



Review

Interaction of chemical contaminants with microplastics: Principles and perspectives



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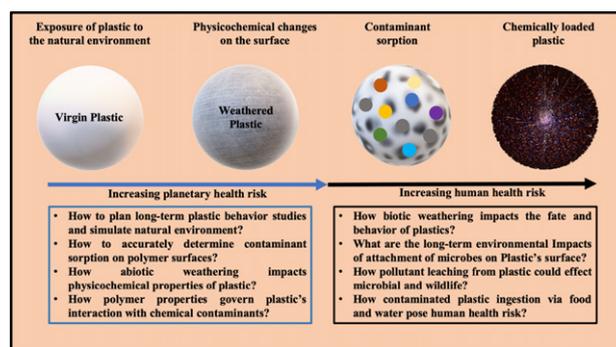
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HIGHLIGHTS

- Plastic additives are multiple stressors of the environment.
- PP-LFERS are more robust and accurate for sorption prediction.
- Sorption is a function of chemical contaminant and polymer properties.

GRAPHICAL ABSTRACT



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ABSTRACT

Scientific evidences abound of the occurrence of plastic pollution, from mega- to nano-sized plastics, in virtually all matrixes of the environment. Apart from the direct effects of plastics and microplastics pollution such as entanglement, inflammation of cells and gut blockage due to ingestion, plastics are also able to act as vectors of various chemical contaminants in the aquatic environment. This paper provides a review of the association of plastic additives with environmental microplastics, how the structure and composition of polymers influence sorption capacities and highlights some of the models that have been employed to interpret experimental data from recent sorption studies. The factors that influence the sorption of chemical contaminants such as the degree of crystallinity, surface weathering, and chemical properties of contaminants, and the implications of chemical sorption by plastics for the marine food web and human health are also discussed. It was however observed that most studies relied on pristine or artificially aged plastics rather than field plastic samples for studies on chemical sorption by plastics.

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1. Introduction

Plastic pollution and pervasive microplastic fragments have gained recognition globally as emerging environmental problems in freshwater and marine ecosystems. It is estimated that over 8 million tons of plastics are discharged annually into the global oceans as marine litter primarily from land-based activities (Jambeck et al., 2015; Erni-Cassola et al., 2019). The ubiquitous nature of plastics can be linked to their low cost of production, light weight, durability, mechanical strength, water and degradation-resistance, which makes it possible to use them extensively for industrial, commercial, and municipal purposes (Andrady, 2011). The global distribution of plastic debris in marine ecosystems driven by ocean currents and wind poses growing threats to marine fauna, human well-being, the environment, and the economy (Napper & Thompson, 2019). Furthermore, the severity of ecological impacts as well as risks associated with marine plastic pollution such as entanglements in fishing gear and ropes, and ingestion of floating and sunken plastics by marine and freshwater organisms at all trophic levels have been reported (Murphy et al., 2017; Nelms et al., 2018; Zarfl et al., 2011). According to the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection, plastic debris are categorized by size into mega- (>1 m), macro- (<1 m), meso- (<2.5 cm), micro- (<5 mm) and nano- (<0.1 µm) plastics (GESAMP, 2015; Thompson, 2016). However, in view of the growing research interest in plastic pollution and the differences in size, origin, colour, polymer type, and shape and structure categorizations posited by experts, a recent systematic gradation framework based on size nomenclature has been proposed and it categorizes marine plastics into: nano- (1–1000 nm), micro- (1–1000 µm), meso- (1–10 mm), and macroplastics (≥1 cm) (Hartmann et al., 2017; Erni-Cassola et al., 2019). Here, we describe microplastics as plastic particles that are <5 mm in length (NOAA, 2018). Microplastics (MPs) have been recognised as prevalent constituents of the marine and coastal environments, and the increasing attention on pollution by MPs are limited in size to those <5 mm (Arthur et al., 2009; Zhang et al., 2018b). This is due to the threat they pose largely associated with their composition, size, and their potential to adsorb, release and partition toxic and endocrine disrupting chemical substances in the marine ecosystems (Wang et al., 2015; Pérez-Lobato et al., 2016; Ramirez et al., 2019).

The ability of the MPs to sorb and accumulate both organic and inorganic contaminants is potentially detrimental as the sorbed contaminants can be released into organisms that ingest them and many studies have demonstrated this (Koelmans et al., 2016; Rochman et al., 2013a; Scopetani et al., 2018). Studies have reported the sorption of various contaminants to microplastics, such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), dioxin-like chemicals, polybrominated diphenyl ethers (PBDEs), toxic metals, hydrophilic organic compounds (ciprofloxacin), and pharmaceuticals (antibiotics and anti-depressants) (Brennecke et al., 2016; Li et al., 2018; Mato et al., 2001; Rios Mendoza and Jones, 2015; Wang and Wang, 2018a; Wang et al., 2018; Rochman et al., 2013a; Westerdahl et al., 2012). Usually, contaminants in water bodies partition between the aqueous phase and natural and anthropogenic particles that may be present, including microplastics. The determination of particle-bound contaminants and the advancement of the theoretical knowledge of the sorption behaviour of microplastics using laboratory studies is vital for comprehensive risk assessment and management (Wang et al., 2018). Furthermore, an understanding of the structure, the chemistry of sorption/desorption kinetics based on partition coefficients of contaminants improves our knowledge of the role of plastic fragments as vectors of contaminants in the fresh and marine waters. The aim of this paper therefore, is to provide an overview of plastics composition and how their physico-chemical properties influence their ability to sorb chemicals in the aquatic environment. Specifically, we (1) describe how the chemical make-up of plastics provide an insight to their toxic properties (2) highlight the factors that influence the sorption capacities of plastics (3) discuss some common models used in explaining sorption data (4) describe the implications of the sorption process to the marine food web and human health and (5) outline research priorities for improved ecosystem and human health risk assessment.

2. Plastic polymers and the role of chemical additives

Plastics are formed by the polymerisation of repeating units called monomers, including other substances such as chemical additives, usually by addition or condensation reactions (Lithner et al., 2011). Additives are chemicals added to polymers to modify their physical and

mechanical properties in order to make processing easy and derive improved performance. Additives change the structure of polymers and consequently the sorptive capacity is also affected (Endo et al., 2005). Plastic fragments surveyed from the marine environment can be essentially associated with two main types of contaminants (i) chemicals deliberately added during manufacturing process and (ii) chemicals sorbed by plastics from the surrounding media (Endo and Koelmans, 2016). However, due to variability in the characteristics of field-surveyed plastic fragments and methodological limitations, it is difficult to distinguish between contaminants (like PBDEs, Bis-phenol A or metals) that are additives and those sorbed from the surrounding water by marine plastics. Although the amount of contaminants that sorb to plastic materials depend on the level of contamination of the surrounding water body, plastic pellets have been shown to accumulate up to 10^6 orders of magnitude organic contaminants (nonylphenols) than in the ambient water column (Mato et al., 2001), suggesting that the excess contamination may have come from leached additives. Since polymers are rarely used in their pure form, most polymers produced for commercial use contain additives. Compounds such as BPA, phthalates and brominated flame retardants are identified as endocrine disruptors which are harmful to both biota and the environment (Hermabessiere et al., 2017). Additives do not usually form covalent bonds with polymers, therefore, they can easily leach from the plastics and enter into the surrounding media (Stringer and Johnston, 2001). There is a varied range of chemicals used to achieve the desired functionality of polymers. Table 1 presents some of the main polymer additives and their functions in plastic products.

