Microplastics in industrial and urban areas in South-West Iran

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Keywords

Microfiber; Polyethylene terephthalate; Polypropylene; Polystyrene; Nylon; Microplastic intake

Abstract

Plastics and their degradation products, microplastics (MPs), are ubiquitous in the environment, and urban and industrial soils may be the most polluted soils by this type of pollution. There is scarce information addressing human exposure to MPs from urban and industrial soils. This study assesses MP pollution in urban and industrial soils of two county capital cities in Iran, i.e. Abadan and Khorramshahr. The concentration of MPs ranged from 17 MPs/100 g industrial soil to 347 MPs/100 g urban soil, with a mean abundance of 122 MPs/100 g soil. Residential areas were the most contaminated by MPs among all the sampling sites. There, microfiber MPs, red and black, were the most abundant MPs, and it was also the case across sampling sites. Polyethylene terephthalate and polypropylene were the most common polymers among the investigated MPs, while nylon and polystyrene MPs were also present. Regarding MP sizes, those in the fraction 250–500 μm prevailed across all investigated sites. Estimated exposures to MPs through contact with the urban and industrial soils, for a normal scenario according to US EPA parameters, would lead to a daily intake < 1 MP for both adult and children, and their respective yearly intake of MPs is estimated in 6–127 and 12–253 MPs. Further research is needed to assess the health impact of current MP types and levels found in urban soil.

1. Introduction

Plastic waste is generated at a rate approaching 400 Mt year−1 (Chamas et al. 2020). The versatility of polymers accounts for continued yearly production growth (Geyer et al. 2017). Plastic products are found in households, agricultural and industrial products in view of their low cost (Rolsky et al. 2020; van den Berg et al. 2020; Yao et al. 2021; Yuan et al. 2022). However, there is now a

growing environmental concern regarding the persistence of plastics in the environment (Haider et al. 2019), and while the use of plastic waste for energy generation and recycling is increasing (PlasticsEurope 2016), some plastics are discarded in landfills or enter the environment (Cole et al. 2011). Mass production and the durability of plastic greatly contribute to their accumulation in different environmental compartments (Nel et al. 2018)

Plastics are now regarded as an indicator of environmental pollution (Waters et al. 2016). Despite plastic's prolonged stability, weathering (abiotic and biotic) degrades plastic. Plastic fragments generated from degradation are commonly referred to as secondary microplastics (MPs) and are defined as fragments between 1 µm and 5 mm in diameter (Sommer et al. 2018; Frias and Nash 2019). Besides secondary MPs, primary MPs are also manufactured as raw materials for making other products (Wagner et al. 2014). During the production of plastic, additives such as initiators, catalysts, stabilizers, flame retardants and pigments are used to regulate the materials attributes and make them suitable for their intended use (Roes et al. 2012; Galloway 2015). Since the additives are not fixed in the plastic matrix, they can be released from the polymer into the surrounding compartments, including water bodies, air, food or body tissues due to their low molecular weight (Galloway 2015). Hence, the degradation of plastic is a threat to the environment due to the leaching of toxic chemicals (Webb et al. 2013) and also due to the release of secondary microplastics, which may even be further degraded (Mateos-Cárdenas et al. 2020). Furthermore, metals and organic contaminants can be adsorbed onto the MPs surface due to the relatively large surface/volume ratio (Ashton et al. 2010; Frias et al. 2010; Karapanagioti and Klontza 2008), and some MPs are transferred through the food chain (Dubaish and Liebezeit 2013; Brennecke et al. 2015; O'Connor et al. 2020). This is a problem in itself, but MPs' toxicity can be exacerbated by

the possible presence of pollutants adsorbed from water. Thus, it is crucial to determine the fate and behaviour of MPs in the environment.

Research monitoring MPs has mainly focused on the marine environment (Carbery et al. 2018; Choy et al. 2019; Galloway and Lewis 2016). However, the fate of MPs in river water and soil is less known (Driedger et al. 2015; Rillig 2012; Steinmetz et al. 2016). The limited understanding regarding the fate of MPs in soil requires much more attention. Specifically, given the incidence of large discarded plastic items in urban and industrial soils, they should be given a priority in MP monitoring studies (Rillig 2012; Bläsing and Amelung 2018).

MP accumulation in soils may decrease soil fertility, have some impact on soil organisms and therefore change the soil ecological role and hence food production (Science Communication Unit 2013; Wuana and Okieimen 2011). Vertical transport of MPs in soil profile is affected by digging organisms; soil ingestion by fauna living in the soil (especially anecic earthworms); agricultural practices (ploughing and harvesting); soil cracking, porosity and movement of soil caused by root elongation. Horizontal distribution can be facilitated by animal hunting activities, movement of epigenic earthworms or agricultural practices (Gabet et al. 2003; Rillig et al. 2017; Guo et al. 2020). Moreover, MPs migration is commonly affected by the plastic type, as microbeads and microfibers demonstrate different interaction with soil aggregation, which in turn may influence the transport of MPs in soil. MP transport through the soil may also be affected by plastic surface properties, which become altered during its degradation (Guo et al. 2020). As other contaminants, MPs can enter the human food chain via adhesion to the surfaces of different edible vegetables and roots (Science Communication Unit 2013; Wuana and Okieimen 2011). They may be ingested by livestock and be carried into the human food chain (Huerta Lwanga et al. 2017). Also children

between 18 months and 2 years of age may ingest large quantities of soil because of hand-to-mouth behaviour (Hong et al. 2016).

