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Role of Microplastics on the Release and Adsorption of Organic Compounds in Natural Waters

Ashton Collins
University of Maine, ashton.collins@maine.edu

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ROLE OF MICROPLASTICS ON THE RELEASE AND ADSORPTION OF ORGANIC COMPOUNDS IN NATURAL WATERS

By

Ashton McCormack Collins

B.S. Susquehanna University, 2019

A DISSERTATION

Submitted in Partial Fulfillment of the

Requirements for the Degree of

Master of Science

(in Civil Engineering)

The Graduate School

The University of Maine

August 2022

Advisory Committee:

Onur Apul, Assistant Professor of Civil Engineering, Advisor

Jean MacRae, Associate Professor of Civil Engineering

François Perreault, Associate Professor of Sustainable Engineering and the Built Environment, Arizona State University

ROLE OF MICROPLASTICS ON THE RELEASE AND ADSORPTION OF ORGANIC COMPOUNDS IN NATURAL WATERS

By: Ashton McCormack Collins

Dissertation Advisor: Dr. Onur Apul

An Abstract of the Dissertation Presented
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Microplastics in the aquatic system are among the many inevitable consequences of plastic pollution, which has cascading environmental and public health impacts. The implications lead to the production of leachate comprised of dissolved organic matter (DOM) and increasing adsorption potential of organic compounds (OCs) onto the microplastic surfaces. In this study, the adsorption potential of organic pollutants and the formation of microplastic leachate from ultraviolet (UV) light were explored.

The adsorption potential was created through a summarization and critical review of the literature on the adsorption of synthetic OCs by microplastics in aqueous environments since their emergence in 2008. A database of 92 articles, reporting 178 OCs, was created to provide a reference for our work. Our findings indicated that phenanthrene was the most commonly investigated OC, appearing in 13 of these studies. The adsorption of OCs were compared between the four most prevalent polymer types: polyethylene (PE), polystyrene (PS), polypropylene (PP), and polyvinyl chloride (PVC) using the linear isotherm parameter, K_D, to determine adsorption variability and understand the controlling factors for adsorption.

Surface interactions and leachate production of six microplastics under UV irradiation were determined. Leachate production was analyzed for the dissolved organic content (DOC), UV absorbance (i.e., UV_{254}), and fluorescence through excitation emission (EEM) to determine the amount of leachate produced and the mechanisms involved in leachate production via UV irradiation. The aged microplastic samples were analyzed for Fourier-transform infrared spectroscopy (FT-IR), Raman, and X-ray photoelectron spectroscopy (XPS), to determine the surface changes in combination with leachate formation. The differences in leachate formation for different polymers were attributed to their chemical makeup and their potency to interact with UV.

Findings indicated that all microplastics showed evidence of surface oxidation, affirming that the leachate formation is an interfacial interaction and could be a significant source of organic compound influx to natural waters due to the abundance of microplastics and their large surface area. In terms of adsorption of organic compounds by microplastics, it was found that the octanol-water partitioning coefficient, K_{ow}, is an appropriate predictor for adsorption capacity with simply structured polymer types, particularly PE. As hydrophobicity increases, PE adsorbs the most, followed by PVC, and then PP. Comparing the K_D to the dipole moments for several different compounds on PE and PS showed that microplastics probably do not have induced electronic interactions, and supported the conclusion that adsorption is most likely driven by a compound's repulsion from water rather than its affinity for microplastics.

ACKNOWLEDGEMENTS

I would like to give thanks to my supervisor Prof. Dr. Onur Apul for his incredible guidance into the world of academic research and continued support throughout my master's experience. I would also like to thank the thesis committee members Prof. Dr. François Perreault and Prof. Dr. Jean McRae for their amazing contributions and insight into our research. I am grateful to the National Science Foundation (NSF) for funding this research and providing financial support to learn, experiment, and discover into the world of microplastic research.

Thank you to everyone who helped throughout my research: Kartik Bhagat, Eliza Costigan, Dr. Mohamed Ateia, and Prof. Dr. Stohm Ohno for helping throughout my experience and contributing in numerous ways to the progression of this research, it would not be what it became without all of you. Lastly, I would like to thank my mother for her continued support and guidance throughout this time period of my life, I would not have been able to finish this thesis without someone having my back and full support throughout my time at the University of Maine.

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LIST OF ABBREVIATIONS

DOC: Dissolved organic carbon

DOM: Dissolved organic matter

EEM: Excitation emission

FOSA: Perfluorooctanesulfonamide

HALS: Hindered amine light stabilizers

MPs: Microplastics

$N0_x$: Nitrogen oxides

OC: Organic compounds

PAHs: Polycyclic aromatic hydrocarbons

PCBs: Polychlorinated biphenyls

PE: Polyethylene

PE_{rec}: Recycled polyethylene

PFAS: Per and polyfluoroalkyl substances

PFOS: Perfluorooctanesulfonic acid

PLA: Polylactic acid

PP: Polypropylene

PS_{col}: Colored polystyrene

PS: Polystyrene

ROS: Reactive oxygen species

UV: Ultraviolet

VOCs: Volatile organic compounds

THESIS OVERVIEW

Chapter I introduces what plastics are, how they are created, and disposed of. The chapter also discusses the formation of microplastics and the many different pathways that breaks down plastics into the micro scale. There are two essential areas of research with the formation of microplastics: the adsorption of organic pollutants onto the surface and leachate production as the microplastics continue to degrade.

Chapter II analyzes the literature present on both areas: Part (A) is the adsorption of organic pollutants literature review published in the Journal of Hazardous Materials Advances:

> *Costigan, E., Collins, A., Hatinoglu, M.D., Bhagat, K., Macrae, J., Perreault, F., and Apul, O. (2022). Adsorption of organic pollutants by microplastics : Overview of a dissonant literature. Journal of Hazardous Materials Advances, 6(March), 100091. doi: 10.1016/j.hazadv.2022.100091*

Part (B) gives an overview of the mechanisms that can breakdown microplastics in the aquatic environment.

Chapter III is the research conducted to help fill in the literature gaps on leachate production with analysis of degradation of different polymer types from simulated sunlight using a UV aging chamber. This study is accepted with major revisions in *Environmental Science: Water Research and* Technology:

> *Collins, A., Ateia, M., Bhagat, K., Ohno, T., Perreault, F., and Apul, O. (2022). Microplastic leachate formation under UV irradiation: extent, characteristics, and mechanisms. Manuscript submitted for publication to Environmental Science: Water Research & Technology.*

Chapter IV discusses the important takeaways from this research and the avenues for further research into understanding the fate of microplastics in the natural aquatic environment.

The thesis helps to fill the necessary gaps on two-way surface interactions of microplastics, analyzing the adsorption potential and leachate release of different polymer types which both have significant impacts on water quality. There are five overarching objectives of the thesis:

(1) To survey, summarize, and analyze the literature in terms of adsorption of organic compounds by microplastics.
(2) Determine the effects of polymer types and UV aging conditions for leachate (dissolved organic matter (DOM)) formation.
(3) Relate the changes in DOM formation to polymer surface degradation changes concurrently.
(4) Determine the physiochemical mechanisms leading to DOM production for each polymer type.
(5) Integrate surface degradation and adsorption potential of different polymer types.

CHAPTER I: INTRODUCTION TO PLASTICS AND MICROPLASTICS

1.1 What are Plastics?

Plastics are synthetic or natural polymers comprised of repeating monomers. The majority of plastics are formed from the reaction of small organic molecules mostly derived from fossil fuels. Plastics are malleable, have low thermal conductivity, cost effective to produce, and are light in mass (Chamas *et al*., (2020)). Due to these properties, they are used in almost every industry and have become essential to everyday life. There are two main types of plastics based on their physical characteristics: thermoplastics and thermoset plastics. Thermoplastics are plastics that can be heated, molded, and cooled repeatedly. Thermoplastics are the most used branch of plastics in each industry, with examples including polyethylene (PE), polypropylene (PP), and polystyrene (PS). Thermoset plastics can only be heated and cooled once and will form permanent chemical bonds across the polymer chain. Polyesters and epoxy resin are examples of thermoset plastics (Chamas *et al.,* (2020)).

Each branch can be broken into two large subsets: aliphatic hydrocarbons and aromatic hydrocarbons. Aliphatic hydrocarbons are comprised of chain carbon (C-C) bonds. They can be alkanes, alkenes, or alkynes with single, double, and triple C-C bonds respectively. A term that gets used often within literature are polyolefins, comprised of the olefin monomer repeating unit. Aromatic hydrocarbons have a phenyl or benzene ring attached to the repeating units (Feldman, (2002)). The formation depends on how the polymer chain is formed and the plastics intended purpose. The comparison of an aromatic and aliphatic hydrocarbon structure is in (Figure 1).

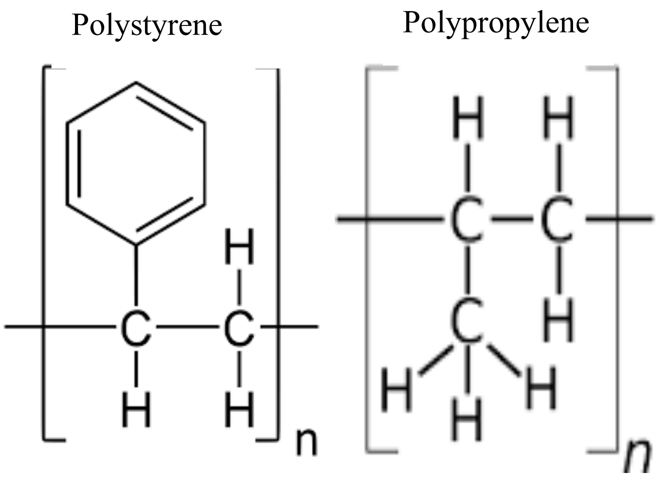


Figure 1. Chemical structure of an aromatic polymer (left) compared to the chemical structure of an aliphatic hydrocarbon (right).

1.2 Plastic Production and Disposal

The innate properties of plastics have led to the production rates to increase exponentially from 2 to 380 Mt per year over the past 70-80 years (Geyer *et al*., (2017)). The production rate is equivalent to over one million water bottles worth of plastic every minute (Van Cauwenberghe & Janssen, (2014)). The heavy usage of plastics leads to millions of tons of being disposed of through landfilling, incineration as well as through runoff and tidal interactions along beaches. Landfilling is a common endpoint for plastic waste, but the properties of plastics allows them to persist for 20-30 years in these landfills due to the anoxic environment, filling large spaces across landfills. Over this time, it is likely that the plastics will leach toxic chemicals into the groundwater. Incineration of plastic waste is another option but burning will release polymer-based gases into the atmosphere including polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds (VOCs), and nitrogen oxides (NO_x). Recycling has been pursued where intensive manufacturing techniques must be applied to remove potential contaminants from the surfaces before use. The process of recycling is expensive, resource intensive, and produces lower quality plastic materials and is why plastic recyclability is limited (Ncube *et al*., (2021)).

The exponential increase in plastic production has also led to an exponential increase in plastic particles in aquatic systems, up to over five trillion pieces of plastic in the worlds water systems (Eriksen *et al*., (2014)). Plastics can enter aquatic systems through a multitude of methods: runoff from irrigation, discharge, direct disposal, cleaning of textiles, and from manufactured products (Singh *et al*., (2022)). Therefore, plastic pollution is encountered in all kinds of environmental systems, including water, soil, air, and sediment. The largest movement of plastic into these systems comes from river discharge, where it is estimated that globally, 12 rivers move over a quarter of total plastic waste into coastal watersheds (Lebreton and Andrady, (2019)). As important as plastics are to daily lives, there is no proper disposal method without severe environmental pollution ramifications.

1.3 Formation of Microplastics

The majority of plastics enter the aquatic environment as macro (>5 mm) sized particles. Over time, these macro sized plastic particles decrease in size into the micro (<5 mm) and nano (<0.0001 mm) size ranges. Micron sized plastics are most common today, called microplastics (MPs), which present numerous environmental dangers due to their small sizes (i.e., high specific surface area). MPs are within the perfect size range for consumption from many different aquatic species where there is potential to block the digestive system. Ecological and ecotoxicological concerns with MPs can be categorized as biological concerns, leaching of organic matter, and adsorption potential due to constantly being transported throughout the environment (Figure 2).

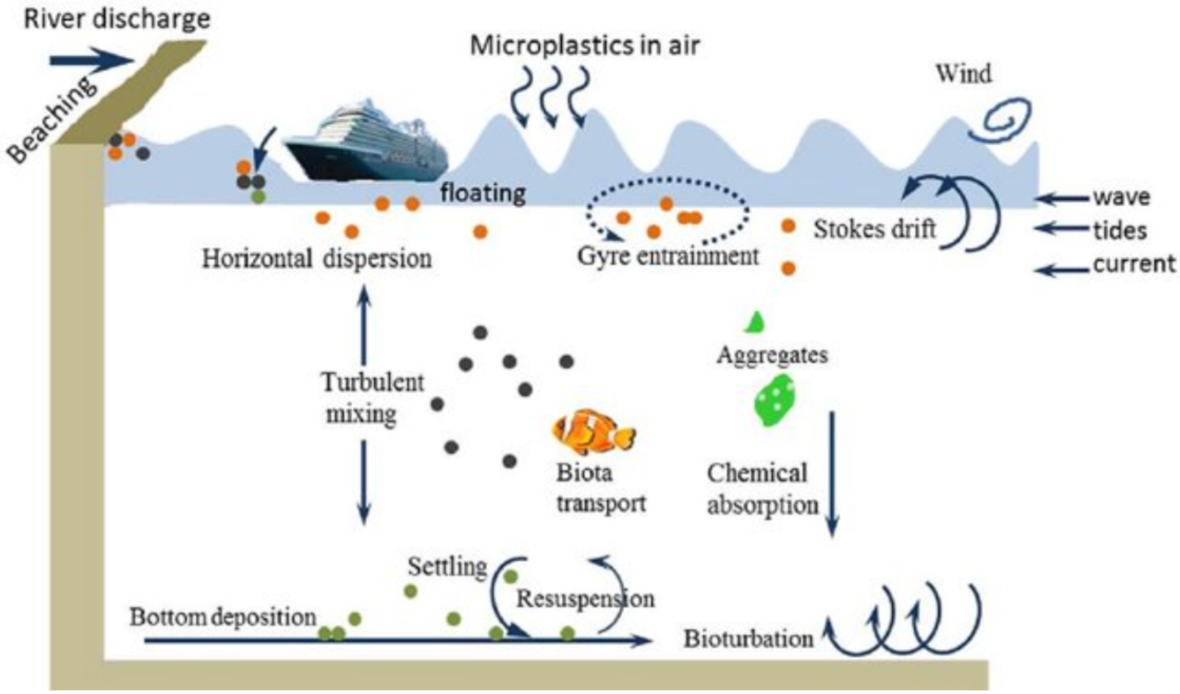


Figure 2. Schematic of the transport pathways of microplastics in the ocean (Welden and Lusher, (2017)).

Plastics entering the environment will constantly be in motion, interacting with wind, the sun, waves, and microbial organisms degrading into MPs over time. These are called secondary MPs where they enter the environment as macro sized particles but degrade into the micro scale through various degradation mechanisms. Degradation will continue even after they become MPs and will continue to decrease in size, while increasing in surface area.

1.4 Two-Way Surface Interactions of Microplastics

Degradation of MPs leaches off organic matter from their surfaces called leachate. It is comprised of fibers and DOM. Organic matter that is dissolved is <0.45 μm in size. Leachate formation occurs through various abiotic and biotic methods including: UV irradiation, hydrolysis, biodegradation, physical abrasion, and thermal degradation (Feldman, (2002)). These mechanisms are discussed further in Chapter II (b) (Figure 3).

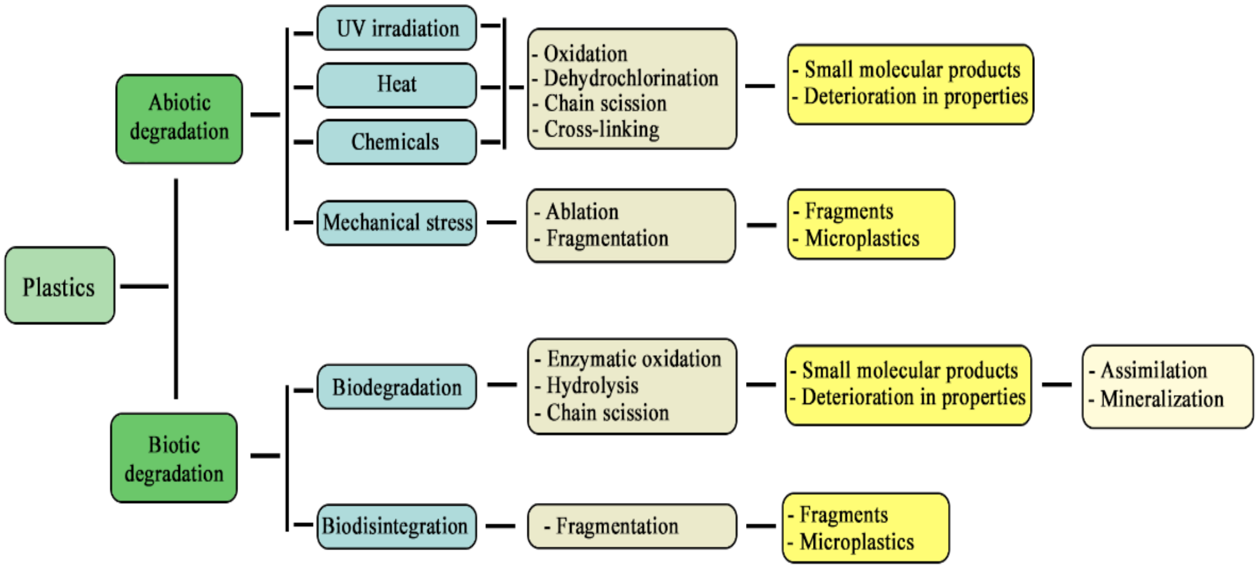


Figure 3. Weathering pathways of microplastics (Alimi *et al.,* (2022)).

All plastic leachate has been determined to have a certain level of toxicity when consumed acting as endocrine disruptors, reproductive inhibitors, and cancer inducing toxins to marine organisms (Ateia *et al*., (2020); Singh *et al*., (2017)). It also has implications on water quality as conventional treatment plants are not purposefully designed to remove microplastic leachate, where human consumption of MP leachate likely will lead to similar health issues as marine based organisms if enough was consumed (Gewert *et al*., (2018)).

The release of organic matter over time causes the morphology of the polymer surfaces to change through fiber generation, micro crack formation, and fragmentation (Mao *et al.,* (2020);

Hüffer *et al*., (2018); Naik *et al*., (2020)). Crack formation and fragmentation allows adsorption sites to form for the ~20,000 organic compounds in marine environments, with the ever-increasing list of emerging contaminants. MPs also have high hydrophobicity and combined with the high surface area with degradation, creates ideal surfaces to form pollutant-laden particles, and shuttle these harmful chemicals into marine-based organisms. Metabolic pathways may transfer the chemicals from the plastic surface to tissues in these organisms where there can be numerous health implications to the organism (Bhagat *et al*., (2022); Galloway *et al*., (2017)). Further adsorption analysis of different polymer types is discussed in Chapter II (a). In addition, the hydrophobic surfaces of the MPs allows for microbial organisms to form a biofilm across the polymer surface creating not only a vessel of toxic pollutants, but microbial organisms as well. These factors bring numerous negative impacts on living organisms, such as toxicity, enzyme inhibition and immobilization (Zettler et al., (2013)). As MPs degrade, there is toxic leachate released from the surface, increasing surface area and forming cracks throughout the polymers surface. This increases adsorption potential creating the importance of two-way surface interactions of MPs in the natural environment. The adsorption potential of polymers and the organic matter it leaches as it degrades both present numerous environmental implications on water quality and the health of marine organisms when consumed. These cascading implications of MPs sanction further research for adsorption and release of organic matter from their surfaces to understand the two-way surface interactions of MPs in the aquatic environment.

CHAPTER II(A): INCREASING POTENTIAL OF ADORPTION OF ORGANIC POLLUTANTS ONTO MICROPLASTICS

Costigan, E., Collins, A., Hatinoglu, M.D., Bhagat, K., Macrae, J., Perreault, F., and Apul, O. (2022). Adsorption of organic pollutants by microplastics : Overview of a dissonant literature. Journal of Hazardous Materials Advances, 6(March), 100091. doi: 10.1016/j.hazadv.2022.100091

* Manuscript is published in Journal of Hazardous Materials Advances but it has been modified to integrate into the thesis.

2.1 Introduction

2.1.1 Introduction to Adsorption of Microplastics

The prevalence of microplastics in the environment has even created a relatively new domain that can be defined as the "microplastisphere". This term has been used to define the soil-plastic interface (Zhou *et al.,* (2021); Ateia *et al.,* (2022)), but microplastics have such a wide breadth of influence in all environments that this term would be lacking if it did not include the water-plastic interface. This term is differentiated from the term "plastisphere", first coined by Zettler *et al.* in 2013, as microplastics may have very different interactions with the environment than macro-plastics. Predominantly, the microplastisphere can possess an overwhelmingly large surface area, which can cause cascading environmental problems such as adsorption/desorption of micropollutants and heavy metals and polymer leaching; this also effects the surrounding hydrosphere and biosphere (Rubin & Zucker, (2022); Liu et al., (2022)). In addition, microplastics often have imperfections such as surface complexity as well as irregularities in shape, which could further increase the already excessively large microplastisphere surface area (Figure 4).

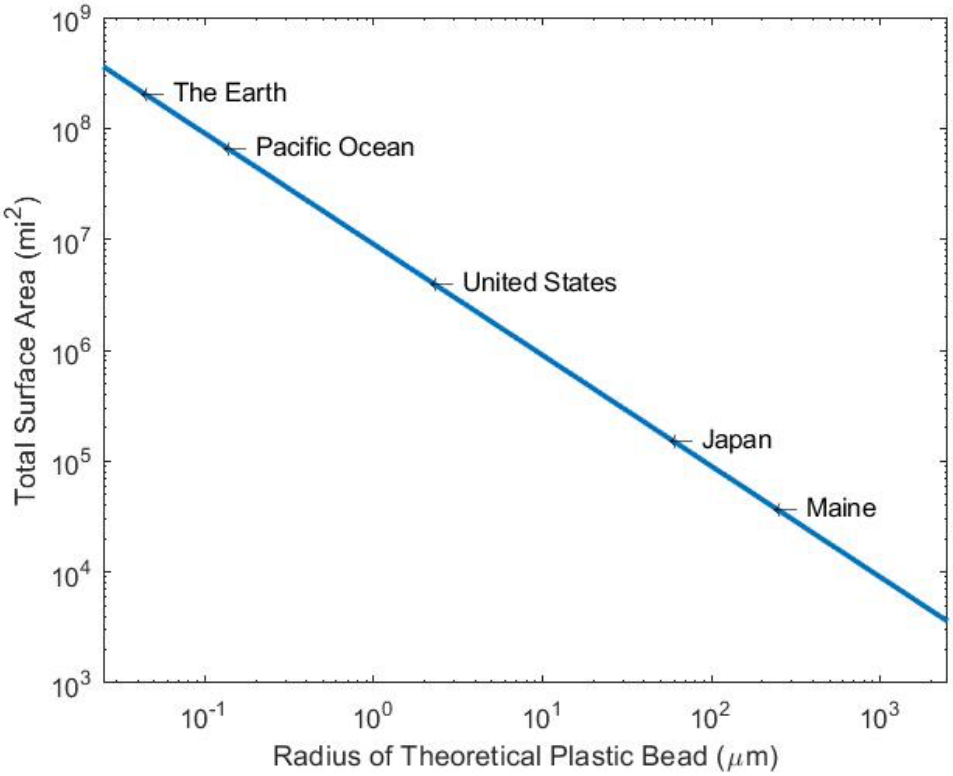


Figure 4. Calculated surface area for global plastic disposed into oceans every year based on the assumption of perfect, uniform pulverization of the plastic debris into microplastics (Jambeck, (2015)).

Considering the large surface area, strong hydrophobicity, and xenobiotic properties of the microplastisphere, investigating adsorptive interactions of microplastics is essential to advance our knowledge on how they interact with and influence the fate, transport, and ecological impacts of micropollutants. Microplastics can serve as adsorbents owing to their high surface areas, generating a significance to their occurrences in the environment (Agboola & Benson, (2021)). One possible impact of their widespread presence is to potentially sorb organic compounds (OCs) from natural waters and contain them on a solid surface, making them more bioavailable to some organisms. In other words, pollutant-laden microplastics can shuttle harmful chemicals into organisms when ingested, and different metabolic pathways may transfer the chemicals from the plastic surface to tissues (Tanaka *et al.,* (2013)). Here, the chemicals can biomagnify and cause harm at higher trophic levels (Hai *et al.,* (2020); Ma *et al.,* (2016)).

Adsorption of OCs by microplastics has been studied in great depth since the late 2000s. To understand and prevent the negative impacts that come along with OC-laden microplastics, the controlling forces causing interactions between OCs and microplastics should be identified. Therefore, this review paper summarizes the results of all of the available adsorption literature, published up to August of 2021. Studies were retrieved using the keywords "microplastics adsorption" with the terms "organic compounds," "pollutants," "adsorption mechanism," "influences," and "behavior" from the databases Science Direct, American Chemical Society, SpringerOpen, and ResearchGate. In total, 91 experimental studies on adsorption of OCs by microplastics were retrieved and analyzed systematically. Multiple other reviews on this topic have been published in recent years (Fu *et al.,* (2021); Guo and Wang, (2019a); Joo *et al.,* (2021); Mei *et al.,* (2021); Wang *et al.,* (2020a); and Yu *et al.,* (2019)). However, some of these reviews only summarize and tabulate a selection of the available adsorption literature (Fu *et al.,* (2021) with 31 studies; Wang *et al.,* (2020) with 17 studies; Guo and Wang, (2019) with 27 studies). These reviews have qualitatively discussed adsorption mechanisms based on the findings of other researchers, concluding that specific interactions depend on surface area, size, pH, salinity, polarity, and compound hydrophobicity. However, none of these studies have attempted to quantify these effects by modeling methods, and none have attempted to establish a correlation between K_{ow} (used as a surrogate for compound hydrophobicity) and K_D (the linear isotherm parameter), as has been done in this study. Therefore, with 91 papers, 68 of which contribute to the adsorption analysis, this review presents a summarization of the largest set of literature currently available, and is the first to attempt a quantitative analysis of such a wide breadth of results. First, an overview of the different microplastic types, sizes, and reporting methods is given. Second, the influence of OC properties and background water characteristics on adsorption by microplastics is investigated. Overall, this work summarizes the existing findings and provides mechanistic insights regarding the adsorption of OCs by microplastics while indicating gaps in literature that may need further research.

2.1.2 Overview of Global Plastic Production and Disposal Patterns

There are dozens of different plastic types and composites in production and use. In this study, plastic production and disposal data were sorted into six major polymer categories: polyethylene (PE), polypropylene (PP), polystyrene (PS), polyethylene terephthalate (PET), polyvinyl chloride (PVC), and polyester, polyamide, and acrylic fibers (PP&A). A final "other" category was included to cover less common plastic types as well as additives and composites. Due to the limited number of studies on different categories of PE (low- or high-density), PE was not subcategorized by density in this study. It is imperative to understand the life-cycle of the most relevant plastics that may end up in natural aquatic systems, as they may eventually break down into microplastics and interact with OCs.

Based on overall global production, the top three most common plastic types are PE, PP, and PP&A, and they constitute 30, 17, and 15% of total production, respectively (Geyer *et al.,* (2017)). The first two of these plastic types are most used for single use, everyday products (e.g., water bottles, plastic bags, plastic containers), while PP&A are fibers are most commonly used in clothing and fabrics (WHO, (2019); Ncube *et al.*, (2021)). The next three plastic types i.e., PVC, PET, and PS, made up 10, 8, and 6% of global plastic production, respectively while the remaining 13% constitutes other miscellaneous plastics. The global yearly disposal of plastics is similar but not identical to their production rates constituting 32 to 34, 19, and 15% of the total for PE, PP and PP&A, respectively (Geyer *et al.,* (2017), Erni-Cassola *et al.,* (2019)). PET and PS were also discarded at similar proportion as they were produced, at 11 and 6%, respectively. The differences between production and disposal of PE, PP and PP&A indicate variations of the plastic life-cycle, regional production and disposal patterns, and possible errors in estimating the global trends. PVC on the other hand made up a much smaller proportion of plastic wasted, at 5% of the total (Geyer *et al.,* (2017)). PVC may be discarded at a smaller proportion because it is generally used in the production of long-lasting construction materials such as piping, exterior building sidings, and window frames; therefore, PVC could have an extended life cycle in contrast to single-use plastics (Hardin, 2021). The amount of plastic currently entering the ocean may be less than 3% of the total plastics wasted yearly, but the cumulative quantity of plastics available to enter the ocean from the land was predicted to increase by an order of magnitude between 2015 and 2025 (Jambeck, (2015)), illustrating the alarming growth of the problem.

2.2 Microplastic Characteristics

2.2.1 Reporting by Size Versus Volume

Accounting of microplastics in the environment is a complex task. Figure 5, as an example, shows the environmental microplastic occurrence percentages by number (Figure 5a) and by calculated volume (Figure 5b) from 10 papers. For Figure 5a, reported microplastic particle numbers were binned by the average diameter of the particles, and the bars represent the bin's percentage of the total number of particles throughout the 10 studies. Figure 5b is a calculation based on Figure 5a and the respective microplastic volumes, with the simplistic assumption that all particles are perfect spheres. The bars represent the percentage of the total volume of microplastics reported in these 10 studies. This calculation is performed to show that there is a marked difference between the representation of number *vs.* volume distribution for the same microplastic occurrence data. Microplastics of size less than 0.5 mm make up the majority of the total number of microplastic particles (~52%) found in the aquatic environment but make up less than 3% of the total volume. This indicates that representing microplastics in terms of number could be misleading. Therefore, reporting the surface area of microplastics that are used in adsorption studies could be an informative approach, especially for understanding the surficial intermolecular interactions.

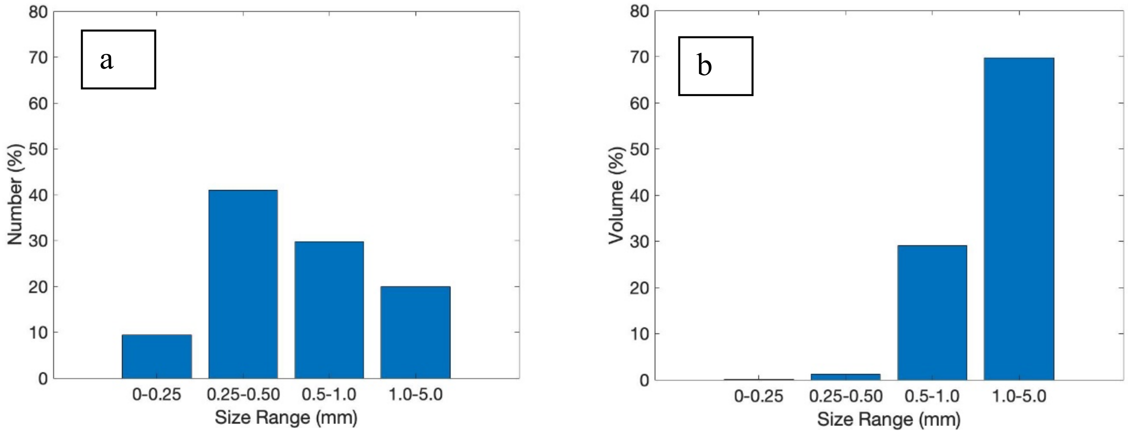


Figure 5. Microplastics found in the environment by **a)** reported number of particles and **b)** calculated volume, globally (Lusher *et al*., (2015); Amelineau *et al*., (2016); Barrows *et al*., (2018); Enders *et al*., (2015); Kanhai *et al*., (2017); Pan *et al*., (2019); Wang *et al*., (2020b); Mu *et al*., (2019); Zhao *et al*., (2015); Frere *et al*., (2017); Klein *et al*., (2015)).

2.2.2 Comparison of Size Versus Surface Area

In an effort to understand the relationship between microplastic size and surface area, the nominal size and surface areas reported for PP, PE, PVC, and PS were compiled from 30 OC adsorption articles, and the data are presented in Figure 6. Compiled size *vs.* surface area data show that surface area has an inverse relationship with size regardless of the plastic type. Overall, polymer type did not have a notable influence on surface area and size relationship but all polymer types had larger surface areas than estimated theoretical spherical particle surface areas (i.e., indicated by the solid lines). It should be noted that some of the microplastic diameters were taken from the manufacturers and some studies used particles sieved to a certain size, which would influence the theoretical calculation of microplastic surface area. However, while sizes ranged from 0.05 to 800 μm, the majority of the data were obtained using plastics with diameters between 50 and 200 μm. The specific surface areas (SSA) ranged from 0.19 to 73 $m^2 g^{-1}$, but most of the SSAs were less than 5 $m^2 g^{-1}$. This indicates that the adsorption literature currently covers micrometer-millimeter scale particles as adsorbents with relatively low surface areas. Additional studies are required using microplastics at the nanoscale or submicron scale with notably higher surface areas to provide pertinent insights regarding their environmental implications. All papers reported here used Brunauer-Emmett-Teller (BET) surface area for microplastics, and studies using chemically-treated microplastics (i.e., the ones artificially aged by UV irradiation, chromic acid etching, and ozonation) were omitted from this analysis for consistency. The effect of surface oxidation, acid washing or any other chemical pretreatment on surface area would require further investigation because the modifications of microplastic surface chemistry may also change the surfaces or internal porosities of microplastics and, consequently, influence OC adsorption (Bhagat *et al.,* (2022); Hüffer *et al.,* (2018a); Liu *et al.,* (2020); Sarkar *et al.,* (2021)). In our summary, some of the size data were averaged into arithmetic means and some were taken as reported with whiskers added to show reported ranges when applicable.

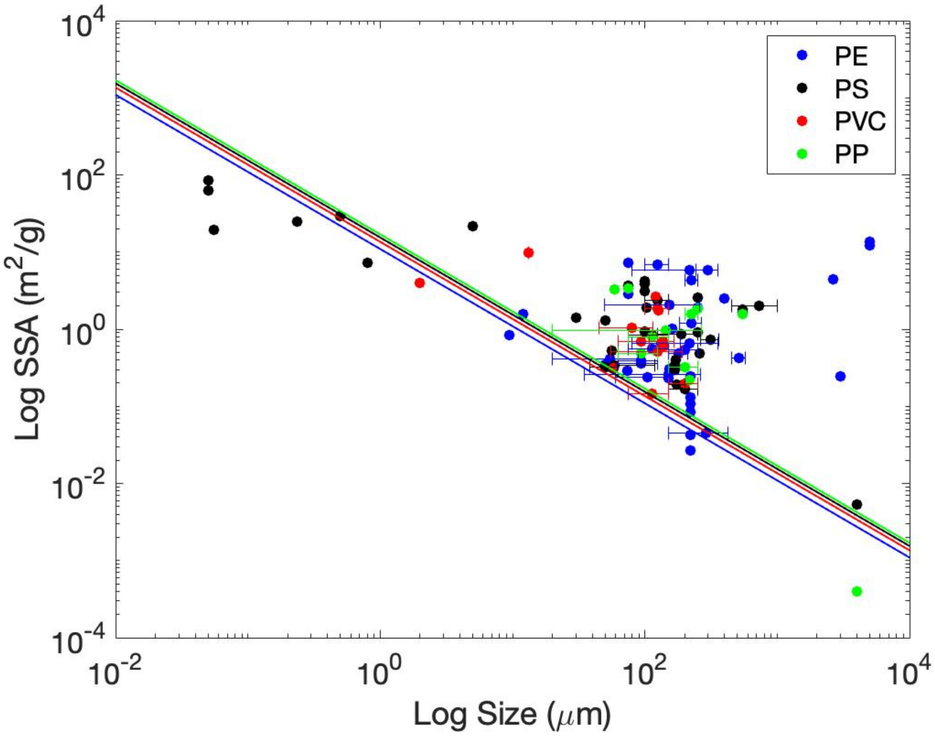


Figure 6. Surface area comparison with diameter for polyethylene, polystyrene, polyvinyl chloride, and polypropylene, compiled from 37 papers (Appendix). Solid lines represent theoretical calculations for surface area with respect to size.

