Microplastic occurrence in settled indoor dust in schools

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Abstract

This study examines for the first time the characteristics and human exposure of microplastics (MPs) in settled indoor dust in schools. An average of 195 MPs·g of dust^{-1} were detected in settled indoor composite dust samples from 28 schools in Shiraz. White-transparent microfibres with lengths 500-1000 µm were the most abundant type of MP found among the samples examined. Polyethylene terephthalate and polypropylene MPs were dominant across all types of MP found including microfibres. MPs had generally smooth morphology with sharp or regular edges which could have been released to the environment as primary MPs. Among all sampling sites, higher concentrations of MPs were found in the south and centre of the city. These were areas affected by high population density, high traffic load and high presence of industrial units and workshops. Principal Component Analysis (PCA) showed a positive strong correlation between sampling sites and MP physical characteristics. The PCA plots revealed that MP sheets and fragments were prevalent in sites in the North of Shiraz, whereas microfibres were mainly associated with sites in the South. The levels of MPs in the South of Shiraz were greater than in the rest of the country and the wind direction and topography were found to be important factors affecting the MP distribution observed. Compared to other population groups, elementary school students had relatively high exposure risk to MPs. This study reveals that microfibres are widespread in Shiraz' schools and pose a high exposure risk to MPs for young students.

Keywords: Microfibre; Human exposure; PCA; Shiraz; Iran

1. Introduction

Microplastics (MPs) are ubiquitous contaminants found in all environmental compartments. They are defined as plastic debris ranging from 1 μm to 5 mm (Frias and Nash et al., 2019). Because of the high durability and slow degradability of plastics, plastic debris remains in the environment for a long time. Some plastics have been manufactured directly within the MP size range. Others will have originated from bigger plastics, which have broken down to the MP size range. Currently, widely-used manufactured plastic materials are mostly composed of polyethylene (PE), polyethylene terephthalate (PET), polypropylene (PP), polystyrene (PS), polyvinyl chloride (PVC), polycarbonate (PC), nylon (NY), and polyurethane (PUR) (Geyer et al., 2017). MPs in the environment are often seen in a variety of shapes, usually named fibre (microfiber or filament or line), fibre bundle, sheet (film or foil), fragment, spherule (bead), pellet and foam (Rochman et al., 2019). This MP shape classification is based on quantifiable shape factors. For instance, microfibres are unbranched, rigid and thread-like filaments (Lehmann et al., 2021) and have a minimum length/diameter ratio of 3/1 (Cole, 2016), sheets (films) are rigid planar (flat) MPs manufactured of thin polymers (Lehmann et al., 2021) and fragments are irregular MP particles derived from fragmentation of larger plastics (Tanaka and Takada, 2016).

Fine airborne particulate matter comprising anthropogenic contaminants is a growing concern due to inducing various respiratory diseases such as asthma (California Air Resources Board, 2015; Keet et al., 2018) and heart disease (Du et al., 2016). The inhalation and ingestion of dust are important routes of human exposure to MPs (Atis et al., 2005; Soltani et al., 2021). MPs may be found in the upper and lower airways of the lungs depending on their size (Gasperi et al., 2018). MPs in dust arise from indoor textiles (e.g., clothing, carpet), soft furnishing (Liebezeit and Liebezeit, 2015; Dris et al., 2016) and outdoor pollution that can come from industrial activities, traffic, clothing, littering and emissions from waste treatment and water

64 treatment (Dris et al., 2016). Population density is an important factor intensifying the release
65 of MPs and reducing air quality (Browne et al., 2011). Recent studies indicate the occurrence
66 of MPs in outdoor dust (e.g., Dehghani et al., 2017; Gasperi et al., 2018; Allen et al., 2019;
67 Narmadha et al., 2020; Patchaiyappan et al., 2021; O'Brien et al., 2021; Strady et al., 2021)
68 and in indoor dust (Dris et al., 2017; Liu et al., 2019a; Zhang et al., 2020a, b; Bahrina et al.,
69 2020; Gaston et al., 2020; Soltani et al., 2021). This study assesses MP occurrence in indoor
70 dust in primary classrooms for the first time. School classrooms are among the most important
71 locations to assess the status of contaminants in urban areas. This is because students are
72 particularly vulnerable to ambient air contaminants (Salvi, 2007; Landrigan et al., 2010).
73 Children, disproportionately, are more vulnerable to air pollution than adults as a result of a
74 higher volume of air pollutants inhaled per their body weight (Landrigan et al., 2010), as well
75 as higher ingestion of settled dust through the hand-to-mouth pathway than adults (Ljung et al.,
76 2006; Keshavarzi et al., 2015). Children's defense mechanisms are under evolution and
77 toxicokinetics of harmful substances are slower in children than in adults. (Schwartz, 2004;
78 Salvi, 2007; Landrigan et al., 2010). This is the first investigation of MPs in the school
79 environment to the best of our knowledge. To achieve coverage in depth in the study, the
80 research has focused on schools located in the Iranian metropolis Shiraz.

81 Air quality of Shiraz has declined over the recent years due to rapid urbanization, occasional
82 dust storm events, fast population growth and increased industrial activities. Among the
83 industrial activities that has affected air quality in Shiraz is an oil refinery (North East Shiraz);
84 a cement factory (West Shiraz) and multiple wood and electronic workshops in an industrial
85 zone in South Shiraz. Other major anthropogenic sources of air pollution in Shiraz are fumes
86 from vehicles and chimneys. Besides, particulate matter from the western neighboring regions
87 (Ahvaz, Persian Gulf, Saudi Arabia, Iraq, Kuwait and Syria) has increased contamination levels
88 in Shiraz in recent years (Keshavarzi et al., 2015). Because of important (in terms of volume)

and varied sources of air pollution in Shiraz and the extensive dust deposition in the city, indoor air quality and MP pollution indoors needs to be studied.

2. Materials and methods

2.1. Research area description

Shiraz (29°37′08″N latitude and 52°31′14″E longitude) is the capital of Fars province of Iran (Fig. 1a). It has an altitude of 1484 m from sea level. Shiraz is the largest populated city in South West Iran and it is the fifth in the country. Shiraz's municipal population is ~1.566 million with a population density of 6525 inhabitants per km^2. Its surface area is 240 km^2 and it forms an elongated area North West-South East surrounded by the Babakouhu and Derak anticlines. Geologically, Shiraz is situated in the Zagros Thrust Belt (Stöcklin, 1968). This setting limits land use and traffic pathways. Traffic becomes concentrated on a few roads and this intensifies traffic load in the municipal area and increases air pollution. Overall, Shiraz has a semiarid climate, with rainy and mild winters and arid summers. The annual average precipitation and temperature are 337 mm and 18°C, respectively. Southwesterly and westerly winds dominate in Shiraz, and the average wind speed is 2.35 m/s (Fig. 1b) (IRIMO, 2020).

2.2. Sampling

Sampling was conducted considering the location of schools along with several important factors including geography, wind direction, the position of industrial units and workshops, population density and traffic load. The sampling pattern included schools that covered the entire range of the city. Sampling was carried out in 28 schools including 12 boys' schools, 8 girls' schools and 8 coed schools. Comprehensive information on each individual school is given in Table S1. Composite dust samples (15 g) were combined from mixture of five dust

subsamples (taken from five representative classrooms) of approximately 3 g each. The samples were collected from a range of horizontal and flat surfaces in each classroom such as wall shelves, cabinets and floors, especially near corners where dust accumulates. Regarding sampling from the floor, samples were taken near corners where students do not usually pass. The sampling was carried out using a brush made of horsetail strands and a steel dustpan and it took place during June 2019. The brush and dustpan were cleaned using deionized water after their use in every school. Collected samples were stored in sealed glass jars and transferred to the laboratory for further examination.

2.3 Sample treatment

Sample treatment was carried out based on the standard protocols described in Dehghani et al. (2017). Briefly, prior to the extraction of MPs, dust samples were passed through a 5mm metal sieve in order to separate unwanted coarse impurities. Then, 10 g of each sample was transferred to 100 ml glass beaker. Sample treatment started with removing organic matter (OM) from the samples, since the presence of OM in the samples may be confused with MPs. The removal of OM was carried out by adding an adequate amount (~ 50 ml) of hydrogen peroxide (30% H_2O_2) until the oxidation reaction was completed in ~10 days (Nuelle et al., 2014; Dehghani et al., 2017; Prata et al., 2019; Nematollahi et al., 2020). Following, the samples including unreacted H_2O_2 were left to dry in a sand bath at 80 °C for 8 hours. Finally, the samples were washed with deionized water and left to dry in the sand bath.

To extract MPs from the digested samples, saline solutions were used to float MPs above the settled solid phase. A total of 50 ml of $ZnCl_2$ solution with a density of 1.6 g cm^{-3}, pre-filtered with 2µm filter paper, was added to each beaker containing digested dust samples and shaken at 4000 rpm for 5 min (Löder and Gerdts, 2015). The dust residue in the beakers was

allowed to settle, and then the supernatant was transferred to falcon tubes for the centrifugation (4000 rpm for 5 min). After the centrifugation, the supernatant was filtered (2-μm S&S filter papers, blue band, grade 589/3) using vacuum. Filter papers, covered with aluminum foil, were left to dry in clean cabinets at room temperature. The extraction process with $ZnCl_2$ was repeated two more times using the same filter papers. Finally, the residual materials on the filter papers were transferred to glass Petri dishes using a natural (non-plastic) hairbrush for further analysis. To minimise loss of MPs in this step, the filters were examined with optical microscopy.

2.4. MPs detection and analysis

An optical microscope (Carl-Zeiss, West Germany) was used to identify and count MPs. ImageJ software was used to count MPs. MPs were classified according to their shape (fragment, sheet and fibre), colour (black/grey, blue/green, red/pink, yellow/orange and white/transparent) and size (50 μm ≤ L < 100 μm, 100 ≤ L< 250 μm, 250 ≤ L< 500 μm, 500 ≤ L< 1000 μm, 1000 ≤ L< 5000 μm) under the optical microscope, where L is the length of the largest aspect of the MP. Representative MPs were selected and characterised for their morphology and composition using a Scanning Electron Microscope (SEM: TESCAN Vega 3, Czech Republic) coupled with an Energy Dispersive X-ray Microanalyzer (EDS). The polymer or mix of polymers, making the MP, was identified with a Raman microscopy (Lab Ram HR Evolution, Horiba Japan). Briefly, the samples were mounted onto two-sided Cu adhesive tapes. For the SEM-EDS analysis, MPs were coated with gold. The analysis of MPs with Raman was performed directly, without coating the sample. The excitation source was a laser irradiating at 785 nm , and the detection was within 400 - 1800 cm^{-1}. The spectra obtained in Raman was compared with reference spectra from the Raman microscope polymer database.

2.5. Quality control

Laboratory equipment and benches were cleaned with ethanol and paper wipes before performing sample treatments and analysis of MPs. Laboratory glassware was washed as indicated by Dehghani et al. (2017). Prior to sample treatment, deionized water and chemical reagents were filtered using filter paper (S&S, 2-μm) to avoid inadvertent plastic particles. Items made of plastic (e.g. suits, gloves and fabrics) were not allowed in the laboratory during the period when this study was in progress. Potential contamination of laboratory experiments was assessed by analyzing a blank control sample consisting of an empty Petri dish left open next to other samples during the analysis (about 3 weeks). Also, a number of blind and blank samples were inserted in the routine analysis of MPs to check the accuracy of the analysis. Detailed controlling protocols of laboratory experiments and analyses of MPs can be found in Dehghani et al. (2017) and Nematollahi et al. (2020).

2.6. Data processing

2.6.1. Software and statistical analysis

The software Arc GIS 10.3 was used to display the spatial distribution of MPs. Calculations were done using Excel 2016 and SPSS22 software. The statistical tests were performed using SPSS 22. These tests included Shapiro-Wilk to assess normality of the data and Kruskal-Wallis to evaluate homogeneity of variance between the abundance of MPs regarding their types in the samples. To find relationships between MP characteristics, Principal Component Analysis (PCA) was carried out using the XLSTAT 2016. The variables were normalized using the Kaiser method before PCA analysis. Sample adequacy for PCA was checked using the Kaiser–Meyer–Olkin (KMO) method.

