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Wind erosion as a driver for transport of light density microplastics

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Abstract

Microplastic pollution in the environment is a growing concern in today's world. Wind-eroded sediment, as an environmental transport pathway of microplastics, can result in environmental and human exposure far beyond its sources. For the first time, this study investigates the presence of microplastics in wind-eroded sediments from different land uses in the Fars Province, Iran. Eleven test plots were selected based on land use and wind erosion risk. On each plot, wind erosion was simulated using a portable wind tunnel and the eroded sediment was collected for further analysis aimed at measuring light density microplastics (LDMP). The LDMPs were extracted in both soil and wind-eroded sediment using a floatation method and then further examined using microscopy. Annual LDMP transport by wind erosion was estimated using wind data from the study areas. LDMPs were detected in six study areas in the Fars Province which are highly prone to wind erosion. Although LDMPs were found mostly in agricultural land, it was also detected in the soils and sediments from two natural areas. The total

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concentrations of LDMPs in polluted areas were 6.91 and 20.27 mg kg^{-1} of microplastics in soil and wind-eroded sediments, respectively. The enrichment ratio for LDMP ranged from 2.83 to 7.63 in different areas. The erosion rate of LDMP ranged from 0.08 to 1.48 mg m^{-2} min^{-1}. The results of this study confirmed the key role of wind erosion in the spread of microplastics in terrestrial environments which could form an exposure risk to humans via direct inhalation of the particles transported with the dust.

Keywords: Microplastic Pollution, Human Health, Soil Erosion, LDMP

1. Introduction

Plastics have a myriad of applications in all aspects of everyday life. They are extensively used in the packaging industry, textile production, household applications, and agriculture. In Iran, each person uses at least three plastic bags a day, on average (IDoE, 2016). 21 tons of these plastic bags are being used in grocery stores, where 96 percent of them end up in the garbage and only four percent are reused (IDoE, 2016).

In developing countries, waste mismanagement is common and the majority of waste is dumped in open landfills. The plastic waste is exposed to sunlight, microbes, the atmosphere, and mechanical stress, all of which causes the plastic to decay and breakdown into microplastics (Dehghani et al, 2017). Although the majority of microplastic research has focused on freshwater and marine environments, it is highly likely that microplastics are pervasive in terrestrial environments. Soils can act as long term sinks for microplastics (Rillig, 2012). Dense polymers are more likely to remain in the soil and ultimately, to be transported into deeper soil layers, whereas lighter polymers are more likely to be transported by wind and water (Horton et al., 2017). Wind action may spread lighter plastic particles to other terrestrial locations or to surface waters (Zylstra, 2013).

As an environmental transport pathway of microplastics, wind-eroded sediment and dust can result in environmental- and human exposure far beyond the original source. Recent studies identified the risk of inhalation of microplastic particles and fibers (House of commons Environmental Audit Committee Oral evidence: Environmental impact of Microplastics, HC 925 Monday 9 May 2016).

Accumulation of wind-dispersed plastic debris and other trash are likely to worsen as urban areas expand and distances between urban and natural areas decrease (McDonald et al., 2009). Zylstra (2013) quantified wind-dispersed trash and plastic debris in a protected desert area in Southern Arizona. He found that discarded plastic bags and balloons were found in high densities in protected desert areas and could pose a threat to these arid environments. He indicated that trash densities were largely independent of road proximity, suggesting that wind could carry plastic bags and balloons more than 2 km into remote areas.

Dris et al. (2015) uncovered the presence of microplastics in atmospheric fallout in Paris for the first time. Researchers used a funnel in a glass bottle on the rooftop of buildings to collect fall out samples. Monitoring was carried out during a 3-month period. Microplastic atmospheric fallout ranged from 29 to 280 particles m^{-2} day^{-1}, with an average of 118 particles m^{-2} day^{-1}. Yang et al. (2015) emphasized the presence, abundances, sources, pathways, and related health effects of microplastics in the urban environment. They stated that microplastics can wind up in the environment due to their low density and therefore be distributed over long distances. Dris et al. (2017) as well investigated textile fibers, including microplastics, in 3 indoor and 1 outdoor air samples from sampling stations about 10 km far from the city of Paris. They also estimated the deposition rate of the fibers and their concentration in deposited dust collected by vacuum cleaner bags and found that outdoor concentrations of microplastics ranged between 0.3 and 1.5 fibers per cubic meter, while indoor concentrations ranged between 1 and 60 fibers per cubic meter. The deposition rate of the fibers in indoor environments was between 1586 and 11130 fibers day^{-1} m^{-2}. Regarding fiber type, 67 percent of the analyzed fibers were made of natural material, primarily cellulosic, while the remaining 33 percent fibers contained petrochemicals with polypropylene being predominant. Microplastics were also found in the atmospheric fallout from Dongguan city, China (Cai et al., 2017). Researchers collected atmospheric fallout samples from three sites of the air monitoring system by using a sampling device equipped with a glass bottle and a fixed support. Microplastics of three different polymers were identified. The concentrations of non-fibrous microplastics and fibers ranged from 175 to 313 particles m^{-2} day^{-1} in the atmospheric fallout. Thus, dust emission and deposition between the atmosphere, land surface, and aquatic environment were associated with the transportation of microplastics.

To our knowledge, there is no study on the wind erosion of microplastics in agricultural and natural areas. Such studies are needed to link the findings in the field studies to laboratory results in order to better understand real risks posed by microplastics.

Without reliable data regarding the severity of the problem in terrestrial environments, the ability of policy-makers to justify restrictions on the use of plastics is limited (Zylstra, 2013). This is also the case in Iran where the Iranian Department of the Environment reported that 7500 tons of waste is generated daily in the capital of Iran and 1 ton of this waste in the form of plastic. Plastics are extensively used in the (semi-)arid agricultural regions to improve the climate and make it more beneficial to plant growth (mulches, shelters or green houses) (Ekebafe et al., 2011). Wind-blown plastics are very common in natural environments in Iran.

This study investigates the occurrence of microplastics in soil and related wind-eroded sediment for different land uses of Fars province in Iran. Using a portable wind tunnel, we looked at the potential of microplastic wind transport as a result of wind erosion. Reporting from these areas is vital considering plastic use is predicted to increase. Unfortunately, there are limited capabilities for recycling and waste management in this region (UNEP, 2014).

2. Material and methods

2.1. Study areas

Eleven study areas were selected in the Fars province. Fars province is located in the south central region of Iran, from 27°2′ to 31°42′ latitude and 50°42′ to 55°36′ longitude (Moradi et al., 2011). Generally, all regions of Fars province are considered arid or semi-arid according to De Martonne aridity index (Nafarzadegan et al., 2012). Actually, most of these areas are considered to be the critical wind erosion regions in this Province (Rezaei et al., 2016). Drought is a common phenomenon in Fars province (Tehrani et al., 2016) due to low annual precipitation, varying between 100 mm to 400 mm (Nafarzadegan et al., 2012). The geographical locations of the study sites and four meteorological stations including Shiraz (29° 32′ N and 52° 36′ E), Abadeh (31° 11′ N and 52° 40′ E), Eghlid (30° 54′ N and 52° 38′ E), and Jam (27° 49′ N and 52° 20′ E) are presented in Figure 1a. The maximum wind speeds measured at these stations are also presented in Figure 1b, which illustrates the occurrence of high wind events in Fars Province.

