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Microplastics in agricultural soils from a semi-arid region and their transport by wind erosion

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ABSTRACT

Despite the importance of agricultural soils, little is known about the fate of microplastics (MPs) in this environment. In the present study, MPs have been determined in soils and wind-eroded sediments from two vegetable-growing fields in the Fars province of Iran, one using plastic mulch for water retention (Field 1) and the other using wastewater for irrigation (Field 2). MPs were heterogeneously distributed in the surface (0–5 cm) and subsurface (5–15 cm) soils of both fields, with a maximum concentration overall of about 1.1 MP g^{-1} and no significant differences in concentrations between either fields or depths. Fibres represented the principal shape of MPs, but spherules, presumably from wastewater, also made a significant (~25%) contribution to MPs in Field 2. Analysis of selected samples by Raman spectroscopy and scanning electron microscopy revealed that polyethylene terephthalate (PET) and nylon were the most abundant polymers and that MPs exhibited varying degrees of weathering. Concentrations of MPs in this study are within the range reported previously for agricultural soils, although the absence of PET observed in earlier studies is attributed to the use of insufficiently dense solutions to isolate plastics. Deployment of a portable wind tunnel revealed threshold wind velocities for soil erosion of up to 7 and 12 m s^{-1} and MP erosion rates up to about 0.4 and 1.1 MP m^{-2} s^{-1} for Fields 1 and 2, respectively. Erosion rates are considerably greater than published depositional rates for MPs and suggest that agricultural soils act as both a temporary sink and dynamic secondary source of MPs that should be considered in risk assessments and global transport budgets.

1. Introduction

Compared with the marine environment, the sources, occurrence, nature, transport and impacts of microplastics (MPs) in the terrestrial environment have received little scientific attention (da Costa et al., 2018; Chia et al., 2021). Terrestrial sources are particularly important because most plastic is generated and disposed of on land and soils appear to be long-term receptors of plastic (Yang et al., 2021). In addition, soil erosion facilitates the transport of MPs from the terrestrial environment to the atmosphere and aquatic ecosystems (Kumar et al., 2020), with soil, therefore, playing a critical role in the global cycling of MPs.

One of the important sources of MPs in soils is the wet and dry deposition of atmospheric material derived from a multitude of anthropogenic activities, although the precise regions of origin of these materials are often difficult to establish because MPs are subject to long-range aeolian transport (Allen et al., 2019; Brahney et al., 2020). Agricultural soils are of particular concern because farming represents an important, global land use whose end-products are usually designed for direct human consumption. MP inputs to agricultural soils are also often augmented through the addition of biosolids as a fertiliser (Crossman et al., 2020), contaminated wastewater for irrigation (Kumar et al.,

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2020), polymer-based, slow-release fertilisers (Weithmann et al., 2018), and by the weathering and fragmentation of plastic films used for mulching (Huang et al., 2020).

In many studies, agricultural soils are considered as a sink for MPs (Rochman, 2018; Waldschläger et al., 2020). However, post-depositional vertical and horizontal transport of MPs in soils is believed to be important (Mai et al., 2018) and water erosion has the potential to carry MPs from agricultural lands to aquatic ecosystems (Rehm et al., 2021). Wind erosion of MPs in agricultural soils is also likely to be significant, especially in arid and semi-arid regions, but this pathway has thus far received little scientific attention (Rezaei et al., 2019).

The present study aims to improve our understanding of the role of agricultural soils as a sink and secondary source of airborne MPs through wind-driven erosion. Specifically, we determine the quantities and characteristics of MPs in surface and subsurface soils from two vegetable-growing fields in Fars province, Iran, that have been subject to different agricultural practices: namely, plastic mulching for water conservation and irrigation by wastewater. We also deploy a portable wind tunnel in both fields to estimate the erodibility and flux of MPs from the soil surface. We hypothesize that different agricultural practices may result in different distributions, types and mobilities of MPs.

2. Materials and methods

2.1. Site description

Fars province, south-central Iran (Fig. 1), has an arid to semi-arid climate and a mean annual precipitation ranging from about 100 to 400 mm (Rezaei et al., 2016). Wind erosion occurs in most areas of the province, with monthly maximum wind velocities often exceeding 25 m s^{-1} at a height of 10 m. In addition, the region is subject to significant dust storm events (Mazidi et al., 2015). Two agricultural fields in the province, shown in Fig. 1, were studied over a seven-day period, beginning June 20th, 2020, during which there was no precipitation, winds of 2–7 m s^{-1} were from the west or north-west and mean daily air temperature ranged from about 25 to 35 °C. Both fields had been used to grow tomatoes for a decade but in Field 1 clear plastic (nylon) mulch was employed to reduce evaporation and in Field 2 wastewater from a water treatment plant was used for irrigation.

2.2. Soil sampling

Soil samples of about 200 g were collected with an auger from five, randomly selected 1 m × 1 m plots in each field (numbered P1 to P5 in Field 1 and P6 to P10 in Field 2). Three soil samples were collected from each plot at depths of 0–5 cm (D1; surface) and 5–15 cm (D2; subsurface), with soils from the same layer bulked together to form a composite sample. Samples were transported to the laboratory in aluminium (Al) foil where measurements were made of soil pH and electrical conductivity (EC) and soil texture was determined using a hydrometer.

