The present document is the Post-Print version of the article entitled “The Sorption of Amoxicillin on Engineered Polyethylene Terephthalate Microplastics”, Journal of Polymers and the Environment (2023) 31:1383–1397, DOI: 10.1007/s10924-022-02690-0. The final version of the manuscript can be found at the following link https://doi.org/10.1007/s10924-022-02690-0.

The sorption of amoxicillin on engineered polyethylene terephthalate microplastics

Francesca Lionetto[1*], Carola Esposito Corcione[1], Francesco Messa[2], Serena Perrone[2], Antonio Salomone[3], Alfonso Maffezzoli[1]

[1]Department of Engineering for Innovation, University of Salento, Via per Monteroni, 73100 Lecce, Italy

[2]Department of Biological and Environmental Sciences and Technologies (DISTEBA), University of Salento, Via per Monteroni, 73100 Lecce, Italy

[3]Dipartimento di Chimica, Università degli Studi di Bari Aldo Moro, Via Orabona 4, 70125 Bari, Italy

[*]Corresponding author: francesca.lionetto@unisalento.it

Orcid IDs:

Francesca Lionetto: 0000-0003-4466-1161

Carola Esposito Corcione: 0000-0002-3056-1162

Francesco Messa: 0000-0002-1664-6151

Serena Perrone: 0000-0003-4887-6805

Antonio Salomone: 0000-0002-3161-385X

Alfonso Maffezzoli: 0000-0002-6371-4030

The sorption of amoxicillin on engineered polyethylene terephthalate microplastics

Francesca Lionetto[1*], Carola Esposito Corcione[1], Francesco Messa[2], Serena Perrone[2], Antonio Salomone[3], Alfonso Maffezzoli[1]

[1]Department of Engineering for Innovation, University of Salento, Via per Monteroni, 73100 Lecce, Italy

[2]Department of Biological and Environmental Sciences and Technologies (DISTEBA), University of Salento, Via per Monteroni, 73100 Lecce, Italy

[3] Dipartimento di Chimica, Università degli Studi di Bari Aldo Moro, Via Orabona 4, 70125 Bari, Italy

[*]Corresponding author: francesca.lionetto@unisalento.it

Abstract

The adsorption studies of contaminants on microplastics (MPs) collected from the marine environment are very hard to carry out mainly due to the difficulties associated with both to filtration of MPs and separation from biofilm and organic matrices. In this work, MPs were produced by a top-down protocol from polyethylene terephthalate (PET) bottles collected on the beach, thus already aged in the natural environment, and compared with engineered MPs obtained from PET pellets. Both types of MPs (size <150 μm) were used to study the adsorption of amoxicillin, which is one of the most widely consumed antibiotics in the world and is found unchanged in the aquatic environment. The results of sorption kinetics and isotherm tests indicated that aged MPs absorbed a higher antibiotic content than unaged ones since the two kinds of microplastics had different specific surface areas.

The experimental results were explained by analysing the thermodynamic affinity among amoxicillin and PET MPs and comparing it with several pharmaceuticals and other microplastics by evaluating Hansen's solubility parameters (HSPs), which account for dispersive, polarizable and hydrogen bonding contributions to the overall cohesive energy of a compound. The possible interaction mechanism among amoxicillin and PET MPs, based on hydrogen bond interactions among the antibiotic and the ester groups of the polymer, was hypothesised. The results of adsorption tests demonstrated that PET MPs can be pollutant carriers with potential long-range transport in the aquatic environment.

Keywords: microplastics, pollutants, adsorption, amoxicillin, polyethylene terephthalate, solubility parameter ~~ocean pollution~~

1. INTRODUCTION

The annual production of plastics worldwide has increased significantly over the past decades. In 2018, almost 360 million tonnes of plastics were produced [1] and by 2050 plastic production is expected to intensify to 2000 million tonnes. Synthetic organic polymers became the fastest growing segment of the municipal waste stream between 1950 and 2003. Due to the global COVID19, the use of plastic based personal protection equipment's has significantly increased, and consequently the plastic waste production has exponentially risen [2,3]. In particular, their presence in the marine environment has emerged as a significant problem as they make up about 80 % to 85% of marine litter [4,5]. Plastics come, in fact, into the aquatic environment in a wide range of sizes [6,7] and they undergo slow degradation under natural conditions, caused either by chemical-physical processes (such as photo-oxidation, thermal activation, hydrolysis, mechanical stress) or by biological activity, into smaller fragments referred to as microplastics (MPs) [8,9]. MPs, particles smaller than or equal to 5 mm in ~~diameter~~ size and with a regular or irregular shape, are ubiquitous environmental contaminants found in seawater and sediments [10], wastewater [11], soil [12,13], air [14] and even food products (e.g. beer, sea salt and tap water) [15]. MPs represent 92% of the 5.25 trillion plastic particles on the ocean surface [4,16] and can originate from two sources, depending on whether primary or secondary microplastics. Primary MPs are manufactured purposely for industrial or domestic applications (toothpastes, exfoliating facial scrubs in personal care products etc.) [4]. Secondary microplastics originate from the breakdown of larger macroplastic debris under environmental conditions [17,18]. Once entered the aquatic environments, MPs may have a great impact on marine life and their ubiquitous nature leads to the inevitable human exposure to these particles, posing risks to human health [19,20]. The most frequently detected MPs in the aquatic environment are composed of polyethylene (PE), polyethylene terephthalate (PET), polypropylene (PP), polyamide (PA), polystyrene (PS) and polyvinyl chloride (PVC) [21,22]. In particular, PET, can be considered the largest utilized polymer for packaging, textiles and bottles, with global manufacture of a minimum 30 million tons per year [23,24]. PET litter in the marine environment has recently been recognised as one of the most hazardous wastes, causing numerous damage, such as the embarrassment of the marine species within the plastic or their ingestion [23]. Even though PET exhibits resistance to weathering, fragmentation processes are not immune to it. Several degradation processes are expected to happen in marine ecosystems, due to abiotic weathering by

photo-oxidation and hydrolysis, and also to the biological activity [25]. The pH change in the ocean may probably modify the chemical balance of microplastics by increasing or decreasing the rate of chemical leaching from their surface, so PET, which is commonly understood to be safe, may become dangerous in the near future. PET microparticles occur in surface waters and sediments around the world but it is hard to define the precise abundance in different regions and environmental matrices because of variability and challenges in sampling techniques [22,26]. According to Burns et al. [26] and Kooi and Koelmans [27], the shares of PET in the aquatic environment is 16.5%. Starting from the lower and upper concentrations reported by Murphy et al. [28], Long et al. [29] and Wolff et al. [30], Schernewski et al. [31] calculated an average minimum of 401 PET particles/m^3 and an average maximum of 14492 PET particles/m^3 in raw wastewater and stormwater. Even more difficult is the quantification of PET nanoplastics, due to the very small size and the limit of actual sampling techniques. However, there is evidence of the occurrence of PET nanoparticles in the environment and living organisms [32].

A wide variety of pollutants can be absorbed in microplastics, including persistent pollutants, pharmaceuticals, pesticides, herbicides and heavy metals [34,35]. The chemical compound, also named sorbate, is distributed depending on its affinity for the sorbent phases. Therefore, the distribution of chemical contaminants in the environment depends directly on the physicochemical properties of both the MPs and the chemical contaminants as well as environmental characteristics [36,37]. It has been recognized that the content adsorbed is higher than the pollutant level in the surrounding environment [38]. Moreover, MPs and NPs can act as carriers or vectors for pollutants to transport these chemicals and release them into the water environment and to biota [39,40]. Currently, there is a heated debate if the adsorption of pollutants on MPs could affect their fate and toxicity in the environment and/or increase their bioavailability, causing enhanced health risks compared to these pollutants alone [41,42].

Pharmaceuticals are emerging organic pollutants frequently detected in natural waters. The huge consumption of pharmaceuticals in medicine and animal farming rises environmental risks. About 40–90% of pharmaceuticals administered are excreted in faeces and urine as parent compounds (in the active form) into the environment [43]. Pharmaceuticals ultimately find their way into the environment through runoffs, leaks, and sewage systems, and these could contaminate soils, water bodies, and plants [43]. Moreover, the use of huge amounts of pharmaceuticals in animal farming can result in agro-ecosystem contamination. This occurs when contaminated manure is applied on agricultural lands as fertilizer and crops are irrigated with wastewater.

Among pharmaceuticals, antibiotics have received increasing attention in the last few decades, due to their impacts on the microbial community as well as the generation of antimicrobial resistance

Furthermore, they represent an important class of contaminants, frequently detected in the aquatic environment worldwide [44].

According to the World health Organization [45], amoxicillin (AMX) is among the most widely consumed antibiotics in the world and it is used as a first-line drug both in human and veterinary medicine in many countries [46]. AMX is a semi-synthetic pharmaceutical product, belonging to the class of β-lactam antibiotics, and it is a broad-spectrum antibiotic used for the treatment and prevention of human respiratory, gastrointestinal, urinary, and skin bacterial infections; moreover, this drug is used extensively for animal diseases as well as growth promoters for many domestic and food animals [46,47]. The original antibiotic is excreted unchanged in the urine and faeces at high rates; therefore, it is likely to find amoxicillin traces in the aquatic environment [48]. Several recent studies demonstrate the persistence of AMX in wastewater treatment plants since they are not completely removed by biological treatments [49]. For example, this pollutant has been found at the ng/L level in various countries such as India [50], Turkey [51], Italy [49], Ghana [52] and Australia [44]. The concentrations of AMX in different water bodies around the world generally range from 1.8 ng/L to 1.6 µg/L [46], but also higher values, for example, 6.9 µg/L [53], have been also detected in some wastewater treatment plants. Khasawneh et al. [54], by reviewing the current data available on the global occurrence of 43 pharmaceutical compounds in municipal wastewater treatment plants in the period from 2010 to 2020, found that amoxicillin recorded the highest daily emissions (944 g/day). Notably, Otoo *et al.* [55] found a concentration of 8.8 mg/L of amoxicillin in hospital effluents. Recent studies have highlighted the potential environmental concerns of this pharmaceutical in some hotspot areas, although further efforts are required to investigate the ecological consequences of AMX [56]. Antibiotics can affect natural microbial communities through the disappearance and/or inhibition of some microbial groups involved in important ecosystem functions [57] or the generation and spread of antibiotic-resistant genes, bacteria, and even superbugs [58]. Therefore, the potential ecotoxicological risk associated with the presence of antibiotics in the aquatic environment has turned the spotlight on the adsorption of antibiotics on microplastics ad their transport.