High wind speeds higher than 7 m s^{-1} and up to 30 m s^{-1} are observed at these stations, and even higher wind speeds being registered at Eghlid. The studied regions included different land uses such as seasonal agricultural land, rangeland, and dried river beds. The average slope of all of the surfaces in the region was very low (less than 1%), and the status of the vegetation cover varied from weak (<5%) to moderate (5-20%) (Ahmadi, 2012). Plastic mulches has been used to reduce evaporation from agricultural fields. Improper removal of plastic from agricultural land after use is a significant source of environmental pollution. Moreover, the studied areas were in close approximation to urban areas where plastic use is higher than would be expected.

2.2. Soil sampling and soil analysis

Soil sampling was done in the first 10 cm of topsoil at 11 study sites in the summer of 2016. Random soil samples were taken in triplicate from the nearest place to the point where wind eroded sediments were collected. In total, 33 soil samples were collected for analysis. The collected soil samples were air-died and crushed to pass through a 2-mm sieve. Physical and chemical properties were then tested in the laboratory. In addition, one soil sample was collected at each site to determine the initial soil moisture content. Physical properties that were examined included soil texture using the hydrometer method (Page et al., 1992a) and soil particle size distribution via dry sieving (Kemper and Rosenau, 1986). Using data obtained from the dry sieving, the mean weight diameter (MWD) of soil particles was determined using the following Equation:

$$MWD = \sum_{i=1}^{n} \overline{X}_i . W_i \tag{1}$$

where $\overline{X}_i$ is the mean diameter of any particular size range of particles and W_i is the weight of particles in that size range expressed as a percentage of the total sample.

Chemical properties, including soil pH of the saturated soil-paste was measured using a pH-meter. In addition, soil organic carbon (OC) was measured using the Walkley and Black (1934) method and Calcium carbonate equivalent (CCE) was determined using the titration method (Pansu and Gautheyrou, 2006).

2.3. Wind-eroded sediment collection

In order to collect the eroded sediment at each study area, wind tunnel experiments were carried out at three distinctive places in each study site using a portable wind tunnel. This device was designed and constructed in the Dry and Desert Regions Research Center of Yazd University, Iran. This wind tunnel consisted of three main parts including (1) a jet fan as a wind generator which could generate wind speeds in the range of 0.5-22 m s^{-1} at the height of 0.25 m, (2) a working section which was 2.5 m long with a cross section of 0.3 m × 0.3 m and a test area measuring 1 m in length and 0.3 m in width at the bottom and (3) a sediment collector which was 8 m long (Fig. 1a and S1).

All wind tunnel experiments were conducted at a constant wind velocity of 12 m s^{-1} and duration of 10 min, which was well above the deflation threshold of the soils, allowing soil erosion rate measurement and eroded sediment generation under controlled conditions.

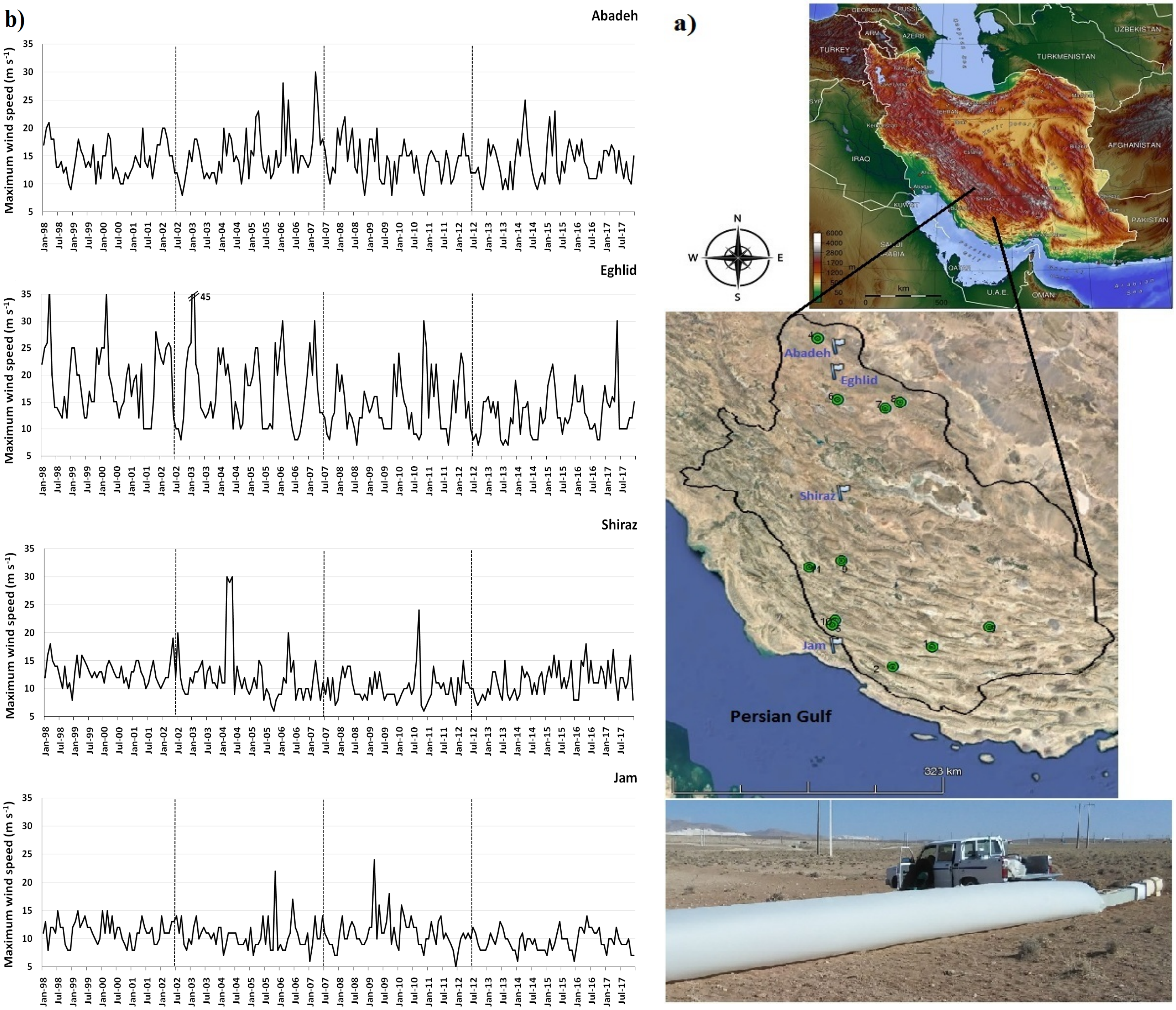

Fig. 1. a) Geographical map of the sampling sites and meteorological stations in the Fars Province, Iran with the view of the wind tunnel set up in the field. b) monthly maximum wind speeds (m s^{-1}) measured between 1998 and 2017 at these meteorological stations.