2.3. Portable wind tunnel experiments and MP flux

To collect the wind-eroded sediment (and associated MPs) in Fields 1 and 2, experiments were carried out at three locations in both fields and close to the corresponding sampling plots (P1, P3, P5 in Field 1; P6, P8, P10 in Field 2) using a portable wind tunnel which is illustrated in Fig. 2 and described in detail by Rezaei et al. (2019). Briefly, the tunnel, of about 10 m in length and consisting of a jet fan, metallic working section of area 0.3 m^2 exposed to the soil and two-layer white nylon cyclone collector, was placed on an undisturbed soil surface along the direction of the prevailing wind that had been ascertained using a Benetech GM8902-plus anemometer. The same wind tunnel has been employed successfully in other wind erosion studies (Asl et al., 2019; Mina et al., 2021), including one that considered MPs (Rezaei et al., 2019).

The threshold velocity of wind erosion was measured by gradually increasing the air flow in the tunnel until the forward movement of soil particles was observed (Belnap et al., 2007). Wind-eroded sediment collection was conducted at a constant velocity of 12 m s^{-1} (above the deflation threshold of the soils) and for a duration of 10 min, allowing soil erosion rate to be determined under controlled conditions. On return to the laboratory, eroded sediments retrieved from the end of the cyclone collector wereweighed using a Shimadzu Libror balance and the

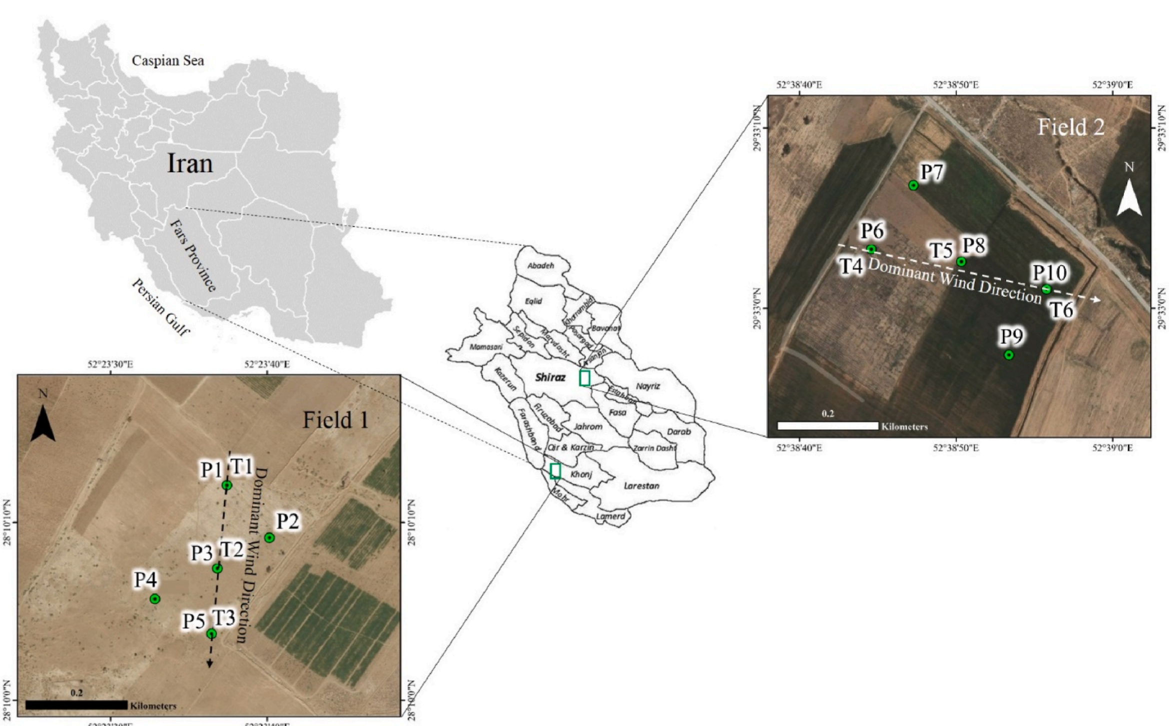

Fig. 1. The two agricultural fields in the Fars province of Iran and locations of the sampling plots (P) and wind tunnel experiments (T).

