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Review

Fate and effects of microplastics in wastewater treatment processes

Mengjie Wu [a,b,1], Wenchang Tang [a,b,1], Shaohua Wu [a,*], Hongyu Liu [b], Chunping Yang [a,b,c,*]

[a] Guangdong Provincial Key Laboratory of Petrochemical Pollution Process and Control, School of Environmental Science and Engineering, Guangdong University of Petrochemical Technology, Maoming, Guangdong 525000, China

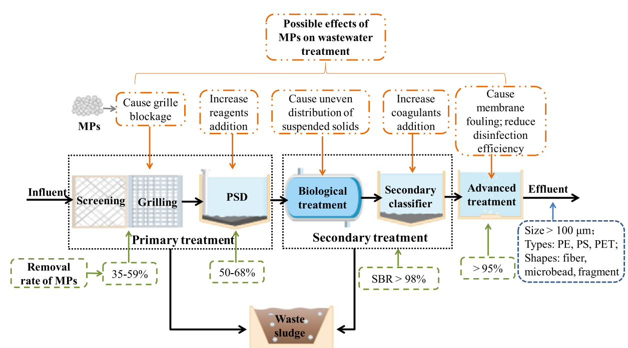
[b] College of Environmental Science and Engineering, Hunan University, and Key Laboratory of Environmental Biology and Pollution Control (Hunan University), Ministry of Education, Changsha, Hunan 410082, China

[c] Hunan Province Environmental Protection Engineering Center for Organic Pollution Control of Urban Water and Wastewater, Changsha, Hunan 410001, China

HIGHLIGHTS

- Secondary treatment is regarded as the most efficient process to remove MPs.
- Presence of MPs can increase reagent addition and cause membrane fouling in WWTPs.
- MPs can affect nitrification and denitrification of AS by disturbing AOB and NOB.
- Mechanisms for affecting AS at the presence of MPs are insufficiently investigated.

GRAPHICAL ABSTRACT



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ABSTRACT

Microplastics (MPs) have garnered growing attention of researchers, as they are proved to be hazardous to the environment and humans. Wastewater treatment plants (WWTPs) are deemed as an important releasing source of MPs to the environment, and thus it is of significance to study the behavior of MPs in WWTPs. In this review, the fate of MPs in WWTPs and their effects on different wastewater treatment processes have been comprehensively discussed. Studies have shown that the secondary treatment is the most efficient process to remove MPs from wastewaters with a removal rate around 98%. The presence of MPs can increase reagent addition dosage, inhibit nitrogen conversion rate, and cause membrane fouling in wastewater treatment processes. Besides, the influences of MPs on activated sludge mainly exert on nitrification and denitrification processes, sludge digestion, and microbial communities. However, it is worth noting that different methods have been employed to determine the concentrations of MPs in WWTPs. As a result, the removal performance on MPs in WWTPs is difficult to be accurately assessed. Moreover, complicated interaction among MPs and other environmental pollutants may expand the impacts of MPs on wastewater treatment processes, which still remains insufficiently investigated. Therefore, this review has also proposed some knowledge gaps existing in present MP studies in WWTPs, and would provide reference to alleviate the adverse effects of MPs for future research.

Abbreviations: MPs, microplastics; PE, polyethylene; PP, polypropylene; PS, polystyrene; PVC, polyvinylchloride; PAM, polyacrylamide; PET, polyethylene terephthalate; HDPE, high density polyethylene; POPs, persistent organic pollutants; WWTPs, wastewater treatment plants; AD, anaerobic digestion; PSD, primary sedimentation; AS, activated sludge; HRT, hydraulic retention time; SRT, sludge retention time; OD, oxidation ditch; A/A/O, anaerobic/anoxic/aerobic; A/O, anaerobic/aerobic; BAF, biological aerated filter; GSF, gravity sand filtration; DF, discfilter; DAF, dissolved air floatation; MBR, membrane bioreactor; UF, ultrafiltration; GAC, granular activated carbon; TS, total solids; SBR, sequence batch reactor; PES, polyester; PA66, polyamide 66; AGS, aerobic granular sludge; AOB, ammonia-oxidizing bacteria; NOB, nitrite-oxidizing bacteria; COD, chemical oxygen demand; NH_4^+-N, ammonia nitrogen; N/A, not available; UASB, upflow granular anaerobic sludge blanket; AnSBR, anaerobic membrane bioreactor.

* Corresponding authors.
E-mail addresses: yangc@hnu.edu.cn (C.P. Yang), wushaohua@hnu.edu.cn (S.H. Wu).

[1] These authors contributed equally to this article.

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Contents

1. Introduction

Microplastics (MPs) refer to tiny plastic particles with upper size boundary less than 5 mm in diameter (GESAMP, 2019). It has been reported that MPs are widely detected in different ecosystems especially in the marine environment (Jambeck et al., 2015; Pan et al., 2019; Sutton et al., 2016). Data showed that the average concentration of MPs in surface water of the mid-west Pacific Ocean and in sub-surface water of the Atlantic Ocean reached 34,039 ± 25,101 pieces/km^2 and 1.15 ± 1.45 particles/m^3, respectively (Wang et al., 2020a; Kanhai et al., 2017). Commonly discovered polymer types of MPs are polyethylene (PE), polypropylene (PP), polystyrene (PS), polyvinylchloride (PVC), polyacrylamide (PAM), and polyethylene terephthalate (PET) (Cho et al., 2019; Pivokonsky et al., 2018). Previous studies demonstrated that MPs could act as vectors of hazardous materials including persistent organic pollutants (POPs) and heavy metals (Brennecke et al., 2016; Chua et al., 2014; Rochman et al., 2015; Rodrigues et al., 2019; Zhou et al., 2019). Consequently, various environmental and health problems may be induced by the ubiquitous MPs (Emadian et al., 2017; Moharir and Kumar, 2019; Wu et al., 2020).

Wastewater treatment plants (WWTPs) are usually considered as barriers to prevent contaminants from entering into the environments; however, they are also significant point sources for MP pollution (Bayo et al., 2020; Browne et al., 2011; Mourgkogiannis et al., 2018; Talvitie et al., 2017a). In addition to industrial wastewater, domestic wastewater is another important origin of MPs in WWTPs (Akarsu et al., 2020; Ben-David et al., 2021; Kelly et al., 2019). MPs come from manifold sources. For instance, cosmetics, personal care products, and textiles wastes comprise a considerable number of MPs and they may discharge into drainage systems and eventually flow into waters and soils (Browne et al., 2007; Edo et al., 2020; Prata, 2018). Several studies suggested that the removal efficiency of MPs in conventional WWTPs was higher than 97% (Carr et al., 2016; Lv et al., 2019; Murphy et al., 2016; Turan et al., 2021; Xu et al., 2019). Nevertheless, as most MPs are retained in sewage sludge, which is a frequently used soil amendment, they may also pose a threat to the terrestrial environment (Gherghel et al., 2019). Moreover, regarding the huge volumes of wastewater discharged by WWTPs, even a small proportion of MPs existing in the effluent could have a great influence on the environment.

Wastewater treatment processes in WWTPs are applied to improve the quality of effluent but not necessarily aim to remove MPs from wastewaters (Mason et al., 2016). However, with the development of advanced final-stage treatment technologies, researchers suggested that the removal efficiency of MPs could be accordingly enhanced (Carr et al., 2016; Mintenig et al., 2017; Ziajahromi et al., 2017). Increasing research investigating the influences of different treatment processes on the removal performance of MPs has emerged in the last two decades (Prata, 2018; Ruan et al., 2019; Talvitie et al., 2017a). For example, Hidayaturrahman and Lee (2019) studied the fate of MPs in different treatment stages of 3 WWTPs and found that the removal rate of MPs by a tertiary treatment could increase to more than 98%. Conley et al. (2019) measured MP loading and removal efficiencies in 3 WWTPs in South Carolina and estimated that these WWTPs could reduce 99.9% of plastic debris input into the environment. Mason et al. (2016) demonstrated that MP concentrations varied in effluent of different WWTPs, proving that different treatment processes played different roles in the removal of MPs. Besides, it was reported that MPs showed a negative impact on activated sludge (AS), as they disturbed the microbial communities and inhibited the functions of sludge such as hydrogen production and anaerobic digestion (AD) (Corradini et al., 2019; Wei et al., 2019a; Wei et al., 2019c). There are also growing number of review articles concerning MPs and WWTPs, whose major interest is analyzing concentrations and compositions of MPs and assessing the removal efficiency of MPs in WWTPs (Enfrin et al., 2019; Turan et al., 2021; Zhang and Chen, 2020; Zhang et al., 2020). However, current reviews discussing both the fate of MPs in WWTPs and their potential impacts on wastewater treatment processes are still scanty. Besides, more research is required to disclose the intricate interaction between MPs and different wastewater treatment processes, so as to minimize the adverse effects of MPs on wastewater treatment performance.

Therefore, the objectives of this paper were to: (i) investigate the fate of MPs in WWTPs by comparing their removal performances in different wastewater treatment processes; (ii) summarize possible effects of MPs on wastewater and sludge treatment; (iii) reveal research gaps in current studies of MPs and wastewater treatment, and propose research prospects on this field.

2. Methodology

Relevant literature from 2010 to June 2020 was retrieved from the database of ScienceDirect. Search strings used were “microplastics” AND (“wastewater treatment” OR “wastewater treatment plant”). A total of 1,239 references were returned and 232 of them were identified as candidate publications. We read the abstract, introduction and conclusion part of all these 232 articles, and grouped them into two subtopics, i.e., fate of MPs in WWTPs and effects of MPs on wastewater treatment processes. These candidate publications were not necessarily shown in the reference list.

3. Fate of MPs in WWTPs

3.1. Effects of pretreatment processes on MP removal

Pretreatment processes are involved in the preliminary and primary treatment in WWTPs. Preliminary treatment is set to separate easily removed particles such as bulky floating or suspended solids and grit

(Enfrin et al., 2019; Pal, 2017). As a result, it is inefficient to remove tiny MPs from wastewaters. It was reported that only 35–59% of MPs could be removed by preliminary treatment in WWTPs (Sun et al., 2019). Primary treatment aims to make relatively heavy substances sink to the bottom of the storage tank as sludge (Forstner et al., 2019). Primary sedimentation (PSD) is used to remove lighter suspended solids, and it is the major process for MP removal in the primary treatment. Suspended and settleable MPs with relatively larger particle sizes can be removed in primary clarifier (Talvitie et al., 2017b). Studies showed that 50–98% of MPs could be removed during the primary treatment (Dris et al., 2015; Michielssen et al., 2016; Murphy et al., 2016).