Therefore, the study of the interactions between antibiotics and MPs has gained increasing attention in recent years for the assessment of environmental risks. Li *et al*. [59] analysed the adsorption of five antibiotics (sulfadiazine, amoxicillin, tetracycline, ciprofloxacin, and trimethoprim) on five types of microplastics (PE, PS, PP, PA, PVC) in the aquatic environment. Adsorption experiments revealed that PA, characterized by a porous structure and hydrogen bonding, had the strongest adsorption capacity for antibiotics [59]. Additionally, Guo *et al*. [60] used tylosin, a widely used veterinary antibiotic, as a model pollutant to test the sorption performance of four types of MPs (PE, PP, PS and

PVC). They found that the sorption ability of tylosin on the sorbent MPs, mostly dominated by electrostatic interactions, surface complexations and hydrophobic interactions, followed the order of PE < PP < PS < PVC [60].

Despite the wide diffusion of both the pollutant and PET MP, very little is known in the literature on the sorption behaviour of amoxicillin on PET, except for the work of Godoy et al. [61], who tested PET microparticles derived from unaged PET bottles with an average size of 2.7 mm. However, due to the continues degradation and fragmentation of plastic debris, the lack of knowledge on the sorption behaviour of micro-, sub-micro and PET nanoplastics, which has experienced natural weathering, has now become critical. Moreover, it is hard to use environmental PET MPs due to the difficulty in gathering large volumes of environmental samples from a single polymer source. Additionally, the difficulty of separating microplastics from environmental samples, the surface impurities and the heterogeneity in plastic sizes and ageing periods also hinder the use of collected MPs for sorption studies [62,63]. For this purpose, some of the authors have previously set up a fast top-down approach based on mechanical fragmentation for producing polyethylene terephthalate (PET) model microplastics and nanoplastics [20]. The obtained PET nanoplastics have been also applied in biological studies through in vitro exposure to mussel hemolymphatic cells [64].

This study focuses on the adsorption of amoxicillin on naturally weathered PET MPs of size lower than 150 μm. To the authors' knowledge, sorption studies of this antibiotic have been carried out only on unaged MPs with 20 times higher size. For this purpose, PET MPs have been produced by a top-down protocol from littered bottles collected on the beach of Torre Colimena (TA) in the South of Italy, thus already aged in the natural environment. The aged MPs have been compared by means of morphological and physical characterization with engineered MPs obtained from PET pellets. The results of the sorption kinetics and equilibrium isotherms have been discussed by analysing the properties of the aged and unaged MPs and the thermodynamic affinity among amoxicillin and PET MPs. This study provides new insights into the mechanism of interactions between PET MPs and antibiotics and highlight the potential for MPs to accumulate and transport these type of pollutants.

3. Experimental

3.1 Materials

Microplastics (MPs) were obtained from beached PET bottles collected on the beach of Torre Colimena (TA) (40°17′58.88″N, 17°44′37″E, Fig. 1) on the Ionian Sea in the South of Italy. The bottles were washed several times in distilled water in order to remove sand and left to dry at room temperature. After that, parts of bottles coated with biofilms were excluded from grinding. Then, the remaining parts were cut into small pieces (about 3 x 5 cm^2) and crushed in a Retsch SM 2000 cutting

mill (RETSCH GmbH & Co., Düsseldorf, Germany). The obtained powder was finely milled to the micrometric size using a RETSCH ZM100 Ultra Centrifugal mill (RETSCH GmbH & Co., Düsseldorf, Germany) at 14.000 rpm. A second type of secondary MPs derived from PET pellets was used in the present study. The pellets, with an average ~~diameter~~ size of 2.4 mm and length of 4 mm, were purchased from Invista Resins & Fibers GmbH (Gersthofen, Germany) and reduced to micrometric size by milling using a RETSCH ZM100 Ultra Centrifugal mill (RETSCH GmbH & Co., Düsseldorf, Germany) at 14.000 rpm [23]. A schematic representation of the two typologies of the secondary microplastics studied in this work is reported in Fig. 1.

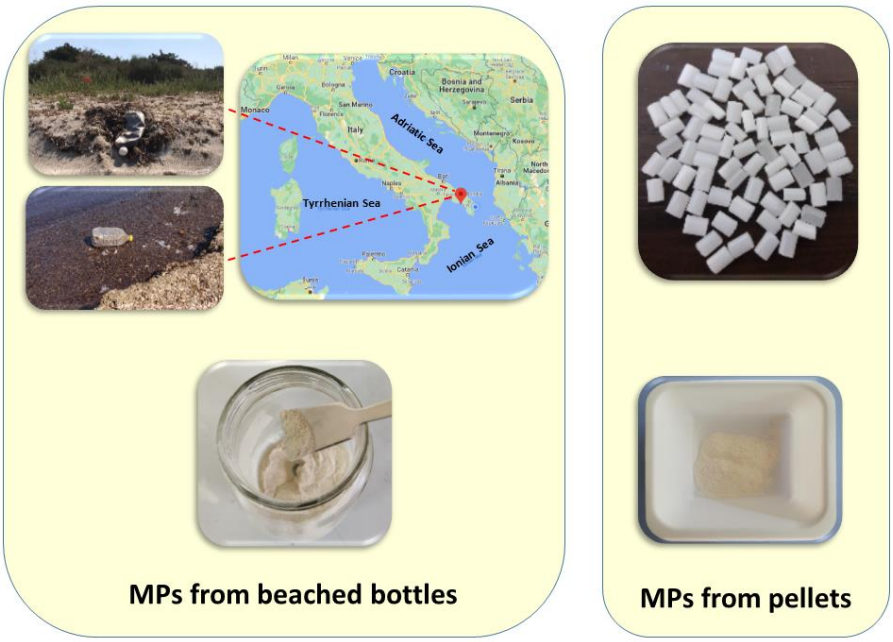

Fig. 1 Typologies and origin of the secondary microplastics studied in this work.

The antibiotic amoxicillin, AMX, (Fig. 2) was used to evaluate the sorption behavior in both types of PET MPs. It was purchased from Sigma-Aldrich in the form of white solid powder with a molecular weight of 419.45 mol/g (CAS: 61336-70-7). Working standard of amoxicillin trihydrate with a purity (HPLC area %) equal to 99.5 % was used for the construction of the calibration curve during sorption experiments while amoxicillin trihydrate of analytical reagent grade (purity ≥ 95%) was used to prepare the solutions for kinetic and isothermal tests. For the sorption test, potassium dihydrogen phosphate (KH_2PO_4) of analytical grade was used while methanol and water of HPLC grade were used without further purification. Commercially available ultrapure water was employed for the preparation of amoxicillin solution for kinetic and isothermal tests. All chemicals for the sorption tests were purchased from Sigma-Aldrich.

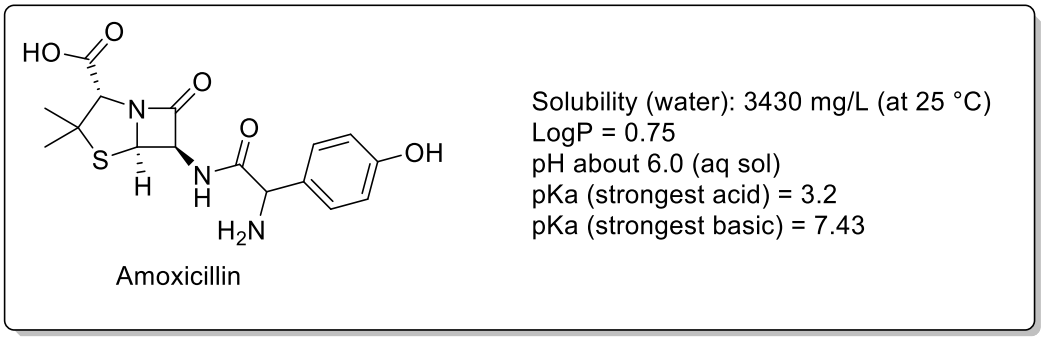

Fig. 2 Structure and main properties of AMX.

3.2 Characterization of microplastics

X-Ray Diffraction (XRD) analysis was carried out on a X-Ray diffractometer (Rigaku, Tokyo, Japan) with CuKα radiation (λ = 1.5418 A°) in the step scanning mode recorded in the 2θ range of 10°–60° with a step size of 0.02° and step duration of 0.5 s. Three replicates for each measurement were performed.

Scanning electron microscopy (SEM) was performed on MP samples spin-coated on a steel plate using a Zeiss EVO 40 SEM instrument at a 20 kV voltage acceleration. The statistical analysis of nanoplastic sizes was performed by measuring the particle diameters and area by using the ImageJ software on more than 500 microparticles for each typology.

Nitrogen adsorption/desorption measurements were carried out at 77 K using a Brunauer-Emmett-Teller (BET) surface area and pore size analyzer (NOVA 2200e, Quantachrome Instruments, USA). The samples were degassed at 50 °C for 24 hours prior to any measurement. At least three measurements were performed for each MP type.

The analysis of particle size distribution was performed by laser diffraction using a CILAS 1190 multi-laser particle size analyzer and Particle Expert® software (CPS Us, Inc., Madison, WI, USA). At least three measurements were performed for each MP type.

3.3 Sorption measurements

The analysis was carried out using a high-pressure liquid chromatography (HPLC) system (model Agilent 1260 series) equipped with a binary pump (G1312B), auto sampler (G1367E), thermo-stated column compartment (G1316C) and UV DAD detector (G1315C), controlled by the Chem Station software Agilent. The analysis was carried out on a binary HPLC pump system using the column Raptor ARC-18 2.7 μm (150 mm x 4.6 mm, 2.7 μm) with a mobile phase flow of 1.00 mL/min, using 20 μL injection volume, with UV detection of 220 nm at 25 °C. The wavelength of 220 nm was

selected for the estimation of amoxicillin peak on the basis of previous studies [65]. The mobile phase was a mixture 95/5 v/v of water solution 0.04 M of potassium dihydrogen phosphate monohydrate KH_2PO_4 (pH = 4.2) and HPLC grade methanol, respectively.