The wind tunnel was placed along the direction of the dominant wind of the region on the undisturbed soil surface. After the end of each experiment, the eroded sediments were collected from the sediment catcher and weighted in the laboratory. The weight of the wind-eroded sediment was converted to an erosion modulus (g m^{-2} s^{-1}) to determine the soil erosion rate via wind (Li et al., 2004). The standard soil erodibility classes determined in the wind tunnel at a wind speed of 12 m s^{-1} can be found in Table S1 (Ahmadi, 2012). Along with the specific sedimentation rate, the threshold wind velocity (U_t) was determined by gradually increasing the wind velocity in the wind tunnel until a forward movement of the soil particles was observed.

2.4. Light density microplastic (LDMP) extraction from soil and wind-eroded sediment

Light density microplastics were extracted from the soil and wind-eroded sediments using a new method developed by Zhang et al. (2018). For this method, soil and sediment samples were air dried and sieved to 2 mm. Microplastics and impurities from the samples were collected via the flotation method. 30 ml of distilled water was added to 15 g of soil/sediment and stirred. After one night, the floating materials were filtered using filter paper (pore diameter < 3 μm). This procedure was done at least four times. The solutions were vibrated for 2 hours using an ultrasonic cleaner agitation (50/60 Hz, Bath ultrasonic, Bransonic 52) and filtered again. The floatation was dried on the filter at 60 °C and evenly distributed on a glass slide using a brush. Plastic particles were distinguished from other particles (soil and impurities) by heating samples for 3-5 seconds at 130 °C. Photos were taken before and after burning using a camera (Leica DFC 425) connected to a microscope (Leica wild M3C, Type S) and analyzed for microplastic detection. After heating, the microplastics in soil and wind-eroded sediments melted and thus their shapes changed. Pictures of these heated slides were then compared to the original photos.

In total, 132 photos were taken for microplastic detection. All the plastic extractions were done in sterile conditions, using glass petri dishes.

2.5. LDMP count and mass calculation

The microplastic counting was performed with the aid of visual and physical properties using images before and after heating. Microplastic size was calculated using ImageJ software (Schindelin et al., 2015).

The area of microplastics after heating was calculated using Image J software under the following conditions: 1) image type was 8 bit and 2) color and resolution of the image was adjusted to make sure all of the particles were included.

The mass of the light density microplastics in the soil and wind-eroded sediments was calculated using the empirical model developed by Zhang et al. (2017):

$$m = \frac{4}{27}\,\rho \sum_{i=1}^{n} \sqrt{\frac{Si^3}{\pi}} \qquad (2)$$

In the above equation, m is the weight of plastics, ρ is the density of plastics, n is the number of microplastics counted, and S_i is the vertical angle of the view area of the plastic i after melting. Due to the fact that microplastics extracted by floatation method were light weighted plastics, their densities could be considered 0.92 g cm^{-3} (Liu et al., 2014a; Steinmetz et al., 2016).

Using the equation 2 and soil erosion rate by wind through a wind tunnel, we can then calculate LDMP erosion rate by wind (mg m^{-2} min^{-1}):

$$LDMP_{we} = WE * m \qquad (3)$$

where WE is the soil erosion rate by wind and m is the weight of LDMP.

2.6. Annual amount of LDMP transported by wind erosion

Using wind tunnel and statistical analysis of the wind velocity and duration at the study areas, we could calculate the annual wind-eroded sediment production (Ekhtesasi & Jahanbakhshi, 2015, Ahmadi, 2012) and thus, the annual amount of LDMP transported by wind erosion using the following equation:

$$WE_a = \Sigma\,(I * T) \tag{4}$$

where WE_a is the annual wind erosion potential (kg ha^{-1} year^{-1}), I is the soil erodibility potential given a specific wind velocity during one hour as measured in a wind tunnel (kg ha^{-1} hr^{-1}), and T is the wind continuity within a specific velocity (hr year^{-1}). Using the same equation, we calculated the annual LDMP wind erosion (kg ha^{-1} year^{-1}). It should be noted that since I is measured using a wind tunnel in the field, the effects of factors including roughness, soil moisture, soil texture etc. on soil erodibility is already considered in the value I.

For the estimation of annual LDMP transport by wind erosion, we used the data of the wind velocity from four meteorological stations. Since we did not have a meteorological station for each study site, the data from the stations where used according to topography and proximity to the study sites.

2.7. Estimation of LDMP intake via ingestion

The amount of microplastic ingestion per year was calculated based on the recommended values of the particle ingestion rate used by the USEPA (2000) which is 200 and 100 mg day^{-1} for children and adults, respectively.

2.8. Statistical Analysis

Quantitative data of microplastic size, number and weight were described as mean ± STD. The normal distribution of the data was tested using the KS test and the significance of the differences ($P<0.05$) were analyzed via ANOVA using SPSS 16 software. E-functions and scatter plots were produced using Excel 2016. Moreover, a principal component analysis (PCA) was performed to determine the relationship between soil properties, transported LDMP by wind erosion and LDMP content. The loading of a given variable was considered meaningful if its absolute value was ≥ 0.40 for a given component (Bento et al., 2017).

3. Results and Discussion

3.1. Soil Properties

Soil physical and chemical properties (mean ± STD) of the study sites are shown in Table 1. Soil moisture content of all studied regions were low and ranged from 1.07% to 2.16%. The highest soil moisture content needed to guarantee wind erosion is about 2% and soil moisture above this value can resist wind erosion (Nourzadeh et al., 2013; Wang et al., 2014; Bento et al., 2017). Therefore, soil moisture is not a limiting factor for wind erosion in the studied areas.

Table 1. Soil physical and chemical properties and soil erosion rate by wind of the study sites (mean ± STD, N=3).

