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Microplastics in Sewage Sludge: Effects of Treatment

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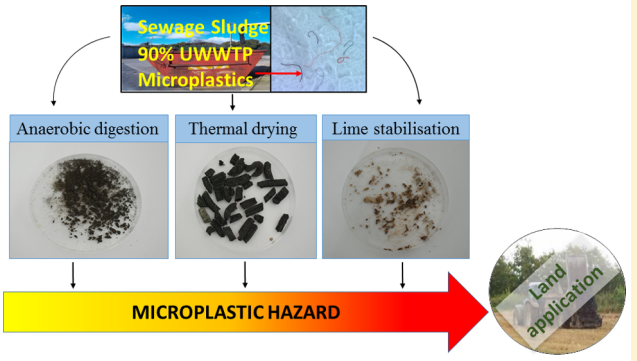
Supporting Information

ABSTRACT: Waste water treatment plants (WWTPs) are receptors for the cumulative loading of microplastics (MPs) derived from industry, landfill, domestic wastewater and stormwater. The partitioning of MPs through the settlement processes of wastewater treatment results in the majority becoming entrained in the sewage sludge. This study characterized MPs in sludge samples from seven WWTPs in Ireland which use anaerobic digestion (AD), thermal drying (TD), or lime stabilization (LS) treatment processes. Abundances ranged from 4196 to 15 385 particles kg^{-1} (dry weight). Results of a general linear mixed model (GLMM) showed significantly higher abundances of MPs in smaller size classes in the LS samples, suggesting that the treatment process of LS shears MP particles. In contrast, lower abundances of MPs found in the AD samples suggests that this process may reduce MP abundances. Surface morphologies examined using scanning electron microscopy (SEM) showed characteristics of melting and blistering of TD MPs and shredding and flaking of LS MPs. This study highlights the potential for sewage sludge treatment processes to affect the risk of MP pollution prior to land spreading and may have implications for legislation governing the application of biosolids to agricultural land.



1. INTRODUCTION

Microplastics (MPs) are synthetic polymers measuring less than 5 mm in diameter and are derived from a wide range of sources including synthetic fibers from clothing,[1,2] polymer manufacturing and processing industries,[3] and personal care products.[4] They have the potential to adsorb persistent organic contaminants[5,6] and priority metals[7−11] from the surrounding environment. These may be released upon digestion by biota or through environmental degradation, leading to possible impacts to human health and ecosystems.[12−14] Over the last 10 years, many studies have investigated the distribution[1,15] and effects[16−19] of MPs within the marine environment. Indeed, MPs have been found in Polar Regions[20] and in a range of freshwater environments worldwide.[21−24] Despite this, few studies have sought to determine land-based sources of MPs.[25] Wastewater treatment plants (WWTPs) have been identified as receptors of MP pollution and effective in capturing the majority of MPs in the sludge during settlement regimes,[26] as first found by Habib et al. when they used synthetic fibers as a proxies for the presence of sewage.[27] More than 10 million tonnes of sewage sludge were produced in WWTPs in the European Union (EU) in 2010.[28] European Union policy on sustainability and recycling of resources[29] favors the recycling of sludge. The introduction of EU legislation such as the Landfill Directive (1999/31/EEC[30]) and the Renewable Energy Directive (2009/28/EC[31]) have diverted sewage sludge from landfill and incineration into use for energy production[32] and agriculture.[33] In some countries, such as Ireland, up to 80% of municipal wastewater sludge is reused in agriculture.[34,35] Guidelines stipulate that the sludge must undergo some type of treatment (after which it is commonly referred to as "biosolids") prior to land application. This may include lime stabilization (LS), anaerobic digestion (AD), composting, or thermal drying (TD).[31] As approximately 99% of MPs are retained in sewage sludge generated in WWTPs,[36] there is a possibility that land applied sludge, even having undergone treatment, could be a source of MPs pollution.

The regulations for the use of biosolids in the EU and USA stipulate limit levels for pathogen content, maximum metal and nutrient application rates to land[37] and vector (flies and rodents) attraction reduction (U.S. only). Restrictions in land application of biosolids vary between the EU and U.S. Under U.S. federal legislation, the application of biosolids to agricultural land can occur without restriction in volume or duration, if the contamination level reaches an exceptional quality "EQ".[37] The concentration limits vary for all contaminants and to date MP pollution is not included.[37] In

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Europe, sewage sludge is dealt with very differently among member states, and application to land is banned in some countries.[38−40]

As most sewage sludge undergoes treatment prior to land-spreading, the effects of these treatments on microplastic morphology is important but remains largely unknown, with some evidence of increased abundance of fibers at a smaller size range for LS sludge[41] which was attributed to alkaline hydrolysis, facilitated by elevated pH, heat, and mechanical mixing.[41,42] Therefore, the aim of this study was to investigate the first stage of the MP pathway post-WWTP, and the impacts of different treatments. In particular, it aimed to determine if (1) MPs are present in treated sewage sludge from a range of WWTPs employing AD, TD, and LS as treatment techniques, and (2) the type of treatment used (TD, AD, LS) employed at the WWTP impacts on MP abundance and characteristics, including size and surface morphology.