It is suggested that PSD would play a vital role in the size and shape distribution of MPs (Murphy et al., 2016). In terms of size distribution in PSD process, larger MPs (particle size ranging from 300 to 5,000 μm) can be efficiently removed. For instance, Talvitie et al. (2017a) indicated that the primary treatment could efficiently remove MPs with the particle size ≥300 μm, and particles in the smallest fraction (20–100 μm) became the most abundant. Dris et al. (2015) showed that the proportion of large MPs (1,000–5,000 μm) decreased from 45% to 7% after PSD. Claessens et al. (2011) pointed out that a small amount of MPs ≥ 300 μm could be collected in the primary clarifier. Regarding MP shapes, Murphy et al. (2016) found that fibers were more readily removed at this stage than microbeads, because microbeads had smaller size and could not be entrapped by coarse and fine screens. Magnusson and Norén (2014) also suggested that plastic fibers were retained to a higher degree than MPs with other shapes. Different results are documented in other studies in which PSD process was reported to remove a considerable number of microbeads (Browne et al., 2011; Sutton et al., 2016; Talvitie et al., 2017a).

3.2. Effects of biological treatments on MP removal

The secondary treatment in WWTPs combines a biological process with a physical phase separation (clarification) (Carr et al., 2016; Tang et al., 2020). AS, trickling filters and rotating biological contactors are three commonly used technologies for the secondary treatment of wastewaters (Chen et al., 2018; Cheng et al., 2016; Fortin et al., 2019; Talvitie et al., 2017a). It was reported that the concentration of MPs could be decreased by 0.2–14% compared with the secondary influent (Mintenig et al., 2017), and sequencing batch reactor (SBR) could remove around 98% of MPs (Alvim et al., 2020).

The main removal mechanisms for MPs in the secondary treatment are that AS or bacterial extracellular polymers take advantage of dissolved oxygen to promote the growth of biological flocs, which may help accumulate the MP debris in the wastewater (Eerkes-Medrano et al., 2015; Magni et al., 2019; Murphy et al., 2016; Peng et al., 2014). When MPs reach a certain amount, they will be settled in the sedimentation tanks. In addition, due to the potential decomposition and ingestion of bacteria and protozoa, a part of MPs might be fused with those flocs (He et al., 2016; Hurley et al., 2018; Yang et al., 2018). However, the removal performance of MPs in the secondary treatment is a complex process, as it is not only related to the treatment technologies applied, but also influenced by the physiochemical characteristics of MPs (Cheung and Fok, 2017; Long et al., 2019). In the study of Long et al. (2019), parameters including particle sizes, polymer types, and shapes were considered when investigating the impacts of different treatment processes on the removal performances of MPs. They found that the removal efficiency of MPs increased as the particle sizes decreased, and two reasons might explain this phenomenon. Firstly, small-sized MPs had shorter retention time caused by the fast fragmentation and degradation rates in wastewater treatment processes; secondly, small-sized MPs were more easily to get aggregated and quickly settled into sludge so that their vertical distribution varied as particle sizes decreased (Enders et al., 2015). Moreover, it also indicated that the removal rate of MPs increased with the increase of polymer density.

Apart from biological process, the addition of chemical flocculants such as ferric sulfate during secondary treatment process is assumed to have a positive impact on the removal of MPs (Sillanpää et al., 2018). The reason is that flocculating agents can produce suspended particulate matters to aggregate other solids including MPs and then form a floc (Jarvis et al., 2005; Murphy et al., 2016). Nevertheless, the relationship between the removal of MPs and microbial or chemical flocs is still unclear. Furthermore, as shown in the research of Carr et al. (2016), some of the MPs in the wastewater might be attached to unstable flocs in the secondary treatment, making them easily be detached from the flocs and consequently leading to a redistribution in the sedimentation tank.

Taking the study of Talvitie et al. (2017a) for example, the secondary treatment of the largest Finnish WWTP, i.e., Viikinmaki WWTP, mainly takes advantage of AS method and pre-, chemical-, and biological treatment. The flocculant ferrous sulfate was used in the sand removal process prior to secondary clarifier. Hydraulic retention time (HRT) of this process was about 25 h and the sludge retention time (SRT) varied from 6 to 12 days. Typical treatment processes in Viikinmaki WWTP as well as the removal performances of MPs are demonstrated in Fig. 1. The results of this study showed that the concentration of MPs

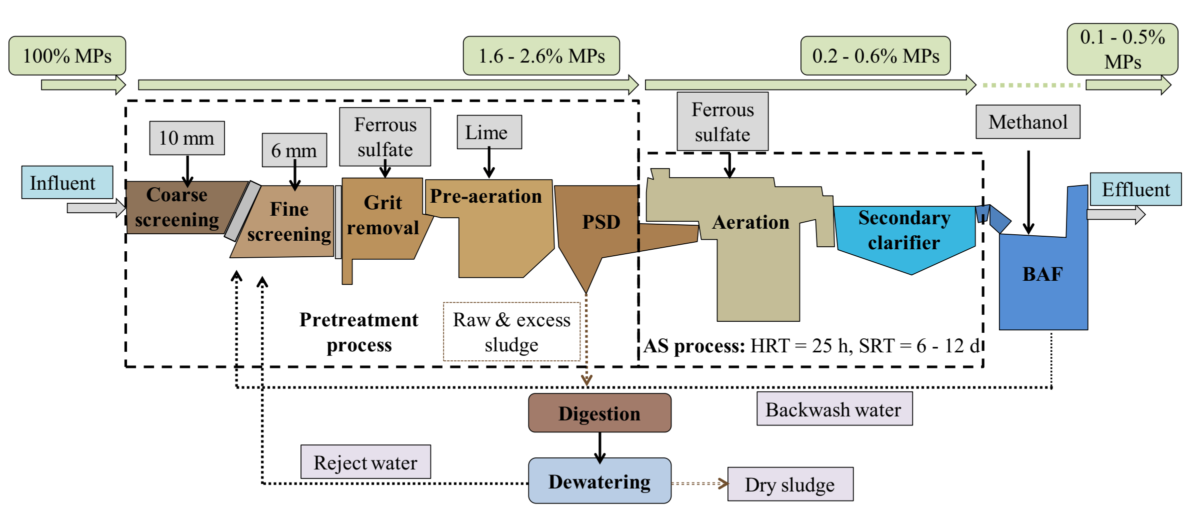

Fig. 1. Schematic illustration of typical pretreatment processes and their removal performance on MPs in Viikinmaki WWTP (PSD: primary sedimentation; AS: activated sludge; BAF: biological aerated filter) (Talvitie et al., 2017a).

was further decreased to 7–20% after AS treatment; MPs mixed with flocs and settled into the sludge during secondary sedimentation but there was still a small portion of MPs that existed in the effluent. This study also suggested that the removal efficiency of MPs might be affected by the SRT, for samples taken from effluents discharged after AS treatment with longer SRT contained lower MP concentration. Similar finding has also been reported by Li et al. (2018). They reported that the average concentrations of MPs in the group which was treated by anaerobic/aerobic (A/O) process were higher than that of oxidation ditch (OD) or sequencing batch reactor (SBR). Reason might be that the OD process usually has longer HRT (>16 h) and SRT (>15 d) than the A/O process.

Moreover, the difference in settling efficiency of sludge could also lead to the difference in removal efficiency of MPs (Estahbanati and Fahrenfeld, 2016). When MPs have a longer contact time with wastewater, chances for biofilm formation will rise, so that the surface of MPs will be coated by biofilm. As a consequence, the relative density of MPs will also be increased, which is favorable to the settling of MPs in the following sedimentation process (Sun et al., 2019). In addition, the nutrient level of the wastewater is associated with the removal efficiency of MPs, because the growth of microbial depends greatly on the nutrient contents of the wastewater (Mahon et al., 2017). However, further investigations into the interactions between these factors and the removal of MPs are needed.

Compared with the primary treatment, secondary treatment could remove more fragment particles than fibers (Carr et al., 2016). Studies showed that the removal of fibers was insignificant in the secondary treatment, and most of the fragment particles were removed during the secondary sedimentation (Murphy et al., 2016; Talvitie et al., 2017b; Mintenig et al., 2017). This may be resulted from that most fibers are easily attached to grit and other large items, and they have already been removed during the pretreatment process (Sun et al., 2019). Nonetheless, different studies display different results about the variation of particle sizes of MPs in WWTPs. For example, Talvitie et al. (2017a) revealed that concentration of MPs with particle sizes of 100–300 μm was decreased during the secondary treatment, and MPs with particle sizes of 20–100 μm accounted for ~ 80% of the total. Mintenig et al. (2017) found that MPs whose sizes were larger than 500 μm were hardly detected in the effluents after the secondary treatment. Michielssen et al. (2016) showed that after the secondary treatment, particles >300 μm were still the major component in the effluents. What leads to the differences in the distribution of particle sizes might be related to the diverse treatment methods and operational conditions adopted by different WWTPs. Moreover, the sampling methods applied by those researchers are also not uniform. In other words, standard sampling methods have not been established yet.

3.3. Effects of advanced treatments on MP removal

Advanced treatment is so called tertiary treatment, and it is an optional process for WWTPs to adopt, because most secondary effluents have met the discharge standard (Sun et al., 2019). However, it could provide a substantially additional treatment process to further improve the quality of effluent before it is discharged to the receiving environments (Gurung et al., 2017; López et al., 2017; Qiao et al., 2008). Multiple techniques can be applied in the tertiary treatment, in which denitrifying biological aerated filter (BAF), gravity sand filtration (GSF), discfilter (DF), dissolved air floatation (DAF), membrane bioreactor (MBR), and advanced oxidation are involved (Yang et al., 2010; Zhang et al., 2014; Talvitie et al., 2017a).

Positive effects of the tertiary treatment on the removal of MPs have been reported (Talvitie et al., 2017a). Table 1 presents the effects of different wastewater treatment processes on the removal performance of MPs. It can be seen that the MP concentration in the effluent after different advanced treatments is further decreased. Given that the primary and secondary treatment have already removed the majority of MPs from wastewaters, the removal performance of the tertiary treatment on MPs is not that remarkable (Browne et al., 2011). It is reported that the concentrations of MPs in the tertiary effluents are only further decreased by 0.2–2%, compared with the tertiary influent (Mahon et al., 2017). In addition, the volumes of samples taken from the effluent and detecting methods used to determine the concentrations of MPs would affect the measurement. Thus, the necessity of adopting a tertiary treatment to further remove MPs from wastewaters depends on the quality requirement of the effluent, but more studies are still needed to understand the effects of tertiary treatment methods on the removal of MPs.