2.1. Association of additives with microplastics

Studies focusing on the detection of plastic additives in field-collected MPs are limited (León et al., 2019; Jang et al., 2017; Net et al., 2015; Rani et al., 2015). However, the recovery of chemical additives from environmental plastic samples is an indication of their persistence and susceptibility to leaching. The release of additives will depend on (i) the pore size of polymer (ii) the size of additive molecule (iii) physico-chemical properties of additives and the surrounding media (Teuten et al., 2009). Luo et al. (2019), investigated the leaching of fluorescent additives from polyurethane (PU) microplastics and reported the release of 3,3'-diaminobenzidine-like substances. The authors concluded that 1.6 g/L of microplastics is sufficient to produce concentrations of leachate to exert inhibitory influence on microalgae cell

photosynthesis. The most common plastic additives include phthalates, bisphenol A (BPA), polybrominated diphenyl ethers (PBDE), nonylphenols (NP), and antioxidants (Hermabessiere et al., 2017), some of which have been detected in microplastics.

Phthalates are phenolic acid esters that possess a wide range of polarity. They are applied to plastics, especially PVC as plasticizers. The use of di(2-ethylhexyl) phthalate (DEHP), di-n-butyl phthalate (DnBP) and benzyl butyl phthalate (BBP) in excess of 0.1% by mass of plastic materials in toys and childcare products was prohibited by the European Parliament Directive 2005/84/EC (Fikarová et al., 2019). DEHP is commonly used in medical devices, cosmetics, personal care products (PCPs), furniture as plasticizer. DEHP and other phthalates have been classified as endocrine disruptors and have shown renal, reproductive, cardio, and neuro-toxicity in organisms (Rowdhwal and Chen, 2018). Phthalates were detected in concentrations up to 6.09 ng/g in beach microplastics collected from Bohai and Yellow sea of China. DEHP accounted for 64% of the total concentrations of phthalates; the report suggests that microplastics are a possible source of additives in the marine environment (Zhang et al., 2018b).

Bisphenol A is identified as an endocrine disrupting chemical that has estrogenic effects in aquatic organisms. BPA forms 65% and 30% by volume of the monomeric units of polycarbonate (PC) and epoxy resin plastics, respectively (Hermabessiere et al., 2017). BPA usually serves as a plasticizer or antioxidant in polymers like PE, PP, and PVC and like other additives, it may leach from the packaging of food and drinks (Rani et al., 2015). Limited studies are available on the interaction of BPA and its analogues (bisphenol F (BPF), bisphenol S (BPS), bisphenol B (BPB) and bisphenol AF (BPAF)) with microplastics. A laboratory study showed the adsorption of BPA to PVC microplastics up to a maximum concentration of 0.19 ± 0.02 mg/g. The maximum adsorption of BPA was attained at a value of 1.5 g/L of PVC hence, the value was used throughout the adsorption experiments (Wu et al., 2019).

Poly-brominated flame retardants are a class of 209 congeners of chemicals that function to reduce the rate of combustion and disperse fire by the release of bromine radicals. PBDEs occur as mixtures and are available commercially as pentaBDE, decaBDE and octaBDE. PBDEs are known to have carcinogenic and teratogenic properties as well as pose reproductive toxicity, liver toxicity, and neurotoxicity in mice (USEPA, 2017). The ability of microplastics to act as vectors of PBDE was demonstrated by Chua et al., (2014) in a study where *Allorchestes compressa* was exposed to PBDE-laden MPs. An uptake of higher brominated BDEs 154 and 153 and lower levels of BDEs 47 and 28 was observed. Rochman et al. (2014), found a positive correlation between the density of plastic debris and the concentration of BDEs 183–209 in the tissues of myctophid fish, which suggests that higher PBDEs that are used as flame retardants may signify plastic contamination in the marine environment. Also, the sorption of PBDEs to PE, PP, PS and PE microplastics was investigated with congeners under different experimental conditions including pH, temperature, salinity and organic matter; congeners BDE 47, 99 and 153 showed dominance in sorption capacity to PS above other polymer types (Xu et al., 2019). Plastic additives (PBDEs, NP, BPA and phthalates) were found present above detection limits in MPs collected from Swiss lakes (Faure et al., 2015).

Nonylphenol is a toxic xenobiotic chemical that is classified as an endocrine disruptor; it is persistent in the environment and is used as antioxidant and plasticizer in some plastics. Nonylphenol surfactants (including Nonylphenol ethoxylates (NPEs)) are also used in household cleaners, lawn care products, laundry detergents, latex paints, spermicides, and cosmetics. (USEPA, 2010). As a result of their uses in household cleaning products, nonylphenol and its surfactants are present in sewage waters, wastewater and soil in micromolar quantities. The ability of nonylphenol to leach from high density-PE, PET and PVC bottles into water was tested by Loyo-Rosales et al., (2004) and concentrations up to 180 and 300 ng/L were detected in water contained in HDPE and PVC, respectively. In another study, the leaching of nonylphenol surfactants from laboratory plastic ware such as pipette tips and microfuge

Table 1
Types of polymer additives and associated applications.

Type of additive	Examples	Purpose
Fillers and mineral reinforcements	CaCO ₃ , mica, talc	To increase bulk, stiffness, surface hardness.
Fiber props	Carbon, aramid, glass	Improve mechanical strength
Colourants	Pigments, liquid colours, dyestuff	Add colours, replacement of heavy metals.
Heat resistance	Anti-oxidants - Octylphenol, nonylphenol, Bis-Phenol A (BPA)	Delay/avoid oxidation of polymer when heated.
UV resistance	Oxanilides, benzophenones, benzotriazoles	Slow down/stop oxidation of plastic under prolonged sunlight exposure.
Flame retardants	Polybrominated diphenylethers (PBDEs), BPA	Prevent ignition of polymer
Anti-statics, conductive	Glycerol monostearate, Carbon black, conductive fibers, and nanomaterials	Enhance electrical conductivity and avoid electrostatic discharge.
Cross-linking, coupling	Styrene, peroxide, ZnO	Improve bonding between polymers and additives
Plasticizers	Phthalates	Improve ease of processing and flexibility (Ambrogio et al., 2016)

tubes which inhibited mitochondrial complex 1 was reported (Belaiche et al., 2009). Microplastics collected from field survey of Swiss surface waters also showed the detection of nonylphenol (Faure et al., 2015).