A stock solution of amoxicillin trihydrate working standard (50 mg/L) was prepared by dissolving 10 mg in 200 mL of ultrapure water. For setting the calibration curve, the stock solution was properly diluted to obtain 11 standard solutions with a final concentration of 18.8, 15.0, 12.0, 10.0, 8.0, 6.0, 4.0, 2.0, 0.8, 0.6, 0.4 mg/L. Each standard preparation was analyzed by HPLC. Each analysis was repeated three times and the medium value was recorded. Calibration curve was created by plotting peak areas *vs.* concentrations. Linearity was assessed by evaluating the coefficient of determination (R^2), using weighted regression method ($1/x$).

Sorption kinetic tests

Sorption kinetic tests were carried out on PET microplastics, derived both from pellets and beached bottles, with amoxicillin trihydrate, a contaminant with a great affinity for PET. The tests were conducted following a methodology adapted from Godoy *et al.*[61]: 0.1 g of microplastics were immersed in a 250 mL Erlenmayer flask containing 100 mL of ultrapure water solution of amoxicillin trihydrate. Three antibiotic concentrations (C_i) were adopted, 5 mg/L, 20 mg/L and 50 mg/L. As reported by recent literature [66], microplastics related studies should quantify the number of involved particles. By considering the density of PET equal to 1.38 g/cm^3 and by hypothesizing a spherical shape of MPs with an average size reported in Table 3 ($D_{average}$), a rough number estimation of about 5000 and 7000 particles for MPs from beached bottles and pellet, respectively, has been obtained. The higher AMX content in aqueous solution compared to those naturally occurring in the aquatic environment was chosen to improve the accuracy and precision of the analyses, as suggested in the literature [59,61,67]. These solutions were placed under magnetic stirring at 100 rpm in a thermostatic water bath at 25 °C for 14 days in order to mimic the agitation occurring in the natural aquatic environment, where MPs are rarely static [61], and also to prevent MP sinking or floating [66]. The kinetic studies were carried out for up to 14 days because, after this time, the degradation of amoxicillin becomes a not negligible phenomenon. As reported by Gozlan *et al*. [68], in an aqueous medium the four-membered *β*-lactam rings of AMX could be hydrolyzed to yield the intermediate AMX-penicilloic acid, which contains an extra free carboxylic acid group. A sample of 1 mL of AMX aqueous solution was taken at different times, filtered over a 0.45 μm membrane and analyzed by HPLC determining the concentration at time t. Each analysis was carried out at least three times. A solution of amoxicillin trihydrate (5 mg/L, 20 mg/L and 50 mg/L) without microplastics was subjected to the same procedure and used as a blank control [68]. The sorption capacity for the microplastic was calculated by the following equations:

$$q_t = \frac{(C_i - C_t)}{m} \times V \quad (1)$$

where $\mathbf{q_t}$ was the amoxicillin concentration (mg/g) adsorbed on MPs at each time value; $\mathbf{C_i}$ was the initial concentration (mg/L), $\mathbf{C_t}$ was the concentration (mg/L) at time t, **m** was the mass of microplastics (g); **V** was the sample volume (L).

Sorption isotherm tests

Sorption isotherm tests were carried out with amoxicillin trihydrate on PET microplastics derived both form pellets and beached bottles. For these experiments, ultrapure water solutions containing AMX at five different concentrations (5, 10, 20, 35 and 50 mg/L) were prepared. The higher AMX content in aqueous solution compared to those naturally occurring in the aquatic environment has been chosen to improve the accuracy and precision of the analyses, as suggested in the literature [59,61,67]. Then, in a 250 mL Erlenmeyer flask containing 100 mL of each solution, 0.1 g of microplastics were immersed; the mixture was placed under magnetic stirring at 100 rpm in a thermostatic water bath at 25 °C, to simulate the motion occurring in the natural aquatic environment. On day 14 of the trial, a sample was taken and the amount of contaminant present in the solution was determined by the HPLC method, as described above. Each analysis was carried out at least three times. A solution of amoxicillin trihydrate at five different concentrations (5, 10, 20, 35 and 50 mg/L) without microplastics was subjected to the same procedure and used as a blank control. The experiment was stopped on day 14 to prevent the antibiotic degradation [68].

4. Results

The time evolution of the sorption capacity of the investigated MPs at the different AMX concentrations is reported in Fig. 3. It is evident the increase of the sorption capacity with the AMX content. At each pollutant concentration, the MPs obtained from beached samples present an increased sorption capacity compared to those produced from pellets.

The curves do not reach the equilibrium in the time of the experiments. This is in agreement with the results of Godoy et al. [61] on bigger microplastics obtained from unaged PET bottles with an average size of 2.7 mm. Failure to reach equilibrium is further proof of the danger of these microplastics further exacerbated by their prolonged stay in the marine environment.

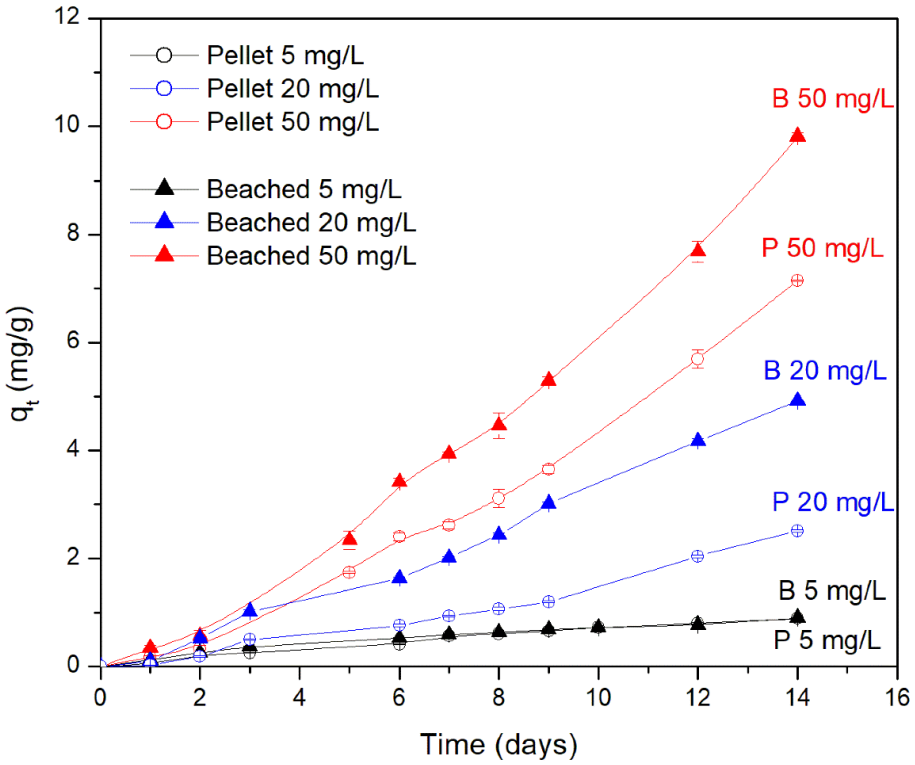

Fig. 3 Amoxicillin sorption capacity onto PET microplastics derived from pellets and beached bottles as a function of time.

Among the numerous adsorption kinetic models reported in the literature [37], the most common ones applied to predict the adsorption of pollutants onto microplastics are the pseudo-first order (PFO) and pseudo second-order (PSO) models [37] [69]. The PFO model is described by the following equation:

$$q_t = q_e(1-e^{-k1t}) \quad (2)$$

where $\mathbf{q_t}$ is the amount (mg/g^{-1}) of a target chemical adsorbed per unit mass of microplastics at time t, $\mathbf{q_e}$ the extrapolated concentration (mg/g^{-1}) at the equilibrium and $\mathbf{k_1}$ is the rate constant (1/day). The q_t concentration has been determined as:

$$q_t = \frac{(c_0 - c_t)V}{W} \quad (3)$$

where $\mathbf{c_0}$ and $\mathbf{c_t}$ are the initial concentration (mg/L) and the concentration (mg/L) at time t of a target chemical in liquid phase; **V** is the volume (L) of the solution and **W** is the mass (g) of microplastic [37].

The Pseudo-second-order (PSO) is expressed by the following equation:

$$\frac{t}{q_t} = \frac{1}{k_2\, q_e^2} + \frac{t}{q_e} \quad (4)$$

where $\mathbf{k_2}$ is the rate constant (g/mg*day) [37].

The experimental values and the fitting curves according to the PFO and PSO models are reported in Fig. 4. The kinetic parameters (equilibrium sorption capacity and kinetic constants) obtained from the two different models are reported in Table 1.

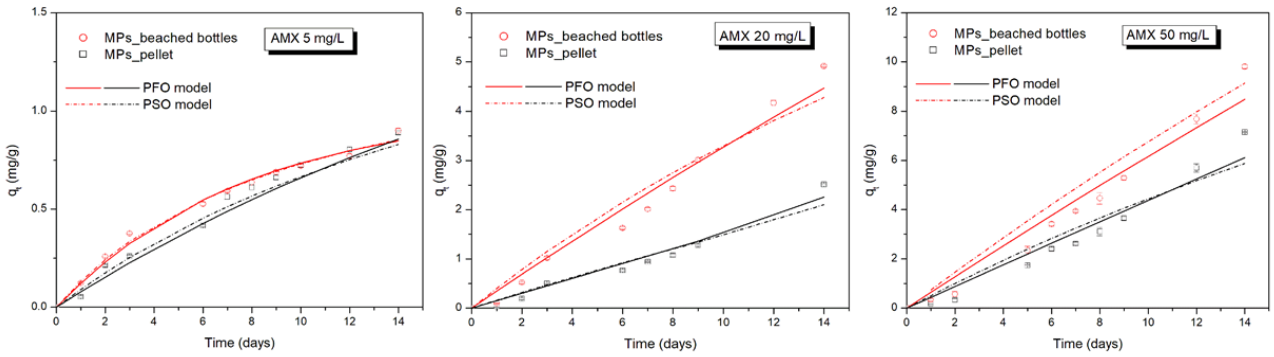

Fig. 4 Kinetic sorption tests at different AMX content (5 mg/L, 20 mg/L and 50 mg/L) fitted by pseudo-first order (PFO) and pseudo-second order (PSO) model.

Table 1 Adsorption kinetic parameters on the investigated PET microplastics.