185

186 **2.6.2. Human exposure to MPs**

187 Exposure to MPs present in the indoor dust was estimated via the ingestion pathway, which
188 is the only exposure route of MPs > 50 μm. The human exposure to contaminants through
189 different pathways was introduced by USEPA (2011) in the Exposure Factors Handbook and
190 Liu et al. (2019a) adapted it for estimating the exposure to MPs through ingestion. In this work,
191 the estimated daily exposure to MPs through the ingestion (EDI; MP kg^{-1} day^{-1}), was estimated
192 using Eq. 1 as follows (Liu et al., 2019a):

193 $$EDI_{ing} = \frac{C \times f \times m_{di}}{BW} \quad \text{Eq. 1}$$

194 where "C" represents the concentration of MP in indoor dust, "f" is the indoor exposure
195 fraction, "m_{di}" is the indoor dust ingestion rate (mg day^{-1}), and BW is body weight. The
196 reference values of the fractions "f " (Johnson-Restrepo and Kannan, 2009) and "m_{di}" (USEPA,
197 2017) for various human age groups including infants (0.5–1 year), toddlers (1–6 years),
198 children (6–12 years), teenagers (12–21 years), and adults (≥ 21 years) were 0.88, 0.79, 0.79,
199 0.88, and 0.88; and 40, 40, 30, 20, and 20 mg day^{-1}, respectively (USEPA, 2017). The average
200 BW considered for infants, toddlers, children, teenagers and adults in Asian countries were 5,
201 19, 29, 53, and 63 kg, respectively (Guo and Kannan, 2011; Liao et al., 2012).

202

203 **3. Results and discussion**

204 **3.1. MP Concentration and distribution**

205 The variability of MP concentrations within dust samples, shown with the coefficient of
206 variation (CV), can be grouped into four categories (Nezhad et al., 2015): CV ≤ 20% (low
207 variability), 20% < CV ≤ 50% (moderate variability), 50% < CV ≤ 100% (high variability),

CV > 100% (exceptionally high variability). The MPs in this study displayed a CV ≥ 90%, reflecting high variability and thus uneven dispersion within the dust samples (included in Table 1). Based on the Shapiro-Wilk test, the MP levels within the dust samples were non-normally distributed (p <0.05). This implies that elevated levels of MPs in some sites may have been affected by hotspots. The statistical Kruskal-Wallis Test indicated significant differences between the abundance of MPs regarding their types (p <0.05). This assumption is in line with the non-normal distribution of MPs in dust samples.

MP concentrations were higher in the indoor dust of schools located in the southern and central parts of the city than in the north (overview shown in Fig. 2a). This can be because of differences in population density, active industrial units and workshops and traffic load. The lifestyle, land topography, geographical environment and dominant wind direction are also important parameters that can affect the distribution of airborne MPs. These factors favour elevated concentrations of MPs (both outdoor and indoors), although MPs can easily enter the indoor environment of the classrooms through windows, doors, and also via soles of shoes (Kurt-Karkus, 2012). Textiles, especially students' clothing, are also potential sources of MPs in schools.

Overall, human lifestyle is a main factor in producing ambient MPs in some areas of Shiraz where residents usually dry their clothing, bed liners and sheets, blankets and pillows by hanging them on lines outdoors and thus exposing them to sunlight and open-air. The frequent exposure to sunlight (and ultraviolet radiation) can facilitate the breakdown of larger synthetic textiles (Song et al., 2017) and this could affect MP ambient concentrations. This washing and drying custom, and also conventional cleaning methods using a brush and dustpan, are more common in the southern parts of the city where there is the highest population density and people with lower socioeconomic status. However, the latter does not have a clear link with the release of MPs. Shiraz has southwesterly and westerly winds predominantly. This will favour

MPs being transported to the centre and south of the city, and this corresponds well with the distribution of MP levels found (Fig. 2). Based on the topography of Shiraz, where elevation reduces towards the south and leads to the dominant wind direction, there is a direct relationship between the increased levels of MPs, dominant wind direction and decreased elevation in the southern parts of the city. The gender of the pupils going to school (Fig. 2b) and their different clothing were not found to be explanatory factors for the MP levels found in the investigated schools.

3.2. Overview of MP characteristics

Representative MPs (5%) visually detected in the treated dust samples were further characterized by complementary analytical techniques. Overall, MPs comprised various physical characteristics, i.e., they had a range of shapes (fibre, fragment, and sheet), colours (black, grey, blue, green, red, pink, yellow, orange, white, and transparent), and sizes (predominantly 250 μm ≤ L ≤ 1000 μm) (see Fig. 4).

Microfibres were the dominant MP in the majority of all the study schools, whilst fragments were the major shape only in S1. An overview of MP properties and levels from monitoring across sampling sites is given in Fig. 4 and Table 1. These results are in agreement with recent studies finding microfibers as the most abundant MPs in indoor dust of urban districts (e.g., Dris et al., 2017; Catarino et al., 2018; Chen et al., 2019; Vianello et al., 2019; Liu et al., 2019a; Zhang et al., 2020a, b; Soltani et al., 2021). This is because synthetic fibres are easily torn and separated from potential indoor sources including clothing and other indoor soft furniture such as carpets and curtains. The levels of indoor microfibres are also largely affected by cleaning habits and ventilation (Dris et al., 2017), and the proportion of natural/ synthetic fibres changes with regions. For instance, in a study conducted in Paris, about 33% of indoor fibres were

synthetic (Dris et al., 2017). In contrast, in a study in an indoor environment in Edinburgh, both natural and synthetic fibres had the same proportion (Vianello et al., 2019). MP fragments were dominant in S1. These may arise from the degradation of the larger plastics including those used in cleaning products, packaging and containers (Wang et al., 2018; Yuan et al., 2019).

White-transparent MPs were dominant in most investigated schools (compiled in Fig. 4b and Table 1). The predominance of white-transparent MPs was also found in outdoor dust of the Asaluyeh county in Iran (Abbasi et al., 2019), but not in other studies characterizing MPs in outdoor dust in Iran (Dehghani et al., 2017; Abbasi et al., 2017). Light coloured MPs in the dust of Shiraz may arise from various disposable plastics used in the commercial and residential areas. For instance, plastic bags are an integral item of shopping centres in the study area. The elevated levels of MPs with light colour may also suggest having had prolonged photobleaching, and if so, this would reflect dust as a sink for MPs. The removal of plastics at an early stage, even from their source, shops, would prevent finding aged MPs in other environmental compartments including dust in school. Dark MPs were among the abundant items found in indoor dust samples and they also prevailed in coastal and marine sediments found elsewhere in Iran (Nematollahi et al., 2020; Jahromi et al., 2021). Although the colour of MPs may suggest their possible source (Fahrenfeld et al., 2019; Zhang et al., 2020c), colour is not permanent. Thus, colour-based identification of the origin of MPs might to some extent be speculative. The colourful MPs found in the schools in Shiraz may arise from various highly durable, consumable plastic materials as proposed elsewhere (Andrady, 2017; Eo et al., 2019), including worn urban and industrial constructive materials. The occurrence of colourful MPs in the indoor environment may also indicate that they are relatively new, and thus have not been affected by photobleaching.

MPs were found in a variety of sizes (compiled in Fig. 4c and Table 1) and that 500 - 1000 μm were dominant (31.5%). In contrast, relatively small MPs (50 - 100 μm) had the lowest

proportion (2.5%). The dominance of middle-sized MPs (500 -1000 μm) is in agreement with a study conducted in outdoor dust in Tehran (Dehghani et al., 2017). Exposure to MPs via ingestion and inhalation pathways is probable (Gall and Thompson, 2015). In particular, exposure to large MPs, especially in children, is more likely to happen via dust intake. In contrast, breathing small MPs is highly likely (Andrady, 2011; Carson et al., 2013; McCormick et al., 2014), for instance, microfibers > 250 μm were identified in 87% of 114 investigated lungs (Pauly et al., 1998), and a study conducted in Shanghai estimated a daily inhaled level of ~21 MPs by residents (Liu et al., 2019b).

3.3. MP polymer type and origin

The composition of the MPs is a good indicator of their origin. Interestingly, some MPs with similar colours were constituted by different polymers. This reflects that MPs with similar appearances can come from various sources. Table S2 includes the polymer type and colour of twenty representative MPs. Raman spectra illustrating the identification of polymers carried out for selected MPs are shown in Fig. S1. The frequency of polymers found in MPs in the schools was PET (50%) > PP (35%) > PS (10 %) > NY (5%). Similar polymer compositions of airborne MPs were found in recent studies elsewhere. For instance, Chen et al. (2019) showed that PP, PE, PS, and PET were the main polymers in MPs in atmospheric fallout. Both PP and PET are among commonly used polymers in producing textiles, fabrics, synthetic fibres and packaging materials (Allen et al., 2019). PS, because of its thermal insulating properties, is commonly used in the manufacturing and packaging industries (Di et al., 2019). Nylon is part of daily life products including clothing, tyres, carpets, ropes, industrial cords, seatbelts, and fishnets (Hu and Yang, 2000). The abundance of microfibres made of PET and PP in the indoor dust of the schools of Shiraz suggests that synthetic textiles are the main MP sources.

306

307 **3.4. MP morphology and elemental composition**

308 The degree of MP weathering was examined by surface morphology analysis of eighteen
309 representatives of MPs. The SEM-EDS spectra, reflecting the MP elemental composition,
310 along with the surface morphology of a number of selected MPs are shown in Fig. 5. MPs
311 mainly had smooth morphology with sharp and regular edges, reflecting that they were less
312 affected by weathering or were primary MPs. However, marks of physicochemical weathering
313 as grooves on microfibres' surface; and irregular edges and cracks on the surface of non-fibrous
314 items were observed. This reflects that these MPs could have originated from a primary plastic,
315 i.e., they are secondary MPs. The weathered surface of MPs is more likely to adsorb
316 contaminants such as toxic metals and organic compounds (Kowalski et al., 2016), and thus
317 weathered MPs are more likely to induce unfavorable impacts. This is because fragmentation
318 and breakdown of plastics result in increased surface, favours the presence of edges, and these
319 features favour adsorption. Most MPs had sharp edges, reflecting that they recently entered
320 into the environment (Hidalgo-Ruz et al., 2012; Rocha-Santos, 2017). It is worth noting that
321 indoor MPs are less affected by the weathering factors (e.g., sunlight and temperature changes)
322 and thus are less likely to break down to the small sizes that the outdoor plastics can degrade
323 to (Gaston et al., 2020). Therefore, the high frequency of large MPs is expected in the indoor
324 environment.

325 The plastic nature of particles was confirmed by the elemental composition of the examined
326 MP items. MPs were composed of a high percentage of C and O with SEM-EDS (Fig. 5). MPs
327 had a minor percentage of other elements including N, Na, Mg, Al, Si, Cl, Ti, Mn, Cu, Zn, Sn,
328 Sb, Hg, and Pb. The presence of major elements including Al, Si, Na, Mg and Mn, which are
329 dominant constituents of silicate minerals (e.g., clays) was likely to be caused by silicates
330 adsorbed onto the surface of MPs. Pb, Hg, Sb, Sn, Zn, Cu, and Ti are well-known urban

elements and likely originated from anthropogenic activities including traffic-related and industrial activities (Ahmady-Birgani et al., 2015; Nematollahi et al., 2021b). In addition, a number of trace elements, such as Pb, are used as additives in paints (Dixon et al., 2009; Brokbartold et al., 2012; Gupta and Gauri, 2013; Ogilo et al., 2017) and could have their origin there. Legacy Pb from fuel may also contribute to its levels in MPs. Zn and Cl largely originate from chemicals used in sample treatment and or salt particles attached to MPs. Trace elements (e.g., Cr, Fe, Cu, and Ti) are also widely used as additives in plastic materials to achieve desired properties such coloured plastic (Nematollahi et al., 2020).

3.5. MPs in the study area compared with other locations

The concentrations of MPs found in the Shiraz schools' indoor dust were compared with those in outdoor dust of Iran and indoor dust from other countries. Table 2 presents a compilation of MP concentrations and characteristics in indoor dust of the study area and other locations. Compared to studies conducted on outdoor dust in Iran, the indoor dust of Shiraz presented higher number of MPs. The very different concentrations of MPs detected in the studies may be ascribed to different land uses, sources distributing MPs to the indoor environments, the wind impact, the population density, and also analytical factors including the number and representativity of the samples considered or the working microscopic range used for the MPs screening. The shape of MPs most commonly found was different across studies: MP fibres were dominant in the indoor dust of Shiraz, in a study carried out in outdoor dust of Bushehr city of Iran (Abbasi et al., 2017) and in most locations included in Table 2. Regarding MP occurrence in indoor dust from worldwide locations, microfibres were the dominant type of MPs identified in indoor air. MPs can be composed of a variety of polymers, though PET and PE were prevalent in the investigated indoor dust samples of most locations.