Solid lines in Figure 6 represent theoretical surface area (SA) calculations with respect to size for PE, PS, and PP. Theoretical SA calculations were performed considering the dimensions and density of microplastics assuming spherical shape with no tortuosity or internal porosity. The theoretical surface areas were in greater agreement with the BET data at the nano- and submicron scales; however, as the size increases, theoretical calculations underestimated the measured SSA. This could be attributed to the non-uniform shape and tortuosity variability of larger microplastic particles, which increases the SSA of microplastics. The smaller microplastics may demonstrate more uniform surfaces, which may indicate the lack of internal volume to create pores or channels within the particle. This is supported by the studies reviewed here, the majority of which used SEM to analyze the surface of their microplastics. In particular, the authors point to the study by Yilimulati *et al.,* (2021) that showed a lack of surface complexity on polystyrene nanoplastics, a stark contrast to the dozens of studies that showed very complex microplastic surfaces at the 100-μm range. However, more data on various polymer types at the submicron scale is required to make a more conclusive statement.

In terms of environmental relevance, Figure 6 shows that theoretical calculations will underestimate surface interactions when surface imperfections are ignored. Therefore, experimental SSA identification such as nitrogen or krypton adsorption, micro-CT or mercury intrusion-based surface area analysis are essential to complement theoretical calculations and mechanistic adsorption experiments. Another uncertainty lies in the way that articles report microplastic properties. Some researchers report the manufacturers' properties, while others measure them. Similarly, some report only one size of the plastic, while others provide a range. In addition, some studies are conducted using plastics as obtained from the manufacturer, while other researchers wash, dry or sieve plastics before surface area measurement. The lack of washing of microplastic particles could be especially problematic, as particles obtained from the manufacturer may contain other compounds that should be cleaned before testing. Additionally, this step can be performed without altering the results of an experiment; Qiu *et al.* (2019) showed that washing PE, PP, and PVC microplastics with n-hexane and dichloromethane did not visually change the surface of the plastics. Therefore, washing should be an integral part of the pretreatment process. As a result of the different pretreatment, sieving, and characterization approaches, there are wide variations in reported data. More uniform testing and reporting of experimental methods throughout the literature would reduce data variability. On the other hand, microplastics in the environment are extremely heterogeneous and over-simplified model materials (e.g., sieved, uniform, pristine, plastic beads) do not accurately reflect real-world conditions. Therefore, for fundamental and mechanistic insights, the simplistic, single polymer type, sieved and washed uniform microplastics will generate more readily interpretable results, while microplastic mixtures with non-uniform shapes, sizes covered with biofilms, co-occurring humic and fulvic acids, adhesives, and plasticizers would be more representative of environmental conditions. To bridge the gap between the fundamental mechanistic studies and those that more accurately reflect environmental conditions, the environmental relevance of adsorption experiments can be gradually increased by controlling the size, types, and surface properties of the plastics while adding complexity under defined laboratory conditions.

2.3 Adsorption of Organic Compounds by Microplastics

2.3.1 Overview of Adsorption Studies and Selection Criteria

Figure 7 shows the types of plastics used in adsorption studies. Consistent with the global production, PE is most studied, at 33% of the total. Many papers group together all PE variations, including HDPE, LDPE, and others, as one category. Although not focused thoroughly, the crystallinity of PE could play a role on the adsorption behavior and further systematic research could answer some key research questions regarding their adsorptive properties. The second most studied polymer is PS, at 22%, followed by PVC and PP at 14% and 12%, respectively. Combined, the remaining categories make up less than 20% of the total, so these categories were not analyzed for adsorption trends.

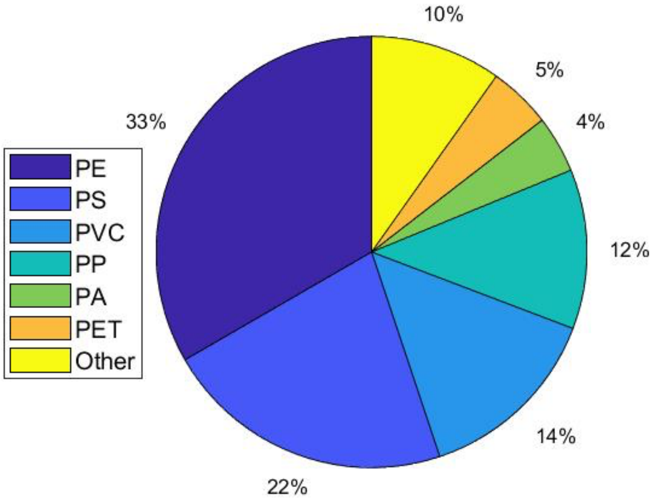


Figure 7. Adsorption studies by polymer type found within the literature as a percentage of 234 total studies performed within 91 different papers.

The 91 studies reviewed for this paper used a wide range of methods and materials to obtain adsorption data. Therefore, to appropriately compare adsorption characteristics, the papers were sorted to capture intermolecular interactions consistently. First, the analysis was constrained to virgin microplastics with different sizes and surface areas, avoiding experiments that altered their microplastics by methods such as UV aging. Therefore, only studies that used physical alterations of the plastics (e.g., grinding, milling, or sieving) were included in the analysis. These methods simply break up or sort the plastics, which change only their size and surface area. In addition, adsorption studies that used water (including synthetic and natural seawater and freshwater) as the background solution were included in our analysis. Furthermore, several papers did not include an analysis of adsorption at (pseudo)-equilibrium, and those articles were not included in the database. The isotherms most commonly used to model the adsorption equilibrium data throughout the dataset were Freundlich, Langmuir, and linear models. In our summary, the partitioning coefficients, K_D, at the Henry's region (linear portion) of the isotherms were indexed and analyzed to represent adsorption. Reporting articles either gave K_D or allowed for attainment of K_D from isotherms. The linear partitioning model is given in Equation (1).

$$q_e = K_D C_e \qquad \text{(eq. 1)}$$

Where q_e is the amount of absorbate on the sorbent (mg g^{-1}), C_e is the amount of absorbate left in solution (mg L^{-1}), and K_D is the linear adsorption constant (L kg^{-1}) under equilibrium. Of the 91 studies reviewed for this paper, 68 studies were deemed applicable for further analysis based on the preliminary screening. The 68 papers yield 234 experiments using various polymer and OC types, totaling 770 isotherms.

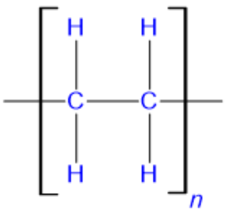
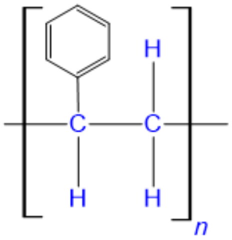
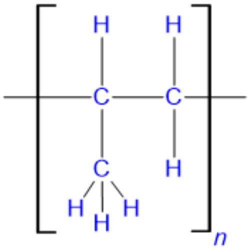
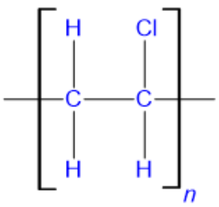
2.3.2 Adsorption Analysis

The remainder of the discussion will focus on the 68 papers deemed applicable for the adsorption analysis. There were 147 OCs tested in the data set, which were grouped into 5 different categories: benzene derivatives (27 compounds), polycyclic aromatic hydrocarbons (PAHs) (33 compounds), polychlorinated biphenyls (PCBs) (17 compounds), per- and polyfluoroalkyl substances (PFAS) (18 compounds), and a final "other" category for the remaining miscellaneous compounds (52 compounds). The "other" category is made up of complex molecules (e.g., heterocyclic, heavily charged compounds) that serve as antibiotics, fungicides, herbicides, pesticides. The most repeatedly tested OC in the data set was phenanthrene, tested in 28 studies; and followed by naphthalene, tested in 17 studies. Both phenanthrene and naphthalene are among the EPA priority pollutant-listed PAHs ("Toxic and Priority Pollutants Under the Clean Water Act", (2021)), a class of pollutants which are challenging to remove from the environment due to their chemical persistence and semi-volatile nature (Yakout *et al.*, (2012)). Among their possible advantages as test contaminants are their simple structures (enabling intermolecular interactions to be observed), wide commercial availability, well established analytical methods, and low cost.

2.3.3 Influence of Polymer Type

To investigate the controlling factors for OC adsorption by microplastics, the K_D for each compound was compared against the compound's octanol-water partitioning coefficient (K_{ow}), which is commonly used as an indication of compound hydrophobicity (Moldoveanu and David, (2021). Some papers compare their adsorption results to K_{ow} and report a positive correlation between compound hydrophobicity and adsorption (Lan *et al.,* (2021); Xu *et al.,* (2021); Xu *et al.,* (2018)). A larger slope of the linear regression indicates a stronger effect of hydrophobicity on adsorption. Figure 8a presents this comparison for the four most commonly studied polymers: PE, PS, PP, and PVC. The polymers with the strongest correlation between K_{ow} and K_D are that of PE and PVC, shown in Figure 8a, with R^2 of 0.71 and 0.68, respectively. These are also the two polymer types that have the simplest chemical structures (Table 1).

Table 1. Structures of the four most studied polymer types throughout the dataset

Plastic Type	Polyethylene	Polystyrene	Polypropylene	Polyvinyl Chloride
Structure				

All four polymers have similar log (K_D) values (order: PS > PE = PP = PVC) for pollutants with log *Kow* of approximately 1.0. However, at log Kow = 6.0, the log (K_D) values were in the order of: PE (contact angle θ = 96°) > PVC (contact angle θ = 85.6°) > PS (contact angle θ = 87.4°) = PP (contact angle θ = 102.1°). This observation implies that hydrophilic compounds may show similar adsorption affinities onto different microplastic types because of the weak intermolecular interactions. On the other hand, more hydrophobic compounds' sorption may differ notably depending on the polymer type. Overall, the log K_D values showed no relationship with polymer's contact angle (θ) with water. This can be attributed to hydrophobic properties of the OCs that may be driving them from water onto microplastic surfaces, but the surface-OC interactions may be weaker in relative terms. However, the variability of data makes achieving a conclusive statement difficult. In fact, none of the R^2 for the data grouped by polymer type are outstanding, as an R^2 of 0.90 or above is generally preferred to make proper conclusions. Additionally, some degree of randomness should be expected, especially when investigating adsorption which may be affected by surface diffusion processes (the result of random Brownian motion). Therefore, a statistical test for randomness was performed in MATLAB on the 770 isotherm results to determine whether the dataset could be considered "random." The test showed that neither the data set as a whole nor the individual polymer groupings can be considered random (*p*-values for all data, PE, PS, PP, and PVC are as follows: $8x10^{-58}$, $1x10^{-42}$, $4x10^{-10}$, $2x10^{-8}$, and $2x10^{-4}$, respectively). The marked differences between the R^2 of each polymer type along with their respective slopes lead us to conclude that the adsorption process is dependent on polymer type, though this is clearly not the only contributing factor.

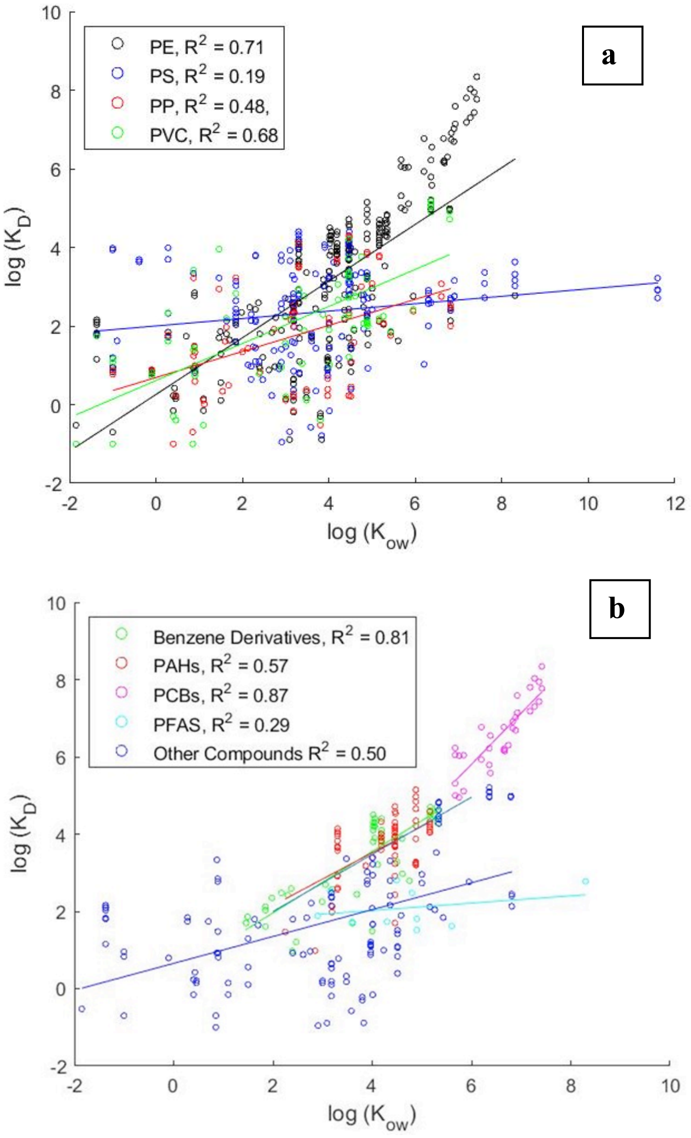


Figure 8. Octanol-water partitioning coefficient, log K_{ow}, versus log K_D: a) grouped by polymer type (polyethylene, polystyrene, polypropylene, and polyvinyl chloride; compiled from 68 papers); b) PE data organized by adsorbate class. Solid lines represent linear regression for points of corresponding color.

Table 2. Correlation coefficients for linear regression analyses of log K_{ow} versus log K_D for polyethylene, organized by compound type.

Group	**Equation: log K_D = a log K_{OW} + b**		**R^2**	**n**
	a	**b**		
Benzene Derivatives	0.80	0.37	0.81	52
PAHs	0.68	0.79	0.57	80
PCBs	1.32	-2.12	0.87	34
PFAS	0.10	1.64	0.29	10
Other	0.35	0.65	0.50	116

2.3.4 Influence of Compound Type

Figure 8 presents a closer look at the correlation between K_{ow} and K_D for the most common plastic type, PE organized by adsorbate class. The same figures for the other three plastic types, PS, PP, and PVC, can be found in the Supplementary Material. The strongest correlations between K_{ow} and K_D are those of the PCBs, the benzene derivatives, and the PAHs at 0.87, 0.81 and 0.57, respectively. These are also the groups that possess the trendlines with the greatest slopes in the same order. PCBs are made up of two linked benzene rings in the biphenyl structure, which can have various degrees of chlorination (Faroon *et al.,* (2015)). Heavy congeners of PCBs are more hydrophobic (Ahmad *et al.,* (2019) and are more likely to partition onto microplastic surfaces, which is supported by the data presented here. PCBs have a flexible bond between benzene rings, and smaller congeners could possibly show an adsorption advantage by configuring themselves according to the adsorbent pore structure (Llorca *et al.,* (2020)). However, there were no observations as such, which is attributed to the non-porous structure of most microplastics (Rozman *et al.,* (2021)). The lack of pores may cause the adsorptive interactions to occur on the external surfaces, and relatively similar and uniform intermolecular interactions indicate that hydrophobicity is a surrogate for their adsorption. For aged microplastics, surface tortuosity and internal porosity could increase the heterogeneity of adsorption sites (Liu *et al.,* 2020). For example, Bhagat *et al.* (2022) observed an increase in phenanthrene (a neutral and hydrophobic compound) adsorption after microplastic aging and attributed this to increased surface roughness. As a result, lighter PCB congeners may also demonstrate an ability to adsorb by adjusting their configuration within the pores of microplastics (Llorca *et al.,* (2020a)). To the authors' knowledge, there are no studies investigating this mechanism of adsorption onto microplastics yet.

The benzene derivatives category is made up of compounds with one benzene ring and one or more functional groups (compounds such as nitrobenzene, trichlorobenzene, and phthalates are included, see the Supplementary Material for the full list of compounds), which are largely non-soluble in water. Similarly, PAHs are composed of fused benzene rings and are largely hydrophobic. While the PAHs have a slope similar to that of the benzene derivatives category, pointing to a possible strong relationship between K_{ow} and K_D, the coefficient of correlation is not strong enough to make conclusive remarks. However, because the benzene derivatives and PAHs are so similar in structure, it seems that hydrophobicity may still be a good predictor of adsorption despite the variability of data sources, water chemistry, and microplastic types--though this should be supported by future work.

The PFAS group has a low coefficient of determination ($R^2 = 0.29$) in addition to a small slope. PFAS are a highly soluble, fluorinated, aliphatic group of compounds (Domingo and Nadal, 2019). Figure 5b shows that a five-magnitude increase in K_{ow} yields less than one magnitude of increase in K_D; clearly, K_{ow} is a poor predictor for PFAS adsorption onto microplastics, possibly indicating that hydrophobicity is not the only intermolecular interaction that takes place. The adsorption mechanism of ionizable PFAS is likely dominated by electrostatic forces acting on them. The study of Wang *et al.* (2015) showed that despite having the same hydrophobic free energy, perfluorooctanesulfonic acid (PFOS) is adsorbed by microplastics less than perfluorooctanesulfonamide (FOSA) due to electrostatic repulsion. Additionally, other variables such as water chemistry, microplastic size, and surface area may have contributed to this poor correlation. Therefore, it may be possible that the clouding of the data produced by the variety of structures of the compounds in the PFAS category prevent K_{ow} from properly indicating adsorption properties. One could argue that hydrophilic and hydrophobic PFAS may have different mechanisms of adsorption that could eventually lead to their partitioning onto microplastics. However, a closer investigation with more data points in uniform solutions would be needed to unravel the predominant intermolecular interactions that control adsorption of PFAS onto microplastics. Salting out of PFAS, active competition between competing anions vs. ionized PFAS as well as cation bridging between negatively charged surface functional groups and PFAS could influence the impact of water chemistry on different PFAS types (Kazwini *et al.,* (2022)). Particularly longer chain PFAS are expected to demonstrate more hydrophobic interactions per net charge while short chain molecules may experience stronger electrostatic interactions (Joo *et al.,* (2021)). In addition, the electronegative C-F bond creates a partial negative "jacket" around the hydrophobic chain of the PFAS further complicating the interactions (Xiao *et al.,* (2011)). A systematic study of PFAS adsorption onto well-characterized microplastics in simple or controlled aquatic conditions would be important to reveal some of the adsorption mechanisms.

2.3.5 Influence of Water Chemistry

In addition to adsorbate and adsorbent characteristics, the aquatic background could also play a significant role on adsorption of OC by microplastics. In the database studied here, the four dominant water type categories were freshwater, synthetic freshwater, purified water (including nanopure, distilled, and lab-grade water), seawater, and synthetic seawater, though many papers did not specify water type used. Of the water type categories, natural freshwater (99 isotherms), natural seawater (72 isotherms), purified water (427 isotherms), synthetic freshwater (89 isotherms) and synthetic seawater (65 results), were studied. In general, the adsorption trends of natural freshwater and synthetic freshwater were similar, though the freshwater adsorption was slightly greater than the synthetic freshwater (Appendix). A similar tendency is observed with the natural seawater and synthetic seawater. A possible explanation for this phenomenon is that the real-world environmental waters contain concentrations of organic matter or ionic compounds that may contribute to polymer bridging during the adsorption process (Brewer *et al.,* (2021)), increasing OC adsorption. Adsorption of OC in both seawaters (real and synthetic) had higher dependance to K_{ow} than the freshwaters. This shows that hydrophobicity may play a larger role in solutions with higher salt, while at low hydrophobicities, adsorption is greater in freshwaters. It has been reported that the salting-out effect, a phenomenon that decreases the solubility of a compound in the presence of salt concentration, increases hydrophobic interactions between OCs and microplastics (Atugoda *et al.,* (2021)). However, the authors once again point out that the variability of the data makes it difficult to draw concrete conclusions; thus, a more specific analysis was investigated to help elucidate the effect of water chemistry.

Ideally, a side-by-side comparison for adsorption of the same OCs onto the same microplastics under different aquatic backgrounds would be available for analysis. This was not available except for a small set of PFAS data as presented in Figure 9. Aquatic chemistry can influence the adsorption affinity of PFAS onto microplastics in numerous ways, including electrostatic attraction/repulsion, intermolecular interactions, hydrophobic interactions, salting-out effect, cation bridging, and water cluster formation (Fu *et al.*, (2021); Mei *et al.*, (2020)). Figure

9, shows partition coefficients (log K_D) of twelve different PFAS compounds onto PS microplastics in freshwater and seawater. The compound list contains nine perfluorocarboxylic acids (PFCAs - C4, C5, C10, C11, C12, C13, C14, C16, C18), two perfluorosulfonic acids (PFSAs - C4, C10), and perfluorooctane sulfonamide (FOSA - C8). All PFCAs and PFSAs exist in their anionic forms except FOSA, which is protonated (neutral) in the database. This figure demonstrates two significant impacts of aquatic chemistry on the adsorption process. First, all ionized compounds adsorb onto PS more strongly in seawater than in freshwater. This can be assigned to the ionic strength of seawater, associated with two adsorption promoting mechanisms. First, the elevated amounts of ions in seawater can decrease the solubility of PFAS in the aqueous phase and make them sorb onto microplastic surfaces (salting-out) (Joo *et al.,* (2021)). Second, divalent cations in seawater can act as a bridge between the anionic head group of PFAS and the negatively charged PS surfaces (Campos Pereira *et al.,* (2018); Llorca *et al.,* (2018)). The only neutral PFAS, FOSA, acted differently and adsorbed more in freshwater. The second finding of this figure is the overwhelming impact of seawater on hydrophobic interactions. There is an increasing trend in adsorption coefficients (clockwise) with PFAS chain length in freshwater. However, this trend disappears in seawater, which is very likely due to the mechanisms mentioned regarding ionic strength.

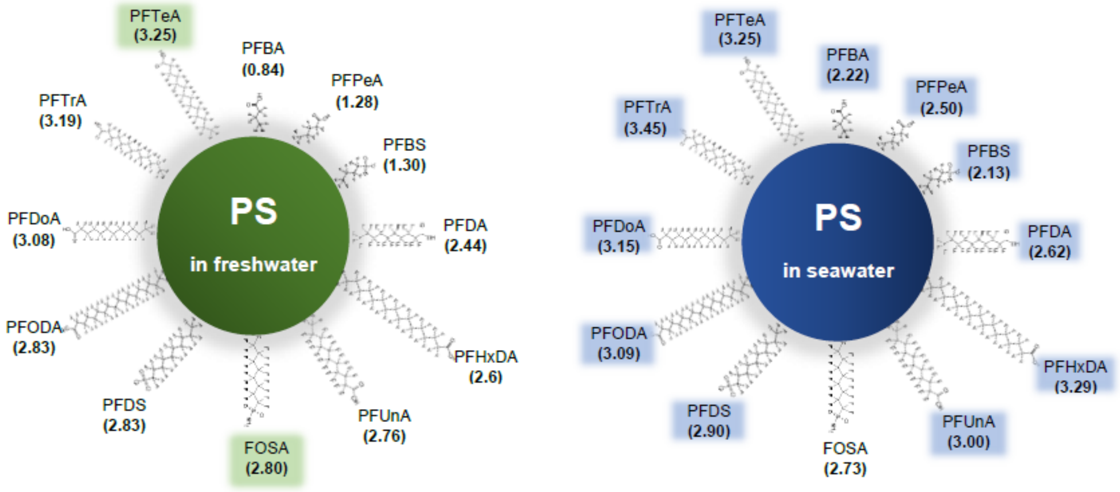


Figure 9. Adsorption of PFAS (log K_D) by PS microplastics. The compounds are ordered in clockwise direction with increasing adsorption in freshwater. The shaded boxes indicate higher adsorption affinity between freshwater and saltwater. (Data from Llorca *et al*., (2018)).

2.3.6 Influence of Polarity

To further investigate the controlling factor in the adsorption process, the results of experiments using five benzene derivative compounds adsorbed to both polyethylene and polystyrene were compared. The compound's average log (K_D) value for each polymer was plotted against the dipole moment of the compound (Figure 10). For the 5 aromatic compounds PS always exhibited stronger adsorption affinity than PE. This could be attributed to the π- π interactions between aromatic compounds and the phenyl ring in the PS structure. A study modeling the interactions between OCs and PS microplastics by Uber *et al.* (2019) also showed the impact of the π- π interactions. The weak negative correlation for both PE and PS, with R^2 of 0.49 and 0.61, respectively, indicates that microplastics do not have strong induced electronic interactions with the benzene derivative compounds, and supports the conclusions drawn from Figure 8 that adsorption may be driven by a compound's repulsion from water (hydrophobicity) rather than its non-specific affinity for microplastics. However, more data on compounds with different polar charges and their adsorption onto microplastics is necessary to draw more meaningful conclusions.

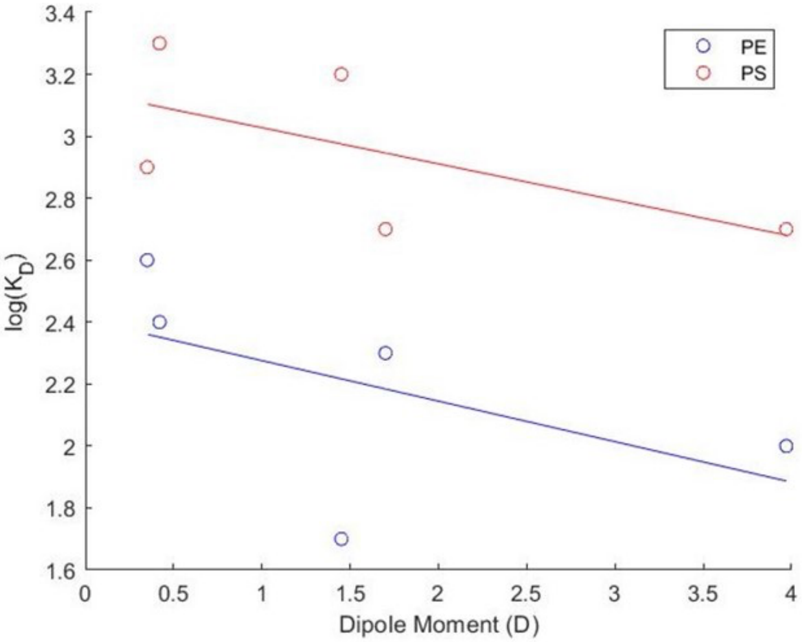


Figure 10. Relationship between polarity vs. log (K_D) for five different compounds on PE and PS.

Table 3. Dipole moments and log (K_D) for five different compounds on PE and PS

Compounds	**log *(K_D)***		**Dipole Moment (D)**
	PE	**PS**	
nitrobenzene	2.0	2.7	3.970
bromobenzene	2.3	2.7	1.700
phenol	1.7	3.2	1.450
n-propylbenzene	2.4	3.3	0.420
m-xylene	2.6	2.9	0.350

2.4 Conclusions

Interactions of OCs with the microplastisphere becomes increasingly important as an additional 8 million tonnes of plastic are discharged into the oceans each year. The environmental fate and transport of OCs could be altered by these interactions and the insights gained could facilitate technology development for OC removal using polymer-based adsorbents. However, our analysis shows that the methodology reported throughout the literature is highly variable, as many researchers have different methods of preparing their microplastics before testing. Additionally, many papers fail to report all important physicochemical characteristics of tested microplastics. Reporting of adsorption results is also non-standardized. Thus, there is a need for more uniform methodology and reporting across the literature, particularly with respect to particle size, surface area, water type, and adsorption isotherm parameters of the respective microplastics.

The adsorption analysis indicated that adsorption is likely dependent on polymer type, though the quantitative effects of each polymer type could not easily be determined from the variable results gathered for this study. Water types used in each test may also contribute to adsorption through the salting-out effect (decreasing solubility), and through potential polymer bridging (increasing adsorption capacity). The variability in these results may be attributed to the microplastic surface tortuosity, complexity, size, or surface area, affecting their adsorption of OCs and their predictability with K_{ow}. Finally, comparing the partitioning of a small subset of aromatic OCs on PE and PS microplastics with regards to their dipole moments showed that OCs do not have notable non-specific attraction or induced electronic interactions to microplastic surfaces, and supported the conclusion that adsorption is most likely driven by a compound's repulsion from water rather than its affinity for microplastics.

As the literature expands, studies on nanoplastic interactions with OCs will become more widespread. Further analysis into this topic may show how nano-scale interactions differ from the micro-scale. Additionally, controlling factors in the adsorption process may be different in weathered, aged, or bio-microplastics. Overall, this study has provided the groundwork for new research into the interactions between microplastics and organic compounds in the environment.

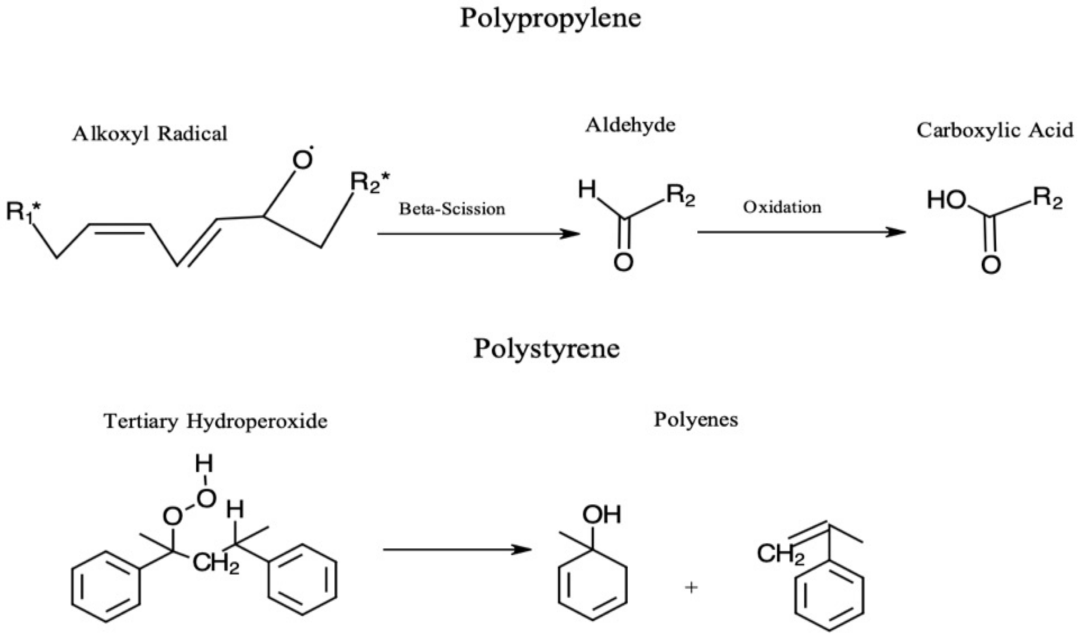
CHAPTER II(B): MICROPLASTIC DEGRADATION PATHWAYS

2.5 Abiotic Degradation Methods

2.5.1 UV Irradiation

UV light is electromagnetic radiation within the wavelengths of 100-400 nm. At most, ~10% of sunlight is comprised of UV light, with ~95% of that being UV-A (315-400 nm) and ~5% being UV-B (280-315 nm) and UV-C (100-280 nm) spectrums. Wavelengths within the UV-A is chosen to act as simulated sunlight most frequently as it creates the most environmentally realistic analysis (Coohil and Sagripanti (2009); Santos *et al.,* (2013); D'Orazio *et al.,* (2013)). Studies investigating how polymers break down under UV irradiation employ polystyrene (PS) as the most common polymer chosen, followed by polypropylene (PP) and polyvinyl chloride (PVC). Due to the polymers being exposed to the atmosphere, when discussing UV irradiation, it is really the photooxidation of the polymer's surface as UV light combines with oxygen to oxidize the polymer's surface (Alimi *et al*., (2022)).

There are three steps to photooxidation: initiation, propagation, and termination. Initiation is the creation of a free radical due to photons being absorbed by chemical bonds through chain scission, cross-linking, or branching. Cross-linking is the attachment of functional groups to the polymer chain where once free radicals form, the presence of oxygen allows these radicals to form hydroperoxyl radicals (ROO·). Branching is the attachment of a polymer side chain with the backbone. Through hydrogen abstraction, an H atom gets removed from the polymer chain, creating a hydroperoxide (ROOH). Hydroperoxides can absorb UV irradiation 300-500x greater than a pure polymer, resulting in the rapid oxidation of a polymer's surface. Termination only occurs when the polymer chain cross-links to a non-radical product, but within the natural environment termination is unlikely and it is why plastics continually degrade from the macro scale into the micro and nano sizes when undergoing photooxidation (Min *et al.,* (2020); Gewert *et al.,* (2015); Alimi *et al*., (2022)). The majority of synthetic polymers are photo-inert to wavelengths >290 nm, but due to the majority of UV light in the UV-B and UV-C spectrums being absorbed by ozone, UV-A light was still most applicable for environmental relevance. This is resulting in photo degradation mostly occurring below 290 nm where light absorbs onto additives or the impurities added during manufacturing (Feldman, (2002)). Once initiation occurs, the polymer surface can oxidize regardless of wavelength. Continued UV irradiation can change the morphology of the surface such as fiber stripping, micro crack formation, and fragmentation, which can modify the molecular structure due to the photons breaking the inter and intramolecular surface bonds (Hüffer *et al.,* (2018); Naik *et al.,* (2020)). As the cycle continues, tertiary reactions take place which can further enhance free radical carbon generation. Examples of tertiary reactions between PP and PS is in Figure 11.


Polypropylene

Polystyrene

Figure 11. Tertiary reactions of the photo-oxidation of PP and PS (Geuskens *et al*., (1978)).

A method for comparison of leachate creation is through analyzing the dissolved organic carbon (DOC) of the leachate per intensity applied across the area of the sample. As a result, polystyrene has been found to leach up to 10 mg DOC per mW/cm^2 intensity provided after 24 days (Lee and Hur*.,* (2020)). PS has also been determined to have large crack formation and mass depletion from the polymer surface after 150 days of aging, but since PS is so reactive, it can overload reactive oxygen species (ROS) forming hydroperoxides and hydroxyl radicals it can inhibit further aging of polymers (Zhu *et al.,* (2020)).

Long term aging using higher wavelengths in the UV-A spectrum produce nonlinear aging with increasing breakdown with time (Brandon *et al*., (2016)). Increasing aging time is also more likely to create oxidation conditions, resulting in negative surface charges, with an increase of electrostatic repulsion between microplastics and other colloids (Zhu *et al.,* (2020)). The charged microplastic surfaces are more likely to stay suspended in the water column with minimal aggregation especially in low ionic strength waters. Using shorter UV wavelengths within the UV-C spectrum decreases the time needed to produce similar results to that of UV-A aging (Tian *et al.,* (2019), Liu *et al*., (2021)). UV-B is not used for microplastic-based leachate formation analysis and some work could fill this gap. Overall, for wavelengths across the UV spectrum, increasing intensity (*I*) and exposure time (*T*) increases aging in exponential growth form. Due to the oxidation of the surface of the polymer, the polymer becomes extremely brittle and likely to fragment or form cracks. This has an array of implications such as alterations of adsorptive interactions between aged microplastics and micropollutants/microorganisms in the environment. It also generates fibers and creates smaller polymer fragments with fragmentation allowing for further surfaces that can release DOM with UV aging (Bhagat *et al.,* (2022); Tubić *et al.,* (2020); Zhang *et al.,* (2020); Wang *et al.,* (2020)).