	Average size (µm)	AMX concentration (mg/L)	q_e (mg/g)	k_1 (1/day)	R_{PFO}^2 (-)	q_e (mg/g)	k_2 (g/mg*day)	R_{PSO}^2 (-)
$MP_{\_beached\ bottles}$	154 ±25	5	1.016	$1.28*10^{-1}$	0.985	1.47	$6.66*10^{-2}$	0.983
		20	26.81	$1.30*10^{-2}$	0.968	16.53	$1.51*10^{-3}$	0.938
		50	71.70	$8.90*10^{-3}$	0.951	73.92	$1.40*10^{-4}$	0.918
$MP_{\_pellet}$	135 ±14	5	1.63	$5.75*10^{-2}$	0.983	2.04	$2.54*10^{-2}$	0.980
		20	39.38	$4.08*10^{-3}$	0.940	25.73	$2.51*10^{-4}$	0.930
		50	198.11	$2.24*10^{-3}$	0.944	152.51	$2.00*10^{-4}$	0.936

The sorption capacity of pollutants on microplastics obtained from aged bottles sampled from coastal environments has the same order of magnitude as the sorption of pollutants on primary microplastics derived from pellets. This proves that the study on the interactions between laboratory-made primary microplastics and pollutants is still meaningful. It can be also observed that the pseudo-first and second-order models allow obtaining a good prediction of the sorption behavior of amoxicillin on PET MPs with average size lower than 150 µm. These results are in agreement with those found by Godoy et al. [61] on almost 20 times bigger PET MPs (chips of 2.7 mm average size) obtained from the grinding of unaged PET bottles.

The values of the kinetic constants k_1 and k_2 indicate that beached derived MPs (from aged PET) adsorb faster than pellet derived ones, these latter being considered as primary MPs. Indeed, it has already been demonstrated that, compared to the primary microplastics, aged microplastics have higher sorption of typical antibiotics, such as sulfamethoxazole (SMX) and sulfamethazine (SMT) [70]. Furthermore, Li et al. [71] suggested that smaller microplastics showed higher sorption affinity to pollutants. Consequently, smaller aged microplastics could collect more pollutants so having severe effects on the environment.

The plots in Fig. 4 and the data reported in Table 1 evidence that both PFO and PSO models are suitable for the prediction of the adsorption experimental data of amoxicillin towards aged and primary PET microplastics. The R^2 value obtained with both models is, in fact very similar, even if the highest value obtained with the pseudo-first order (PFO) model suggest that the sorption of the studied antibiotic on the microplastic occurs mainly via physical processes.

In order to better understand this behavior, a second sorption run has been carried out on the same MPs. After reaching the 14 days of kinetic experiments, the MPs have been washed several times and then a second sorption run has been performed. As reported in Fig. 5, the sorption behavior of the second run is very similar to that obtained in the first run thus proving that the sorption-desorption phenomena are reversible, in the adopted test conditions. Moreover, it can be assumed that all binding sites for the AMX are easily accessible. This scenario supports once more the role of MPs as dangerous carriers of pollutants**.**

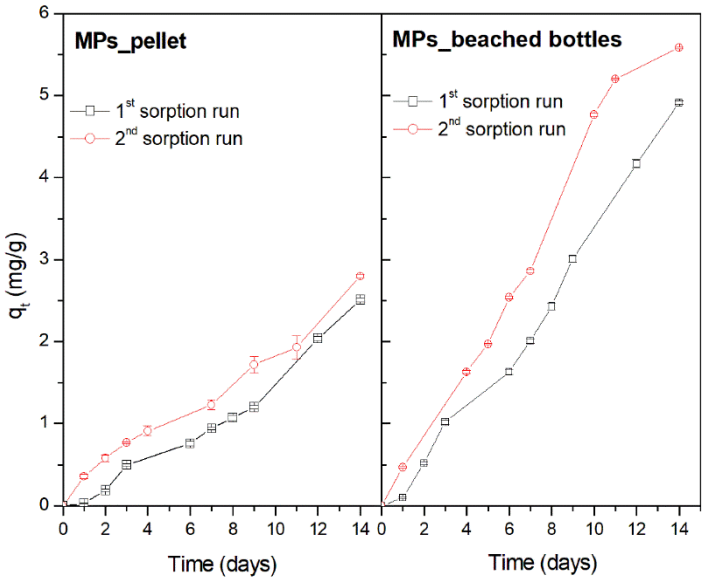

Fig. 5 First and second sorption run at AMX content of 20 mg/L.

The results of adsorption isothermal experiments at different pollutant concentration have been modeled with Langmuir, ~~and~~ Freundlich and Temkin models in order to get information about the adsorption capacity and interaction of AMX molecules with PET microplastic.

The Langmuir model is described by the following equation [72]:

$$\frac{1}{q_i} = \frac{1}{c_e q_m k_L} + \frac{1}{q_m} \quad (5)$$

where $\mathbf{q_i}$ represents the sorption capacity (mg/g) during an adsorption isotherm test at each external concentration c_e (mg/L) of AMX in aqueous phase, $\mathbf{q_m}$ the maximum amount of antibiotic per unit weigh of microplastic (mg/g) and $\mathbf{k_L}$ represents the Langmuir adsorption constant (L/mg). Langmuir isotherm modelling has been carried out assuming that the monolayer coverage of the antibiotic molecules on the PET MP is homogenous.

Langmuir isotherm is applicable to the homogeneous surfaces, which have equal adsorption affinity sites, while Freundlich model assumes heterogeneous adsorption sites [73,74]:

$$q_i = k_F c_e^{\frac{1}{n}} \quad (6)$$

where $\mathbf{q_i}$ represents the sorption capacity (mg/g) during an adsorption isotherm test at each external concentration c_e (mg/L) of AMX in aqueous phase, while $\mathbf{k_F}$ and **n** are adsorption capacity and adsorption intensity, respectively.

Temkin isotherm can be described by the following equation:

$$q_i = \frac{RT}{b} \ln k_T + \frac{RT}{b} \ln c_e \quad (7)$$

where **b** is Temkin constant (J/mol), **R** is the universal gas constant (8.314 J/mol K), **T** is the temperature (K) and $\mathbf{k_T}$ is Temkin isotherm constant (L/mg).

Table 2 reports the fitting of adsorption parameters for Langmuir, Freundlich and Temkin models.

Table 2 Adsorption isotherm parameters according to the fitting with Langmuir (subscript L) ~~and~~ Freundlich (subscript F) and Temkin (T) models.

	q_m (mg/g)	k_L (L/mg)	R^2 (-)	k_F (mg/g)	n (L/mg)	R^2 (-)	b (J/mol)	k_T (L/mg)	R^2 (-)
$MP_{\_beached\ bottles}$	29.16	$1.03*10^{-2}$	0.994	0.13	2.09	0.958	644.86	0.23	0.956
$MP_{\_pellet}$	28.38	$6.67*10^{-3}$	0.991	0.09	2.17	0.986	909.53	0.22	0.922

Fig. 6 reports the adsorption curves at different AMX concentrations fitted by Langmuir and Freundlich models for each kind of PET microplastic. Since the correlation coefficient R obtained with Temkin model was lower than those obtained from Langmuir and Freundlich models (see Table 2), the fitting with Temkin model is not reported in Fig. 6, for sake of clarity. The sorption capacity q_i of MPs from beached bottles is always higher than that of MPs from pellets since the first ones derive from PET which is likely more degraded due to the exposure to UV/oxygen and water leading to a higher specific surface area. Both the adsorption models fit well the experimental data but the Langmuir model fits better as reflected by the high value of correlation coefficient R^2 in Table 2, thus suggesting a monolayer adsorption of pollutants on the MP surface.

The maximum adsorption capacity q_m obtained from fitting with Langmuir model is slightly higher for MPs from beached bottles (29.16 mg/g) than that of MPs from pellets (28.38 mg/g), but the two values are quite close, thus confirming the role of PET microplastics as carriers of pollutants. Both the adsorption constants, k_L and k_F, of beached derived MPs are higher than the corresponding values of pellet derived MPs. This suggests a higher affinity of beached derived MPs to AMX in agreement with their higher measured specific surface area. The difference in the experimental results of adsorption tests, is a further demonstration of the reliability of the laboratory-made MPs to be applied for pollution studies.

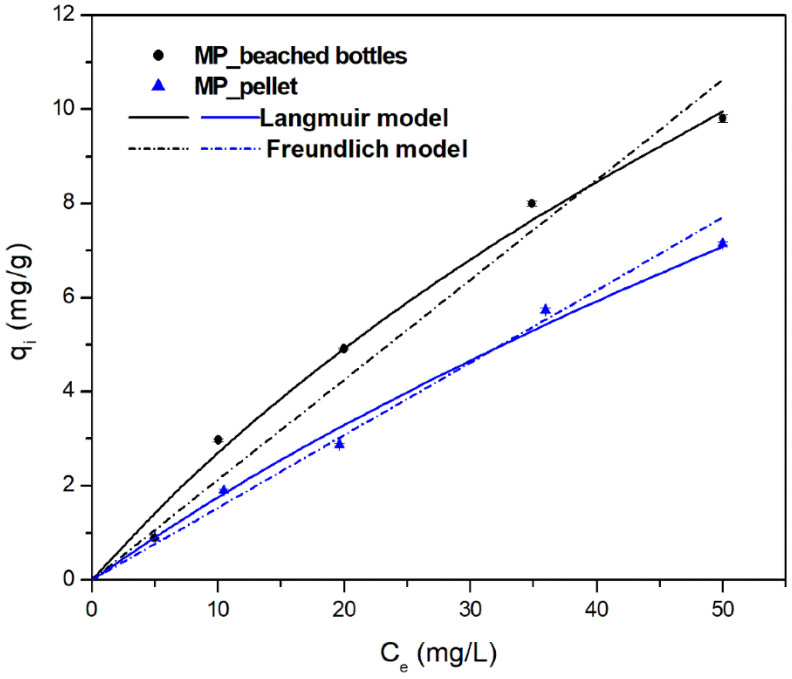

Fig. 6 Adsorption isotherms curves at different AMX concentration and fitting with Langmuir and Freundlich model.

5. Discussion

The difference in the sorption behavior between MPs from beached bottles and MPs from pellets could be explained by the eventual differences in the morphology of the two kinds of MPs and increased surface area of the MPs from beached bottles (see Table 3), due to the abrasion and photo-oxidative degradation experienced in the marine environment by the beached bottles-[75].