355

356 **3.6. Interrelationship of MP characteristics within sites**

357 Relationships and correlations between MP characteristics (shape, colour, and size) and
358 sampling sites were found using a PCA biplot (Fig. 6a). A KMO of 0.8 indicated that the PCA
359 analysis carried out provides important results for the interrelationships of the investigated
360 variables (Nematollahi et al., 2021b). The PCA biplot reflects both principal components (PCs)
361 accounting for a total variance of 79.4%, thereby, PC1 and PC2 explained 68.6% and 10.8%
362 of the total variance, respectively. The sheet and fragment MPs from the dust samples in the
363 PCA plot have contribution from both PC1 and PC2. In contrast, fibre MPs lie in PC1 and have
364 no significant positive correlation with sheets and fragments, i.e., their eigenvectors make a
365 very large angle. In addition, surprisingly, sheets and fragments have low correlation with MP
366 characteristics. Fibre's eigenvector has an opposite direction to sheet and fragment's. This
367 reflects a different distribution of sheets and fragments in the samples and thus it likely
368 originates from different sources and distribution pathways. More specifically, the sheet and
369 fragment's eigenvectors are directed towards S1-S4, reflecting that the distribution of these
370 types of MPs prevails in the sites. Hence, the MP sources in these sites are likely different. On
371 the contrary, fibre MPs have a high positive correlation with MP characteristics. Fibre's
372 eigenvector makes a close angle with the eigenvectors of MP characteristics including white-
373 transparent, black-grey, blue-green, $500 \leq L < 1000$ μm, $1000 \leq L < 5000$ μm, and $250 \leq L < 500$
374 μm. This signifies that the majority of fibre MPs in most sites have the above-mentioned
375 characteristics. Moreover, fibre's eigenvector is biased towards S13 and S21: a reflection that
376 fibre MPs have higher distribution in these sites.

377 In all schools, microfibres have the highest distribution, though the distribution of sheets and
378 fragments in the northern part of Shiraz, including sites S1-S4 is greater (Fig. 6b). Similarly,
379 there is a higher distribution of red-pink MPs in schools located in the northern regions of the

city. These results comply with the multivariate statistical analysis with PCA (Fig. 6a). Hence, MP characteristics explain well the distribution of MPs, and thus their distinct origins within the sampling sites. PCA plots inform about the characteristics of the MPs in the 28 sampled schools in a very concise and visual manner, however, the different abundancy of MPs in the different sites does not come across.

3.7. Human exposure to MP

The statistical summary of EDIs of fibres, sheets, fragments, non-fibrous particles, and all MPs for the different age groups is presented in Table 3. The mean EDI (in MP kg^{-1} day^{-1}) of all MPs for the infants (13.7) was the highest, followed by toddlers (3), children (1.61), teenagers (0.7), and adults (0.6). Since, in general, the students and teachers in schools are older than 6 years of age, infants and toddlers are not exposed to MPs in the school. Hence, the exposure to MPs contained in indoor dust in elementary school with students between 6 and 12 years old is higher than in the middle and high schools with teenage students. The higher exposure risk posed by elementary school students is due to the childrens' lower weight and higher indoor dust ingestion rate than middle and high school students. There was a daily exposure risk of 0.14, 0.93, and 4.86 MP items kg^{-1} day^{-1} for elementary school students at 5^{th}, 50^{th}, and 95^{th} percentiles of the total estimated exposure, respectively, while the corresponding values for middle and high school students (teenagers) were 0.1, 0.4, and 2.0 MP items kg^{-1} day^{-1}, respectively. Among the different types of MPs, the mean EDI of MP fibres for all age groups was higher than that of non-fibrous MPs. Therefore, the exposure risk posed by MP fibres in all schools was higher than that of non-fibrous particles. This is because of the higher abundance of fibre MPs in indoor dust than other MP types. PET and PP were the main types of plastics composing the MPs found. They are among the plastics with lower density and

hence can be transported easier. The exposure to MPs will cause exposure to toxic metals such as Hg or Pb, (as shown in the EDS results in Fig. 5) but this has unknown consequences.

MPs may have some capacity to adsorb pollutants from the environment and may also contain various toxic additives. Thus, MPs can act as carriers of pollutants such as metals, as found in this study and elsewhere (Holmes et al., 2012), bisphenol A (Teuten et al., 2009), organochlorine pesticides (Ogata et al., 2009), polyaromatic hydrocarbons (Fries and Zarfl, 2012), phthalates (Fries et al., 2013), polychlorinated biphenyls (Velzeboer et al., 2014), and organic flame retardants (Jang et al., 2017). The pollutants carried by MPs might be released in the human body after their ingestion, although this is currently unclear. What has become evident, is that some MPs (and their sorbed pollutants) can bioaccumulate and be transferred to organisms at higher trophic levels (Savoca et al., 2020).

If pollutants sorbed onto MPs were released after their ingestion, this would imply an additional potential health risk, especially for elementary school students. From the results of this study, measures to reduce MPs in the school classrooms are recommended and these would have greater impact when addressing microfibre pollution. Future work could include the introduction of passive samplers in schools to relate the levels of MPs with the duration of the exposure and this will improve risk assessment to the exposure of MPs in classrooms. Furthermore, the study should consider seasonability, exposure to suspended dust and it could be expanded to other types of settings, especially those related to occupational exposure to MPs.

Conclusion

The present study is the first one to examine MPs in settled indoor dust in schools and it is based in Iran. MPs have been assessed for their concentration, distribution, physicochemical

properties and human health risk. Environmental factors, including wind direction, land topography, population density, presence of MP hotspots sources, traffic load and human lifestyle have impacted the distribution and concentrations of MPs in the dust and have helped to explain why there is a greater abundance of microplastics in the southern parts of Shiraz. The dominant physicochemical properties of the MPs found in the study reflected that items made of synthetic textiles had the largest contribution to the MP pollution in the dust. PET and PP microfibres, mainly white-transparent and with 500-1000 μm length, were the most abundant MPs in indoor dust. Findings of this study also indicated the importance of MPs as a carrier for potentially toxic pollutants which may be used as additives in the matrix of MPs and or subsequently be adsorbed on their surface. The dominance of MP fibres highlighted the importance of indoor plastic items as sources of microfibres. The elevated levels of MP fibres in indoor environments meant an increased daily exposure to MPs through the ingestion pathway in elementary school students. These results can be regarded as a baseline for the occurrence of MPs in indoor dust of school classrooms in future studies. Although the health effects of MPs on children are largely unknown, the ubiquity of MPs in indoor and outdoor environments underscores the importance of future research in this area. A prospective study with detailed respiratory exposure to MPs is needed. Future work should compare MPs in dust suspended in schools' air with MPs in settled dust. Seasonal sampling as well as examining outdoor dust is highly recommended to identify the sources of MPs in indoor dust.

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References

Abbasi, S., Keshavarzi, B., Moore, F., Delshab, H., Soltani, N. and Sorooshian, A., 2017. Investigation of microrubbers, microplastics and heavy metals in street dust: a study in Bushehr city, Iran. *Environmental earth sciences*, *76*(23), pp.1-19. https://doi.org/10.1007/s12665-017-7137-0.

Abbasi, S., Keshavarzi, B., Moore, F., Turner, A., Kelly, F.J., Dominguez, A.O. and Jaafarzadeh, N., 2019. Distribution and potential health impacts of microplastics and microrubbers in air and street dusts from Asaluyeh County, Iran. *Environmental pollution*, *244*, pp.153-164. https://doi.org/10.1016/j.envpol.2018.10.039.

Ahmady-Birgani, H., Mirnejad, H., Feiznia, S. and McQueen, K.G., 2015. Mineralogy and geochemistry of atmospheric particulates in western Iran. *Atmospheric Environment*, *119*, pp.262-272. https://doi.org/10.1016/j.atmosenv.2015.08.021.

Allen, S., Allen, D., Phoenix, V.R., Le Roux, G., Jiménez, P.D., Simonneau, A., Binet, S. and Galop, D., 2019. Atmospheric transport and deposition of microplastics in a remote mountain catchment. *Nature Geoscience*, *12*(5), pp.339-344. https://doi.org/10.1038/s41561-019-0335-5.

Andrady, A.L., 2011. Microplastics in the marine environment. *Marine pollution bulletin*, *62*(8), pp.1596-1605. https://doi.org/10.1016/j.marpolbul.2011.05.030.

Andrady, A.L., 2017. The plastic in microplastics: A review. *Marine pollution bulletin*, *119*(1), pp.12-22. https://doi.org/10.1016/j.marpolbul.2017.01.082.

Atis, S., Tutluoglu, B., Levent, E., Ozturk, C., Tunaci, A., Sahin, K., Saral, A., Oktay, I., Kanik, A. and Nemery, B., 2005. The respiratory effects of occupational polypropylene flock exposure. European Respiratory Journal, 25(1), pp.110-117. https://doi.org/10.1183/09031936.04.00138403.

Bahrina, I., Syafei, A.D., Satoto, R., Jiang, J.J., Nurasrin, N.R., Assomadi, A.F., Boedisantoso, R., Hermana, J. and Nasir, M., 2020. An Occupant-Based Overview of Microplastics in Indoor Environments in the City of Surabaya, Indonesia. *Journal of Ecological Engineering*, *21*(8), pp.236-242. https://doi.org/10.12911/22998993/126876.

Brokbartold, M., Wischermann, M. and Marschner, B., 2012. Plant availability and uptake of lead, zinc, and cadmium in soils contaminated with anti-corrosion paint from pylons in comparison to heavy metal contaminated urban soils. *Water, Air, & Soil Pollution*, *223*(1), pp.199-213. https://doi.org/10.1007/s11270-011-0851-4.

Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T. and Thompson, R., 2011. Accumulation of microplastic on shorelines woldwide: sources and sinks. *Environmental science & technology*, *45*(21), pp.9175-9179. https://doi.org/10.1021/es201811s.

California Air Resources Board. "Particulate Matter Program". 2015. https://ww3.arb.ca.gov/desig/pm.htm [accessed 30 Mar 2020].

Carson, H.S., Nerheim, M.S., Carroll, K.A. and Eriksen, M., 2013. The plastic-associated microorganisms of the North Pacific Gyre. *Marine pollution bulletin*, *75*(1-2), pp.126-132. https://doi.org/10.1016/j.marpolbul.2013.07.054.

Catarino, A.I., Macchia, V., Sanderson, W.G., Thompson, R.C. and Henry, T.B., 2018. Low levels of microplastics (MP) in wild mussels indicate that MP ingestion by humans is minimal compared to exposure via household fibres fallout during a meal. *Environmental pollution*, *237*, pp.675-684. https://doi.org/10.1016/j.envpol.2018.02.069.

Chen, G., Feng, Q. and Wang, J., 2020. Mini-review of microplastics in the atmosphere and their risks to humans. *Science of The Total Environment*, *703*, p.135504. https://doi.org/10.1016/j.scitotenv.2019.135504.

Cole, M., 2016. A novel method for preparing microplastic fibers. Scientific reports, 6(1), pp.1-7. https://doi.org/10.1038/srep34519.

Dehghani, S., Moore, F. and Akhbarizadeh, R., 2017. Microplastic pollution in deposited urban dust, Tehran metropolis, Iran. *Environmental Science and Pollution Research*, *24*(25), pp.20360-20371. https://doi.org/10.1007/s11356-017-9674-1.

Di, M., Liu, X., Wang, W. and Wang, J., 2019. Manuscript prepared for submission to environmental toxicology and pharmacology pollution in drinking water source areas: microplastics in the Danjiangkou Reservoir, China. *Environmental toxicology and pharmacology*, *65*, pp.82-89. https://doi.org/10.1016/j.etap.2018.12.009.

Dixon, S.L., Gaitens, J.M., Jacobs, D.E., Strauss, W., Nagaraja, J., Pivetz, T., Wilson, J.W. and Ashley, P.J., 2009. Exposure of US children to residential dust lead, 1999–2004: II. The contribution of lead-contaminated dust to children's blood lead levels. *Environmental Health Perspectives*, *117*(3), pp.468-474. https://doi.org/10.1289/ehp.11918.

Dris, R., Gasperi, J., Mirande, C., Mandin, C., Guerrouache, M., Langlois, V. and Tassin, B., 2017. A first overview of textile fibers, including microplastics, in indoor and outdoor environments. *Environmental pollution*, *221*, pp.453-458. https://doi.org/10.1016/j.envpol.2016.12.013.