2.5.2 Chemical Additives

The mechanisms formed from UV irradiation are powerful tools at degrading plastics rapidly with time. UV stabilizers are common to reduce the effects of photodegradation, specifically to inhibit free radical generation. There are three types of UV stabilizers (1) absorbers, (2) quenchers, and (3) hindered amine light stabilizers (HALS). Absorbers act by taking the photon energy from UV light and dissipating its energy as heat across the polymer chain. One of the most common absorbers are benzotriazoles. They are organic, hydrophobic compounds having a very low water solubility and high octanol-water partitioning coefficient (log K_{ow}) showing a tendency to partition to suspended particulate matter and sediment within the environment. When benzotriazoles attach to sediment and sink to the sea floor they have long expected half-lives of 250-550 days. This poses a great threat to benthic dwelling organisms as once these stabilizers attach to sediment they persist within the benthic zone in aquatic environments (Apel *et al.,* (2018)); Wick *et al.,* (2016); Allinson *et al.,* (2018)).

Quenchers are additives that stabilize an excited polymer state by decomposing the hydroperoxides preventing bond cleavage, returning the polymer back to ground state. Nickel is the most common quencher used on the market, used mostly in agricultural films. These films are used to increase and secure the yield of agricultural crops as they can: modify soil temperature, limit weed growth, improve crop yield, maintain nutritional value, and stop fermentation processes. HALS come from the amine class where hydrogen atoms of ammonia are replaced by organic groups. HALS work by consuming free radicals through nitroxide radical formation through the Denisov cycle (Hodgson and Coote, (2010)). Amine solutions produce many toxic tertiary products including volatile aldehydes, nitrosamines, and alkylamines that present further ecological health issues (Lu *et al.,* (2016)). HALS are also soluble and can travel for prolonged distances. The use of UV stabilizers is one of the fastest growing industries and there is minimal understanding of how stabilized polymer chains breakdown within the environment. Further research is needed to understand their breakdown pathways as these polymers are ubiquitously found in the natural environment.

2.5.3 Mechanical Abrasion

Polymers in the environment will always undergo physical stressors creating shear force on the polymers. These include wave propagation, turbulence, and wind. The photooxidation of the polymer surface creates a brittle surface, combined with physical abrasion conditions in the environment can breakdown the MPs into fragments. There has been leaching studies analyzing DOM formation from polymers undergoing rapid shaking, simulating turbulent conditions found within the environment. The results show that DOM formation can occur without UV light and just pure shear force (Chen *et al*., (2021); Mortula *et al.,* (2021)). The antagonistic aspect of the environmental complexity is that both forces combine and create rapid DOM formation. Depending on the location of the polymers in the aquatic environment, the MPs can freeze and thaw in sea ice, causing potential fragmentation as well.

2.5.4 Thermal Degradation

Increasing temperatures of the environment around a polymer, causes the polymer to absorb heat through the polymer chain. Depending on the melting point of the polymer, the absorbed heat can begin to form free radicals. With oxygen, the hydroperoxides begin to form and the oxidation begins. The rising temperatures can also increase the rate and the total mass of DOM produced. Cross-linking and branching may shorten or lengthen the polymer chain further causing the MPs to lose tensile strength by causing changing the molecular weight. It could be presumed that plastic waste along the Equator will degrade at faster rates than that of higher or lower latitudes leading to dangers in all locations based on toxic leachate formation or the consumption of microplastics based on latitude.

2.5.5 Other Environmental Factors Influencing the Degradation Pathways

Environmental conditions are essential in determining the rate of degradation including leachate formation. Water temperature and salinity controls the overall density of the water column, causing different polymer types to lie at the surface and varying depths. In salt water, more microplastics will float to that of freshwater. As a result, each polymer will succumb to different degradation pathways depending on where the microplastics lie in the water body including UV irradiation and mechanical abrasion. There mechanisms can be interrupted by other temperature controlled environmental conditions including the production of algae, where the rate of sunlight initiating photooxidation becomes greatly reduced with decreasing sun exposure. Biofouling (discussed in Section 2.6) is seasonally controlled by water temperature increasing or decreasing microbial activity. When microbes degrade organic matter from polymer surfaces, it forces the microplastics to sink regardless of water type (Kooi *et al*., (2017)). Temperature can initiate the thermal degradation and increase the rate of photooxidation.

Salinity effects not only the density, but can also affect the pH, ionic strength, polarity, and dipole moment (Mao *et al*., (2020); Suhrhoof and Böttcher, (2016); Karlsson *et al*., (2018)). Changing pH levels to more acidic and basic levels has been determined to increase leachate production and fiber generation compared to that of neutral pH levels. Changing polarity and dipole moment for different polymer types can increase or decrease leachate production (Mortula *et al.,* (2021)). Salinity is an essential condition that can affect degradation rates and pathways but is extremely variable depending on the polymer type.

The presence of oxygen is essential for photo and thermal degradation. Although in anaerobic conditions, hydrolysis can occur, where warm enough water temperatures allows free radical generation (Alimi *et al*., (2022)). These free radicals can react with water causing the polymer chain to degrade, similar to that of photooxidation. Hydrolysis can occur in aerobic conditions, but is not an essential mechanism for degradation compared to that of anaerobic conditions.

The amount of natural organic matter (NOM) in a water body also creates interactions with microplastics. These interactions include microbes, enzymes, and reactive oxygen species leading to exopolymeric aggregation, where cells are embodied during biofilm development. This meshes a combination of NOM, microplastics, and minerals called marine plastic snow. Environmental conditions such as salinity, redox conditions, and Ca^{2+} concentrations which can enhance exopolymeric aggregation leading to further production of marine plastic snow (Santschi *et al.,* (2021)). NOM in the environment can trap microplastics, creating larger molecules which provide their own dangers to water quality and aquatic organisms.

Environmental conditions can greatly affect the rates of degradation and the important mechanisms depending on the polymer type. There is no safe location for microplastics in the aquatic environment as microplastics that sink, are more likely to adsorb pollutants and cause potential harm to aquatic species, compared to microplastics that float will most likely be transported by wind where it will produce leachate into the water system as it is transported (Rezaei *et al*., (2019)). There are numerous environmental conditions that control the movement, degradation, and effects of microplastics in the aquatic environment with this as a brief overview of the potential connections of environmental conditions and degradation pathways.

2.6 Biotic Degradation Methods

There are five major processes to biodegradation: colonization, biodeteriation, biofragmentation, assimilation, and mineralization. Colonization occurs when microorganisms attach to a polymer surface in the natural environment creating a biofilm across the surface of the polymer, which is the anchoring of these microbial communities. Biofilm formation could be a product of algae, fungus, proteobacteria, and bacteria. Common examples are bacteroidetes, and firmicutes (Rummel *et al.,* (2017)). The organisms begin to eat away at the organic matter known as biodeteriation. Over time with enough organic matter consumed, the microorganisms can enter the polymer and begin to secret enzymes to bind to the polymer chain. Hydrolytic cleavage occurs and they catalyze. Through catalyzation, intermediate products begin to form through biofragmentation, providing carbon-based energy for growth for the microorganisms. Once the consumed material has reached the microbial cells, they mineralize, producing ATP and biomass through aerobic or anaerobic respiration (Ali *et al*., (2021); Shah *et al*., (2008)). In aquatic environments, biodegradation is essential degradation mechanisms for the breakdown of MPs over time.

Biodisintegration is the process of consuming biological material until it has been fully removed from the environment, meaning the microplastics mineralize into carbon dioxide and water. Biodegradable plastics are new areas of manufacturing, where the materials are designed to be consumed through biodisintegration methods. The purpose is to design plastics that degrade through microorganisms, helping to negate the effects of leachate formation and adsorption potential (Ali *et al.,* (2021)). Biodisintegration rarely occurs from synthetic polymers, but as biodegradable polymers continue to grow in market shares, it will become more essential to understand the mechanisms of biodisintegration.

CHAPTER III: MICROPLASTIC LEACHATE FORMATION UNDER UV IRRADIATION

Collins, A., Ateia, M., Bhagat, K., Ohno, T., Perreault, F., and Apul, O. (2022). Microplastic leachate formation under UV irradiation: extent, characteristics, and mechanisms. Manuscript submitted for publication to *Environmental Science*: *Water Research & Technology.*

*Manuscript is accepted with major revisions by the journal during the submission of the thesis but it has been modified to integrate into the thesis.

3.1 Introduction

The overarching goal of this study was to report a compound analysis of microplastic surface characterization comparison for six polymer types under UV aging and to analyze the DOM mixtures leached after UV aging for the first time. This study had four specific objectives: (1) conduct surface analysis using FT-IR, Raman spectroscopy, and XPS to understand the changes in surface characteristics of post UV aged microplastics; (2) determine the effect of polymers types and UV aging conditions for DOM leaching; (3) relate the changes in surface characteristics and the amount of DOM leached concurrently; (4) determine the physiochemical mechanisms leading to DOM production for each polymer type.

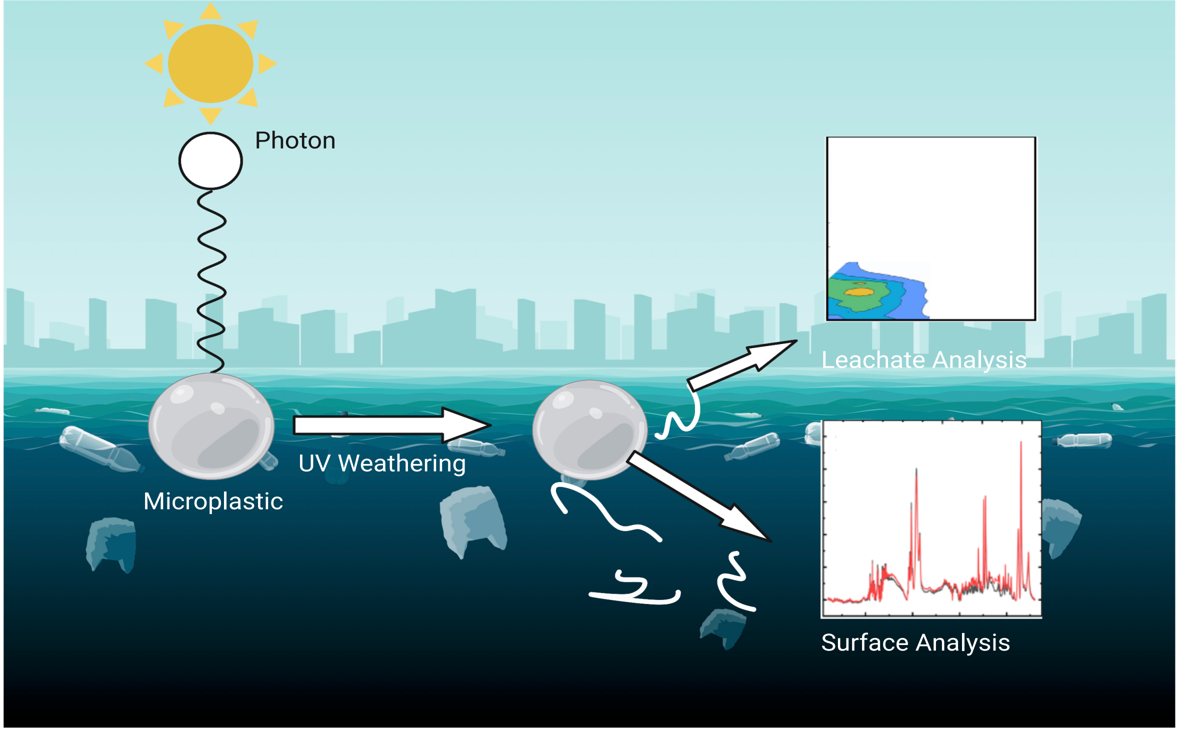


Figure 12. Conceptual model behind experimentation, showing the two important research elements in UV irradiation, the leachate formation and the polymer surface oxidation.

3.2 Materials and Methods

3.2.1 Microplastic Samples and Preparation

Sources: Five types of non-biodegradable (fossil based) microplastic samples with sizes ranging between 3-4 mm were purchased from: TOTAL, Flint Hills Resources, and Verbatim: Polyfil antistatic additive polyethylene resin (PE), black polystyrene plastic pellets (PS_{col}), polyethylene recycled re-grinded shreds (PE_{rec}), polypropylene plastic resin pellets natural injected (PP), and high impact polystyrene plastic pellets (PS). One biodegradable microplastic was purchased from Verbatim: 3D printing material polylactic acid (PLA). The PLA was shipped in tube form and was ground to ~5 mm using a Hamilton Beach coffee grinder. Liquid nitrogen was added to prevent overheating during grinding processes. No visible changes in the microplastic morphology were noted due to liquid nitrogen addition prior to grinding.

Pre-washing process**:** All microplastics were washed for 2 minutes under running tap water and rinsed with distilled and deionized (DDI) water (18.2 MΩ-cm) for 30 s to ensure no residual impurity was left on their surfaces. All DDI water used for cleaning and analysis was stored in glass containers to prevent plastic contamination. The amount of each polymer was left out in the open air for approximately 4 h to ensure the polymers were completely dry before experimentation. Washed and dried microplastic were placed in sealed glass containers and stored in a dark space. Photographs of each microplastic type and their respective sizes are shown in Figure 13.

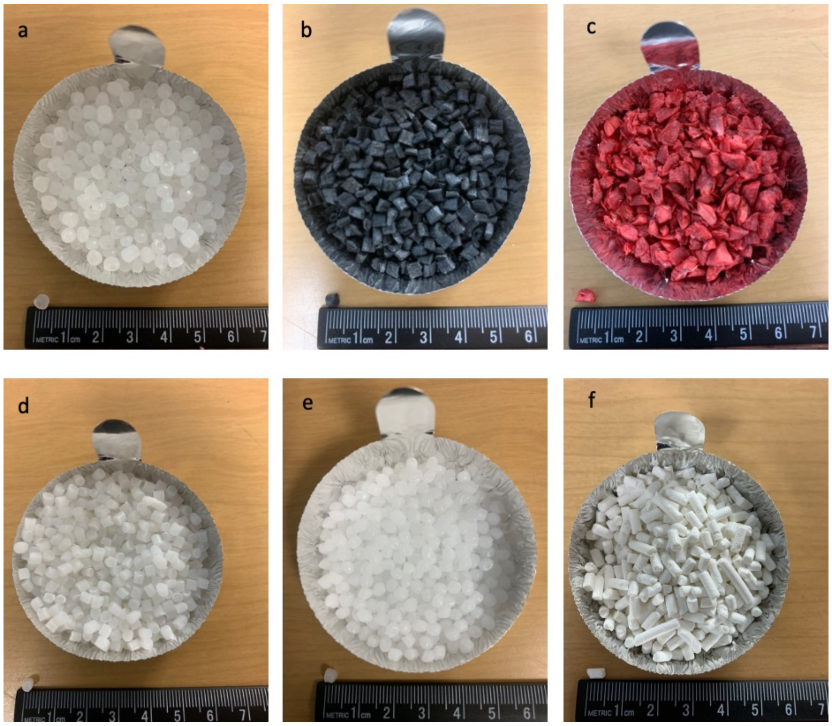
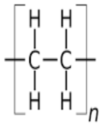
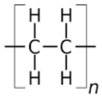
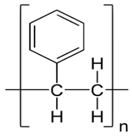
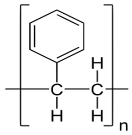
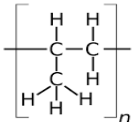
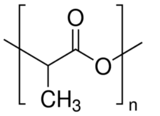


Figure 13. Photographs of each polymer used for analysis: polyethylene (a), polystyrene colored (b), polyethylene recycled (c), polystyrene (d), polypropylene (e), and polylactic acid (f).

The bulk surface characteristics for each polymer are shown in Table 4. The average diameter was found by the average of ten beads for each polymer. The average diameter was used to formulate the surface area and multiplying the surface area by the quantity of beads per one gram of each polymer type, the specific surface area (SSA) for each microplastic was computed ($m^2 \cdot g^{-1}$).

Table 4. Bulk surface characteristics of six polymer microplastics

Microplastic Polymer	**Abbreviations**	**Chemical Formula**	**Molecular Structure**	**Avg. Diameter (mm)**	**Specific Surface Area ($m^2 \cdot g^{-1}$)**
Polyethylene	PE	$(C_2H_4)_n$		3.45 ± 0.33	7.07×10^{-3}
Polyethylene Recycled	PE_{rec}	$(C_2H_4)_n$		3.85 ± 0.63	4.71×10^{-3}
Polystyrene	PS	$(C_8H_8)_n$		3.00 ± 0.00	9.64×10^{-3}
Polystyrene Colored	PS_{col}	$(C_8H_8)_n$		3.60 ± 0.15	3.23×10^{-3}
Polypropylene	PP	$(C_3H_6)_n$		3.35 ± 0.25	1.06×10^{-2}
Polylactic Acid	PLA	$(C_3H_4O_2)_n$		3.45 ± 0.92	8.61×10^{-3}

3.2.2 Characterization of Microplastic Surfaces

A WITec alpha 300 R confocal Raman spectrometer was used for Raman spectroscopy. The spectrometer uses a 532 nm laser with grating at 600 g mm^{-1} and accusation time of 3 s. Raman spectroscopy was used for characterizing both pristine and post UV aged microplastics. FT-IR was performed using a Bruker IFS66V/S FT-IR system equipped with a mercury cadmium telluride standard detector and a KBr beam splitter with a diamond attenuated total reflectance module using a 500 nm laser. Carbonyl indices were calculated for each polymer type using the ratio of absorbance intensity of the carbonyl peak to methyl peak for the detection wavelength of 1716 cm^{-}

^{1} and 720 cm^{-1} respectively. A standard unpaired t-test was performed between pristine and aged carbonyl indices to determine if the variability was considered statistically significant. XPS was conducted using a VG 220i-XL equipped with a monochromated Al K-alpha X-ray source. The data analysis on the carbon and oxygen content was done using the CasaXPS software.

3.2.3 Microplastic Aging Experiments

UV irradiation experiments were completed for each of the six polymer types under identical conditions to determine how much organic matter leaches during UV weathering. A photochemical aging chamber (Rayonet RPR 100) was used with 16 lamps (RPR-3500Å) arranged in a circular pattern emitting light at ~350 nm UV-A light. UV-A light was chosen as it incorporates the most of UV spectrum from natural sunlight (Coohil and Sagripanti (2009); Santos *et al.,* (2013)). The monochromatic intensity of the bulbs produced 12.2 mW cm^{-2} at a distance of 12.5 cm from the center of the quartz beaker in the chamber. Over 24 h, the beads received ~ 0.293 kWh m^{-2} of simulated solar UV irradiance. The total solar irradiance off the coast of the southeastern United States receives ~4.87 kWh m^{-2} per day. Between 5-10% of this is UV-A light, making the solar UV irradiance between 0.244-0.487 kWh m^{-2} of simulated solar UV irradiance showing the microplastics received a similar solar UV irradiance to that in the natural environment off the coast off the southeastern United States (Gewert *et al*., (2018)).

The experiments were conducted using 50 g of each polymer type weighed and placed in into their own 250 mL quartz beaker with 150 mL of DDI water. A magnetic stir rod was added, and the beaker was placed on a stir plate inside the aging chamber. The chamber was turned on and the UV irradiation experiment was carried out for 24 h. To analyze the kinetics, 4 mL samples were taken after 1, 3, and 6 h of UV irradiation. Triplicate samples were analyzed after the completion of 24 h aging for replication of analysis techniques. Three separate experiments were run for PS and PE to ensure consistency in results for each aging experiments. The leftover solutions were poured over a filter into an amber bottle to remove aged microplastic beads from the solution and were left out to dry for 4 h and stored in a sealed glass container in the fridge after drying. Polyethylene was replicated three times to ensure statistical robustness. A FLIR EX-Series thermal camera was used to take a thermal images at the same time intervals as the 4 mL samples to ensure thermal degradation is not a significant factor and all bulbs were functional and emitting light uniformly. Photograph of the aging chamber and an example thermal image are shown in Figure 14.

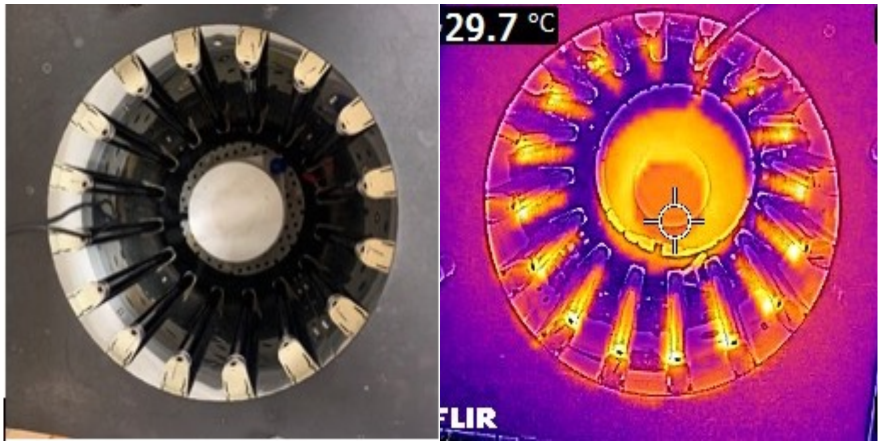


Figure 14. A top view photograph of UV aging chamber with the stir plate inside (left). Example of a thermal image taken at one of the four different time intervals for each experiment (right). Temperature shown is average inside target circle. Thermometer was used for actual temperature analysis as the thermal image was used to ensure consistency of UV intensity.

In addition, PS was tested at two other temperatures to represent a range of environmentally relevant conditions between $\sim 18.3\ ^{\circ}C$ and $\sim 35.0\ ^{\circ}C$. Temperatures were maintained using a New Brunswick Scientific C25KC Incubator Shaker Classic Series. The Rayonet RPR 100 UV aging chamber was placed inside the incubator for experimentation and temperatures were monitored throughout experimentation using a thermometer. Experiments were replicated in triplicates for DOC concentration at 1, 3, 6, and 24 h. Excitation emission (EEM) was conducted after 1 and 24 h. Same UV aging protocol was employed for all temperatures. Finally, PS DOM leachates were compared against natural organic matter (NOM) isolated from Suwannee and Mississippi Rivers obtained from International Humic Substances Society.

Collected water samples (4 mL each and triplicates for 24 h) for each polymer were analyzed via UV absorbance, DOC, and fluorescence through EEM. Excitation emission is a 3D matrix of fluorescence intensity from scanning excitation and emission wavelengths. Intensity is a unitless value, but is directly proportional to concentration, with increasing intensity relating to increasing concentrations of the organic matter present in solution. The analytical tools to characterize the leachate is discussed in the following section. Each polymer was replicated with UV light off in the same chamber for 1 h of stirring conditions with identical microplastic masses and DDI water volumes as the stirring control. The control experiment was conducted to capture the impact of stirring alone on leachate formation. In addition, a dark turbulent shaking experiment was set up with identical microplastic mass and DDI water volumes. In brief, 50 g of each polymer type were placed in a 250 mL amber bottle with 150 mL of DDI water. The amber bottles were secured in a sealed box and placed on a shaker and shaken at 180 rpm for 1 h. For dark turbulent shaking experiment, triplicates were created for each of the six polymer types. The microplastic aging experimental set-up for these dark and light conditions are summarized in Figure 15.

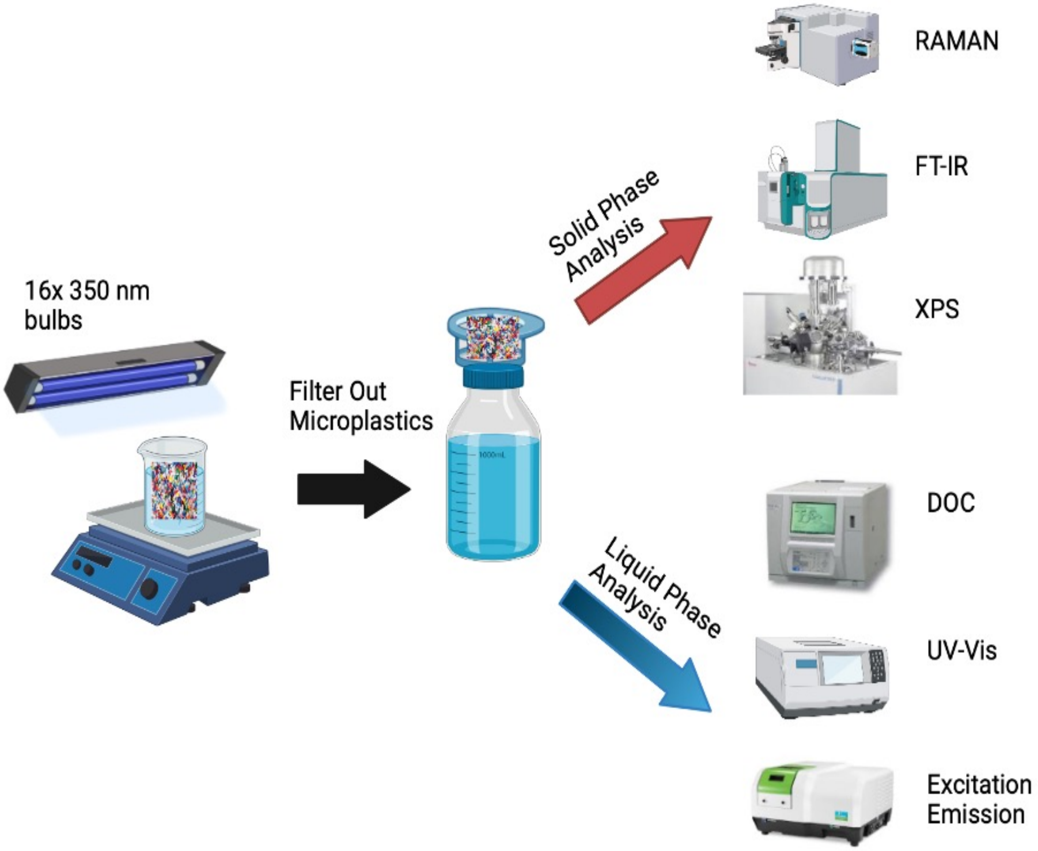


Figure 15. Schematic of experimental process for UV aging experiment

3.3 Results and Discussion

3.3.1 DOM Release

The UV aging released notable amounts of soluble organic matter from all six types of microplastics tested. UV spectroscopy after 1 h and 24 h UV aging and compares the data against dark control conditions (stirring and shaking), at a spectral range from 200-400 nm (Figure 16).

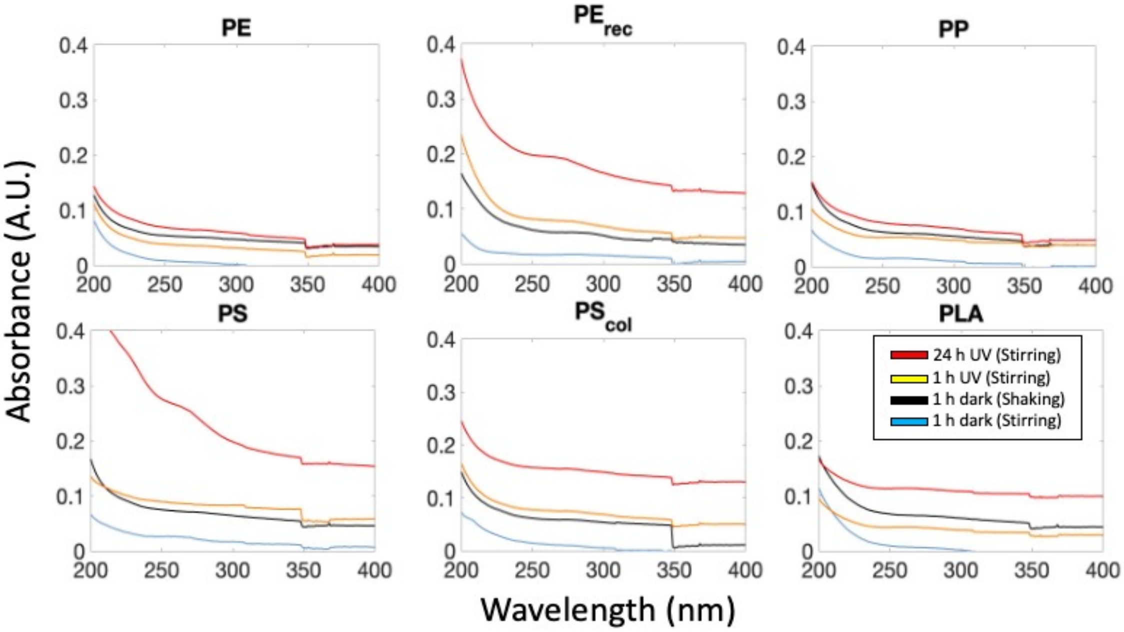


Figure 16. UV absorbance values of leachate solutions for six types of microplastics aged by 1 h and 24 h UV irradiation and associated dark controls.

All tested polymer leachates showed an increase in UV absorbance spectra when 24 h UV irradiation is applied, which was greater than 1 h UV irradiation indicating that the photo-oxidation of microplastic surfaces release dissolved organic matter into solution. The corresponding 1 h dark control conditions (i.e., stirring with no UV) showed no significant increase in UV absorbance of the leachate. However, 1 h shaking conditions released leachate comparable to 1 h UV aging regardless of the microplastic type, which was attributed to mechanical abrasion of surfaces and the subsequent breakdown of microplastics over time (Lee *et al.,* (2020a); Gewert *et al*., (2015)). For PE, PP and PLA, 1 h shaking experiments resulted in even greater DOM release when compared to 1 h UV stirring. Further comparison is shown in Table 5.

Table 5. DOC and UV_{254} for 1 h turbulent conditions and 1 h stirring under dark conditions without UV irradiation (± represents 1 standard deviation)

Microplastic	**DOC Turbulent (mg/L)**	**UV_{254} Turbulent**	**DOC Stirring (mg/L)**	**UV_{254} Stirring**
PS	3.23±0.31	0.0734	0.767	0.0266
PS_{col}	2.23±0.21	0.0601	0.356	0.0133
PE	1.89±0.27	0.0534	0.468	0.0189
PE_{rec}	2.41±0.31	0.0897	0.563	0.0162
PP	1.98±0.34	0.0608	0.422	0.0150
PLA	2.83±0.36	0.0761	0.784	0.0232

This indicates that UV aging is not the only factor generating leachate and in the environment, both turbulence and UV irradiation should be considered for microplastic aging. Especially for microplastic types that are resistant to UV aging, mechanical abrasion could be the predominant mechanism for leachate formation. To understand the leachate formation kinetics, UV_{254} and DOC concentrations for each polymer type were analyzed at 1, 3, 6, and 24-h (Figure 17). The data is further summarized in Figure 18.

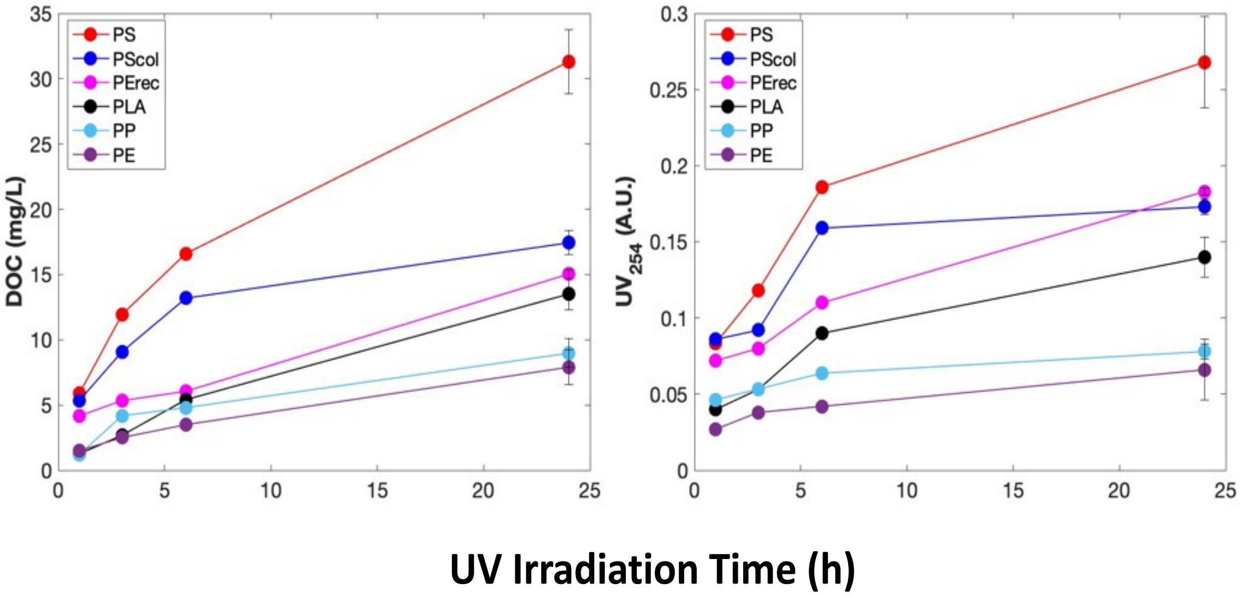


Figure 17. UV_{254} absorbance (top) and DOC concentration in mg/L (bottom), for UV irradiated samples taken at 1,3,6, and 24 h. Error bars represent standard deviation between triplicates. A comparison figure of UV_{254} and DOC results for each polymer after 24 h is presented in (Figure 18).

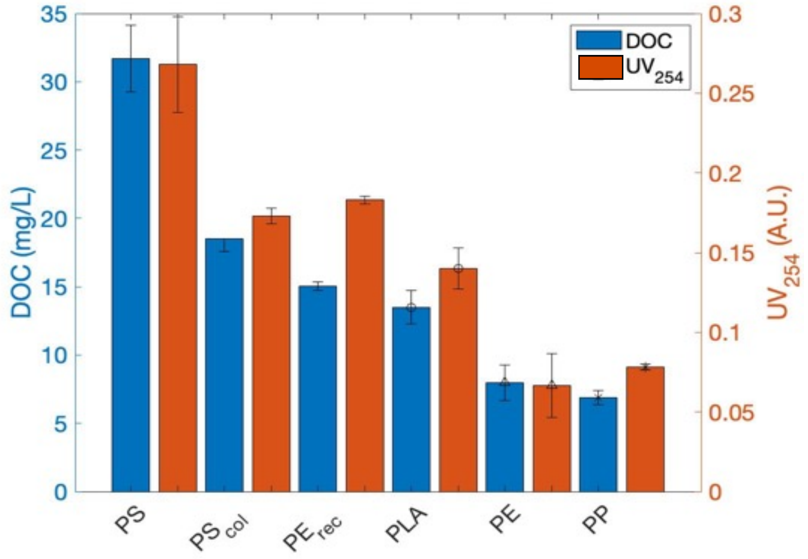


Figure 18. Comparison of UV_{254} and DOC concentrations after 24 h for all six polymers

Consistent with UV spectral analysis, all six polymers had increasing DOC concentrations during 24 h of UV aging. The aromatic hydrocarbons, PS and PS_{col} leached the largest DOC concentrations up to 31.7 mg L^{-1} and 18.5 mg L^{-1} (50 g microplastics in 150 mL DDI). The olefins leached the least, PP and PE, leaching up to 6.97 mg L^{-1} and 7.89 mg L^{-1} after 24 h irradiation. The initial mechanisms of leachate formation are similar between polymer types (Feldman, (2002)). UV irradiation can change the morphology of the surface such as fiber stripping, micro crack formation, and fragmentation, which can modify the molecular structure due to the photons breaking the inter and intramolecular surface bonds (Feldman, (2002); Mao *et al.,* (2020); Hüffer *et al.,* (2018) Naik *et al.,* (2020)). During irradiation, the photooxidation of olefins or aromatics creates three major processes: initiation, propagation, and termination. Initiation is the creation of free radical carbons (R·) due to photons being absorbed by chemical bonds through chain scission, cross-linking, or branching. The presence of oxygen allows these radicals to form hydroperoxyl radicals (ROO·) (Min *et al.,* (2020); Gewert *et al.,* (2015); Alimi et *al.,* (2022)). Through hydrogen abstraction, an H molecule gets removed from the polymer chain, creating a hydroperoxide (ROOH) (see equations 2-4)

$$Polymer \ \rightarrow R\cdot \qquad \text{(eq. 2)}$$

$$R* + O_2 \rightarrow ROOH + R\cdot \qquad \text{(eq. 3)}$$

$$ROOH \rightarrow 2RO\cdot \qquad \text{(eq. 4)}$$

Hydroperoxides can absorb UV irradiation 300-500x greater than a pure polymer (Brandon *et al.,* (2016); ter Halle *et al.,* (2017). Hydroperoxides readily undergo ketone photolysis, which gets divided into Norrish I and Norrish II reactions. Norrish I produces a free radical where the chain remains intact without cleavage. The Norrish II reaction is when chain cleavage occurs allowing the introduction of ketones onto the back of polymer chain. Ketones are able to absorb photons, breaking C-C bonds, and scission the polymer backbone (Feldman, (2002)).