After the milling process, the obtained microplastics, both from pellets and beached bottles, present a similar size distribution, as reported in Fig. 7a. However, the MPs derived from beached bottles present a slightly higher size and wider size distribution, as also reported in Table 3. D_{10}, D_{50}, and D_{90} values indicate that 10%, 50% and 90% by volume, respectively, of the analyzed particles, are below the corresponding values.

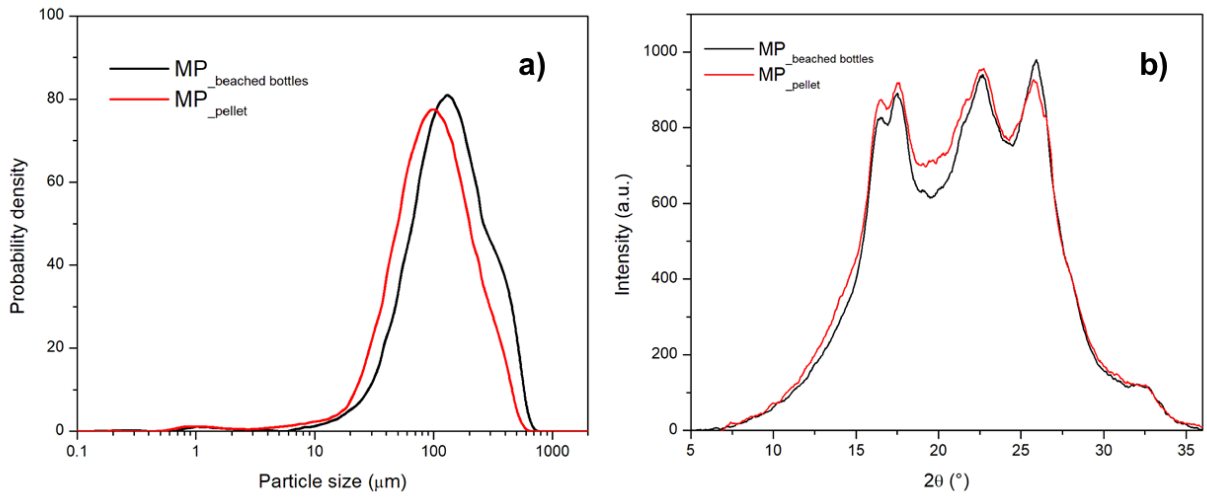

Fig. 7 a) Particle size distribution and b) X-Ray diffraction spectra of the obtained PET MPs.

Table 3 Results of MP characterization: several ~~diameters~~ dimensions (D) by laser diffraction; degree of crystallinity (X_{XRD}) by X-Ray diffraction; specific surface area (SSA) by BET analysis and circularity by SEM.

	$D_{average}$ (µm)	D_{10} (µm)	D_{50} (µm)	D_{90} (µm)	X_{XRD} (-)	SSA (m^2/g)	Circularity (-)
$MP_{\_beached\ bottles}$	154±25	44±4	123±25	319±43	0.17±0.01	0.14	0.70± 0.18
$MP_{\_pellet}$	135±14	37±0	103±8	287±41	0.16±0.01	0.11	0.76± 0.15

The XRD spectra in Fig. 7b show crystalline peaks at 2θ equal to 16.5°, 17.6°, 22.5° and 25.9°, corresponding to the $(0\bar{1}1)$), (010), $(\bar{1}10)$ and (100) planes, respectively [76]. The spectra of both MPs are very close with slight pronounced crystalline peaks for the MPs derived from beached bottles. The degree of crystallinity (X_{XRD}), reported in Table 3, has been determined from the integrated areas under the crystalline peaks $\mathbf{A_c}$ and broad amorphous halo $\mathbf{A_a}$ of the XRD spectra:

$$X_{XRD} = \frac{A_c}{A_c + A_a} \quad (8)$$

Both MPs have similar degree of crystallinity. The observed little increase of X_{XRD} in $MP_{\_beached\ bottles}$, which is however within the standard deviation, could be ascribed to the possible degradation of the amorphous regions of the beached bottles.

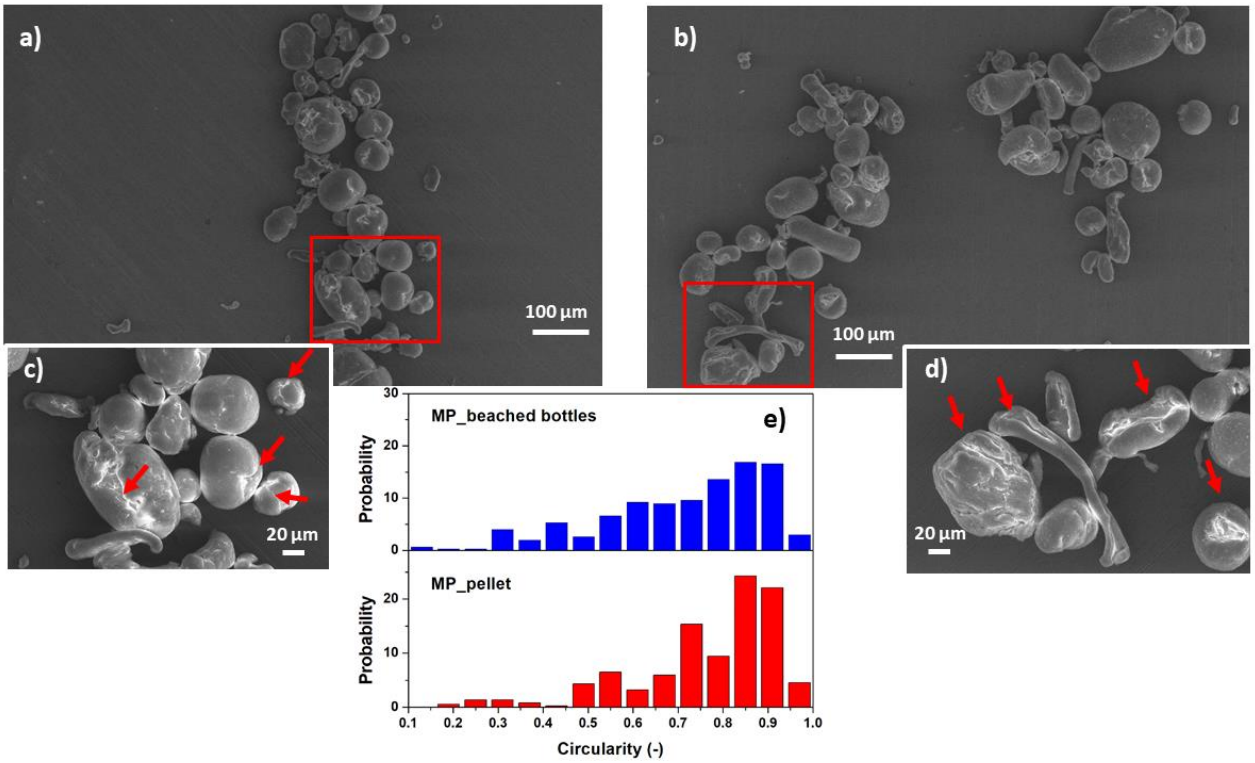

Fig. 8 SEM micrographs of the PET MPs obtained from pellet (a, c) and beached bottles (b, d) and e) results of the image analysis related to the circularity of the particles.

SEM analysis confirms that the morphology of the obtained MPs is quite similar. The microparticles present rough and uneven surface which resembles that of weathered MPs reported in the literature, for example by Liu et al. [77] for PS and PE weathered MPS. In the images at higher magnification, the presence of several cracks on the surface is evidenced by red arrows. The presence of cracks has been also reported for naturally weathered plastic surfaces [17]. The fragmentation process used for producing the MPs brings about some surface cracks on the MPs obtained by pellet without a significant difference form MPs obtained from beached bottles.

Some particles present severely damaged surfaces together with a significant change in size. In order to examine the particle shape in the overall population, an analysis of the circularity by means of ImageJ software has been carried out on a series of SEM images of both MPs, reported in Fig. 8 and in Table 3. The software calculates object circularity using the formula:

$$\text{Circularity} = 4\pi \frac{\text{area}}{\text{perimeter}^2} \quad (9)$$

A circularity value of 1 indicates a perfect circle. As observable also in Fig. 8, the particles are not perfectly spherical but elongated, which has been confirmed by the circularity values in Table 3, which are 0.70± 0.18 and 0.76± 0.15 for MPs from beached bottles and pellet, respectively.

The observed differences in size and shape between the aged and unaged seem to be not significant. Therefore, the higher adsorption capacity of MPs from weathered beached bottles compared to that of MPs derived from pellets can be ascribed to the higher specific surface area (SSA), obtained from BET measurements. This could be due to the degradation experienced by the beached bottles in the natural environment which leads to a higher amount of surface irregularities and microcracks, as also reported by Dabrowska et al. [78].

~~4.3. Thermodynamic affinity among microplastics and pollutants~~

To better understand the interactions responsible for the adsorption of amoxicillin on PET MPs, the affinity among microplastics and some pollutants among active principles of the most common drugs have been studied by evaluating Hansen's solubility parameters (HSPs), which account for dispersive, polarizable and hydrogen bonding contributions to the overall cohesive energy of a compound [79]. HSPs are widely used to predict the miscibility of polymers, the adsorption of pigments to surfaces, the surface wettability and the interaction between pharmaceuticals and polymers or excipients, etc. [79,80]. The affinity between amoxicillin and polyethylene terephthalate has been compared with that of other pharmaceutical and microplastic typologies. Eleven widely used Active Pharmaceutical Ingredients (APIs) have been considered: Aciclovir (antiviral), Levodopa (anti-Parkinson), Amoxicillin, Ciprofloxacin, Cyclosporine (three antibiotics), Piretanide, Diltiazem, Nifedipine (three antihypertensives), Oxprenolo (beta blocker), Terbinafine HCl (antifungal) and Ibuprofen (anti-inflammatory). Moreover, three polymers, largely diffused as microplastics (PS, PE, PET) have been selected.

Following the regular solution theory of Hildebrand, Hansen's solubility parameter (δ) is given as the sum of three contributions, accounting for three different interactions among molecules: non-polar (dispersion) interactions (**δ_d**), polar (dipole-dipole) interactions (**δ_p**) and hydrogen bonding interactions (**δ_h**) as given by Eq. (10)

$$\delta^2 = \delta_d^2 + \delta_p^2 + \delta_h^2 \quad (10)$$

In Table 4, the HSP and its contributions are reported for each considered pharmaceutical and polymer.