Dris, R., Gasperi, J., Saad, M., Mirande, C. and Tassin, B., 2016. Synthetic fibers in atmospheric fallout: a source of microplastics in the environment?. *Marine pollution bulletin*, *104*(1-2), pp.290-293. https://doi.org/10.1016/j.marpolbul.2016.01.006.

Du, Y., Xu, X., Chu, M., Guo, Y. and Wang, J., 2016. Air particulate matter and cardiovascular disease: the epidemiological, biomedical and clinical evidence. Journal of thoracic disease, 8(1), p.E8. https://doi.org/10.3978/j.issn.2072-1439.2015.11.37

Eo, S., Hong, S.H., Song, Y.K., Han, G.M. and Shim, W.J., 2019. Spatiotemporal distribution and annual load of microplastics in the Nakdong River, South Korea. *Water research*, *160*, pp.228-237. https://doi.org/10.1016/j.watres.2019.05.053.

Fahrenfeld, N.L., Arbuckle-Keil, G., Beni, N.N. and Bartelt-Hunt, S.L., 2019. Source tracking microplastics in the freshwater environment. *TrAC Trends in Analytical Chemistry*, *112*, pp.248-254. https://doi.org/10.1016/j.trac.2018.11.030.

Frias, J.P.G.L., Nash, R., 2019. Microplastics: Finding a consensus on the definition. Mar. Pollut. Bull. 138, 145–147. https://doi.org/10.1016/j.marpolbul.2018.11.022.

Fries, E., Dekiff, J.H., Willmeyer, J., Nuelle, M.T., Ebert, M., Remy, D., 2013. Identification of polymer types and additives in marine microplastic particles using pyrolysis-GC/MS and scanning electron microscopy. Environ. Sci. Proc. Impacts. 15, 1949–1956. https://doi.org/10.1039/c3em00214d.

Fries, E., Zarfl, C., 2012. Sorption of polycyclic aromatic hydrocarbons (PAHs) to low and high density polyethylene (PE). Environ. Sci. Pollut. Res. 19, 1296–1304. https://doi.org/10.1007/s11356-011-0655-5.

Gall, S.C. and Thompson, R.C., 2015. The impact of debris on marine life. *Marine pollution bulletin*, *92*(1-2), pp.170-179. https://doi.org/10.1016/j.marpolbul.2014.12.041.

Gasperi, J., Wright, S.L., Dris, R., Collard, F., Mandin, C., Guerrouache, M., Langlois, V., Kelly, F.J. and Tassin, B., 2018. Microplastics in air: are we breathing it in?. *Current Opinion in Environmental Science & Health*, *1*, pp.1-5. https://doi.org/10.1016/j.coesh.2017.10.002.

Gaston, E., Woo, M., Steele, C., Sukumaran, S. and Anderson, S., 2020. Microplastics differ between indoor and outdoor air masses: insights from multiple microscopy methodologies. *Applied spectroscopy*, *74*(9), pp.1079-1098. https://doi.org/10.1177/0003702820920652.

Geyer, R., Jambeck, J.R. and Law, K.L., 2017. Production, use, and fate of all plastics ever made. Science advances, 3(7), p.e1700782. https://doi.org/10.1126/sciadv.1700782.

Guo, Y. and Kannan, K., 2011. Comparative assessment of human exposure to phthalate esters from house dust in China and the United States. *Environmental science & technology*, *45*(8), pp.3788-3794. https://doi.org/10.1021/es2002106.

Gupta, A. and Gauri, S.K., 2013. Determination of optimal quantities of different types of driers for addition in the batches of paint formulation. *International Journal of Engineering, Science and Technology*, *5*(4), pp.1-13. https://doi.org/10.4314/ijest.v5i4.1.

Hidalgo-Ruz, V., Gutow, L., Thompson, R.C. and Thiel, M., 2012. Microplastics in the marine environment: a review of the methods used for identification and quantification. *Environmental science & technology*, *46*(6), pp.3060-3075. https://doi.org/10.1021/es2031505.

Holmes, L.A., Turner, A., Thompson, R.C., 2012. Adsorption of trace metals to plastic resin pellets in the marine environment. Environ. Pollut. 160, 42–48. https://doi.org/10.1016/j.envpol.2011.08.052.

549 Hu, X-C, Yang, H. H., 2000. 1.11 - Polyamide and Polyester Fibers, Editor(s): Anthony Kelly, Carl Zweben,
550 Comprehensive Composite Materials, Pages 327-344, https://doi.org/10.1016/B0-08-042993-9/00060-7.

551 IRIMO, 2020. Islamic Republic of Iran Meteorological Organizationhttps (IRIMO).
552 https://www.irimo.ir/eng/index.php.

553 Jahromi, F.A., Keshavarzi, B., Moore, F., Abbasi, S., Busquets, R., Hooda, P.S. and Jaafarzadeh, N., 2021. Source
554 and risk assessment of heavy metals and microplastics in bivalves and coastal sediments of the Northern Persian
555 Gulf, Hormogzan Province. Environmental Research, 196, p.110963.
556 https://doi.org/10.1016/j.envres.2021.110963.

557 Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R. and Law, K.L., 2015.
558 Plastic waste inputs from land into the ocean. *Science*, *347*(6223), pp.768-771.
559 https://doi.org/10.1126/science.1260352.

560 Jang, M., Shim, W.J., Han, G.M., Rani, M., Song, Y.K., Hong, S.H., 2017. Widespread detection of a brominated
561 flame retardant, hexabromocyclododecane, in expanded polystyrene marine debris and microplastics from South
562 Korea and the Asia-Pacific coastal region. Environ. Pollut. 231, 785–794.
563 https://doi.org/10.1016/j.envpol.2017.08.066.

564 Johnson-Restrepo, B., Kannan, K., 2009. An assessment of sources and pathways of human exposure to
565 polybrominated diphenyl ethers in the United States. Chemosphere 76, 542–548.
566 https://doi.org/10.1016/j.chemosphere.2009.02.068.

567 Keet, C.A., Keller, J.P. and Peng, R.D., 2018. Long-term coarse particulate matter exposure is associated with
568 asthma among children in Medicaid. American journal of respiratory and critical care medicine, 197(6), pp.737-
569 746. https://doi.org/10.1164/rccm.201706-1267OC.

570 Keshavarzi, B., Tazarvi, Z., Rajabzadeh, M.A. and Najmeddin, A., 2015. Chemical speciation, human health risk
571 assessment and pollution level of selected heavy metals in urban street dust of Shiraz, Iran. *Atmospheric*
572 *Environment*, *119*, pp.1-10. https://doi.org/10.1016/j.atmosenv.2015.08.001.

573 Klein, M. and Fischer, E.K., 2019. Microplastic abundance in atmospheric deposition within the Metropolitan
574 area of Hamburg, Germany. Science of the Total Environment, 685, pp.96-103.
575 https://doi.org/10.1016/j.scitotenv.2019.05.405.

576 Kowalski, N., Reichardt, A.M. and Waniek, J.J., 2016. Sinking rates of microplastics and potential implications
577 of their alteration by physical, biological, and chemical factors. *Marine pollution bulletin*, *109*(1), pp.310-319.
578 https://doi.org/10.1016/j.marpolbul.2016.05.064.

579 Kurt-Karakus, P.B., 2012. Determination of heavy metals in indoor dust from Istanbul, Turkey: estimation of the
580 health risk. *Environment international*, *50*, pp.47-55. https://doi.org/10.1016/j.envint.2012.09.011.

581 Landrigan, P.J., Rauh, V.A. and Galvez, M.P., 2010. Environmental justice and the health of children. Mount
582 Sinai Journal of Medicine: A Journal of Translational and Personalized Medicine: A Journal of Translational and
583 Personalized Medicine, 77(2), pp.178-187. https://doi.org/10.1002/msj.20173.

584 Lehmann, A., Leifheit, E.F., Gerdawischke, M. and Rillig, M.C., 2021. Microplastics have shape-and polymer-
585 dependent effects on soil aggregation and organic matter loss–an experimental and meta-analytical approach.
586 Microplastics and Nanoplastics, 1(1), pp.1-14. https://doi.org/10.1186/s43591-021-00007-x.

587 Liao, C., Liu, F., Guo, Y., Moon, H.B., Nakata, H., Wu, Q. and Kannan, K., 2012. Occurrence of eight bisphenol
588 analogues in indoor dust from the United States and several Asian countries: implications for human
589 exposure. *Environmental science & technology*, *46*(16), pp.9138-9145. https://doi.org/10.1021/es302004w.

590 Liebezeit, G. and Liebezeit, E., 2015. Origin of synthetic particles in honeys. *Polish Journal of Food and Nutrition*
591 *Sciences*, *65*(2). https://doi.org/10.1515/pjfns-2015-0025.

592 Liu, C., Li, J., Zhang, Y., Wang, L., Deng, J., Gao, Y., Yu, L., Zhang, J. and Sun, H., 2019a. Widespread
593 distribution of PET and PC microplastics in dust in urban China and their estimated human exposure. *Environment*
594 *international*, *128*, pp.116-124. https://doi.org/10.1016/j.envint.2019.04.024.

595 Liu, K., Wang, X., Fang, T., Xu, P., Zhu, L. and Li, D., 2019b. Source and potential risk assessment of suspended
596 atmospheric microplastics in Shanghai. *Science of the total environment*, *675*, pp.462-471.
597 https://doi.org/10.1016/j.scitotenv.2019.04.110.

598 Ljung, K., Selinus, O., Otabbong, E. and Berglund, M., 2006. Metal and arsenic distribution in soil particle sizes
599 relevant to soil ingestion by children. *Applied Geochemistry*, *21*(9), pp.1613-1624.
600 https://doi.org/10.1016/j.apgeochem.2006.05.005.

601 Löder, M.G. and Gerdts, G., 2015. Methodology used for the detection and identification of microplastics—A
602 critical appraisal. *Marine anthropogenic litter*, pp.201-227. https://doi.org/10.1007/978-3-319-16510-3_8.

603 McCormick, A., Hoellein, T.J., Mason, S.A., Schluep, J. and Kelly, J.J., 2014. Microplastic is an abundant and
604 distinct microbial habitat in an urban river. *Environmental science & technology*, *48*(20), pp.11863-11871.
605 https://doi.org/10.1021/es503610r.

606 Mertoglu-Elmas, G., 2017. The effect of colorants on the content of heavy metals in recycled corrugated board
607 papers. *BioResources*, *12*(2), pp.2690-2698. https://doi.org/10.15376/biores.12.2.2690-2698.

608 Narmadha, V.V., Jose, J., Patil, S., Farooqui, M.O., Srimuruganandam, B., Saravanadevi, S. and Krishnamurthi,
609 K., 2020. Assessment of microplastics in roadside suspended dust from urban and rural environment of Nagpur,
610 India. International Journal of Environmental Research, 14(6), pp.629-640. https://doi.org/10.1007/s41742-020-
611 00283-0.

612 Nematollahi, M.J., Dehdaran, S., Moore, F. and Keshavarzi, B., 2021b. Potentially toxic elements and polycyclic
613 aromatic hydrocarbons in street dust of Yazd, a central capital city in Iran: contamination level, source
614 identification, and ecological–health risk assessment. *Environmental geochemistry and health*, *43*(1), pp.485-519.
615 https://doi.org/10.1007/s10653-020-00682-4.

616 Nematollahi, M.J., Keshavarzi, B., Moore, F., Esmaeili, H.R., Saravi, H.N. and Sorooshian, A., 2021a.
617 Microplastic fibers in the gut of highly consumed fish species from the southern Caspian Sea. Marine Pollution
618 Bulletin, 168, p.112461. https://doi.org/10.1016/j.marpolbul.2021.112461.

619 Nematollahi, M.J., Moore, F., Keshavarzi, B., Vogt, R.D., Saravi, H.N. and Busquets, R., 2020. Microplastic
620 particles in sediments and waters, south of Caspian Sea: frequency, distribution, characteristics, and chemical
621 composition. Ecotoxicology and Environmental Safety, 206, p.111137.
622 https://doi.org/10.1016/j.ecoenv.2020.111137.

623 Nezhad, M.T.K., Tabatabaii, S.M., Gholami, A., 2015. Geochemical assessment of steel smelter-impacted urban
624 soils, Ahvaz, Iran. J. Geochem. Explor. 152, 91–109. https://doi.org/10.1016/j.gexplo.2015.02.005.