Antioxidants are used to inhibit the occurrence of aging in plastics by delaying oxidation. Antioxidant plastic additives have been shown to migrate from different polymer types to oil and food (Reinas et al., 2012). The transfer of the antioxidant plastic additives pentaerythrityl tetrakis(3,5-di-tert-butyl-4-hydroxyphenyl)propionate (Irganox 1010) and tris(2,4-di-tert-butylphenyl)phosphite (Irgafos 168) from polyolefinic packaging (polypropylene, high density polyethylene and ethylene propylene) into five oils, representing lipophilic media was investigated by Marcato et al. (2003). The leaching of the two antioxidants were reported to vary significantly with respect to polyolefin crystallinity, structure and variation in temperature. Nonylphenol and bisphenol A are also antioxidants that have been associated with microplastics. Supporting this reasoning, Marcato et al. (2003) demonstrated the leaching ability of antioxidant additives due to strong affinities to the aforementioned factors.

It is evident from various studies that both pristine plastics and environmentally collected plastics release additives into the surrounding media. Additives in plastics can therefore also be viewed as multiple stressors in the environment due to (i) leaching into the surrounding media posing potential risk (ii) long range transportation (iii) significant contribution to the persistence of plastic materials by reducing their ability to biodegrade.

3. Plastic sorption and the role of structure and composition

Sorption is the process of transfer of chemicals from a fluid phase like air or water, to a solid phase. The term "sorption" encompasses both adsorption and absorption. During absorption, the chemical contaminant molecules penetrate and become embedded within the matrix of the solid phase while molecules of the chemical contaminant remain on the interface between the fluid and the solid phases in an adsorption process. Adsorption to the solid surface may include interactions by ionic, van der Waals, steric or covalent bonds while absorption occurs by the partitioning of the sorbate molecules into the sorbent matrix and are held by weak van der Waals forces. This process strongly depends on the hydrophobic properties of the sorbate chemical, the properties of the solid phase and the surface to volume ratio of the solid (Endo and Koelmans, 2016). Surface polarity of microplastics determine how strongly an organic contaminant interacts; hydrophobic organic contaminants (HOCs) preferably adhere to non-polar surfaces (Mato et al., 2002). Usually, the resulting partition coefficient (K_{pw}) between water and plastic is related to the octanol-water partition ratios (K_{ow}) (O'Connor et al., 2016). When concentration of sorbate is low, the partitioning of the organic chemical between the fluid phase and the solid matrix is high, yielding elevated partition ratios, due to stronger forces of interaction at the surface. Therefore, low concentration of sorbate favours adsorption and at high concentration of the sorbate, absorption is more likely due to larger volume (Cornelissen et al., 2005; Hartmann et al., 2017). Sorption can also be considered in terms of chemical (chemisorption) and physical (physisorption) association of sorbate with the sorbent. During chemisorption, the chemicals interact with the solid phase by forming covalent bonds; physisorption on the hand, requires association by non-covalent van der Waals forces. Chemisorption can however be irreversible except when the covalent bonds are broken, making desorption of the chemicals from the solid phase difficult (Endo and Koelmans, 2016). Hence, sorption phenomena leading to desorption of contaminants in the tissue of marine organisms is largely by physisorption and the most significant interactions in the sorption of organic chemicals to plastics are van der Waals forces, hydrogen bonding and cavity formation (Goss and Schwarzenbach, 2003).

Carbon is the most critical element in polymers. It has four valence electrons and achieves stability by sharing four more electrons, forming a wide range of covalent bonds. Most notably, carbon catenates and

form strong bonds with itself. The functional groups on a polymer, the presence of unsaturated bonds and hydrophobicity are relevant in the determination of the type(s) and strength of its secondary bonds. For example, PTFE (Teflon) has a structure similar to polyethylene (PE) except that fluorine atoms have replaced all the hydrogen atoms. This confers a nonpolar character on the polymer, without any exposed positive part of the molecule to interact with the negative part of a neighbouring molecule. This effect was corroborated by a study that investigated the sorption of PAHs to PE, PP, PTFE and stainless steel. PTFE and stainless steel were reported to have better recovery rates than PE and PP. It was concluded that loss of analyte is lower when PTFE and stainless steel containers are used in sorption experiments (Krüger et al., 2014). Planar molecules like PAHs typically have higher sorption coefficients than non-planar molecules of similar hydrophobicity (Velzeboer et al., 2014). DLCs are coplanar molecules, hence they have comparatively higher sorption capacities toward plastics (Chen et al., 2019). In humans, toxic effects due to dioxin-like PAHs and PCBs arise from aryl hydrocarbon receptors which bind coplanar molecules with high sorption affinities and transport them into the nucleus (Eichbaum et al., 2014). The character of the pi-electrons in PS is also expected to affect its sorption capacity (Goedecke et al., 2017). The planarity of a molecule affects how close it can move toward the particle surface, thus facilitating adsorption (Velzeboer et al., 2014).

4. Factors influencing the sorption of organic chemicals by microplastics

4.1. Plastic polymer type

The properties of plastics, such as surface charge, surface area, molecular chain arrangement (Fig. 1), functional groups present, and the acid-base character, influence the sorption of chemicals (Fotopoulou and Karapanagioti, 2012). Additionally, in case of polymer (e.g. nanocomposites or hybrids) that exclusively possess groups that form complexes with the contaminants e.g. metals, the chain structure could prove crucial while understanding the role of planarity on adsorption (Rivas et al., 2018). A measurement of the surface area of plastic types used by Seidensticker et al. (2018), showed PE to be non-porous while PS is mesoporous having a pore size of about 195 Angstrom. This observation resulted in greater sorption of contaminants to PS than PE which is confirmed by a pore-filling mechanism depicted by non-linear sorption isotherms in the study. However, some other studies reported that PE microplastics had higher sorption capacities for hydrophobic organic compounds (HOCs) than polyvinyl chloride (PVC) and polystyrene (PS) particles (Wang and Wang, 2018b; Zuo et al., 2019). The authors attributed their finding to the arrangement of the plastic's molecular chain. The efficiency of sorption also depends on the molecular structure and the composition of microplastics (Wang et al., 2015). A similar occurrence was reported by Lee et al. (2014), where the authors observed that the concentrations of PAHs and PCBs and the partitioning coefficients between the different types of microplastics in seawater followed the order of PS > PE > PP. In another study, PS acted as a sink for PAHs and had 8–200 times order of magnitude sorption for PAHs than PET, PVC, PE and PP (Rochman et al., 2013b). The high sorption capacity of PS has also been related to its amorphous structure. Although sorption capacities vary with polymer types, there appears to be other factors that determine sorption levels.