2. MATERIALS AND METHODS

2.1. WWTP Sludge Sample Collection and Preparation. Sewage sludge, having undergone treatment including TD, AD or LS, was collected from seven waste WWTPs with population equivalents (PEs) ranging from 6500 to 2.4 million (Table 1). These WWTPs received wastewater from industry, stormwater runoff and domestic sources, all of which comprised up to 30% of the influent organic loading (measured as biochemical oxygen demand, BOD) (Table 1). Due to an intense difficulty in gaining access to these WWTPs and the primary focus being in the chacterization of sewage sludge applied to land (post-treatment), pretreatment samples were not taken. Three replicate samples of 30 g were obtained from each WWTP and stored at −20 °C prior to sample preparation. The treated sewage sludge had dry matter (DM) contents ranging from 24% (AD) to 87% (TD). Pellets of TD sludge were placed in water for 1 week to induce softening, transferred to a water bath (30 °C) for 24 h, and placed in an "end-over-end" shaker (Parvalux, UK) for 12 h. This shaking procedure was repeated until the pellets were sufficiently softened without compromising the physical characteristics of the MPs. The samples were subsequently washed through a 250 μm sieve, which resulted in complete degradation of the pelleted clumps prior to elutriation. As handling and identification of MPs becomes unfeasible below 200 μm using available instrumentation, and most studies have examined MPs above this size,[44] the sieve size (250 μm) was selected to optimize efficiency of MP extraction and identification. Although fibers are only 10−20 μm in diameter, they are generally longer than 250 μm and curved to some degree and the majority are therefore trapped in the 250 μm sieve.[20] A proportion of the washed through fraction (WTF) was retained and passed through 212, 63, and 45 μm sieves for particle size determination or particle size fractionation. These sieve sizes were chosen to include the majority of smallest particles and therefore give an impression of treatment effects of sludge particle size.

Anaerobically digested and LS sludge were soaked in filtered tap water to soften and homogenize them, and were also washed through 250, 212, 63, and 45 μm sieves to determine particle size fractions. As the LS sludge had an oily appearance, thought to be derived from the breakdown of cellulosic material through alkaline hydrolysis, it was decided that the elutriation and other density separation techniques were unsuitable for extraction of MPs. Instead, 10 g from each replicate sample were examined by passing it directly through a filter (GF/C: Whatman, 1.2 μm) using vacuum filtration.

2.2. Microplastics Extraction. *2.2.1. Elutriation.* The principal of elutriation was used as the first step in the separation of MPs from other sample components. Elutriation separates lighter particles from heavier ones through an upward flow of liquid and/or gas, and has been widely used in the separation of biota within sediment samples.[42] To separate MPs from the sludge samples, an elutriation column, based on the design of Claessens et al.[43] was constructed.

2.2.1.1. Column Extraction Efficiency Estimation. To check for efficiency of the column in extracting MP, three sediment samples, each weighing 40 g, were spiked with 50 microplastic particles of high density polyethylene (HDPE) (three colors) and PVC, and run through the column. The HDPE samples used were shavings of approximately 1.0 (L) × 4.0 (W) × 2.0 mm (B). The PVC particles were of a similar dimension, but were more brittle. Therefore, each particle was marked with a blue marker to ensure that particles were not counted twice upon recovery. The number of particles, separated from the sediment matrix, that exited the column, was enumerated and the percentage efficiency was calculated.

2.2.2. Zinc Chloride ($ZnCl_2$) Extraction. The microplastic extraction was filtered through 250 μm mesh, rinsed into a separatory funnel with 1 molar $ZnCl_2$ solution, and brought to a volume of 300 mL. The funnel was plugged, vigorously shaken for 1 min, and allowed to settle (20 min). The settled material was drained and the remainder of the sample was filtered onto glass fiber filters (GF/C: Whatman, 1.2 μm). The oily appearance of the LS samples rendered this density separation technique unsuitable for extraction of MP.