625 Nuelle, M.T., Dekiff, J.H., Remy, D. and Fries, E., 2014. A new analytical approach for monitoring microplastics
626 in marine sediments. *Environmental Pollution*, *184*, pp.161-169. https://doi.org/10.1016/j.envpol.2013.07.027.

627 O'Brien, S., Okoffo, E.D., Rauert, C., O'Brien, J.W., Ribeiro, F., Burrows, S.D., Toapanta, T., Wang, X. and
628 Thomas, K.V., 2021. Quantification of selected microplastics in Australian urban road dust. Journal of Hazardous
629 Materials, 416, p.125811. https://doi.org/10.1016/j.jhazmat.2021.125811.

630 Ogata, Y., Takada, H., Mizukawa, K., Hirai, H., Iwasa, S., Endo, S., Mato, Y., Saha, M., Okuda, K., Nakashima,
631 A. and Murakami, M., 2009. International Pellet Watch: Global monitoring of persistent organic pollutants (POPs)
632 in coastal waters. 1. Initial phase data on PCBs, DDTs, and HCHs. *Marine pollution bulletin*, *58*(10), pp.1437-
633 1446. 1437–1446. https://doi.org/10.1016/j.marpolbul.2009.06.014.

634 Ogilo, J. K., Onditi, A. O., Salim, A. M., Yusuf, A. O., 2017. Assessment of levels of heavy metals in paints from
635 interior walls and indoor dust from residential houses in Nairobi City County, Kenya.
636 https://doi.org/10.9734/CSJI/2017/37392.

637 Patchaiyappan, A., Dowarah, K., Ahmed, S.Z., Prabakaran, M., Jayakumar, S., Thirunavukkarasu, C. and
638 Devipriya, S.P., 2021. Prevalence and characteristics of microplastics present in the street dust collected from
639 Chennai metropolitan city, India. Chemosphere, 269, p.128757.
640 https://doi.org/10.1016/j.chemosphere.2020.128757.

641 Pauly, J.L., Stegmeier, S.J., Allaart, H.A., Cheney, R.T., Zhang, P.J., Mayer, A.G. and Streck, R.J., 1998. Inhaled
642 cellulosic and plastic fibers found in human lung tissue. *Cancer Epidemiology and Prevention Biomarkers*, *7*(5),
643 pp.419-428.

644 Prata, J.C., da Costa, J.P., Duarte, A.C. and Rocha-Santos, T., 2019. Methods for sampling and detection of
645 microplastics in water and sediment: A critical review. *TrAC Trends in Analytical Chemistry*, *110*, pp.150-159.
646 https://doi.org/10.1016/j.trac.2018.10.029.

647 Rocha-Santos, Teresa Duarte, 2017. Characterization and Analysis of Microplastics. Elsevier.

Rochman, C.M., Brookson, C., Bikker, J., Djuric, N., Earn, A., Bucci, K., Athey, S., Huntington, A., McIlwraith, H., Munno, K. and De Frond, H., 2019. Rethinking microplastics as a diverse contaminant suite. Environmental toxicology and chemistry, 38(4), pp.703-711. https://doi.org/10.1002/etc.4371.

Salvi, S., 2007. Health effects of ambient air pollution in children. Paediatric respiratory reviews, 8(4), pp.275-280. https://doi.org/10.1016/j.prrv.2007.08.008.

Savoca, S., Bottari, T., Fazio, E., Bonsignore, M., Mancuso, M., Luna, G.M., Romeo, T., D'Urso, L., Capillo, G., Panarello, G. and Greco, S., 2020. Plastics occurrence in juveniles of Engraulis encrasicolus and Sardina pilchardus in the Southern Tyrrhenian Sea. Science of the Total Environment, 718, p.137457. https://doi.org/10.1016/j.scitotenv.2020.137457..

Schwartz, J., 2004. Air pollution and children's health. Pediatrics, 113(Supplement 3), pp.1037-1043.

Soltani, N.S., Taylor, M.P. and Wilson, S.P., 2021. Quantification and exposure assessment of microplastics in Australian indoor house dust. *Environmental Pollution*, p.117064. https://doi.org/10.1016/j.envpol.2021.117064.

Song, Y.K., Hong, S.H., Jang, M., Han, G.M., Jung, S.W. and Shim, W.J., 2017. Combined effects of UV exposure duration and mechanical abrasion on microplastic fragmentation by polymer type. *Environmental science & technology*, *51*(8), pp.4368-4376. https://doi.org/10.1021/acs.est.6b06155.

Stöcklin, J., 1968. Structural history and tectonics of Iran: a review. AAPG Bulletin, 52(7), pp.1229-1258.

Strady, E., Kieu-Le, T.C., Tran, Q.V. and Thuong, Q.T., 2021. Microplastic in atmospheric fallouts of a developing Southeast Asian megacity under tropical climate. Chemosphere, 272, p.129874. https://doi.org/10.1016/j.chemosphere.2021.129874.

Tanaka, K. and Takada, H., 2016. Microplastic fragments and microbeads in digestive tracts of planktivorous fish from urban coastal waters. Scientific reports, 6(1), pp.1-8. https://doi.org/10.1038/srep34351.

Teuten, E.L., Saquing, J.M., Knappe, D.R., Barlaz, M.A., Jonsson, S., Björn, A., Rowland, S.J., Thompson, R.C., Galloway, T.S., Yamashita, R. and Ochi, D., 2009. Transport and release of chemicals from plastics to the environment and to wildlife. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *364*(1526), pp.2027-2045. https://doi.org/10.1098/rstb.2008.0284.

US EPA, 2011. Exposure Factors Handbook Chapter 8: Body-Weight Studies. https://cfpub.epa.gov/si/si_public_record_report.cfm?direntryid=236252, Accessed date: 22 March 2019.

Velzeboer, I., Kwadijk, C.J., Koelmans, A.A., 2014. Strong sorption of PCBs to nanoplastics, microplastics, carbon nanotubes, and fullerenes. Environ. Sci. Technol. 48, 4869–4876. https://doi.org/10.1021/es405721v.

Vianello, A., Jensen, R.L., Liu, L., Vollertsen, J., 2019. Simulating human exposure to indoor airborne microplastics using a Breathing Thermal Manikin. Sci. Rep. 9, 8670. https://doi.org/10.1038/s41598-019-45054-w.

Wang, F., Wong, C.S., Chen, D., Lu, X., Wang, F. and Zeng, E.Y., 2018. Interaction of toxic chemicals with microplastics: a critical review. *Water research*, *139*, pp.208-219. https://doi.org/10.1016/j.watres.2018.04.003.

Yuan, W., Liu, X., Wang, W., Di, M. and Wang, J., 2019. Microplastic abundance, distribution and composition in water, sediments, and wild fish from Poyang Lake, China. *Ecotoxicology and environmental safety*, *170*, pp.180-187. https://doi.org/10.1016/j.ecoenv.2018.11.126.

Zhang, J., Wang, L. and Kannan, K., 2020b. Microplastics in house dust from 12 countries and associated human exposure. *Environment international*, *134*, p.105314. https://doi.org/10.1016/j.envint.2019.105314.

Zhang, Q., Zhao, Y., Du, F., Cai, H., Wang, G. and Shi, H., 2020a. Microplastic fallout in different indoor environments. *Environmental science & technology*, *54*(11), pp.6530-6539. https://doi.org/10.1021/acs.est.0c00087.

Zhang, Y., Kang, S., Allen, S., Allen, D., Gao, T. and Sillanpää, M., 2020c. Atmospheric microplastics: A review on current status and perspectives. *Earth-Science Reviews*, *203*, p.103118. https://doi.org/10.1016/j.earscirev.2020.103118.

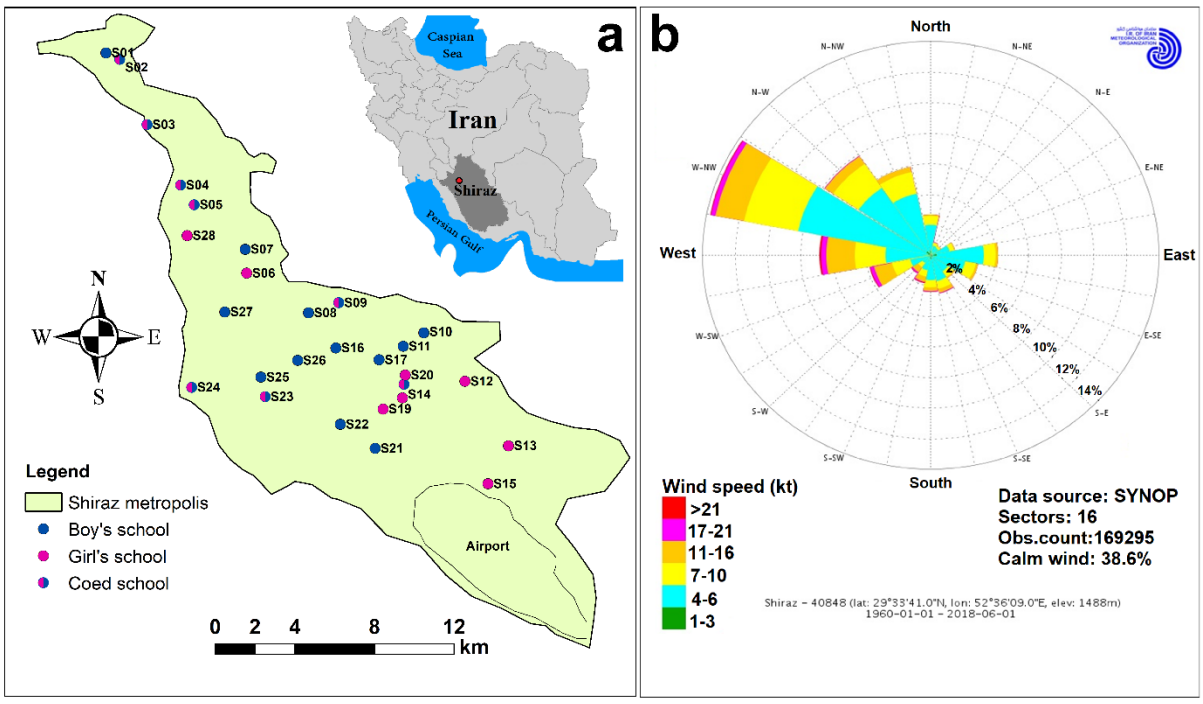

Fig. 1 a) Map illustrating the location of the research area along with the spatial distribution of the sampling sites S01-S28 and their correspondence school types, and b) rose diagram

showing the dominant wind direction in Shiraz metropolis. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.