4.2. Crystallinity

Polymers can be categorized into crystalline, semi-crystalline and amorphous phases, depending on the extent of the alignment of their molecular chains. The degree of crystallinity of a polymer is the fraction of the polymer that is crystalline and is expressed as a mass fraction or

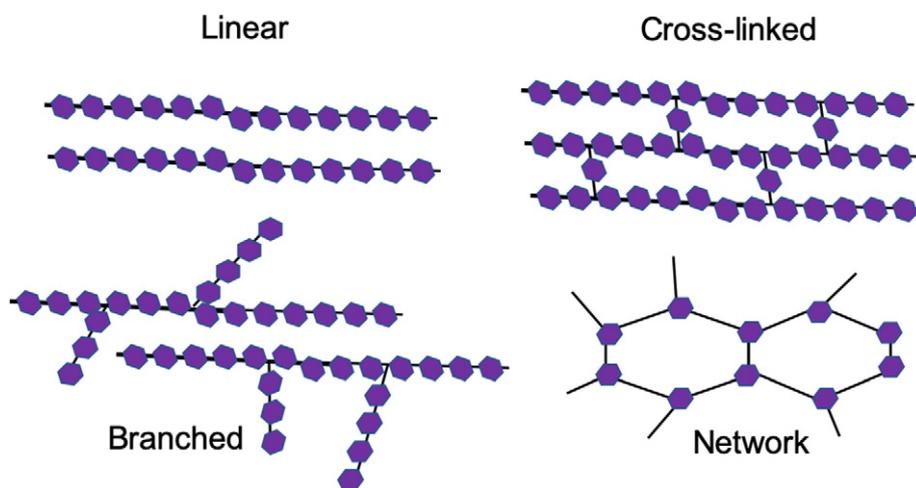


Fig. 1. Polymer chain types.

the volume fraction (Joshi et al., 2017). The degree of crystallinity usually ranges from a few percentage points to about 90%. X-ray diffraction (XRD) is a standard analytical method for the determination of polymer crystallinity. Some examples of semi-crystalline polymers are PE, PP, PTFE, PET and polyacrylonitrile (PAN). The presence of straight chains with regularly spaced side groups promotes crystallization in polymer molecules. Increased crystallinity in pristine plastics was observed to reduce adsorption of ciprofloxacin (G. Liu et al., 2019b). Amorphous polymers, on the other hand, are characterized by chains that are incapable of forming a rigid arrangement. In the solid state, they possess a short-range order of repeating units. These polymers form an amorphous glassy solid in which the molecular chains are randomly arranged. Poly (methyl methacrylate) and polycarbonate are typical examples of amorphous polymers. Fig. 2 is a pictorial representation of the regions present in a semi-crystalline polymer.

Some of the factors that affect polymer crystallinity include polymer complexity, chain configuration, isomerism, and rate of cooling during solidification. Another important factor to consider is the glass transition property which is only associated with the amorphous domains of a polymer. At low temperature, the amorphous regions of a polymer are in a glassy state so that the molecules only vibrate because they are rigid and frozen. When heat is applied, the polymer will reach a state where it changes from being glassy to a rubbery state. The temperature at which the change occurs is the glass transition temperature (T_g). The glass transition temperatures of commonly detected plastic polymers in the environment is presented in Table 2. The rubbery segments are

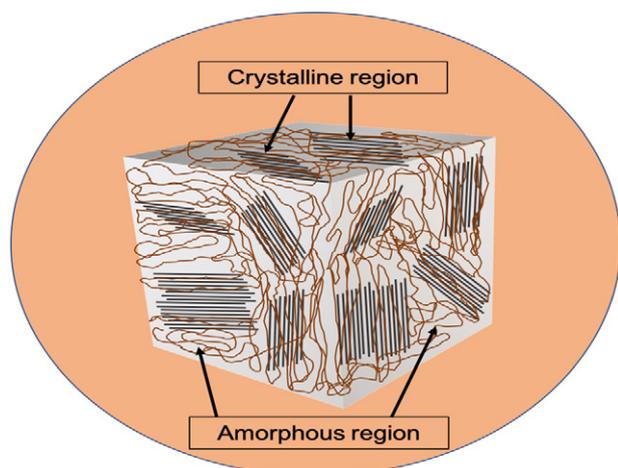


Fig. 2. Structure showing crystalline and amorphous portions of a polymer.

highly flexible because molecules have higher freedom to move hence absorption of organic chemicals in this region is enhanced. The sorption isotherm of a rubbery polymer is essentially linear and the absorption process is reversible and non-competitive. Conversely, glassy polymers undergo adsorption, showing competition with any coexisting chemicals and their isotherms are nonlinear (Endo and Koelmans, 2016). The crystalline region within a polymer does not favour the sorption of chemicals due to the high energy that is required to destabilize the strongly ordered polymer chains. However, the amorphous portion of the polymer which has randomly oriented polymeric chains are more available for greater sorption of chemical compounds. This is also corroborated by Guo et al. (2012), where it was reported that the sorption of HOCs was enhanced as the mobile and rubbery domains on the polymers increased. This implies that crystallinity has a profound influence on the sorption capacities of HOCs by plastics.

4.3. Size of the plastic particles

From the knowledge of conventional risk assessment in particulates, decreased particle size increases surface area to volume ratio (Endo and Koelmans, 2016) and this is an important parameter in the determination of the extent of risk. Cryo-milling of polypropylene particles to 0.2–0.6 mm to increase the surface area enhanced the sorption capacity of metformin to the microplastics (Goedecke et al., 2017). During the sorption studies of 3,6 dibromocarbazole and 1,3,6,8 tetrabromocarbazole to polypropylene microplastics, Zhang et al. (2019) observed an increased sorption capacity of the polypropylene particles as particle size decreased between 0.45 and 5 mm. Conversely, Chen et al. (2019), reported that the differences in concentration of

Table 2
Glass transition temperatures (T_g) and densities of selected polymers.

Polymer	T_g ($^{\circ}\text{C}$)	Density (g/cm^3)
High density polyethylene (HDPE)	–125	0.94–0.97
Low density polyethylene (LDPE)	–130	0.91–0.93
Polypropylene (atactic)	–20 to –5	0.85–0.94
Polyethylene terephthalate (PET)	70 to 80	1.34–1.39
Polytetrafluoroethylene (PTFE)	120 to 130	2.2
Polyvinyl alcohol	80 to 90	1.19
Polyvinyl chloride (PVC)	65 to 85	1.16–1.20
Polypropylene (isotactic)	100	0.92
Polystyrene (PS)	90 to 110	1.04–1.09
Polyurethane (PU)	120 to 160	1.2
Poly(methyl methacrylate)	85 to 105	1.18
Polyacrylamide	160 to 170	1.11
Polyamide (Nylon 6,6)	50 to 60	1.13–1.15

Data adapted from Perkinelmer (2019) and Sundt et al. (2014).

PAHs in small, medium and large plastic particle sizes collected from the open ocean were insignificant. They concluded that pollutant concentrations versus particle size did not occur, which they attributed to the partition sorption mechanisms of PAHs and PCBs. They suggested that the insignificant contribution of particle size effects might become more evident if the particle size used in their study had been smaller. It can then be implied that particle size plays a significant role where sorption of chemical contaminants is concerned. Although, the effects of nano-scale particles are yet to be quantified, the size effect is a matter of concern especially through ambient air.