Different advanced treatment techniques show different removal efficiency of MPs. The removal efficiencies of MPs of aforementioned advanced treatment techniques have been compared in a couple of studies, and tertiary treatment that employs membrane-related techniques is proved to have better removal performance. Talvitie et al. (2017a) have investigated the removal efficiency of MPs by different advanced treatment technologies in different WWTPs in Finland. Four WWTPs were studied: the Viikinmaki WWTP which is equipped with a DF and a BAF for the tertiary treatment; the Kakolanmaki WWTP that employs GSF as full-scale tertiary treatment; the Paroinen WWTP that uses DAF technology; and the Kenkaveronniemi WWTP that sets MBR pilot unit. In these WWTPs, all advanced final treatment stage technologies could remove more than 95% of MPs with particle size >20 μm, and the highest removal efficiency was achieved by the MBR, followed by GSF and DAF. Furthermore, Michielssen et al. (2016) reported that the Northfield WWTP with a novel pilot-scale AnMBR system had the highest removal efficiency for MPs. They suggested that the MP concentrations in the final effluent of those investigated WWTPs that adopted MBR were much lower than WWTPs using advanced filtration technologies. Nonetheless, opposite results have been reported by other researchers. For instance, Leslie et al. (2017) stated that MBR systems were not better at retaining MPs in effluent than conventional WWTP systems. The research conducted by Lee and Kim (2018) suggested that MBR had a relatively low removal efficiency of 44.7% on MPs. Moreover, inefficient removal performance of other tertiary treatment techniques on MPs has also been revealed. For example, Carr et al. (2016) found that WWTP using GSF as a tertiary treatment showed low removal efficiency of MPs. Mintenig et al. (2017) showed that BAF method did not have significant impact on the removal of MPs, which was in accordance with the finding of Talvitie et al. (2017a).

As for the particle sizes of MPs, it is reported that the final effluents are dominated by MPs with particle sizes <20 μm (Carr et al., 2016). This is also related to the technologies that the tertiary treatment applies, because various treatment techniques exerted different influences on the removal of MPs with different sizes. For example, in the effluent of the tertiary treatment with a GSF technology, MPs > 45 μm are not detected (Carr et al., 2016). The tertiary advanced filtration is reported to completely remove MPs > 500 μm and the concentrations of MPs (20–500 μm) have decreased from 0.2 to 0.01 items/L (with a removal efficiency of 95%) (Mintenig et al., 2017). WWTPs using MBR as the tertiary treatment method could remove approximately 90% of MPs whose particle size >300 μm, and MPs with the particle sizes of 20–100 μm made up the majority of the MPs in the effluent (Talvitie et al., 2017a). With regards to the shapes and polymer types of MPs in the effluent after a tertiary treatment, fibers and PE are dominant in the final effluent in most studies (Dris et al., 2015; Lee and Kim, 2018; Talvitie et al., 2017a; Ziajahromi et al., 2017). Although most of the fibers have been removed by the primary and secondary treatments, due to the property that fibers could escape from filters or membranes more easily, their relative abundance would increase in the final effluent (Shen et al., 2013).

Many other techniques are also utilized in advanced treatment processes. For instance, coagulation-flocculation is one of the main methods for solid-liquid separation, and it is usually used as a primary

Table 1
The presence of MPs in different WWTPs.

Location	Facility capacity (m^3/d)	Average MP concentration in influent	Average MP concentration in effluent	Average MP concentration in dry sludge	Sampling size (μm)	Dominant size (μm)	Dominant polymer types	Dominant shape	Treatment processes	Removal efficiency	References
France	240,000	2.6–3.2 × 10^5 particles/m^3	1.4–5 × 10^4 particles/m^3	N/A	>100	100–5000	PE	Fiber	Preliminary Primary Secondary	63–81% 58–72% 83–95%	Dris et al. (2015)
UK	260,954	15.7 ± 5.2 MPs/L	0.25 ± 0.04 MPs/L	N/A	>65	598 ± 89	PE	Flake	Preliminary Primary Secondary	44.59% 78.34% 98.41%	Murphy et al. (2016)
Korea	110,000	13.5 MPs/L	0.09 MPs/L	7.34 MPs/g	>106	106–300	N/A	Fiber	A/A/O SBR Media	49.3% 44.7% 49.0%	Lee and Kim (2018)
USA	2,500,000	133.0 ± 35.6 MPs/L	5.9 MPs/L	N/A	>100	100–1000	PE	Microbeads	Preliminary Primary Secondary	58.6% 84.1% 93.8%	Michielssen et al. (2016)
Finland	N/A	0.7–6.9 MPs/L	0.005–0.3 MPs/L	N/A	20–300	20–100	PES	Fiber	DF RSF DAF MBR	98.5% 97% 95% 99.9%	Talvitie et al. (2017a)
Australia	13,000	N/A	0.28 MPs/L	N/A	25–500	100–190	PE	Fiber	Primary Secondary Tertiary	N/A N/AA >90%	Ziajahromi et al. (2017)
China	120,000	5.6 ± 0.09 mg/L	0.168 ± 0.02/ 0.028 ± 0.01 mg/L	N/A	25–500	>500	PET	Fragment	OD MBR	97% 99.5%	Lv et al. (2019)
South Korea	26,545	4200 MPs/L	33 MPs/L	710 MPs/L	N/A	400.83	N/A	Fragment	Primary Secondary Coagulation Ozone	62.7% 54.7% 53.8% 89.9%	Hidayaturrahman and Lee (2019)
Italy	18,000	3.6 MPs/L	0.52 MPs/L	5.3 MPs/gTS	N/A	100–500	PE	Fiber	UASB AnMBR	52.6% 41.4%	Pittura et al. (2021)
Australia	130,000	92.0 particles/L	0.18 particles/L	56.5 particles/g	25–500	>25	PET	Fiber	Primary Secondary	97.6% >98%	Ziajahromi et al. (2021)
Iran	22,000	12,667 MPs/m^3	423 MPs/m^3	3,514 MPs/m^3	37–500	37–300	PE	Fiber	Primary	96.7%	Petroody et al. (2020)
UK	111,496–184,703	3–10 MPs/L	<1–3 MPs/L	N/A	2800	20–190	PP	Fiber	Preliminary Primary Secondary Tertiary	6% 60–76% 92% 96%	Blair et al. (2019)
Spain	28,400	171 ± 42 particles/L	10.7 ± 5.2 particles/L	133 ± 59 particles/g	25–375	25–104	PE	Fragment	Secondary	93.7%	Edo et al. (2020)
Thailand	200,000	12.2 pieces/L	2.0 pieces/L	103.4 pieces/L	330–475	N/A	PEs	Fiber	Primary Aeration	0% 83.6%	Hongprasith et al. (2020)

N/A: not available; TS: total solids; PE: polyethylene; PES: polyester; PET: polyethylene terephthalate; PP: polypropylene; UASB: upflow granular anaerobic sludge blanket; AnSBR: anaerobic membrane bioreactor.

or secondary treatment in wastewater disposal, and widely-used coagulants are iron and aluminum salts (Ayekoe et al., 2017; Verma et al., 2012). In addition, ultrafiltration (UF) is also a frequently applied method when separating solid from liquid. Both coagulation-flocculation and UF technology are reported to have been applied in the removal of MPs from waters, but both of them have relatively low removal efficiency when used separately (Ziajahromi et al., 2017). Therefore, method that combines the two technologies has been employed to remove MPs. For example, Ma et al. (2019b) used aluminum-based salts as the first step to remove PE and then used UF membrane to further remove MPs. Their results showed that coagulation together with UF technology had essentially 100% removal efficiency of MPs and could have potential application in drinking water treatment. Fe-based salts are also utilized as coagulant to remove MPs from waters, but low removal efficiency (<15%) was observed (Ma et al., 2019a). Whereas, secondary pollution caused by the coagulants and flocculants and membrane fouling are inevitable shortcomings of these technologies (Chen et al., 2007; Zahrim et al., 2017).

Furthermore, ozonation combined with granular activated carbon (GAC) filtration have been applied as advanced water treatment technologies and they are especially utilized to eliminate emerging contaminants (Gang et al., 2018; Nasuhoglu et al., 2018; Sbardella et al., 2018). Wang et al. (2020b) investigated the effects of ozonation integrated with GAC filtration on the removal performances of MPs and found that the MP concentration in the effluent of ozonation treatment was slightly increased, but 56.8–60.9% of the MPs were removed after the GAC filtration process. They also showed that MPs might be broken into smaller size during the ozonation process, which would benefit the following GAC filtration, as this process was efficient in removing small-sized particles. It has also been reported that PE, PP, and PAM were among the top three polymer types removed by GAC filtration.

In addition to conventional wastewater treatment methods, novel synthetic materials that can be used to remove MPs have been developed in recent years. For instance, innovative inorganic-organic hybrid silica gels which can remove hydrophobic MPs like PE, PP, and PET from wastewater have been synthesized by Herbort and Schuhen (2017), and the new materials are supposed to be cost-effective and environmentally-friendly. Moreover, synthetic amorphous silica is used as a carrier to combine with bio-inspired materials or catalysts, and it showed that this material has the potential to efficiently remove MPs from wastewaters (Herbort et al., 2018). Nevertheless, the efficiency of these novel materials in removing MPs have not been specifically reported so far. Future research should put more focus on this aspect.

To conclude, the removal efficiencies of different treatment processes in WWTPs on MPs are different. Generally, the preliminary and primary treatment has relatively low removal rate on MPs, the secondary treatment with a SBR can remove over 98% of MPs from the wastewater, and the performance of the tertiary treatment on removing MPs is limited. Besides, MPs-targeted treatment is needed so as to reduce the numbers of MPs discharging into the environments, and novel materials should be synthesized to more efficiently remove MPs from wastewaters.