Table 4 Hansen's solubility parameters of the eleven selected APIs.

	δ_d	δ_p	δ_h	δ	Ref.
API	$(MPa)^{0.5}$	$(MPa)^{0.5}$	$(MPa)^{0.5}$		

				$(MPa)^{0.5}$	
Aciclovir	21.80	14.64	24.06	35.61	[81]
Levodopa	22.34	6.26	21.86	31.88	[81]
Amoxicillin	23.13	7.38	15.65	28.89	[81]
Ciprofloxacin	22.60	7.30	10.97	26.16	[81]
Cyclosporine	18.83	3.25	14.10	23.74	[81]
Piretanide	22.48	6.75	12.18	26.44	[81]
Diltiazem	20.44	4.86	8.40	22.62	[81]
Nifedipine	19.61	5.15	8.59	22.02	[81]
Oxprenolol	18.32	3.87	11.34	21.89	[81]
Terbinafine HCL	17.10	13.90	17.20	27.95	[81]
Ibuprofen	16.60	6.91	9.97	20.56	[82]
Polymer					
PET	19.10	6.31	9.10	22.96	[79]
PS	22.28	5.75	4.30	24.10	[79]
PE	16.51	2.70	6.13	18.00	[79]

In pharmaceutical science, the most commonly studied interactions are correlated to solubility between active pharmaceutical ingredients and polymers or excipients. For this purpose, different methods based on the solubility parameters have been developed [83]. One widely used method is Greenhalgh's approach [84], which uses the difference in the total solubility parameters (δ_t) of the substances to compare:

$$\Delta\delta_t = \left|\delta_1 - \delta_2\right| \tag{11}$$

where $\boldsymbol{\delta_1}$ and $\boldsymbol{\delta_2}$ are the solubility parameter of the polymer and the API, respectively.

The results of many combinations of active substances and excipients give a clear trend of solubility at $\Delta\delta_t$ < 7 $MPa^{0.5}$. This methodology has been applied to compare the affinities of the pharmacologically active molecules with the selected polymers. As can be seen from Fig. 9, six of the selected APIs are poorly soluble in PE, being their $\Delta\delta_t$ values higher than the limit value of 7 $MPa^{0.5}$ ~~outside the yellow area of Fig. 10~~, while affinity increases considerably for PET and PS, except for the case of levodopa and aciclovir. This trend can be explained by the high non-polar character of PE compared to PET and PS. Specifically, amoxicillin appears to have a good affinity with PET and PS, with a $\Delta\delta_t$ value equal to 5.9 and 4.8 $MPa^{0.5}$, respectively.

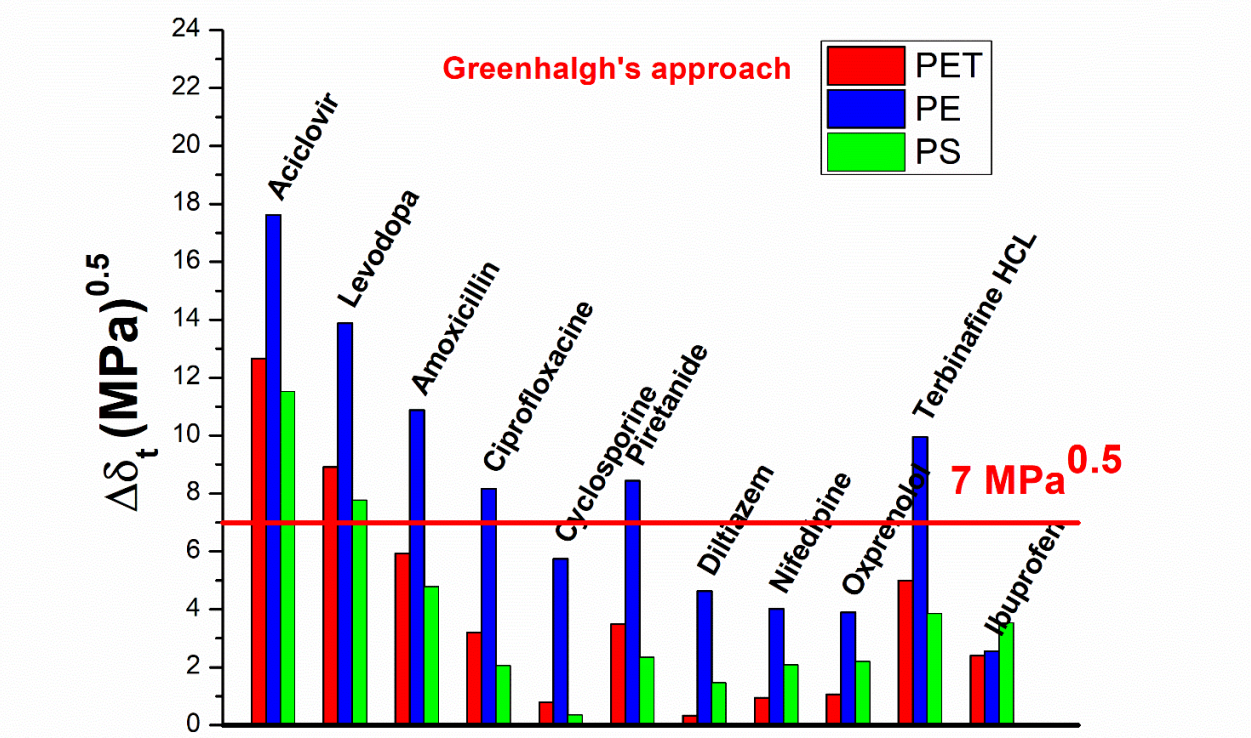

Fig. 9 Greenhalgh's approach applied to study the affinity of some pharmacologically active molecules with three polymers (PET: polyethylene terephthalate, PE: polyethylene, PS: polystyrene).

Another approach used to compare the miscibility of the examined compounds, has been reported by Khansary *et al.*[80]. In order to predict the solubility of compound 1 (API), in compound 2 (polymer), as the first step two values R_0 and R_1 are defined for each pharmacologically active molecule:

$$R_0 = \sqrt{\delta_{d1}^2 + \delta_{p1}^2} - \delta_{h1} \tag{12}$$

$$R_1 = \sqrt{\delta_{d1}^2 + \delta_{p1}^2} - \delta_{d1} \tag{13}$$

The polymer is highly compatible, poorly compatible or incompatible with API according to the following relationships:

$$|\delta_{p1} - \delta_{p2}| \leq R_0 \quad \text{highly compatible} \tag{14}$$

$$R_0 \leq |\delta_{p1} - \delta_{p2}| \leq R_1 \quad \text{poorly compatible} \tag{15}$$

$$|\delta_{p1} - \delta_{p2}| > R_1 \quad \text{incompatible} \tag{16}$$

$$|\delta_{h1} - \delta_{h2}| \leq R_0 \quad \text{highly compatible} \tag{17}$$

$$R_0 \leq |\delta_{h1} - \delta_{h2}| \leq R_1 \quad \text{poorly compatible} \tag{18}$$

$$|\delta_{h1} - \delta_{h2}| > R_1 \quad \text{incompatible} \tag{19}$$

Once the conditions illustrated have been verified, for each polymer related to each API, it is possible to represent the data in a graph (Fig. 10), in which the bars represent the range R_0 - R_1, the red dots the difference between the polar dispersion forces (δ_p) of the two components, and the black squares the difference between the hydrogen interaction forces (δ_h).

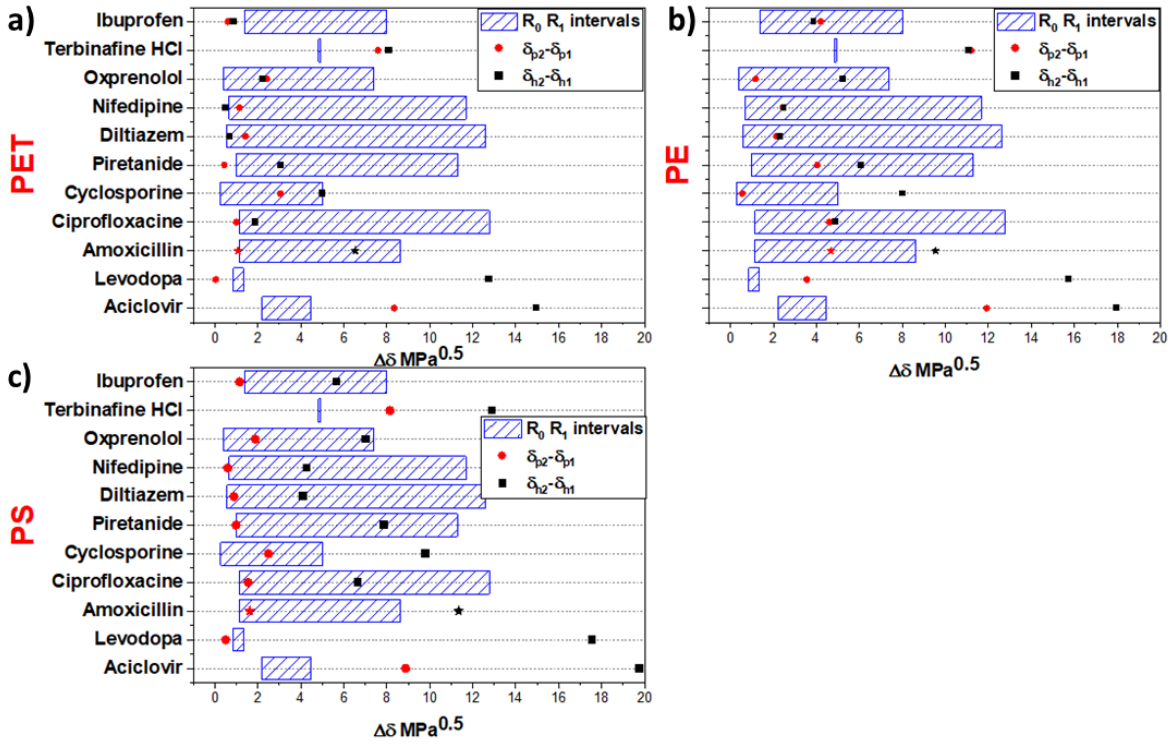

Fig. 10 Application of the Khansary's method to the selected APIs and polymers: a) polyethylene terephthalate, b) polyethylene, c) polystyrene.