The variation between aromatics and olefins starts with the monomer structure, with aromatics more likely to degrade with time shown due to π bonds creating delocalized electrons and a phenyl ring easily excited by UV light, helping to create rapid radical formation. This allows propagation or branching to occur continuing radical formation and continued degradation (Yousif and Haddad, (2013)). The rapid formation of hydroperoxides and hydroxyl radicals for PS can inhibit aging at a faster rate (Zhu *et al.,* (2020)).

PP degrades due to alkoxyl radical formation undergoing $\beta -$ scission forming an acyl-ketone which is the main source of breaking the backbone of the molecule (Carlsson and Wiles, (1969); Gijsman and Sampers, (1996)). Polyethylene undergoes random cross-linking leading to the formation of many products including ethane, propane, and hexene. This weakens the structure for the polymer, resulting in a brittle structure that is easily fragmented (Gewert *et al*., (2015)). Longer term studies have shown differing results than this study with PP leaching larger DOM concentrations versus PE. This can be attributed to the alkyl groups on the PP polymer chain likely causing faster chain scission of the polymer backbone over time (Lee *et al.,* (2020b); Aslanzadeh and Haghighat Kishm, (2010)).

PE_{rec} leached 15 mg L^{-1} compared to the pristine PE of 8 mg L^{-1} after 24. This is most likely attributed to the introduction polycyclic aromatic hydrocarbons (PAH's) during manufacturing processes through the heating and remolding processes. Phenanthrene is a common compound introduced and can be easily adsorbed to the microplastic surfaces during remolding (Alassali *et al.,* (2020)). The introduction of PAH's on a polymer's surface can create fast radical formation increasing the kinetic rate of PE degradation.

PLA is a biodegradable plastic that is an aliphatic hydrocarbon. It leached up to 13.5 mg L^{-1} after the 24 h of UV irradiation. PLA follows similar oxidation mechanisms: radicals are formed allowing a tertiary hydrogen from PLA chain to form a R·. This radical can react with oxygen to form a peroxide radical followed by hydroperoxides and can undergo scission and begin to breakdown (Gardette *et al.,* (2011); Bocchini and Frache, (2013)). PLA is photo-inert so hydroperoxides also acted as a photo-initiator for surface oxidation.

There are varying kinetic rates of degradation when undergoing the 24 hours of UV irradiation. That results in photons released from the UV bulbs having the ability to break down the intermolecular bonds to create fracturing and the release the DOM from the surface of each polymer type. That means over time the intensity applied from the UV bulbs will create a set DOM mass release. The DOM release for each polymer type was analyzed further using the SSA of each polymer type, the mass of DOM release, and the intensity applied (Figure 19).

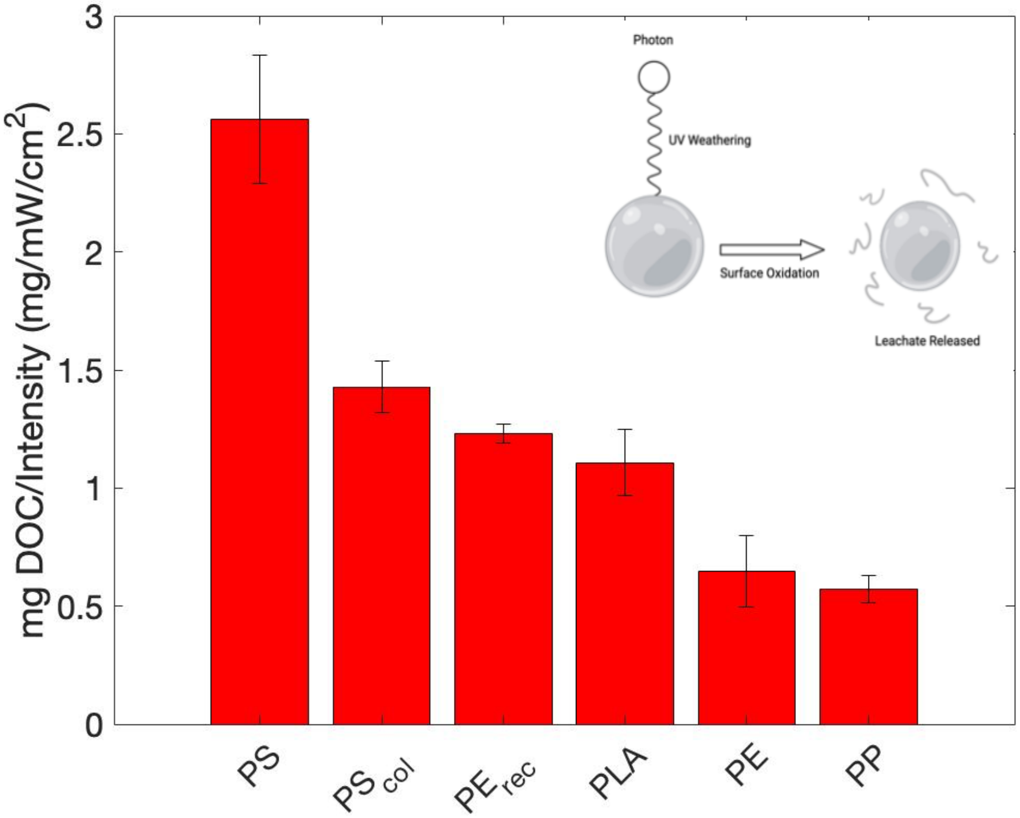


Figure 19. Leached DOC in mg/L per energy intensity (mW/cm^2) produced from UV bulbs.

There was a direct correlation to UV absorbance, DOC, and leached DOC per the intensity produced across all six polymers. Since intensities were checked throughout the 24 h for each compound it can be assumed that similar amounts of photons were hitting the microplastics over the course of the 24 h. This allows an estimation on the amount of DOM leached per photon energy that can be used at various energy spectrums depending on length and intensity of UV irradiation is used.

3.3.2 DOM Characterization

To characterize the DOM from these polymers, fluorescence was measured using excitation (240-400 nm) and emission (300-460 nm) to characterize the leachate based on the sector that fluoresces (Chen *et al.,* (2003)) (Figure 20).

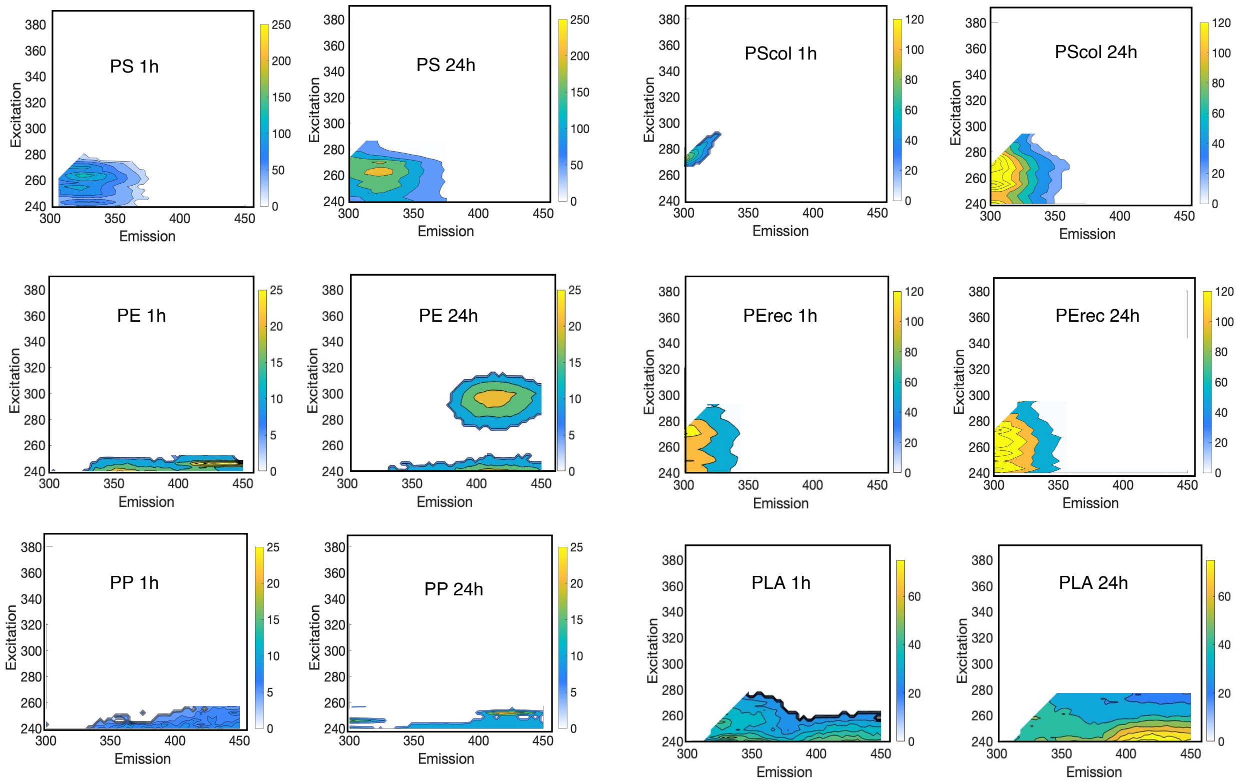


Figure 20. Fluorescence analysis using excitation (240-400 nm) and emission (300-450 nm) after 1 h UV irradiation and 24 h UV irradiation.

Intensity peaks of EEM increased in all six polymers from 1 to 24 h UV aging. PS showed the largest intensity peak, measured in arbitrary units, representing the fluorescence maxima (143-237) followed by PS$_{col}$ (100-182), representing the large PE$_{rec}$ also had a larger increase in peak intensity (248-329) showing the aromatic compounds had highest fluorescence and the introduction of the PAH's for PE_{rec} was theorized to be present through fluorescence analysis with large peaks in the aromatic region and the minor peaks in the humic and fluvic sectors was left as scatter due to the large intensity disparity. The aliphatic microplastics, PP (9.69-24.7) and PE (25.9-34.8) showed much smaller peak intensities in the fulvic acid sector due to the ethylene addition to form each polymer. PLA had the highest intensity for an aliphatic (55.2-97.0) coinciding with their larger DOM concentrations (13.5 mgL^{-1})

3.3.3 Importance of Thermal Degradation

Polystyrene leached the most DOM seen through the highest UV absorption, DOC concentration, and fluorescence intensity. This microplastic was chosen for UV aging comparison at two temperatures to observe leachate production over time. It should be noted that previous experiments were carried out at room temperature, but due to heat from the UV bulbs, enclosed space the temperature that each microplastic endured over the course of the full 24 h was $\sim 26.7\ °C$. To create a comparison at environmentally relevant temperatures w $18.3\ °C$ and $35.0\ °C$ were chosen. The resulting DOC concentrations are in Figure 21.

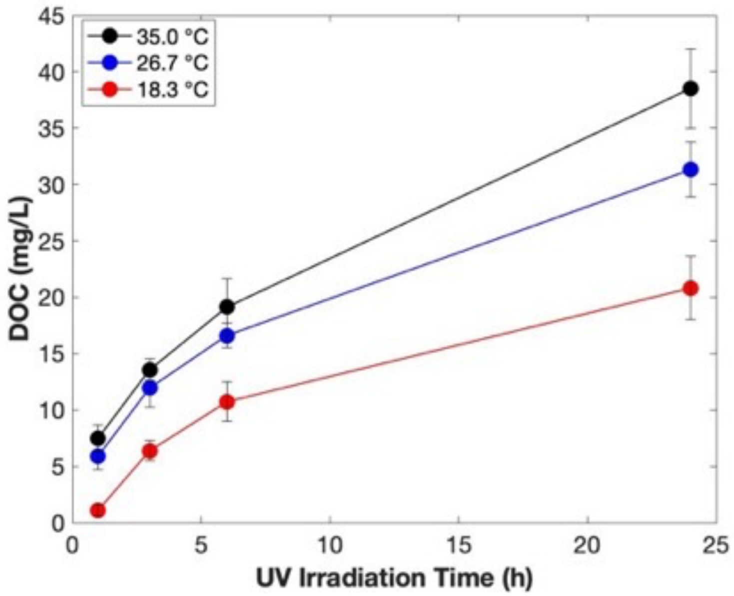


Figure 21. DOC formation kinetics comparison of PS at 26.7 and 18.3 °C after 24 h irradiation

The results show that UV aging depends on the temperature and decreasing the temperature to $\sim 18.3\,°C$ helped reduce the DOC concentration from 32 to 20 mg L^{-1}. Increasing the temperature to $\sim 35.0\,°C$ resulted in DOC concentrations increases from 32 to 37 mg L^{-1}. This indicates that microplastics in warmer environments will degrade faster and the light intensity required decreases as temperature increases providing further evidence that thermal degradation is an essential element in the breakdown of microplastics with time.[3] The kinetic leachate formation constants (k_T) were calculated fitting the formation data to zero order reaction kinetics for each temperature used. Zero-order kinetics provided the best correlation between each kinetic rate order. The following rate constants at each temperature were: $k_{18.3} = 0.84$ mg L^{-1} h^{-1}, $k_{26.7} = 1.06$ mg L^{-1} h^{-1}, and $k_{35.0} = 1.26$ mg L^{-1} h^{-1}. The rate constants were plotted against 1/T using the Arrhenius Equation (eq. 5).

$$k = Ae^{\frac{-E_A}{RT}} \qquad \text{(eq. 5)}$$

and the resulting slope was the minimum activation energy over the real gas constant ($\frac{E_A}{R}$). The minimum energy required for the formation of PS leachate to occur was +18.3 kJ mol^{-1} aligning well with previously reported values.[45] To further show the impact of temperature, the EEM results complemented the DOC results as shown in Figure 22. Peak intensities diminished at the lower temperatures after 24 h (237-94.1) and increased at the higher temperatures (237-387). Microplastics such as PS would degrade at different rates depending on the temperature; however, other microplastic types may need to be evaluated to make a conclusive statement about the overall leachate formation.

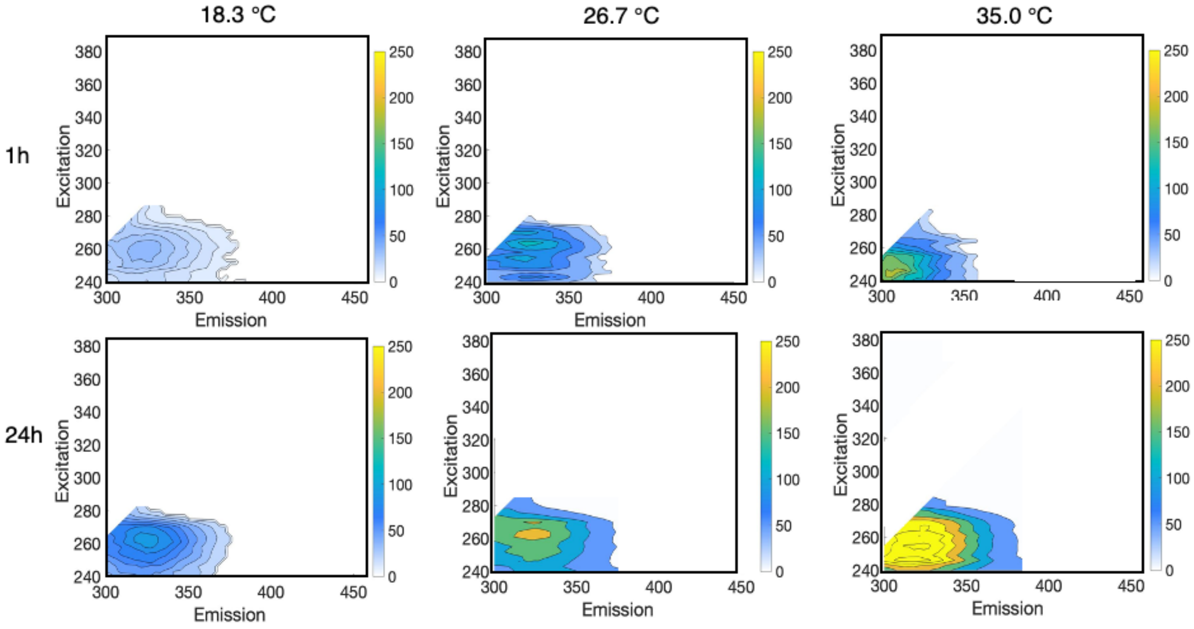


Figure 22. EEM comparison of PS at 35, 26.7 and 18.3 °C after 1 h and 24 h irradiation. It should be noted that different temperature graphs have different intensity ranges to pronounce the changes in EEM.

3.3.4 Characterization of Pristine and Aged Microplastic Surfaces

Raman spectroscopy (Figure 23) and FT-IR (Figure 24) were performed on all six polymers to compare the pristine and post 24 h UV aged surface characteristics. Raman spectroscopy shows relative frequencies to distinguish between molecular bond changes, FT-IR shows absolute frequencies of radiation absorbance to determine functional group vibrations and polar bonds. Each can analyze surface oxidation by the formation of hydroxyl, carboxyl, and carbonyl groups (Gewert *et al*., (2015)). Raman spectroscopy was performed on all six polymers using a 532 nm laser. Two dyed polymers did not have clear signals during Raman spectroscopy analysis (Saviello *et al.,* (2019)); however, the four white polymers (PE, PS, PLA, and PP) presented clear Raman spectral data (Figure 23).

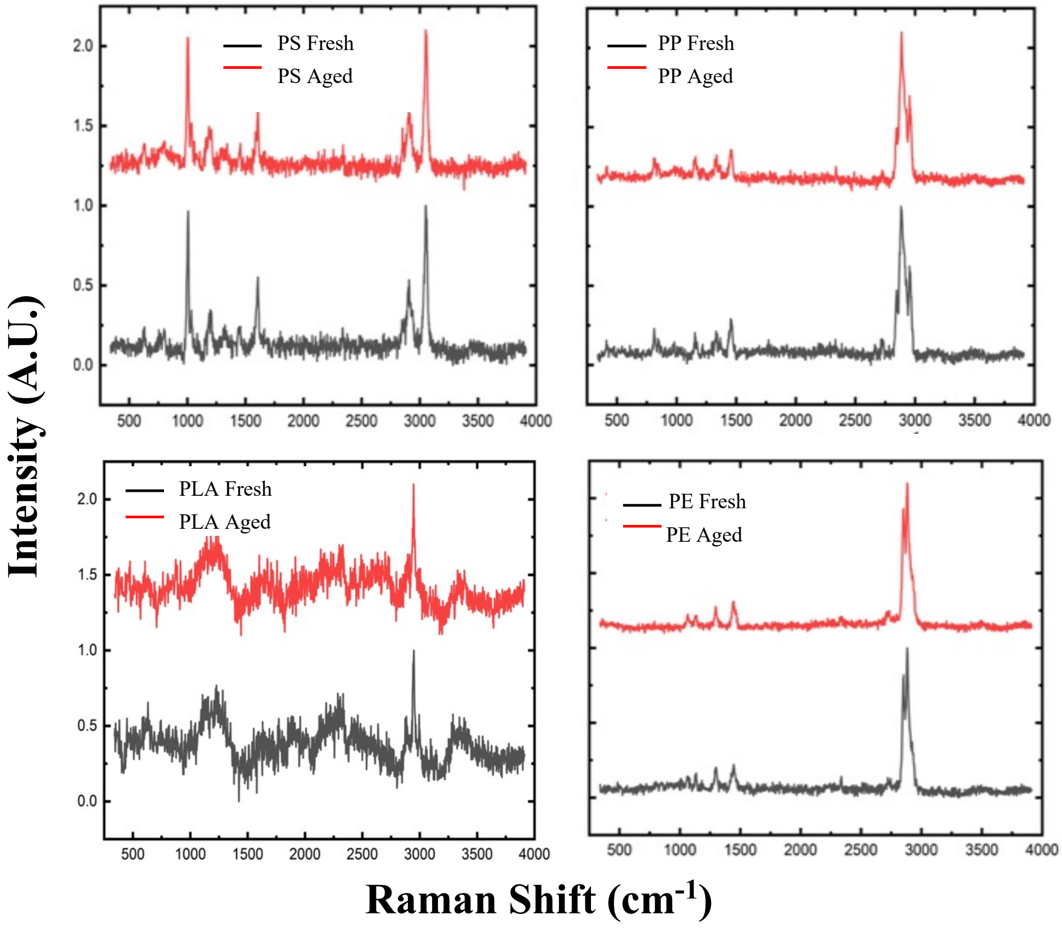


Figure 23. Raman spectroscopy on pristine and 24 h UV aged microplastic polymers with black lines for fresh polymers and red lines for the aged polymers.

The characteristic peaks with decreasing intensity between 1 and 24 h are present for each polymer, PS at 1000 cm^{-1}, PP at 695 cm^{-1}, and PE at 1059 cm^{-1}, presenting evidence for UV aging (Sobhani *et al.,* (2019)). On the other hand, PLA was tested thrice, and the Raman results remained scattered. The scattering may be attributed to the molecular stability of the PLA (Araújo *et al.,* (2013)). The methyl, methylene and methine peaks at wavenumber of 2800-3000 cm^{-1} for all four microplastics becomes attenuated with aging, signaled through CH_2 and CH stretching. The CH_2 bending from 1400-1450 cm^{-1} for PS and peak from 1680-1800 cm^{-1} for PLA signifies that PS and

PLA formed aldehydes from the formation of a carbonyl group post UV aging (Dong *et al.,* (2020)). This occurs due to free radicals causing the chain scission of the C-H bonds of PLA and PS. The symmetry at 1600 cm^{-1} signifies that PP and PE followed similar hydroperoxide formation, resulting in alkene formation post UV aging (Hüffer *et al.,* (2018)).

The FT-IR spectrums and corresponding carbonyl indices for all microplastics are presented in Figure 24. The carbonyl index was calculated from FT-IR by measuring the ratio of carbonyl peaks to reference peaks for each microplastic type. The peaks were determined based and previous literature findings of: PE (1716, 1375 cm^{-1}), PS (1720, 1450 cm^{-1}), PLA (1748, 1451 cm^{-1}), and PP (1712, 1456 cm^{-1}) (Rouillon *et al.,* (2016); Yousif *et al.,* (2012); Medeiros *et al.,* (2018)).

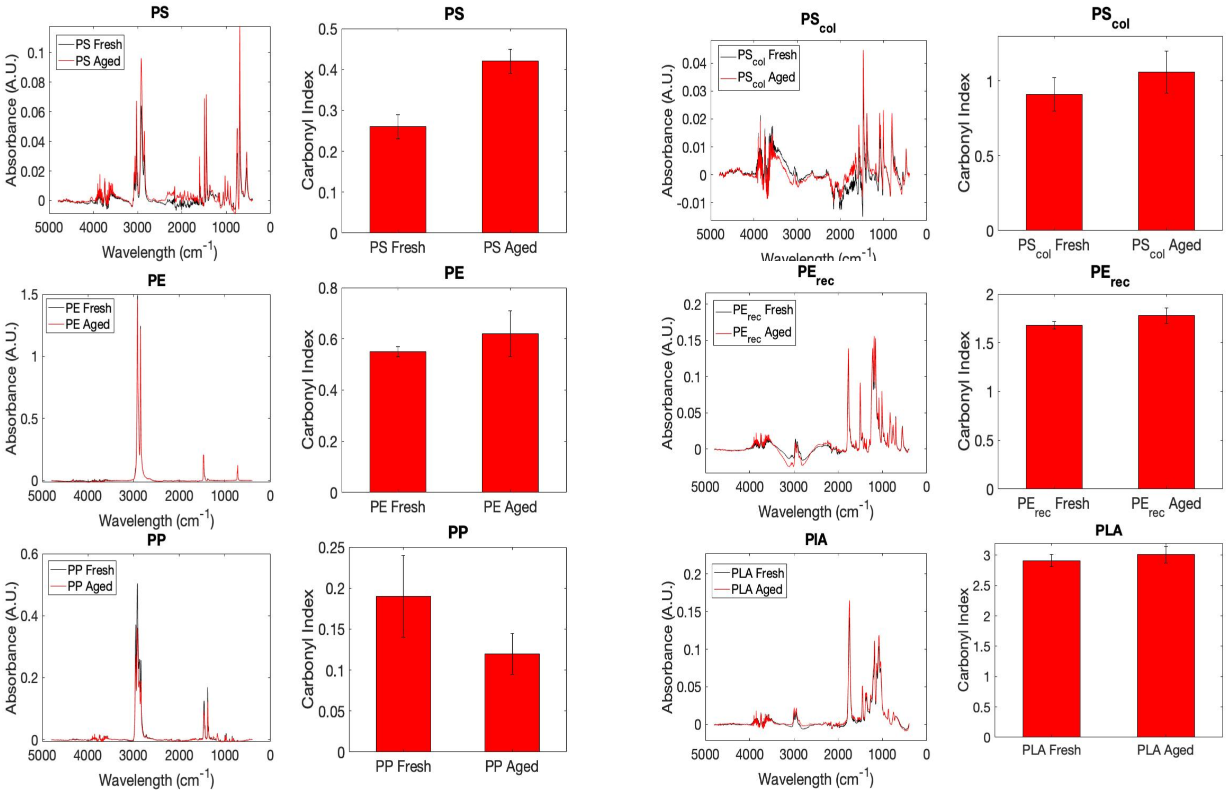


Figure 24. FT-IR absorbance from 4000-1000 cm^{-1} (Left). Carbonyl index with one standard deviation error bars (Right).

Carbonyl groups are also likely in aromatics allowing similar processes to occur of ketone formation from abstracted hydrogen molecules from the hydrogen chain followed by hydroperoxide formation. ROOH's can undergo photolysis forming similar alkoxyl or hydroxyl radicals (Weir and Whiting, (1989); Geuskens *et al.,* (1978)).

FT-IR can further signify UV aging based on the intensity and location of absorbance bands. Decreasing intensities signifies a movement towards micro based structures as the outer layers of the polymers begin to weaken and break off due to the irradiation. There were also characteristic peaks at 1470 cm^{-1} and 720 cm^{-1} for each polymer associated with the C-H bending of CH_2 bonds and rocking mode of CH_3 bonds showing UV aging.[55] Increasing number of peaks also can signify formation of functional groups, where all six polymers, there were increasing number of peaks from 1780-1684 cm^{-1} showing the formation of oxygen containing functional groups such as a carbonyl group, ester formation, or γ −lactones (Hüffer *et al.,* (2018)).

The change in PS was considered statistically significant (p-value < 0.05) after the 24 h UV aging. PP showed no statistical change in carbonyl index between pristine and aged microplastics. The increasing carbonyl index of PS (0.26-0.42) and PS_{col} (0.91-1.07) is due to these polymers being favorable to aging due to the cleavage of weak C-H bonds after chain scission forming stable radicals continuing aging processes (Min *et al.,* (2018)).

In all microplastics (except PP), it was a clear indication that the surfaces of the microplastics were oxidizing. Certain microplastics with aromatic monomers, e.g., PS and $PS_{col,}$ were oxidizing more than the aliphatic ones. To quantify the surface oxidation of each polymer pre and post aging, XPS wide scanning was also conducted, and carbon and oxygen content was quantified. It is a surface sensitive method to analyze elemental percentage on the surface of each microplastics as shown in Table 6.

Table 6. Carbon and oxygen content with carbon to oxygen ratio

Sample	**C (%)**	**O (%)**	**C/O**
PE Fresh	98.4	1.63	60.4
PE Aged	78.0	2.78	28.1
PP Fresh	99.0	0.99	99.0
PP Aged	51.4	4.61	11.1
PLA Fresh	46.1	32.6	1.41

table cont.

PLA Aged	41.6	48.0	0.867
PS Fresh	94.6	5.41	17.5
PS Aged	92.9	7.10	13.1
Recycled PE Fresh	74.5	23.5	3.17
Recycled PE Aged	45.2	16.2	2.79
Colored PS Fresh	62.7	10.4	6.03
Colored PS Aged	59.5	14.4	4.13

Two important trends to analyze the oxidation of a polymer during UV aging processes through XPS, is increasing oxygen percent and decreasing C/O ratio as surface oxidation occurs. For all six polymers there was increasing oxygen content through surface oxidation and decreasing C/O ratio. XPS results differed from FT-IR as PP showed the largest decrease in C/O ratio showing (80%) followed by PE (54%) and PS_{col} (32%). The two largest reductions in C/O ratio were from aliphatic hydrocarbons. The remaining percentages of composition can be attributed to fluorine and neon. Many plastics are fluorinated with fluorine gas to create a surface barrier (Kranz *et al.,* 1994). The introduced neon could best be assumed to occur through the UV-C bulbs, where neon is introduced for germicidal purposes and may leak out overtime. Neon has been found to bond to hydrogen fluoride over time (Losonczy and Moskowitz, (1974)). PLA had a low starting carbon content, but is consistent with values presented within the literature (Hamid *et al.,* (2018); Lednev *et al.,* (2021); Boonmee *et al.,* (2016)).

3.4 Implications on Water Quality

Microplastic DOM release lies within the possible harm to drinking water sources. Many facilities are designed to remove natural organic matter (NOM) from water but are not necessarily equipped for higher aromatic leachate removal as most NOM is comprised of humic and fulvic acids. This difference is demonstrated with EEM comparison of PS, Mississippi River NOM, and Suwannee River NOM at 32 mgL^{-1} (Figure 25).

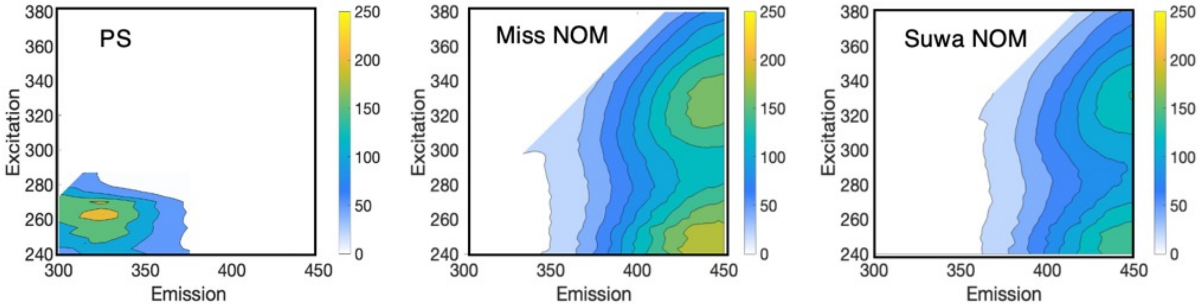


Figure 25. EEM comparison of PS and DOM from Suwanee and Mississippi Rivers at 32 ppm

The inherent difference between the DOM makeup combined with the excessively large surface area of microplastisphere, is an overwhelming new anthropogenic domain that may change the dissolved organic matter concentration in the environment especially in locations where microplastics prevail. The UV irradiation and thermolysis could accelerate the leachate formation and further threaten the ecological health with a potential burden to existing water treatment infrastructure that is dating back to the Victorian era.

Temperature is a key element in the release of DOM. There are numerous other environmental factors that can affect the kinetic rates of DOM release including pH, density, and salinity, effectively increasing or decreasing the rates of DOM release based on polymer type Also, whether a polymer chain has been UV stabilized has a significant impact on the rate of DOM release from UV light, but more studies are needed into how UV stabilized polymers can handle mechanical abrasion, hydrolysis, and biodegradation. Environmental conditions and polymer changes can significantly affect water treatment and the ability to protect human consumption of organic matter over time.

3.5 Conclusions

This study analyzed six different polymer types of microplastics undergoing 24 h UV irradiation. The purpose was to create a compound analysis that combined microplastic surface analysis techniques of FT-IR and Raman Spectroscopy with DOM analysis after the 24 h to understand surface changes and leachate production. The results suggest that aromatic hydrocarbons breakdown much more rapidly from UV irradiation compared to aliphatic hydrocarbons. This was determined through leachate analysis with largest DOC concentrations, peak intensity through EEM analysis, and absorption results. The aromatics also showed the largest Raman intensity shift, carbonyl index increase, and surface oxidation results through XPS. It is clear that aromatic microplastic compounds will break down more rapidly within the environment compared to aliphatic hydrocarbons. Further leachate analysis is needed to determine the toxicity of each leachate produced and the environmental dangers created through varying degradation rates. Comparing whether microplastics that degrade rapidly versus microplastics that can persist within the environment is also an area of further research to determine benefits and costs of persistence and degradation of microplastics within aquatic systems.

CHAPTER IV: SUMMARY OF FINDINGS AND FURTURE RESEARCH

4.1 Important Findings of Two-Way Surface Interactions

The first goal of the thesis was to: (1) survey, summarize, and analyze the literature in terms of adsorption of organic compounds by microplastics. It was indicated that adsorption is heavily dependent on polymer type. Each type of MP has changing tortuosity, complexity, size, shape, and surface area, affecting their adsorption of OCs and their predictability with K_{ow}. The water type is also essential in adsorption potential, where salt water is likely to lead to a salting-out effect, reducing solubility or polymer bridging which increases adsorption potential. The polymer type in each water type might have different adsorption potential based on surface characteristics. According to our findings adsorption is speculated to be controlled by a compounds' repulsion from water rather than its affinity to bond to the MP surface. There is also a need for more uniform analysis techniques and data reporting to better represent adsorption using MPs.

The second goal of the thesis was to: (2) determine the effects of polymer types and UV aging conditions for leachate DOM formation. It was found that aromatic polymers will leach at faster rates than that of aliphatic hydrocarbons undergoing UV irradiation. These results show that the chemical structure of each polymer type will have a significant impact on the kinetic rate and mass of DOM formation. This study was conducted in pristine water as it was a fundamental analysis, where in the natural environment there will be numerous other variables that will affect the rate of leachate production.

The third goal of the thesis was to: (3) relate the changes in DOM formation to polymer surface degradation changes concurrently. The results presented showed a direct link between surface oxidation and DOM formation, filling a needed gap within the literature to show that as a polymer oxidizes it is releasing DOM.

The fourth goal was to: (4) determine the physiochemical mechanisms leading to DOM production for each polymer type. A brief overview of the physiochemical mechanisms was presented, where different polymer types follow similar mechanistic pathways. The importance is the initiation of free radical formation. The chemical structure is essential in the rate of free radical generation and likely lead to the significant differences seen in DOM formation between aromatic and aliphatic polymer types.

The fifth goal of the thesis was to: (5) integrate surface degradation into adsorption potential of different polymer types. The literature review combined with the leachate analysis showed that oxidizing polymers increases adsorption potential through the weakening and breaking of a polymer chain. This creates a brittle polymer surface allowing fragmentation and cracks to form. It is important to note that adsorption potential is heavily dependent on the polymer type, organic compound used, and the water chemistry.