4.4. pH

Freshwater aquatic life requires a pH range of 6.5–9.0 for existence, and the seawater pH typically ranges between 8.08 and 8.33 (Marion et al., 2011). In aqueous systems, pH is a critical parameter as it affects the chemical and biological reactions and the equilibrium of such systems. Some studies have investigated the influence of pH on the strength of sorption of contaminants to plastics. (F. Liu et al., 2019a), found no significant influence of pH in the sorption of phthalate esters to PE, PS and PVC due to high electrostatic repulsion; as pH increased there was a reduction of hydrophobic interactions between the charged sorbate and sorbent. This was also corroborated by another study where there was a limited effect of variable pH conditions observed during the sorption experiment of sulfamethoxazole on PE (Xu et al., 2018). This was ascribed to electrostatic repulsion of surface charges. At pH above 8, both PE and sulfamethoxazole surfaces had negative charges. However, pH was found to enhance the adsorption of difenoconazole on PP but not on PA and PS. Liu et al. (2019b), reported differing sorption behaviours of aged and pristine PVC and PS in response to pH variation. For PVC, sorption efficiency increased with pH except at pH > 9, while for PS, optimum sorption was achieved for aged and pristine PS at pH 8 and 5, respectively. The sorption of oxytetracycline (an antibiotic compound) on pristine PS was unaffected by changes in pH, but the aged PS had maximum adsorption at pH 5. The authors reported that at pH 5, electrostatic repulsion between the microplastic surface and the oxytetracycline zwitterion was minimal, which is an indication that electrostatic interaction may have a regulatory role in the adsorption process. The role pH can be explained as follows (i) electrostatic repulsion will increase as solution pH increases, suppressing electrostatic interaction between different charged sorbate and sorbent; (ii) an increase in pH may promote the dissociation of the hydrophobic neutral sorbate molecules into hydrophilic, negatively charged species, thereby resulting in reduced hydrophobic interaction; (iii) a high pH can increase π -donor ability of sorbate, thus enhancing π - π interactions (Liu et al., 2019a).

4.5. Salinity

The effect of salinity is dependent by the extent of electrostatic interaction or ion-exchange mechanism of the sorption process. The sorption capacity of virgin microplastics for ciprofloxacin was found to decrease as the degree of salinity increased; salinity significantly lowered adsorption on all the tested microplastics (Liu et al., 2019b). Enhanced salinity had a weak positive effect on the adsorption of difenoconazole on virgin polyamide (PA) and PP but slowed down adsorption in PS (Goedecke et al., 2017). The impact of salinity on the sorption of carbazole on PP was observed to vary in a study by Zhang et al. (2019). The adsorption capacity and rate initially decreased with increasing salinity but later increased and reached a maximum at about 14% salinity of the solution. Yet, salinity range from 0.05% to 3.5% for sulfamethazole sorption showed no significant effects, suggesting that electrostatic interactions had limited influence in the process (Xu et al., 2018). P. Xu et al., (2019) also reported no significant effects of enhanced salinity in the sorption of PBDEs to four microplastics (PE, PP, PA and PS). It appears that sorption by MPs may be contaminant-specific and MPs can sorb

contaminants irrespective of the level of salinity of the surrounding media.

4.6. Ionic strength

Ionic strength is the total ion concentration in solution. It is a property governing the shielding of charges in solution. Natural water bodies generally contain some level of dissolved salts which in turn translate to their ionic strength. The effect of ionic strength due to the presence of NaCl, and CaCl₂ in solution during the sorption of oxytetracycline to PS foams was investigated by Zhang et al. (2018a). It was found that oxytetracycline sorption to both virgin and aged PS decreased as ionic strength increased. Oxytetracycline sorption to the PS microplastics was stronger in the presence of CaCl₂ than NaCl, suggesting that sorption of the antibiotic is enhanced by the presence of multivalent cations like Ca²⁺ through the formation of ternary complexes between the antibiotic, the cations and the surface functional groups (MacKay and Canterbury, 2005). In another study, the adsorption of tylosin on PE, PP, PS and polyvinylchloride (PVC) initially increased in all microplastic types and then decreased as the ionic strength increased (Guo et al., 2018). It was attributed to increased competition for sorption sites as the concentration of K⁺ ions increased. Ionic strength is also a crucial factor for the sorption of metals like lead, copper, uranium and cadmium in the aquatic ecosystem (Guo et al., 2018; Mayer et al., 2014; Wang et al., 2013). Sorption process had been found to decrease by 50% as the concentration of Na⁺ ions increased from 25 to 300 mg/L (Xiao et al., 2017). Competition for binding sites between ions in solution and contaminants appears to be the determinant factor for the effects of ionic strength in sorption.

4.7. Age and degree of weathering of plastics

Pristine or virgin plastics are those which are manufactured newly and are yet to undergo any abrasion in the environment. Aged or weathered plastics on the other hand, are those that have been exposed to thermal, mechanical, biological, radiative and oxidative pressures which have caused their surfaces to be altered (Albertsson et al., 1987). The alteration of plastic surfaces has been demonstrated to enhance the sorption of metals and organic compounds. Aging enhanced the concentration of some PAH congeners on microplastics (Chen et al., 2019). In a sorption study by Liu et al. (2019), the interaction of ciprofloxacin (a hydrophilic organic compound) on UV-accelerated aged polystyrene and polyvinylchloride showed an elevated sorption capacity than in pristine plastic materials. The authors attributed this to the formation of oxygen-containing functional groups on the plastic surface, and the occurrence of light-induced surface oxidation of the plastics. Peaks characteristic of the hydroxyl group were observed in the aged PS and PVC but were absent in virgin plastics. Results from Hüffer and Hofmann (2016) also suggests that aging reduces hydrophobicity of plastics thereby increasing their sorption capacity for hydrophilic contaminants. Goedecke et al. (2017) studied the interaction of a polar drug (metformin) and a non-polar fungicide (difenoconazole) on virgin MPs of polyamide (PA), polypropylene (PP) and polystyrene (PS). It was found that agitation played a significant role in the level of adsorption of difenoconazole to the plastic particles. Artificial aging in PA and increased surface area in PP enhanced the adsorption of the contaminants. The non-polar compound was found to sorb more to the MPs while the polar compound hardly interacted with the MPs. These evidences suggest that weathered or aged plastics are potentially more likely to be toxic than “unweathered” or pristine plastics of the same polymer type due to higher sorption of contaminants.

4.8. Chemical properties of contaminant

Most organic contaminants of environmental concern occur as non-polar species with limited water solubility making them hydrophobic in

Table 3
Recent studies of chemical sorption by microplastics and reported results.