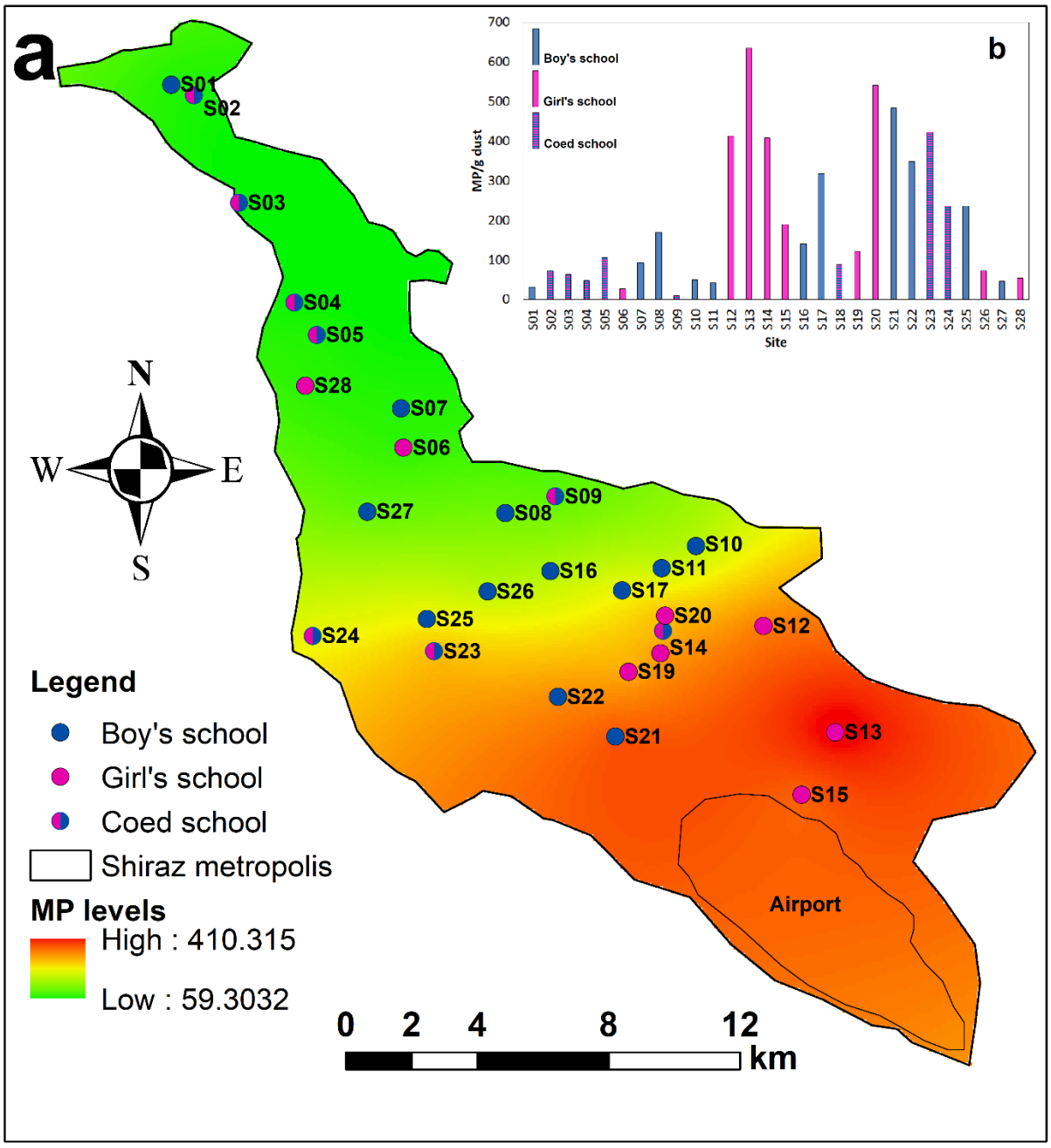

Fig. 2. a) Heat map of Shiraz including spatial distribution of MP levels in indoor dust of school in Shiraz and b) Detail of MP concentration at each sampling site.

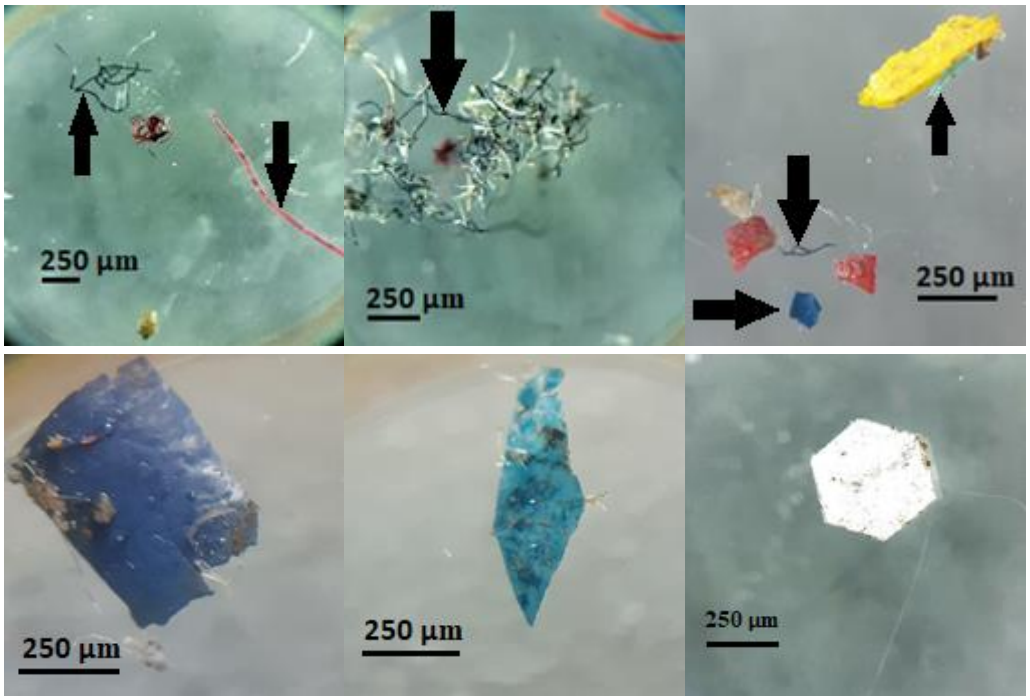

Fig. 3 Examples of MP items detected below optical microscope with optical zoom of 40X (arrows point to MP particles).

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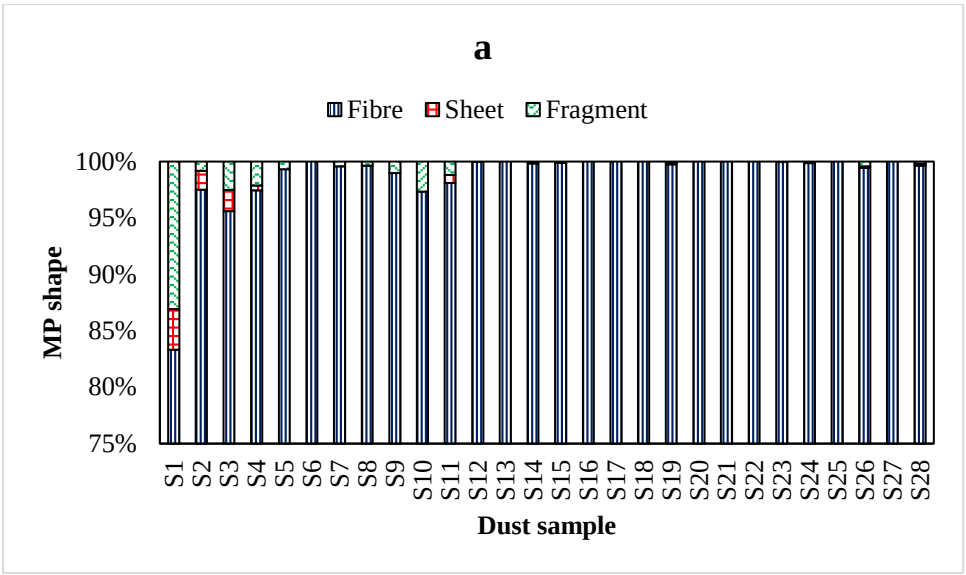

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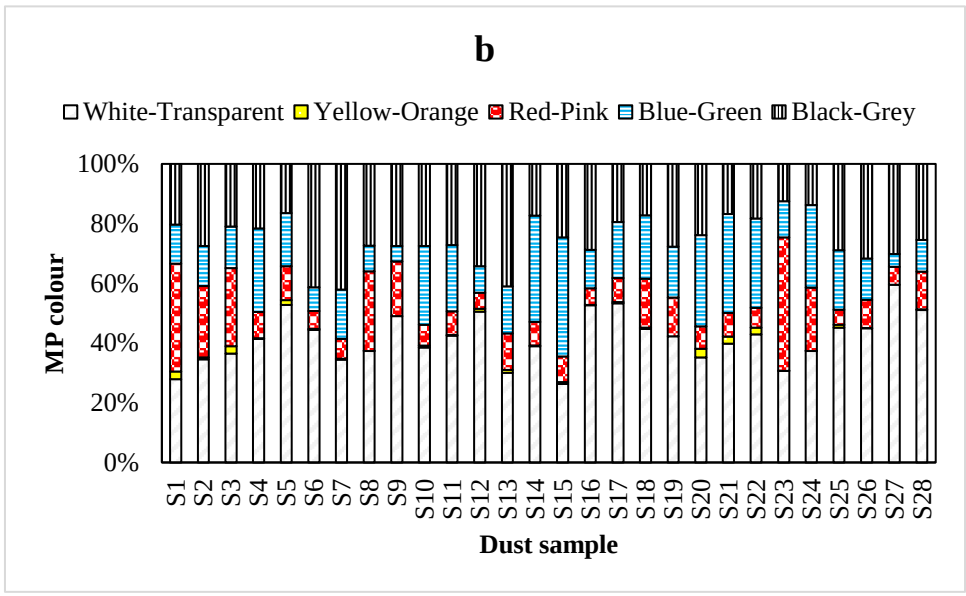

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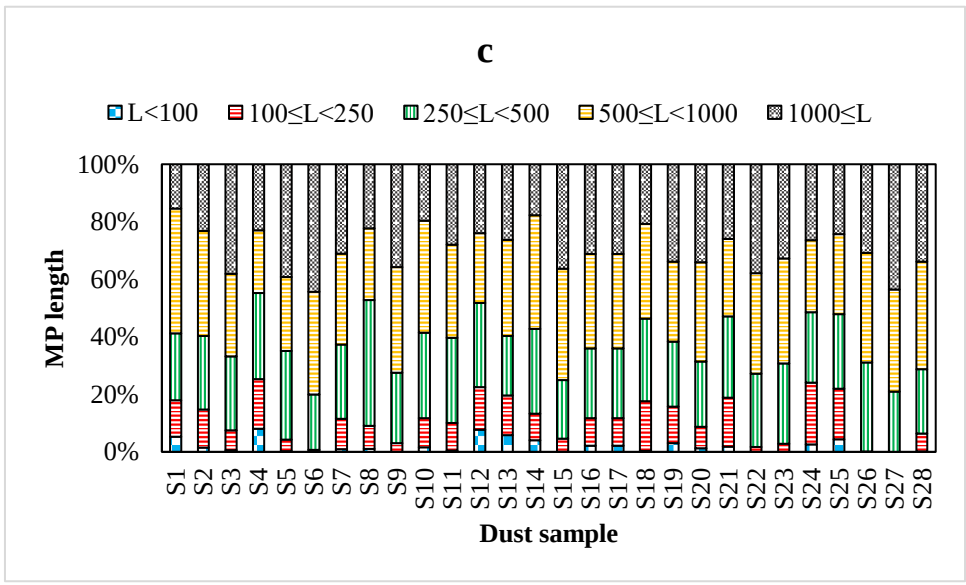

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746 **Fig. 4**. Cumulative percentage of MPs classified by a) shape, b) colour, c) size (μm) within
747 indoor dust of Shiraz's schools.

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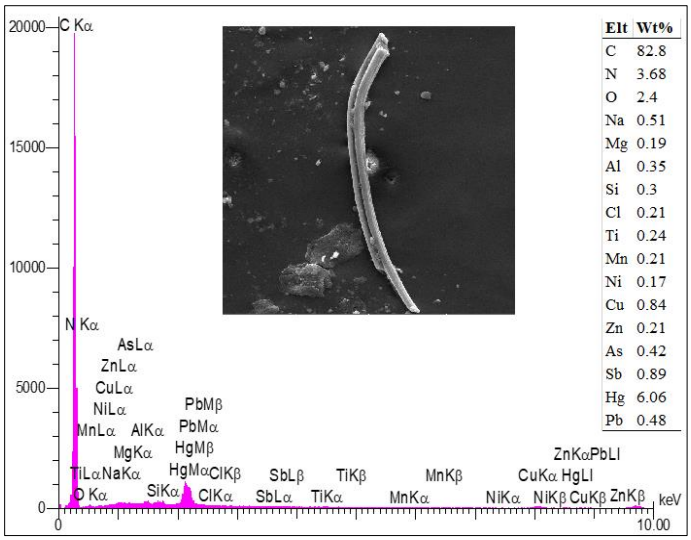

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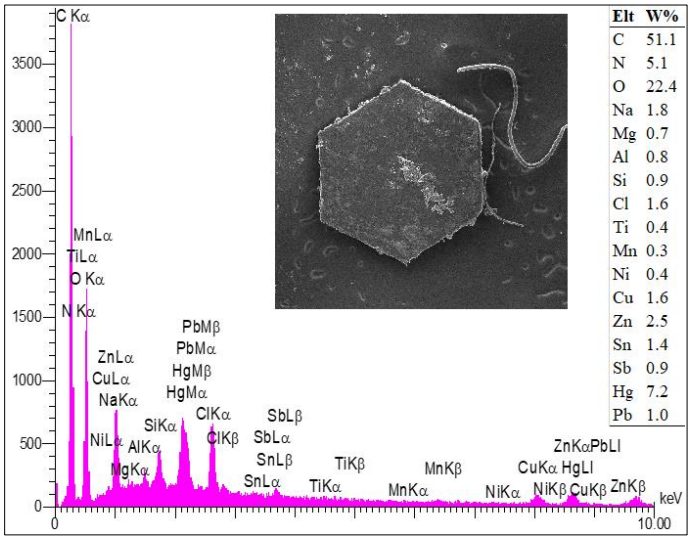

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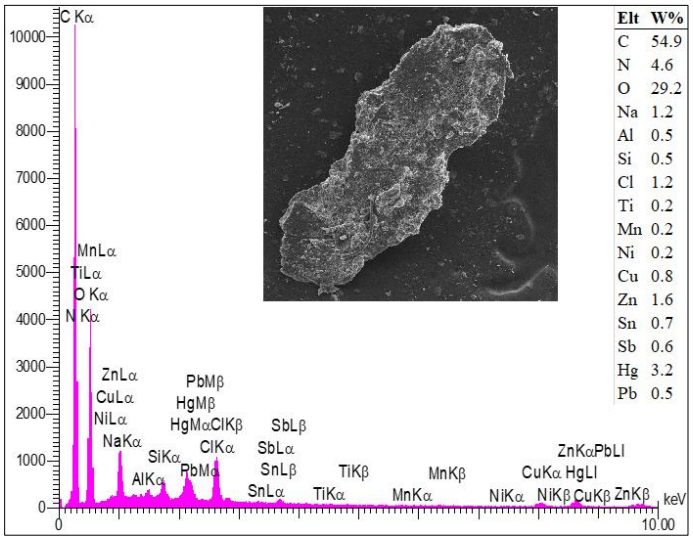

751 **Fig. 5** SEM micrographs and their corresponding EDS spectra showing the elemental composition of
752 representative MPs (with different shapes) in indoor dust of Shiraz's schools.