2.3. Characterization of MPs. The filters were examined using stereomicroscopy equipped with a polarizer (Olympus SZX10) attachment and a Qimaging Retiga 2000R digital camera. Microplastics were identified and enumerated based on several criteria including form, color and sheen used in previous studies as described by Hidalgo-Ruz et al.[44] For fibers, the form of a synthetic fiber should not taper at either end, while not having rigidly straight form. Any polymer will not have cellular

Table 1. Characteristics of Municipal Wastewater Treatment Sites Investigated (adapted from Healy et al., 2016)

site	WWTP/ agglomeration size (PEs)	landfill leachate as % of influent BOD load	industrial, and domestic septic tank sludge[a] as % of influent BOD load	type of treatment
1A	2 362 329	<0.01	<0.01	thermal drying, anaerobic digestion
1B	284 696	0.3	24	thermal drying
2	179 000	unknown	30	anaerobic digestion
3	130 000	unknown	0.008	thermal drying
4	101 000	2.0	unknown	lime stabilization
5	31 788	0.25	unknown	lime stabilization
6	25 000	0.7	0	thermal drying
7	6500	unknown	unknown	thermal drying

[a]Most recent available figures in all WWTPs (2013).

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structure or other organic structures. Artificial fibers/particles also have uniformity of color and exhibit a sheen once passed through the polarized light. Where ambiguity remained following these observations, the suspected polymer was manipulated with a hot pin by which a melted form indicated a positive result. Microplastics were measured and allotted to the following size categories: 250−400 μm, 400−600 μm, 600−1000 μm, and 1000−4000 μm. Suspected microplastics were enumerated and measured and approximately 10% of MP specimens from each filter paper were set aside for polymer identification. This was deemed sufficient as the cumulative abundance of MP concentration within the sludge systems made it unfeasible to examine each MP particle. As MPs were randomly selected, they covered a range of size-classes and morphological types. Microplastics for which any ambiguity remained as to whether it was a polymer or not were automatically selected for analyses.

Attenuated total reflectance (ATR) and Fourier transform infrared spectroscopy (FTIR) (PerkinElmer, Spectrum Two with Universal ATR Accessory and Thermo Scientific, UK, Nicolet iN10 FTIR microscope with germanium Tip Slide-on-ATR) were used to analyze approximately 10% of MP specimens. The spectra were obtained with 3-s data collection (16 scans per sample) over the wavenumber range of 600−4000 cm^{-1} using a liquid nitrogen-cooled MCT-A detector at 8 cm^{-1} resolution. SEM analyses was carried out in order to examine the surface structures of MPs of LS and TD sludges. A selection of MP samples (~20) extracted from the sludge (and pristine plastics for comparative purposes) were gold-coated (Emitecg K550, Quorum technologies, Ltd., UK) and subjected to variable pressure scanning electron microscopy (SEM) in secondary electron mode using a Hitachi model S2600N (Hitachinaka, Japan). The analyses were performed at accelerating voltages of 10−20 kV; an emission current (I_c) of 10 μA; working distance of 12−24 mm.[45] MP samples chosen where mostly those which were easily divided in 2 to allow for simultaneous polymer identification and SEM analyses. There was therefore a bias to larger MPs

2.4. Quality Control and Contamination Prevention. Cotton laboratory coats and nitrile gloves were used during the sample preparation and analyses. In addition, synthetic clothing was avoided and samples were covered at all times and working surfaces were cleaned with alcohol prior to use. When analyzing filter papers, a blank filter paper was exposed to the open laboratory conditions to assess the possibility of air-borne contamination.

2.5. Data Analyses. Statistical analyses were carried out using Minitab 17 (2010) and R (R Core Team, 2012). As data were not normally distributed, nonparametric tests were used to test for differences in MP abundances among locations (Mann−Whitney Test). To investigate if there were any possible effects of PE on abundance, a Spearman's rank correlation analysis test was utilized. With the exception of one WWTP, there was only one treatment method employed per site (Table 1), so in-site correlation was not possible. Each site was treated as an independent measurement and plotted using a box plot. A GLMM (generalized linear mixed effect model; eq 1) was used to investigate the high number of MP particles in the smaller class sizes at WWTPs in which LS was used.

$$\text{microplastic counts} = \text{Treatment Type} + \text{population equivalent} + \frac{1}{\text{treatment plant}} \quad (1)$$

Where *1/treatment plant* specifies a random intercept model.