4. Effects of MPs on wastewater treatment processes

4.1. Effects of MPs on three wastewater treatment stages

Table 2 summarizes some possible impacts of MPs on some typical wastewater treatment processes. It can be seen that different influences of MPs may be exerted on different treatment processes. For the preliminary treatment, the major negative effect of MPs is blockage. Although small-sized MPs would not block coarse grilles whose grid distance generally ranges from 16 – 25 mm, they might cause blockage to fine grilles with grid distance of 3–10 mm, due to the large volumes of wastewaters (Zhang and Chen, 2020). As MPs can suspend in the wastewaters, more flocculant (i.e. ferrous sulfate) addition are required to precipitate suspended solids during the primary treatment (Zhang and Chen, 2020). Moreover, MPs are hydrophobic and have high adsorptive capacity to other pollutants such as heavy metals and polychlorinated biphenyls since these pollutants are more likely accumulated in WWTPs (Bakir et al., 2014). Consequently, the removal of toxic contaminants by the primary treatment processes might also be affected by the presence of MPs.

MPs could have various effects on the biological treatment in WWTPs. It was reported that MPs had high retention potential during the biological treatment processes (Liu et al., 2019a). Therefore, biological conversion rates in the secondary treatment processes could be related to the existence of MPs. Recent studies have shown that the presence of MPs might affect nitrogen removal efficiency, concentration

Table 2
Possible effects of MPs on different wastewater treatment processes.

Treatment processes	Affected objects	Polymer types	MP concentrations	MP sizes (μm)	Possible effects	Experiment condition	References
Preliminary treatment	Fine grilling	N/A	N/A	N/A	Cause blockage	N/A	Zhang and Chen (2020)
Primary treatment	Coagulation/flocculation	PS	0.1–6.7 mg/L	1.0–6.3	Increase reagents addition dosage	Laboratory	Rajala et al. (2020)
Secondary treatment	SBR	PE	13.9 mg/m^3	<100	No obvious effects	Laboratory	Kalčíková et al. (2017)
Secondary treatment	BAF	PE	2.5 particles/L	100–300	Cause uneven distribution of water	Field	Talvitie et al. (2017b)
Advanced treatment	Coagulation	PE	N/A	N/A	Increase Fe-based coagulant addition	Laboratory	Ma et al. (2019a)
Advanced treatment	DAF	PES	2.0 MPs/L	20–100	Reduce floatation ability of bubbles	Field	Talvitie et al. (2017b)
Advanced treatment	UF	PE	628–3,605 MPs/L	1–100	Result in pore blocking and membrane fouling	laboratory	Enfrin et al. (2020)
Advanced treatment	Chlorine disinfection	HDPE, PP	N/A	<5,000	Form new chlorine-carbon bonds and increase ecotoxicology	Laboratory	Kelkar et al. (2019)
Advanced treatment	UV disinfection	PE, PP	11.80 ± 1.10 MPs/L	<5,000	Decrease disinfection efficiency	Field	Raju et al. (2020)
Advanced treatment	Ozonation	PET	3,760 ± 726 MPs/L	1–5	Reduce removal efficiency of ozonation	Field	Wang et al. (2020b)

N/A: not available; SBR: sequence batch reactor; BAF: biological aerated filter; DAF: dissolved air floatation; UF: ultrafiltration; PS: polystyrene; PE: polyethylene; PES: polyester; HDPE: high density polyethylene; PET: polyethylene terephthalate.

of biochemical oxygen demand, dissolved oxygen, total nitrogen, and total phosphorus (Caruso et al., 2018; Kataoka et al., 2019; Liu et al., 2017; Xiao et al., 2015). The study of Cluzard et al. (2015) suggested that MPs, especially microbeads, had the potential to disturb the cycling of ammonium in water by affecting the bioconversion of inorganic nitrogen. Specifically, nitrogen conversion bacteria have been proved to be affected by MPs. For example, Sun et al. (2018) showed that the conversion efficiencies of NH_4^+-N of *Halomonas alkaliphila* were increased by the treatment of MPs, while the conversion efficiencies of NO_3^--N and NO_2^--N were not significantly impacted. The denitrification process in biological wastewater treatment might be inhibited by MPs which could change the microbial-mediated processes, so that ammonium would be accumulated rather than be removed (Sun et al., 2018). No clear evidences have shown that MPs had influences on the removal of phosphorus, and it is speculated that phosphorous accumulation related microorganisms are less sensitive to the presence of MPs than nitrogen-conversion related bacteria (Chen et al., 2012). MPs could easily adsorb suspended solids from wastewaters and form spheres, which could cause uneven distribution of suspended solids in wastewater (Zhang and Chen, 2020). Nevertheless, Kalčíková et al. (2017) demonstrated that MPs had no effect on SBR.

The impacts of MPs on advanced treatment processes in WWTPs are manifold as well. For instance, the intricate interaction between the negatively surface-charged MPs and flocculants/coagulants could reduce the efficacy of flocculation/coagulation (Zhang and Chen, 2020), which requires increasing amount of reagent addition. In addition to this, MPs may also affect different treatment stages including air floatation, membrane filtration, and disinfection. MPs could adsorb various substances from wastewaters and form agglomerates and thus reduce the floatation ability of bubbles that are designed for contaminants removal in the air floatation process (Raju et al., 2020; Zhang and Chen, 2020). Moreover, the existence of MPs in wastewaters increases the number of suspended solids, leading to higher energy consumption for generating air bubbles (Bilgin et al., 2020). Filtration technologies such as UF, microfiltration, and reverse osmosis are frequently used in advanced wastewater treatment. The presence of MPs is detrimental for the filtration performance of membranes. The research of Lai et al. (2014) evidenced that current membranes, especially reverse osmosis membrane usually had short lifespan and suffered irreversible wear by abrasive particles including MPs. In other words, MPs with irregular shapes would threaten the filtration performance of membranes. The work of Enfrin et al. (2020) reported that MPs might also result in membrane fouling and blockage because of the interaction between MPs and membranes by absorbing various substances onto the surface of membranes. As for the disinfection process, MPs are reported to impede chloride and ultraviolet disinfection by providing a protective shield for bacteria (Kelkar et al., 2019). Additionally, MPs might reduce the efficiency of ozonation process due to two reasons. Firstly, ozone has strong affinity for many organic contaminants adsorbed onto the surface of MPs and can generate corresponding derivatives (Ahmed et al., 2017; Benner et al., 2013; Von Sonntag and von Gunten, 2012). Secondly, ozone as a strong oxidant can also oxidize MPs (Wang et al., 2020b; Zhang and Chen, 2020).

4.2. Effects of MPs on AS

AS is a commonly utilized process treating municipal and industrial wastewater, and it is vital to the whole treatment processes in WWTPs. Therefore, possible effects of MPs on AS are specifically discussed in this secssion. It has been reported that more than 90% of MPs in WWTPs are eventually remained in sludge (Li et al., 2020a), and the retention rate of MPs could reach 99% (Gies et al., 2018). It was estimated that the concentrations of MPs in waste sludge ranged from 1.5×10^3 to 2.4×10^4 MPs/kg (Mahon et al., 2017; Mintenig et al., 2017). To the best of our knowledge, studies on the impacts of MPs on sludge are inadequate until now. We have summarized some major effects from available literature as shown in Table 3.

These effects can be grouped into three aspects. First of all, MPs are proved to affect the nitrification and denitrification of AS. Ammonia-oxidizing bacteria (AOB) and nitrite-oxidizing bacteria (NOB) are two major bacteria groups that play the leading role in nitrification and denitrification (Capodaglio et al., 2016; Kampschreur et al., 2009). Different MPs exert different degrees of influence on nitrification and denitrification processes by either promoting or inhibiting the activity of the two bacteria groups (An et al., 2018; Xu et al., 2013). For instance, Li et al. (2020b) utilized five types of MPs to investigate their impacts on the nitrification and denitrification of AS. They found that MPs showed adverse effects on ammonia oxidation efficiency but had no obvious influence on nitrite oxidation efficiency. When the abundance of MPs was

Table 3
Effects of MPs on AS in WWTPs.

MP types	MP concentration	Exposure time	Major effects[a] (compared with the blank control)	Possible mechanisms	Experiment condition	References
PVC, PES	5,000 particles/L	3 h	PVC and PES increased the specific denitrification rate by around 23.2% and 25%, respectively; PVC and PES inhibited AS nitrification by around 11.0% and 12.3%, respectively	Influencing the activity of AOB and NOB	Laboratory	Li et al. (2020b)
Polyether sulfone	0.5 g/L	30 d	Inhibited removal efficiency of ammonia nitrogen by 70.7%	Inhibiting the performance of nitrite oxidase in the AGS	Laboratory	Qin et al. (2020)
PET	10–60 particles/g-TS	21 d	Reduced hydrogen production by 11.6–29.3%	Leaching the toxic di-n-butyl phthalate and causing shift of the microbial community toward the direction against hydrolysis-acidification	Laboratory	Wei et al. (2019c)
PE	100–200 particles/g-TS	44 d	Significantly decreased methane production by 12.4–27.5%	Inducing reactive oxygen species to reduce cell viability	Laboratory	Wei et al. (2019a)
PS	100 μg/mL	210 h	Decreased methane production by 17.5% in the first cycle but recovered in the second cycle	Changing the protein secondary structure of extracellular polymeric substances	Laboratory	Feng et al. (2018b)
PA66	0.1–0.5 g/L	30 d	Caused damage to sludge structure; slightly promoted the performance of AGS: the removal efficiencies of COD and NH_4^+-N were increased by 0.44–1.13% and 0.33–1.36%, respectively	Inhibiting microbial cell growth; promoting the metabolism of microorganisms	Laboratory	Zhao et al. (2020)

AS, activated sludge; PVC: polyvinylchloride; PES: polyester; PET: polyethylene terephthalate; PE: polyethylene; PS: polystyrene; PA66: polyamide 66; TS: total solids; AOB, ammonia-oxidizing bacteria; NOB, nitrite-oxidizing bacteria; AGS: aerobic granular sludge; COD: chemical oxygen demand; NH_4^+-N: ammonia nitrogen.
[a] All the data were converted into percentage from the original data in the references.

as high as 1,000–10,000 particles/L, the nitrification rate was slightly decreased while the denitrification rate was increased. This might due to the fact that MPs could inhibit the activity of NO_2^--oxidizing in the nitrification process. Similarly, Song et al. (2020) studied the effect of MPs on the partial nitrification process with PVC concentrations at 0, 1000, 5000, and 10,000 particles/L, and inhibition effects were seen in this process. They also concluded that PVC could suppress AOB and NOB activities. Besides, an acceleration of denitrification rate was observed when 5,000 particles/L of PVC and polyester (PES) was added, and high concentration of PVC also led to notable increasing of nitrous oxide emission during denitrification rather than nitrification. This process was closely related to AOB, demonstrating that high-concentration PVC could promote the activity of them (Massara et al., 2017). Meanwhile, Chen et al. (2020) showed that MP biofilms could affect nitrogen cycle by accelerating ammonia and nitrite oxidation as well as denitrification, and they suggested that the underlying mechanism might be that MPs could provid additional substrates for the attachment of microbes. However, the research of Wei et al. (2019b) indicated that high concentration of PVC could release some toxic chemical substances to undermine denitrification process by inhibiting the activity of AOB. Besides, Qin et al. (2020) proved that polyether sulfone could slightly inhibit the removal efficiency of ammonia nitrogen, and the total removal rate of nitrogen was 5.6% higher than the blank control. However, further investigations should be conducted to reveal deeper mechanisms behind these effects.