Study site	**Sand**	**Silt**	**Clay**	**OC**	**CCE**	**pH**	**MWD**	**Moisture**	**Soil erosion rate**
	%	%	%	%	%	-	mm	%	$g\ m^{-2}\ s^{-1}$
1A	19.6 ± 0.00	50.7 ± 0.00	29.7 ± 0.01	0.53 ± 0.00	44.3 ± 0.08	7.77 ± 0.08	0.54 ± 0.01	2.01 ± 0.01	0.30 ± 0.04h
2A	25.6 ± 0.00	44.7 ± 0.00	29.7 ± 0.00	1.07 ± 0.00	46.9 ± 0.05	7.33 ± 0.05	0.38 ± 0.00	1.52 ± 0.01	1.10 ± 0.10ef
3A	82.9 ± 0.01	9.39 ± 0.00	7.72 ± 0.04	0.11 ± 0.01	83.8 ± 0.09	7.78 ± 0.10	0.40 ± 0.01	1.30 ± 0.03	1.95 ± 0.34d
4R	27.6 ± 0.00	46.7 ± 0.00	25.7 ± 0.00	0.41 ± 0.08	36.0 ± 0.02	7.79 ± 0.02	0.38 ± 0.01	1.55 ± 0.01	3.04 ± 0.26c
5R	89.6 ± 0.03	6.72 ± 0.01	3.72 ± 0.02	0.10 ± 0.00	64.6 ± 0.11	7.75 ± 0.11	0.20 ± 0.00	1.07 ± 0.02	10.3 ± 0.55a
6R	21.6 ± 0.00	44.7 ± 0.00	33.7 ± 0.00	0.28 ± 0.05	24.5 ± 0.19	7.76 ± 0.19	0.44 ± 0.01	2.16 ± 0.02	1.45 ± 0.15e
7R	33.6 ± 0.00	42.7 ± 0.02	23.7 ± 0.00	0.47 ± 0.05	30.1 ± 0.08	7.78 ± 0.08	0.44 ± 0.01	2.01 ± 0.02	0.86 ± 0.32fg
8R	35.6 ± 0.01	38.7 ± 0.00	25.7 ± 0.01	0.33 ± 0.01	29.4 ± 0.11	7.82 ± 0.11	0.38 ± 0.03	2.10 ± 0.01	0.52 ± 0.03gh
9A	35.6 ± 0.01	36.7 ± 0.01	27.7 ± 0.00	1.37 ± 0.01	56.3 ± 0.05	7.35 ± 0.05	0.34 ± 0.02	1.98 ± 0.03	1.00 ± 0.05f
10DR	23.6 ± 0.01	52.7 ± 0.02	23.7 ± 0.01	0.91 ± 0.00	55.8 ± 0.05	7.52 ± 0.05	0.42 ± 0.00	1.75 ± 0.05	0.96 ± 0.15fg
11A	85.6 ± 0.00	6.72 ± 0.01	7.72 ± 0.00	0.18 ± 0.01	72.7 ± 0.07	7.87 ± 0.07	0.24 ± 0.01	1.14 ± 0.01	4.72 ± 0.39b

A – Agricultural lands; R – Rangelands; DR – Dried river; OC – organic carbon; CCE – calcium carbonate equivalent; MWD – mean weight diameter. Values followed by different letters are significantly different (P<0.05).

3.2. Wind Erosion Rate

The measured soil erosion rates by wind at the study sites are listed in Table 1. According to the standard table of the wind tunnel (Table S1), all the study areas were considered as very sensitive to wind erosion. Study area 5R was the most sensitive to wind erosion (10.31 $g\ m^{-2}\ s^{-1}$) and study area 1A was the least sensitive to wind erosion (0.3 $g\ m^{-2}\ s^{-1}$).

The relationship between threshold wind velocity (U_t) and the soil erosion rate is shown in Figure 2a. High regression coefficients (R^2=0.96, P<0.01) were observed between the threshold velocity and the wind erosion intensity. With the increase in threshold velocity, the intensity of wind erosion decreases with a nonlinear trend. Different soils with different properties have a different sensitivity to wind erosion. As a result, the wind threshold velocities for particle movement vary considerably in different soils. As seen in figure 2a, the soils in the study areas showed a significant decrease in wind erosion rates with an increase in the threshold velocity, especially at velocities more than 8 m s^{-1}.

3.3. Relationship between aggregate size and the soil erosion rate

The relationship between aggregate size distribution (MWD) and soil erosion rate is shown in Figure 2b. The results pointed out that the soil erosion rate decreased significantly with an increase in MWD, following a power function (R^2=0.72, P<0.001). According to figure 2b, soil particles with a MWD larger than 0.3 mm could significantly reduce wind erosion. In fact, the larger particles acted as a kind of shelter on the soil surface against wind erosion (Mackinnon et al., 2004).

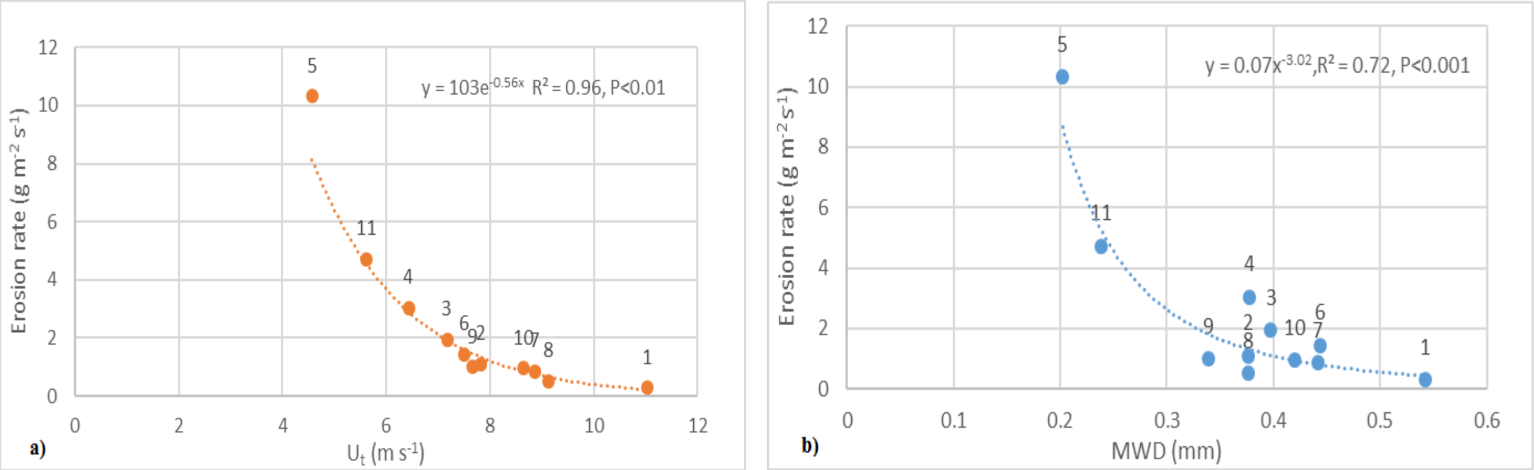

Fig. 2. a) The relationship between threshold wind velocities (U_t) and soil erosion rates in the studied sites, b) The relationship between MWD and the soil erosion rate at the studied sites.

3.4. Microplastics in soil and wind-eroded sediment

Microplastics were detected in 6 study areas of the Fars Province that are prone to wind erosion. Figure 4 illustrates the microscopic image of a few samples before and after heating. Abundance distribution of microplastics in each study area and in the province is shown in Fig. 3. Among them, the study area 1 had the most microplastics both in the soil and the sediments.

Microplastics were found mostly in agricultural lands. However, they were also found in two natural areas. Plastic mulches, containing mostly LDPE[2] with a density of < 0.93 g cm^{-3}, have been used to reduce evaporation from agricultural fields. Improper removal of plastic after use in agricultural lands is a significant source of environmental pollution. In addition, large quantities of microplastic particles from cosmetic, textile and industrial processes could be ending up on agricultural land that is fertilized with urban sewage sludge (Nizzetto et al., 2016). Moreover, anthropogenic effects on natural areas are ubiquitous. Most of the studied areas in this research were in close proximity to urban areas where plastics were being used at higher levels than expected. Finding microplastics in natural areas is alarming for policy makers.

A total of 5866 microplastic particles were detected in all of the soils and sediments. The minimum and maximum concentration of microplastics in polluted areas ranged from 67 to 400 particles per kg in soil and from 67 to 1133 particles per kg in sediments (Table 2).