MP type and size	Chemical sorbate	Instrumentation	Results highlight	Reference
Bio-degradable PBAT, PE, PS (2338 ± 486, 400 and 250 µm, respectively)	Phenanthrene (Phe)	Thermo gravimetric analyzer (TGA), Fourier transform infra-red (FTIR), XRD, DSC, pyrolysis GC-MS.	Bio-degradable PBAT sorbed 3 times and 40 times more Phe than PE and PS.	Zuo et al. (2019)
UV-accelerated PS and PVC (75 µm)	Ciprofloxacin (CIP)	XRD, FTIR, UV-VIS spectrophotometer	UV-induced aging increased sorption of CIP by 123.3% and 20.4% to PS and PVC, respectively.	Liu et al. (2019)
Pristine and aged PS (125–250 µm)	Organic compounds (not named)	Scanning electron microscope (SEM), ATR-FTIR, GC-MS	UV-induced aging decreased sorption of sorbate to PS MPs.	Hüffer et al. (2018)
Beached and virgin PS (0.45–1 mm)	Oxytetracycline	ATR-FTIR, Elemental analyzer, potentiometric titration, HPLC	Adsorption of sorbate on beached MPs was 2 times that of virgin MPs.	Zhang et al. (2018a)
HDPE, PS and PVC (100–150 µm)	Pyrene	SEM, MP surface analyzer	Affinity for pyrene followed the order PE > PS > PVC	Wang and Wang (2018a)
PE (45–48 µm)	Sulfamethoxazole (SMX), propranolol (PRP), sertraline (SER)	SEM, UPLC/MS/MS	Sorption order on PE was SER > PRP > SMX	Razanajatovo et al. (2018)
PE, PS, PP, PA and PVC (75–180 µm)	Sulfadiazine (SDZ), amoxicillin (AMX), tetracycline (TC), ciprofloxacin (CIP), trimethoprim (TMP)	SEM, XRD, HPLC	On PS, PE, PP and PVC, sorption decreased in order CIP > AMX > TMP > SDZ > TC. PA had high affinity for AMX, TC and CIP in freshwater.	Li et al. (2018)
PE (150 µm)	Sulfamethoxazole	SEM, electro kinetic analyzer, HPLC	Negligible effects of pH, salinity and dissolved organic matter (DOM) on sorption capacity.	Xu et al. (2018)
PE, PP, PS and PVC (<200 mesh)	Tylosin	SEM, Brunner-Emmet-Teller (BET), FTIR, Zeta potential	Sorption of tylosin increased in the order PE < PP < PS < PVC	Guo et al. (2018)
PE, PS and PVC (100–150 µm)	Phenanthrene and pyrene	BET, SEM. GC-MS	MPs affinity for Phe followed the order PE > PS > PVC but the sorption level decreased in the presence of pyrene.	Wang and Wang (2018b)
Virgin PA, PP and PS (3–5 mm)	Metformin and Difenconazole	Dynamic Scanning Calorimetry (DSC), Gel Permeation Chromatography (GPC), FTIR, LC-DAD, GC-MS	Difenconazole adsorption followed the order PS > PP > PA while no adsorption occurred for metformin on all MP pellets	Goedecke et al. (2017)
PP (0.18–0.425, 0.425–0.85, 0.85–2, 2–5 mm)	3,3',4,4' tetrachloro biphenyl (PCB 77)	GC-ECD	Particle size, temperature and solution environment had significant impact on sorption behaviour.	Zhan et al. (2016)
PE, PA, PS and PVC	n-hexane, cyclohexane, benzene, toluene, chlorobenzene, ethyl benzene and naphthalene	Particle size and shape analyzer, GC-MS	Sorption increased in the order PA < PE < PVC < PS	Hüffer and Hofmann (2016)

nature. The sorption of HOCs by microplastics is highly dependent on the properties of the specific contaminant and the polymer (Bakir et al., 2014). Studies demonstrating the sorption of different contaminants on the same type of MP have shown different sorption behaviour attributed to differences in contaminant octanol-water coefficient (K_{ow}) values (Guo et al., 2012). Hydrophobic interaction mechanism was found to dominate in the sorption of two perfluorochemicals to PE, PS and PVC microplastics (Wang et al., 2015). The ability to partition between water and an organic phase measured by the plastic-water coefficient (K_{pw}), octanol-water coefficient (K_{ow}) and aqueous solubility (S_w) are some of the hydrophobic factors considered in the prediction of the sorption behaviour of organic contaminants.

In recent years, researchers have investigated the sorption behaviour and mechanisms of different chemical compounds including persistent organic pollutants, pharmaceuticals, antibiotics, hydrophilic compounds, aliphatic and aromatic compounds. Table 3 shows some of such studies:

5. Sorption models for estimating sorption coefficient

Many models exist for the study of the adsorption and desorption mechanisms of contaminants sorbed to microplastics. The most commonly used adsorption kinetic models include the first-order adsorption kinetics model (Skrip et al., 2013) and pseudo-second-order adsorption kinetics model (Miyake et al., 2013). Others are intra-particle diffusion and film diffusion models (Zhang et al., 2018a). Sorption equilibrium is established when the concentration of sorbate in the

bulk solution is in dynamic balance with that of the interface. The equilibrium data are usually fitted into Henry's isotherm (one parameter model), linear or non-linear Langmuir and Freundlich isotherms which are two-parameter models (Ayawei et al., 2017). Adsorption isotherms are used to predict the amount of sorbate that can sorb to a solid surface while kinetics models are used to estimate the efficiency of sorption. The isotherms are a graphical representation of the association between the sorbate concentration per unit weight of the sorbent and the remnant amount of sorbate at equilibrium (Desta, 2013). Table 4 presents some common models used to describe the sorption process. Adsorption kinetics typically involves four independent consecutive or parallel steps (a) bulk transport, (b) external mass transfer (film transport), (c) intra-particle diffusion and (d) chemisorption (Ho et al., 2000). A representation of the sorption process is depicted in Fig. 3.

The Langmuir model is typically designed for gas-solid adsorption. It balances relative adsorption and desorption by accounting for adsorbent surface coverage such that adsorption is directly proportional to the open adsorbent surface as desorption is to the covered adsorbent surface (Ayawei et al., 2017). The model is based on the following basic assumptions:

- A dynamic equilibrium exists between adsorbed ions or molecules and the free ions or molecules.
- The whole surface of the adsorbent is uniform.
- The molecules that have adsorbed have no form of interaction with each other.
- The mechanism is the same for to all adsorptions.

Table 4
Common kinetics and isotherm models used in sorption/desorption studies.

Type of model	Equation	References
Kinetic models		
Pseudo first order	$\text{Log}(q_m - q_t) = \log q_m - k_1/2.303 t$	Ho et al. (2000)
Pseudo second order	$1/q_t = 1/k_2 q_m^2 + 1/q_m t$	Ho et al. (2000)
Intra-particle diffusion	$q_t = K_{id} \times t^{1/2} + C_i$	Weber & Morris (1962)
Film diffusion	$B_t = -\ln(1 - q_t/q_e) - 0.4977$	Martins et al. (2015)
Intra-particle diffusion (desorption)	$f_{desorbed}(t) = 1 - M_t/M_0 = 1 - 6/\pi^2 \sum 1/n^2 \exp(-n^2 \pi^2 t D_p/r^2)$	Endo et al. (2013)
Aqueous boundary layer diffusion (desorption)	$f_{desorbed}(t) = 1 - M_t/M_0 = 1 - \exp(-3D_w S A_p t / \delta V_p K_{p/w}) = 1 - \exp(-3D_w t / r \delta K_{p/w})$	Endo et al. (2013)
Isotherm models		
Henry's isotherm (linear)	$q_e = K_d \times C_e$	Ayawei et al. (2017)
Langmuir (nonlinear)	$q_e = Q_{max} \times K_L \times C_e / 1 + K_L \times C_e$	Teuten et al. (2007)
Langmuir (linear)	$C_e/q_e = 1/q_m K_L + C_e/q_m$ $q_e/C_e = K_L q_m - K_L q_e$ $1/q_e = [1/q_m K_L] 1/C_e + 1/q_m$	Ayawei et al. (2017)
Freundlich (nonlinear)	$q_e = K_f \times C_e^{1/n}$	Teuten et al. (2007)
Freundlich (linear)	$\log q_e = \log K_f + 1/n \log C_e$	Teuten et al. (2007)

e) At maximum adsorption, molecules only deposit on the adsorbent surface and not on each other (monolayer adsorption) (vlab.amrita.edu, 2011).