From Fig. 10a it is clear how amoxicillin has a great affinity toward PET. As shown from this theoretical study, non-polar (δ_d) and dipolar (δ_p) interactions are of little importance for the adsorption of amoxicillin on PET; on the other hand, amoxicillin has a good interaction with PET, substantially based on its ability to generate hydrogen bond interactions with the ester groups of the polyethylene terephthalate monomers. To the best of the authors' knowledge, this is the first study on the predictive approach of affinity between AMX and PET. The results presented in Fig. 9 and Fig. 10 confirm the importance of studying the interactions of a worldwide present antibiotic as amoxicillin, with the ubiquitously present PET microplastic.

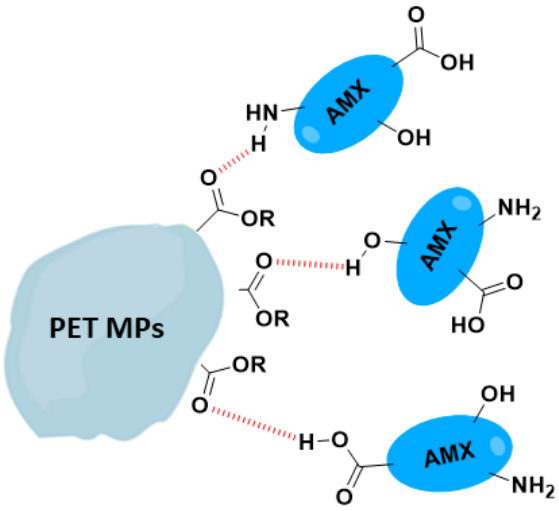

Fig. 11 Sketch of the possible interaction mechanism among amoxicillin molecules and PET MPs.

The affinity of AMX toward PET could be explained by the possible dipolar and hydrogen bond interactions between the ester functional groups of PET MPs and the hydroxyl and amino groups of the antibiotic molecule, as schematically reported in Fig. 11.

6. Conclusions

A top-down approach based on mechanical fragmentation has enabled has enabled the production of model PET microparticles from beached bottles with irregular shapes and surfaces by means of a process close to the mechanical abrasion of microplastics by small sand granules occurring in the aquatic environment. For comparison purposes, the same protocol has been applied to produce unweathered MPs from pellet. The two kinds of MPs from beached bottles and pellets do not present statistically significant differences in size, shape, crystallinity but only in the specific surface area which can be ascribed to the degradation undergone by the beached bottles in the marine environment. The present work demonstrates the importance of studying the interactions of amoxicillin, a worldwide present antibiotic, with the ubiquitously present PET microplastic. The results of the sorption tests have proved that PET microplastics act as carriers of pollutants in the aquatic environment. The interactions responsible for the adsorption of amoxicillin on PET MPs have been evaluated by studying the affinity between amoxicillin and PET through the Hansen's solubility parameters (HSPs) and comparing it with that of other pharmaceutical and microplastic typologies. Amoxicillin presents a good interaction with PET, substantially based on its ability to generate hydrogen bond interactions with the ester groups of the polyethylene terephthalate monomers. This study provides new insights into the mechanism of interactions between PET MPs and antibiotics and highlights the potential for MPs to accumulate and transport pollutants. Finally, the reliability and

relevance of the laboratory-made PET microplastics, produced in a fast and reproducible way, have been demonstrated. Therefore, the proposed approach to produce microplastics from beached bottles is a viable alternative for sorption studies of contaminants of microplastics.

Funding: Francesca Lionetto acknowledges Regione Puglia for funding REFIN—Research for Innovation project "NANOPLASTIC", project no. EF42B557, in the framework of POR PUGLIA FESR-FSE 2014/2020 projects.

Acknowledgement: Mr Donato Cannoletta is kindly acknowledged for the support in XRD and SEM measurements.

References

[1] Plastics - the facts 2019. Plast Eur 2019 n.d. https://www.plasticseurope.org/application/files/9715/7129/9584/FINAL_web_version_Plastics_the_facts2019_14102019.pdf (accessed December 5, 2020).

[2] Issac MN, Kandasubramanian B. Effect of microplastics in water and aquatic systems. Environ Sci Pollut Res 2021;28:19544–62.

[3] Gündogdu S, Rathod N, Hassoun A, Jamroz E, Kulawik P, Gokbulut C, et al. The impact of nano/micro-plastics toxicity on seafood quality and human health: facts and gaps. Crit Rev Food Sci Nutr 2022:1–19.

[4] Auta HS, Emenike CU, Fauziah SH. Distribution and importance of microplastics in the marine environment: a review of the sources, fate, effects, and potential solutions. Environ Int 2017;102:165–76.

[5] Thompson RC. Plastic debris in the marine environment: consequences and solutions. Mar Nat Conserv Eur 2006;193:107–15.

[6] Hidalgo-Ruz V, Gutow L, Thompson RC, Thiel M. Microplastics in the marine environment: a review of the methods used for identification and quantification. Environ Sci Technol 2012;46:3060–75.

[7] Cole M, Lindeque P, Halsband C, Galloway TS. Microplastics as contaminants in the marine environment: a review. Mar Pollut Bull 2011;62:2588–97.

[8] De Gisi S, Gadaleta G, Gorrasi G, La Mantia FP, Notarnicola M, Sorrentino A. The role of (bio) degradability on the management of petrochemical and bio-based plastic waste. J Environ Manage 2022;310:114769.

[9] Kida M, Koszelnik P. Investigation of the presence and possible migration from

microplastics of phthalic acid esters and polycyclic aromatic hydrocarbons. J Polym Environ 2021;29:599–611.

[10] Rodrigues MO, Abrantes N, Gonçalves FJM, Nogueira H, Marques JC, Gonçalves AMM. Spatial and temporal distribution of microplastics in water and sediments of a freshwater system (Antuã River, Portugal). Sci Total Environ 2018;633:1549–59.

[11] Nikpay M. Wastewater Fines Influence the Adsorption Behavior of Pollutants onto Microplastics. J Polym Environ 2022;30:776–83.

[12] Zhang S, Yang X, Gertsen H, Peters P, Salánki T, Geissen V. A simple method for the extraction and identification of light density microplastics from soil. Sci Total Environ 2018;616:1056–65.

[13] Ruggero F, Gori R, Lubello C. Methodologies for microplastics recovery and identification in heterogeneous solid matrices: a review. J Polym Environ 2020;28:739–48.

[14] Abbasi S, Keshavarzi B, Moore F, Turner A, Kelly FJ, Dominguez AO, et al. Distribution and potential health impacts of microplastics and microrubbers in air and street dusts from Asaluyeh County, Iran. Environ Pollut 2019;244:153–64.

[15] Kosuth M, Mason SA, Wattenberg E V. Anthropogenic contamination of tap water, beer, and sea salt. PLoS One 2018;13:e0194970.

[16] Eriksen M, Lebreton LCM, Carson HS, Thiel M, Moore CJ, Borerro JC, et al. Plastic pollution in the world's oceans: more than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. PLoS One 2014;9:e111913.

[17] Andrady AL. Microplastics in the marine environment. Mar Pollut Bull 2011;62:1596–605.

[18] Wagner M, Scherer C, Alvarez-Muñoz D, Brennholt N, Bourrain X, Buchinger S, et al. Microplastics in freshwater ecosystems: what we know and what we need to know. Environ Sci Eur 2014;26:1–9.

[19] Ajith N, Arumugam S, Parthasarathy S, Manupoori S, Janakiraman S. Global distribution of microplastics and its impact on marine environment—a review. Environ Sci Pollut Res 2020;27:25970–86.

[20] Prata JC, da Costa JP, Lopes I, Duarte AC, Rocha-Santos T. Environmental exposure to microplastics: An overview on possible human health effects. Sci Total Environ 2020;702:134455.

[21] Carr SA, Liu J, Tesoro AG. Transport and fate of microplastic particles in wastewater treatment plants. Water Res 2016;91:174–82.

[22] Bond T, Ferrandiz-Mas V, Felipe-Sotelo M, Van Sebille E. The occurrence and degradation of aquatic plastic litter based on polymer physicochemical properties: A review. Crit Rev

Environ Sci Technol 2018;48:685–722.

[23] Lionetto F, Corcione CE, Rizzo A, Maffezzoli A. Production and Characterization of Polyethylene Terephthalate Nanoparticles. Polymers (Basel) 2021;13:3745.

[24] Volpe V, Lanzillo MS, Molaro A, Affinita G, Pantani R. Characterization of Recycled/Virgin Polyethylene Terephthalate Composite Reinforced with Glass Fiber for Automotive Applications. J Compos Sci 2022;6:59.

[25] Saygin H, Baysal A. Insights into the Degradation Behavior of Submicroplastics by Klebsiella pneumoniae. J Polym Environ 2021;29:958–66.

[26] Burns EE, Boxall ABA. Microplastics in the aquatic environment: Evidence for or against adverse impacts and major knowledge gaps. Environ Toxicol Chem 2018;37:2776–96.

[27] Kooi M, Koelmans AA. Simplifying microplastic via continuous probability distributions for size, shape, and density. Environ Sci Technol Lett 2019;6:551–7.

[28] Murphy F, Ewins C, Carbonnier F, Quinn B. Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment. Environ Sci Technol 2016;50:5800–8.

[29] Long Z, Pan Z, Wang W, Ren J, Yu X, Lin L, et al. Microplastic abundance, characteristics, and removal in wastewater treatment plants in a coastal city of China. Water Res 2019;155:255–65.

[30] Wolff S, Kerpen J, Prediger J, Barkmann L, Müller L. Determination of the microplastics emission in the effluent of a municipal waste water treatment plant using Raman microspectroscopy. Water Res X 2019;2:100014.

[31] Schernewski G, Radtke H, Hauk R, Baresel C, Olshammar M, Osinski R, et al. Transport and behavior of microplastics emissions from urban sources in the Baltic Sea. Front Environ Sci 2020;8:579361.

[32] Nolasco ME, Lemos VAS, López G, Soares SA, Feitosa JPM, Araújo BS, et al. Morphology, Chemical Characterization and Sources of Microplastics in a Coastal City in the Equatorial Zone with Diverse Anthropogenic Activities (Fortaleza city, Brazil). J Polym Environ 2022:1–13.

[33] Leslie HA, van Velzen MJM, Brandsma SH, Vethaak D, Garcia-Vallejo JJ, Lamoree MH. Discovery and quantification of plastic particle pollution in human blood. Environ Int 2022:107199.