Secondly, sludge digestion could also be impacted by the presence of MPs. AD is a frequently adopted method to stabilize sludge in WWTPs, and four processes are involved in AD, namely hydrolysis, acidogenesis, acetogenesis, and methanogenesis (Wei et al., 2019b). It was reported by Zhao et al. (2010) that MPs could negatively inhibit the hydrolysis of proteins and polysaccharides, and consequently, the content of acidified matters decreased, which would result in less gas production. The study of Wei et al. (2019c) provided evidence that the hydrogen production of AS under alkaline anaerobic condition was reduced by the presence of PET. Possible mechanism was that PET could leach a toxic chemical, namely di-n-butyl phthalate, which would increse the number of reactive oxygen species and further lead to more microbial death. It has also been shown by Fu et al. (2018) that hydrogen yield of AD was rapidly increased in the first 6 days when exposing to nanoplastics; however, the final hydrogen production was slightly decreased. Their further analysis revealed that many nanoplastics attached on the cell membrane of *Acetobacteroides hydrogenigenes*, and formed considerable number of nano-size pores. As a result, permeability of cell membrane as well as the redox cycling in the cytoso could be changed (Liang et al., 2010), affecting the activity of anaerobic bacteria. The stage of methanogenesis is affected by MPs most among the four processes (Wei et al., 2019a). Li et al. (2020a) showed that methane production would not be much reduced when adding different types of MPs, and the methane production rate was 88.53%–95.08% in all studied polymer types compared with the control group. Besides, MP concentrations might exert different influence on methane production as reported by Wei et al. (2019b). They have shown that 10 particles/g of TS (total solids) of PVC MPs could enhance anaerobic methane production by $5.9 \pm 0.1\%$, while higher levels of MPs (≥20 particles/g TS) would inhibit the methane production. The reason might be that bisphenol A which was leached from PVC MPs would affect sludge stabilization and further influence the methane production. Furthermore, Feng et al. (2018a) elucidated that cationic PS nanoparticles showed higher inhibition capacity on methane production than anionic PS nanoparticles.

Lastly, some MP types have been reported to have an effect on the microbial communities in AS of WWTPs. Microbial community structures could be affected by the presence of MPs, and different microorganisms showed different responses (Fu et al., 2018). The experiments of Zhu et al. (2018) revealed that the bacterial diversity was greatly enhanced by the exposure to MPs, which changed the microbiota of the collembolan gut. MPs polyamide 66 (PA66) showed adverse effects on the microbial structure in aerobic granular sludge (AGS) as reported by Zhao et al. (2020). In their study, both promotion and inhibition on microbial growth were observed as 0.2 g/L and 0.5 g/L PA66 would increase microbial diversity while 0.1 g/L PA66 would negatively affect bacterial diversity. More specifically, because *Actinobacteria* was positively related to the increase of sludge flocculation by maintaining the sludge structure (Liu et al., 2019b), the fact that PA66 could slightly promote the flocculation of AGS indicated that PA66 might favor the growth of *Actinobacteria* (Zhao et al., 2020). The major way that MPs might influence microorganisms is by affecting some enzymes and metabolic intermediates of them (Zhang and Chen, 2020). For instance, Wei et al. (2019b) analyzed several key enzymes in AD of AS and found that protease and acetate kinase presented dosage-dependent relationship with the concentration of PVC (20–60 particles/g-TS), and 60 particles/g of PVC significantly reduced the activity of coenzyme F_{420} to $79.3 \pm 0.02\%$. However, MPs, their surface-adsorbed pollutants, and chemical additives may be capable of affecting microorganisms, making the mechanism analysis a difficult issue to tackle (Anbumani and Kakkar, 2018).

Studies on how MPs affect AS in WWTPs are still scanty. MPs properties including sizes, shapes, and polymer types may play different roles in changing the structures and functions of sludge; however, present studies have shown limited information about this. As a result, more evidence should be collected to support this statement. Worse still, once the toxic materials are desorbed from MPs, their impacts on AS will be difficult to estimate. Therefore, more research should be conducted in the future to further reveal the effects of MPs on AS.

In summary, potential effects of MPs on wastewater treatment processes have been gradually reported in recent years, despite the lack of mechanism revealing. Current investigations did not show how the characteristics of MPs including size and polymer type affected the performance of different wastewater treatment technologies and AS. Moreover, the interaction between surface-adsorbed pollutants on MPs and wastewater treatment processes remains unclear. Once these contaminants are desorbed, they might induce much severer inhibition effects on microorganisms in biofilms and AS than MPs alone. Therefore, studies focusing on these aspects are still needed in the future work. Noting that many studies on the removal performance of MPs in WWTPs are conducted in lab-scale, the results reported by these studies are still environmentally relevant due to two facts: (1) The wastewaters used in these experiments were taken from WWTPs, and the treatment processes were based on actual conditions of WWTPs. (2) The compositions, sizes, shapes, and concentrations of MPs used in the experiments were all typical in real wastewaters. Nevertheless, it is difficult to completely simulate the real environment of WWTPs in the laboratory. Thus, more parameters such as temperature, pH, and dissolved oxygen concentration should be taken into consideration so that the results obtained by the experiments will be more reliable. All in all, disclosing the underlying mechanisms of how MPs affect wastewater treatment processes will help improve removal performance of WWTPs and alleviate MP pollution in the environment.

5. Conclusions

This paper has thoroughly reviewed the fate of MPs in in WWTPs and discussed the potential effects of MPs on different wastewater treatment processes. Main conclusions are listed as follows:

(1) The preliminary and primary treatment in WWTPs showed relatively low MP removal efficiency, but the secondary treatment (represented by SBR) can reduce about 98% of the MPs content. MPs with size larger than 100 μm are most abundant in the final effluents, and PE, PS, and PES are three dominant polymer types in most WWTPs. Besides, fiber, microbead, and fragment are three dominant shapes in wastewaters.

(2) It is reported that MPs could exert various influences on different wastewater treatment processes. Large numbers of MPs could cause blockage to fine grilles. More reagents should be added in flocculation and coagulation process because of the high adsorptive capacity of MPs. Furthermore, MPs also show negative effects on denitrification process, filtration, and disinfection.
(3) MPs may affect nitrification and denitrification of AS by disturbing AOB and NOB groups. In the AD, methanogenesis process is most affected by the presence of MPs. Both increased and decreased methane production has been observed by researchers. Besides, microbial diversity and growth rate as well as the activity of several key enzymes may be impacted by MPs.

There are still some knowledge gaps needing to be filled in the near future as the studies on MPs and wastewater treatment processes are relatively limited.

(1) At present, the removal of MPs in WWTPs relies on the adopted treatment processes and none of them are designed with the initial purpose to remove MPs. The increasing production of synthetic plastics calls for novel technologies to efficiently and purposefully remove MPs from wastewaters.
(2) Studies on the effects of MPs on wastewater treatment processes are inadequate. Mechanisms revealing the impacts of MPs on different wastewater treatment processes including the structure and function of AS are still unclearly elucidated. Additionally, MPs can easily adsorb other pollutants (i.e. heavy metals and organic pollutants) onto their surface, and they also contain many chemical additives. Once these toxics are leached/desorbed from MPs, they may pose severe risks to microorganisms in biofilms and AS. Therefore, future research should attach much importance to this aspect so as to avoid the detrimental effects of MPs on the performance of wastewater treatment processes.
(3) As the majority of current investigations into the potential effects of MPs on wastewater treatment processes are based on laboratory study, it is of significance to consider the actual environment in WWTPs when designing experiments. Factors including temperature, pH, and dissolved oxygen concentration of wastewaters should be taken into account when researching the effects of MPs on wastewater treatment processes.

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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References

Ahmed, M.B., Zhou, J.L., Ngo, H.H., Guo, W., Thomaidis, N.S., Xu, J., 2017. Progress in the biological and chemical treatment technologies for emerging contaminant removal from wastewater: a critical review. J. Hazard. Mater. 323, 274–298.

Akarsu, C., Kumbur, H., Gökdağ, K., Kıdeyş, A.E., Sanchez-Vidal, A., 2020. Microplastics composition and load from three wastewater treatment plants discharging into Mersin Bay, north eastern Mediterranean Sea. Mar. Pollut. Bull. 150, 110776–110788.

Alvim, C.B., Castelluccio, S., Ferrer-Polonio, E., Bes-Piá, M.A., Mendoza-Roca, J.A., Fernández-Navarro, J., Alonso, J.L., Amorós, I., 2020. Effect of polyethylene microplastics on activated sludge process - Accumulation in the sludge and influence on the process and on biomass characteristics. Process Saf. Environ. Prot. 148, 536–547.

An, H., Liu, J., Li, X., Yang, Q., Wang, D., Xie, T., Yi, K., 2018. The fate of cyanuric acid in biological wastewater treatment system and its impact on biological nutrient removal. J. Environ. Manag. 206, 901–909.

Anbumani, S., Kakkar, P., 2018. Ecotoxicological effects of microplastics on biota: a review. Environ. Sci. Pollut. Res. 25, 14373–14396.

Ayekoe, C.Y.P., Robert, D., Gone, L.D., 2017. Combination of coagulation-flocculation and heterogeneous photocatalysis for improving the removal of humic substances in real treated water from Agbô river (Ivory-coast). Catal. Today 281, 2–13.

Bakir, A., Rowland, S.J., Thompson, R.C., 2014. Transport of persistent organic pollutants by microplastics in estuarine conditions. Estuar. Coast. Shelf Sci. 140, 14–21.

Bayo, J., Olmos, S., López-Castellanoss, J., 2020. Microplastics in an urban wastewater treatment plant: the influence of physicochemical parameters and environmental factors. Chemosphere 238, 124593.

