16.3 \pm 4.1 a
Fragments (n/g dw)	0.33 \pm 0.33 d	0.89 \pm 0.26 c	2.16 \pm 0.54 b	5.11 \pm 1.47 a
Fibres (n/g dw)	1.67 \pm 0.33 b	0.78 \pm 0.22 c	1 \pm 0.21 bc	4.66 \pm 0.77 a
TWPs (n/g dw)	n.d.	0.56 \pm 0.34 c	2.63 \pm 0.71 b	6.57 \pm 2.63 a
Fragments (μm)	199 \pm 11 a	87 \pm 24 c	151 \pm 18 b	219 \pm 24 a
Fibres (μm)	847 \pm 191 b	1039 \pm 414 ab	1288 \pm 137 a	1291 \pm 87 a
TWPs (μm)	n.d.	82 \pm 15 c	119 \pm 8 a	101 \pm 7 b
Deposition (MPs/m ² /d)	1.1–3.3 d	4.3–9.8 c	12.1–43.8 b	20.7–142.6 a
F_V/F_M	0.688 \pm 0.067	0.664 \pm 0.062	0.641 \pm 0.059	0.607 \pm 0.053

Table 2

Exposed-to-control ratios in *Evernia prunastri* for urban parks and parking lots calculated as the ratio between the number of MPs after the exposure to that of the samples prior to exposure for fragments, fibres, tyre wear particles (TWPs) and total MPs. Data followed by a different letter are statistically different (Mann-Whitney *U* test, $p < 0.05$).

	Exposed to Control (EC) ratios	
	Urban parks	Parking lots
Fragments	6.11 \pm 3.71 b	15.26 \pm 10.67 a
Fibres	1.28 \pm 0.66 b	2.73 \pm 1.62 a
TWPs	4.34 \pm 3.72	11.53 \pm 13.84
Total MPs	3.30 \pm 1.59 b	8.04 \pm 6.37 a

control site) were characterized by low deposition (1–10 MP/m²/d) and with the following shape distribution: fragments (40 %), fibres (35 %), and TWPs (25 %) based on just 20 MPs detected. Polyethylene terephthalate (PET) was the polymer most frequently detected in the samples irrespective of the site (Fig. 2; Supplementary Material – Fig. S2). No differences emerged when comparing the length of the fibres between parking lots and urban parks, while longer fragments and shorter TWPs were found in parking lots compared with urban parks (Table 1, Supplementary Material – Fig. S3). Overall, the dominant length proportion for MPs was shorter than 400 μm , particularly for fragments and TWPs (Fig. 3).

The transplants maintained their overall vitality after the exposure in the urban environment (similar to their condition pre-exposure and to that of native samples from rural areas), as witnessed firstly by the visual assessment (category 3 – thalli with no visual injury) and secondly by the analysis of chlorophyll *a* fluorescence emission (F_V/F_M , Table 1), suggesting that under the experimental conditions the status of the samples did not influence MP accumulation.

4. Discussion

4.1. Number of MPs and deposition rates

The results highlight significant differences in MP accumulation by *E. prunastri* exposed across urban areas in Pisa; on the whole, the number of MPs in parking lots was 2.3 times higher than in urban parks (greenspaces), suggesting that despite the minimal distance between parking lots and public parks, the latter can be considered as less impacted by MP contamination. This further suggests that green urban areas may play a positive role in buffering atmospheric MPs. Noteworthy, our estimated daily MP deposition in the urban area of Pisa (12–143 MP/m²/d) is comparable to a similar study in Milan (that used *E. prunastri*), which was in the range 43–119 MP/m²/d (Jafarova et al., 2022). Consistent with Jafarova et al. (2022), our transplants were exposed within 2 m from the ground, which is the height relevant for human respiration. In this context, the estimated daily MP deposition highlights that the residents of Pisa are regularly exposed to varying levels of airborne MPs; however, we do not know what levels pose a