Where q_m and q_e are the amount of adsorbate at equilibrium or adsorption capacity (mg/g); q_t is the adsorbed amount at time t ; K_{id} and C_i are intra-particle diffusion constants, k_1 and k_2 are the rate constants; B_t is Boyd constant; K_d is partition coefficient between the sorbent and the solution at equilibrium; C_e is the sorbate concentration at equilibrium; Q_{max} is the maximum adsorption capacity; K_L is Langmuir constant (mg/g); K_f is sorption affinity coefficient (L/mg) and $1/n$ is adsorption intensity (Freundlich constants); f_{sorbed} is cumulative fraction of desorbed contaminant between time 0 and t , M_0 and M_t are initial mass and mass of sorbate remaining at time t , respectively; D_p is diffusion coefficient in plastic; r is radius of plastic, D_w

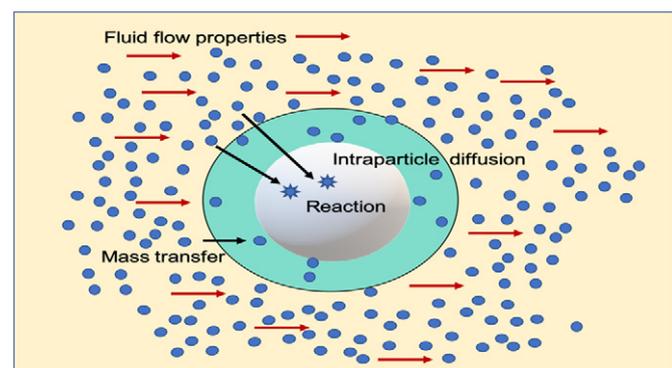


Fig. 3. Sorption of sorbate on a solid surface.

is diffusion coefficient in water, S_{ap} is surface area of plastic; δ is the thickness of aqueous boundary layer; V_p is volume of the plastic; K_p/w is partition coefficient between water and plastic.

The Freundlich model is used to describe both linear and nonlinear sorption. In recent sorption experiments, both PE and nylon fibers showed linear isotherms toward phenanthrene thus indicating their strong affinity and sorption capacity. However, the evaluated Freundlich $\log K_f$ parameter revealed that PE fibers had 1–2 orders of magnitude sorption capacity than the nylon fibers (Wang et al., 2018). Zuo et al. (2019), fitted sorption data for kinetics study into pseudo second-order model as a result of the success of the model from similar studies. The authors found the Freundlich linear model more appropriate to explain the sorption of phenanthrene on biodegradable plastics of PE and poly (butylene adipate co-terephthalate) (PBAT). Generally, adsorption studies begin with a series of batch experiments where sampled data are analyzed over equal time intervals. Experiments are usually conducted at room temperature except where the effects of variable temperature on sorption is being considered. A high correlation coefficient (R^2) value and the closeness of the experimental and theoretical values of the adsorption capacities of the sorbate (q_e) are essential. In a series of experiments conducted to establish model adsorption isotherms, three isotherm models namely Freundlich, Langmuir and Dubinin-Radushkevich were tested to determine the correlation of four sets of data. The differences in the R^2 values and standard error values were used as parameters to determine the best model for the data. Langmuir linear model was found to have the most significant value of R^2 and the lowest of standard error hence it was considered the best model for the dataset (Chen, 2015). In another sorption study, the R^2 value for pristine PVC and PS were found to be higher when fitted into a pseudo first order than the pseudo second-order model while the data for the aged version of the same plastics fitted into a pseudo second-order model (Liu et al., 2019b). The decision of fitting experimental data into a kinetic or isotherm model appears to be based on their success with previous studies or a trial and error process. The one-parameter linear free energy relationships (OP-LFERs) (i.e. K_{pw} , K_{ow} , S_w) and two-parameter models are limited in that they can only predict variability within a single class of compounds (Endo and Koelmans, 2016) and are inadequate for contaminants with tendency for strong hydrogen bonding and specific interactions with the organic phase (Nguyen et al., 2005). Hence, models known as poly-parameter linear free-energy relationships (PP-LFERs) which accounts for various intermolecular interactions to $\text{Log } K_{pw}$ have been used to describe sorption more accurately.

5.1. Poly-parameter linear free-energy relationships (PP-LFERs)

PP-LFERs are reliable models that can be used to predict the partition coefficients for several thousands of environmental phases with high accuracy. They can be applied to almost any neutral compound irrespective of chemical class with prediction errors <0.3 log units and accounts for contributions toward free energy change from wide range of molecular interactions with both water and bulk organic phases (Stenzel et al., 2013). For example, the Abraham Linear Solvation Energy Relationships (LSERs) for K_{pw} is given as.

$$\text{Log } K_{pw} = c + eE + sS + aA + bB + vV \quad (1)$$

where E = excess molar fraction, S = solute dipolarity/polarizability, A = solute H-bond donor property (acidity), B = solute H-bond acceptor property (basicity), V = molar volume.

E , S , A , B and V are the solute descriptors while c , e , s , a , b and v are regression coefficients. The solute descriptors describe the ability of the contaminant to undergo molecular interactions and the regression coefficients are indicative of the differences in particular interaction characteristics between plastic and water (Endo and Koelmans, 2016). Few recent studies on sorption of HOCs to plastics have applied the

Abraham LSERs (Hüffer et al., 2018; Uber et al., 2019). In the study by Uber et al. (2019), the sorption properties of HDPE for non-ionic organic compounds was investigated using the linear, Freundlich and PP-LFER models for interpretation of data. At aqueous concentrations below 10^{-2} $\mu\text{g/L}$, linear isotherm resulted in a $R^2 > 0.5$ ($p < 0.5$) while the non-linear Freundlich isotherm yielded $R^2 > 0.75$ ($p < 0.05$) suggesting that pore-filling adsorption is more relevant in the sorption process and that the Freundlich nonlinear isotherm gives a better fit than the linear isotherm. However, using the PP-LFER model, $R^2 > 0.84$ and root mean square error (RMSE) of 0.27 was obtained compared to RMSE of 0.52 for Freundlich isotherm indicating the robustness of the PP-LFER model for sorption by HDPE. There are also other PP-LFERs commercial software that do not require experimental descriptors and fitting coefficients; they rely solely on the molecular structures of polymer and the contaminants as inputs. Examples are COSMOtherm and SPARC; their prediction accuracy averagely lies within 1.0 log unit. Their application for polymer sorption is however limited (Endo and Koelmans, 2016). The descriptors for many compounds are available but for most environmentally relevant plastic types, regression coefficients are still lacking.