This study will characterize MP pollution in urban and industrial soils from two populous cities in Iran and will include an assessment of the level of exposure of citizens to MPs. This study will focus on Abadan and Khorramshahr as two neighbouring cities with similar environmental conditions and no previous records of MP pollution.

2. Materials and Method

2.1. Study area and sample collection

The study was carried out in Abadan and Khorramshahr cities, located in Khuzestan province, South-West Iran. Khorramshahr (30° 26″ 21′ N, 48° 10″45′ E; 3 m a.s.l.) is an inland port city located ~ 10 km north of Abadan (Fig. 1). Its municipal population is 164,797 inhabitants (2020). Abadan city is located next to the Iraq–Iran border and lies in Abadan Island (68 kms long, 3–19 kms wide). Abadan city is located 53 km south of the Persian Gulf. Abadan is surrounded by the Arvand River to the West and the Bahmanshir branch of the Karoon River to the East. The ambient temperature of the study area reaches up to 52 °C in summer, and it has arid and hot climate (Ghavidel Rahimi and Ahmadi 2015). In contrast, winters are short and mild and annual rainfall is 200 mm. The relative humidity in the study area varies between 50 and 80%, throughout the year. In recent years, Abadan and Khorramshahr cities have been adversely affected by air pollution and dust storms. The annual particulate matter (PM10) mean concentrations reached 169, 187 and 201 µg/m3 in 2014, 2015 and 2016, respectively (Momtazan et al. 2019).

In October 2017, ten urban and industrial surface soils were collected in the study area (see S1–10 in Fig. 1). The urban soils had markedly different traffic loads (Table S1, in Supplementary

Materials, SM). Possible contaminant sources (residential, traffic and presence of industry) and different kinds of land use were the factors considered in sampling. Each sample consisted of five subsamples of 1–10 cm surface soil. Subsamples were 5 m apart. Subsamples were mixed to represent the sampling site. Extraneous materials were removed from the samples. In order to remove other coarse particles, the samples were passed through a 5-mm metal sieve.

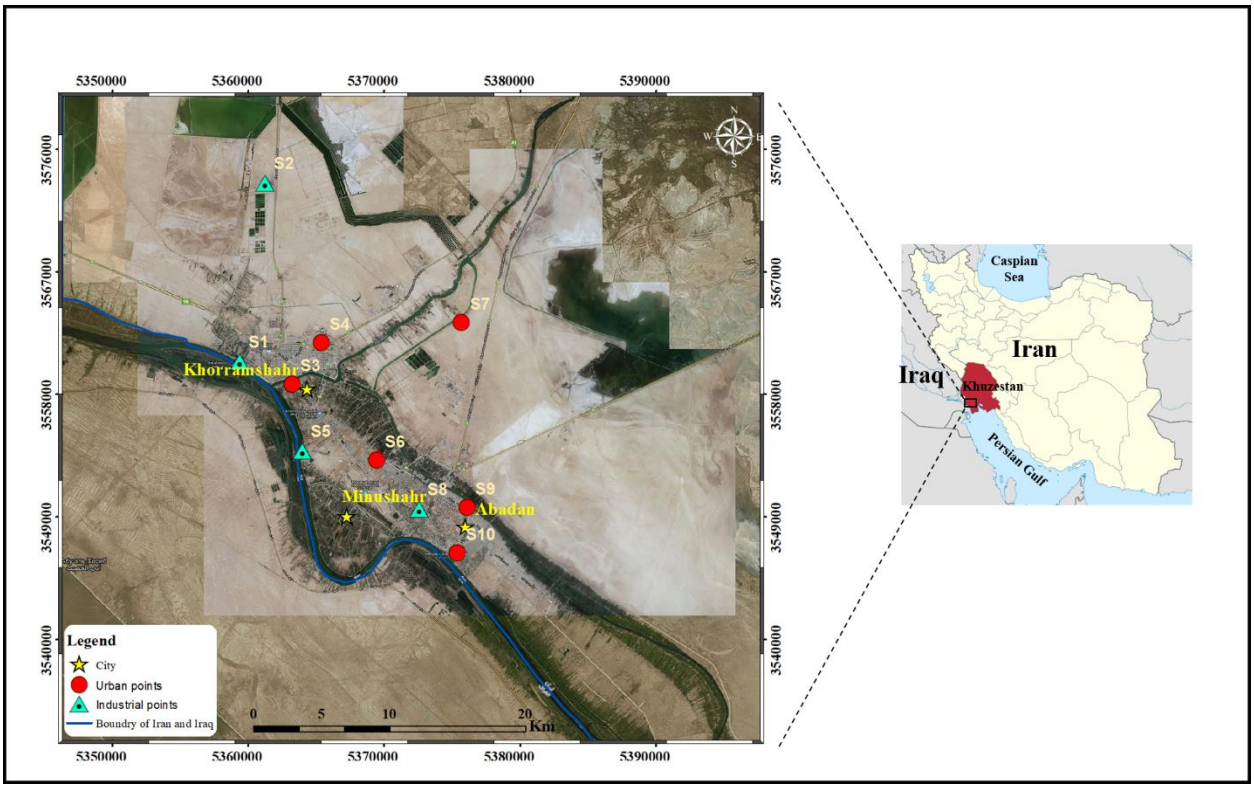


Fig. 1. Spatial distribution of surface soil samples

2.2. Sample preparation and MP particle extraction

MP extraction, purification and analysis were carried out on composite samples using density separation. Hydrogen peroxide was used to remove organic matter (Abbasi et al. 2017). Briefly, 100 g of each soil composite sample was mixed with 100 ml of 30% H2O2 until there was no more organic matter being oxidized. (This was observed by the cessation of bubbling.) This process took