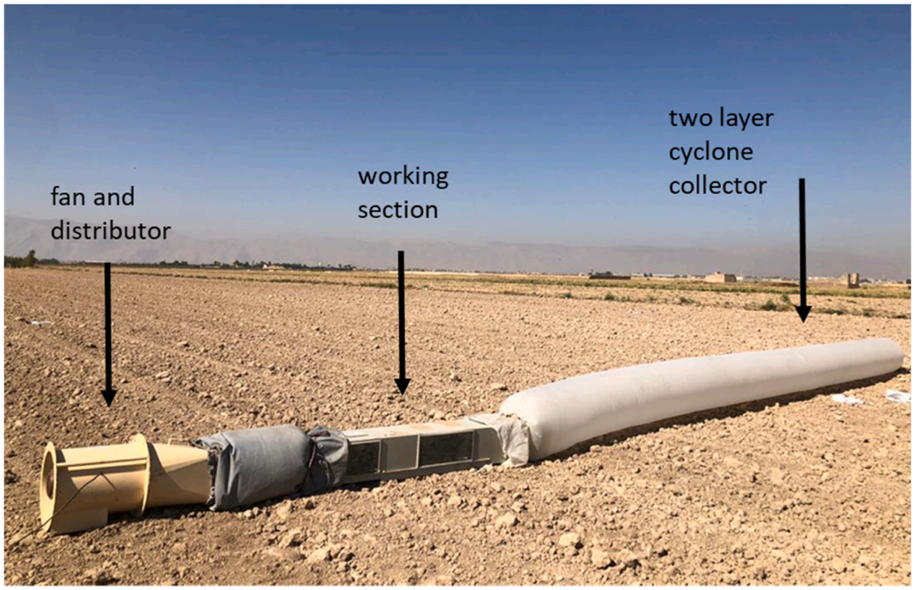

Fig. 2. The portable wind tunnel deployed along a section of agricultural soil.

wind erosion rate, in g m^{-2} s^{-1}, was determined by dividing the sediment mass by soil surface area and duration of the experiment (Liu et al., 2007). MP flux (MP m^{-2} s^{-1}) was then calculated by multiplying the soil erosion rate by the numbers of MPs per g of eroded sediment.

2.4. Microplastic extraction, identification and characterisation

Composite soil samples and eroded sediments retrieved from the wind tunnel were transferred to individual, 600-mL glass beakers using a stainless steel spoon. The contents were covered with Al foil and dried for 24 h at 25 °C before being sieved through a 5-mm stainless steel mesh to remove coarse material. In clean 600-mL glass beakers, organic matter was decomposed by oxidation of 200 g of each composite soil sample and 3–6 g of wind tunnel soil with 15–200 mL of 30% H_2O_2 solution (Arman Sina, Tehran) at 25 °C until bubble formation ceased. Residual material was washed through a 150 mm-diameter S&S filter paper (blue ribbon cellulose circles, grade 589/3, 2 μm pore size) using filtered, deionized water before being dried in a sand bath at 60 °C for 2 h.

MPs were separated by flotation for 5 min in a saturated 300 mL solution of $ZnCl_2$ (Arman Sina, Tehran; density 1.6–1.8 g cm^{-3}) in clean glass beakers after an initial period of agitation at 350 rpm. The decanted contents were subsequently centrifuged at 4000 rpm and the supernatants vacuum-filtered through S&S filter papers. This procedure was repeated twice, with resulting filters air-dried for 48 h at 25 °C in a clean room under laminar flow and transferred to glass Petri dishes for physical and chemical characterisation.

MPs on filters were visually identified and counted under a binocular microscope (Carl-Zeiss) at up to 200× magnification. Identification criteria were based on established visual characteristics and reaction to a hot, 250-μm stainless steel needle (Hidalgo-Ruz et al., 2012), and particle size was subsequently determined on imagery using ImageJ software (Abbasi et al., 2019). MP colour was categorised as black-grey, yellow-orange, white-transparent, red-pink or blue-green, shape or type was classified as fibre, primary (pellet, granule) or secondary (fragment, film), and size was scaled according to length, L, along the longest axis ($L \leq 100$ μm, $100 < L \leq 250$ μm, $250 < L \leq 500$ μm, $500 < L \leq 1000$ μm, $L > 1000$ μm).

Twenty six randomly selected MP samples were analysed for surface characteristics and polymer composition by scanning electron microscope (SEM) and Raman spectrometry. The micro-Raman spectrometer (LabRAM HR, Horiba, Japan) employed a laser of 785 nm and Raman shift of 400–1800 cm^{-1} with acquisition times between 20 and 30 s. A high vacuum SEM (TESCAN Vega 3, Czech Republic) was operated with a resolution of 2 nm at 20 kV with MP mounted on double-sided copper adhesive tape on microscope slides and gold-coated.

3. Results

The characteristics of the agricultural soils from the two fields are shown in Table 1. Soil in Field 2 with waste-water application was finer (more loamy), slightly more acidic and had a higher EC than soil in Field 1 where plastic mulch had been applied to prevent evaporation.

Table 2 shows the threshold velocity and wind erosion rate at each location where the wind tunnel was deployed. The threshold velocity is lower and wind erosion rate is greater in Field 1.