A separate GLMM for each size class was carried out using a Poisson distribution and a random effect term to account for nesting of replicates within WWTPs to determine which explanatory variable was responsible for larger proportions of smaller MP particles at lime stabilized WWTPs.

3. RESULTS AND DISCUSSION

3.1. Characterization of Treated Sewage Sludge. The characteristics of the sewage sludge treated using AD, LS and TD had varying physical characteristics. The particle size fractionation (g/kg) of the AD samples was smaller than the LS and TD samples (Table 2), and had a sandy appearance. The

Table 2. Particle Size Fraction (g) of Lime Stabilized (LS), Anaerobically Digested (AD) and Thermally Dried (TD) Samples (40 g)

	treatment type		
size fraction	LS	AD	TD
>212 μm	3.004 ± 0.550	31.753 ± 0.578	35.503 ± 0.661
>63 μm	27.410 ± 0.840	7.948 ± 0.7778	3.593 ± 0.894
>45 μm	9.400 ± 1.166	0.327 ± 0.241	0.930 ± 0.486
<45 μm	0.200 ± 0.213	0.000 ± 0.00	0.000 ± 0.000

AD samples were very dark and heavy with some cellulosic material, whereas the TD samples had a lot of cellulosic material entrained, which was difficult to separate during elutriation and zinc chloride extraction. Although this cellulosic material was distinctive from MP material (in that its fibers tapered at the ends and it was often branched) and therefore easy to disqualify, its presence in the samples increased greatly the time and consumables (filter papers) utilized during the filtration process. High levels of cellulose derived from toilet paper in sewage may merit the inclusion of a digestion process using the cellulase enzyme as has been previously used for the isolation of MPs in North Sea sediments.[46]

3.2. Microplastics Extraction. *3.2.1. Elutriation Column Extraction Efficiency Estimation.* The average extraction efficiency rate of the elutriation column for the spiked sediment samples was 90%, 94% and 91% for the red, blue and black HDPE particles, respectively. The elutriation process was less efficient for the PVC particles, which resulted in an average extraction efficiency of 80%. This is an indication that results of MP abundance in this study may be an underestimation. As the efficiency test was carried only for fragments at one size only, it may not be representative of efficiency of fiber removal.

3.3. Characterization of MPs. *3.3.1. Microplastics Abundance.* MPs extracted from the biosolids ranged from an average of 4196 to 15 385 particles kg^{-1} (DM) among the seven sites, with significant differences in MP abundances between some sites and within Site 1 (1A, 1B) between AD samples and TD samples (Mann−Whitney, $w = 15$, $p = 0.0809$; Figure 1). This is likely to be an underestimation due to loses in column efficiency (approximately 20%) and through the use of a 250 μm sieve from which a proportion of fibers may be lost.[20] Airbourne contamination on the control filter papers was

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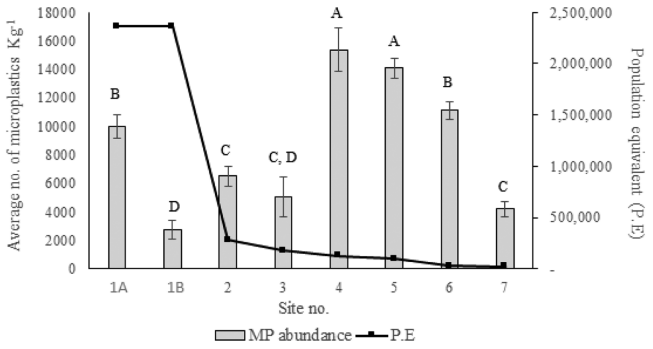

Figure 1. Average abundances and corresponding population equivalents of microplastics at seven sites. Sites sharing the same letter are not significantly different (Mann-Whitney-U test, $p > 0.005$).