up to 21 days for most soil samples. Then, S&S filter papers (blue band, grade 589/3, 2 μm pore size) were used to filter the samples using vacuum. The filtrates were dried in a sand bath at 60 °C. To separate MPs from the treated soil samples, 130 ml of NaI in water (with density of 1.68 g cm−3) was added to each sample (Zhang et al. 2018). The samples were mixed (5 min in a 300 rpm in a shaker) and let to settle for 12 h to precipitate soil particles. The supernatant of each sample was transferred to a Falcon tube and centrifuged at 5000 rpm for 5 min to separate the remaining soil particles. The supernatant was then filtered with an S&S filter paper to separate MPs from the NaI solution. To separate the MPs from the soil in the pellet, the process of mixing soil with NaI solution, followed by density separation, was repeated four times, sequentially, for each sample, to increase the recovery of MPs, and the fifth flotation did not extract any more MPs. The last step involved transferring MPs from the surface of the filters to a petri dish using a clean horsehair brush.

2.3. Microplastic detection and quantification

MPs were identified using light microscopy (Carl-Zeiss, Germany) and polarized light microscopy with an Olympus BX41-P (Olympus, Japan). The MPs were optically counted and characterized under a light microscope with a final magnification of 200. Particles < 100 μm and > 50 μm could be detected with this magnification with sufficient quality. A digital camera was connected to the microscope via a phototube, and then, the physical characteristics of MP particles including shape, lustre, hardness, elastic attribute structures and homogeneous colours were optically identified by a needle (Rocha-Santos and Duarte 2015). MPs were classified into five categories, where L indicates the length of the longest diameter: first-category ($L \leq 100$ μm); second-category ($100 < L \leq 250$ μm); third-category ($250 < L \leq 500$ μm); fourth-category ($500 < L \leq 1000$ μm); and fifth-category ($1000 < L \leq 5000$ μm). MPs were also categorized according to their shapes as

fibres, films, fragments and spherules. Considering colour, MPs were grouped as follows: blue-green, white-transparent, black-grey, yellow-orange, and red-pink. Morphological characteristics and elemental composition of selected MPs were investigated by scanning electron microscopy-energy-dispersive X-ray spectroscopy (SEM–EDX). The elemental composition of the MPs is helpful in discriminating carbon-predominant MPs. The composition of the particles was used to identify the likely MPs and reject the non-plastic ones. Before analysis, MPs were fixed on a 10 mm-diameter cylindrical SEMs tub and covered with a thin conductive gold layer. A Tescan VEGA 3 SEM with an accelerating voltage up to 10 kV (TESCAN, Czech Republic) and an Oxford Instruments X-Max 50 silicon drift detector were used. The SEM–EDX was equipped with AZtec and INCA software, respectively. A total of 10 MPs were selected for SEM–EDX analysis. A confocal Raman microscope (RS, Lab Ram HR, Horiba Japan) with a laser beam of 785 nm and Raman shift of 400–1800 cm−1 was used for the determination of the polymer composition of 9 set of selected MPs. Microscope slides covered by double-sided adhesive tapes and containing MPs were used for the analysis (Kniggendorf et al. 2019).

2.4. Quality control

Materials used in the lab for treating and in contact with the samples were protected with aluminum foil except when the samples were drying on the sand bath. Lab equipment in contact with the samples was washed and rinsed with double-distilled water before and after each procedure. . Metal and glass tools were utilized throughout the analysis. For the determination of possible presence of airborne MPs in the laboratory working space, Petri dishes which had been cleaned and inspected with the optical microscope, were left open in the laboratory during the tests and examined under the same conditions in which the MPs were extracted from the samples. During this study no MP contamination was detected in the control dishes.

2.5. Statistical analysis

Kolmogorov-Smirnov (K-S) and Shapiro-Wilk (S-W) tests were applied to investigate the normality of the data. Significant differences ($p < 0.05$) in MPs concentrations were evaluated between urban and industrial soil samples by non-parametric Mann-Whitney (M-W) U-test. Furthermore, a cluster analysis was carried out with the soil data to classify sampling sites to group them based on MPs shapes. Spearman correlation was used to determine correlation between various shapes and abundance of MPs.

2.6. Microplastic intake assessment through ingestion pathway

The daily intake of MPs was computed based on U.S. Environmental Protection Agency (2011) which estimates the mean soil particles ingestion rate at 200 mg day^{-1} for children (1 - 6 years) and 100 mg day^{-1} for adults. These rates were determined for a normal exposure scenario, whilst for an acute exposure, 1 g and 330 mg soil per day are proposed for children and outdoor or construction workers, respectively (Harris and Harper 2004). The intake was estimated using the number of microplastic particles in 100g soil and the total amount of the daily ingested soil (Eq.1), where "X" is the daily microplastic intake, "a" is the number of MPs in 100 g soil sample, "b" is the total amount of daily ingested soil recommended in normal and acute exposures.

$$X = \frac{a \times b}{100} \qquad (Eq.1)$$

3. Results and discussion

3.1. Microplastic detection in soil samples

Abadan and Khorramshahr cities constitute the so-called Arvand Free Zone, which is a long-established oil-rich area with good transportation facilities, including the Arvand and Karoon rivers, ports, railway, roads and airports. This area hosts different industries, such as the Abadan oil refinery and petrochemical complex, ports and warehouses. Heavy transit, trade and tourism are carried out on land of this zone. However, there are no available data on MPs pollution in urban and industrial soils in Iran, and the Arvand zone is a site of special significance, with numerous potential sources of MPs. MPs representative of the ones found in soil composite samples in Abadan and Khorramshahr are displayed in Fig. 2.