The size of the microplastics ranged between 40 and 740 µm. Most of the Microplastics were less than 100 µm in size. Microplastics below 100 µm in size are the most susceptible size fraction to be ingested or re-suspended to the atmospheric load and thereby, the most likely to be inhaled (Dehghani et al., 2017).

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[2] Low Density Polyethylene

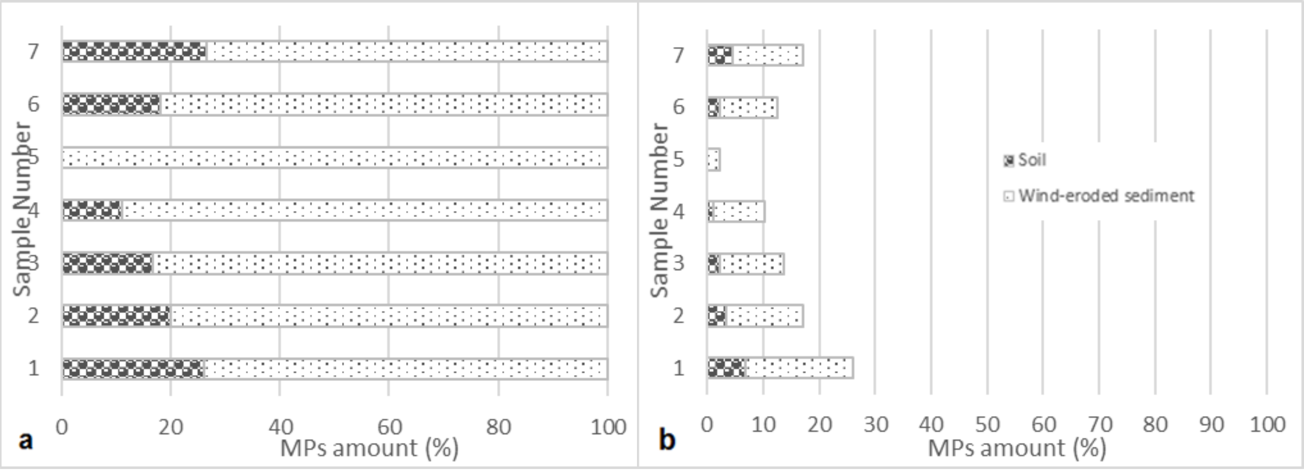


Fig. 3. Abundance distribution of microplastics in soil and wind-eroded sediments in a) % of the total number of microplastics in the Fars Province and b) % of the total number of microplastics in each study area.

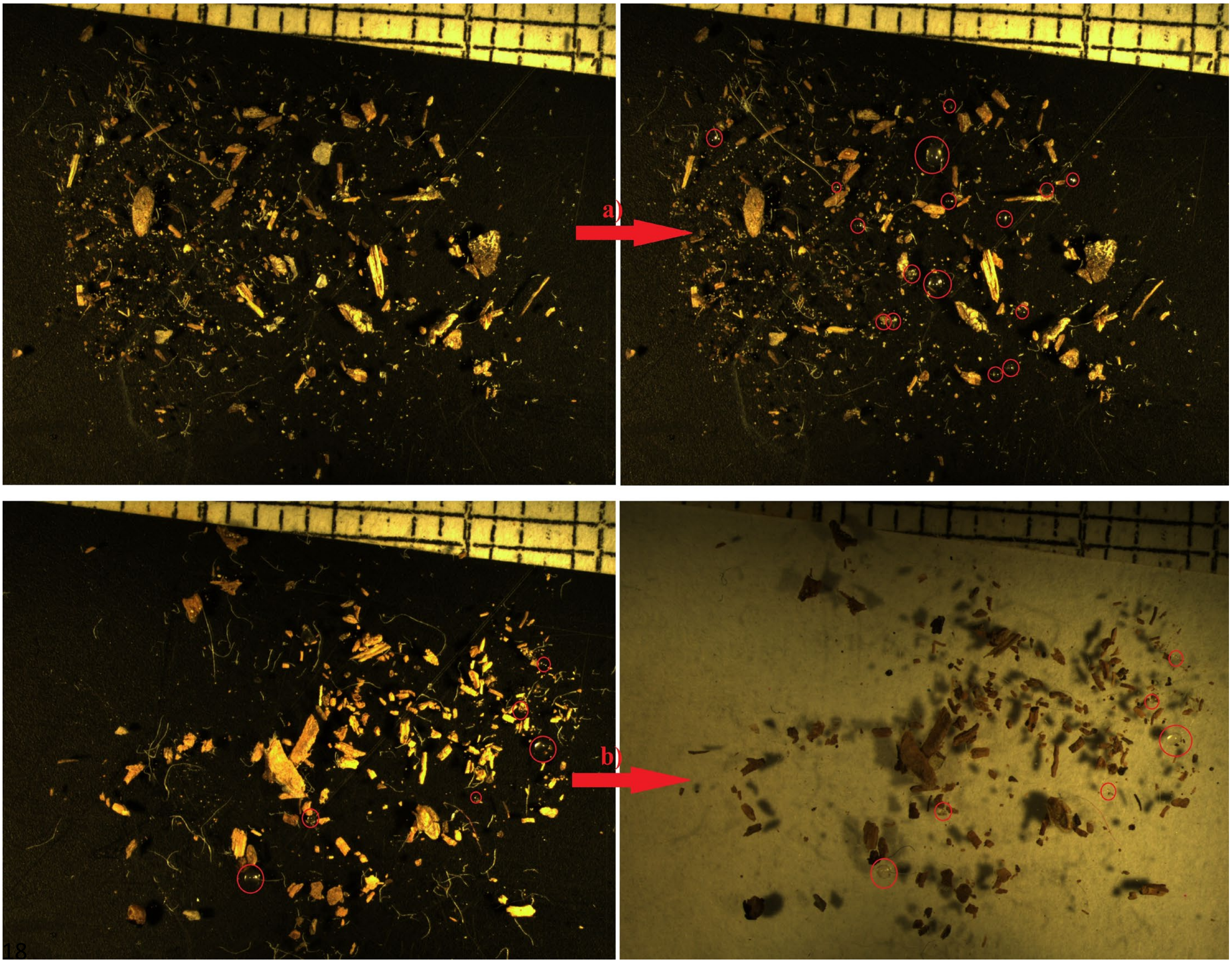


Fig. 4. a) Microplastic detection before and after heating in sample no.1, b) Visual differentiation of Microplastics from other particles after heating in sample no. 4.

Moreover, the results revealed a significant positive correlation (0.79) between the microplastics in the original soil with the abundance of microplastics in wind-eroded sediments. This implies that the number of microplastics in wind-eroded sediments increases with increasing numbers of microplastics in the soil. Relative standard deviation of counted microplastics was 58.1% and 46.1% for soil and sediment, respectively. This indicated heterogeneity among samples from different areas. The coefficient of variation of triplicate samples of soil and sediments shows the heterogenic nature of each soil/sediment sample.