negligible (1−2 fibers per day) due to high abundance of MPs in these samples. The use of a different method using a smaller sample size after sieving (250 μm) for LS samples (sites 5,6), may have had an effect on the quantification as no loss of MPs would have occurred during the elutriation process. This may have led to the possible inflation of MP abundances in LS samples compared to TD and AD sludges. The abundances found in this study are in the same order of magnitude as the study by Zubris and Richards[40] which reported between 3000 and 4000 particles kg^{-1}. In this study, a lack of correlation between PE and MP abundance kg^{-1} (Spearman's rank, $r = -0.308$, $p = 0.458$) implies that these differences may have been due to the variation of source inputs (industrial, stormwater, landfill etc.). However, as no data exists on the temporal variation of MPs in sewage sludge, the possibility also exists that these variations are a result of fluxes in MP input which could be a result of peak MP emission times in relation to household and industrial activity etc. In this study, the significantly lower abundance of MPs reported for an anaerobically digested biosolid sample (1B) which was collected from the same site as sample 1A (TD) compared to all other sample except site 3 (also AD) (taken roughly at the same time), posits an interesting question over the possible role of AD in the degradation of MPs. Without pretreatment samples, there is no evidence to prove that the Mesophilic anaerobic digestion (MAD) used at the AD treatment plants in this study, facilitated the breakdown of MPs and few studies have investigated the breakdown of polymers in anaerobic digesters. However, one pilot study investigated the effect of plastic waste on the functioning of anaerobic digestion, finding that digesters from which plastic was removed produced less gas than those to which plastic was added.[47] As there is already substantial evidence of microbial breakdown of polymers through the activity of exoenzymes (promoting depolymerization) and assimilation of smaller articles resulting in mineralization,[48−50] the role of degradation by microorganisms within the AD systems should be further investigated.

3.3.2. Morphological Categorization and Polymer Identification of MPs. This study confirmed that MPs are retained in the sewage sludge and are largely composed of fibers, similar to what was found by Talvite et al.[47] and Magnusson and Norén.[36]

Approximately 75.8% of the MP consisted of fibers, followed by fragments, films, other unidentified particles, and spheres, which accounted for only 0.3% of total MP abundance (Table 3). The greatest proportion of MP fragments was found at the LS WWTPs, with Site 6 being the only site to have marginally more fragments than fibers (Table 3; Figure 2). Polymers,

Table 3. Breakdown of Types of Average Microplastic Abundance kg^{-1} (Dry Matter) Among Sites

		microplastic types				
site no.	treatment	fibres	fragments	films	spheres	other
1A	TD	9113	511	255	89	44
1B	AD	2065	611	67	0	0
2	TD	5583	588	222	44	67
3	AD	4007	855	111	33	150
4	TD	13 675	1143	366	33	178
5	LS	10 778	3075	122	11	78
6	LS	4762	5228	11	0	11
7	TD	3463	511	167	0	56
total		**53 447**	**12 521**	**1321**	**211**	**583**
%		78.5	18.4	1.9	0.3	0.9

identified by FTIR, comprised HDPE, polyethylene (PE) polyester, acrylic, polyethylene terephthalate (PET), polypropylene, and polyamide (Figure 3). Some of these contained minerals. Although wastewater derived from households generate high quantities of fibers, principally derived from clothes washing of >1900 fibers per wash,[1] other industrial sources of fibers such as the fiber manufacturing industry may also be important contributors.

3.3.3. Size of MPs. Using the fitted coefficients from the GLMM, hypotheses of no difference between all pairwise combinations of the treatment effects were tested. At small and medium particle sizes, the LS treatment was significantly different from both TD and AD treatments (Figure 4; $P < 0.001$; sizes classes A and C; $P < 0.05$ size class B). During the analyses, raising the MP characteristics (abundance and size), from the smaller sample size processed for LS samples (10 g) to the equivalent of that in the TD and AD samples (30 g), may have reduced the representativeness of the size-classes within LS samples. However, from the sludge particle size fractionation, where 40 g were sampled from LS, TD and AD sludges, there was also a larger proportion of sludge (g) in size classes <45 μm, > 45 μm and >63 μm, while lower sludge mass at the larger size fraction >212 μm (3 g compared to 32 and 35 respectively). This corresponds to the higher number of MPs at smaller size classes in LS samples derived from our analyses and therefore suggests that the mechanisms that resulted in smaller sludge particle size may be responsible for MP size distribution alike. In this study, as it was not possible to obtain pretreatment samples, it is not possible to wholly assign the differences in size classes to the treatment processes. However, the elevated numbers at the small size classes for LS samples, are in agreement with results reported by Zubris and Richards.[40] In their study, there was evidence of elevated abundance of MPs at smaller size classes for LS samples which were derived from the same sludge pretreatment batch. The authors attributed this to elevated pH combined with mechanical mixing. In a more recent study, Cole et al.[17] found partial destruction of nylon fibers and melding of polyethylene fragments using 10 M NaOH at 60 °C. It is likely, therefore that in this study, the combination of elevated pH, temperature and mechanical mixing could be responsible for the elevated numbers of MPs in smaller sizes classes. As alkaline-stabilized sludges have been associated with higher levels of metals compared to other sludges,[51] the possibility of shearing effects should be investigated further to identify potential impacts.