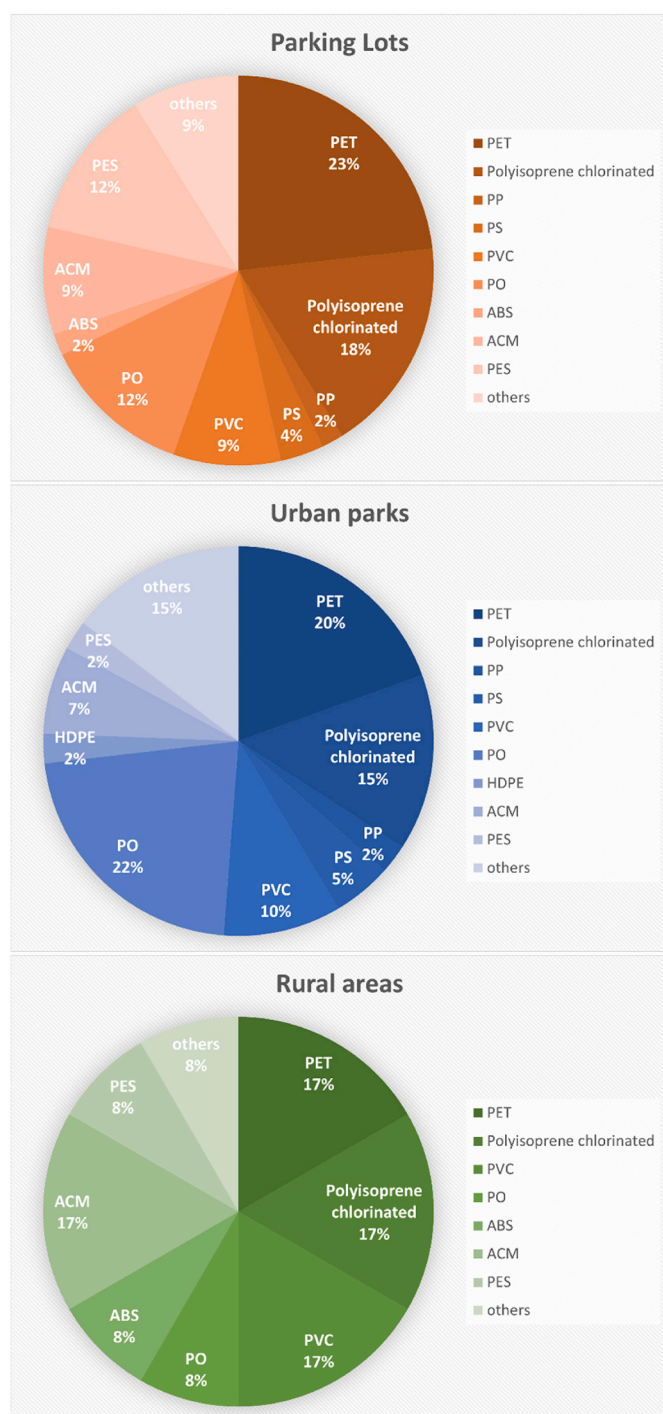


Fig. 2. Polymer composition under the experimental conditions. Notes: PET – polyethylene terephthalate; PP – polypropylene; PS – polystyrene; PVC – polyvinyl chloride; PO – polyolefins; ABS – acrylonitrile butadiene styrene; ACM – acrylic rubber; PES – polyester; HDPE – high density polyethylene.

direct health risk to individuals in urban environments. As a comparison, Welsh et al. (2022) reported an average MP deposition rate of 7 MP/m²/d in rural areas of Canada, using bulk precipitation collectors, and Roblin et al. (2020) reported 12 fibres/m²/d in rural areas of Ireland using similar collectors. Our observations from rural areas, including the control site (2 MP/g dw), are consistent with those found in the lichen *Flavoparmelia caperata* (3–9 MP/g dw) from other rural sites in Tuscany (Loppi et al., 2021). On the other hand, they are lower than those reported for autochthonous *E. prunastri* from other rural sites in Central

Italy (8–12 MP/g dw; Jafarova et al., 2023) and for transplants of *E. prunastri* exposed for 3 months in a rural area of Northern Italy (12–24 MP/g dw; Jafarova et al., 2022). Noteworthy, MPs produced by road traffic (such as TWPs – tyre wear particles and BWPs – brake wear particles), can be efficiently transported via the atmosphere and contaminate remote regions (Evangelidou et al., 2020). This highlights the pervasive nature of MP pollution, as airborne particles are potentially capable of traveling long distances through atmospheric transport mechanisms, such as wind and weather patterns. Nonetheless, the abundance of atmospheric MPs is directly related to the intensity of local anthropogenic activity (Loppi et al., 2021; Jafarova et al., 2022, 2023; Taurozzi et al., 2024). The presence of MPs in remote areas indicates that even ecosystems seemingly untouched by human activity are not immune to contamination. This underscores the global scope of MP contamination and raises concern about its potential impact on remote environments.