Table 3 presents a summary of the numbers, concentrations and characteristics of the MPs in the agricultural soil samples and eroded sediments captured by the portable wind tunnel from the two fields. (The full data set is given in the Supporting Information.) A total of 768 and 1018 MP were retrieved from the soils of Fields 1 and 2, respectively, but these were distributed highly heterogeneously between plots and depths. For example, the number of MPs ranged from 18 to 195 and from 8 to 126 for surface and subsurface soils, respectively in Field 1, and from 44 to 222 and from 40 to 123 for surface and subsurface soils, respectively in Field 2. Overall, and when normalised to the dry mass of soil, concentrations ranged from 0.04 MP g^{-1} to 0.83 MP g^{-1} in Field 1 (mean = 0.38 MP g^{-1} and median = 0.27 MP g^{-1}) and from 0.20 MP g^{-1} to about 1.1 MP g^{-1} in Field 2 (mean = 0.51 MP g^{-1} and median = 0.38 MP g^{-1}, respectively). There were no clear patterns in MP numbers or concentrations with soil depth, with values in surface soils (D1; 0–5 cm) that were either greater than, lower than or similar to corresponding values in subsurface soils (D2; 5–15 cm). Consequently, and despite

Table 1
Soil texture, pH and electrical conductivity (EC) of the agricultural soils.

	Field 1	Field 2
pH	8.10	7.48
EC, dS m^{-1}	0.27	0.58
Sand, %	77.1	29.5
Silt, %	14.7	44.7
Clay, %	8.2	25.8
Texture	loamy sand	loam

Table 2
Results from the portable wind tunnel experiments in Fields 1 and 2.

Field	Location	Threshold speed, m s^{-1}	Eroded sediment, g	Erosion rate, g m^{-2} s^{-1}
1	T1	7	5.5	0.031
	T2	6	5.9	0.033
	T3	7	3.7	0.021
2	T4	10	2.6	0.014
	T5	11	3.8	0.021
	T6	12	2.7	0.015

different agricultural practices, there were no significant differences in MP concentrations between fields and between the different depths according Kruskal-Wallis tests performed in Minitab *v*19.

MP shape was also variable but, overall, fibres were the dominant type of MP in the soils, with percentage contributions at the different plots ranging from 5.6 to 100 in Field 1 and 4.5 to 100 in Field 2. White-transparent was the most important colour among the soil MPs, with percentage contributions at the different plots ranging from 5.6 to 100 in Field 1 and 15.9 to 90.5 in Field 2. MPs in the smallest size category ($L \leq$ 100 μm) were absent in some soil samples and made a maximum contribution of <50% in both fields. Likewise, MPs in the largest size category ($L >$ 1000 μm) were absent in some soil samples, but with maximum contributions in Fields 1 and 2 of about 50% and 65%, respectively.

In the wind-eroded sediments, MPs were distributed heterogeneously between and within fields, and with respect to size, type and colour. However, non-fibrous MPs were always present (and up to 82.5% for sample T6), with relative abundance in the order: film > fragment > spherule; and MP concentrations normalised to soil mass (MP g^{-1}) were considerably higher than in the agricultural soil samples and were significantly higher ($p < 0.05$) in Field 2 (mean and median of 45.26 MP g^{-1} and 42.89 MP g^{-1}, respectively) than in Field 1 (mean and median of 9.08 MP g^{-1} and 11.35 MP g^{-1}, respectively) according to a Kruskal Wallis test performed in Minitab *v*19.

Fig. 3 shows the MP flux (MP m^{-2} s^{-1}) for each location where the wind tunnel was deployed. Using the wind erosion rates given in Table 2 and the numbers of MPs per g of eroded sediment in Table 3, MP fluxes were found to be in the range of about 0.13 MP m^{-2} s^{-1} at T1 to about 1.1 MP m^{-2} s^{-1} at T4 and with a median of about 0.3 MP m^{-2} s^{-1}.

Because of the heterogeneity of the data, MPs by type (shape) were pooled for the surface soils and wind-eroded sediments in both agricultural fields and as shown in Fig. 4. Overall, fibres made the dominant contribution to surface soil MPs in Field 1, but in Field 2 about one-quarter of MPs were spherules. In Field 1, the wind tunnel suspended proportionally more films and fewer fibres than in the corresponding original soils, while in Field 2, proportionally more fibres and less fragments were suspended than in the corresponding soils. In both fields, the percentage of fragments in the wind-eroded sediments was lower than in the corresponding soils.

Fig. 5 shows the frequency distribution of polymer types among the 26 MPs retrieved from the soils and wind tunnel. The majority of MPs were polyethylene terephthalate fibres and nylon fibres and fragments of various lengths and colours, with contributions also arising from polypropylene and polystyrene fibres and fragments.

Fig. 6 illustrates SEM images for four MPs retrieved from the

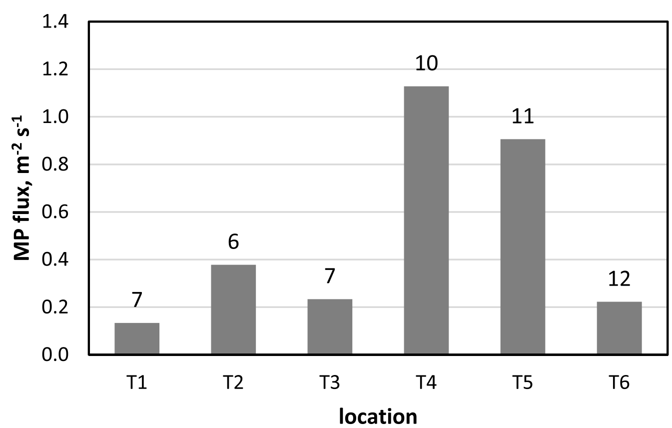

Fig. 3. MP fluxes for the six locations where the wind tunnel was deployed in the two fields. Annotated above each bar are corresponding threshold wind velocities for soil erosion in m s^{-1}.