[34] Teuten EL, Rowland SJ, Galloway TS, Thompson RC. Potential for plastics to transport hydrophobic contaminants. Environ Sci Technol 2007;41:7759–64.

[35] Antunes JC, Frias JGL, Micaelo AC, Sobral P. Resin pellets from beaches of the Portuguese coast and adsorbed persistent organic pollutants. Estuar Coast Shelf Sci 2013;130:62–9.

[36] Tourinho PS, Kočí V, Loureiro S, van Gestel CAM. Partitioning of chemical contaminants to microplastics: Sorption mechanisms, environmental distribution and effects on toxicity and bioaccumulation. Environ Pollut 2019;252:1246–56.

[37] Lionetto F, Esposito Corcione C. An Overview of the Sorption Studies of Contaminants on Poly (Ethylene Terephthalate) Microplastics in the Marine Environment. J Mar Sci Eng 2021;9:445.

[38] Baysal A, Saygin H. Co-occurence of antibiotics and micro (nano) plastics: a systematic review between 2016-2021. J Environ Sci Heal Part A 2022:1–21.

[39] Wang F, Wong CS, Chen D, Lu X, Wang F, Zeng EY. Interaction of toxic chemicals with microplastics: a critical review. Water Res 2018;139:208–19.

[40] Teuten EL, Saquing JM, Knappe DRU, Barlaz MA, Jonsson S, Björn A, et al. Transport and release of chemicals from plastics to the environment and to wildlife. Philos Trans R Soc B Biol Sci 2009;364:2027–45.

[41] Ma Y, Huang A, Cao S, Sun F, Wang L, Guo H, et al. Effects of nanoplastics and microplastics on toxicity, bioaccumulation, and environmental fate of phenanthrene in fresh water. Environ Pollut 2016;219:166–73.

[42] Deng Y, Zhang Y, Qiao R, Bonilla MM, Yang X, Ren H, et al. Evidence that microplastics aggravate the toxicity of organophosphorus flame retardants in mice (Mus musculus). J Hazard Mater 2018;357:348–54.

[43] Polianciuc SI, Gurzău AE, Kiss B, Ştefan MG, Loghin F. Antibiotics in the environment: causes and consequences. Med Pharm Reports 2020;93:231.

[44] Watkinson AJ, Murby EJ, Kolpin DW, Costanzo SD. The occurrence of antibiotics in an urban watershed: from wastewater to drinking water. Sci Total Environ 2009;407:2711–23.

[45] Organization WH. WHO report on surveillance of antibiotic consumption: 2016-2018 early implementation 2018.

[46] Elizalde-Velázquez A, Gómez-Oliván LM, Galar-Martínez M, Islas-Flores H, Dublán-García O, SanJuan-Reyes N. Amoxicillin in the aquatic environment, its fate and environmental risk. Environ Heal Risk-Hazardous Factors to Living Species 2016;1:247–67.

[47] Kaur SP, Rao R, Nanda S. Amoxicillin: a broad spectrum antibiotic. Int J Pharm Pharm Sci 2011;3:30–7.

[48] Mahmodi Sheikh Sarmast Z, Sedaghat S, Derakhshi P, Azar PA. Facile Fabrication of Silver Nanoparticles Grafted with Fe3O4-Chitosan for Efficient Removal of Amoxicillin from Aqueous Solution: Application of Central Composite Design. J Polym Environ 2022;30:2990–3004.

[49] Spataro F, Ademollo N, Pescatore T, Rauseo J, Patrolecco L. Antibiotic residues and endocrine disrupting compounds in municipal wastewater treatment plants in Rome, Italy. Microchem J 2019;148:634–42.

[50] Mutiyar PK, Mittal AK. Occurrences and fate of an antibiotic amoxicillin in extended aeration-based sewage treatment plant in Delhi, India: a case study of emerging pollutant. Desalin Water Treat 2013;51:6158–64.

[51] Aydin E, Talinli I. Analysis, occurrence and fate of commonly used pharmaceuticals and hormones in the Buyukcekmece Watershed, Turkey. Chemosphere 2013;90:2004–12.

[52] Azanu D, Styrishave B, Darko G, Weisser JJ, Abaidoo RC. Occurrence and risk assessment of antibiotics in water and lettuce in Ghana. Sci Total Environ 2018;622:293–305.

[53] Bhagat C, Kumar M, Tyagi VK, Mohapatra PK. Proclivities for prevalence and treatment of antibiotics in the ambient water: a review. Npj Clean Water 2020;3:1–18.

[54] Khasawneh OFS, Palaniandy P. Occurrence and removal of pharmaceuticals in wastewater treatment plants. Process Saf Environ Prot 2021;150:532–56.

[55] Otoo BA, Amoabeng IA, Darko G, Borquaye LS. Antibiotic and analgesic residues in the environment–occurrence and ecological risk study from the Sunyani municipality, Ghana. Toxicol Reports 2022.

[56] Lee S, Kim C, Liu X, Lee S, Kho Y, Kim W-K, et al. Ecological risk assessment of amoxicillin, enrofloxacin, and neomycin: Are their current levels in the freshwater environment safe? Toxics 2021;9:196.

[57] Grenni P, Ancona V, Caracciolo AB. Ecological effects of antibiotics on natural ecosystems: A review. Microchem J 2018;136:25–39.

[58] Shi H, Ni J, Zheng T, Wang X, Wu C, Wang Q. Remediation of wastewater contaminated by antibiotics. A review. Environ Chem Lett 2020;18:345–60.

[59] Li J, Zhang K, Zhang H. Adsorption of antibiotics on microplastics. Environ Pollut 2018;237:460–7.

[60] Guo X, Pang J, Chen S, Jia H. Sorption properties of tylosin on four different microplastics. Chemosphere 2018;209:240–5.

[61] Godoy V, Martín-Lara MA, Calero M, Blázquez G. The relevance of interaction of chemicals/pollutants and microplastic samples as route for transporting contaminants. Process Saf Environ Prot 2020;138:312–23.

[62] Sarkar AK, Rubin AE, Zucker I. Engineered Polystyrene-Based Microplastics of High Environmental Relevance. Environ Sci Technol 2021;55:10491–501.

[63] Ducoli S, Federici S, Nicsanu R, Zendrini A, Marchesi C, Paolini L, et al. A different protein

corona cloaks “true-to-life” nanoplastics with respect to synthetic polystyrene nanobeads. Environ Sci Nano 2022;9:1414–26.

[64] Lionetto F, Lionetto MG, Mele C, Corcione CE, Bagheri S, Udayan G, et al. Autofluorescence of Model Polyethylene Terephthalate Nanoplastics for Cell Interaction Studies. Nanomaterials 2022;12:1560.

[65] Tippa DMR, Singh N. Development and validation of stability indicating HPLC method for simultaneous estimation of amoxicillin and clavulanic acid in injection. Am J Anal Chem 2010;1:95.

[66] Tewari A, Almuhtaram H, McKie MJ, Andrews RC. Microplastics for Use in Environmental Research. J Polym Environ 2022;30:4320–32.

[67] Li J, Huang X, Hou Z, Ding T. Sorption of diclofenac by polystyrene microplastics: Kinetics, isotherms and particle size effects. Chemosphere 2022;290:133311.

[68] Gozlan I, Rotstein A, Avisar D. Amoxicillin-degradation products formed under controlled environmental conditions: identification and determination in the aquatic environment. Chemosphere 2013;91:985–92.

[69] Ho YS, Wase DAJ, Forster CF. Removal of lead ions from aqueous solution using sphagnum moss peat as adsorbent. WATER SA-PRETORIA- 1996;22:219–24.

[70] Guo X, Chen C, Wang J. Sorption of sulfamethoxazole onto six types of microplastics. Chemosphere 2019;228:300–8.

[71] Li Y, Li M, Li Z, Yang L, Liu X. Effects of particle size and solution chemistry on Triclosan sorption on polystyrene microplastic. Chemosphere 2019;231:308–14.

[72] Langmuir I. The adsorption of gases on plane surfaces of glass, mica and platinum. J Am Chem Soc 1918;40:1361–403.

[73] Pathan S, Pandita N, Kishore N. Acid functionalized-nanoporous carbon/MnO2 composite for removal of arsenic from aqueous medium. Arab J Chem 2019;12:5200–11.

[74] Vadi M, Mansoorabad AO, Mohammadi M, Rostami N. Investigation of Langmuir, Freundlich and Temkin adsorption isotherm of tramadol by multi-wall carbon nanotube. Asian J Chem 2013;25:5467.

[75] Gewert B, Plassmann MM, MacLeod M. Pathways for degradation of plastic polymers floating in the marine environment. Environ Sci Process Impacts 2015;17:1513–21.

[76] Gates-Rector S, Blanton T. XRD JCPDS No. 50-2275. International Centre for Diffraction Data, 2000.

[77] Liu P, Qian L, Wang H, Zhan X, Lu K, Gu C, et al. New insights into the aging behavior of microplastics accelerated by advanced oxidation processes. Environ Sci Technol

2019;53:3579–88.

[78] Dąbrowska A, Gniadek M, Machowski P. The Proposal and Necessity of the Numerical Description of Nano-and Microplastics' Surfaces (Plastisphere). Polymers (Basel) 2021;13:2255.

[79] Hansen CM. Hansen solubility parameters: a user's handbook. CRC press; 2007.

[80] Khansary MA, Mellat M, Saadat SH, Fasihi-Ramandi M, Kamali M, Taheri RA. An enquiry on appropriate selection of polymers for preparation of polymeric nanosorbents and nanofiltration/ultrafiltration membranes for hormone micropollutants removal from water effluents. Chemosphere 2017;168:91–9.

[81] Breitkreutz J. Prediction of intestinal drug absorption properties by three-dimensional solubility parameters. Pharm Res 1998;15:1370–5.

[82] Kitak T, Dumičić A, Planinšek O, Šibanc R, Srčič S. Determination of solubility parameters of ibuprofen and ibuprofen lysinate. Molecules 2015;20:21549–68.

[83] Mosca Angelucci D, Tomei MC. Uptake/release of organic contaminants by microplastics: A critical review of influencing factors, mechanistic modeling, and thermodynamic prediction methods. Crit Rev Environ Sci Technol 2022;52:1356–400.

[84] Greenhalgh DJ, Williams AC, Timmins P, York P. Solubility parameters as predictors of miscibility in solid dispersions. J Pharm Sci 1999;88:1182–90.