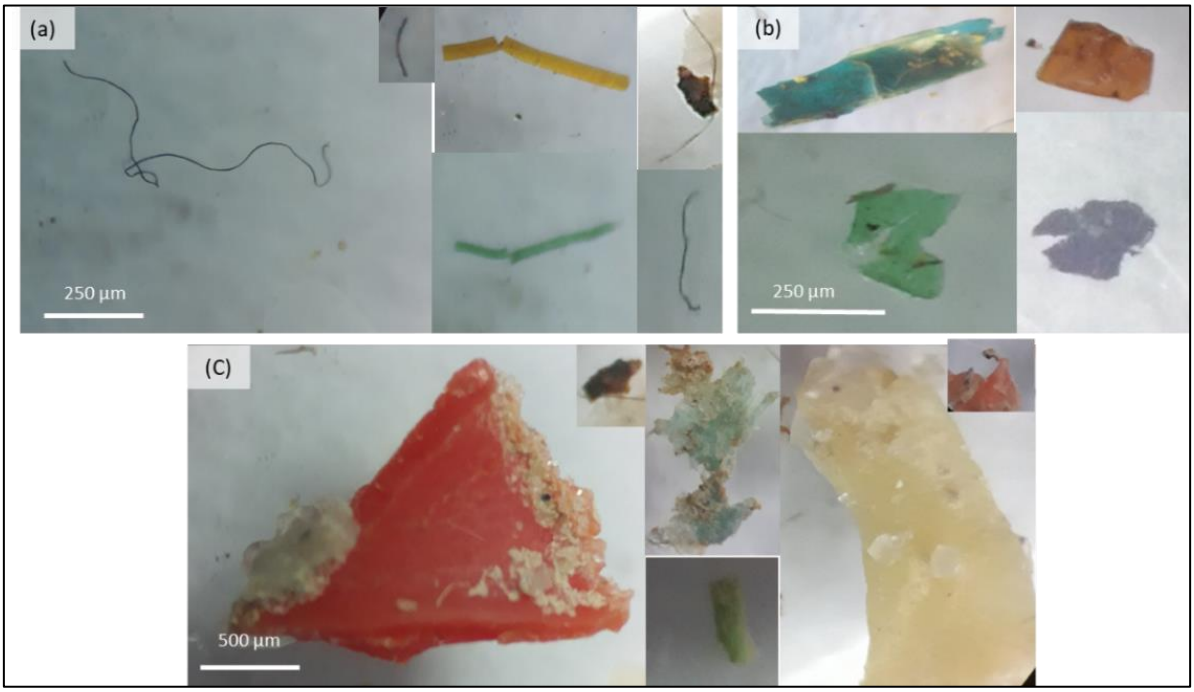


Fig. 2. Light microscopy images of various types of MPs **(a)** fibres **(b)** films **(c)** fragments.

The identification of MPs is facilitated with the colour change of MPs when rotated under polarized light (Abbasi et al. 2017). However, sometimes several other substances like wood and paper may also show similar colour changes under a polarizing microscope. Nevertheless, attention to details can help with their differentiation, and improved preparation steps can limit misidentification. Generally, materials like wood show grooved shapes and cellulose forms under optical and polarized light (Fig. S1). Despite the advantages that microscopes with polarized light offer for the identification of MPs, optical microscopy still prevails in the identification of MPs (Pan et al. 2019; Taghizadeh Rahmat Abadi et al. 2021). Figure 3 illustrates different types of MPs showing colourful shapes under polarized light.

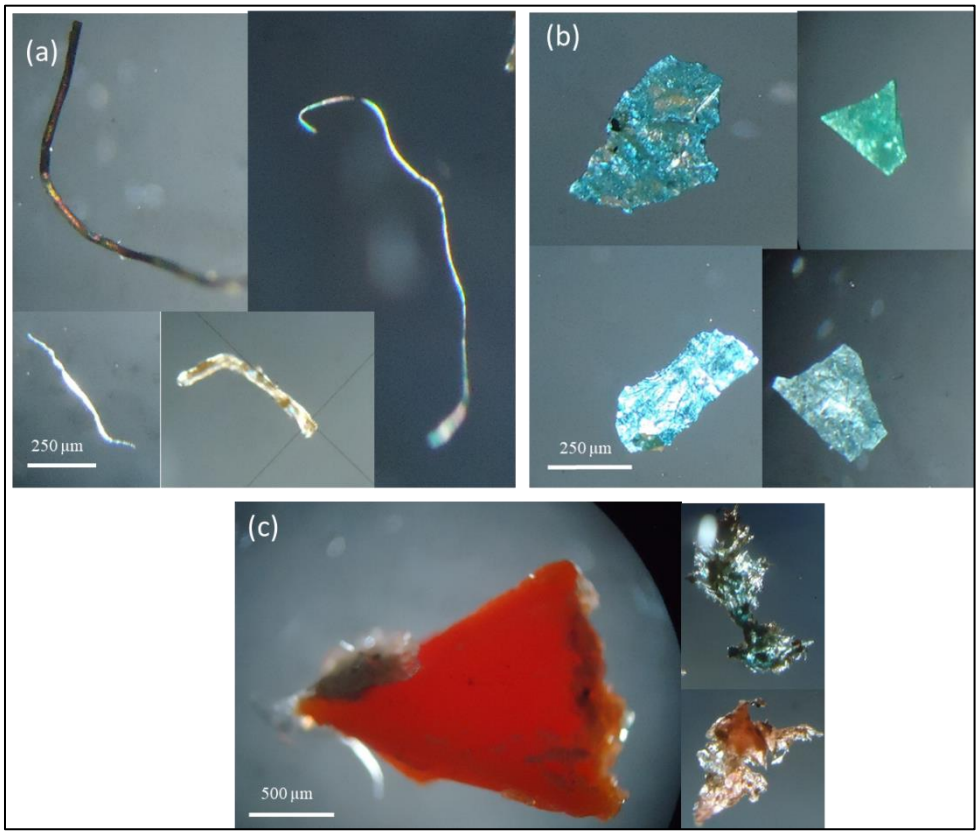


Fig. 3. Polarized light microscopy images of different types of MPs **(a)** fibres **(b)** films **(c)** fragments.