There is no simple solution to fix the dangers presented by microplastics. UV irradiation will photooxidize a polymer's surface creating toxic leachate and pathways for pollutants to adsorb to the surface. This creates a balance of microplastic such as polystyrene that degrade rapidly within the environment creating different challanges to water treatment and marine organisms than that of microplastics that can persist for longer periods. Microplastic degradation stems from the overall usage of plastics in the numerous industries they are used. Plastics have necessary uses that currently have less practical replacement, but the unnecessary uses that have replacements such as water bottle usage is where reduction can begin. There are two avenues that can help: legislative and non-legislative. Legislative changes come in the form of taxes and levies, which encourage conscious decision making from consumers. The consumer level is all the way from businesses to a single customer purchasing merchandise. When consumers are forced to pay for purchasing a plastic product, they are less likely to purchase that item. Legislation can work on the national, regional, and municipal levels (Xanthos and Walker, (2017); Schnurr *et al*., (2018)). With increasing social awareness of the dangers of plastics, companies can self-impose phase outs of plastics to improve social image and be a new standard for environmental protection (Molloy *et al*., (2022)).

For companies to pursue these options, there needs to be incentives or better alternatives to move towards that can result in the reduction of plastics within the environment. There are alternatives to single use plastics such as fabric bags can be a replacement for plastic bags at stores, packaging has been made to be compostable and potentially reusable. These alternatives can be pursued if companies believe that the long-term financial benefits outweigh the short-term ones. For example, biodegradable plastics are expensive to produce and require additional farming practices to meet the demand necessary for replacing fossil-based plastics. However, if companies are incentivized enough for using biodegradable plastics, and the farmers are aided in the production of materials for plastics then changes could be made. The issue that consumers face is whether to purchase an item or pass on the purchase for environmental concerns. However, the majority of consumers will still purchase that item, and this has aided in the exponential increase in plastics in the natural environment (Heidbreder *et al*., (2020)). Legislative and non-legislative initiatives can facilitate change and hopefully reduce the amount of plastic in the environment.

4.2 Future Research Recommendations

As plastic continue to degrade from the micro scale into nano sized particles, understanding the interactions of organic contaminants with nanoplastics will be an area of research importance. It can be expected that with the increasing surface area, nano-sized particles may become invisible pollutant filled particles that will be destructive to marine based organisms as well as water treatment methods. If MPs are used for further research, introducing more factors that can affect leachate production and adsorption potential is a need for future research. Changes to the water type, age, and density of polymers, stabilizing the polymer chain, and introducing microbes to experimentation to see how microbial degradation can affect analysis are all essential aspects that need further understanding. Analysis on pristine MPs aging is becoming more well-understood, but within the environment there are so many conditions that can affect analysis that more environmentally realistic studies are needed.

The UV stabilization of plastics is becoming a very large industry, and there is preliminary findings that these additives breakdown in the aquatic environment at faster rates than manufacturing companies suggest. They suggest they can withstand UV irradiation for 10-15 years but may breakdown in the aquatic environment from other methods, releasing new toxic compounds into the natural environment while still providing the same water pollution effects that pristine plastics provide (Apel *et al.,* (2018)). Understanding how these compounds are released and the effect of UV stabilized plastics have in the aquatic environment is becoming a need within the scientific community. MPs are a danger to water quality and marine organisms, through two-way surface interactions. Analysis in understanding the adsorption potential and leachate production will only benefit the process in reducing plastic waste in natural environments and hopefully lead to changes moving forward.

REFERENCES

Abbasi, S., Moore, F., & Keshavarzi, B. (2021). “PET-microplastics as a vector for polycyclic aromatic hydrocarbons in a simulated plant rhizosphere zone.” Environmental Technology and Innovation, 21, 101370. doi: 10.1016/j.eti.2021.101370

Agboola, O.D., & Benson, N.U. (2021). “Physisorption and Chemisorption Mechanisms Influencing Micro (Nano) Plastics-Organic Chemical Contaminants Interactions: A Review.” Frontiers in Environmental Science, 28. doi: 10.3389/fenvs.2021.678574

Ahmad, I., Weng, J., Stromberg, A., Hilt, J.Z., & Dziubla, T.D. (2020). “Fluorescence based detection of polychlorinated biphenyls (PCBs) in water using hydrophobic interactions.” Analyst, 144 (2019): 677-684. doi: 10.1039/C8AN00867A

Alassali, A., Wolfgang C., Evangelos G., and Kerstin K. (2020). “The Degree and Source of Plastic Recyclates Contamination with Polycyclic Aromatic Hydrocarbons.” RSC Advances 10(73):44989–96. doi: 10.1039/d0ra08554e.

Alimi, O. S., Claveau-Mallet, D., Kurusu, R., Lapointe, M., Bayen, S., and Tufenkji. N. (2022). “Weathering Pathways and Protocols for Environmentally Relevant Microplastics and Nanoplastics: What Are We Missing?” Journal of Hazardous Materials 423 9(10): 255-265. doi: 10.1016/j.jhazmat.2021.126955.

Allinson, M., Yutaka, K., Kumiko, K., and Graeme, A. (2018). “Occurrence and Assessment of the Risk of Ultraviolet Filters and Light Stabilizers in Victorian Estuaries.” Environmental Science and Pollution Research 25(12):12022–33. doi: 10.1007/s11356-018-1386-7.

Ali, S.S., Elsamahy, T., Al-Tohamy, R., Zhu, D., Mahmoud, Y.A.G., Koutra, E., Metwally, M.A., Kornaros, M., Sun, J., (2021a). “Plastic wastes biodegradation: Mechanisms, challenges and future prospects.” Sci. Total Environ. 780: 146590. doi: 10.1016/j.scitotenv.2021.146590

Amelineau, F., Bonnet, D., Heitz, O., Mortreux, V., Harding, A.M.A., Karnovsky, N., Walkusz, W., Fort, & J., Gremillet, D. (2016). “Microplastic pollution in the Greenland Sea: Background levels and selective contamination of planktivorous diving seabirds.” Environmental Pollution 219: 1131-1139. doi: 10.1016/j.envpol.2016.09.017

Apel, C., Hanna J., and Ebinghaus, R. (2018). “Environmental Occurrence and Hazard of Organic UV Stabilizers and UV Filters in the Sediment of European North and Baltic Seas.” Chemosphere 212: 254–261. doi: 10.1016/j.chemosphere.2018.08.105.

Araújo, A., Botelho, G.L., Silva, M., and Machado, A.V. (2013). “UV Stability of Poly(Lactic Acid) Nanocomposites.” Journal of Materials Science and Engineering B 3(2). doi: 10.17265/2161-6221/2013.02.001.

Aslanzadeh, S., and M. Haghighat Kish. 2010. “Photo-Oxidation of Polypropylene Fibers Exposed to Short Wavelength UV Radiations.” Fibers and Polymers 11(5):710–18. doi: 10.1007/s12221-010-0710-8.

Ateia, M., Ersan, G., Alalm, M.G., Boffito, D.C., and Karanfil, T. (2022). “Emerging Investigator Series: Microplastic Sources, Fate, Toxicity, Detection, and Interactions with Micropollutants in Aquatic Ecosystems – a Review of Reviews.” Environmental Science: Processes & Impacts 24(2):172–95. doi: 10.1039/d1em00443c.

Ateia, M., Kanan, A. and Karanfil, T. (2020). “Microplastics Release Precursors of Chlorinated and Brominated Disinfection Byproducts in Water.” Chemosphere 251. doi: 10.1016/j.chemosphere.2020.126452.

Atugoda, T., Wijesekara, H., Werellagama, D.R.I.B., Jinadasa, K.B.S.N., Bolan, N.S., and Vithanage, M. (2020). “Adsorptive interaction of antibiotic ciprofloxacin on polyethylene microplastics: Implications for vector transport in water.” Environmental Technology and Innovation, 19: 100971. doi: 10.1016/j.eti.2020.100971

Bakir, A., Rowland, S. J., and Thompson, R. C. (2012). “Competitive sorption of persistent organic pollutants onto microplastics in the marine environment.” Marine Pollution Bulletin, 64(12): 2782–2789. doi: 10.1016/j.marpolbul.2012.09.010

Bakir, A., Rowland, S. J., and Thompson, R. C. (2014). “Transport of persistent organic pollutants by microplastics in estuarine conditions.” Estuarine, Coastal and Shelf Science, 140: 14–21. doi: 10.1016/j.ecss.2014.01.004

Bao, Z. Z., Chen, Z. F., Lu, S. Q., Wang, G., Qi, Z., and Cai, Z. (2021). “Effects of hydroxyl group content on adsorption and desorption of anthracene and anthrol by polyvinyl chloride microplastics.” Science of the Total Environment, 790. doi: 10.1016/j.scitotenv.2021.148077

Bao, Z. Z., Chen, Z. F., Zhong, Y., Wang, G., Qi, Z., and Cai, Z. (2021). “Adsorption of phenanthrene and its monohydroxy derivatives on polyvinyl chloride microplastics in aqueous solution: Model fitting and mechanism analysis.” Science of the Total Environment, 764: 142889. doi: 10.1016/j.scitotenv.2020.142889

Barrows, A.P.W., Cathey, S.E., and Peterson, C.W. (2018). “Marine environment microfiber contamination: Global patterns and the diversity of microparticle origins.” Environmental Pollution, 237: 275-284. doi: 10.1016/j.envpol.2018.02.062

Bhagat, K., Barrios, A.C., Rajwade, K., Kumar, A., Oswald, J., Apul, O., and Perreault, F. (2022). “Aging of Microplastics Increases Their Adsorption Affinity towards Organic Contaminants.” Chemosphere 298 (February); 134298 doi: 10.1016/j.chemosphere.2022.134238.

Bocchini, S., and Frache, A. (2013). “Comparative Study of Filler Influence on Polylactide Photooxidation.” Express Polymer Letters 7(5):431–42. doi: 10.3144/expresspolymlett.2013.40.

Boonmee, C., Kositanont, C., and Leejarkpai, T. (2016). “Degradation of Poly (Lactic Acid) under Simulated Landfill Conditions.” Environment and Natural Resources Journal 14(2):1–9. doi: 10.14456/ennrj.2016.8.

Brandon, J., Goldstein, M., and. Ohman, M.D. (2016). “Long-Term Aging and Degradation of Microplastic Particles: Comparing in Situ Oceanic and Experimental Weathering Patterns.” Marine Pollution Bulletin 110(1):299–308. doi: 10.1016/j.marpolbul.2016.06.048.

Brewer, A., Dror, I., and Berkowitz, B. (2021). “The Mobility of Plastic Nanoparticles in Aqueous and Soil Environments: A Critical Review.” Environmental Science and Technology, Water, 2021(1): 48-57. doi: 10.1021/acsestwater.0c00130

Campos Pereira, H., Ullberg, M., Kleja, D. B., Gustafsson, J. P., and Ahrens, L. (2018). “Sorption of perfluoroalkyl substances (PFASs) to an organic soil horizon – Effect of cation composition and pH.” Chemosphere, 207: 183-191. doi: 10.1016/j.chemosphere.2018.05.012

Calsson, D.J., and Wiles. D.M. 1969. “The Photodegradation of Polypropylene Films. III. Photolysis of Polypropylene Hydroperoxides. American Chemical Society. D597(6):597–606.

Chamas, A., Moon, H., Zheng, J., Qiu, Y., Tabassum, T., Jang, J.H., Abu-Omar, M., Scott, S.L., Suh, S., 2020. “Degradation Rates of Plastics in the Environment.” ACS Sustain. Chem. Eng. 8, 3494–3511. doi: 10.1021/acssuschemeng.9b06635

Chen, Q., Qian W., Chao Z., Jiawen Z., Zihang D., and Xu, Q. (2021). “Aging Simulation of Thin-Film Plastics in Different Environments to Examine the Formation of Microplastic.” Water Research 202(July):117462. doi: 10.1016/j.watres.2021.117462.

Chen, S., Tan, Z., Qi, Y., and Ouyang, C. (2019). “Sorption of tri-n-butyl phosphate and tris(2-chloroethyl) phosphate on polyethylene and polyvinyl chloride microplastics in seawater.” Marine Pollution Bulletin, 149(May): 110490. doi: 10.1016/j.marpolbul.2019.110490

Chen, W., Westerhoff, P., Leenheer, J.A., and Booksh, K. (2003). “Fluorescence Excitation-Emission Matrix Regional Integration to Quantify Spectra for Dissolved Organic Matter.” Environmental Science and Technology 37(24):5701–10. doi: 10.1021/es034354c.

Chen, X., Gu, X., Bao, L., Ma, S., and Mu, Y. (2021a). “Comparison of adsorption and desorption of triclosan between microplastics and soil particles.” Chemosphere, 263: 127947. doi: 10.1016/j.chemosphere.2020.127947

Chen, Y., Li, J., Wang, F., Yang, H., and Liu, L. (2021b). "Adsorption of tetracyclines onto polyethylene microplastics: A combined study of experiment and molecular dynamics simulation." Chemosphere, 265: 129133. doi: 10.1016/j.chemosphere.2020.129133

Coohill, T.P., and Sagripanti.J.L (2009). "Bacterial Inactivation by Solar Ultraviolet Radiation Compared with Sensitivity to 254 Nm Radiation." Photochemistry and Photobiology 85(5):1043–52. doi: 10.1111/j.1751-1097.2009.00586.x.

Cui, R., Jong, M. C., You, L., Mao, F., Yao, D., Gin, K. Y. H., and He, Y. (2022). "Size-dependent adsorption of waterborne Benzophenone-3 on microplastics and its desorption under simulated gastrointestinal conditions." Chemosphere, 286(P3): 131735. doi: 10.1016/j.chemosphere.2021.131735

Domingo, J.L., and Nadal, M. (2019). "Human exposure to per- and polyfluoroalkyl substances (PFAS) through drinking water: A review of the recent scientific literature." Environmental Research, 177: 108648. doi: 10.1016/j.envres.2019.108648

D'Orazio, J., Stuart, J., Amaro-Ortiz, A., and Scott, T. (2013). "UV Radiation and the Skin." International Journal of Molecular Sciences 14(6):12222–48. doi: 10.3390/ijms140612222.

Dong, M., Qiaoqiao Z., Xinli X., Chen, W., She, Z., and Luo, Z. (2020). "Raman Spectra and Surface Changes of Microplastics Weathered under Natural Environments." Science of the Total Environment 739:139990. doi: 10.1016/j.scitotenv.2020.139990.

Dong, X., Zheng, M., Qu, L., Shi, L., Wang, L., Zhang, Y., Liu, X., Qiu, Y., and Zhu, H. (2019). "Sorption of Tonalide, Musk Xylene, Galaxolide, and Musk Ketone by microplastics of polyethylene and polyvinyl chloride." Marine Pollution Bulletin, 144(April): 129–133. doi: 10.1016/j.marpolbul.2019.04.046

Elizalde-Velázquez, A., Subbiah, S., Anderson, T. A., Green, M. J., Zhao, X., and Cañas-Carrell, J. E. (2020). "Sorption of three common nonsteroidal anti-inflammatory drugs (NSAIDs) to microplastics." Science of the Total Environment, 715: 136974. doi: 10.1016/j.scitotenv.2020.136974

Enders, K., Lenz, R., Stedmon, C.A., and Nielsen, T.G. (2015). "Abundance, size and polymer composition of marine microplastics ≥ 10 μm in the Atlantic Ocean and their modelled vertical distribution." Marine Pollution Bulletin, 100(1): 70-81. doi: 10.1016/j.marpolbul.2015.09.027

Eriksen, M., Laurent C.M., Lebreton, H.S., Carson, M.T., Moore, C.J. Borerro, J.C., Galgani, F., Ryan, P.J., and Reisser, J. (2014). "Plastic Pollution in the World's Oceans: More than 5 Trillion Plastic Pieces Weighing over 250,000 Tons Afloat at Sea." PLoS ONE 9(12):1–15. doi: 10.1371/journal.pone.0111913.

Erni-Cassola, G., Zadjelovic, V., Gibson, M.I., Christie-Oleza, J.A. (2019). "Distribution of plastic polymer types in the marine environment; A meta-analysis." Journal of Hazardous Materials, 369: 691-698. doi: 10.1016/j.jhazmat.2019.02.067

Fan, X., Gan, R., Liu, J., Xie, Y., Xu, D., Xiang, Y., Su, J., Teng, Z., and Hou, J. (2021). “Adsorption and desorption behaviors of antibiotics by tire wear particles and polyethylene microplastics with or without aging processes.” Science of the Total Environment, 771: 145451. doi: 10.1016/j.scitotenv.2021.145451

Fang, S., Yu, W., Li, C., Liu, Y., Qiu, J., and Kong, F. (2019). “Adsorption behavior of three triazole fungicides on polystyrene microplastics.” Science of the Total Environment, 691: 1119–1126. doi: 10.1016/j.scitotenv.2019.07.176

Faroon, O., and Ruiz, P. (2015). “Polychlorinated biphenyls.” Toxicology and Industrial Health, 32(11): 1825-1847. doi: 10.1177%2F0748233715587849

Feldman, D. 2002. “Polymer Weathering: Photo-Oxidation.” Journal of Polymers and the Environment 10(4):163–73. doi: 10.1023/A:1021148205366.

Fries, E., and Zarfl, C. (2012). “Sorption of polycyclic aromatic hydrocarbons (PAHs) to low and high density polyethylene (PE)." Environmental Science and Pollution Research, 19(4): 1296–1304. doi: 10.1007/s11356-011-0655-5

Frere, L., Paul-Pont, I., Rinnert, E., Petton, S., Jaffre, J., Bihannic, I., Soudant, P., Lambert, C., and Huvet, A. (2017). “Influence of environmental and anthropogenic factors on the composition, concentration and spatial distribution of microplastics: A case study of the Bay of Brest (Brittany, France).” Environmental Pollution, 225: 211-222. doi: 10.1016/j.envpol.2017.03.023.

Fu, L., Li, J., Wang, G., Luan, Y., Dai, W. (2021). “Adsorption behavior of organic pollutants on microplastics.” Ecotoxicology and Environmental Safety, 217. doi: 10.1016/j.ecoenv.2021.112207

Gajst, T., Bizjak, T., Palatinus, A., Liubartseva, S., and Krzan, A. (2016). “Sea surface microplastics in Slovenian part of the Northern Adriatic.” Marine Pollution Bulletin, 113(1-2): 392-399. doi: 10.1016/j.marpolbul.2016.10.031

Galloway, T.S., Cole, M., Lewis, C., (2017). Interactions of microplastic debris throughout the marine ecosystem. Nat. Ecol. Evol. 1, 1–8. doi: 10.1038/s41559-017-0116

Gardette, M., Sandrine, T., Gardette, J.L., Murariu, M., and Dubois, P. (2011). “Photooxidation of Polylactide/Calcium Sulphate Composites.” Polymer Degradation and Stability 96(4):616–23. doi: 10.1016/j.polymdegradstab.2010.12.023.

Geuskens, G.,. Baeyens-Volant, D., Delaunois, G., Lu-Vinh, Q., Piret, W., and David, C. (1978). “Photo-Oxidation of Polymers-I. A Quantitative Study of the Chemical Reactions Resulting from Irradiation of Polystyrene at 253.7 Nm in the Presence of Oxygen.” European Polymer Journal 14(4):291–97. doi: 10.1016/0014-3057(78)90051-4.

Gewert, B.,. Plassmann, M.M., and Macleod, M. (2015). “Pathways for Degradation of Plastic Polymers Floating in the Marine Environment.” Environmental Sciences: Processes and Impacts 17(9):1513–21. doi: 10.1039/c5em00207a.

Gewert, B., Plassmann, M.M., Sandblom, O., and Macleod, M. (2018). “Identification of Chain Scission Products Released to Water by Plastic Exposed to Ultraviolet Light.” Environmental Science and Technology Letters 5(5):272–76. doi: 10.1021/acs.estlett.8b00119.

Geyer, R., Jambeck, J.R., and Law, K.L. (2017). “Production, Use, and Fate of All Plastics Ever Made [Producción, Uso y Destino de Todos Los Plásticos Jamás Fabricados].” Science Advances 3(7):e1700782.

Gijsman, P., and Sampers. J. (1997). “The Influence of Oxygen Pressure and Temperature on the UV-Degradation Chemistry of Polyethylene.” Polymer Degradation and Stability 58(1–2):55–59. doi: 10.1016/s0141-3910(97)00012-8.

Godoy, V., Martín-Lara, M. A., Calero, M., and Blázquez, G. (2020). “The relevance of interaction of chemicals/pollutants and microplastic samples as route for transporting contaminants.” Process Safety and Environmental Protection, 138: 312–323. doi: 10.1016/j.psep.2020.03.033

Gong, W., Jiang, M., Han, P., Liang, G., Zhang, T., and Liu, G. (2019). “Comparative analysis on the sorption kinetics and isotherms of fipronil on nondegradable and biodegradable microplastics.” Environmental Pollution, 254: 112927. doi: 10.1016/j.envpol.2019.07.095

Gui, B., Xu, X., Zhang, S., Wang, Y., Li, C., Zhang, D., Su, L., and Zhao, Y. (2021). “Prediction of organic compounds adsorbed by polyethylene and chlorinated polyethylene microplastics in freshwater using QSAR.” Environmental Research, 197(March): 111001. doi: 10.1016/j.envres.2021.111001

Guo, X., Wang, X., Zhou, X., Kong, X., Tao, S., and Xing, B. (2012). “Sorption of four hydrophobic organic compounds by three chemically distinct polymers: Role of chemical and physical composition.” Environmental Science and Technology, 46(13): 7252–7259. doi: 10.1021/es301386z

Guo, X., Chen, C., and Wang, J. (2019a). “Sorption of sulfamethoxazole onto six types of microplastics.” Chemosphere, 228: 300–308. doi: 10.1016/j.chemosphere.2019.04.155

Guo, X., and Wang, J. (2019). “The chemical behaviors of microplastics in marine environment: A review.” Marine Pollution Bulletin, 142 (2019): 1-14. doi: 10.1016/j.marpolbul.2019.03.019

Guo, X., Pang, J., Chen, S., and Jia, H. (2018). “Sorption properties of tylosin on four different microplastics.” Chemosphere, 209: 240–245. doi: 10.1016/j.chemosphere.2018.06.100

Hai, N., Liu, X., Li, Y., Kong, F., Zhang, Y., and Fang, S. (2020). “Effects of Microplastics on the Adsorption and Bioavailability of Three Strobilurin Fungicides.” ACS Omega, 5(47): 30679–30686. doi: 10.1021/acsomega.0c04787

Hamid, Z.A., Tham, C.Y., and Ahmad, Z. (2018). “Preparation and Optimization of Surface-Engineered Poly(Lactic Acid) Microspheres as a Drug Delivery Device.” Journal of Materials Science 53(7):4745–58. doi: 10.1007/s10853-017-1840-9

Hardin, T. (2021). Plastic: It’s Not All the Same. PlasticOceans. Retrieved from: https://plasticoceans.org/7-types-of-plastic/

Ho, W.K., Law, J.C.F., Zhang, T., and Leung, K.S.Y. (2020). “Effects of Weathering on the Sorption Behavior and Toxicity of Polystyrene Microplastics in Multi-solute Systems.” Water Research, 187. doi: 10.1016/j.watres.2020.116419

Hu, E., Shang, S., Fu, Z., Zhao, X., Nan, X., Du, Y., and Chen, X. (2020). “Cotransport of naphthalene with polystyrene nanoplastics (PSNP) in saturated porous media: Effects of PSNP/naphthalene ratio and ionic strength.” Chemosphere, 245: 125602. doi: 10.1016/j.chemosphere.2019.125602

Hüffer, T., Weniger, A.K., and Hofmann., T. (2018). “Sorption of Organic Compounds by Aged Polystyrene Microplastic Particles.” Environmental Pollution 236:218–25. doi: 10.1016/j.envpol.2018.01.022.

Hüffer, T., and Hofmann, T. (2016). “Sorption of non-polar organic compounds by micro-sized plastic particles in aqueous solution.” Environmental Pollution, 214, 194–201. doi: 10.1016/j.envpol.2016.04.018

Hüffer, T., Wehrhahn, M., and Hofmann, T. (2020). “The molecular interactions of organic compounds with tire crumb materials differ substantially from those with other microplastics.” Environmental Science: Processes and Impacts, 22(1): 121–130. doi: 10.1039/c9em00423h

Hu, J.Q., Yang, S.Z., Guo, L., Xu, X., Yao, T., and Xie, F. (2017). “Microscopic investigation on the adsorption of lubrication oil on microplastics.” Journal of Molecular Liquids, 227: 351–355. doi: 10.1016/j.molliq.2016.12.043

Hu, M., Hou, N., Li, Y., Liu, Y., Zhang, H., Zeng, D., and Tan, H. (2021). “The effect of microplastics on behaviors of chiral imidazolinone herbicides in the aquatic environment: Residue, degradation and distribution.” Journal of Hazardous Materials, 418(May): 126176. doi: 10.1016/j.jhazmat.2021.126176

Hummel, D., Fath, A., Hofmann, T., and Hüffer, T. (2021). “Additives and polymer composition influence the interaction of microplastics with xenobiotics.” Environmental Chemistry, 18(3): 101–110. doi: 10.1071/EN21030

Jahan, S., Strezov, V., Weldekidan, H., Kumar, R., Kan, T., Sarkodie, S.A., He, J., Dastjerdi, B., Wilson, S.P., (2019). "Interrelationship of microplastic pollution in sediments and oysters in a seaport environment of the eastern coast of Australia." Sci. Total Environ. 695, 133924. doi: 10.1016/j.scitotenv.2019.133924

Jambeck, J.R. (2015). "Plastic waste inputs from land into the ocean." Science, 347(6223): 768-771. doi: 10.1126/science.1260352

Jiang, M., Hu, L., Lu, A., Liang, G., Lin, Z., Zhang, T., Xu, L., Li, B., and Gong, W. (2020). "Strong sorption of two fungicides onto biodegradable microplastics with emphasis on the negligible role of environmental factors." Environmental Pollution, 267. doi: 10.1016/j.envpol.2020.115496

Joo, S.H., Liang, Y., Kim, M., Byun, J., Choi, H. (2021). "Microplastics with adsorbed contaminants: Mechanisms and Treatment." Environmental Challenges 3 (2021): 100042. doi: 10.1016/j.envc.2021.100042

Kanhai, L.D.K., Officer, R., Lyashevska, O., Thompson, R.C., and O'Connor, I. (2017). "Microplastic abundance, distribution and composition along a latitudinal gradient in the Atlantic Ocean." Marine Pollution Bulletin, 115(1-2): 307-314. doi: 10.1016/j.marpolbul.2016.12.025

Karapanagioti, H.K., and Klontza, I. (2008). "Testing phenanthrene distribution properties of virgin plastic pellets and plastic eroded pellets found on Lesvos island beaches (Greece)." Marine Environmental Research, 65 (2008): 283-290. doi: 10.1016/j.marenvres.2007.11.005

Karlsson, T.M., Hassellöv, M., and Jakubowicz. I. (2018). "Influence of Thermooxidative Degradation on the in Situ Fate of Polyethylene in Temperate Coastal Waters." Marine Pollution Bulletin 135(May):187–94. doi: 10.1016/j.marpolbul.2018.07.015.

Kazwini, T., Yadav, S., Ibrar, I., Al-Juboori, R.A., Singh, L., Ganbat, N., Karbassiyazdi, E., Samal, A.K., Subbiah, S., and Altaee, A. (2022). "Updated review on emerging technologies for PFAS contaminated water treatment." Chemical Engineering Research and Design. doi: 10.1016/j.cherd.2022.04.009

Klein, S., Worch, E., and Knepper, T.P. (2015). "Occurrence and Spatial Distribution of Microplastics in River Shore Sediments of the Rhine-Main Area in Germany." Environmental Science and Technology, 49(10): 6070-6076. doi: 10.1021/acs.est.5b00492

Kooi, M., Van Nes, E.H., Scheffer, M., and Koelmans, A.A. (2017). "Ups and Downs in the Ocean: Effects of Biofouling on Vertical Transport of Microplastics." Environmental Science and Technology, 51(14), 7963–7971. doi: 10.1021/acs.est.6b04702

Kranz, G., Lüschen, R., Gesang, T., Schlett, V., Hennemann, O.D., and Stohrer. W.D. (1994). "The Effect of Fluorination on the Surface Characteristics and Adhesive Properties of Polyethylene and Polypropylene." International Journal of Adhesion and Adhesives 14(4):243–53. doi: 10.1016/0143-7496(94)90037-X.

Lan, T., Wang, T., Cao, F., Yu, C., Chu, Q., and Wang, F. (2021). "A comparative study on the adsorption behavior of pesticides by pristine and aged microplastics from agricultural polyethylene soil films." Ecotoxicology and Environmental Safety, 209: 111781. doi: 10.1016/j.ecoenv.2020.111781

Lara, L.Z., Bertoldi, C., Alves, N.M., and Fernandes, A.N. (2021). "Sorption of endocrine disrupting compounds onto polyamide microplastics under different environmental conditions: Behaviour and mechanism." Science of the Total Environment, 796: 148983. doi: 10.1016/j.scitotenv.2021.148983

Lebreton, L., and Andrady. A. (2019). "Future Scenarios of Global Plastic Waste Generation and Disposal." Palgrave Communications 5(1):1–11. doi: 10.1057/s41599-018-0212-7.

Lednev, I., Salomatina, E., Ilyina, S., Zaitsev, S., Kovylin, R., and Smirnova, L. (2021). "Development of Biodegradable Polymer Blends Based on Chitosan and Polylactide and Study of Their Properties." Materials 14(17):1–17. doi: 10.3390/ma14174900.

Lee, H., Shim, W.J., and Kwon, J.H. (2014). "Sorption capacity of plastic debris for hydrophobic organic chemicals." Science of the Total Environment, 470–471: 1545–1552. doi: 10.1016/j.scitotenv.2013.08.023

Lee, Y.K., and Hur, J. (2020). "Adsorption of Microplastic-Derived Organic Matter onto Minerals." Water Research 187:116426. doi: 10.1016/j.watres.2020.116426.

Lee, Yun Kyung, Kathleen R. Murphy, and Jin Hur. (2020a). "Fluorescence Signatures of Dissolved Organic Matter Leached from Microplastics: Polymers and Additives." Environmental Science and Technology 54(19):11905–14. doi: 10.1021/acs.est.0c00942.

Lee, Y.K., Romera-Castillo, K., Hong, S., and Hur. J (2020b). "Characteristics of Microplastic Polymer-Derived Dissolved Organic Matter and Its Potential as a Disinfection Byproduct Precursor." Water Research 175:115678. doi: 10.1016/j.watres.2020.115678.

Li, H., Wang, F., Li, J., Deng, S., and Zhang, S. (2021a). "Adsorption of three pesticides on polyethylene microplastics in aqueous solutions: Kinetics, isotherms, thermodynamics, and molecular dynamics simulation." Chemosphere, 264: 128556. doi: 10.1016/j.chemosphere.2020.128556

Li, J., Zhang, K., and Zhang, H. (2018). "Adsorption of antibiotics on microplastics." Environmental Pollution, 237: 460–467. doi: 10.1016/j.envpol.2018.02.050

Li, S., Ma, R., Zhu, X., Liu, C., Li, L., Yu, Z., Chen, X., Li, Z., and Yang, Y. (2021b). “Sorption of tetrabromobisphenol A onto microplastics: Behavior, mechanisms, and the effects of sorbent and environmental factors.” Ecotoxicology and Environmental Safety, 210: 111842. doi: 10.1016/j.ecoenv.2020.111842

Li, Y., Zhang, H., and Tang, C. (2020). “A review of possible pathways of marine microplastics transport in the ocean.” Anthropocene Coasts, 3(1), 6–13. doi: 10.1139/anc-2018-0030

Li, Z., Hu, X., Qin, L., and Yin, D. (2020). “Evaluating the effect of different modified microplastics on the availability of polycyclic aromatic hydrocarbons.” Water Research, 170: 115290. doi: 10.1016/j.watres.2019.115290

Liu, F., Liu, G., Zhu, Z., Wang, S., and Zhao, F. (2019a). “Interactions between microplastics and phthalate esters as affected by microplastics characteristics and solution chemistry.” Chemosphere, 214: 688–694. doi: 10.1016/j.chemosphere.2018.09.174

Liu, G., Zhu, Z., Yang, Y., Sun, Y., Yu, F., and Ma, J. (2019b). “Sorption behavior and mechanism of hydrophilic organic chemicals to virgin and aged microplastics in freshwater and seawater.” Environmental Pollution, 246: 26–33. doi: 10.1016/j.envpol.2018.11.100

Liu, L., Fokkink, R., and Koelmans, A.A. (2016). “Sorption of polycyclic aromatic hydrocarbons to polystyrene nanoplastic.” Environmental Toxicology and Chemistry, 35(7): 1650–1655. doi: 10.1002/etc.3311

Liu, P., Yanqi S., Xiaowei W., Wang, H., Huang, H., Guo, X., and Gao, S. (2021). “Review of the Artificially-Accelerated Aging Technology and Ecological Risk of Microplastics.” Science of the Total Environment 768:144969. doi: 10.1016/j.scitotenv.2021.144969.

Liu, P., Zhan, X., Wu, X., Li, J., Wang, H., and Gao, S. (2020a). “Effect of Weathering on Environmental Behavior of Microplastics: Properties, Sorption and Potential Risks.” Chemosphere, 242(March 2020). doi: 10.1016/j.chemosphere.2019.125193

Liu, P., Lu, K., Li, J., Wu, X., Qian, L., Wang, M., and Gao, S. (2020b). “Effect of aging on adsorption behavior of polystyrene microplastics for pharmaceuticals: Adsorption mechanism and role of aging intermediates.” Journal of Hazardous Materials, 384(September 2019): 121193. doi: 10.1016/j.jhazmat.2019.121193

Liu, S., Huang, J., Zhang, W., Shi, L., Yi, K., Yu, H., Zhang, C., Li, S., and Li, J. (2022). “Microplastics as a vehicle of heavy metals in aquatic environments: A review of adsorption factors, mechanisms, and biological effects.” Journal of Environmental Management, 302(Part A): 113995.

Liu, X., Shi, H., Xie, B., Dionysiou, D.D., Zhao, Y. (2019c). “Microplastics as Both a Sink and a Source of Bisphenol A in the Marine Environment.” Environmental Science and Technology 2019, 53: 10188-10196. doi: 10.1021/acs.est.9b02834

Liu, X., Zheng, M., Wang, L., Ke, R., Lou, Y., Zhang, X., Dong, X., and Zhang, Y. (2018). “Sorption behaviors of tris-(2,3-dibromopropyl) isocyanurate and hexabromocyclododecanes on polypropylene microplastics.” Marine Pollution Bulletin, 135(July): 581–586. doi: 10.1016/j.marpolbul.2018.07.061

Llorca, M., Ábalos, M., Vega-Herrera, A., Adrados, M.A., Abad, E., and Farré, M. (2020). “Adsorption and desorption behaviour of polychlorinated biphenyls onto microplastics’ surfaces in water/sediment systems.” Toxics, 8(3). doi: 10.3390/TOXICS8030059

Llorca, M., Schirinzi, G., Martínez, M., Barceló, D., and Farré, M. (2018). “Adsorption of perfluoroalkyl substances on microplastics under environmental conditions.” Environmental Pollution, 235: 680–691. doi: 10.1016/j.envpol.2017.12.075

Lončarski, M., Gvoić, V., Prica, M., Cveticanin, L., Agbaba, J., and Tubić, A. (2021). “Sorption behavior of polycyclic aromatic hydrocarbons on biodegradable polylactic acid and various nondegradable microplastics: Model fitting and mechanism analysis.” Science of the Total Environment, 785. doi: 10.1016/j.scitotenv.2021.147289

Losonczy, M., Moskowitz, M.J. and Stillinger. F.H. (1974). “Hydrogen Bonding between Neon and Hydrogen Fluoride.” The Journal of Chemical Physics 61(6):2438–41. doi: 10.1063/1.1682346.