4.2. Shape, size and polymer composition

Our dataset revealed a partition of MPs between fragments (31 %), fibres (30 %) and TWPs (39 %). Using lichens as biomonitors, Jafarova et al. (2022) found that 94.6 % of MPs in Milan were classified as fibres, while just 5.4 % as fragments. Loppi et al. (2021) reported that the percentage of microfibrils was lower in lichens close to a landfill dumping site (41 %) and much higher at remote sites (73 %). The morphology of MPs is influenced by their original shape, the processes of degradation and erosion they undergo, as well as by their time spent in the environment (Zhang et al., 2020). In general, fibres appear more subject to long-distance transport due to differences in weight and density compared with other forms, as heavier fragments tend to settle more easily (Ding et al., 2021; Welsh et al., 2022). Furthermore, fibres have a higher surface-to-volume ratio, which facilitates their transport and reduces their sedimentation rate (Preston et al., 2020). However, the data available so far, suggest wide variability in transport and fate of atmospheric MPs; studies conducted in several urban areas, such as Dongguan, Shanghai, Yantai, Paris, Bahia Blanca, London, and Krakow, reported fibres as the dominant form of atmospheric MPs (Dris et al., 2016; Cai et al., 2017; Zhou et al., 2017; Liu et al., 2019c; Wright et al., 2020; Jarosz et al., 2022; Villafaña et al., 2023). Similarly, fibres were reported as the dominant form in remote areas of Italy, Switzerland, and Canada (Ambrosini et al., 2019; Taurozzi et al., 2024; Negrete Velasco et al., 2020; Welsh et al., 2022), while in other studies, fragments dominated both urban areas, such as Hamburg (Klein and Fischer, 2019), and remote areas, such as the French Pyrenees (Allen et al., 2019).

In our dataset, no significant differences emerged between the average length of the fibres in parking lots (1291 µm) and urban parks (1288 µm). As a comparison, using transplants of *E. prunastri* in Milan, Jafarova et al. (2022) found fibres in the range 128–1839 µm in urban parks and 111–3980 µm in the city centre (away from urban parks).

The size of atmospheric MPs influences their interaction with organisms and their environmental fate; the smaller the size, the greater the degree of dispersion (Roblin et al., 2020). In our study, based on the few MPs observed in remote sites, it is possible to argue that shorter fibres were found at the control site and the shortest fragments in the other remote areas. In fact, according to several studies, when MPs are present due to long-range atmospheric transport in remote or rural sites distant from urban or industrial areas, they chiefly consist of smaller particles (Allen et al., 2019; Brahney et al., 2020). However, compared to MPs from aquatic environments and sediments, the predominant size of atmospheric MPs is generally smaller (Zhang et al., 2020).

Nevertheless, given that standardized protocols for sampling atmospheric MPs have not yet been fully established, the size ranges reported in literature are heavily influenced by the methodology adopted (Prata et al., 2020; Rochman et al., 2019).

Polyester, or polyethylene terephthalate (PET), was the most

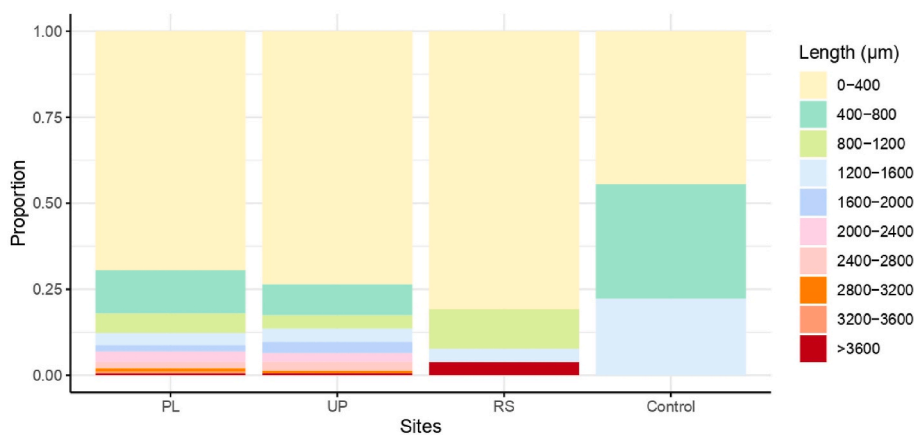


Fig. 3. Length proportion for MPs in *Evernia prunastri* under the experimental conditions. PL parking lots; UP = urban parks; RS = remote sites; Control = source site for the transplant experiment.

frequently detected polymer in our samples, reflecting its status as the most prevalent textile fibre globally (Textile Exchange, 2023). Similarly, Loppi et al. (2021) found PET at 68 %, followed by polyethylene (PE) at 26 %, and polystyrene (PS) at 5 % in *F. caperata* around a landfill site in Tuscany. Capozzi et al. (2023) reported PET as the most abundant (75 %), followed by polypropylene (PP) at 15 %, and PS at 8 % in the moss *Hypnum cupressiforme* from rural sites in Campania (Southern Italy). Roblin et al. (2020) investigated wet and bulk rainfall in coastal areas around Ireland and reported PET as the most abundant polymer (71 %), followed by polyacrylonitrile at 11 %, PE at 11 %, and PP at 4 %. Similarly, Welsh et al. (2022) reported PET and polyamide as the most abundant polymers in remote sites far from anthropogenic emission sources in Ontario (Canada).