Table 3
Numbers, concentrations and characteristics of MPs retrieved from the agricultural soils samples (P1 to P10; D1 = 0–5 cm, D2 = 5–15 cm) and wind-eroded sediments (T1 to T6) in Fields 1 and 2.

	Location	Depth	Total MPs	MP g^{-1}	% Fibres	% White/trans	% $L \leq$ 100 mm	% $L >$ 1000 mm
Field 1	P1	D1	76	0.38	56.6	50.0	0	23.7
		D2	126	0.63	100	22.2	49.2	1.6
	P2	D1	195	0.98	96.9	20.5	20.5	4.6
		D2	8	0.04	37.5	37.5	0	50.0
	P3	D1	21	0.11	81.0	100	0	0
		D2	31	0.16	100	96.8	19.4	0
	P4	D1	165	0.83	97.6	96.4	23.0	3.0
		D2	33	0.17	100	48.5	48.5	0
	P5	D1	18	0.09	5.6	5.6	5.6	22.2
		D2	95	0.48	80	72.6	0	51.6
	T1		24	4.36	45.8	54.2	0	16.7
	T2		68	11.53	54.4	89.7	80.9	1.5
	T3		42	11.35	64.3	21.4	7.1	21.4
Field 2	P6	D1	165	0.83	58.8	61.2	6.7	35.2
		D2	40	0.20	60.0	82.5	12.5	10.0
	P7	D1	63	0.32	81.0	90.5	0	65.1
		D2	87	0.44	60.9	29.9	20.7	21.8
	P8	D1	44	0.22	4.5	15.9	0	0
		D2	45	0.23	100	84.4	40.0	42.2
	P9	D1	177	0.89	52.5	38.4	6.2	32.2
		D2	52	0.26	65.4	86.5	0	37.8
	P10	D1	222	1.11	34.2	55.9	26.1	7.7
		D2	123	0.62	40.7	24.4	26.0	33.3
	T4		203	78.08	86.2	56.7	3.0	62.6
	T5		163	42.89	65.6	58.3	17.8	24.5
	T6		40	14.81	17.5	37.5	7.5	0

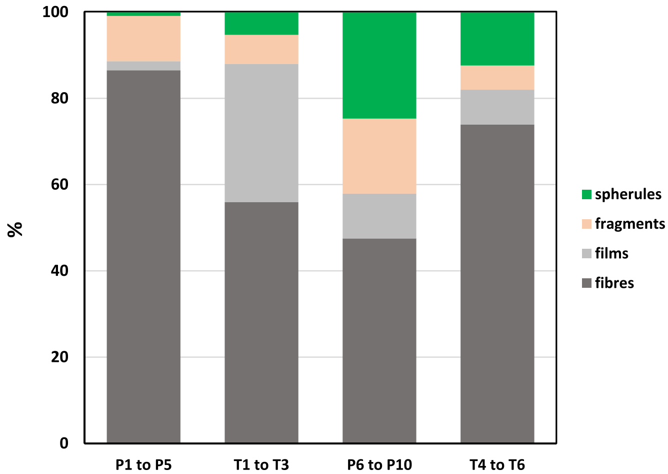

Fig. 4. Overall percentage distribution of MPs by shape retrieved from the agricultural soils and wind tunnel in Fields 1 and 2.

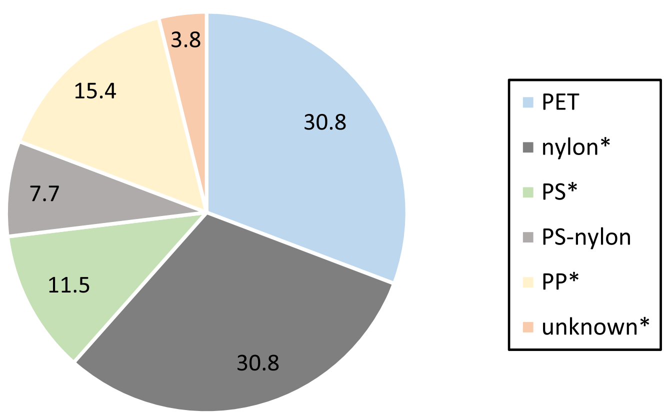

Fig. 5. Relative frequency distribution of polymers (as percentages) amongst 26 MPs randomly selected from surface soils and wind-eroded sediments. PET = polyethylene terephthalate, PS = polystyrene; PP = polypropylene; an asterisk denotes the presence of non-fibrous MPs.

agricultural soils. While all fragments analysed (n = 5) exhibited considerable weathering, the fibres (n = 21) displayed variable degrees of weathering, with some havingsmooth surfaces and others exhibitingirregular, cracked and pitted surfaces.