Lusher, A.L., Tirelli, V., O’Conner, I., Officer, R. (2015). “Microplastics in Arctic polar waters: the first reported values of particles in surface and sub-surface samples.” Scientific Reports 5: 14947. doi: 10.1038/srep14947

Lu, Z., Chen, S., and Zhang, J. (2013). “Effect of UV Absorbers and Hindered Amine Light Stabilizers on the Photodegradation of Ethylene-Octene Copolymer.” Journal of Applied Polymer Science 127(2):1135–47. doi: 10.1002/app.37955.

Magadini, D.L., Goes, J.I., Ortiz, S., Lipscomb, J., Pitiranggon, M., and Yan, B. (2020). “Assessing the sorption of pharmaceuticals to microplastics through in-situ experiments in New York City waterways.” Science of the Total Environment, 729: 138766. doi: 10.1016/j.scitotenv.2020.138766

Mao, R., Mengfan L., Yu, X., Wu, R., Yang, X., and Guo, X., (2020). “Aging Mechanism of Microplastics with UV Irradiation and Its Effects on the Adsorption of Heavy Metals.” Journal of Hazardous Materials 393 122515. doi: 10.1016/j.jhazmat.2020.122515.

Ma, Y., Huang, A., Cao, S., Sun, F., Wang, L., Guo, H., and Ji, R. (2016). “Effects of nanoplastics and microplastics on toxicity, bioaccumulation, and environmental fate of phenanthrene in fresh water.” Environmental Pollution, 219: 166-173. doi: 10.1016/j.envpol.2016.10.061

Medeiros, E.S., Oliveira, J.E., Rego, M., Rego, M., and Branco, C. (2018). “Soil Biodegradation of PLA / CNW Nanocomposites Modified with Ethylene Oxide Derivatives 2” . Materials and Methods. 1–6.

Mei, W., Chen, G., Bao, J., Song, M., Li, Y., Luo, C. (2020). “Interactions between microplastics and organic compounds in aquatic environments: A mini review.” Science of the Total Environment, 736. doi: 10.1016/j.scitotenv.2020.139472

Min, K., Cuiffi, J.D., and Mathers, R.T. (2020). “Ranking Environmental Degradation Trends of Plastic Marine Debris Based on Physical Properties and Molecular Structure.” Nature Communications 11(1). doi: 10.1038/s41467-020-14538-z.

Moldoveanu, S. and David, V. (2021). “Chapter 5 – Phase transfer in sample preparation. Modern Sample Preparation for Chromatography (Second Edition).” 2021: 151-190.

Mortula, M.M, Atabay, S. Fattah, K.P., and Madbuly, A. (2021). “Leachability of Microplastic from Different Plastic Materials.” Journal of Environmental Management 294 (November 2020). doi: 10.1016/j.jenvman.2021.112995.

Mu, J., Zhang, S., Qu, L., Jin, F., Fang, C., Ma, X., Zhang, W., and Wang, J. (2019). “Microplastics abundance and characteristics in surface waters from the Northwest Pacific, the Bering Sea, and the Chukchi Sea.” Marine Pollution Bulletin, 143: 58-65. doi: 10.1016/j.marpolbul.2019.04.023

Müller, A., Becker, R., Dorgerloh, U., Simon, F. G., and Braun, U. (2018). “The effect of polymer aging on the uptake of fuel aromatics and ethers by microplastics.” Environmental Pollution, 240: 639–646. doi: 10.1016/j.envpol.2018.04.127

Murphy, K.R., Stedmon, C.A., Graeber, D., and Bro, R. (2013). “Fluorescence Spectroscopy and Multi-Way Techniques. PARAFAC.” Analytical Methods 5(23):6557–66. doi: 10.1039/c3ay41160e.

Naik, R.A., Rowles, L.S., Hossain, A.I., Yen, M.Y., Aldossary, R.A., Apul, O.G., Conkle, J., and Saleh. N.B. (2020). “Microplastic Particle versus Fiber Generation during Photo-Transformation in Simulated Seawater.” Science of the Total Environment 736:139690. doi: 10.1016/j.scitotenv.2020.139690.

Ncube, L.K., Ude, A.U., Ogunmuyiwa, E.N., Zulkifli, R., Beas, I.N. (2020). “An Overview of Plastic Waste Generation and Management in Food Packaging Industries.” Recycling, 6(1). doi: 10.3390/recycling6010012

Pan, Z., Guo, H., Chen, H., Wang, S., Sun, X., Zou, Q., Zhang, Y., Lin, H., Cai, S., and Huang, J. (2019). “Microplastics in the Northwestern Pacific: Abundance, distribution, and characteristics.” Science of the Total Environment, 650, Part 2: 1913-1922. doi: 10.1016/j.scitotenv.2018.09.244

PlasticsEurope. (2021). “Plastics - the Facts 2021: an Analysis of European Plastics Production, Demand and Waste Data.”

Puckowski, A., Cwięk, W., Mioduszewska, K., Stepnowski, P., and Białk-Bielińska, A. (2021). “Sorption of pharmaceuticals on the surface of microplastics.” Chemosphere, 263. doi: 10.1016/j.chemosphere.2020.127976

Qiu, Y., Zheng, M., Wang, L., Zhao, Q., Lou, Y., Shi, L., and Qu, L. (2019). Sorption of polyhalogenated carbazoles (PHCs) to microplastics. Marine Pollution Bulletin, 146(July): 718–728. doi: 10.1016/j.marpolbul.2019.07.034

Razanajatovo, R.M., Ding, J., Zhang, S., Jiang, H., and Zou, H. (2018). “Sorption and desorption of selected pharmaceuticals by polyethylene microplastics.” Marine Pollution Bulletin, 136(September): 516–523. doi: 10.1016/j.marpolbul.2018.09.048

Rezaei, M., Riksen, M.J. P.M., Sirjani, E., Sameni, A., and Geissen, V. (2019). “Wind erosion as a driver for transport of light density microplastics.” Science of the Total Environment, 669, 273–281. doi: 10.1016/j.scitotenv.2019.02.382

Ritchie, H., and Roser, M. (2018). "Plastic Pollution". Our World in Data.” Retrieved from: https://ourworldindata.org/plastic-pollution [Online Resource]

Rochman, C.M., Manzano, C., Hentschel, B.T., Simonich, S.L.M., and Hoh, E. (2013). “Polystyrene plastic: A source and sink for polycyclic aromatic hydrocarbons in the marine environment.” Environmental Science and Technology, 47(24): 13976–13984. doi: 10.1021/es403605f

Rouillon, C., Bussiere, P.O., Desnoux, E., Collin, S., Vial, C., Therias, S., and Gardette. J.L. (2016). “Is Carbonyl Index a Quantitative Probe to Monitor Polypropylene Photodegradation?” Polymer Degradation and Stability 128:200–208. doi: 10.1016/j.polymdegradstab.2015.12.011

Rozman, U., and Kalcikova, G. (2022). “Seeking for a perfect (non-spherical) microplastic particle – The most comprehensive review on microplasgtic laboratory research.” Journal of Hazardous Materials, 424 Part C: 127529. doi: 10.1016/j.jhazmat.2021.127529

Rozman, U., Turk, T., Skalar, T., Zupančič, M., Korošin, N.C., Marinšek, M., Olivero-Verbel, J., and Kalčíková, G. (2021). “An Extensive Characterization of Various Environmentally Relevant Microplastics – Material Properties, Leaching and Ecotoxicity Testing.” Science of the Total Environment 773. doi: 10.1016/j.scitotenv.2021.145576.

Rubin, A.E., and Zucker, I. (2022). “Interactions of microplastics and organic compounds in aquatic environments: A case study of augmented joint toxicity.” Chemosphere, 289 (2022): 133212. doi: 10.1016/j.chemosphere.2021.133212

Rummel, C.D., Jahnke, A., Gorokhova, E., Kühnel, D., and Schmitt-Jansen. M. (2017). “Impacts of Biofilm Formation on the Fate and Potential Effects of Microplastic in the Aquatic Environment.” Environmental Science and Technology Letters 4(7):258–67. doi: 10.1021/acs.estlett.7b00164.

Santos, A.L., Oliveira, V., Baptista, I., Henriques, I., Gomes, N.C.M., Almeida, A., Correia, A., and Cunha, A. (2013). “Wavelength Dependence of Biological Damage Induced by UV Radiation on Bacteria.” Archives of Microbiology 195(1):63–74. doi: 10.1007/s00203-012-0847-5.

Santschi, P.H., Chin, W., Quigg, A., Xu, C., Kamalanathan, M., and Lin, P. (2021). “How does Natural Organic Matter (NOM) affect Micro- and Nano- Plastic Pollution in the Environment ? - The biophysical Mechanisms leading to the formation of ‘ Marine Plastic Snow’” Universal Journal of Engineering Mechanics. 9(1):32–42.

Sarkar, A.K., Rubin, A.E., and Zucker, I. (2021). “Engineered Polystyrene-Based Microplastics of High Environmental Relevance.” Environmental Science and Technology 55(15): 10491-10501. doi: 10.1021/acs.est.1c02196

Saviello, D., Trabace, M., Alyami, A., Mirabile, A., Baglioni, P., Giorgi, R., and Iacopino, D. (2019). “Raman Spectroscopy and Surface Enhanced Raman Scattering (SERS) for the Analysis of Blue and Black Writing Inks: Identification of Dye Content and Degradation Processes.” Frontiers in Chemistry 7(October):1–9. doi: 10.3389/fchem.2019.00727.

Scutariu, R.E., Puiu, D., Nechifor, G., Niculescu, M., Pascu, L.F., and Galaon, T. (2019). “In vitro sorption study of some organochlorine pesticides on polyethylene terephthalate microplastics.” Revista de Chimie, 70(12): 4620–4626. doi: 10.37358/RC.19.12.7803

Seidensticker, S., Grathwohl, P., Lamprecht, J., and Zarfl, C. (2018). “A combined experimental and modeling study to evaluate pH-dependent sorption of polar and non-polar compounds to polyethylene and polystyrene microplastics.” Environmental Sciences Europe, 30(1): 1–13. doi: 10.1186/s12302-018-0155-z

Shah, A.A., Hasan, F., Hameed, A., Ahmed, S., (2008). “Biological degradation of plastics: A comprehensive review.” Biotechnol. Adv. 26, 246–265. doi: 10.1016/j.biotechadv.2007.12.005

Shan, J., Wang, J., Zhan, J., Liu, L., Wu, F., and Wang, X. (2020). “Sorption behaviors of crude oil on polyethylene microplastics in seawater and digestive tract under simulated real-world conditions.” Chemosphere, 257: 127225. doi: 10.1016/j.chemosphere.2020.127225

Singh, B., and Sharma, N. (2008). “Mechanistic Implications of Plastic Degradation.” Polymer Degradation and Stability 93(3):561–84. doi: 10.1016/j.polymdegradstab.2007.11.008.

Sobhani, Z., Amin, M.D.A., Naidu, R., Megharaj, M., and Fang, C. (2019). “Identification and Visualisation of Microplastics by Raman Mapping.” Analytica Chimica Acta 1077:191–99. doi: 10.1016/j.aca.2019.05.021.

Song, X., Wu, X., Song, X., Shi, C., and Zhang, Z. (2021). “Sorption and desorption of petroleum hydrocarbons on biodegradable and nondegradable microplastics.” Chemosphere, 273: 128553. doi: 10.1016/j.chemosphere.2020.128553

Suhrhoff, T.J., and Scholz-Böttcher. B.M. (2016). “Qualitative Impact of Salinity, UV Radiation and Turbulence on Leaching of Organic Plastic Additives from Four Common Plastics - A Lab Experiment.” Marine Pollution Bulletin 102(1):84–94. doi: 10.1016/j.marpolbul.2015.11.054.

Tanaka, K., Takada, H., Yamashita, R., Mizukawa, K., Fukuwaka, M., and Watanuki, Y. (2013). “Accumulation of plastic-derived chemicals in tissues of seabirds ingesting marine plastics.” Marine Pollution Bulletin, 69(1-2): 219-222. doi: 10.1016/j.marpolbul.2012.12.010

ter Halle, A., Ladirat, L., Martignac, M., Mingotaud, A.F., Boyron, O., and Perez, E. (2017). “To What Extent Are Microplastics from the Open Ocean Weathered?” Environmental Pollution 227:167–74. doi: 10.1016/j.envpol.2017.04.051.

Teuten, E.L., Rowland, S.J., Galloway, T.S., and Thompson, R.C. (2007). “Potential for plastics to transport hydrophobic contaminants.” Environmental Science and Technology, 41(22): 7759–7764. doi: 10.1021/es071737s

Tian, L., Chen, Q., Jiang, W., Wang, L., Xie, H., Kalogerakis, N., Ma, Y., and Ji, R. (2019). “A Carbon-14 Radiotracer-Based Study on the Phototransformation of Polystyrene Nanoplastics in Water: Versus in Air.” Environmental Science: Nano 6(9):2907–17. doi: 10.1039/c9en00662a.

Tong, H., Hu, X., Zhong, X., and Jiang, Q. (2021). “Adsorption and Desorption of Triclosan on Biodegradable Polyhydroxybutyrate Microplastics.” Environmental Toxicology and Chemistry, 40(1): 72–78. doi: 10.1002/etc.4902

“Toxic and Priority Pollutants Under the Clean Water Act.” (2021). Environmental Protection Agency. Retrieved from: https://www.epa.gov/eg/toxic-and-priority-pollutants-under-clean-water-act [Online Resource]

Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W.G., McGonigle, D., and Russell, A.E. (2004). “Lost at Sea: Where Is All the Plastic?” Science, 304(5672): 838. doi: 10.1126/science.1094559

Tubić, A., Lončarski, A., Apostolović, T., Isakovski, M.K., Tričković, J., Jazić, J.M., and Agbaba, J. (2021). “Adsorption Mechanisms of Chlorobenzenes and Trifluralin on Primary Polyethylene Microplastics in the Aquatic Environment.” Environmental Science and Pollution Research 28(42):59416–29. doi: 10.1007/s11356-020-11875-w.

Uber, T.H., Hüffer, T., Planitz, S., and Schmidt, T.C. (2019). “Sorption of non-ionic organic compounds by polystyrene in water.” Science of the Total Environment, 682: 348–355. doi: 10.1016/j.scitotenv.2019.05.040

Van Cauwenberghe, L., and Janssen, C.R. (2014). “Microplastics in Bivalves Cultured for Human Consumption.” Environmental Pollution, 193: 65–70. doi: 10.1016/j.envpol.2014.06.010

Velzeboer, I., Kwadijk, C.J.A.F., and Koelmans, A.A. (2014). “Strong sorption of PCBs to nanoplastics, microplastics, carbon nanotubes, and fullerenes.” Environmental Science and Technology, 48(9): 4869–4876. doi: 10.1021/es405721v

Wang, F., Shih, K.M., and Li, X.Y. (2015). “The partition behavior of perfluorooctanesulfonate (PFOS) and perfluorooctanesulfonamide (FOSA) on microplastics.” Chemosphere, 119: 841–847. doi: 10.1016/j.chemosphere.2014.08.047

Wang, J., Liu, X., and Liu, G. (2019a). “Sorption behaviors of phenanthrene, nitrobenzene, and naphthalene on mesoplastics and microplastics.” Environmental Science and Pollution Research: 12563–12573. doi: 10.1007/s11356-019-04735-9

Wang, J., Liu, X., Liu, G., Zhang, Z., Wu, H., Cui, B., Bai, J., and Zhang, W. (2019b). “Size effect of polystyrene microplastics on sorption of phenanthrene and nitrobenzene.” Ecotoxicology and Environmental Safety, 173(February): 331–338. doi: 10.1016/j.ecoenv.2019.02.037

Wang, S., Chen, H., Zhou, X., Tian, Y., Lin, C., Wang, W., Kaiwen, Z., Zhang, Y., and Lin, H. (2020). “Microplastic abundance, distribution and composition in the mid-west Pacific Ocean.” Environmental Pollution, 264: 114125. doi: 10.1016/j.envpol.2020.114125

Wang, W., and Wang, J. (2018a). “Comparative evaluation of sorption kinetics and isotherms of pyrene onto microplastics.” Chemosphere, 193: 567–573. doi: 10.1016/j.chemosphere.2017.11.078

Wang, W., and Wang, J. (2018b). “Different partition of polycyclic aromatic hydrocarbon on environmental particulates in freshwater: Microplastics in comparison to natural sediment.” Ecotoxicology and Environmental Safety, 147(41907352): 648–655. doi: 10.1016/j.ecoenv.2017.09.029

Wang, Y., Wang, X., Li, Y., Li, J., Wang, F., Xia, S., and Zhao, J. (2020). “Biofilm alters tetracycline and copper adsorption behaviors onto polyethylene microplastics.” Chemical Engineering Journal, 392(October 2019): 123808. doi: 10.1016/j.cej.2019.123808

Wang, Z., Chen, M., Zhang, L., Wang, K., Yu, X., Zheng, Z., and Zheng, R. (2018). “Sorption behaviors of phenanthrene on the microplastics identified in a mariculture farm in Xiangshan Bay, southeastern China.” Science of the Total Environment, 628–629: 1617–1626. doi: 10.1016/j.scitotenv.2018.02.146

Wang, F., Zhang, M., Sha, W., Wang, Y., Hao, H., Dou, Y., Li, Y. (2020). “Sorption Behavior and Mechanisms of Organic Contaminants to Nano and Microplastics.” Molecules 2020, 25: 1827. http://dx.doi.org/10.3390/molecules25081827

Wang, T., Yu, C., Chu, Q., Wang, F., Lan, T., and Wang, J. (2020). “Adsorption Behavior and Mechanism of Five Pesticides on Microplastics from Agricultural Polyethylene Films.” Chemosphere 244. doi: 10.1016/j.chemosphere.2019.125491.

Weir, N.A., and Whiting, K. (1989). “Initial Steps in the Photolysis and Photooxidation of Polystyrene.” European Polymer Journal 25(3):291–95. doi: 10.1016/0014-3057(89)90234-6.

WHO. (2019). “Microplastics in drinking-water.” Retrieved from: https://www.who.int/publications/i/item/9789241516198

Wick, A., Jacobs, B., Kunkel, U., Heininger, P., and Ternes. T.A. (2016). “Benzotriazole UV Stabilizers in Sediments, Suspended Particulate Matter and Fish of German Rivers: New Insights into Occurrence, Time Trends and Persistency.” Environmental Pollution 212:401–12. doi: 10.1016/j.envpol.2016.01.024.

Wu, J., Xu, P., Chen, Q., Ma, D., Ge, W., Jiang, T., and Chai, C. (2020a). “Effects of polymer aging on sorption of 2,2′,4,4′-tetrabromodiphenyl ether by polystyrene microplastics.” Chemosphere, 253: 1–9. doi: 10.1016/j.chemosphere.2020.126706

Wu, P., Cai, Z., Jin, H., and Tang, Y. (2019). “Adsorption mechanisms of five bisphenol analogues on PVC microplastics.” Science of the Total Environment, 650: 671–678. doi: 10.1016/j.scitotenv.2018.09.049

Wu, X., Liu, P., Huang, H., and Gao, S. (2020b). “Adsorption of triclosan onto different aged polypropylene microplastics: Critical effect of cations.” Science of the Total Environment, 717: 137033. doi: 10.1016/j.scitotenv.2020.137033

Xia, Y., Zhou, J.J., Gong, Y.Y., Li, Z.J., and Zeng, E.Y. (2020). “Strong influence of surfactants on virgin hydrophobic microplastics adsorbing ionic organic pollutants.” Environmental Pollution, 265: 115061. doi: 10.1016/j.envpol.2020.115061

Xiao, F., Zhang, X., Penn, L., Gulliver, J.S., and Simcik, M.F. (2011). “Effects of monovalent cations on the competitive adsorption of perfluoroalkyl substances by kaolinite: Experimental studies and modeling.” Environmental Science and Technology, 45(23): 10028-10035. doi: 10.1021/es202524y

Xu, B., Liu, F., Brookes, P.C., and Xu, J. (2018). “The sorption kinetics and isotherms of sulfamethoxazole with polyethylene microplastics.” Marine Pollution Bulletin, 131(April): 191–196. doi: 10.1016/j.marpolbul.2018.04.027

Xu, J., Wang, L., and Sun, H. (2021). “Adsorption of neutral organic compounds on polar and nonpolar microplastics: Prediction and insight into mechanisms based on pp-LFERs.” Journal of Hazardous Materials, 408(October 2020): 124857. doi: 10.1016/j.jhazmat.2020.124857

Xu, P., Ge, W., Chai, C., Zhang, Y., Jiang, T., and Xia, B. (2019). “Sorption of polybrominated diphenyl ethers by microplastics.” Marine Pollution Bulletin, 145(February): 260–269. doi: 10.1016/j.marpolbul.2019.05.050

Yakout, S.M., Daifullah, A.A.M., and El-Reefy, S.A. (2012). “Adsorption of Naphthalene, Phenanthrene, and Pyrene from Aqueous Solution Using Low-Cost Activated Carbon Derived from Agricultural Wastes.” Adsorption Science and Technology, 31(4): 293-302. doi: 10.1260%2F0263-6174.31.4.293

Yilimulati, M., Wang, L., Ma, X., Yang, C., and Habibul, N. (2021). “Adsorption of ciprofloxacin to functionalized nano-sized polystyrene plastic: Kinetics, thermochemistry and toxicity.” Science of the Total Environment, 750: 142370. doi: 10.1016/j.scitotenv.2020.142370

Yousif, E., and Haddad, R. (2013). “Photodegradation and Photostabilization of Polymers, Especially Polystyrene: Review.” SpringerPlus 2(1):1–32. doi: 10.1186/2193-1801-2-398.

Yousif, E., Salimon, J., and Salih, N. (2012). “New stabilizers for polystyrene based on 2-N-salicylidene-5-(substituted)-1,3,4-thiadiazole compounds.” Journal of Saudi Chemical Society, 16(3), 299–306. doi: 10.1016/j.jscs.2011.01.011

Yu, F., Yang, C., Huang, G., Zhou, T., Zhao, Y., and Ma, J. (2020). “Interfacial interaction between diverse microplastics and tetracycline by adsorption in an aqueous solution.” Science of the Total Environment, 721: 137729. doi: 10.1016/j.scitotenv.2020.137729

Yu, H., Yang, B., Waigi, M. G., Peng, F., Li, Z., and Hu, X. (2020). “The effects of functional groups on the sorption of naphthalene on microplastics.” Chemosphere, 261: 127592. doi: 10.1016/j.chemosphere.2020.127592

Yu, F., Yang, C., Zhu, Z., Bai, X., Ma, J. (2019). “Adsorption behavior of organic pollutants and metals on micro/nanoplastics in the aquatic environment.” Science of the Total Environment, 694 (2019): 133643. doi: 10.1016/j.scitotenv.2019.133643

Yurtsever, M., Oz, N., Aksu, A., Balkis, N., Altug, G., and Taskin, Ö. S. (2020). “Hydrophobic pesticide endosulfan (α + β) and endrin sorption on different types of microplastics.” Journal of the Chemical Society of Pakistan, 42(5): 789–797. doi: 10.52568/000685/jcsp/42.05.2020

Zettler, E.R., Mincer, T.J., Amaral-Zettler, L.A. (2013). “Life in the “Plastisphere”: Microbial Communities on Plastic Marine Debris.” Environmental Science and Technology, 47(13). http://dx.doi.org/10.1021/es401288x

Zhang, Haibo, Wang, J., Zhou, B., Zhou, Y., Dai, Z., Zhou, Q., Chriestie, P., and Luo, Y. (2018). Enhanced adsorption of oxytetracycline to weathered microplastic polystyrene: Kinetics, isotherms and influencing factors. Environmental Pollution, 243: 1550–1557. doi: 10.1016/j.envpol.2018.09.122

Zhang, H., Liu, F., Wang, S., Huang, T., Li, M., Zhu, Z., and Liu, G. (2020). “Sorption of fluoroquinolones to nanoplastics as affected by surface functionalization and solution chemistry.” Environmental Pollution, 262: 114347. doi: 10.1016/j.envpol.2020.114347

Zhang, J.Y., Boyd, I.W., and Esrom, H. (1997). “UV Intensity Measurement for a Novel 222 Nm Excimer Lamp Using Chemical Actinometer.” Applied Surface Science 109–110:482–86. doi: 10.1016/S0169-4332(96)00789-1.

Zhang, J., Huangbo, C., Huan, H., Xinying, C., Tao, M., Jiapeng, H., Shaogui, Y., Shiyin, L., and Limin, Z. (2020). “Adsorption Behavior and Mechanism of 9-Nitroanthracene on Typical Microplastics in Aqueous Solutions.” Chemosphere 245:125628. doi: 10.1016/j.chemosphere.2019.125628.

Zhang, X., Zheng, M., Yin, X., Wang, L., Lou, Y., Qu, L., Liu, X., Zhu, H., and Qiu, Y. (2019). “Sorption of 3,6-dibromocarbazole and 1,3,6,8-tetrabromocarbazole by microplastics.” Marine Pollution Bulletin, 138(September 2018): 458–463. doi: 10.1016/j.marpolbul.2018.11.055

Zhang, Y., Ni, F., He, J., Shen, F., Deng, S., Tian, D., Zhang, Y., Liu, Y., Chen, C., and Zou, J. (2021). “Mechanistic insight into different adsorption of norfloxacin on microplastics in simulated natural water and real surface water.” Environmental Pollution, 284(May): 117537. doi: 10.1016/j.envpol.2021.117537

Zhao, L., Rong, L., Xu, J., Lian, J., Wang, L., and Sun, H. (2020). “Sorption of five organic compounds by polar and nonpolar microplastics.” Chemosphere, 257: 127206. doi: 10.1016/j.chemosphere.2020.127206

Zhao, S., Zhu, L., and Li, D. (2015). “Microplastic in three urban estuaries, China.” Environmental Pollution, 206: 597-604. doi: 10.1016/j.envpol.2015.08.027

Zhan, Z., Wang, J., Peng, J., Xie, Q., Huang, Y., and Gao, Y. (2016). “Sorption of 3,3′,4,4′-tetrachlorobiphenyl by microplastics: A case study of polypropylene.” Marine Pollution Bulletin, 110(1): 559–563. doi: 10.1016/j.marpolbul.2016.05.036

Zhou, J., Gui, H., Banfield, C.C., Wen, Y., Zang, H., Dippold, M.A., Charlton, A., and Jones, D.L. (2021). “The Microplastisphere: Biodegradable Microplastics Addition Alters Soil Microbial Community Structure and Function.” Soil Biology and Biochemistry 156(October 2020):108211. doi: 10.1016/j.soilbio.2021.108211.

Zhou, J., Gui, H., Banfield, C.C., Wen, Y., Zang, H., Dippold, M.A., Charlton, A., Jones, D.L. (2021). “The microplastisphere: Biodegradable microplastics addition alters soil microbial community structure and function.” Soil Biology and Biochemistry, 156 (2021): 108211. doi: 10.1016/j.soilbio.2021.108211

Zhu, K., Hanzhong J., Yajiao S., Yunchao D., Chi Z., Xuetao G., Tiecheng W., and Lingyan Z. (2020). “Long-Term Phototransformation of Microplastics under Simulated Sunlight Irradiation in Aquatic Environments: Roles of Reactive Oxygen Species.” Water Research 173:115564. doi: 10.1016/j.watres.2020.115564.

Zuo, L.Z., Li, H.X., Lin, L., Sun, Y.X., Diao, Z.H., Liu, S., Zhang, Z.Y., and Xu, X.R. (2019). "Sorption and desorption of phenanthrene on biodegradable poly(butylene adipate co-terephtalate) microplastics." Chemosphere, 215: 25–32. doi: 10.1016/j.chemosphere.2018.09

APPENDIX

Database of 66 papers with adsorption parameter K_D, log K_{ow}, polymer type, compound type, water type, size, and surface area (NR = not reported).

Authors	Title	Plastic	Category	Compound	Kd (L/kg)	log(Kow)	SSA (m2/g)	Size (um)	Mean Size (um)	Water Type	Data Contributed to Figure:
Lan et al. 2021	A comparative study on the adsorption behavior of pesticides by pristine and aged microplastics from agricultural polyethylene soil films	polyethylene	PE	carbendazim (CAR)	1.3	1.5	12.1	<5000	5000	DI	2,3,4a,4b
				diflubenzuron (DIF)	628.9	3.9	12.1	<5000	5000	DI	
				malathion (MAL)	8.4	2.4	12.1	<5000	5000	DI	
				difenoconazole (DIFE)	255.5	4.36	12.1	<5000	5000	DI	
		agricultural polyethylene	PE	carbendazim (CAR)	19.7	1.5	13.6	<5000	5000	DI	
				diflubenzuron (DIF)	1045.8	3.9	13.6	<5000	5000	DI	
				malathion (MAL)	62.6	2.4	13.6	<5000	5000	DI	
				difenoconazole (DIFE)	624.8	4.36	13.6	<5000	5000	DI	
Hummel et al. 2021	Additives and polymer composition influence the interaction of microplastics with xenobiotics	PVC	PVC	oestrone (E1)	572.8	3.43	0.517	124.2 +/- 68.62	124.2	DI	2,3,4a
					75.7	3.43	0.689	137.4 +/- 28.67	137.4	DI	
				17-a-ethinyloestradiol (EE2)	1140.2	4.21	0.517	124.2 +/- 68.62	124.2	DI	
					461.3	4.21	0.689	137.4 +/- 28.67	137.4	DI	
				norethisterone (NOR)	171.0	2.99	0.517	124.2 +/- 68.62	124.2	DI	
					36.2	2.99	0.689	137.4 +/- 28.67	137.4	DI	
Tong et al. 2020	Adsorption and Desorption of Triclosan on Biodegradable Polyhydroxybutyrate Microplastics	polyhydroxybutyrate (PHB)	OTHER	triclosan	21412.0	5	NR	1222 +/- 104	1222	DI	3,4a,4b
		polyethylene	PE		536.2	5	NR	1255 +/- 144	1255	DI	

Zhang et al. 2020	Adsorption behavior and mechanism of 9-Nitroanthracene on typical microplastics in aqueous solutions	polyethylene	PE	9-nitroanthracene	34000.0	4.47	NR	100 - 150	125	DI	3,4a
		polypropylene	PP		17940.0	4.47	NR	100 - 150	125	DI	
		polystyrene	PS		24810.0	4.47	NR	100 - 150	125	DI	
Wang et al. 2020	Adsorption behavior and mechanism of five pesticides on microplastics from agricultural polyethylene films	agricultural polyethylene	PE	carbendazim (CAR)	3.6	1.5	NR	<5000	5000	DI	3,4a,4b
				dipterex (DIP)	2.6	0.43	NR	<5000	5000	DI	
				diflubenzuron (DIF)	513.0	3.9	NR	<5000	5000	DI	
				malathion (MAL)	9.2	2.75	NR	<5000	5000	DI	
				difenoconazole (DIFE)	144.6	4.36	NR	<5000	5000	DI	
Fang et al. 2019	Adsorption behavior of three triazole fungicides on polystyrene microplastics	polystyrene	PS	triadimenol (TRI)	0.1	2.91	0.93	100	100	DI	3,4a
				myclobutanil (MYC)	0.3	3.58	0.93	100	100	DI	
				hexaconazole (HEX)	0.7	4.01	0.93	100	100	DI	
Tubic et al. 2020	Adsorption mechanisms of chlorobenzenes and trifluralin on primary polyethylene microplastics in the aquatic environment	polyethylene	PE	1,2,3-trichlorobenzene (TeCB)	24545.5	4.05	2.088	49.7 - 259	154.35	SF	2,3,4a,4b
					24545.5	4.05	2.088	49.7 - 259	154.35	F	
				1,3,5-trichlorobenzene (TeCB)	23275.9	4.19	2.088	49.7 - 259	154.35	SF	
					8421.1	4.19	2.088	49.7 - 259	154.35	F	
				1,2,4-trichlorobenzene (TeCB)	5800.0	4.02	2.088	49.7 - 259	154.35	SF	
					12000.0	4.02	2.088	49.7 - 259	154.35	F	
				pentachlorobenzene (PeCB)	30769.2	5.17	2.088	49.7 - 259	154.35	SF	
					42000.0	5.17	2.088	49.7 - 259	154.35	F	
				hexachlorobenzene (HeCB)	19029.9	5.31	2.088	49.7 - 259	154.35	SF	
					37272.7	5.31	2.088	49.7 - 259	154.35	F	
				trifluralin (TFL)	24090.9	5.34	2.088	49.7 - 259	154.35	SF	
					28888.9	5.34	2.088	49.7 - 259	154.35	F	
		polyethylene_pcp1	PE	1,2,3-trichlorobenzene (TeCB)	8800.0	4.05	0.66	80 - 185	132.5	SF	
					18214.3	4.05	0.66	80 - 185	132.5	F	
				1,3,5-trichlorobenzene (TeCB)	5735.3	4.19	0.66	80 - 185	132.5	SF	
					9183.7	4.19	0.66	80 - 185	132.5	F	
				1,2,4-trichlorobenzene (TeCB)	21371.0	4.02	0.66	80 - 185	132.5	SF	
					29347.8	4.02	0.66	80 - 185	132.5	F	
				pentachlorobenzene (PeCB)	24090.9	5.17	0.66	80 - 185	132.5	SF	
					46721.3	5.17	0.66	80 - 185	132.5	F	

				hexachlorobenzene (HeCB)	25238.1	5.31	0.66	80 - 185	132.5	SF	
					47500.0	5.31	0.66	80 - 185	132.5	F	
				trifluralin (TFL)	18750.0	5.34	0.66	80 - 185	132.5	SF	
					63333.3	5.34	0.66	80 - 185	132.5	F	
		polyethylene_pcp2	PE	1,2,3-trichlorobenzene (TeCB)	8269.2	4.05	5.791	244 - 358	301	SF	
					17000.0	4.05	5.791	244 - 358	301	F	
				1,3,5-trichlorobenzene (TeCB)	5820.9	4.19	5.791	244 - 358	301	SF	
					12631.6	4.19	5.791	244 - 358	301	F	
				1,2,4-trichlorobenzene (TeCB)	17000.0	4.02	5.791	244 - 358	301	SF	
					30000.0	4.02	5.791	244 - 358	301	F	
				pentachlorobenzene (PeCB)	31764.7	5.17	5.791	244 - 358	301	SF	
					43076.9	5.17	5.791	244 - 358	301	F	
				hexachlorobenzene (HeCB)	37671.2	5.31	5.791	244 - 358	301	SF	
					45833.3	5.31	5.791	244 - 358	301	F	
				trifluralin (TFL)	42307.7	5.34	5.791	244 - 358	301	SF	
					67857.1	5.34	5.791	244 - 358	301	F	
Li et al. 2018	Adsorption of antibiotics on microplastics	polypropylene	PP	ciprofloxacin (CIP)	57.1	0.28	NR	117.29	117.29	DI	3,4a,4b
				trimethoprim (TMP)	9.7	0.9	NR	117.29	117.29	DI	
					3.9	0.9	NR	117.29	117.29	S	
				sulfadiazine (SDZ)	7.9	-0.1	NR	117.29	117.29	DI	
					7.1	-0.1	NR	117.29	117.29	S	
				amoxicillin (AMX)	17.5	0.87	NR	117.29	117.29	DI	
		polystyrene	PS	ciprofloxacin (CIP)	51.5	0.28	NR	132.05	132.05	DI	
				trimethoprim (TMP)	9.5	0.9	NR	132.05	132.05	DI	
					7.3	0.9	NR	132.05	132.05	S	
				sulfadiazine (SDZ)	7.4	-0.1	NR	132.05	132.05	DI	
					6.8	-0.1	NR	132.05	132.05	S	
		PVC	PVC	ciprofloxacin (CIP)	41.5	0.28	NR	86.92	86.92	DI	
				trimethoprim (TMP)	8.4	0.9	NR	86.92	86.92	DI	
					5.5	0.9	NR	86.92	86.92	S	
				sulfadiazine (SDZ)	6.6	-0.1	NR	86.92	86.92	DI	