Ben-David, E.A., Habibi, M., Haddad, E., Hasanin, M., Angel, D.L., Booth, A.M., Sabbah, I., 2021. Microplastic distributions in a domestic wastewater treatment plant: removal efficiency, seasonal variation and influence of sampling technique. Sci. Total Environ. 752, 141880.

Benner, J., Helbling, D.E., Kohler, H.P.E., Wittebol, J., Kaiser, E., Prasse, C., Ternes, T.A., Albers, C.N., Aamand, J., Horemans, B., Springael, D., Walravens, E., Boon, N., 2013. Is biological treatment a viable alternative for micropollutant removal in drinking water treatment processes? Water Res. 47, 5955–5976.

Bilgin, M., Yurtsever, M., Karadagli, F., 2020. Microplastic removal by aerated grit chambers versus settling tanks of a municipal wastewater treatment plant. J. Water Process Eng. 38, 101604.

Blair, R.M., Waldron, S., Gauchotte-Lindsay, C., 2019. Average daily flow of microplastics through a tertiary wastewater treatment plant over a ten-month period. Water Res. 163, 114909.

Brennecke, D., Duarte, B., Paiva, F., Caçador, I., Canning-Clode, J., 2016. Microplastics as vector for heavy metal contamination from the marine environment. Estuar. Coast. Shelf Sci. 178, 189–195.

Browne, M.A., Galloway, T., Thompson, R., 2007. Microplastic - an emerging contaminant of potential concern? Integr. Environ. Assess. Manag. 3, 559–561.

Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., Thompson, R., 2011. Accumulation of microplastic on shorelines woldwide: sources and sinks. Environ. Sci. Technol. 45, 9175–9179.

Capodaglio, A.G., Hlavínek, P., Raboni, M., 2016. Advances in wastewater nitrogen removal by biological processes: state of the art review. Revista Ambiente & Agua 11, 250–267.

Carr, S.A., Liu, J., Tesoro, A.G., 2016. Transport and fate of microplastic particles in wastewater treatment plants. Water Res. 91, 174–182.

Caruso, G., Pedà, C., Cappello, S., Leonardi, M., La Ferla, R., Lo Giudice, A., Maricchiolo, G., Rizzo, C., Maimone, G., Rappazzo, A.C., Genovese, L., Romeo, T., 2018. Effects of microplastics on trophic parameters, abundance and metabolic activities of seawater and fish gut bacteria in mesocosm conditions. Environ. Sci. Pollut. Res. 25, 30067–30083.

Chen, Y., Dong, B.Z., Gao, N.Y., Fan, J.C., 2007. Effect of coagulation pretreatment on fouling of an ultrafiltration membrane. Desalination 204, 181–188.

Chen, Y.G., Su, Y.L., Zheng, X., Chen, H., Yang, H., 2012. Alumina nanoparticles-induced effects on wastewater nitrogen and phosphorus removal after short-term and long-term exposure. Water Res. 46, 4379–4386.

Chen, Y.J., He, H.J., Liu, H.Y., Li, H.R., Zeng, G.M., Xia, X., Yang, C.P., 2018. Effect of salinity on removal performance and activated sludge characteristics in sequencing batch reactors. Bioresour. Technol. 249, 890–899.

Chen, X.C., Chen, X.F., Zhao, Y.H., Zhou, H.E., Xiong, X., Wu, C.X., 2020. Effects of microplastic biofilms on nutrient cycling in simulated freshwater systems. Sci. Total Environ. 719, 137276.

Cheng, Y., He, H.J., Yang, C.P., Zeng, G.M., Li, X., Chen, H., Yu, G.L., 2016. Challenges and solutions for biofiltration of hydrophobic volatile organic compounds. Biotechnol. Adv. 34, 1091–1102.

Cheung, P.K., Fok, L., 2017. Characterisation of plastic microbeads in facial scrubs and their estimated emissions in Mainland China. Water Res. 122, 53–61.

Cho, Y., Shim, W.J., Jang, M., Han, G.M., Hong, S.H., 2019. Abundance and characteristics of microplastics in market bivalves from South Korea. Environ. Pollut. 245, 1107–1116.

Chua, E.M., Shimeta, J., Nugegoda, D., Morrison, P.D., Clarke, B.O., 2014. Assimilation of polybrominated diphenyl ethers from microplastics by the marine amphipod, *Allorchestes Compressa*. Environ. Sci. Technol. 48, 8127–8134.

Claessens, M., Meester, S.D., Landuyt, L.V., Clerck, K.D., Janssen, C.R., 2011. Occurrence and distribution of microplastics in marine sediments along the Belgian coast. Mar. Pollut. Bull. 62, 2199–2204.

Cluzard, M., Kazmiruk, T.N., Kazmiruk, V.D., Bendell, L.I., 2015. Intertidal concentrations of microplastics and their influence on ammonium cycling as related to the shellfish industry. Arch. Environ. Contam. Toxicol. 69, 310–319.

Conley, K., Clum, A., Deepe, J., Lane, H., Beckingham, B., 2019. Wastewater treatment plants as a source of microplastics to an urban estuary: removal efficiencies and loading per capita over one year. Water Res. X 10030. https://doi.org/10.1016/j.wroa.2019.100030.

Corradini, F., Meza, P., Eguiluz, R., Casado, F., Huerta-Lwanga, E., Geissen, V., 2019. Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. Sci. Total Environ. 671, 411–420.

Dris, R., Gasperi, J., Rocher, V., Mohamed, S., Tassin, B., 2015. Microplastic contamination in an urban area: a case in Greater Paris. Environ. Chem. 12, 592–599.

Edo, C., Gonzalez-Pleiter, M., Leganes, F., Fernandez-Pinas, F., Rosal, R., 2020. Fate of microplastics in wastewater treatment plants and their environmental dispersion with effluent and sludge. Environ. Pollut. 259, 113837–113845.

Eerkes-Medrano, D., Thompson, R.C., Aldridge, D.C., 2015. Microplastics in freshwater systems: a review of the emerging threats, identification of knowledge gaps and prioritisation of research needs. Water Res. 75, 63–82.

Emadian, M.S., Onay, T.T., Demirel, B., 2017. Biodegradation of bioplastics in natural environments. Waste Manag. 59, 526–536.

Enders, K., Lenz, R.C., Stedmon, A., Nielsen, T.G., 2015. Abundance, size and polymer composition of marine microplastics ≥10 μm in the Atlantic Ocean and their modelled vertical distribution. Mar. Pollut. Bull. 100, 70–81.

Enfrin, M., Dumée, L.F., Lee, J., 2019. Nano/microplastics in water and wastewater treatment processes - origin, impact and potential solutions. Water Res. 161, 621–638.

Enfrin, M., Lee, J., Le-Clech, P., Dumée, L.F., 2020. Kinetic and mechanistic aspects of ultrafiltration membrane fouling by nano- and microplastics. J. Membr. Sci. 601, 117890. https://doi.org/10.1016/j.memsci.2020.117890.

Estahbanati, S., Fahrenfeld, N.L., 2016. Influence of wastewater treatment plant discharges on microplastic concentrations in surface water. Chemosphere 162, 277–284.

Feng, Y., Feng, L.J., Liu, S.C., Duan, J.L., Zhang, Y.B., Li, S.C., Sun, X.D., Wang, S.G., Yuan, X.Z., 2018a. Emerging investigator series: inhibition and recovery of anaerobic granular sludge performance in response to short-term polystyrene nanoparticle exposure. Environ. Sci.: Water Res. Technol. 4, 1902–1911.

Feng, L., Wang, J., Liu, S., Sun, X., Yuan, X., Wang, S., 2018b. Role of extracellular polymeric substances in the acute inhibition of activated sludge by polystyrene nanoparticles. Environ. Pollut. 238, 859–865.

Forstner, C., Orton, T.G., Wang, P., Kopittke, P.M., Dennis, P.G., 2019. Soil chloride content influences the response of bacterial but not fungal diversity to silver nanoparticles entering soil via wastewater treatment processing. Environ. Pollut. 255, 113274–113283.

Fortin, S., Song, B., Burbage, C., 2019. Quantifying and identifying microplastics in the effluent of advanced wastewater treatment systems using Raman microspectroscopy. Mar. Pollut. Bull. 149, 110579–110585.

Fu, S., Ding, J., Zhang, Y., Li, Y., Zhu, R., Yuan, X., Zou, H., 2018. Exposure to polystyrene nanoplastic leads to inhibition of anaerobic digestion system. Sci. Total Environ. 625, 64–70.

Gang, L., Ben, W., Hui, Y., Dong, Z., Qiang, Z., 2018. Performance of ozonation and biological activated carbon in eliminating sulfonamides and sulfonamide-resistant bacteria: a pilot-scale study. Chem. Eng. J. 341, 327–334.

GESAMP (2019). Guidelines or the monitoring and assessment of plastic litter and microplastics in the ocean (Kershaw P.J., Turra A. and Galgani F. editors), (IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP/ISA Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). Rep. Stud. GESAMP No. 99, 130p.

Gherghel, A., Teodosiu, C., De Gisi, S., 2019. A review on wastewater sludge valorisation and its challenges in the context of circular economy. J. Clean. Prod. 228, 244–263.

Gies, E.A., LeNoble, J.L., Noël, M., Etemadifar, A., Bishay, F., Hall, E.R., Ross, P.S., 2018. Retention of microplastics in a major secondary wastewater treatment plant in Vancouver, Canada. Mar. Pollut. Bull. 133, 553–561.

Gurung, K., Ncibi, M.C., Sillanpää, M., 2017. Assessing membrane fouling and the performance of pilot-scale membrane bioreactor (MBR) to treat real municipal wastewater during winter season in Nordic regions. Sci. Total Environ. 579, 1289–1297.

He, H.J., Chen, Y.J., Xiang, L., Yan, C., Yang, C.P., Zeng, G.M., 2016. Influence of salinity on microorganisms in activated sludge processes: a review. Int. Biodeterior. Biodegradation 119, 520–527.

Herbort, A.F., Schuhen, K., 2017. A concept for the removal of microplastics from the marine environment with innovative host-guest relationships. Environ. Sci. Pollut. Res. 24, 11061–11065.

Herbort, A.F., Sturm, M.T., Schuhen, K., 2018. A new approach for the agglomeration and subsequent removal of polyethylene, polypropylene, and mixtures of both from freshwater systems - a case study. Environ. Sci. Pollut. Res. 25, 1–9.