4.3. Vitality of the samples

Under the outdoor exposure conditions (7 weeks) the transplants maintained their overall vitality (as indicated by chlorophyll *a* fluorescence emission analysis – F_v/F_m ratio) and did not differ significantly from control samples or from samples growing in remote areas, suggesting that in the short-term there was a limited effect on lichen metabolism. Most likely, MPs (as well as heavy metals) were intercepted by the lichen cortex and did not reach the algal layer or penetrate into the cytoplasm of the cells. Stress symptoms due to MPs on lichen metabolism still need to be investigated. To date, studies on marine and freshwater algae have shown alterations of the potential quantum yield of PSII (F_v/F_m ratio) under increasing micro and nanoplastics concentrations (Gao et al., 2021), reduction of chlorophyll content (e.g., Sjollem et al., 2016; Wu et al., 2019), increased oxidative stress and malondialdehyde production, with effects on algal growth detectable only after exposure to high plastic concentrations (Gao et al., 2021). Some studies on plants suggest no significant impact on plant health (Li et al., 2022), while others report that plastics can stimulate the growth of higher plants, inasmuch as, being organic polymers, some plastics contain elements such as nitrogen, which can act as a stimulant and fertilizer for plants (e.g., De Souza Machado et al., 2019). Other studies have found that MPs, and especially nanoplastics (<1 µm), may have inhibitory and phytotoxic effects. These include inhibition of plant growth, cytotoxicity, genotoxicity, and oxidative stress in roots, compromised germination, alteration of antioxidant enzyme activity, reduction of photosynthetic activity, inhibition of gene expression, alteration of normal metabolism and tissues, and reduction of biomass (Giorgetti et al., 2020; Li et al., 2022; Spanò et al., 2022; Menicagli et al., 2022). Smaller particles can be absorbed by the roots and translocated to the shoots (Spanò et al., 2022), or accumulate on the surface of vascular plants and potentially penetrate the tissues of both aquatic and

terrestrial plants (Yin et al., 2021).

In addition, experiments carried out with human cultured cells have shown that sharp MP particles can induce toxicity and affect cell structures (Choi et al., 2021), while smooth particles, e.g., those made from high-density PE did not result in severe toxicity. In our study, the 7 weeks exposure to the urban atmospheric environment did not alter the potential quantum yield of primary photochemistry in the green algal photobiont. Nonetheless, prolonged exposure in the urban environment may affect the transplants, due to the overall load of pollutants and their synergistic effects on the lichen thalli. However, whether the vitality of the thalli is relevant or not during the accumulation of MPs still needs to be investigated. On the whole, the fruticose lichen *E. prunastri*, often employed to detect major and trace elements (e.g., Paoli et al., 2011) and PAHs deposition (e.g., Loppi et al., 2015), can also be profitably used to detect airborne MPs in urban and remote areas (Jafarova et al., 2022, 2023).

5. Conclusions

Our study characterized atmospheric MPs (fragments, fibres, and tyre wear particles) along a gradient of increasing anthropogenic activity, ranging from rural areas (far from direct anthropogenic sources) to urban environments (urban parks and parking lots). Further, we assessed the positive role of green urban areas in buffering atmospheric MPs. We found MPs in all lichen samples collected, with concentrations significantly higher in urban sites where human activity and pollution were more intense. However, atmospheric MPs were significantly lower (less than half) in urban parks compared with parking lots, suggesting that greenspaces may play a positive role in buffering atmospheric MPs. Our study suggests that *E. prunastri* is especially suitable for detecting airborne MPs owing to its morphological characteristics and capacity to accumulate atmospheric particles, offering a non-invasive method to assess relative levels of environmental pollution across different ecosystems. Further, lichen transplants appear to maintain their overall vitality (under a seven week exposure).

CRedit authorship contribution statement

Carolina Stringa Basile: Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Mehriban Jafarova:** Writing – review & editing, Methodology, Investigation. **Julian Aherne:** Writing – review & editing, Supervision, Methodology. **Jacob Wyonch:** Writing – review & editing, Investigation. **Monica Ruffini Castiglione:** Writing – review & editing, Investigation. **Carmelina Spanò:** Writing – review & editing, Investigation. **Zuzana Fačková:** Writing – review & editing, Investigation.

Stefano Loppi: Writing – review & editing, Investigation. **Luca Paoli:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

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.

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.envres.2025.122334>.

Data availability

Data are included in the supplementary materials

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