					5.4	-0.1	NR	86.92	86.92	S	
				amoxicillin (AMX)	24.7	0.87	NR	86.92	86.92	DI	
		polyethylene	PE	ciprofloxacin (CIP)	55.1	0.28	NR	149.1	149.1	DI	
				trimethoprim (TMP)	8.4	0.9	NR	149.1	149.1	DI	
					6.5	0.9	NR	149.1	149.1	S	
				sulfadiazine (SDZ)	6.2	-0.1	NR	149.1	149.1	DI	
					6.3	-0.1	NR	149.1	149.1	S	
				amoxicillin (AMX)	8.4	0.87	NR	149.1	149.1	DI	
		polyamide	PA	ciprofloxacin (CIP)	96.5	0.28	NR	134.2	134.2	DI	
				trimethoprim (TMP)	17.1	0.9	NR	134.2	134.2	DI	
					5.9	0.9	NR	134.2	134.2	S	
				sulfadiazine (SDZ)	7.4	-0.1	NR	134.2	134.2	DI	
					6.6	-0.1	NR	134.2	134.2	S	
				amoxicillin (AMX)	756.0	0.87	NR	134.2	134.2	DI	
				tetracycline (TC)	356.0	-1.37	NR	134.2	134.2	DI	
					4.4	-1.37	NR	134.2	134.2	S	
Liu et al. 2020	Adsorption of chlorophenols on polyethylene terephthalate microplastics from aqueous environments: Kinetics, mechanisms and influencing factors*	polyethylene terephthalate	PET	4-chlorophenol (MCP)	10.6	2.42	0.864	<150	150	DI	2,3,4a
					11.4	2.42	0.864	<150	150	S	
					10.7	2.42	0.864	<150	150	F	
				2,4-dichlorophenol (DCP)	29.4	3.17	0.864	<150	150	DI	
					21.7	3.17	0.864	<150	150	S	
					18.0	3.17	0.864	<150	150	F	
				2,4,6-trichlorophenol (TCP)	10.5	3.66	0.864	<150	150	DI	
					5.7	3.66	0.864	<150	150	S	
					4.6	3.66	0.864	<150	150	F	
Yilimulati et al. 2021	Adsorption of ciprofloxacin to functionalized nano-sized polystyrene plastic: Kinetics, thermochemistry and toxicity	polystyrene	PS	ciprofloxacin (CIP)	9782.6	0.28	NR	0.2	0.2	DI	3,4a
					4923.1	0.28	NR	0.5	0.5	DI	
Xu et al. 2021	Adsorption of neutral organic compounds on polar and nonpolar microplastics: Prediction and insight into mechanisms based on pp-LFERs	low-density polyethylene	PE	diethyl phthalate (DEP)	16.0	2.47	NR	100 - 200	150	DI	3,4a,4b
				dibutyl phthalate (DBP)	1901.1	4.5	NR	100 - 200	150	DI	
				diisobutyl phthalate (DiBP)	1622.5	4.11	NR	100 - 200	150	DI	
				bisphenol A (BPA)	6.0	3.32	NR	100 - 200	150	DI	

				naphthalene	404.9	3.3	NR	100 - 200	150	DI
				phenanthrene	8530.3	4.46	NR	100 - 200	150	DI
				atrazine (ATR)	7.6	2.61	NR	100 - 200	150	DI
				diuron (DIU)	68.6	2.68	NR	100 - 200	150	DI
				1-naphthol (NAP)	9.6	2.85	NR	100 - 200	150	DI
				1-nitronaphthalene (1-Mnap)	134.2	3.19	NR	100 - 200	150	DI
				1-chloronaphthalene	2414.3	4	NR	100 - 200	150	DI
				1-methylnaphthalene	1510.4	3.87	NR	100 - 200	150	DI
				2-tertbutyl-4-methyl phenol (2,4-Tmphenol)	49.5	3.6	NR	100 - 200	150	DI
				2,3,5-trimethyl phenol (2,3,5-Tmphenol)	30.8	4	NR	100 - 200	150	DI
				n-propylbenzene (PB)	276.0	3.69	NR	100 - 200	150	DI
				bromobenzene (BRB)	178.1	2.99	NR	100 - 200	150	DI
				biphenyl (BIP)	1661.7	4.01	NR	100 - 200	150	DI
				m-xylene (m-Xyl)	376.3	3.2	NR	100 - 200	150	DI
				diethyl phthalate (DEP)	4.3	2.47	NR	100 - 200	150	DI
				dibutyl phthalate (DBP)	9992.1	4.5	NR	100 - 200	150	DI
				diisobutyl phthalate (DiBP)	4173.9	4.11	NR	100 - 200	150	DI
				bisphenol A (BPA)	6.5	3.32	NR	100 - 200	150	DI
		polystyrene	PS	naphthalene	649.1	3.3	NR	100 - 200	150	DI
				phenanthrene	1822.8	4.46	NR	100 - 200	150	DI
				diuron (DIU)	1.7	2.68	NR	100 - 200	150	DI
				1-naphthol (NAP)	8.1	2.85	NR	100 - 200	150	DI
				1-nitronaphthalene (1-Mnap)	133.5	3.19	NR	100 - 200	150	DI
				1-chloronaphthalene	1597.4	4	NR	100 - 200	150	DI
				1-methylnaphthalene	1022.5	3.87	NR	100 - 200	150	DI
				2-tertbutyl-4-methyl phenol (2,4-Tmphenol)	41.1	3.6	NR	100 - 200	150	DI
				2,3,5-trimethyl phenol (2,3,5-Tmphenol)	26.7	4	NR	100 - 200	150	DI
				n-propylbenzene (PB)	2110.1	3.69	NR	100 - 200	150	DI
				bromobenzene (BRB)	509.7	2.99	NR	100 - 200	150	DI
				biphenyl (BIP)	1838.6	4.01	NR	100 - 200	150	DI
				m-xylene (m-Xyl)	1563.8	3.2	NR	100 - 200	150	DI

		polybutylene succinate	OTHER	diethyl phthalate (DEP)	135.8	2.47	NR	100 - 200	150	DI	
				dibutyl phthalate (DBP)	5953.6	4.5	NR	100 - 200	150	DI	
				diisobutyl phthalate (DiBP)	3481.5	4.11	NR	100 - 200	150	DI	
				bisphenol A (BPA)	1179.1	3.32	NR	100 - 200	150	DI	
				naphthalene	1026.6	3.3	NR	100 - 200	150	DI	
				phenanthrene	22516.0	4.46	NR	100 - 200	150	DI	
				atrazine (ATR)	112.8	2.61	NR	100 - 200	150	DI	
				diuron (DIU)	160.2	2.68	NR	100 - 200	150	DI	
				1-naphthol (NAP)	609.0	2.85	NR	100 - 200	150	DI	
				1-nitronaphthalene (1-Mnap)	1531.6	3.19	NR	100 - 200	150	DI	
				1-chloronaphthalene	4825.1	4	NR	100 - 200	150	DI	
				1-methylnaphthalene	2637.9	3.87	NR	100 - 200	150	DI	
				2-tertbutyl-4-methyl phenol (2,4-Tmphenol)	1772.5	3.6	NR	100 - 200	150	DI	
				2,3,5-trimethyl phenol (2,3,5-Tmphenol)	200.0	4	NR	100 - 200	150	DI	
				n-propylbenzene (PB)	763.9	3.69	NR	100 - 200	150	DI	
				bromobenzene (BRB)	328.9	2.99	NR	100 - 200	150	DI	
				biphenyl (BIP)	4815.1	4.01	NR	100 - 200	150	DI	
				m-xylene (m-Xyl)	496.8	3.2	NR	100 - 200	150	DI	
		polycaprolactone	OTHER	diethyl phthalate (DEP)	124.9	2.47	NR	100 - 200	150	DI	
				dibutyl phthalate (DBP)	7394.3	4.5	NR	100 - 200	150	DI	
				diisobutyl phthalate (DiBP)	7280.6	4.11	NR	100 - 200	150	DI	
				bisphenol A (BPA)	205.8	3.32	NR	100 - 200	150	DI	
				naphthalene	1696.7	3.3	NR	100 - 200	150	DI	
				phenanthrene	41290.0	4.46	NR	100 - 200	150	DI	
				atrazine (ATR)	174.5	2.61	NR	100 - 200	150	DI	
				diuron (DIU)	170.5	2.68	NR	100 - 200	150	DI	
				1-naphthol (NAP)	869.9	2.85	NR	100 - 200	150	DI	
				1-nitronaphthalene (1-Mnap)	2301.1	3.19	NR	100 - 200	150	DI	
				1-chloronaphthalene	7789.5	4	NR	100 - 200	150	DI	
				1-methylnaphthalene	4333.9	3.87	NR	100 - 200	150	DI	
				2-tertbutyl-4-methyl phenol (2,4-Tmphenol)	2869.0	3.6	NR	100 - 200	150	DI	

				2,3,5-trimethyl phenol (2,3,5-Tmphenol)	266.7	4	NR	100 - 200	150	DI	
				n-propylbenzene (PB)	1243.8	3.69	NR	100 - 200	150	DI	
				bromobenzene (BRB)	757.9	2.99	NR	100 - 200	150	DI	
				biphenyl (BIP)	8858.7	4.01	NR	100 - 200	150	DI	
				m-xylene (m-Xyl)	818.7	3.2	NR	100 - 200	150	DI	
Llorca et al. 2018	Adsorption of perfluoroalkyl substances on microplastics under environmental conditions*	high-density polyethylene	PE	PFPeA	76.9	2.9	NR	3 - 16	9.5	S	3,4a,4b
				PFHxA	53.3	3.6	NR	3 - 16	9.5	S	
				PFHpA	54.3	4.3	NR	3 - 16	9.5	S	
				PFOA	64.1	4.9	NR	3 - 16	9.5	S	
				PFNA	41.7	5.6	NR	3 - 16	9.5	S	
				PFTrA	595.9	8.3	NR	3 - 16	9.5	F	
				PFTeA	642.3	4.49	NR	3 - 16	9.5	F	
				PFHxDA	341.3	3.16	NR	3 - 16	9.5	F	
		polystyrene	PS	PFBA	4.4	2.2	NR	3 - 12	9.5	F	
					136.0	2.2	NR	3 - 12	9.5	S	
				PFPeA	12.0	2.9	NR	3 - 12	9.5	F	
					270.5	2.9	NR	3 - 12	9.5	S	
				PFHxA	57.6	3.6	NR	3 - 12	9.5	S	
				PFHpA	53.9	4.3	NR	3 - 12	9.5	S	
				PFOA	8.4	4.9	NR	3 - 12	9.5	F	
				PFDA	455.1	6.3	NR	3 - 12	9.5	F	
					466.4	6.3	NR	3 - 12	9.5	S	
				PFUnA	565.0	6.9	NR	3 - 12	9.5	S	
				PFDoA	524.8	7.6	NR	3 - 12	9.5	S	
				PFTrA	1052.1	8.3	NR	3 - 12	9.5	F	
					1451.9	8.3	NR	3 - 12	9.5	S	
				PFTeA	1223.5	4.49	NR	3 - 12	9.5	S	
				PFHxDA	396.3	3.16	NR	3 - 12	9.5	F	
				PFODA	515.1	11.6	NR	3 - 12	9.5	F	
					1657.9	11.6	NR	3 - 12	9.5	S	
				PFBS	27.7	2.3	NR	3 - 12	9.5	F	

					140.4	2.3	NR	3 - 12	9.5	S	
				PFHxS	27.1	3.7	NR	3 - 12	9.5	F	
				PFOS	501.4	4.9	NR	3 - 12	9.5	F	
				FOSA	619.3	4.8	NR	3 - 12	9.5	S	
		polystyrene - COOH	PS	PFBA	9.3	2.2	NR	3 - 12	9.5	F	
					197.1	2.2	NR	3 - 12	9.5	S	
				PFPeA	25.8	2.9	NR	3 - 12	9.5	F	
					366.9	2.9	NR	3 - 12	9.5	S	
				PFHxA	64.0	3.6	NR	3 - 12	9.5	S	
				PFDA	100.2	6.3	NR	3 - 12	9.5	F	
					370.9	6.3	NR	3 - 12	9.5	S	
				PFUnA	578.0	6.9	NR	3 - 12	9.5	F	
					1432.1	6.9	NR	3 - 12	9.5	S	
				PFDoA	1209.0	7.6	NR	3 - 12	9.5	F	
					2293.6	7.6	NR	3 - 12	9.5	S	
				PFTrA	2071.6	8.3	NR	3 - 12	9.5	F	
					4240.6	8.3	NR	3 - 12	9.5	S	
				PFTeA	1773.7	4.49	NR	3 - 12	9.5	F	
				PFHxDA	1931.2	3.16	NR	3 - 12	9.5	S	
				PFODA	832.8	11.6	NR	3 - 12	9.5	F	
					802.5	11.6	NR	3 - 12	9.5	S	
				PFBS	12.5	2.3	NR	3 - 12	9.5	F	
					128.3	2.3	NR	3 - 12	9.5	S	
				PFHxS	2.2	3.7	NR	3 - 12	9.5	F	
				PFOS	100.5	4.9	NR	3 - 12	9.5	F	
				PFDS	673.4	6.3	NR	3 - 12	9.5	F	
					801.3	6.3	NR	3 - 12	9.5	S	
				FOSA	630.4	4.8	NR	3 - 12	9.5	F	
					459.6	4.8	NR	3 - 12	9.5	S	
Wu et al. 2020	Adsorption of triclosan onto different aged polypropylene microplastics_ Critical effect of cations	polypropylene	PP	triclosan (TCS)	1200.0	5	NR	<180	180	DI	3,4a

Atugoda et al. 2020	Adsorptive interaction of antibiotic ciprofloxacin on polyethylene microplastics: Implications for vector transport in water	polyethylene	PE	ciprofloxacin (CIP)	69.3	0.28	NR	100	100	DI	3,4a,4b
Wang et al. 2020	Biofilm alters tetracycline and copper adsorption behaviors onto polyethylene microplastics	polyethylene	PE	tetracycline (TC)	14.3	-1.37	0.2354	60 - 150	105	DI	3,4a,4b
					62.5	-1.37	0.2354	60 - 150	105	DI	
Gong et al. 2019	Comparative analysis on the sorption kinetics and isotherms of fipronil	polyethylene	PE	fipronil	120.0	4	0.553	75 - 150	112.5	DI	2,3,4a,4b
		polystyrene	PS		110.0	4	0.843	75 - 150	112.5	DI	
		PVC	PVC		80.0	4	0.144	75 - 150	112.5	DI	
		polypropylene	PP		160.0	4	0.789	75 - 150	112.5	DI	
		polylactic acid	OTHER		420.0	4	0.444	75 - 150	112.5	DI	
		polybutylene succinate	OTHER		3260.0	4	0.601	75 - 150	112.5	DI	
Wang et al. 2018	Comparative evaluation of sorption kinetics and isotherms of pyrene onto microplastics	polyethylene	PE	pyrene	7500.0	4.88	6.91	100 - 150	125	DI	2,3,4a,4b
		polystyrene	PS		2272.7	4.88	2.35	100 - 150	125	DI	
		PVC	PVC		1666.7	4.88	1.87	100 - 150	125	DI	
Chen et al. 2021	Comparison of adsorption and desorption of triclosan between microplastics and soil particles	polyethylene	PE	triclosan (TCS)	900.0	5	1.19	225 +/- 41	225	DI	2,3,4a,4b
		polystyrene	PS		260.0	5	0.73	313 +/- 48	313	DI	
Bakir et al. 2012	Competitive sorption of persistent organic pollutants onto microplastics in the marine environment	unplasticized PVC	PVC	phenanthrene	2285.0	4.46	NR	200 - 250	225	S	3,4a,4b
					5428.6	4.46	NR	200 - 250	225	S	
				4,4'-DDT	82500.0	6.79	NR	200 - 250	225	S	
					52000.0	6.79	NR	200 - 250	225	S	
		ultra-high molecular weight polyethylene	PE	phenanthrene	51532.0	4.46	NR	200 - 250	225	S	
					20000.0	4.46	NR	200 - 250	225	S	
				4,4'-DDT	90000.0	6.79	NR	200 - 250	225	S	
					97142.9	6.79	NR	200 - 250	225	S	
Hu et al. 2020	Cotransport of naphthalene with polystyrene nanoplastics (PSNP) in saturated porous media: Effects of PSNP/naphthalene ratio and ionic strength	polystyrene nanoparticles	PS	naphthalene	16000.0	3.3	NR	0.1219 +/- 0.0018	0.1219	DI	3,4a
					25000.0	3.3	NR	0.1219 +/- 0.0018	0.1219	DI	
					20681.8	3.3	NR	0.1219 +/- 0.0018	0.1219	DI	
Wang and Wang, 2018	Different partition of polycyclic aromatic hydrocarbon on environmental particulates in freshwater - Microplastics in comparison to natural sediment	polyethylene	PE	pyrene	1900.0	4.88	6.91	100 - 150	125	SF	2,3,4a,4b
					1850.0	4.88	6.91	100 - 150	125	SF	
					1650.0	4.88	6.91	100 - 150	125	SF	
					1600.0	4.88	6.91	100 - 150	125	SF	
					1580.0	4.88	6.91	100 - 150	125	SF	

		polystyrene	PS		210.0	4.88	2.35	100 - 150	125	SF	
					190.0	4.88	2.35	100 - 150	125	SF	
					180.0	4.88	2.35	100 - 150	125	SF	
					170.0	4.88	2.35	100 - 150	125	SF	
					170.0	4.88	2.35	100 - 150	125	SF	
		PVC	PVC		150.0	4.88	1.87	100 - 150	125	SF	
					130.0	4.88	1.87	100 - 150	125	SF	
					120.0	4.88	1.87	100 - 150	125	SF	
					110.0	4.88	1.87	100 - 150	125	SF	
					110.0	4.88	1.87	100 - 150	125	SF	
Liu et al. 2020	Effect of aging on adsorption behavior of polystyrene microplastics for pharmaceuticals_ Adsorption mechanism and role of aging intermediates	polystyrene	PS	atorvastatin (ATV)	390.0	6.36	0.32	50.4 +/- 11.9	50.4	DI	2,3,4a
				amlodipine (AML)	30.0	3	0.32	50.4 +/- 11.9	50.4	DI	
Bao et al. 2021	Effects of hydroxyl group content on adsorption and desorption of anthracene and anthrol by polyvinyl chloride microplastics	PVC	PVC	anthracene (ANT)	525.0	4.45	NR	140	140	DI	3,4a
				2-hydroxyanthracene (MOHA)	195.0	3.86	NR	140	140	DI	
				2,6-dihydroxyanthracene (DOHA)	11.0	3.38	NR	140	140	DI	
				1,8,9-trihydroxyanthracene (TOHA)	9.0	2.9	NR	140	140	DI	
Hai et al. 2020	Effects of Microplastics on the Adsorption and Bioavailability of Three Strobilurin Fungicides	polystyrene	PS	azoxystrobin	0.2	3.09	3.09	100	100	DI	2,3,4a,4b
				picoxystrobin	0.2	3.83	3.83	100	100	DI	
				pyraclostrobin	14.8	4.23	4.23	100	100	DI	
		polyethylene	PE	azoxystrobin	0.1	3.09	3.09	100	100	DI	
				picoxystrobin	0.1	3.83	3.83	100	100	DI	
				pyraclostrobin	9.9	4.23	4.23	100	100	DI	
Wu et al. 2020	Effects of polymer aging on sorption of 2,20 ,4,40 -tetrabromodiphenyl ether by polystyrene microplastics	polystyrene	PS	BDE-47	10.6	6.2	NR	4000	4000	DI	3,4a
Zhang et al. 2018	Enhanced adsorption of oxytetracycline to weathered microplastic polystyrene_ Kinetics, isotherms and influencing factors	polystyrene	PS	oxytetracycline	41.7	-0.9	2.03	450 - 1000	725	DI	2,3,4a
Li et al. 2020	Evaluating the effect of different modified microplastics on the availability of polycyclic aromatic hydrocarbons	high density polyethylene	PE	phenanthrene	10720.0	4.46	0.49	150 - 210	180	DI	2,3,4a,4b
Liu et al. 2019	Hydrophobic sorption behaviors of 17β-Estradiol on environmental microplastics	polyamide	PA	17B-estradiol (E2)	53900.0	4.01	0.544	10 - 60, 29.7	29.7	DI	2,3,4a,4b
		high-density polyethylene	PE		2462.5	4.01	0.415	20 - 120, 54.4	54.4	DI	
		linear low-density polyethylene	PE		863.6	4.01	0.428	450 - 600, 510.9	510.9	DI	
		polypropylene	PP		123.3	4.01	0.964	20 - 250, 144.4	144.4	DI	

		medium-density polyethylene	PE		46.8	4.01	0.257	20 - 250, 150	150	DI	
		low-density polyethylene	PE		45.8	4.01	0.843	2 - 20, 9.5	9.5	DI	
		polystyrene	PS		27.1	4.01	0.343	10 - 130, 59	59	DI	
		polycarbonate	OTHER		20.6	4.01	0.319	10 - 40, 21.4	21.4	DI	
		PVC	PVC		10.3	4.01	1.03	50 - 120, 80	80	DI	
		polymethyl methacrylate	OTHER		13.5	4.01	0.164	250 - 500, 350	350	DI	
Liu et al. 2019	Interactions between microplastics and phthalate esters as affected by microplastics characteristics and solution chemistry	PVC	PVC	diethyl phthalate (DEP)	6.1	2.39	NR	<75	75	DI	2,3,4a,4b
				dibutyl phthalate (DBP)	573.9	4.61	NR	<75	75	DI	
		polyethylene	PE	diethyl phthalate (DEP)	9.3	2.39	7.158	<75	75	DI	
				dibutyl phthalate (DBP)	940.4	4.61	7.158	<75	75	DI	
		polystyrene	PS	diethyl phthalate (DEP)	12.6	2.39	NR	<75	75	DI	
				dibutyl phthalate (DBP)	1966.0	4.61	NR	<75	75	DI	
Yu et al. 2020	Interfacial interaction between diverse microplastics and tetracycline by adsorption in an aqueous solution	polyethylene	PE	tetracycline (TC)	143.0	-1.37	NR	28	28	DI	3,4a,4b
					125.0	-1.37	NR	48	48	DI	
					67.0	-1.37	NR	125	125	DI	
					123.0	-1.37	NR	250	250	DI	
					107.0	-1.37	NR	590	590	DI	
		polystyrene	PS		58.0	-1.37	NR	74	74	DI	
		PVC	PVC		55.0	-1.37	NR	74	74	DI	
Zhang et al. 2021	Mechanistic insight into different adsorption of norfloxacin on microplastics in simulated natural water and real surface water☆	PVC	PVC	norfloxacin (NOR)	14.6	-1	0.196	150 - 250	200	SF	2,3,4a,4b
					11.9	-1	0.196	150 - 250	200	F	
		polystyrene	PS		8.6	-1	0.165	150 - 250	200	SF	
					15.6	-1	0.165	150 - 250	200	F	
		polyethylene	PE		6.8	-1	0.544	150 - 250	200	SF	
					9.0	-1	0.544	150 - 250	200	F	
		polypropylene	PP		5.9	-1	0.322	150 - 250	200	SF	
					7.6	-1	0.322	150 - 250	200	F	
Liu et al. 2019	Microplastics as Both a Sink and a Source of Bisphenol A in the Marine Environment	low-density polyethylene	PE	bisphenol A (BPA)	0.0	3.32	0.843	2 - 20, 9.5	9.5	DI	2,3,4,4b
		linear low-density polyethylene	PE		90.0	3.32	0.428	450 - 600, 510.9	510.9	DI	
		medium-density polyethylene	PE		12.5	3.32	0.257	20 - 250, 150	150	DI	

		high-density polyethylene	PE		48.0	3.32	0.415	20 - 120, 54.4	54.4	DI	
		polypropylene	PP		2.0	3.32	0.964	20 - 250, 144.4	144.4	DI	
		polystyrene	PS		49.0	3.32	0.343	10 - 130, 59	59	DI	
		polycarbonate	OTHER		0.0	3.32	0.319	10 - 40, 21.4	21.4	DI	
		PVC	PVC		0.0	3.32	1.03	50 - 120, 80	80	DI	
		polyamide	PA		4000.0	3.32	0.544	10 - 60, 29.7	29.7	DI	
		polymethyl methacrylate	OTHER		1.0	3.32	0.164	250 - 500, 350	350	DI	
		polyethylene terephthalate	PET		5.1	3.32	NR	NR	NR	DI	
		polyurethane	OTHER		950.0	3.32	NR	NR	NR	DI	
Abbasi et al. 2021	PET-microplastics as a vector for polycyclic aromatic hydrocarbons in a simulated plant rhizosphere zone	polyethylene terephthalate	PET	naphthalene	622.9	3.3	NR	<2000	2000	DI	3,4a
					22.9	3.3	NR	<2000	2000	DI	
					32.2	3.3	NR	<2000	2000	DI	
					10.5	3.3	NR	<2000	2000	DI	
				phenanthrene	205.0	4.46	NR	<2000	2000	DI	
					23.3	4.46	NR	<2000	2000	DI	
					200.0	4.46	NR	<2000	2000	DI	
					5.0	4.46	NR	<2000	2000	DI	
					980.0	4.46	NR	<2000	2000	DI	
Teuten et al. 2007	Potential for Plastics to Transport Hydrophobic Contaminants	polyethylene	PE	phenanthrene	38100.0	4.46	4.37	200 - 250	225	S	2,3,4a,4b
		polypropylene	PP		2190.0	4.46	1.56	200 - 250	225	S	
		PVC	PVC		1650.0	4.46	NR	200 - 250	225	S	
					1690.0	4.46	1.76	127	127	S	
Gui et al. 2021	Prediction of organic compounds adsorbed by polyethylene and chlorinated polyethylene microplastics in freshwater using QSAR	polyethylene	PE	phenanthrene	1716.0	4.46	NR	<280	280	DI	3,4a,4b
				naphthalene	855.0	3.3	NR	<280	280	DI	
				nitrobenzene	133.0	1.85	NR	<280	280	DI	
				2-nitrotoluene	265.0	2.3	NR	<280	280	DI	
				1,3-dinitrobenzene	72.0	1.49	NR	<280	280	DI	
				4-nitrotoluene	388.0	2.37	NR	<280	280	DI	
				2,4-dinitrotoluene	223.0	1.98	NR	<280	280	DI	
				1,2,3,4-tetrachlorobenzene	7711.0	4.6	NR	<280	280	DI	
				2,4-dinitrochlorobenzene	300.0	2.17	NR	<280	280	DI	

				2,4-dichlorophenol	491.0	3.06	NR	<280	280	DI	
				tricyclazole	44.0	1.7	NR	<280	280	DI	
				triadimefon	164.0	2.77	NR	<280	280	DI	
				clothianidin	55.0	0.7	NR	<280	280	DI	
Wang et al. 2019	Size effect of polystyrene microplastics on sorption of phenanthrene nitrobenzene	polystyrene	PS	phenanthrene	1174.9	4.46	0.4	170	170	DI	2,3,4a
					2570.4	4.46	0.9	102	102	DI	
					5370.3	4.46	1.3	50	50	DI	
					6456.5	4.46	1.4	30	30	DI	
					13489.6	4.46	7.2	0.8	0.8	DI	
					15848.9	4.46	24.6	0.235	0.235	DI	
					11749.0	4.46	63.4	0.05	0.05	DI	
				nitrobenzene	38.0	1.85	0.4	170	170	DI	
					120.2	1.85	0.9	102	102	DI	
					213.8	1.85	1.3	50	50	DI	
					269.2	1.85	1.4	30	30	DI	
					436.5	1.85	7.2	0.8	0.8	DI	
					1380.4	1.85	24.6	0.235	0.235	DI	
					1071.5	1.85	63.4	0.05	0.05	DI	
Cui et al. 2022	Size-dependent adsorption of waterborne Benzophenone-3 on microplastics and its desorption under simulated gastrointestinal conditions	polystyrene	PS	benzophenone-3	215.4	3.18	1.79	550	550	DI	2,3,4a,4b
					296.0	3.18	2.57	250	250	DI	
					375.0	3.18	3.63	75	75	DI	
					536.4	3.18	21.55	5	5	DI	
					657.1	3.18	28.97	0.5	0.5	DI	
		polyethylene	PE		141.3	3.18	1.64	550	550	DI	
					227.3	3.18	1.90	250	250	DI	
					300.0	3.18	2.87	75	75	DI	
		polypropylene	PP		94.7	3.18	1.55	550	550	DI	
					211.1	3.18	1.84	250	250	DI	
					337.5	3.18	3.33	75	75	DI	
Song et al. 2021	Sorption and desorption of petroleum hydrocarbons on biodegradable and nondegradable microplastics	polylactic acid	OTHER	diesel oil (No. 10)	272.0	3.72	NR	50 - 200	125	DI	3,4a,4b
		polyethylene terephthalate	PET		325.0	3.72	NR	50 - 200	125	DI	

		polystyrene	PS		363.0	3.72	NR	50 - 200	125	DI	
		PVC	PVC		213.0	3.72	NR	50 - 200	125	DI	
		polyamide	PA		978.0	3.72	NR	50 - 200	125	DI	
		polyethylene	PE		415.0	3.72	NR	50 - 200	125	DI	
Zuo et al. 2019	Sorption and desorption of phenanthrene on biodegradable poly(butylene adipate co-terephthalate) microplastics	poly(butylene adipate co-terephtalate)	OTHER	phenanthrene	54800.0	4.46	2.21	2338 +/- 486	2338	SS	2,3,4a,4b
		polyethylene	PE		15600.0	4.46	4.4	2628 +/- 623	2628	SS	
					14100.0	4.46	2.47	400	400	SS	
		polystyrene	PS		1340.0	4.46	0.92	250	250	SS	
Razanajatovo et al. 2018	Sorption and desorption of selected pharmaceuticals by polyethylene microplastics	ultra-high molecular weight polyethylene	PE	sulfamethoxazole (SMX)	700.0	0.89	NR	45 - 48	46.5	DI	3,4a,4b
				propranolol (PRO)	2300.0	3.48	NR	45 - 48	46.5	DI	
				sertraline (SER)	3330.0	5.29	NR	45 - 48	46.5	DI	
Liu et al. 2019	Sorption behavior and mechanism of hydrophilic organic chemicals to virgin and aged microplastics in freshwater and seawater	polystyrene	PS	ciprofloxacin (CIP)	210.0	0.28	NR	75	75	DI	3,4a
		PVC	PVC		215.0	0.28	NR	75	75	DI	
Loncarski et al. 2021	Sorption behavior of polycyclic aromatic hydrocarbons on biodegradable polylactic acid and various nondegradable microplastics: Model fitting and mechanism analysis	powdered polyethylene	PE	naphthalene	4736.8	3.3	2.088	49.7 - 259	154.35	SF	2,3,4a,4b
					6666.7	3.3	2.088	49.7 - 259	154.35	F	
				fluorene	3928.6	4.18	2.088	49.7 - 259	154.35	SF	
					8250.0	4.18	2.088	49.7 - 259	154.35	F	
				fluoranthene	17727.3	5.16	2.088	49.7 - 259	154.35	SF	
					20833.3	5.16	2.088	49.7 - 259	154.35	F	
				pyrene	69500.0	4.88	2.088	49.7 - 259	154.35	SF	
					69500.0	4.88	2.088	49.7 - 259	154.35	F	
		granulated polyethylene	PE	naphthalene	4411.8	3.3	0.248	3000	3000	SF	
					3750.0	3.3	0.248	3000	3000	F	
				fluorene	6666.7	4.18	0.248	3000	3000	SF	
					5000.0	4.18	0.248	3000	3000	F	
				fluoranthene	11000.0	5.16	0.248	3000	3000	SF	
					11363.6	5.16	0.248	3000	3000	F	
				pyrene	22166.7	4.88	0.248	3000	3000	SF	
					16333.3	4.88	0.248	3000	3000	F	
		polyethylene_pcp1	PE	naphthalene	8863.6	3.3	0.66	80 - 358	219	SF	

					11500.0	3.3	0.66	80 - 358	219	F	
				fluorene	10227.3	4.18	0.66	80 - 358	219	SF	
					21000.0	4.18	0.66	80 - 358	219	F	
				fluoranthene	13000.0	5.16	0.66	80 - 358	219	SF	
					38285.7	5.16	0.66	80 - 358	219	F	
				pyrene	92666.7	4.88	0.66	80 - 358	219	SF	
					45000.0	4.88	0.66	80 - 358	219	F	
		polyethylene_pcp2	PE	naphthalene	13923.1	3.3	5.781	80 - 358	219	SF	
					10136.4	3.3	5.781	80 - 358	219	F	
				fluorene	10090.9	4.18	5.781	80 - 358	219	SF	
					6787.9	4.18	5.781	80 - 358	219	F	
				fluoranthene	15000.0	5.16	5.781	80 - 358	219	SF	
					38285.7	5.16	5.781	80 - 358	219	F	
				pyrene	141000.0	4.88	5.781	80 - 358	219	SF	
					140000.0	4.88	5.781	80 - 358	219	F	
		polyethylene terephthalate	PET	naphthalene	8076.9	3.3	0.164	3000	3000	SF	
					8684.2	3.3	0.164	3000	3000	F	
				fluorene	8750.0	4.18	0.164	3000	3000	SF	
					9545.5	4.18	0.164	3000	3000	F	
				fluoranthene	17727.3	5.16	0.164	3000	3000	SF	
					8076.9	5.16	0.164	3000	3000	F	
				pyrene	68750.0	4.88	0.164	3000	3000	SF	
					71666.7	4.88	0.164	3000	3000	F	
		polypropylene	PP	naphthalene	3315.8	3.3	NR	3000	3000	SF	
					3107.1	3.3	NR	3000	3000	F	
				fluorene	5000.0	4.18	NR	3000	3000	SF	
					4318.2	4.18	NR	3000	3000	F	
				fluoranthene	5625.0	5.16	NR	3000	3000	SF	
					6153.8	5.16	NR	3000	3000	F	
				pyrene	6562.5	4.88	NR	3000	3000	SF	
					7500.0	4.88	NR	3000	3000	F	

		polylactic acid	OTHER	naphthalene	675.7	3.3	0.094	3000	3000	SF	
					777.8	3.3	0.094	3000	3000	F	
				fluorene	561.4	4.18	0.094	3000	3000	SF	
					581.8	4.18	0.094	3000	3000	F	
				fluoranthene	673.1	5.16	0.094	3000	3000	SF	
					673.1	5.16	0.094	3000	3000	F	
				pyrene	1046.5	4.88	0.094	3000	3000	SF	
					1071.4	4.88	0.094	3000	3000	F	
Shan et al. 2020	Sorption behaviors of crude oil on polyethylene microplastics in seawater and digestive tract under simulated real-world conditions	polyethylene	PE	oil	276.0	3.72	NR	NR	NR	SS	3,4a,4b
					1222.2	3.72	NR	NR	NR	SS	
					520.8	3.72	NR	NR	NR	SS	
Wang et al. 2018	Sorption behaviors of phenanthrene on the microplastics identified in a mariculture farm in Xiangshan Bay, southeastern China	polyethylene	PE	phenanthrene	8140.0	4.46	0.129	220	220	SS	2,3,4a,4b
					4670.0	4.46	0.042	220	220	SS	
					2760.0	4.46	0.027	220	220	SS	
					8840.0	4.46	0.248	220	220	SS	
					8080.0	4.46	0.107	220	220	SS	
					6800.0	4.46	0.085	220	220	SS	
					2780.0	4.46	0.129	220	220	SS	
				phenol	50.0	1.46	0.129	220	220	SS	
		nylon	PA	phenanthrene	2240.0	4.46	NR	220	220	SS	
					2090.0	4.46	NR	220	220	SS	
				phenol	30.0	1.46	NR	220	220	SS	
Wang et al. 2019	Sorption behaviors of phenanthrene, nitrobenzene, and napthalene on mesoplastics and microplastics	high-density polyethylene	PE	phenanthrene	15848.9	4.46	0.352	50	50	DI	2,3,4a,4b
				nitrobenzene	107.2	1.85	0.352	50	50	DI	
				naphthalene	371.5	3.3	0.352	50	50	DI	
		low-density polyethylene	PE	phenanthrene	12589.3	4.46	1.579	12	12	DI	
				nitrobenzene	40.7	1.85	1.579	12	12	DI	
				naphthalene	812.8	3.3	1.579	12	12	DI	
		PVC	PVC	phenanthrene	16218.1	4.46	3.944	2	2	DI	
				nitrobenzene	676.1	1.85	3.944	2	2	DI	
				naphthalene	831.8	3.3	3.944	2	2	DI	

		polypropylene	PP	phenanthrene	19498.4	4.46	3.327	59	59	DI	
				nitrobenzene	1698.2	1.85	3.327	59	59	DI	
				naphthalene	12302.7	3.3	3.327	59	59	DI	
		polystyrene	PS	phenanthrene	602.6	4.46	0.482	261	261	DI	
				nitrobenzene	173.8	1.85	0.482	261	261	DI	
				naphthalene	588.8	3.3	0.482	261	261	DI	
Zhan et al. 2016	Sorption of 3,3′,4,4′-tetrachlorobiphenyl by microplastics_ A case study of polypropylene	polypropylene	PP	PCB-77	1182.6	6.6	NR	425 - 850	637.5	SS	3,4a
					556.4	6.6	NR	425 - 850	637.5	DI	
Lara et al. 2021	Sorption of endocrine disrupting compounds onto polyamide microplastics under different environmental conditions: Behaviour and mechanism	polyamide	PA	17B-estradiol (E2)	1000.0	4.01	0.3	<350	350	DI	2,3,4a
					1710.5	4.01	0.3	<350	350	SS	
				17a-ethynylestradiol (EE2)	1115.4	3.67	0.3	<350	350	DI	
					916.7	3.67	0.3	<350	350	SS	
				estriol (E3)	458.3	2.45	0.3	<350	350	DI	
					511.4	2.45	0.3	<350	350	SS	
Zhao et al. 2020	Sorption of five organic compounds by polar and nonpolar microplastics	polybutylene succinate	OTHER	pyrene	121000.0	4.88	0.04	150 - 200	175	DI	2,3,4a
				phenanthrene	15000.0	4.46	0.04	150 - 200	175	DI	
				1-nitronaphthalene (1-Mnap)	714.0	3.19	0.04	150 - 200	175	DI	
				1-naphthylamine	507.0	2.25	0.04	150 - 200	175	DI	
				atrazine (ATR)	68.8	2.61	0.04	150 - 200	175	DI	
		polycaprolactone	OTHER	pyrene	142000.0	4.88	NR	150 - 200	175	DI	
				phenanthrene	31800.0	4.46	NR	150 - 200	175	DI	
				1-nitronaphthalene (1-Mnap)	931.0	3.19	NR	150 - 200	175	DI	
				1-naphthylamine	518.0	2.25	NR	150 - 200	175	DI	
				atrazine (ATR)	58.1	2.61	NR	150 - 200	175	DI	
		polyurethane	OTHER	1-nitronaphthalene (1-Mnap)	1500.0	3.19	0.1	150 - 200	175	DI	
				1-naphthylamine	858.0	2.25	0.1	150 - 200	175	DI	
				atrazine (ATR)	76.1	2.61	0.1	150 - 200	175	DI	
		polystyrene	PS	pyrene	1780.0	4.88	0.19	150 - 200	175	DI	
				phenanthrene	820.0	4.46	0.19	150 - 200	175	DI	
				1-nitronaphthalene (1-Mnap)	95.3	3.19	0.19	150 - 200	175	DI	
				1-naphthylamine	29.6	2.25	0.19	150 - 200	175	DI	