5.2. Implications of the sorption constants for the sorption of organic chemicals by plastics

The respective sorption coefficients, K_L and K_f in the Langmuir and Freundlich isotherms describe the partition of the organic sorbate between the existing aqueous solution and the plastic particles. The values of the sorption coefficients depend largely on the intermolecular interaction of the organic chemical in/on plastic and the surrounding water. Generally, hydrophobic organic chemicals (HOCs) exhibit favourable energy for interaction with plastic than the water phase; hence the chemical is more enriched in the plastic-water phase at equilibrium. (Endo and Koelmans, 2016). The parameter, $1/n$ in the Freundlich model is the exponent of non-linearity. It is associated with compounds of very low hydrophobicity. In most cases, the value of $1/n$ ranges between 0.7 and 1 showing that when the sorbate concentration increases, its relative adsorption decreases. This is also an indication that no adsorption site is available on the adsorbent for further sorption, suggesting a saturation and less adsorption occurs. $1/n$ values are rarely >1 except in compounds containing polar functional groups at low concentrations. However, $1/n$ values of <0.7 are indicative of highly curved, nonlinear isotherms. The R^2 value is an expression of the reproducibility of the data used to generate the Freundlich and Langmuir adsorption coefficients, K_f and K_L , respectively. The closer the value is to 1, the more reproducible the data (European Centre for Ecotoxicology and Toxicology of Chemicals, 2013).

6. Fate of plastic-sorbed contaminants in the environment and health risk

Common environmental organic contaminants such as PAHs show high partition coefficients with respect to plastics, indicating their strong affinities for plastic polymers. Partition coefficients of phenantrene to PE, PP and PVC were reported to be $38,100 \pm 5600$, 2190 ± 170 and 1650 ± 200 , respectively (Teuten et al., 2007). Some of the frequently detected environmental plastics are less dense than seawater (density 1.02 g/cm^3) hence they float and are able to sorb contaminants in the surface microlayer (SML) of water bodies (refer to Table 2). The floating contaminant-laden plastics are accessible to seabirds and marine organisms in the epipelagic zone for ingestion. Conversely, polymers like PVC, PS and PU and plastics with fouled surfaces, including the effects of additives have higher density than seawater or freshwater. Therefore, contaminant-sorbed plastics of this nature sink to the bottom of the water and are available to benthic organisms for ingestion (Teuten et al., 2007). The sorbed chemical contaminants have capacity to desorb from plastics into organisms (Bakir

et al., 2014; Rochman, 2015) and biomagnify from lower trophic organisms to fish (Kelly et al., 2007). The bioaccumulation of PCBs associated with plastics was demonstrated in lugworms, which resulted in weight loss and reduction in their feeding activity (Besseling et al., 2013).

6.1. Sorption phenomena and its implications to marine food-web

One of the most crucial problems with marine microplastics is their ability to act as vectors of harmful chemical contaminants. They are regarded as multiple stressors because they have the potential to impact organisms both physically and chemically (Rochman, 2013). Microplastics ingestion has been reported in organisms at virtually every trophic level, including zooplanktons, mussels and lugworms, shell fish, cetaceans, fish, seabirds and large sea mammals (Cho et al., 2019; Sun et al., 2018; Van Cauwenberghe et al., 2015; Neves et al., 2015). In addition, digestive fluids promote the leaching of sorbed contaminants on microplastic particles (Voparil and Mayer, 2000). The longer the ingested MPs stay in the guts of the organism, the higher the likelihood that the contaminants will translocate into the body tissue (Chua et al., 2014). In another study, Bakir et al., (2014) showed that the presence of organic matter (gut surfactants) significantly enhanced the desorption of organic pollutants from microplastics. Polybrominated diphenyl ethers (PBDEs) were loaded into MPs isolated from commercially sold cleansing soap and exposed to *Allorchestes compressa* (marine amphipods) for 72 h. Thereafter, gut clearance was allowed for 48 h in fresh seawater before oven-drying, homogenization, extraction and analysis. It was found that PBDEs sorbed on microplastics were assimilated into the tissue of the marine amphipods (Chua et al., 2014). A study established an artificial food chain between *Artemia nauplii* and *Dania rerio* (zebrafish) and monitored the uptake and transfer of two sets of virgin microplastics ($1-20 \mu\text{m}$). The first set was loaded with benzo(a)pyrene (BaP) while the other set was without BaP. Although no physical harm was recorded with MPs not loaded with BaP, some of the particles were retained in the guts of the zebrafish. On the other hand, the transfer of BaP by the loaded microplastic particles to zebrafish from the ingestion of *Artemia nauplii* was confirmed by fluorescence analysis. The study suggests that pollutant-loaded microplastics in invertebrate organisms ingested as food can leach in the intestine of vertebrate organisms and translocate to the stomach walls and the liver (Batel et al., 2016). These studies provide useful information that indicate the complex interaction of sorbed contaminants with biological systems.

6.2. Plastics - chemical interaction and its implications for human health risk assessment

Humans are exposed to microplastics uptake via the consumption of sea foods, drinking water, plastic-packaged drinks and inhalation (Digka et al., 2018; Eerkes-Medrano et al., 2015; Peixoto et al., 2019), including fishes from aquaculture (Cheung et al., 2018). There have also been reports of microplastics detection in food products like salt (Yang et al., 2015), sugar and honey (Liebezeit and Liebezeit, 2013), beer (Liebezeit and Liebezeit, 2014) and canned foods (Karami et al., 2018). Although scientific evidences abound on the occurrence of microplastics in foods consumed by humans and the capacity of microplastics to sorb harmful contaminants, no information is available yet about the fate of microplastics in the human body on the basis of microplastics ingestion (Rist et al., 2018). However, the bioaccumulation of MPs and sorbed POPs from lower nutrient levels may eventually impact human health (Miranda and de Carvalho-Souza, 2016). The risk of microplastics exposure to humans increases when sea organisms are consumed whole compared to when the digestive tracts are removed before consumption (Carbery et al., 2018). Particle toxicity is a viable perspective to determine potential toxic effects in humans. Drawing from other research fields involved in human toxicity studies due to particulate matter like air pollution, fiber technology and

nanotoxicology, it can be concluded that smaller particles, in the nanometer range may be able to cross cell membranes into body fluids resulting in cell damage, inflammation, and oxidative stress (Rist et al., 2018; Vethaak and Leslie, 2016). Further investigation and synchronization of collected environmental data will aid the performance of risk assessment of human health risk due to exposure to contaminated food. As microplastic pollution is increasing in the environment, it is expedient that research efforts be intensified on risk prediction, assessment and management concerning human health.

7. Conclusion and future outlook

The ability of plastics of various sizes to adsorb organic and inorganic contaminants in the environment is not in doubt. However, an understanding of the fate and transport mechanisms is required for effective toxicity studies, risk management and the prediction associated health effects. The sorption and transfer of toxic contaminants ingested with microplastics may likely impact the health of humans in the long term. Therefore, for effective and comprehensive risk prediction and assessment of the impact of plastic and microplastics pollution, future research should focus on (a) developing regression descriptors for frequently detected environmental plastics is required for accurate estimation of sorption coefficients (b) the sorption and desorption of chemical additives on microplastics (c) sorption studies of field-collected microplastics in the long term.

Declaration of competing interest

The authors declare they have no actual or potential competing financial interests.

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