SEM analysis complemented optical microscopy. As an example, Fig. S4 illustrates selected fragments of MPs with different surface roughness. These MPs were mainly smooth, even though different levels of erosion were also observed on their surface. Mechanical and chemical weathering could have caused the observed irregular edges. This morphology may favour their capacity to absorb other contaminants. As shown in Fig S4, pits (Fig. S4a), fractures (Fig. S4a, b and c), flakes (Fig. S4c) and adhering particles (Fig. S4c) are the usual types of degradation in MPs found in the study urban and industrial soils.

The presence of MPs was evidenced by a high amount of carbon content (more than 50%) in samples, shown by energy-dispersive X-ray spectra (Tiwari et al. 2019). Carbon and oxygen are indeed main constituents of polymer materials (Ivar do Sul and Costa 2014). EDX results showed that specimens collected from the composite samples, which had properties plausible with plastics according to optical and polarized light microscopy, are mostly made of C and O together with other elements including Al, Mg, Ca, Si, K and Fe. Some of these elements could be from clay and silt (which contain Al- and Si oxides) adsorbed onto the surface of MP particles (Nematollahi et al. 2020). Part of the elements could be residues from the reagents used in the extraction and purification of the MPs (Na and I) (Fig. S4) (Abbasi et al. 2018). Some could also be additives in the polymer. For example, the improvement of physical strength and deformation resistance of thermoplastics are achieved by including inorganic additives, such as silica; furthermore, red and yellow colours in polymers are commonly achieved by Fe and Cd as inorganic pigments (Bolgar et al. 2015). Moreover, in order to increase plastic duration, Al and Si are commonly added to the polymer as antioxidant elements.

3.2. Microplastics concentration and characteristics in soil samples

A quantitative summary of MPs extracted from the soil samples in the study area are presented in Table 1. The frequency of MP particles was non-normally distributed ($p<0.05$) in the soil samples. Overall, a total of 1208 MP particles were identified in 10 soil samples. The mean and median frequency of MPs in soil samples were 122 and 64 MPs/100 g soil, respectively.

Table 1. Statistical summary of MPs frequency in soil samples from Abadan and Khorramshahr cities

	unit	Total No.	Minimum.	Maximum	Mean	Median	S. D.	RSD (%)	S. W.
MPs in soil	MPs/100g soil	1208	17	347	122	64	122	100	1.453

S.D: Standard deviation
RSD: Relative standard deviation
S.W.: Skewness

Table S1, in SM, shows the concentration of MPs in the sampling sites. The concentration of MPs presented the following decreasing order: S4 (347 MPs/100g residential soil)> S9 (338 MPs/100g residential soil)> S2 (144 MPs/100g industrial soil)> S3 (97 MPs/100g urban soil)> S7 (76 MPs/100g urban soil)> S8 (51 MPs/100g Industrial soil)> S5 (50 MPs/100g industrial soil)> S10 (49 MPs/100g urban soil)> S6 (39 MPs/100g urban soil)> S1 (17 MPs/100g industrial soil). The sampling sites are presented in Fig 1. The highest concentration of MPs was found in residential areas (Table S1). When this data is compared to the levels of MPs found in soil from other urban and non-urban areas (Table S2) it becomes apparent that urban and industrial soils are two be hotspots with MP levels in the high pollution range.

The values found here are lower than those in storm water reservoirs, where a larger MP accumulation occurs over time (e.g. 5754 MPs/100 g soil) and is also linked to urban pollution (Braga Moruzzi et al. 2020). The level of MPs at S4 is certainly high and similar to what found in sediments of the Rhine river (Germany) (Klein et al. 2015). Half of the sampling sites in this study

had MP concentrations greater than those found in river sediments (66–8 MPs/100 g) reviewed elsewhere (Braga Moruzzi et al. 2020). For instance, in tributaries to the Thames, 33.2 ± 16.1 MPs/100 g of river sediment (10 cm surface) were found in the most sewage effluent impacted site (Horton et al. 2017).

The colour, size distribution and shape of MPs in all soil samples are presented in Fig. 4. More than 96% of the MPs in S4 and 62% in S9 were fibres (Fig. 4). Fibres ranged from < 100 μm to > 1000 μm and were dominant in the study area (abundance on average across sites 61%, Fig. 4b) and mostly (70%) originate from S4 (low-density, single-family homes) and S9 (moderate-density, apartments and single-family homes). This was also confirmed by the cluster dendrogram (Fig. S2), which was used to classify MPs based on shape. Figure S2 shows that MPs from residential areas (Sites 4 and 9) were different from other sampling sites, probably due to a high percentage of MP fibres. The high fibre count may be due to the proximity of the source as torn clothes and many house furniture, which would eventually produce fibres (polyethylene-terephthalate, polyamide or polypropylene textiles, curtains, carpets, etc.) (Dris et al. 2016). Such fibres that originate from the degradation of cloth and house appliances also contribute to atmospheric fallout, as suggested from the higher number of fibres in indoor air compared with outdoor (Dris et al. 2016; Wagner and Lambert 2018). Other studies outside Iran have found that fibres are the dominant type of MP in sediments from urban rivers (Horton et al. 2017). Fibres' low-density and high adherence capacity due to their sharp tip facilitate their elevated concentration in the environment, which ends up impacting exposure to them (Ebrahimi et al. 2022).