3.3.4. Surface Morphologies of MPs. Scanning electron micrographs of surface textures of polymers entrained in the

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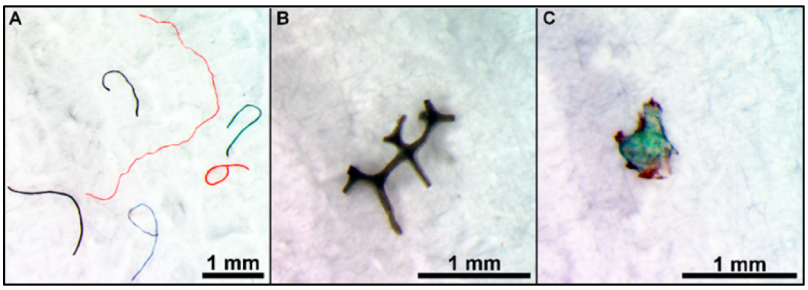

Figure 2. Stereomicrograph of mircoplastics fibers (A), other (B), and fragment (C).

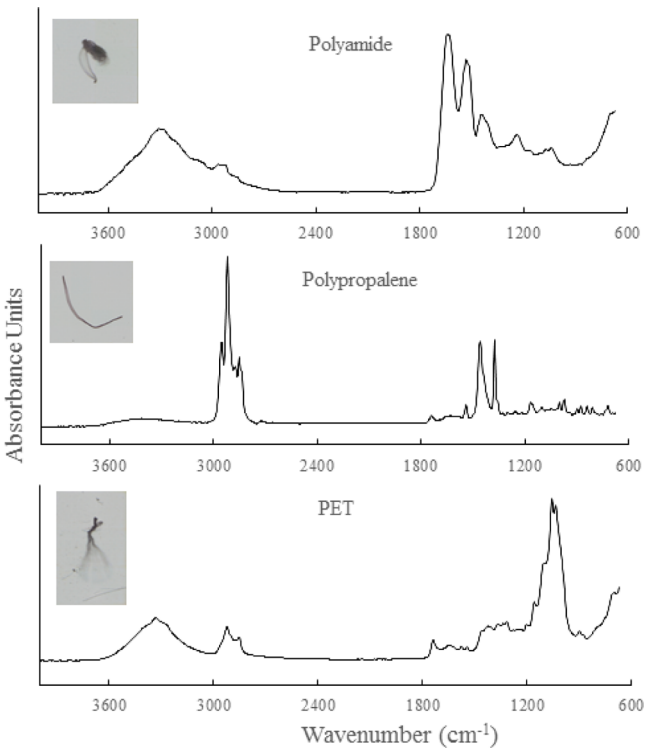

Figure 3. Fourier Transform Infrared Spectroscopy (FTIR) spectra within specimen photographs of polyamide, polypropylene, and polyethylene terephthalate (PET).

treated biosolids had some surface morphologies, which varied among treatment type. An unknown polymer fiber, which was thermally dried, had distinct blistering and fracturing, particularly in the fiber curves (Figure 5A–C). Additionally, polymer fragments from TD samples, identified as HDPE and PE fragments, showed wrinkling, melding and some fracturing, which was quite distinct from pretreatment samples (Figure 6: G-I; Figure 7D–F). Surface morphologies of MPs originating from LS biosolids had a more shredded and flaked appearance for the unknown polymer (Figure 5D–F) and a HDPE specimen (Figure 5D–F). This is in accordance to some degree with the study by Zubris and Richards,[40] which reported surface morphologies exhibiting high abrasion and appeared "very brittle". Anaerobically digested samples of an unknown polymer had deep cleavage, which was distinct from any other observations (Figure 5G–I). It is not possible to wholly assign these observed surface morphologies to effects treatment processes due to low replication and the absence of comparison of surface morphologies of these MPs prior to entering the treatment plant. However, these images provide some idea of initial fracturing patterns which can be observed in MPs which have traveled thus far in their pollution pathway.

4. CONCLUSIONS

Although it was not possible to assign wholly the abundances or size distributions to the treatment processes, results suggest that treatment processes may have an effect. If MPs are altered by treatment, the potential for impact may also be influenced accordingly. This could add to the unknown risks associated