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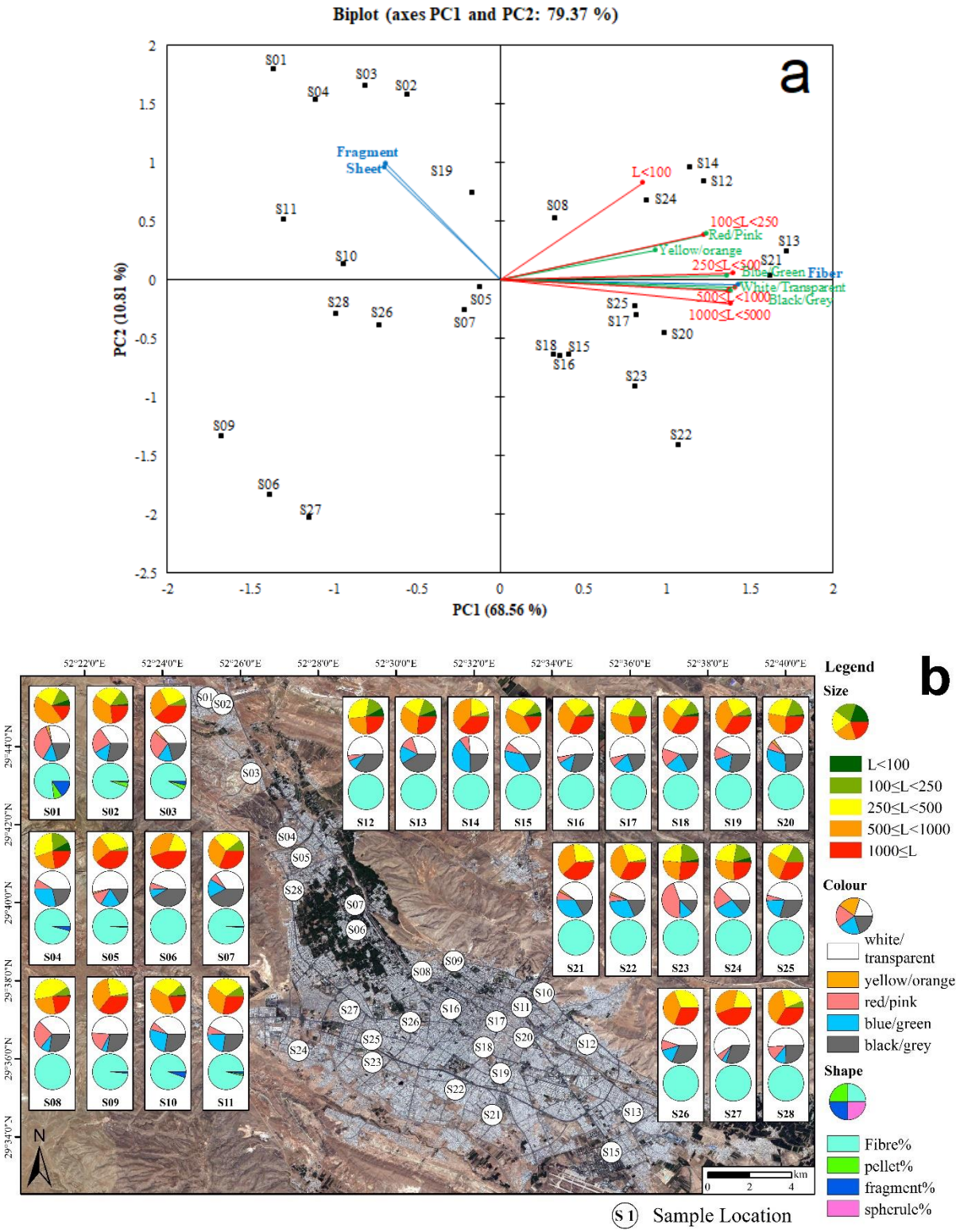

Fig. 6 PCA biplot showing interrelations of MP characteristics **(a)**, and spatial distribution of MP characteristics within indoor dust in the Shiraz's schools (sampling sites S1-28) **(b)**

762 **Table 1** Properties of MPs in indoor dust of Shiraz's schools

MPs properties	**%**	Total N. g^{-3} of dust	Min. (N)	Max. (N)	Mean (N)	Med. (N)	S.D.	C.V. (%)
All MPs	100	5463.5	9.8	634.5	195.1	113.9	179.2	90
Shape								
Fibre	99.7	54464	97	6345	1945.1	1134.0	1796.4	90
Fragment	0.2	126	0	40	4.5	1.0	8.2	180
Sheet	0.1	45	0	12	1.6	0.0	3.6	230
Color								
White-Transparent	39.7	21678	48	2084	774.2	507.0	674.7	90
Yellow-orange	1.0	571	0	157	20.4	3.5	38.0	190
Red-pink	12.9	7039	16	1891	251.4	151.5	367.2	150
Blue-green	21.9	11959	5	1658	427.1	183.0	496.5	120
Black-gray	24.5	13393	27	2600	478.3	331.5	542.0	110
Size (µm)								
50≤L<100	2.5	1396	0	370	49.9	11.5	93.4	190
100≤L<250	11.6	6051	0	919	216.1	93.5	288.2	130
250≤L<500	26.4	14215	24	1532	507.7	301.0	467.9	90
500≤L<1000	31.5	17011	36	2120	607.5	357.5	573.9	90
1000≤L	28.0	15001	35	1831	535.8	396.0	496.6	90

763 N = Number of MPs from 50µm to 1000µm

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Table 2 MP characteristics in indoor dust of the study area relative to other locations

Region	Sample type	MPs frequency	Major Polymers	Major shapes	Reference
Shiraz metropolis, Iran	School indoor dust	5464[a], 195[b] MPs/g, 10-635[c] MPs/g	PET, PP	Fibre	This study
Tehran, Iran	Urban outdoor dust	2649[a], 83±10-605±10[c] MPs/30g		Fragment	Dehghani et al., 2017
Bushehr City, Iran	Urban outdoor dust	744.8[b], 210-1658[c] MPs/10g		Fibre	Abbasi et al., 2017
Asaluyeh county, Iran	Urban outdoor dust	13132/15S, 900[b] MPs/15g		Bead	Abbasi et al., 2019
39 major China's city	Indoor PET MPs	1550 - 120000 mg/kg	PET	Fibre	Liu et al., 2019a
	Indoor PC MPs	NI - 107 mg/kg	PC	Fibre	
Normal University, Shanghai, China	Dormitory dust	9.9 × 103 MPs/m^2/day	PES, RY	Fibre	Zhang et al., 2020a
	Office indoor dust	1.8 × 103 MPs/m^2/day	PES, RY	Fibre	
	Corridor dust	1.5 × 103 MPs/m^2/day	PES, RY	Fibre	
House dust of 12 countries	House indoor dust PET MPs	38–120,000 µg/g	PET	Fibre	Zhang et al., 2020b
	House indoor dust PC MPs	<1–1700 µg/g	PC	Fibre	
Australia	House indoor dust	7483[a]	PE, PET, PA, PV	Fibre	Soltani et al., 2021
CSUCI, California, USA	Indoor dust	3.3±2.9 fibers & 12.6±8.0 fragments m–3	PVC, PS, PE, PET	Fragment, Fibre	Gaston et al., 2020
Paris, France	Indoor air	1–60 MPs/m-3	RY, PE, PA, PP	Fibre	Dris et al., 2017
Edinburgh, UK	Indoor air	5±3.3 – 10±4.2 MPs/m^3	PET, PUR	Fibre	Catarino et al., 2018
Aarhus, Denmark	Indoor air	9.3±5.8 MPs/m^3	PAN, PE, PES, PP, AR	Fiber, Fragment	Vianello et al., 2019
Surabaya, Indonesia	Office dust	1179[a] MPs		Fibre	
	School dust	1107[a] MPs		Fibre	
	Appartment dust	463[a] MPs		Fibre	

a= total abundance, b= mean, c= range, NI= Not-Identified, PC= polycarbonate MP, PES= polyester, RY= Rayon, PA= polyamide, PV= polyvinyl, PUR= polyurethane, PAN= polyacrylonitrile, AR= acrylic resin

Table 3 Statistical summary of EDI[a] values for fibres, sheets, fragments and other non-fibrous MP particles, for the different age groups. The EDI values were calculated as described by Liu et al., 2019a)[a].

Age group	MP type	Min.	Max.	Mean	Percentile		
					5th	50th	95th
Infant[b]	Fibre	0.683	44.669	13.694	1.183	7.983	41.710
Toddlers[c]	Fibre	0.161	10.533	3.229	0.279	1.882	9.835
Children[d]	Fibre	0.080	5.203	1.595	0.138	0.930	4.858
Teenagers[e]	Fibre	0.032	2.094	0.642	0.055	0.374	1.955
Adults[f]	Fibre	0.027	1.777	0.545	0.047	0.318	1.659
Infant	Sheet	0.000	0.084	0.011	0.000	0.000	0.084
Toddlers	Sheet	0.000	0.020	0.003	0.000	0.000	0.020
Children	Sheet	0.000	0.010	0.001	0.000	0.000	0.010
Teenagers	Sheet	0.000	0.004	0.001	0.000	0.000	0.004
Adults	Sheet	0.000	0.024	0.003	0.000	0.000	0.024
Infant	Fragment	0.000	0.282	0.032	0.000	0.007	0.206
Toddlers	Fragment	0.000	0.066	0.007	0.000	0.002	0.048
Children	Fragment	0.000	0.033	0.004	0.000	0.001	0.024
Teenagers	Fragment	0.000	0.013	0.001	0.000	0.000	0.010
Adults	Fragment	0.000	0.079	0.009	0.000	0.002	0.058
Infant	Non-fibre	0.000	0.359	0.043	0.000	0.014	0.286
Toddlers	Non-fibre	0.000	0.085	0.010	0.000	0.003	0.067
Children	Non-fibre	0.000	0.042	0.005	0.000	0.002	0.033
Teenagers	Non-fibre	0.000	0.017	0.002	0.000	0.001	0.013
Adults	Non-fibre	0.000	0.101	0.012	0.000	0.004	0.080
Infant	All MPs	0.690	44.669	13.737	1.222	8.019	41.710
Toddlers	All MPs	0.163	10.533	3.239	0.288	1.891	9.835
Children	All MPs	0.080	5.203	1.600	0.142	0.934	4.858
Teenagers	All MPs	0.032	2.094	0.644	0.057	0.376	1.955
Adults	All MPs	0.029	1.777	0.557	0.050	0.327	1.659

[a] the indoor exposure fraction and dust ingestion rate (mg day^{-1}) respectively were: 0.88 and 40[b]; 0.79 and 40[c]; 0.79 and 30[d]; 0.88 and 20[e,f] .