Wind erosion is a very material sorting and removing process (Nerger et al., 2017); as a result, fine particles of microplastics in the saltation or suspension transport can be enriched. This is expressed through the enrichment ratio (ER), the ratio of LDMP content in the eroded material compared to the LDMP content in the original soil. Sterk et al., (1996) used an enrichment ratio for SOC content in wind-eroded sediment for the first time. Bach (2008) derived an ER of 0.98 for SOC in the saltation layer in wind tunnel studies in Germany. The enrichment ratio for LDMP (Table 2) ranged from 2.83 to 7.63 in different areas, which indicates that the number of microplastics were higher in the wind-eroded sediments as compared to the original soils. This can be attributed to the fact that the microplastics deposited in the surface layers were lighter than soil particles and they were easily blown away by wind erosion. In other words, microplastics have lower threshold velocities than soil particles. This shows that light, small-sized microplastics can be easy carried away by soil erosion in both agricultural and natural lands and pose a significant risk to human health. Wind-eroded sediment generated by the wind tunnel with a cross-sectional area of 0.3 m * 0.3 m included material transported by both saltation and suspension processes.

In addition, the proximity of the erodible study areas to urban environments results in direct ingestion and inhalation of dust particles and consequently, of microplastics by humans (Abrahams, 2002). This is particularly important for those potentially vulnerable segments of the population, such as children whose digestion systems are more susceptible to any negative health effects of environmental contaminants (Leotsinidis et al., 2005; Mielke et al.,1999;

Rojas-Bracho et al., 2002). Farmers and other people, who work in outdoor spaces, are another group of people who are exposed to wind-blown microplastics.

Table 2. Number and mass of LDMP in soil and sediment samples using the model and LDMP erosion rate by wind (mean ± STD, N=3).

Site	Sample	Number (N kg^{-1})	CV	Content (mg kg^{-1})	ER	LDMP erosion rate by wind (mg m^{-2} min^{-1})
1A	Soil	400 ± 305a	0.76	2.80 ± 0.59a	2.83	0.08 ± 0.04
	Sediment	1133 ± 67a	0.06	4.57 ± 2.06a		
2A	Soil	0 ± 0a	0	0	0.00	0
	Sediment	89 ± 38c	0.43	0		
3A	Soil	200 ± 231a	1.15	1.25 ±1.24bc	4.11	0.43 ± 0.02
	Sediment	822 ± 367b	0.45	3.69 ± 0.18a		
4R	Soil	100 ± 47a	0.47	0.23 ± 0.09c	5.02	0.61 ± 0.31
	Sediment	667 ± 377b	0.57	3.36 ± 1.71a		
5R	Soil	89 ± 38a	0.43	0.68 ± 0.32bc	7.63	1.48 ± 1.20
	Sediment	511 ± 203b	0.40	2.40 ± 1.94a		
6R	Soil	0 ± 0	0	0	0.00	0
	Sediment	0 ± 0	0	0		
7R	Soil	0 ± 0	0	0	0.00	0
	Sediment	111 ± 38c	0.34	0		
8R	Soil	0 ± 0	0	0	0.00	0
	Sediment	0 ± 0	0	0		
9A	Soil	156 ± 153a	0.98	0.41 ± 0.21bc	4.68	0.16 ± 0.02
	Sediment	622 ± 385b	0.62	2.60 ± 0.34a		
10DR	Soil	0 ± 0	0	0	0.00	0
	Sediment	0 ± 0	0	0		
11A	Soil	267 ± 240a	0.90	1.54 ± 0.94ab	2.83	1.03 ± 0.77
	Sediment	756 ± 328ab	0.43	3.65 ± 2.76a		

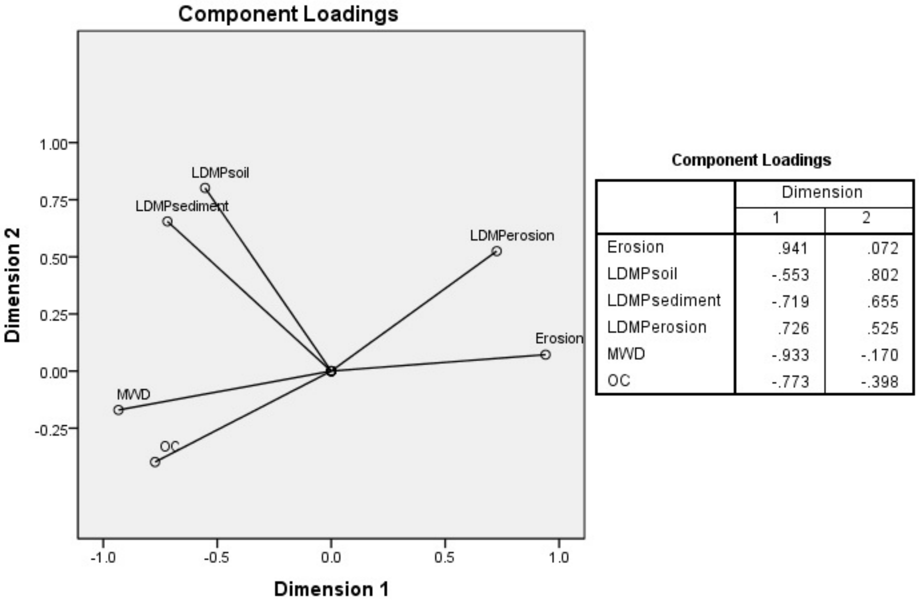
CV – Coefficient of Variation. ER – Enrichment Ratio. Values followed by different letters within the same sample (soil/sediment) are significantly different ($P<0.05$).

3.5. Mass estimation and LDMP transport related to wind erosion

Due to the great variance in the shape of microplastics in natural soils, it is difficult to quantify the volume of the microplastics. However, all of the microplastic particles in the soil samples were transformed into circular bubble-like shapes after heating, so it was easy to calculate the volume of the microplastics. The mass of microplastics in the soil and wind-eroded sediments are shown in Table 2. A total of 6.91 and 20.27 mg kg^{-1} of microplastics were present in the soil and sediments of the six polluted areas ranging from 0.23 to 2.8 mg kg^{-1} in soil and from 2.4 to 4.57 mg kg^{-1} in sediments. As for the number of microplastic particles, the results indicate that the mass of microplastics was higher in the wind-eroded sediments. The coefficient of variation of the triplicate samples of soil and sediments shows the heterogenic nature of each soil/sediment sample.

Results of the mass estimation were in the same order as the number of MP for different sites. This indicates that microplastics (number and mass) decrease in the order of site: 1A> 3A> 11A> 4R> 9A> 5R for wind-eroded sediments.

The minimum and maximum transport rate of microplastics by wind erosion ranged from 0.08 (1A) to 1.48 (5R) mg m^{-2} min^{-1}. Figure 5 shows the results of the principal component analysis. All the variables analyzed in this study loaded well in the first component. However, LDMP in soil and sediment samples loaded in the second component together with the LDMP erosion rate. The proportion of variance accounted for by the first component is 87.61%, whereas the second component accounted for 12.34%. Thus, the two components together accounted for a considerable proportion (99.96%) of the variance.