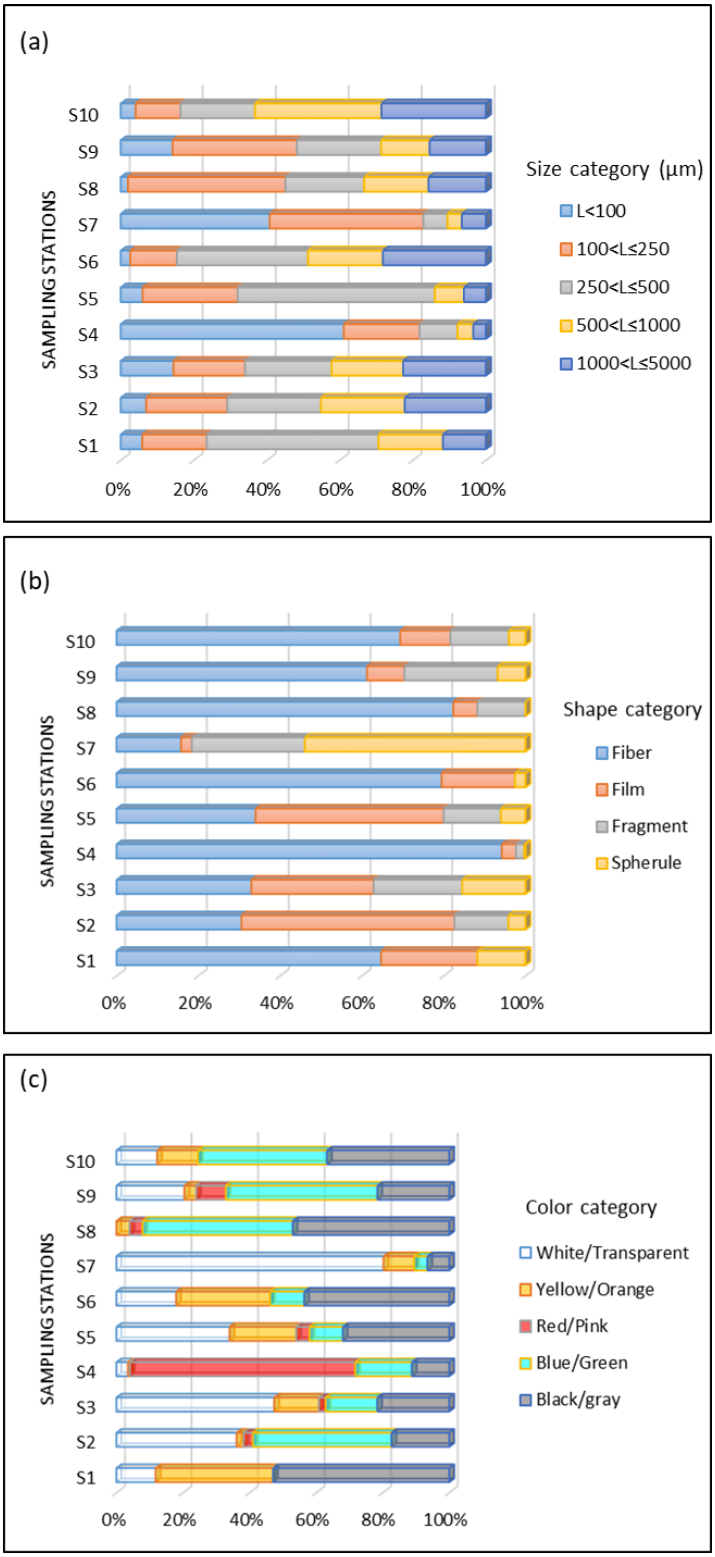
The sampling site located between the Port and Khorramshahr custom (S2), with 17 MPs/100 g soil had the lowest MP concentration in the study area. Considering different traffic loads in the

streets of the study area, S3 (Khorramshahr–Abadan junction) with high traffic load had the highest MP particle count following residential sites (97 items/100 g soil) among urban soils (Table S1). Similar results were also reported in street dust from Bushehr City in Iran (Abbasi et al. 2017). The relative standard deviation of total MPs was 100%, indicating very high heterogeneity of MP concentrations among soil samples. Furthermore, Mann–Whitney U-test revealed no significant difference between the number of MPs in the urban and industrial soil samples at $p < 0.05$.

The dominant MP particle size was 250–500 µm, ranging from 7 to 54%, and 27% on average, followed by microplastics 100–250 µm, varying between 12 and 43%, and 25% on average (Fig. 4). The < 100 µm category showed great variability across sites and had particular incidence in S4. The maximum of $500 < L \leq 1000$ and $1000 < L \leq 5000$ categories occurred in S10 with 16% on average (Fig. 4a).

The blue/green (28%, on average), white/transparent (22%, on average), red/ pink (23%, on average), black/grey (20%, on average) and yellow/orange (6%, on average) were the dominant MPs colours found in the study area (Fig. 4c). This variety of colours could be reflecting various sources of MPs. Among MP fibres, 35% comprised red/pink colour followed by black/gray colour (29%). Within the spherule particles, the relative abundance of white/transparent colour was 86%. This might indicate that they could be primary MPs. A relatively high percentage of white/transparent film MPs were also found (46%). A similar abundance was found for MP fragments with blue/green colour (43%). The application of chromatic plastic products in daily life, for instance packing and clothing, produces a great amount of plastic waste, and by subsequent decomposition, the resultant waste is coloured MPs. However, photobleaching of MPs colour may occur through exposure to sunlight during transportation in the environment (Di and Wang 2018).