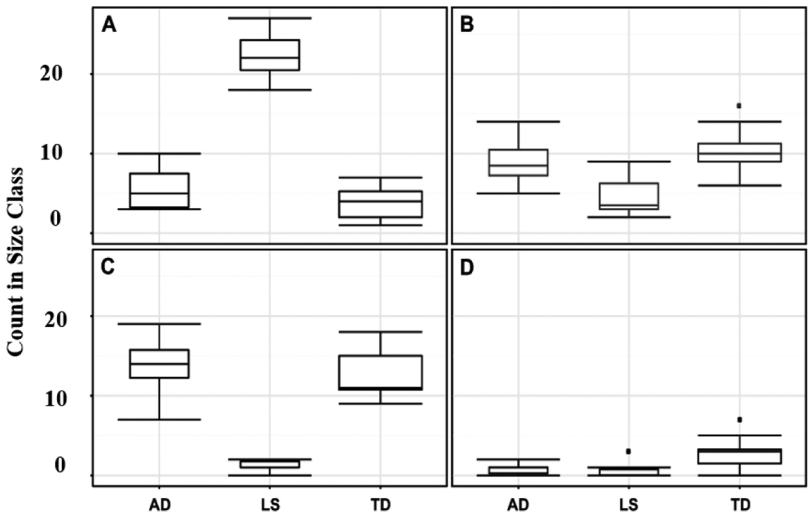

Figure 4. Abundance of microplastics in different size classes (A: 250–400 μm, B: 400–600 μm, C: 600–1000 μm, D: 1000–4000 μm) as a function of treatment type.

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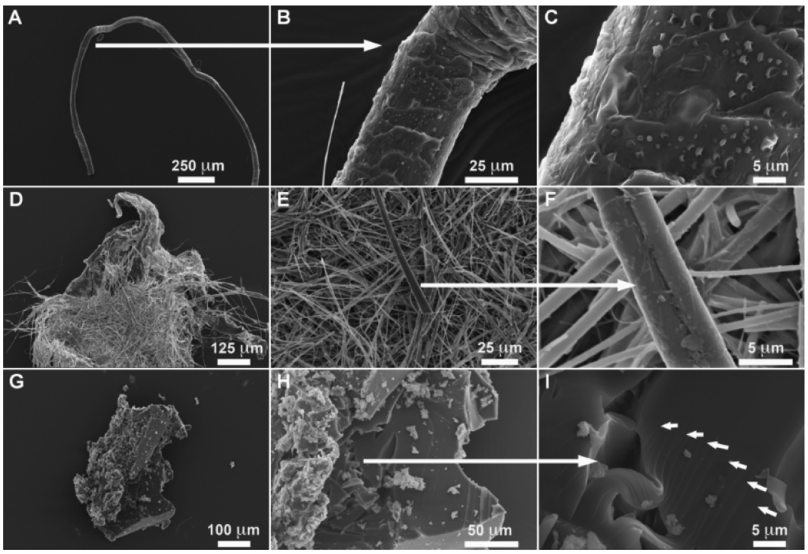

Figure 5. Diversity in morphology and surface texture of microplastics isolated from treated sewage sludge. Scanning electron micrographs of fibrous particle from thermally dried (TD) biosolids (A–C). Multi fibrous particle from lime stabilized (LS) biosolids (D–F). Overview of nonfibrous particle from anaerobically digested (AD) biosolids (G–H). Presence of lamellae or cleavage planes (arrow heads) on microplastic surface (I).

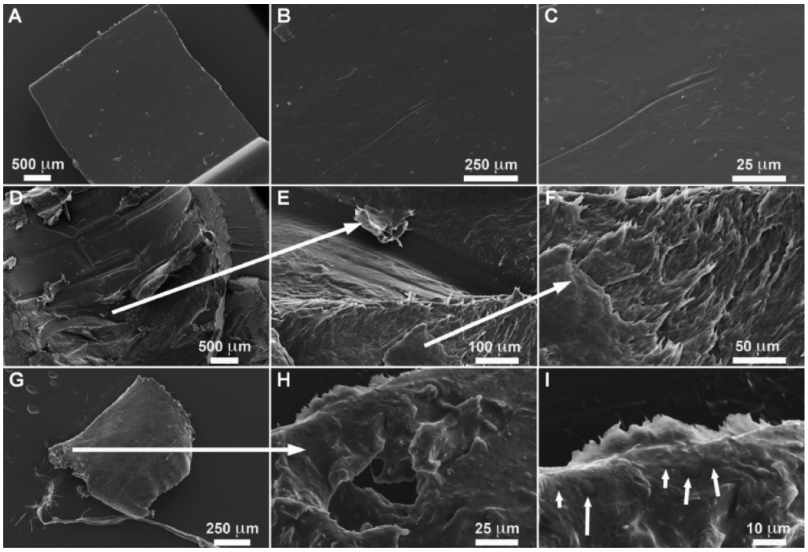

Figure 6. Morphological and surface texture comparison between pretreatment high density polyethylene (HDPE) and HDPE particles isolated from treated sewage sludge. Scanning electron images of pretreatment HDPE (A–C) showing smooth nondegraded surface. Scanning electron micrographs of HDPE particle from lime stabilized (LS) biosolids (D–F) showing altered and weathered surface texture. Scanning electron micrograph of HDPE particle from thermally dried (TD) biosolids (G–I) with evidence of blistering effect (arrow heads) on polymer surface (I).