Component Loadings

	Dimension	
	1	2
Erosion	.941	.072
LDMPsoil	-.553	.802
LDMPsediment	-.719	.655
LDMPerosion	.726	.525
MWD	-.933	-.170
OC	-.773	-.398

Fig. 5. Principal components analysis. OC – organic carbon; MWD – mean weight diameter; LDMP – Light Density Microplastic.

3.6. The primary estimation of annual LDMP transport by wind erosion

Considering a logarithmic wind profile and roughness length generally observed for arid areas, wind velocity data (at 10m) from the four meteorological stations were transformed for wind tunnel height (0.25m) in order to explain the LDMP flux during wind erosion. According to the wind data, the frequency of erosive winds in Shiraz, Abadeh, Eghlid and Jam stations were 8.33, 25, 16.67, and 10.7 percent for 2016, respectively (NRWMO, 2005). Annual LDMP transport related to wind erosion calculated based on equation 4 (Table 3).

This can be evidence to suggest that microplastics can be transported and deposited by wind erosion. We used this procedure to present the first idea of the importance of the potential problem which should be further studied.

Table 3. Estimation of annual LDMP erosion based on meteorological data (2016).

Study site	Meteorological station	frequency of erosive winds (%)	frequency of erosive winds (hr year^{-1})	LDMP erosion rate (kg ha^{-1} hr^{-1})	Annual LDMP erosion (kg ha^{-1} year^{-1})
1A	Jam	10.7	937	0.05	46.9
3A	Shiraz	8.33	730	0.26	190
4R	Abadeh	25	2190	0.37	810
5R	Jam	10.7	937	0.89	834
9A	Shiraz	8.33	730	0.10	73
11A	Jam	10.7	937	0.62	581

hr – Hours

Due to the fact that I is measured in a wind tunnel which was in contact with one meter of soil surface, we need to correct the length for the natural settings. Studies showed that with an increase in the length, sedimentation rate will also increase (Chepil, 1956). On the other hand, sedimentation rate will decrease as the duration of wind erosion increases therefore, we leave these opposite effects out of our calculations for simplicity.

3.7. The primary estimation of LDMP exposure

The risk posed by microplastics in agricultural and natural areas usually stems from their potential to enter the groundwater via runoff or to be transported as dust and essentially, be inhaled by humans (Dehghani et al., 2017). Small particles of microplastics, which have lower densities in comparison to mineralogical dust (2.65 g cm^{-3}) (Hidalgo-Ruz et al., 2012), can be transported by suspension more easily than mineralogical particles during wind erosion events. Besides, agricultural soils are a significant source of airborne particulate matter due to of wind erosion and tillage activities (Gill et al., 2006, Bento et al., 2017). Therefore, offsite airborne transport of microplastics from farmland is very likely. This off-site transport is mostly associated with the finest particles (dust) which can travel over large distances (Bento et al., 2017). Inhalation of microplastics can be linked to human diseases and the effects could be intensified when the microplastics adsorb pollutants from water and soil on their surfaces (Rillig, 2012). Table 4 shows the yearly intake of microplastics based on the recommended 100 and 200 mg day^{-1} by USEPA (2000) as the mean dust ingestion rate for adults and children, respectively.

Table 4. Estimation of yearly intake of MPs in humans in normal exposure scenarios.

		Number of LDMP	
Study area	LDMPs (N kg^{-1})	Normal exposure for adults (100 mg day^{-1})	Normal exposure for children (200 mg day^{-1})
		Per year	Per year
1A	1133	41	83
3A	800	29	58
4R	667	24	49
5R	533	19	39
9A	600	22	44
11A	733	27	54

4. Conclusion

Wind erosion is a serious problem in arid and semi-arid areas of Iran. The present study shows, for the first time, that wind erosion should be considered as a transport pathway of microplastics in terrestrial environments. Microplastics were found in soil and wind-eroded sediments from both agricultural and natural areas. Wind-eroded sediments were enriched with microplastics as compared to the original soils. The risk of off-site transport of microplastics with wind-eroded sediment and dust is, therefore, very high. From literature, it becomes clear that microplastics measuring below 100 μm can be re-suspended to the atmospheric load and thereby, can be inhaled, contributing to the risk of human exposure. Therefore, more attention should be paid to this route of exposure in environmental and human health risk assessment studies. Moreover, source separation of waste materials should be encouraged in developing countries in order to make the recycling process of polymeric material easier. Furthermore, it is important to pay special attention to educational programs for the inhabitants of these developing countries in order to reduce plastic usage. It should be noted that this study gives a first indication of the role of wind erosion in the transport of microplastics. To get a better understanding of the actual transport rates, a more extended field measurement campaign with sediment catchers is recommended. Further studies are also needed to assess the mechanism of wind erosion in transporting microplastics in laboratory and field conditions.

5. References

Abrahams, P., 2002. Soils: their implications to human health. Sci. Total Environ. 291:1-32.

Ahmadi, H., 2012. Applied geomorphology (desert-wind erosion). Tehran University Publication, 706 p.

Bach, M., 2008. Aolische Stofftransporte in Agrarlandschaften – Experimentelle Untersuchungen und raumliche Modellierung von Bodenerosionsprozessen durch Wind (Ph.D. Thesis). University of Kiel, Germany, 129 pp.

Barnes, D.K., Galgani, F., Thompson, R.C., Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. Philos Trans Royal Soc Lond B Biol Sci. 364(1526):1985-1998.

Bento, C.P.M., Goossens, D., Rezaei, M., Riksen, M., Mol, H.G.J., Ritsema, C.J., Geissen, V., 2017. Glyphosate and AMPA in wind-eroded sediment derived from loess soil. Environmental Pollution. 220: 1079-1089.

Cai, L., Wang, J., Peng, J., Tan, Z., Zhan, Z., Tan, X., Chen, q., 2017. Characteristic of microplastics in the atmospheric fallout from Dongguan city, China: preliminary research and first evidence. Environ Sci Pollut Res. 24(32):24928-24935.

Chepil, W.S., 1956. Influence of moisture on erodibility of soil by wind. Proc. Soil Sci. Soc. Am. 20:288-292.

Dehghani, Sh., Moore, F., Akhbarizadeh, R., 2017. Microplastic pollution in deposited urban dust, Tehran metropolis, Iran. Environ Sci Pollut Res. 24(25):20360-20371.

Dris, R., Gasperi, J., Mirande, C., Mandin C., Guerrouache, M., Langlois, V., Tassin, B., 2017. A first overview of textile fibers, including microplastics, in indoor and outdoor environments. Environ Pollut. 221:452-458.

Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N., Tassin, B., 2015. Microplastic contamination in an urban area: a case study in greater Paris. Environ Chem. 12(5):592-599.

Ekebafe, L., Ogbeifun, D., Okieimen, F., 2011. Polymer application in agriculture. Biokemistri 23.

Ekhtesasi, M.R., Akhavan Ghalibaf, M., Azimzadeh, H.R., Emtehani, M.H., 2003. Effects of salts on erodibility of soils by wind. Iranian Journal of Natural Resources. 56 (1-2), 17-28 (In Persian).

Ekhtesasi, M.R., Jahanbakhshi, F., 2015. Models and Tools for Estimating and Measuring Wind Erosion and Dust Storms. Yazd University Publication. 238p.