Hidayaturrahman, H., Lee, T.G., 2019. A study on characteristics of microplastic in wastewater of South Korea: identification, quantification, and fate of microplastics during treatment process. Mar. Pollut. Bull. 146, 696–702.

Hongprasith, N., Kittimethawong, C., Lertluksanaporn, R., Eamchotchawalit, T., Kittipongvises, S., Lohwacharin, J., 2020. IR microspectroscopic identification of microplastics in municipal wastewater treatment plants. Environ. Sci. Pollut. Res. 27, 18557–18564.

Hurley, R.R., Lusher, A.L., Olsen, M., Nizzetto, L., 2018. Validation of a method for extracting microplastics from complex, organic-rich, environmental matrices. Environ. Sci. Technol. 52, 7409–7417.

Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R., Law, K.L., 2015. Plastic waste inputs from land into the ocean. Science 347, 768–771.

Jarvis, P., Jefferson, B., Parson, S.A., 2005. Breakage, re-growth, and fractal nature of natural organic matter flocs. Environ. Sci. Technol. 39, 2307–2314.

Kalčíková, G., Alič, B., Skalar, T., Bundschuh, M., Gotvajn, A.Ž., 2017. Wastewater treatment plant effluents as source of cosmetic polyethylene microbeads to freshwater. Chemosphere 188, 25–31.

Kampschreur, M.J., Temmink, H., Kleerebezem, R., Jetten, M.S., van Loosdrecht, M.C., 2009. Nitrous oxide emission during wastewater treatment. Water Res. 43, 4093–4103.

Kanhai, L.D.K., Officer, R., Lyashevska, O., Thompson, R.C., O'Connor, I., 2017. Microplastic abundance, distribution and composition along a latitudinal gradient in the Atlantic Ocean. Mar. Pollut. Bull. 115, 307–314.

Kataoka, T., Nihei, Y., Kudou, K., Hinata, H., 2019. Assessment of the sources and inflow processes of microplastics in the river environments of Japan. Environ. Pollut. 244, 958–965.

Kelkar, V.P., Rolsky, C.B., Pant, A., Green, M.D., Tongay, S., Halden, R.U., 2019. Chemical and physical changes of microplastics during sterilization by chlorination. Water Res. 163, 114871. https://doi.org/10.1016/j.watres.2019.114871.

Kelly, M.R., Lant, N.J., Kurr, M., Burgess, J.G., 2019. Importance of water-volume on the release of microplastic fibers from laundry. Environ. Sci. Technol. 53, 11735–11744.

Lai, C.Y., Groth, A., Gray, S., Duke, M., 2014. Enhanced abrasion resistant PVDF/nanoclay hollow fibre composite membranes for water treatment. J. Membr. Sci. 449, 146–157.

Lee, H., Kim, Y., 2018. Treatment characteristics of microplastics at biological sewage treatment facilities in Korea. Mar. Pollut. Bull. 137, 1–8.

Leslie, H.A., Brandsma, S.H., Velzen, M.J.M.V., Vethaak, A.D., 2017. Microplastics en route: field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota. Environ. Int. 101, 133–142.

Li, X.W., Chen, L.B., Mei, Q.Q., Dong, B., Dai, X.H., Ding, G.J., Zeng, E.Y., 2018. Microplastics in sewage sludge from the wastewater treatment plants in China. Water Res. 142, 75–85.

Li, L., Geng, S.X., Li, Z.Y., Song, K., 2020a. Effect of microplastic on anaerobic digestion of wasted activated sludge. Chemosphere 247, 125874–125881.

Li, L., Song, K., Yeerken, S., Geng, S.X., Liu, D., Dai, Z.L., Xie, F.Z., Zhou, X.H., Wang, Q.L., 2020b. Effect evaluation of microplastics on activated sludge nitrification and denitrification. Sci. Total Environ. 707, 135953–135958.

Liang, Z., Das, A., Hu, Z., 2010. Bacterial response to a shock load of nanosilver in an activated sludge treatment system. Water Res. 44, 5432–5438.

Liu, H., Yang, X., Liu, G., Liang, C., Xue, S., Chen, H., Ritsema, C.J., Geissen, V., 2017. Response of soil dissolved organic matter to microplastic addition in Chinese loess soil. Chemosphere 185, 907–917.

Liu, X., Yuan, W., Di, M., Li, Z., Wang, J., 2019a. Transfer and fate of microplastics during the conventional activated sludge process in one wastewater treatment plant of China. Chem. Eng. J. 362, 176–182.

Liu, Y.R., Wei, D., Xu, W.Y., Feng, R., Du, B., Wei, Q., 2019b. Nitrogen removal in a combined aerobic granular sludge and solid-phase biological denitrification system: system evaluation and community structure. Bioresour. Technol. 288, 121504–121511.

Long, Z.X., Pan, Z., Wang, W.L., Ren, J.Y., Yu, X.G., Lin, L.Y., Lin, H., Chen, H.Z., Jin, X.L., 2019. Microplastic abundance, characteristics, and removal in wastewater treatment plants in a coastal city of China. Water Res. 155, 255–265.

López, M.E., Rene, E.R., Boger, Z., Veiga, M.C., Kennes, C., 2017. Modelling the removal of volatile pollutants under transient conditions in a two-stage bioreactor using artificial neural networks. J. Hazard. Mater. 324, 100–109.

Lv, X.M., Dong, Q., Zuo, Z.Q., Liu, Y.C., Huang, X., Wu, W.M., 2019. Microplastics in a municipal wastewater treatment plant: fate, dynamic distribution, removal efficiencies, and control strategies. J. Clean. Prod. 225, 579–586.

Ma, B.W., Xue, W.J., Ding, Y.Y., Hu, C.Z., Liu, H.J., Qu, J.H., 2019a. Removal characteristics of microplastics by Fe-based coagulants during drinking water treatment. J. Environ. Sci. 78, 267–275.

Ma, B.W., Xue, W.J., Hu, H.J., Qu, J.H., Li, L.L., 2019b. Characteristics of microplastic removal via coagulation and ultrafiltration during drinking water treatment. Chem. Eng. J. 359, 159–167.

Magni, S., Binelli, A., Pittura, L., Avio, C.G., Della Torre, C., Parenti, C.C., Gorbi, S., Regoli, F., 2019. The fate of microplastics in an Italian wastewater treatment plant. Sci. Total Environ. 652, 602–610.

Magnusson, K., Norén, F., 2014. Screening of Microplastic Particles in and Down-Stream a Wastewater Treatment Plant. Report Number C55. IVL Swedish Environmental Research Institute, Stockholm.

Mahon, A.M., Connell, B.O., Healy, M.G., Connor, I.O., Officer, R., Nash, R., Morrison, L., 2017. Microplastics in sewage sludge: effects of treatment. Environ. Sci. Technol. 51, 810–818.

Mason, S., Garneau, D., Sutton, R., Chu, Y., Ehmann, K., 2016. Microplastic pollution is widely detected in US municipal wastewater treatment plant effluent. Environ. Pollut. 218, 1045–1054.

Massara, T.M., Malamis, S., Guisasola, A., Baeza, J.A., Noutsopoulos, C., Katsou, E., 2017. A review on nitrous oxide (N_2O) emissions during biological nutrient removal from municipal wastewater and sludge reject water. Sci. Total Environ. 596, 106–123.

Michielssen, M.R., Michielssen, E.R., Ni, J., Duhaime, M.B., 2016. Fate of microplastics and other small anthropogenic litter (SAL) in wastewater treatment plants depends on unit processes employed. Environ. Sci.: Water Res. Technol. 2, 1064–1073.

Mintenig, S.M., Int-Veen, I., Löder, M.G.J., Primpke, S., Gerdts, G., 2017. Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging. Water Res. 108, 365–372.

Moharir, R.V., Kumar, S., 2019. Challenges associated with plastic waste disposal and allied microbial routes for its effective degradation: a comprehensive review. J. Clean. Prod. 208, 65–76.

Mourgkogiannis, N., Kalavrouziotis, I.K., Karapanagioti, H.K., 2018. Questionnaire based survey to managers of 101 wastewater treatment plants in Greece confirms their potential as plastic marine litter sources. Mar. Pollut. Bull. 133, 822–827.

Murphy, F., Ewins, C., Carbonnier, F., Quinn, B., 2016. Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment. Environ. Sci. Technol. 50, 5800–5808.

Nasuhoglu, D., Isazadeh, S., Westlund, P., Neamatallah, S., Yargeau, V., 2018. Chemical, microbial and toxicological assessment of wastewater treatment plant effluents during disinfection by ozonation. Chem. Eng. J. 346, 466–476.