4. Discussion

4.1. MP distributions in agricultural soils

The net accumulation of MPs in both fields was highly heterogeneous, and despite different agricultural practices (mulching versus irrigation) and differences in soil properties, there were no clear differences in MP concentrations in topsoils or subsurface soils, or in the types of plastic (polymeric makeup) or their colour between the fields. One might have expected, for example, more film-like MPs in Field 1, where nylon mulching sheets have been used and large (>10 cm) fragments were often visible on the soil surface, and more fibrous MPs in Field 2 where wastewater contaminated with industrial and textile fibres has been used for irrigation. Similarly, more MPs might be expected in Field 2, and especially at the surface, where there was a greater proportion of clay-sized soil particles that can capture and trap MPs. Although, more generally, it is predicted that higher numbers of MPs should be found in topsoil where immediate deposition takes place (Liu et al., 2018), their abundance below the surface suggests ready but heterogeneous migration and percolation to lower layers during periods of precipitation (Cao et al., 2021; Yu et al., 2021).

The only distinct differences between the two agricultural fields were a greater number and proportion of sphere-like MPs and higher threshold wind velocities for soil in Field 2. The former observation may be attributed to the presence of MP beads in wastewater derived from, for example, abrasive agents in personal care and cosmetic products (Cheung and Fok, 2017), while the latter observation may be attributed to the higher proportion of cohesive (clay) material in the soil that acts to hold surficial MPs more tightly in the substrate.

4.2. Comparison with literature data

Table 4 summarises the concentrations and characteristics of MPs reported in the literature for various agricultural soils. Consistent with the present study, MPs appear to be distributed heterogeneously, with a range in concentration spanning one or two orders of magnitude for some locations, and are encountered both in topsoils and in soils below the surface. The mean surface and subsurface concentrations in Fars province are similar to the mean concentration observed in Chilean arable soils, to the lower concentrations reported for arable soils in Shouguang City and for soils used for vegetable growing in the suburbs of Wuhan, and to the upper concentrations reported for various agricultural soils in western Greece and northwest China. Mean concentrations in this study are, however, an order of magnitude greater than mean concentrations in vegetable-growing soils in the Yangtze River and suburban Shanghai regions of China. Note that all concentrations reported in the literature are above the mean concentration recently derived for remote, subtropical desert soils (0.02 MP g^{-1}; Abbasi et al., 2021), suggesting that agricultural soils are more generally contaminated compared with a soil "baseline", regardless of their location, use and history. Sources of contamination referred to in the studies above include the use of mulch and irrigation water (Liu et al., 2018), flooding by river water (Yu et al., 2021) and the inputs from centres of urbanisation and industrialisation (Chen et al., 2020). The variability observed in the literature is, presumably, related to factors like climate, soil texture, proximity to urbanised-industrialised areas and the precise nature, source and frequency of irrigation or mulching, as well as differences in means of sample collection and, as elaborated below, MP isolation and identification.

Many published studies on agricultural soils report fibres and fragments (that appear to embrace "films") as an important (or the dominant) type of MP, and polypropylene appears to be the most commonly documented polymer. Although we only determined the polymeric makeup of about 1.5% of MPs identified and may have overlooked some important types of polymer (e.g., polyethylene), the present study is the first to demonstrate the potential significance of polyethylene terephthalate (PET) fibres. These fibres are derived from polyester textiles and are commonly encountered in sewage sludge and waste water (Gaylarde et al., 2021; Zhang et al., 2021) and appear to be an important component of the MP pool in the atmosphere (Liu et al., 2019). A careful examination of the methods of MP isolation in the independent studies shown in Table 4 reveals that all but one employed either distilled water or saturated NaCl solution (density ~ 1.2 g cm^{-3}) for flotation. Consequently, and with a density of 1.38 g cm^{-3}, PET particles would have evaded capture during soil processing. Moreover, in the only independent study employing a sufficiently dense solution to retain PET (NaI, 1.7 g cm^{-3}; van den Berg et al., 2020), MPs were not analysed for polymeric construction. Thus, it would appear that in many published studies in agricultural soils, MP abundance is restricted to "light" plastics and the total content is likely underreported.

4.3. Windblown MP fluxes from agricultural soils

MPs captured by the wind tunnel revealed enrichment in the wind-eroded sediments (in MP g^{-1}) compared to corresponding surface or