Gill, T.E., Zobeck, T.M., Stout, J.E., 2006. Technologies for laboratory generation of dust from geological materials. J. Hazard. Mater. 132, 1-13.

Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: a review of methods used for identification and quantification. Environ Sci Technol. 46(6):3060-3075.

Horton, A.A., Walton, A., Spurgeon, D.J., Lahive, E., Svendsen, C., 2017. Microplastics in freshwater and terrestrial environments: Evaluating the current understanding to identify the knowledge gaps and future research priorities. Science of The Total Environment. 586: 127-141.

House of commons Environmental Audit Committee Oral evidence: Environmental impact of Microplastics, HC 925 Monday 9 May 2016.

Iranian Department of Environment (IDoE), 2016. Annual Internal Report. http://www.doe.ir

Kemper, W.D., Rosenau, R.C., 1986. Aggregate stability and size distribution. In: Klute A, editor, Methods of Soil Analysis, Madison (WI): ASA and SSSA, p. 425-442.

Leotsinidis, M., Alexopoulos, A., Kostopoulou-Farri, E., 2005. Toxic and essential trace elements in human milk from Greek lactating women: association with dietary habits and other factors. Chemosphere. 61(2):238-247.

Li, X.Y., Liu, L.Y., Wang, J.H., 2004. Wind tunnel simulation of Aeolian sandy soil erodibility under human disturbance. Geomorphology. 59(1-4), pp. 3-11.

Liu, E.K., He, W.Q., Yan, C.R., 2014a. “White revolution” to “white pollution”-agricultural plastic film mulch in China. Environ. Res. Lett. 9 (9), 3.

Mackinnon, D.J., Clow, G.D., Tigges, R.K., Reynols, R.L., Chavez, P.S., 2004. Comparison of aerodynamically and model derived roughness lengths (Z_0) over diverse surfaces, central Mojave Desert, California, USA. Geomorphology. 63, 103-113.

McDonald, R.I., Forman, R.T.T., Kareiva, P., Neugarten, R., Saer, D., Fisher, J., 2009. Urban effects, distance, and protected areas in an urbanizing world. Landscape and Urban Planning. 93, 63-75.

Mielke, H.W., Gonzales, C.R., Smith, M.K., Mielke, P.W., 1999. The urban environment and children's health: soils as an integrator of lead, zinc, and cadmium in New Orleans, Louisiana, USA. Environ Res. 81(2):117-129.

Moradi, H.R., Rajabi, M., Faragzadeh, M., 2011. Investigation of meteorological drought characteristics in Fars province, Iran. Catena. 84, 35-46.

Nafarzadegan, A.R., Rezaeian Zadeh, M., Kherad, M., Ahani, H., Gharehkhani, A., Karampoor, M.A., Kousari, M.R., 2012. Drought area monitoring during the past three decades in Fars province, Iran. Quat. Int. 250: 27-36.

Natural Resources and Watershed Management Office (NRWMO) of Fars Province. 2005. The studies of action plan of desert region management of Dowlatabad-Farashband region. Fars: Natural Resources and Watershed Management Office, 5-22.

Nerger, R., Funk, R., Cordsen, E., Fohrer, N., 2017. Application of a modeling approach to designate soil and soil organic carbon loss to wind erosion on long-term monitoring sites (BDF) in Northern Germany. Aeolian Research. 25: 135-147.

Nizzetto, L., Futter, M., Langaas, S., 2016. Are Agricultural Soils Dumps for Microplastics of Urban Origin? Environmental Science & Technology. 50(20): 10777-10779.

Nourzadeh, M., Bahrami, H.A., Goossens, D., Fryrear, D.W., 2013. Determining soil erosion and threshold friction velocity at different soil moisture conditions using a portable wind tunnel. Z. für Geomorphol. 57, 97-109.

Page, A.L., Miller, r.h., Jeeney, d.r., 1992a. Methods of Soil Analysis, Part 1. Physical and mineralogical methods. SSSA, Madison, p 1750.

Pansu, M., Gautheyrou, J., 2006. Handbook of Soil Analysis, Mineralogical, Organic and Inorganic Methods. Springer. 993 p.

Rezaei, M., Sameni, A., Fallah Shamsi, S.R., Bartholomeus, H., 2016. Remote sensing of land use/cover changes and its effect on wind erosion potential in Southern Iran. PeerJ. 4: e1948; DOI 10.7717/peerj. 1948.

Rillig, M.C., 2012. Microplastic in terrestrial ecosystems and the soil? Environ. Sci. Technol. 46(12), 6453-6454.

Rojas-Bracho, L., Suh, H.H., Oyola, P., Koutrakis, P., 2002. Measurements of children's exposure to particles and nitrogen dioxide in Santiago, Chile. Sci. Total Environ. 287(3):249-264.

Schindelin, J., Rueden, C.T., Hiner, M.C., Eliceiri, K.W., 2015. The ImageJ ecosystem: an open platform for biomedical image analysis. Mol. Reprod. Dev. 82(7-8), 518-529.

Steinmetz, Z., Wollmann, C., Schaefer, M., Buchmann, C., David, J., Troeger, J., Munoz, K., Fror, O., Schaumann, G.E., 2016. Plastic mulching in agriculture. Trading short-term agronomic benefits for long-term soil degradation? Sci. Total Enciron. 550, 690-705.

Sterk, G., Herrmann, L., Bantiono, A., 1996. Wind-blown nutrient transport and soil productivity changes in southwest Niger. Land Degradation & Development. 7, 325-335.

Tehrani, M.M., Kamgar-Haghighi, A.A., Razzaghi, F., Sepaskhah, A.R., Zand-Parsa, S., Eshghi, S., 2016. Physiological and yield responses of rainfed grapevine under different supplemental irrigation regimes in Fars province, Iran. Sci. Hortic. 202, 133-141.

UNEP, 2014. Valuing Plastic: The Business Case for Measuring, Managing and Disclosing Plastic Use in the Consumer Goods Industry. Division of Environmental Policy Implementation, United Nations Environment Programme (ISBN:978-92-807-3400-3).

USEPA (2000) N. Child-specific exposure factors handbook. NCEA-W-0853. External Review Draft, 2002.

Walkley, A., Black, I.A., 1934. An examination of the Degtjareff method for determining soil organic matter, and proposed modification of the chromic acid titration method. Soil Science. 37, 29-38.

Wang, L., Shi, Z.H., Wu, G.L., Fang, N.F., 2014. Freeze/thaw and soil moisture effects on wind erosion. Geomorphology. 207, 141-148.

Yang, Y.Y., Rodrigez-Jorquera, L.A., McGuire, M., Toor, G.S., 2015. Contaminants in the urban environment: microplastics. UF/IFAS Extension.

Zhang, S., Yang X., Gertsen, H., Peters, P., Salanki, T., Geissen, V., 2018. A simple method for the extraction and identification of light density microplastics from soil. Science of the Total Environment. 616-617:1056-1065.

Zylstra, E.R., 2013. Accumulation of wind-dispersed trash in desert environments. Journal of Arid Environments. 89, 13-15.