Pal, P., 2017. Industrial Water Treatment Process Technology. Butterworth-Hei-nemann, Oxford, UK, pp. 537–544.
Pan, Z., Sun, X.W., Guo, H.G., Cai, S.Z., Chen, H.Z., Wang, S.M., Zhang, Y.B., Lin, H., Huang, J., 2019. Prevalence of microplastic pollution in the Northwestern Pacific Ocean. Chemosphere 225, 735–744.
Peng, L.Y., Yang, C.P., Zeng, G.M., Lu, W., Dai, C.H., Long, Z.Y., Liu, H.Y., Zhong, Y.Y., 2014. Characterization and application of bioflocculant prepared by *Rhodococcus erythropolis* using sludge and livestock wastewater as cheap culture media. Appl. Microbiol. Biotechnol. 98, 6847–6858.
Petroody, S.S.A., Hashemi, S.H., van Gestel, C.A.M., 2020. Factors affecting microplastic retention and emission by a wastewater treatment plant on the southern coast of Caspian Sea. Chemosphere 261, 128179.
Pittura, L., Foglia, A., Akyol, Ç., Cipolletta, G., Benedetti, M., Regoli, F., Eusebi, A.L., Sabbatini, S., Tseng, L.Y., Katsou, E., Gorbi, S., Fatone, F., 2021. Microplastics in real wastewater treatment schemes: comparative assessment and relevant inhibition effects on anaerobic processes. Chemosphere 262, 128415.
Pivokonsky, M., Cermakova, L., Novotna, K., Peer, P., Caithaml, T., Janda, V., 2018. Occurrence of microplastics in raw and treated drinking water. Sci. Total Environ. 643, 1644–1651.
Prata, J.C., 2018. Microplastics in wastewater: state of the knowledge on sources, fate and solutions. Mar. Pollut. Bull. 129, 262–265.
Qiao, X.L., Zhang, Z.J., Wang, N.C., Victor, W., Megan, L., Loh, C.S., Ng, T.H., 2008. Coagulation pretreatment for a large-scale ultrafiltration process treating water from the Taihu river. Desalination 230, 305–313.
Qin, R.H., Su, C.Y., Liu, W.H., Tang, L.Q., Li, X.J., Deng, X., Wang, A.L., Chen, Z.P., 2020. Effects of exposure to polyether sulfone microplastic on the nitrifying process and microbial community structure in aerobic granular sludge. Bioresour. Technol. 302, 122827–122835.
Rajala, K., Grönfors, O., Hesampour, M., Mikola, A., 2020. Removal of microplastics from secondary wastewater treatment plant effluent by coagulation/flocculation with iron, aluminum and polyamine-based chemicals. Water Res. 183, 116045.
Raju, S., Carbery, M., Kuttykattil, A., Senthirajah, K., Lundmark, A., Rogers, Z., SCB, S., Evans, G., Palanisami, T., 2020. Improved methodology to determine the fate and transport of microplastics in a secondary wastewater treatment plant. Water Res., 115549 https://doi.org/10.1016/j.watres.2020.115549.
Rochman, C.M., Kross, S.M., Armstrong, J.B., Bogan, M.T., Darling, E.S., Green, S.J., Smyth, A.R., Veríssimo, D., 2015. Scientific evidence supports a ban on microbeads. Environ. Sci. Technol. 49, 10759–10761.
Rodrigues, J.P., Duarte, A.C., Santos-Echeandía, J., Rocha-Santos, T., 2019. Significance of interactions between microplastics and POPs in the marine environment: a critical overview. TrAC Trends Anal. Chem. 111, 252–260.
Ruan, Y.F., Zhang, K., Wu, C.X., Wu, R.B., Lam, P.K.S., 2019. A preliminary screening of HBCD enantiomers transported by microplastics in wastewater treatment plants. Sci. Total Environ. 674, 171–178.
Sbardella, L., Comas, J., Fenu, A., Rodriguez-Roda, I., Weemaes, M., 2018. Advanced biological activated carbon filter for removing pharmaceutically active compounds from treated wastewater. Sci. Total Environ. 636, 519–529.
Shen, L., Xiao, H., Yang, W.Q., Miao, D.R., Li, X.M., 2013. Study on biological aerated filter as enhanced pretreatment process of coagulation and sedimentation. Adv. Mater. Res. 726-731, 1940–1944.
Sillanpää, M., Ncibi, M.C., Matilainen, A., Vepsäläinen, M., 2018. Removal of natural organic matter in drinking water treatment by coagulation: a comprehensive review. Chemosphere 190, 54–71.
Song, K., Li, Z.Y., Liu, D., Li, L., 2020. Analysis of the partial nitrification process affected by polyvinylchloride microplastics in treating high-ammonia anaerobic digestates. ACS Omega 5, 23836–23842.
Sun, X.M., Chen, B.J., Li, Q.F., Liu, N., Xia, B., Zhu, L., Qu, K.M., 2018. Toxicities of polystyrene nano- and microplastics toward marine bacterium *Halomonas alkaliphila*. Sci. Total Environ. 642, 1378–1385.
Sun, J., Dai, X.H., Wang, Q.L., Van Loosdrecht, M.C.M., Ni, B.J., 2019. Microplastics in wastewater treatment plants: detection, occurrence and removal. Water Res. 152, 21–37.
Sutton, R., Mason, S.A., Stanek, S.K., Willis-Norton, E., Wren, I.F., Box, C., 2016. Microplastic contamination in the San Francisco Bay, California, USA. Mar. Pollut. Bull. 109, 230–235.
Talvitie, J., Mikola, A., Koistinen, A., Setälä, O., 2017a. Solutions to microplastic pollution - removal of microplastics from wastewater effluent with advanced wastewater treatment technologies. Water Res. 123, 401–407.
Talvitie, J., Mikola, A., Setälä, O., Heinonen, M., Koistinen, A., 2017b. How well is microlitter purified from wastewater? - a detailed study on the stepwise removal of microlitter in a tertiary level wastewater treatment plant. Water Res. 109, 164–172.
Tang, W.C., Li, X., Liu, H.Y., Wu, S.H., Zhou, Q., Du, C., Teng, Q., Zhong, Y.Y., Yang, C.P., 2020. Sequential vertical flow trickling filter and horizontal flow reactor for treatment of decentralized domestic wastewater with sodium dodecyl benzene sulfonate. Bioresour. Technol. 300, 122634. https://doi.org/10.1016/j.biortech.2019.122634.
Turan, N.B., Erkan, H.S., Engin, G.O., 2021. Microplastics in wastewater treatment plants: occurrence, fate and identification. Process Saf. Environ. Prot. 146, 77–84.
Verma, A.K., Dash, R.R., Bhunia, P., 2012. A review on chemical coagulation/flocculation technologies for removal of colour from textile wastewaters. J. Environ. Manag. 93, 154–168.
Von Sonntag, C., von Gunten, U., 2012. Chemistry of Ozone in Water and Wastewater Treatment: From Basic Principles to Applications. IWA Publishing, London, UK.
Wang, S.M., Chen, H.Z., Zhou, X.W., Tian, Y.Q., Lin, C., Wang, W.L., Zhou, K.W., Zhang, Y.B., Lin, H., 2020a. Microplastic abundance, distribution and composition in the mid-west Pacific Ocean. Environ. Pollut. 114125. https://doi.org/10.1016/j.envpol.2020.114125.
Wang, Z.F., Lin, T., Chen, W., 2020b. Occurrence and removal of microplastics in an advanced drinking water treatment plant (ADWTP). Sci. Total Environ. 700, 134520–134528.
Wei, W., Huang, Q., Sun, J., Dai, X., Ni, B., 2019a. Revealing the mechanisms of polyethylene microplastics affecting anaerobic digestion of waste activated sludge. Environ. Sci. Technol. 53, 9604–9613.
Wei, W., Huang, Q.S., Sun, J., Wang, J.Y., Wu, S.L., Ni, B.J., 2019b. Polyvinylchloride microplastics affect methane production from the anaerobic digestion of waste activated sludge through leaching toxic bisphenol-A. Environ. Sci. Technol. 53, 2509–2517.
Wei, W., Zhang, Y., Huang, Q.S., Ni, B.J., 2019c. Polyethylene terephthalate microplastics affect hydrogen production from alkaline anaerobic fermentation of waste activated sludge through altering viability and activity of anaerobic microorganisms. Water Res. 163, 114881.
Wu, M.J., Yang, C.P., Du, C., Liu, H.Y., 2020. Microplastics in waters and soils: occurrence, analytical methods and ecotoxicological effects. Ecotoxicol. Environ. Saf. 202, 110910–110921.
Xiao, Y., De Araujo, C., Sze, C.C., Stuckey, D.C., 2015. Toxicity measurement in biological wastewater treatment processes: a review. J. Hazard. Mater. 286, 15–29.
Xu, S., Zhang, Y., Sims, A., Bernards, M., Hu, Z., 2013. Fate and toxicity of melamine in activated sludge treatment systems after a long-term sludge adaptation. Water Res. 47, 2307–2314.
Xu, X., Jian, Y., Xue, Y.G., Hou, Q.T., Wang, L.P., 2019. Microplastics in the wastewater treatment plants (WWTPs): occurrence and removal. Chemosphere 235, 1089–1096.
Yang, C.P., Chen, H., Zeng, G.M., Yu, G.L., Luo, S.L., 2010. Biomass accumulation and control strategies in gas biofiltration. Biotechnol. Adv. 28, 531–540.
Yang, C.P., Qian, H., Li, X., Cheng, Y., He, H.J., Zeng, G.M., Xi, J.Y., 2018. Simultaneous removal of multicomponent VOCs in biofilters. Trends Biotechnol. 36, 673–685.
Zahrim, A.Y., Dexter, Z.D., Joseph, C.G., Hilal, N., 2017. Effective coagulation-flocculation treatment of highly polluted palm oil mill biogas plant wastewater using dual coagulants: Decolourisation, kinetics and phytotoxicity studies. J. Water Process Eng. 16, 258–269.
Zhang, Z.Q., Chen, Y.G., 2020. Effects of microplastics on wastewater and sewage sludge treatment and their removal: a review. Chem. Eng. J. 382, 122955–122970.
Zhang, Q.D., Liu, S.J., Yang, C.P., Chen, F.M., Lu, S.L., 2014. Bioreactor consisting of pressurized aeration and dissolved air flotation for domestic wastewater treatment. Sep. Purif. Technol. 138, 186–190.
Zhang, X.L., Chen, J.X., Li, J., 2020. The removal of microplastics in the wastewater treatment process and their potential impact on anaerobic digestion due to pollutants association. Chemosphere 251, 126360.
Zhao, Y.X., Chen, Y.G., Zhang, D., Zhu, X.Y., 2010. Waste activated sludge fermentation for hydrogen production enhanced by anaerobic process improvement and acetobacteria inhibition: the role of fermentation pH. Environ. Sci. Technol. 44, 3317–3323.
Zhao, L.J., Su, C.Y., Liu, W.H., Qin, R.H., Tang, L.Q., Deng, X., Wu, S.M., Chen, M.L., 2020. Exposure to polyamide 66 microplastic leads to effects performance and microbial community structure of aerobic granular sludge. Ecotoxicol. Environ. Saf. 190, 110070–110079.
Zhou, Q., Li, X., Lin, Y., Yang, C.P., Tang, W.C., Wu, S.H., Li, D.H., Lou, W., 2019. Effects of copper ions on removal of nutrients from swine wastewater and release of dissolved organic matters in duckweed systems. Water Res. 2019 (158), 171–181.
Zhu, D., Chen, Q.L., An, X.L., Yang, X.R., Christie, P., Ke, X., Wu, L.H., Zhu, Y.G., 2018. Exposure of soil collembolans to microplastics perturbs their gut microbiota and alters their isotopic composition. Soil Biol. Biochem. 116, 302–310.
Ziajahromi, S., Neale, P.A., Rintoul, L., Leusch, F.D.L., 2017. Wastewater treatment plants as a pathway for microplastics: development of a new approach to sample wastewater-based microplastics. Water Res. 112, 93–99.
Ziajahromi, S., Neale, P.A., Silveira, I.T., Chua, A., Leusch, F.D.L., 2021. An audit of microplastic abundance throughout three Australian wastewater treatment plants. Chemosphere 263, 128294.