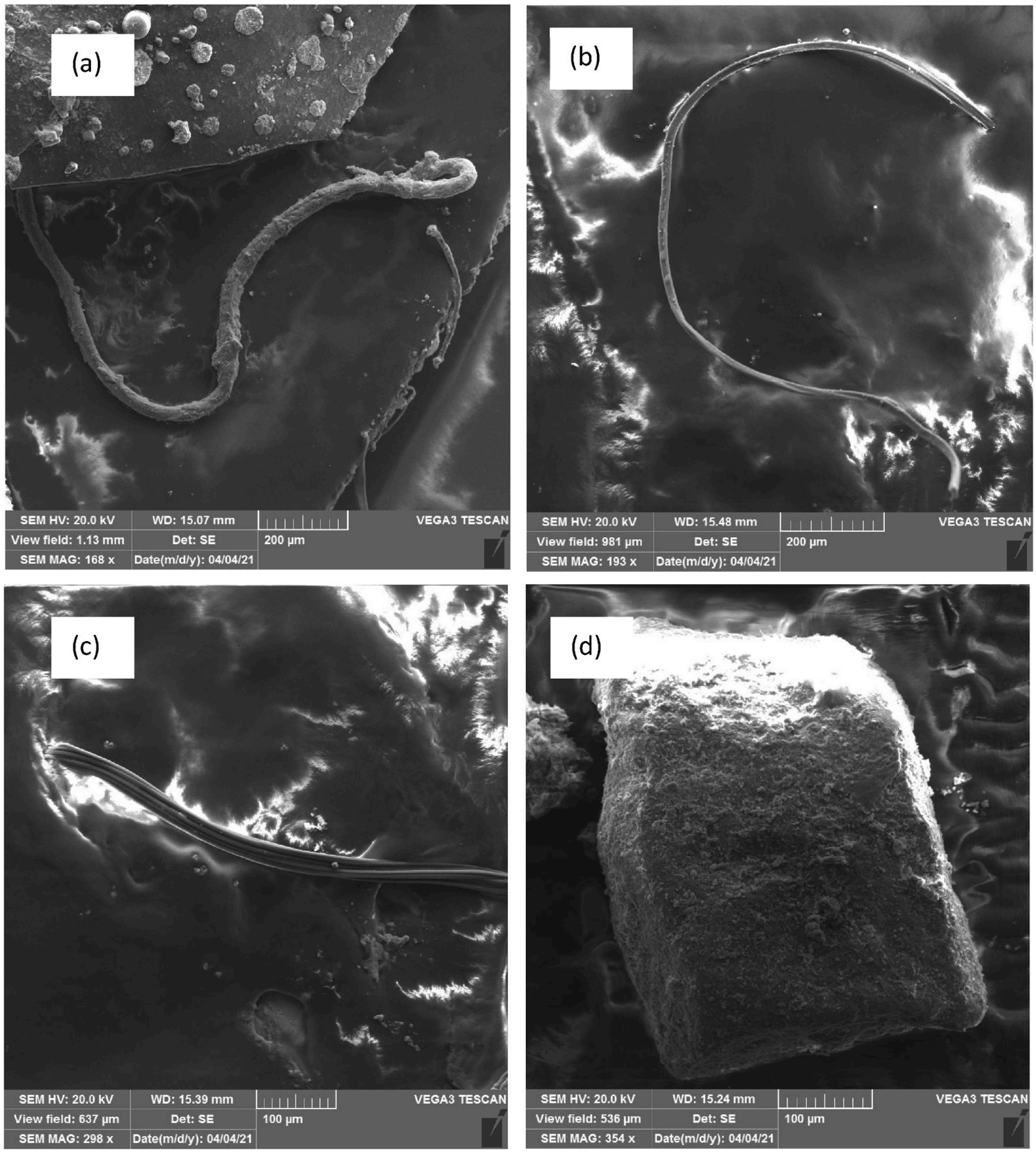

Fig. 6. SEM images of four MPs retrieved from the agricultural soils. (a) A single stranded, blue nylon fibre exhibiting oxidative weathering, (b) a smooth, single stranded, green PET fibre with little evidence of weathering, (c) a smooth, multi-stranded, blue nylon fibre with little evidence of weathering, and (d) a yellow polystyrene fragment exhibiting significant oxidative weathering.

Table 4
Concentrations and characteristics of MPs in agricultural soils (and, as a baseline and in italics, remote desert soils) reported in the literature. PA = polyamide; PET = polyethylene terephthalate; PP = polypropylene; PU = polyurethane; ns = not specified.

Location	Crops	MP g^{-1}	Measure	Main polymers	Main shape	Reference
Yangtze River region	various	0.037	(mean)	PP	fragments	Cao et al. (2021)
		0.042	(mean for vegetable)			
		0.005 to 0.253	(range)			
Wuhan suburbs	vegetable	0.32 to 12.6	(range)	PA and PP	beads and fibres	Chen et al. (2020)
Northern Chile	various	0.54 ± 0.32	(mean ± 1 sd)	acrylates, PU	fibres	Corradini et al. (2021)
Western Greece	fruit, vegetable	0.04 to 0.56	(range)	PE	films	Isari et al. (2021)
Suburban Shanghai	vegetable	0.078	(mean, topsoil)	PP	fibres	Liu et al. (2018)
		0.063	(mean, subsurface)			
Valencia, Spain	cereal, fruit	5.2	(mean)	ns	fragments	van den Berg et al. (2020)
Shouguang City, N. China	various	1.44	(mean)	PP	fragments	Yu et al. (2021)
		0.31 to 5.7	(range)			
Northwest China	various	0.04 to 0.32	(range)	PE	ns	Zhang et al. (2018)
Fars province	vegetable	0.57	(mean, topsoil)	nylon, PET	fibres	this study
		0.32	(mean, subsurface)			
Lut and Kavir	*desert*	*0.02*	*(mean)*	*nylon, PET*	*fibres*	Abbasi et al. (2021)