with MPs in sewage sludge. Regardless of treatment regimes, over time, there may be consequences for the accumulation of MPs in terrestrial, freshwater, or marine ecosystems derived from land-spreading of sewage sludge or biosolids.

MPs entrained in biosolids which are applied to land, may be degraded through photodegradation and thermo-oxidative degradation.[52,53] exacerbating the problem of land-spread MP pollution. The interaction of MPs with contaminants in the soil, could have major consequences for the absorption and transportation of contamination elsewhere. Surface weathering and the subsequent attachment of organic matter and the resulting negative charge attracts metals including cadmium (Cd), lead (Pb), and zinc (Zn).[54,55] Whether agricultural land is a sink or a source of MP pollution remains unclear. Microplastic fibers have been found on land 15 years post application, and some evidence of vertical translocation through the soil has also been found.[40] Possible impacts arising from land-applied MPs begin in the terrestrial ecosystem with implications for terrestrial species such as earth worms[56] and birds feeding on terrestrial ecosystems.[57] As legislation in the EU and the U.S. generally permit the land application of sewage sludge, there is a strong possibility that large amounts of MPs are emitted to freshwater, where currently little is known on their impacts to species and habitats.[58] Furthermore, buffer zones around freshwater bodies, which may be stipulated in "codes of good practice", do not take into account the mechanisms of transportation of MP vertically through the soil or with surface runoff following a precipitation event. While legislation currently takes into account pathogens as well as nutrient and metal concentrations of treated sludge[59] it does not consider the presence of MPs within the sludge, and their associated risks. The predicted exponential growth of the

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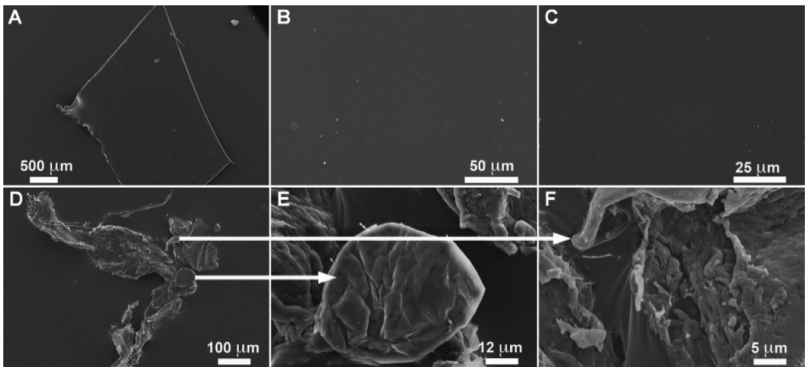

Figure 7. Morphological and surface texture comparison between pretreatment polyethylene (PE) and PE particle isolated from sewage sludge. Scanning electron images of pretreatment PE (A–C) with unaltered surface. Scanning electron micrographs of PE particle from thermally dried (TD) biosolids (D–F) showing wrinkling and fracturing of polymer surface.

plastics industry for the coming years[60] may be accompanied by a significant increase in MPs in the waste stream. Therefore, vigilant management of cumulative sources of MPs such as sewage sludge or biosolids is necessary. In particular, this study has highlighted the potential for treatment processes to alter the counts of MPs which in-turn increases the available area for absorption/desorption of organic pollutants

Recommendations for Further Studies. Further investigations are required to investigate accelerated proliferation of MP pollution through sludge treatment processes. In particular, the role of degradation by microorganisms within the AD systems should be further investigated as a potential remediation method. Knowledge gaps regarding the factors critical for the mobilization and transport of MPs which are likely to affect the pathway attenuation of land-spread sewage sludge MP pollution should to be addressed in order to determine MP flow within the terrestrial system and to freshwater systems. Only when the knowledge is acquired, can we estimate exposure and associated risks to the environment from MP pollution.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge on the ACS Publications website at DOI: 10.1021/acs.est.6b04048.

Detailed description of the dimensions of the elutriation column, accompanied by a photograph and schematic representation. Flow rates and technique used for extraction of MPs using the elutriation column are also included (PDF)

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