Correlation between various shapes and total abundance of MPs was carried out using Spearman correlation. The results showed that MP shapes, except spherule MPs, were positively correlated with total concentration of MPs at $p < 0.05$ (0.71 for fibre, 0.64 for film and 0.68 for fragmented MPs). This implies that there is a mixture of MP shapes at different levels of pollution. However, the no-concentration trend found for spherules may point out to spherules belonging to different sources of pollution. They could be primary plastics from industrial origin for producing plastic products.



311 **Fig. 5.** Abundance and distribution of microplastics within (a) size, (b) shape, and (c) color categories

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3.3. Chemical composition of microplastic particles

The chemical composition of a set of 9 selected main types of MPs (based on their abundance in the soil samples as well as different shape and colour) was determined by Raman spectroscopy (Fig. S3). A total of 4 polymers were identified, including polypropylene (PP), polystyrene (PS), polyethylene terephthalate (PET), and nylon (NY). Specifically, red and transparent fragment MPs were made of PS, while blue fragment MPs were made of PP. The blue and green fibres were made of PP, while black and red fibres were mostly made of PET. Transparent fibres and MPs films were composed of PP and NY, respectively (Fig. S3). These types of polymers are common and have been reported in studies elsewhere, with some variations. For example, polyvinyl chloride (PVC), not identified in this study, is reported to be the main component (> 80%) in the soil samples in Sydney (Fuller and Gautam 2016). The MPs found in a coastal zone in China with high MP pollution were granules (75%) and fragments (20%) and presented a variety of compositions (Zhou et al. 2016). Furthermore, Zhou et al. (2018) demonstrated that MPs concentration in soils of a coastal beach (Shandong, China) was in the range of 1.3–14,712.5 particles kg−1 (dry weight) and (PE, PP, PS, polyether urethane were the polymer composition of MPs), probably indicating a variety of sources generating the MPs (Zhou et al. 2018). PP and PE were also the main types of MPs in soil from Shanghai farmlands (Liu et al. 2018). Styrene-butadiene (SBR) was found to be the typical composition (PE, PS and PVC) in Switzerland (Scheurer and Bigalke 2018) (Table S2).

Raman analysis of selected MPs in soils of the study area revealed that the red and black MPs were made of PET, and the blue ones were made of PP. In the present study, red and black fibre MPs were the prevailing microplastic in the soil samples (22 and 19%, respectively). Following PET, PP was also widespread (17% of the MPs) in blue fragments and fibres MPs. PET is extensively used as a polymer resin in polyester, and it has been estimated to account for relatively 64% of the

fibres recovered from the soils in the study area. The ubiquity of PET fibres in the ecosystem has already been reported in the previous research (e.g. Dris et al. 2017; Imhof et al. 2016; Mathalon and Hill 2014). PET fibres are especially widespread in sediment and soil due to their relative high density (Imhof et al. 2016; Zhang et al. 2018). Under natural conditions, these PET MPs could have resulted from degradation of PET bottles over 15 years (Ioakeimidis et al. 2016). In the study area, the 75% of PET fibres were found in residential areas (S4 and S9), and their source could be clothing, house furniture, different types of packaging containers and PET bottles.

The S9 study area displayed high content of PP particles (50%) followed by S4 (20%). A likely source of PP MPs could be textiles and packaging containers (Auta et al. 2018).

3.4. Assessment of microplastic human intake

There is a great concern regarding the health risks of MPs contamination in soil (Zhu et al. 2019). Metabolism disorders, oxidative stress and inflammatory reactions can be triggered in organisms by exposure to MPs (Chang et al. 2019). Furthermore, besides the potential harm caused by plastics particles, there may also be a contribution to the toxic effect of MPs by leachable additives from the plastic and adsorbed contaminants.

Microplastics can accumulate in human body via different exposure routes, such as dust inhalation, food ingestion or drinking water (Oßmann et al. 2018; Schymanski et al. 2018). Recent research indicates that MPs are persistent to chemical degradation in vivo due to the relatively short residence time (hours to days). If inhaled or ingested, they may also resist mechanical clearance, becoming lodged or embedded (Wright and Kelly 2017). Thus, physical harm would probably cause stress and health problems earlier than that related to the released chemicals (Ebrahimi et al. 2022). MPs biopersistence is an important factor contributing to their risk, along with dose. Current

research shows that some microplastics can translocate across living cells to the lymphatic and/or circulatory system, potentially accumulating in secondary organs, or affecting the immune system and cells health (Wright and Kelly 2017). After ingestion of MPs, microfold cells (M-cells), covering an intestinal lymphoid tissue—Peyer's patches, transport particles from the intestinal lumen to the mucosal lymphoid tissues (Ensign et al. 2012). The formation of corona proteins during digestion increases the solubility of insoluble particles and therefore enhances MPs bioavailability and intestinal uptake rate and hence the subsequent toxicological impact and health risks (Prata et al. 2020). The MPs elimination occurs through bile, urine, pulmonary alveoli, peritoneal cavity, cerebrospinal fluid and milk (Wright and Kelly 2017). Persistence and clearance rates are affected by the size, shape, chemical composition and additive chemicals of ingested MPs by humans (Smith et al. 2018).