subsurface soils (also in MP g^{-1}). With respect to surface soils, enrichment of MPs in eroded sediments ranged from about 11 at T1 in Field 1 and where mulching had been applied to about 200 at T5 in Field 2 and where wastewater was used for irrigation, with a median value for the six locations of about 100. By comparison, wind tunnel experiments in various soils of Fars province undertaken by Rezaei et al. (2019) reported enrichment of low-density (<1 g cm^{-3}) plastics in erodible sediments of 2.8–7.6. It is possible that the white nylon collector contributed to enrichment of MPs in the eroded sediments. However, based on the fractions of white fibres retrieved from the sediments (about 0.4) and MPs that were constructed of nylon (about 0.3), we estimate that this contribution should, at most, be around 12%. Therefore, MPs are much more mobile (through suspension and saltation) than the soil particles themselves, and appear to be considerably more mobile in agricultural soils than in soils associated with other land uses. The mobility of MPs can be attributed to their lightness and low density relative to the original soils, and for fibres, a high surface area–to–volume ratio that increases drag forces and reduces settling velocity (Brahney et al., 2020). The high mobility of fibres compared with other shapes was also recently demonstrated in laboratory wind erosion simulations conducted by Bullard et al. (2021). Clearly, it follows that MPs also have lower threshold velocities for erosion, although this would not be observable (or measurable) in conventional wind tunnel experiments.

Fig. 3 also indicates some fractionation of MPs by shape that are suspended in the wind tunnel. For example, the proportion of fibres as MPs is reduced from surface soils to eroded sediments in Field 1 but is increased from soils to eroded sediments in Field 2. This may reflect differences in soil texture or the means of delivery of the microfibres to the soils. Thus, it is possible that fibres delivered in wastewater have a greater propensity to be trapped in the soil interstitial environment and become less mobile than those deposited on the soil surface from atmospheric fallout.

Overall, the median MP flux arising from the wind tunnel experiments and shown in Fig. 3 is about 0.3 MP m^{-2} s^{-1}. Assuming that threshold wind velocities (for soil erosion) are attained for 10% of the time in the region (Rezaei et al., 2019), and bearing in mind that some MPs may be eroded below the threshold wind velocity, this is equivalent to a net flux of at least 0.03 MP m^{-2} s^{-1}, or about 2600 MP m^{-2} d^{-1}. The wet and dry depositional flux of MP from the atmosphere has been empirically calculated for various remote environments and ranges from about 100 to 500 MP m^{-2} d^{-1} (Allen et al., 2019; Brahney et al., 2020), or about 0.001–0.005 MP m^{-2} s^{-1}. That is, the erosional flux of MPs from agricultural soils is about one order of magnitude greater than the depositional flux in remote areas.

There are two possible explanations for this discrepancy. Firstly, greater MP erosion from agricultural soils may result from the in situ addition of resuspendable MPs in mulch and contaminated irrigation water. The presence of microfibres in the latter, for example, may account for some of the MPs observed by SEM exhibiting limited oxidative weathering (Fig. 5). Secondly, depositional fluxes of MPs are normally measured a few meters above the ground and away from any immediate airflow-surface interactions whereas the wind tunnel captures MPs transported in suspension and by saltation into a layer of <1 m from the ground. These observations suggest that agricultural soils act as both an important temporary reservoir for MPs and a dynamic secondary source of MPs that should be considered in MP inventories, atmospheric transport models, and risk assessments for soil biota, wildlife and workers in the agriculture sector.

5. Conclusions

Despite differences in agricultural practices (wastewater application versus mulching), there were no clear differences in MP abundance in soils sampled from fields of the Fars province, Iran. Rather, MPs were heterogeneously distributed within surface and subsurface soils, both within and between fields. Fibres constituted the main type of MP and nylon and PET were the most common polymers identified. Wind tunnel experiments revealed that MPs were highly enriched (up to 200-fold) in eroded sediments compared to original soils, with fluxes of up to 1.1 MP m^{-2} s^{-1} calculated for (soil) threshold velocities. These observations reflect the high mobility of terrestrial MPs and the propensity of agricultural soils to act as both a sink and secondary source of MPs that should be factored into global inventories and transport models.

CRediT authorship contribution statement

Mahrooz Rezaei: Supervisor, Methodology, Initial Idea, Funding, Review and Editing. **Sajjad Abbasi:** Writing-original draft, Methodology, Conceptualization, Investigation, Interpretation, Review and Editing. **Haniye Pourmahmood:** MSc student, Sampling, Lab Activity. **Patryk Oleszczuk:** Investigation, Review and Editing. **Coen Ritsema:** Investigation, Review and Editing. **Andrew Turner:** Conceptualization, Investigation, Writing-final draft, Interpretation, Review and Editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envres.2022.113213.

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