Science of the Total Environment

A first study of microplastic occurrence in settled indoor dust in schools. A case study based in Shiraz

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Table S1 Comprehensive information on each site sampling site

Site	Longitude	Latitude	MP/g	School type	Gender	Address
S01	637253	3292398	30.6	Elementary school	B	Golestan Town
S02	637859	3292085	72.4	Elementary school	G/B	Golestan Town
S03	639089	3288837	63.6	Elementary school	G/B	Beheshti Town
S04	640597	3285836	47.1	Elementary school	G/B	Sanaye Sq.
S05	641206	3284861	105.7	Elementary school	G/B	Goldasht-E-Ma'aliabad
S06	643543	3281476	26.6	High school	G	Ghasredasht Sq.
S07	643466	3282664	92.5	High school	B	Shahed Blvd.
S08	646276	3279526	169.9	High school	B	Chamran Blvd.
S09	647593	3280051	9.8	Elementary school	G/B	Eram Sq.
S10	651332	3278595	48.8	Middle school	B	Haft Tanan Blvd.
S11	650435	3277909	41.9	Elementary school	B	Azadi Blvd.
S12	653162	3276200	412.3	Middle school	G	Nasr Blvd.
S13	655108	3273003	634.5	High school	G	Forsat-E-Shirazi Sq.
S14	650439	3275338	408.3	Elementary school	G	Zamzam junction
S15	654243	3271095	189.1	High school	G	Modarres Blvd.
S16	647490	3277790	140.2	High school	B	Mollasadra street
S17	649392	3277224	318	High school	B	Zand Blvd.
S18	650498	3276013	87.7	Elementary school	G/B	Darvazeh Kazeroun
S19	649600	3274763	122.1	High school	G	Sibouyeh Blvd.
S20	650545	3276476	541.1	Middle school	G	Valiasr Sq.
S21	649279	3272795	483.9	Elementary school	B	Navvab-E-Safavi street
S22	647737	3273978	348.2	Elementary school	B	Pansad Dastgah
S23	644434	3275315	422.9	Elementary school	G/B	Seraj street
S24	641213	3275742	236	Elementary school	G/B	Valfajr Town
S25	644228	3276291	236.3	Elementary school	B	Amir Kabir Blvd.
S26	645826	3277146	73	High school	B	Havabord junction
S27	642610	3279522	46.4	Elementary school	B	Pasdaran Blvd.
S28	640916	3283321	54.6	High school	G	Farhangshahr

MP/g = number of MPs found per gram of dust, B = Boy, G = Girl

832 **Table S2** Polymer type of twenty representatives of MPs with various colours and shapes.

MP shape	Color	Polymer
Fibrous	White	PP
Fibrous	White	PP
Fibrous	White	PET
Fibrous	White	PET
Fibrous	White	PET
Fibrous	Black	PS
Fibrous	Black	PET
Fibrous	Red	PP
Fibrous	Red	PP
Fibrous	Red	PET
Fibrous	Red	PET
Fibrous	Blue	PP
Fibrous	Blue	PP
Fibrous	Blue	PET
Fibrous	Blue	PS
Non-Fibrous	Red	NYL
Non-Fibrous	Red	PET
Non-Fibrous	Red	PET
Non-Fibrous	Blue	PP
Non-Fibrous	Blue	PET

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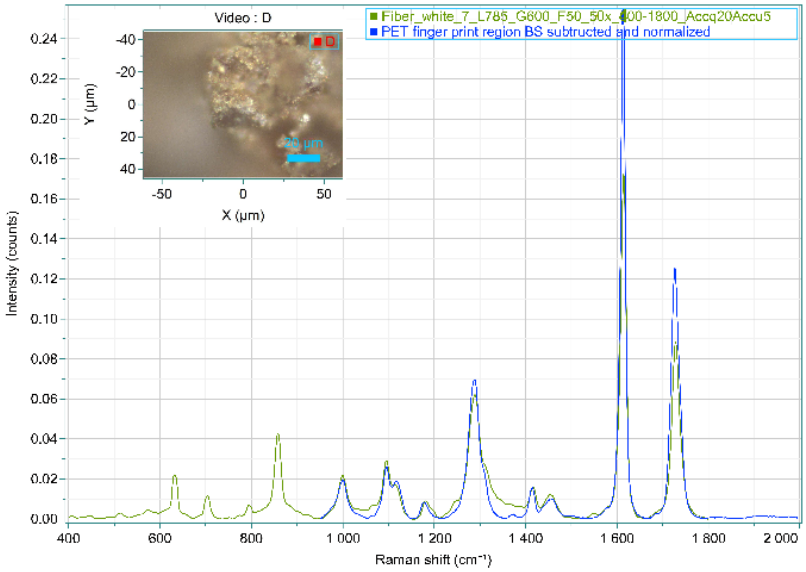
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Intensity (counts)
Raman shift (cm⁻¹)
X (µm)
Y (µm)

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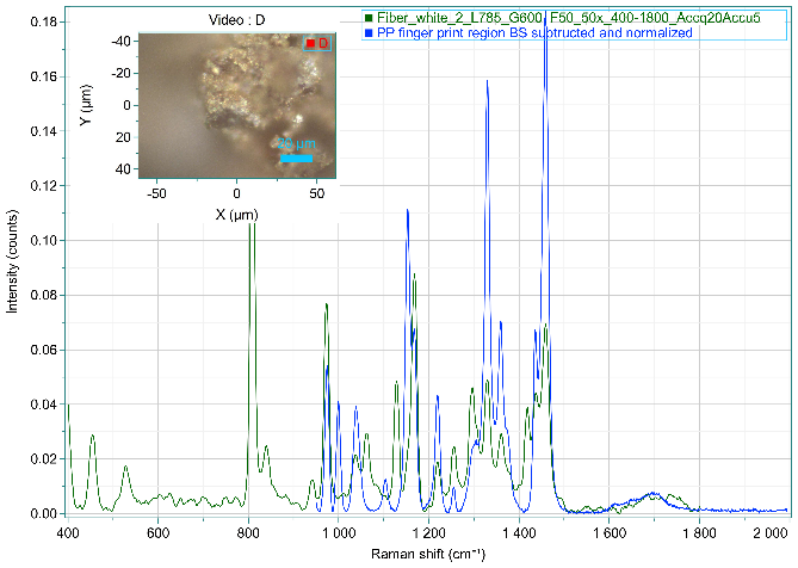
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PP finger print region BS subtructed and normalized
Intensity (counts)
Raman shift (cm⁻¹)
X (µm)
Y (µm)

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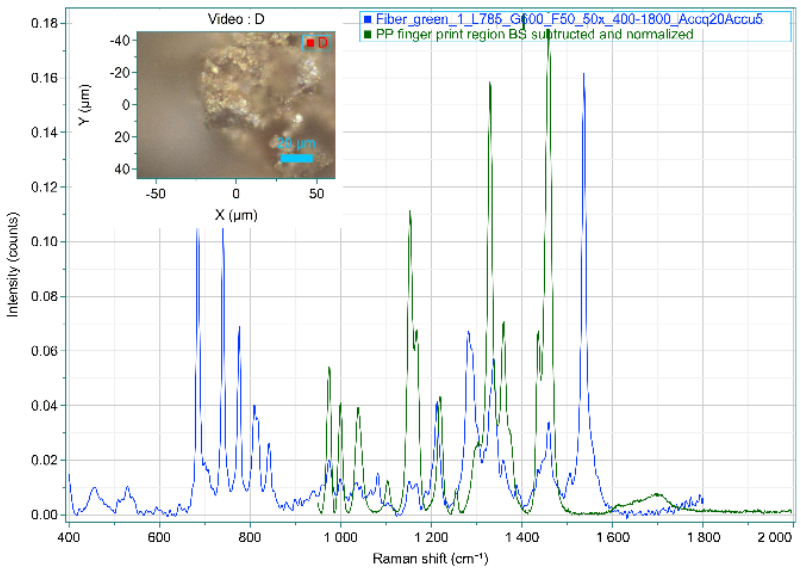
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Intensity (counts)
Raman shift (cm⁻¹)
X (µm)
Y (µm)

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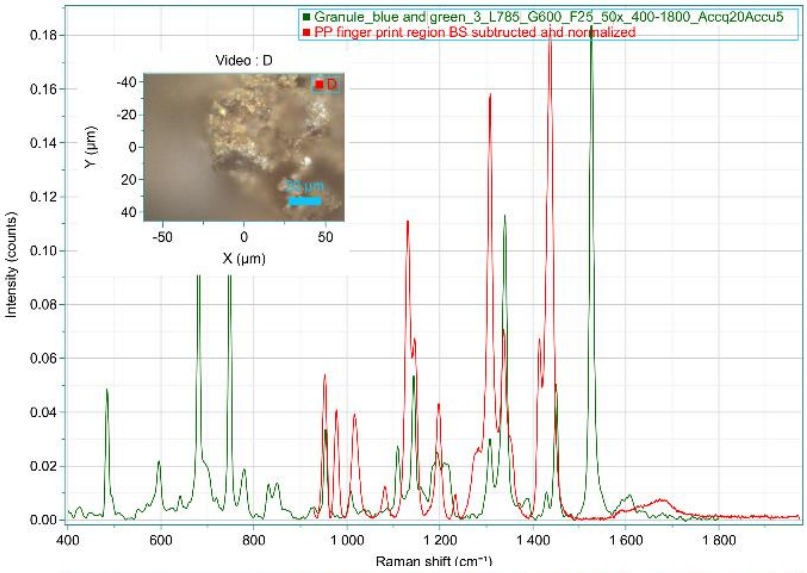

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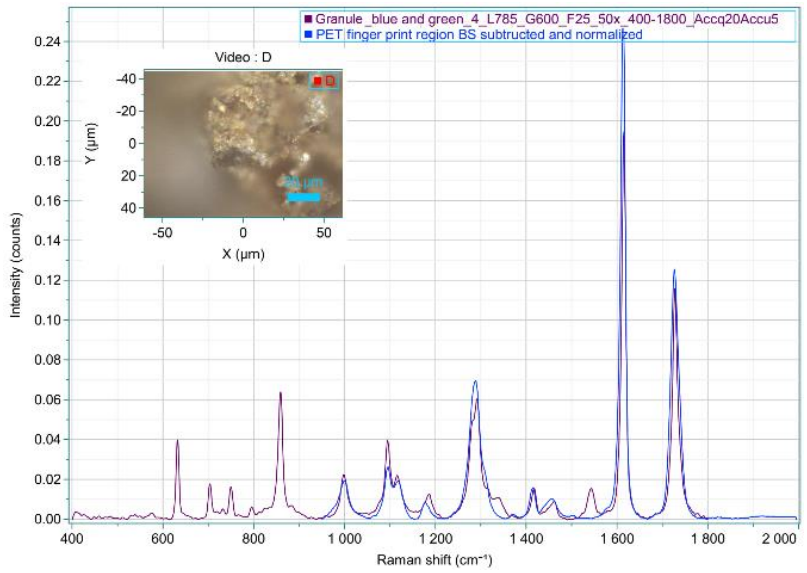

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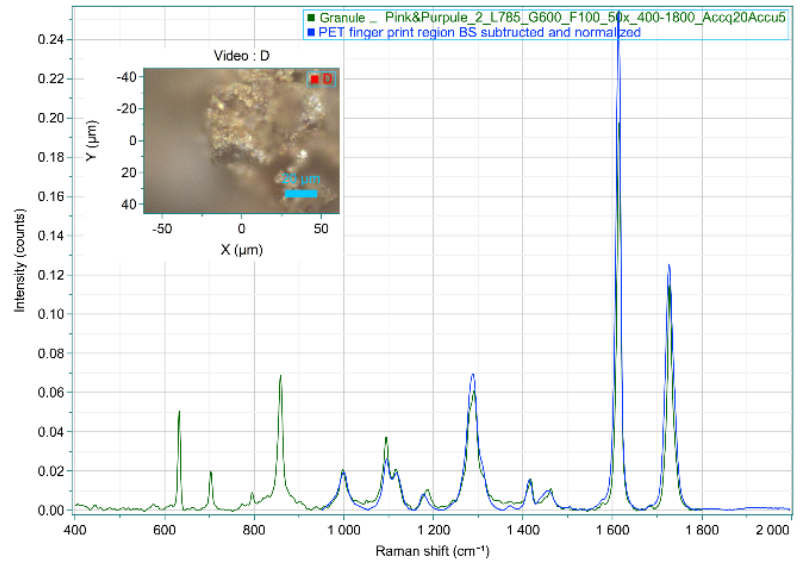

841 **Fig. S1** Raman spectra showing polymer type of representative MP items in indoor dust of Shiraz's
842 schools.

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