Exposure to MPs may affect particularly vulnerable age groups. For instance, children are likely to accidentally ingest considerable amounts of soil or dust because of hand or finger sucking. Daily soil ingestion has been estimated according to some assumptions with regard to hand soil loading; the frequency of hand-to-mouth activity; hands' moisture, and efficiency of transfer (Dehghani et al. 2017). We have estimated daily and yearly MP intakes via soil ingestion, considering normal and acute exposure conditions, from the MP abundances found in this study and recommended values for the daily exposure rate of bulk soil. According to our estimation, in normal exposure scenarios, the MP daily intake is less than 1 particle per day, across sampling stations S1–S10 (see Table 2). The calculated yearly MPs ingestion, considering a normal exposure scenario, is 6–127 MPs. In condition of acute exposure in adults, the yearly exposure would be 20–418 MPs. Furthermore, the yearly intakes of MPs were in the range of 12–253 MPs in a normal exposure scenario, while 62–1267 MPs in acute exposure for children. Moreover, the median of computed

yearly ingestion of MPs under acute exposure was 3.3 and 5 times greater than normal exposure for adults and children, respectively (Table 2).

In Asaluye City (Iran), acute exposure approximations via dust ingestion are reported to be 15 and 5 particles per day for children and construction workers, respectively (Abbasi et al. 2018). Dehghani et al. (2017) showed that average intake of MPs through dust ingestion (Tehran, Iran) was 0.68 and 1.75 particles per day in normal exposure for adults and children, respectively, while it was 1063 and 3223 particles per year in acute exposure for adults and children, respectively. Hence, compared with the present study, exposure to MPs via dust can be greater than via industrial and urban soil in hotspots of in Iran.

Tissue-accumulation kinetics and distribution patterns of MPs are significantly size-dependent (Ebrahimi et al. 2022). Thus, it is also important to consider the size of particles that can be ingested and its impact on human health. Previous research indicated that fine particles of soil with modal size of about 250 μm have high chance of adhering to the skin surface and finer particles (< 50 μm) are likely to be potentially ingested involuntarily (Padoan et al. 2017; Mokhtarzadeh et al. 2020). Plastic particles, when sufficiently small (i.e. ~ 0.2 μm), can affect cells (Rothen-Rutishauser et al. 2006). The results of the current study show that the 250–500 μm class (with a mean occurrence of 27%) followed by 100–250 μm class (with a mean occurrence value of 25%) are the most abundant MPs that can adhere to surface of skin and be ingested by human, assuming that all MPs have the same behaviour in adhering to hands.

Table 2. Daily and yearly intake of microplastic particles in normal and acute exposure for adults and children. NE and AE refer to normal and acute exposure, respectively.

Sampling stations	MPs/100g soil	Adult				Children			
		NE (100 mg/day)		AE (330 mg/day)		NE (200 mg/day)		AE (1000 mg/day)	
		per day	per year	per day	per year	per day	per year	per day	per year
S1	17	0.02	6	0.06	20	0.03	12	0.17	62
S2	144	0.14	53	0.48	173	0.29	105	1.44	526
S3	97	0.10	35	0.32	117	0.19	71	0.97	354
S4	347	0.35	127	1.15	418	0.69	253	3.47	1267
S5	50	0.05	18	0.17	60	0.1	37	0.5	183
S6	39	0.04	14	0.13	47	0.08	28	0.39	142
S7	76	0.08	28	0.25	92	0.15	55	0.76	277
S8	51	0.05	19	0.17	61	0.1	37	0.51	186
S9	338	0.34	123	1.12	407	0.68	247	3.38	1234
S10	49	0.05	18	0.16	59	0.1	36	0.49	179
Median	63.50	0.06	23	0.21	76	0.13	46	0.64	232

At this stage, the information from this study informs about drivers of microplastic pollution, their risk evaluation, and need of mitigation strategies. However, more research is needed to indicate the risk assessment associated with MPs on human health.

4. Conclusion

T This study has found that urban and industrial soils of Abadan and Khorramshahr cities are contaminated with MP particles at a median concentration level of 64 MPs/100 g soil. The concentration of MPs found across the sampling sites did not allow identifying significant differences (p 0.05) between MP levels in the industrial and urban soils. Remarkably, levels as high as 347 MPs/100 g soil were found in a residential soil. The main types of MPs in the residential soil were red and black PET fibres. Indeed, they prevailed across the sampling sites. PET fibres could originate from clothing and household packaging materials or bottles. The fibres may move via air exchange and precipitate via atmospheric fallout. All MP shapes, except

spherules, were found to be correlated with the total concentration of MPs (p 0.05). This could be the result of diverse sources for the different types/shapes of MPs. However, spherules, which were white/transparent, could have a common origin and are probably primary MPs. The identification of MPs was carried out with complementary techniques, among them polarized light microscopy images provided superior characterization power than optical microscopy for the analysis of MPs.

The yearly intake of MPs from the study soils estimated in a normal scenario for adults and children was 6–127 MPs and 12–253 MPs, respectively. Under acute exposure, the intake increased to 20–418 MPs and 62–1267 MPs. The sizes of MPs more likely to be ingested by the citizens exposed to soil from the study area were 100–250 μm and 250–500 μm. MPs < 100 μm showed high variability across the study area. Urban and industrial soils present high pollution range of MPs when compared to other soil types. Remediation measures, reducing plastic litter and waste management policies should be taken to reduce further spread of MPs in the environment.

Data availability

The datasets analysed during the current study are available from the corresponding author on reasonable request.

Acknowledgements

The authors would like to acknowledge the help of Shiraz University Medical Geology Research Centre and Shiraz University research committee for analytical and logistic assistance.

Funding

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

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