Zhang et al. 2020	Sorption of fluoroquinolones to nanoplastics as affected by surface functionalization and solution chemistry*	nano-polystyrene	PS	norfloxacin (NOR)	8596.5	-1	19.48	0.055	0.055	DI	2,3,4a
					9500.0	-1	19.48	0.055	0.055	DI	
				levofloxacin (LEV)	4861.1	-0.39	85.63	0.05	0.05	DI	
					4285.7	-0.39	85.63	0.05	0.05	DI	
Puckowski et al. 2021	Sorption of pharmaceuticals on the surface of microplastics	low-density polyethylene	PE	enrofloxacin (ENR)	1.4	1.1	0.3884	63 - 125	94	DI	2,3,4a,4b
				ciprofloxacin (CIP)	1.7	0.4	0.3884	63 - 125	94	DI	
				norfloxacin (NOR)	1.6	0.46	0.3884	63 - 125	94	DI	
				propranolol (PRO)	1.5	3.48	0.3884	63 - 125	94	DI	
				nadolol (NAD)	0.2	0.85	0.3884	63 - 125	94	DI	
				methotrexate (MET)	0.3	-1.85	0.3884	63 - 125	94	DI	
				flubendazole (FLU)	1.4	3	0.3884	63 - 125	94	DI	
				fenbendazole (FEN)	0.6	3.8	0.3884	63 - 125	94	DI	
				phenanthrene	50.1	4.46	0.3884	63 - 125	94	DI	
					269.2	4.46	0.3884	63 - 125	94	DI	
		high-density polyethylene	PE	enrofloxacin (ENR)	0.7	1.1	0.3548	63 - 125	94	DI	
				ciprofloxacin (CIP)	0.7	0.4	0.3548	63 - 125	94	DI	
				norfloxacin (NOR)	1.4	0.46	0.3548	63 - 125	94	DI	
				propranolol (PRO)	2.1	3.48	0.3548	63 - 125	94	DI	
				nadolol (NAD)	0.1	0.85	0.3548	63 - 125	94	DI	
				5-fluorouracil (5-FU)	0.2	-1	0.3548	63 - 125	94	DI	
				flubendazole (FLU)	1.6	3	0.3548	63 - 125	94	DI	
				fenbendazole (FEN)	0.5	3.8	0.3548	63 - 125	94	DI	
		polypropylene	PP	enrofloxacin (ENR)	1.3	1.1	0.4836	63 - 125	94	DI	
				ciprofloxacin (CIP)	0.1	0.4	0.4836	63 - 125	94	DI	
				norfloxacin (NOR)	1.7	0.46	0.4836	63 - 125	94	DI	
				propranolol (PRO)	2.4	3.48	0.4836	63 - 125	94	DI	
				nadolol (NAD)	0.2	0.85	0.4836	63 - 125	94	DI	
				5-fluorouracil (5-FU)	0.1	-1	0.4836	63 - 125	94	DI	
				flubendazole (FLU)	1.3	3	0.4836	63 - 125	94	DI	
				fenbendazole (FEN)	0.3	3.8	0.4836	63 - 125	94	DI	
				phenanthrene	1.7	4.46	0.4836	63 - 125	94	DI	

		PVC	PVC		215.7	4.46	0.4836	63 - 125	94	DI	
				enrofloxacin (ENR)	0.3	1.1	0.6929	63 - 125	94	DI	
				ciprofloxacin (CIP)	0.5	0.4	0.6929	63 - 125	94	DI	
				norfloxacin (NOR)	0.4	0.46	0.6929	63 - 125	94	DI	
				propranolol (PRO)	1.3	3.48	0.6929	63 - 125	94	DI	
				nadolol (NAD)	0.1	0.85	0.6929	63 - 125	94	DI	
				5-fluorouracil (5-FU)	0.1	-1	0.6929	63 - 125	94	DI	
				methotrexate (MET)	0.1	-1.85	0.6929	63 - 125	94	DI	
				flubendazole (FLU)	1.6	3	0.6929	63 - 125	94	DI	
				fenbendazole (FEN)	0.4	3.8	0.6929	63 - 125	94	DI	
				phenanthrene	16.2	4.46	0.6929	63 - 125	94	DI	
					184.5	4.46	0.6929	63 - 125	94	DI	
Xu et al. 2019	Sorption of polybrominated diphenyl ethers by microplastics	polyethylene	PE	BDE-47	134.9	6.81	NR	4000	4000	DI	2,3,4a,4b
					251.2	6.81	NR	4000	4000	DI	
					281.8	6.81	NR	4000	4000	DI	
		polypropylene	PP		100.0	6.81	0.0004	4000	4000	DI	
					302.0	6.81	0.0004	4000	4000	DI	
					346.7	6.81	0.0004	4000	4000	DI	
		polyamide	PA		323.6	6.81	0.0017	4000	4000	DI	
					416.9	6.81	0.0017	4000	4000	DI	
					478.6	6.81	0.0017	4000	4000	DI	
		polystyrene	PS		245.5	6.81	0.0054	4000	4000	DI	
					457.1	6.81	0.0054	4000	4000	DI	
					512.9	6.81	0.0054	4000	4000	DI	
Liu et al. 2016	SORPTION OF POLYCYCLIC AROMATIC HYDROCARBONS TO POLYSTYRENE NANOPLASTIC	polystyrene	PS	chrysene	########	5.8	NR	65 +/- 25	65	DI	3,4a
					########	5.8	NR	65 +/- 25	65	DI	
					########	5.8	NR	65 +/- 25	65	DI	
				benzo(b)fluoranthene (BbF)	########	5.8	NR	65 +/- 25	65	DI	
					########	5.8	NR	65 +/- 25	65	DI	
					########	5.8	NR	65 +/- 25	65	DI	
				benzo(a)anthracene (BaA)	########	5.9	NR	65 +/- 25	65	DI	

					########	5.9	NR	65 +/- 25	65	DI	
					########	5.9	NR	65 +/- 25	65	DI	
				benzo(a)pyrene (BaP)	########	6	NR	65 +/- 25	65	DI	
					########	6	NR	65 +/- 25	65	DI	
					########	6	NR	65 +/- 25	65	DI	
				benzo(k)fluoranthene (BkF)	########	6.2	NR	65 +/- 25	65	DI	
					########	6.2	NR	65 +/- 25	65	DI	
					########	6.2	NR	65 +/- 25	65	DI	
				benzo(g,h,i)perylene (BgP)	########	6.9	NR	65 +/- 25	65	DI	
					########	6.9	NR	65 +/- 25	65	DI	
					########	6.9	NR	65 +/- 25	65	DI	
Qiu et al. 2019	Sorption of polyhalogenated carbazoles (PHCs) to microplastics	polypropylene	PP	3-BCZ	38.0	4.58	NR	<150	150	SS	3,4a,4b
				3,6-BCZ	58.0	5.43	NR	<150	150	SS	
				3,6-CCZ	136.0	5.16	NR	<150	150	SS	
				3,6-ICZ	242.0	5.95	NR	<150	150	SS	
				2,7-BCZ	118.0	5.26	NR	<150	150	SS	
		polyethylene	PE	3-BCZ	78.3	4.58	NR	<150	150	SS	
				3,6-BCZ	69.4	5.43	NR	<150	150	SS	
				3,6-CCZ	131.0	5.16	NR	<150	150	SS	
				2,6-ICZ	576.0	5.95	NR	<150	150	SS	
				2,7-BCZ	110.0	5.26	NR	<150	150	SS	
		PVC	PVC	3-BCZ	85.1	4.58	NR	<150	150	SS	
				3,6-BCZ	73.4	5.43	NR	<150	150	SS	
				3,6-CCZ	153.0	5.16	NR	<150	150	SS	
				2,6-ICZ	262.0	5.95	NR	<150	150	SS	
				2,7-BCZ	176.0	5.26	NR	<150	150	SS	
Guo et al. 2019	Sorption of sulfamethoxazole onto six types of microplastics	polyamide	PA	sulfamethoxazole (SMX)	284.0	0.89	NR	100 - 150	125	DI	3,4a,4b
		polyethylene	PE		30.0	0.89	NR	100 - 150	125	DI	
		polystyrene	PS		29.7	0.89	NR	100 - 150	125	DI	
		polyethylene terephthalate	PET		22.2	0.89	NR	100 - 150	125	DI	
		PVC	PVC		28.2	0.89	NR	100 - 150	125	DI	

		polypropylene	PP		30.9	0.89	NR	100 - 150	125	DI	
Li et al. 2021	Sorption of tetrabromobisphenol A onto microplastics: Behavior, mechanisms, and the effects of sorbent and environmental factors	polyethylene	PE	tetrabromobisphenol-A	616.0	4.75	1.0159	161	161	DI	2,3,4a,4b
		polypropylene	PP		993.0	4.75	0.2218	218	218	DI	
		polystyrene	PS		2263.0	4.75	1.9241	103	103	DI	
		PVC	PVC		5814.0	4.75	2.6334	120	120	DI	
Elizalde-Velazquez et al. 2020	Sorption of three common nonsteroidal anti-inflammatory drugs to microplastics	ultra high molecular weight polyethylene	PE	ibuprofen (IBU)	7.5	3.97	NR	<10	10	SS	3,4a,4b
					16.0	3.97	NR	<10	10	SF	
					14.0	3.97	NR	<10	10	SF	
				diclofenac (DCF)	6.9	4.51	NR	<10	10	SS	
					32.3	4.51	NR	<10	10	SF	
					12.4	4.51	NR	<10	10	SF	
				naproxen (NPX)	1.6	3.18	NR	<10	10	SS	
					4.5	3.18	NR	<10	10	SF	
					3.4	3.18	NR	<10	10	SF	
		average molecular weight polyethylene	PE	ibuprofen (IBU)	8.8	3.97	NR	300 - 400	350	SS	
					13.3	3.97	NR	300 - 400	350	SF	
					12.5	3.97	NR	300 - 400	350	SF	
				diclofenac (DCF)	2.5	4.51	NR	300 - 400	350	SS	
					26.2	4.51	NR	300 - 400	350	SF	
					11.5	4.51	NR	300 - 400	350	SF	
				naproxen (NPX)	0.6	3.18	NR	300 - 400	350	SS	
					4.3	3.18	NR	300 - 400	350	SF	
					1.3	3.18	NR	300 - 400	350	SF	
		polystyrene	PS	ibuprofen (IBU)	5.5	3.97	NR	600 - 800	700	SS	
					7.8	3.97	NR	600 - 800	700	SF	
					8.4	3.97	NR	600 - 800	700	SF	
				diclofenac (DCF)	2.5	4.51	NR	600 - 800	700	SS	
					27.9	4.51	NR	600 - 800	700	SF	
					12.6	4.51	NR	600 - 800	700	SF	
				naproxen (NPX)	1.6	3.18	NR	600 - 800	700	SS	
					2.4	3.18	NR	600 - 800	700	SF	

					7.1	3.18	NR	600 - 800	700	SF	
		polypropylene	PP	ibuprofen (IBU)	1.7	3.97	NR	1000	1000	SS	
					2.5	3.97	NR	1000	1000	SF	
					4.5	3.97	NR	1000	1000	SF	
				diclofenac (DCF)	1.6	4.51	NR	1000	1000	SS	
					12.3	4.51	NR	1000	1000	SF	
					2.8	4.51	NR	1000	1000	SF	
				naproxen (NPX)	0.7	3.18	NR	1000	1000	SS	
					1.9	3.18	NR	1000	1000	SF	
					1.6	3.18	NR	1000	1000	SF	
Guo et al. 2018	Sorption properties of tylosin on four different microplastics	polyethylene	PE	tylosin	62.8	1.63	0.173	NR	NR	DI	3,4a,4b
		polypropylene	PP		94.1	1.63	0.348	NR	NR	DI	
		polystyrene	PS		134.1	1.63	0.508	NR	NR	DI	
		PVC	PVC		155.3	1.63	0.836	NR	NR	DI	
Velzeboer et al. 2014	Strong Sorption of PCBs to Nanoplastics, Microplastics, Carbon Nanotubes, and Fullerenes	polyethylene	PE	PCB-28	204173.8	5.67	NR	100	100	F	3,4a,4b
					1698243.7	5.67	NR	100	100	S	
				PCB-31	100000.0	5.67	NR	100	100	F	
					1122018.5	5.67	NR	100	100	S	
				PCB-44	89125.1	5.75	NR	100	100	F	
					1071519.3	5.75	NR	100	100	S	
				PCB-52	128825.0	5.84	NR	100	100	F	
					1096478.2	5.84	NR	100	100	S	
				PCB-74	851138.0	6.2	NR	100	100	F	
					########	6.2	NR	100	100	S	
				PCB-101	380189.4	6.38	NR	100	100	F	
					3548133.9	6.38	NR	100	100	S	
				PCB-105	1513561.2	6.65	NR	100	100	F	
					########	6.65	NR	100	100	S	
				PCB-149	1621810.1	6.67	NR	100	100	F	
					1412537.5	6.67	NR	100	100	S	
				PCB-118	1995262.3	6.74	NR	100	100	F	

					1995262.3	6.74	NR	100	100	S	
				PCB-138	8128305.2	6.83	NR	100	100	F	
					########	6.83	NR	100	100	S	
				PCB-153	########	6.92	NR	100	100	F	
					########	6.92	NR	100	100	S	
				PCB-156	########	7.18	NR	100	100	F	
					########	7.18	NR	100	100	S	
				PCB-170	########	7.27	NR	100	100	F	
					########	7.27	NR	100	100	S	
				PCB-180	########	7.36	NR	100	100	F	
					########	7.36	NR	100	100	S	
				PCB-77	1659586.9	6.36	NR	100	100	F	
					616595.0	6.36	NR	100	100	S	
				PCB-126	########	6.89	NR	100	100	F	
					########	6.89	NR	100	100	S	
				PCB-169	########	7.42	NR	100	100	F	
					########	7.42	NR	100	100	S	
Jiang et al. 2020	Strong sorption of two fungicides onto biodegradable microplastics with emphasis on the negligible role of environmental factors*	polybutylene succinate	OTHER	triadimefon	1000.0	3.18	0.601	75 - 150	112.5	DI	2,3,4a,4b
				difenoconazole (DIFE)	28890.0	4.36	0.601	75 - 150	112.5	DI	
		polyethylene	PE	triadimefon	240.0	3.18	0.553	75 - 150	112.5	DI	
				difenoconazole (DIFE)	2040.0	4.36	0.553	75 - 150	112.5	DI	
		PVC	PVC	triadimefon	150.0	3.18	0.144	75 - 150	112.5	DI	
				difenoconazole (DIFE)	160.0	4.36	0.144	75 - 150	112.5	DI	
Karapanagioti and Klontza 2008	Testing phenanthrene distribution properties of virgin plastic pellets and plastic eroded pellets found on Lesvos island beaches (Greece)	polyethylene	PE	phenanthrene	9200.0	4.46	NR	2000-3000	2500	SF	3,4a,4b
					13000.0	4.46	NR	2000-3000	2500	SS	
		polypropylene	PP		270.0	4.46	NR	2000	2000	SF	
					380.0	4.46	NR	2000	2000	SS	
		polyoxymethylene	OTHER		5300.0	4.46	NR	2000-3000	2500	SF	
					7400.0	4.46	NR	2000-3000	2500	SS	
Hu et al. 2021	The effect of microplastics on behaviors of chiral imidazolinone herbicides in the aquatic environment: Residue, degradation and distribution	high-density polypropylene	PP	imazapic	40.3	1.6	NR	1 - 10	4.5	F	3,4a
				imazamox	21.4	0.9	NR	1 - 10	4.5	F	

				imazethapyr	21.7	2	NR	1 - 10	4.5	F	
Muller et al. 2018	The effect of polymer aging on the uptake of fuel aromatics and ethers by microplastics*	polypropylene	PP	MTBE	1.0	1.12	NR	3000 - 5000	4000	DI	3,4a
				ETBE	2.2	1.54	NR	3000 - 5000	4000	DI	
				TAME	3.1	1.68	NR	3000 - 5000	4000	DI	
				TAEE	6.6	2.41	NR	3000 - 5000	4000	DI	
				benzene	26.9	2.13	NR	3000 - 5000	4000	DI	
				toluene	83.2	2.73	NR	3000 - 5000	4000	DI	
				ethyl-benzene	199.5	3.15	NR	3000 - 5000	4000	DI	
				m-xylene (m-Xyl)	218.8	3.2	NR	3000 - 5000	4000	DI	
				o-xylene	144.5	3.12	NR	3000 - 5000	4000	DI	
		polystyrene	PS	benzene	10.2	2.13	NR	3500	3500	DI	
				toluene	23.4	2.73	NR	3500	3500	DI	
				ethyl-benzene	39.8	3.15	NR	3500	3500	DI	
				m-xylene (m-Xyl)	37.2	3.2	NR	3500	3500	DI	
				o-xylene	18.2	3.12	NR	3500	3500	DI	
Yu et al. 2020	The effects of functional groups on the sorption of naphthalene on microplastics	polystyrene	PS	naphthalene	11965.0	3.3	NR	10	10	DI	3,4a
					6047.0	2.3	NR	10	10	DI	
					11640.0	3.9	NR	10	10	DI	
					8394.0	3.1	NR	10	10	DI	
					6350.0	2.9	NR	10	10	DI	
					9351.0	3.3	NR	10	10	DI	
					6321.0	2.3	NR	10	10	DI	
					9974.0	3.9	NR	10	10	DI	
					4536.0	3.1	NR	10	10	DI	
					4551.0	2.9	NR	10	10	DI	
Huffer et al. 2020	The molecular interaction of organic compounds with tire crumb materials differ substaintially from those with other microplastics	styrene butadiene	OTHER	n-hexane	10232.9	3.29	NR	125 - 250	187.5	DI	3,4a
				cyclohexane	3890.5	3.21	NR	125 - 250	187.5	DI	
				benzene	151.4	1.99	NR	125 - 250	187.5	DI	
				chlorobenzene	660.7	2.64	NR	125 - 250	187.5	DI	
				di-n-propylether	182.0	2.03	NR	125 - 250	187.5	DI	
				2,6-dimethylheptan-2-ol	251.2	3.11	NR	125 - 250	187.5	DI	

		TCR-S	OTHER	n-hexane	7762.5	3.29	NR	125 - 250	187.5	DI	
				cyclohexane	2238.7	3.21	NR	125 - 250	187.5	DI	
				benzene	107.2	1.99	NR	125 - 250	187.5	DI	
				chlorobenzene	316.2	2.64	NR	125 - 250	187.5	DI	
				di-n-propylether	229.1	2.03	NR	125 - 250	187.5	DI	
				2,6-dimethylheptan-2-ol	602.6	3.11	NR	125 - 250	187.5	DI	
		TCR-R	OTHER	n-hexane	22908.7	3.29	NR	125 - 250	187.5	DI	
				cyclohexane	3981.1	3.21	NR	125 - 250	187.5	DI	
				benzene	186.2	1.99	NR	125 - 250	187.5	DI	
				chlorobenzene	288.4	2.64	NR	125 - 250	187.5	DI	
				di-n-propylether	123.0	2.03	NR	125 - 250	187.5	DI	
				2,6-dimethylheptan-2-ol	457.1	3.11	NR	125 - 250	187.5	DI	
Wang et al. 2015	The partition behavior of perfluorooctanesulfonate (PFOS) and perfluorooctanesulfonamide (FOSA) on microplastics	polyethylene	PE	PFOS	32.8	4.9	NR	150	150	DI	3,4a,4b
				FOSA	298.3	4.8	NR	150	150	DI	
		polystyrene	PS	FOSA	84.9	4.8	NR	250	250	DI	
		PVC	PVC	PFOS	100.5	4.9	NR	230	230	DI	
				FOSA	115.7	4.8	NR	230	230	DI	
Godoy et al. 2020	The relevance of interaction of chemicals/pollutants and microplastic samples as route for transporting contaminants	polyethylene terephthalate	PET	amoxicillin	1222.2	0.87	NR	2700	2700	DI	3,4a,4b
				phenol	2076.9	1.46	NR	2700	2700	DI	
		polystyrene	PS	amoxicillin	2181.8	0.87	NR	3100	3100	DI	
				phenol	1666.7	1.46	NR	3100	3100	DI	
		PVC	PVC	amoxicillin	2636.4	0.87	NR	4300	4300	DI	
				phenol	9000.0	1.46	NR	4300	4300	DI	
		polypropylene	PP	amoxicillin	1666.7	0.87	NR	3400	3400	DI	
				phenol	863.6	1.46	NR	3400	3400	DI	
		polyethylene	PE	amoxicillin	2166.7	0.87	NR	3500	3500	DI	
Xu et al. 2018	The sorption kinetics and isotherms of sulfamethoxazole with polyethylenemicroplastics	polyethylene	PE	sulfamethoxazole (SMX)	591.7	0.89	0.2341	150	150	DI	2,3,4a,4b
Bakir et al. 2014	Transport of persistent organic pollutants by microplastics in estuarine conditions	PVC	PVC	phenanthrene	2000.0	4.46	NR	200 - 250	225	DI	3,4a,4b
					2250.0	4.46	NR	200 - 250	225	SS	
					2200.0	4.46	NR	200 - 250	225	SS	

					2000.0	4.46	NR	200 - 250	225	SS	
					2250.0	4.46	NR	200 - 250	225	SS	
				DDT	150000.0	6.36	NR	200 - 250	225	DI	
					80000.0	6.36	NR	200 - 250	225	SS	
					125000.0	6.36	NR	200 - 250	225	SS	
					125000.0	6.36	NR	200 - 250	225	SS	
					110000.0	6.36	NR	200 - 250	225	SS	
		polyethylene	PE	phenanthrene	4900.0	4.46	NR	200 - 250	225	DI	
					4850.0	4.46	NR	200 - 250	225	SS	
					5000.0	4.46	NR	200 - 250	225	SS	
					5300.0	4.46	NR	200 - 250	225	SS	
					5200.0	4.46	NR	200 - 250	225	SS	
				DDT	160000.0	6.36	NR	200 - 250	225	DI	
					90000.0	6.36	NR	200 - 250	225	SS	
					125000.0	6.36	NR	200 - 250	225	SS	
					120000.0	6.36	NR	200 - 250	225	SS	
					95000.0	6.36	NR	200 - 250	225	SS	
Papers with No Kd (or unable to interpret Kd from results)											
Seidensticker et al. 2018	A combined experimental and modeling study to evaluate pH-dependent sorption of polar and non-polar compounds to polyethylene and polystyrene microplastics	polyethylene	PE	List of 19 compounds			NR	260	260	DI	3
		polystyrene	PS				NR	250	260	DI	
Llorca et al. 2020	Adsorption and Desorption Behaviour of Polychlorinated Biphenyls onto Microplastics' Surfaces in Water/Sediment Systems	polyethylene	PE	PCB-28			NR	1 - 300	150.5	SS	3
				PCB-52			NR	1 - 300	150.5	SS	
				PCB-101			NR	1 - 300	150.5	SS	
				PCB-118			NR	1 - 300	150.5	SS	
				PCB-153			NR	1 - 300	150.5	SS	
				PCB-138			NR	1 - 300	150.5	SS	
				PCB-180			NR	1 - 300	150.5	SS	
		polyethylene terephthalate	PET	PCB-28			NR	1 - 300	150.5	SS	
				PCB-52			NR	1 - 300	150.5	SS	
				PCB-101			NR	1 - 300	150.5	SS	
				PCB-118			NR	1 - 300	150.5	SS	

		polystyrene	PS	PCB-153		NR	1 - 300	150.5	SS	
				PCB-138		NR	1 - 300	150.5	SS	
				PCB-180		NR	1 - 300	150.5	SS	
				PCB-28		NR	1 - 300	150.5	SS	
				PCB-52		NR	1 - 300	150.5	SS	
				PCB-101		NR	1 - 300	150.5	SS	
				PCB-118		NR	1 - 300	150.5	SS	
				PCB-153		NR	1 - 300	150.5	SS	
				PCB-138		NR	1 - 300	150.5	SS	
				PCB-180		NR	1 - 300	150.5	SS	
Fan et al. 2021	Adsorption and desorption behaviors of antibiotics by tire wear particles and polyethylene microplastics with or without aging processes	tire wear particles	OTHER	chlortetracycline (CTC)		0.401	>74	74	DI	2,3
				amoxicillin (AMX)		0.401	>74	74	DI	
		polyethylene	PE	chlortetracycline (CTC)		0.287	>74	74	DI	
				amoxicillin (AMX)		0.287	>74	74	DI	
Wu et al. 2019	Adsorption mechanisms of five bisphenol analogues on PVC microplastics	PVC	PVC	bisphenol A (BPA)		9.77	13.2	13.2	DI	2,3
				bisphenol AF (BPAF)		9.77	13.2	13.2	DI	
				bisphenol B (BPB)		9.77	13.2	13.2	DI	
				bisphenol F (BPF)		9.77	13.2	13.2	DI	
				bisphenol S (BPS)		9.77	13.2	13.2	DI	
Bao et al. 2021	Adsorption of phenanthrene and its monohydroxy derivatives on polyvinyl chloride microplastics in aqueous solution: Model fitting and mechanism analysis	PVC	PVC	1-OHP		0.595	140	140	DI	2,3
				2-OHP		0.595	140	140	DI	
				4-OHP		0.595	140	140	DI	
				9-OHP		0.595	140	140	DI	
				phenanthrene		0.595	140	140	DI	
Chen et al. 2021	Adsorption of tetracyclines onto polyethylene microplastics: A combined study of experiment and molecular dynamics simulation	polyethylene	PE	tetracycline (TC)		0.0454	150 - 425	287.5	OTHER	2,3
				chlortetracycline (CTC)		0.0454	150 - 425	287.5	OTHER	
				oxytetracycline (OTC)		0.0454	150 - 425	287.5	OTHER	
Li et al. 2021	Adsorption of three pesticides on polyethylene microplastics in aqueous solutions: Kinetics, isotherms, thermodynamics, and molecular dynamics simulation	polyethylene	PE	imidacloprid		NR	710 - 850	780	SS	3
				buprofezin		NR	710 - 850	780	SS	
				difenoconazole (DIFE)		NR	710 - 850	780	SS	
			PET	atenolol		NR	2300 - 5000	3650	F	3

Magadini et al. 2020	Assessing the sorption of pharmaceuticals to microplastics through in-situ experiments in New York City waterways	polyethylene terephthalate		ibuprofen (IBU)		NR	2300 - 5000	3650	F	
				sulfamethoxazole (SMX)		NR	2300 - 5000	3650	F	
		high-density polyethylene	PE	atenolol		NR	2300 - 5000	3650	F	
				ibuprofen (IBU)		NR	2300 - 5000	3650	F	
				sulfamethoxazole (SMX)		NR	2300 - 5000	3650	F	
		PVC	PVC	atenolol		NR	2300 - 5000	3650	F	
				ibuprofen (IBU)		NR	2300 - 5000	3650	F	
				sulfamethoxazole (SMX)		NR	2300 - 5000	3650	F	
		low-density polyethylene	PE	atenolol		NR	2300 - 5000	3650	F	
				ibuprofen (IBU)		NR	2300 - 5000	3650	F	
				sulfamethoxazole (SMX)		NR	2300 - 5000	3650	F	
		polypropylene	PP	atenolol		NR	2300 - 5000	3650	F	
				ibuprofen (IBU)		NR	2300 - 5000	3650	F	
				sulfamethoxazole (SMX)		NR	2300 - 5000	3650	F	
Yurtsever et al. 2020	Hydrophobic Pesticide Endosulfan (α + β) and Endrin Sorption on Different Types of Microplastics	low-density polyethylene	PE	endosulfan		NR	100 - 800	450	DI/F	3
				endrin		NR	100- 800	450	DI/F	
		polypropylene	PP	endosulfan		NR	25 - 250	137.5	DI/F	
				endrin		NR	25 - 250	137.5	DI/F	
		PVC	PVC	endosulfan		NR	50 - 350	200	DI/F	
				endrin		NR	50 - 350	200	DI/F	
		polystyrene	PS	endosulfan		NR	3000	3000	DI/F	
				endrin		NR	3000	3000	DI/F	
		polyamide	PA	endosulfan		NR	3000	3000	DI/F	
				endrin		NR	3000	3000	DI/F	
		polyethylene terephthalate	PET	endosulfan		NR	5000	3000	DI/F	
				endrin		NR	5000	3000	DI/F	
Scutariu et al. 2019	In Vitro Sorption Study of Some Organochlorine Pesticides on Polyethylene Terephthalate Microplastics	PET	PET	a-HCH		NR	500 - 5000	2750	F	3
				b-HCH		NR	500 - 5000	2750	F	
				gamma-HCH		NR	500 - 5000	2750	F	
				sigma-HCH		NR	500 - 5000	2750	F	
				heptachlor		NR	500 - 5000	2750	F	

				aldrin		NR	500 - 5000	2750	F	
				heptachlor epoxide		NR	500 - 5000	2750	F	
				endosulfan I		NR	500 - 5000	2750	F	
				4,4-DDE		NR	500 - 5000	2750	F	
				dieldrin		NR	500 - 5000	2750	F	
				endrin		NR	500 - 5000	2750	F	
				endosulfan II		NR	500 - 5000	2750	F	
				4,4-DDD		NR	500 - 5000	2750	F	
				endrin aldehyde		NR	500 - 5000	2750	F	
				4,4-DDT		NR	500 - 5000	2750	F	
				endosulfan sulfate		NR	500 - 5000	2750	F	
Hu et al. 2017	Microscopic investigation on the adsorption of lubrication oil on microplastics	micropolystyrene	PS	oil		0.294	169	169	NR	2,3
		nanopolyethylene	PE	oil		0.532	56.45	56.45	NR	
Rochman et al. 2013	Polystyrene Plastic: A Source and Sink for Polycyclic Aromatic Hydrocarbons in the Marine Environment	polystyrene	PS	List of 25 compounds		NR	2000	2000	S	3
Liu et al. 2018	Sorption behaviors of tris-(2,3-dibromopropyl) isocyanurate and hexabromocyclododecanes on polypropylene microplastics	polystyrene	PS	TBC		NR	450 - 850	650	SS	3
				a-HBCD		NR	450 - 850	650	SS	
				b-HBCD		NR	450 - 850	650	SS	
				gamma-HBCD		NR	450 - 850	650	SS	
Lee et al. 2014	Sorption capacity of plastic debris for hydrophobic organic chemicals	polyethylene	PE			NR	420	420	SS	3
		polypropylene	PP			NR	440	440	SS	
		polystyrene	PS			NR	320	320	SS	
Zhang et al. 2019	Sorption of 3,6-dibromocarbazole and 1,3,6,8-tetrabromocarbazole by microplastics	polypropylene	PP	3,6-BCZ		NR	2000 - 5000	3500	SS	3
				3,6-BCZ		NR	150 - 450	300	SS	
				1,3,6,8-BCZ		NR	2000 - 5000	3500	SS	
				1,3,6,8-BCZ		NR	150 - 450	300	SS	
Guo et al. 2012	Sorption of Four Hydrophobic Organic Compounds by Three Chemically Distinct Polymers_ Role of Chemical and Physical Composition	polyethylene	PE	phenanthrene, lidane, naphthalene, and 1-naphthol		6.1	>150	150	NR	2,3
						1.4	>150	150	NR	
						5.1	>150	150	NR	
						2.2	>150	150	NR	
						10.7	>150	150	NR	

						1.6	>150	150	NR	
						0.2	>150	150	NR	
		polystyrene	PS			0.2	>150	150	NR	
						70.2	>150	150	NR	
		polyphenyleneoxide (PPO)	OTHER			64.6	>150	150	NR	
Uber et al. 2019	Sorption of non-ionic organic compounds by polystyrene in water	polystyrene	PS	List of 24 compounds		NR	29	29	DI	3
Huffer and Hofmann, 2016	Sorption of non-polar organic compounds by micro-sized plastic particles in aqueous solution	polyethylene	PE	n-hexane, cyclohexane, benzene, toluene, chlorobenzene, ethylbenzoate, naphthalene		0.308	152.53	152.53	NR	2,3
		polystyrene	PS			0.338	168.55	168.55	NR	
		polyamide	PA			0.156	109.44	109.44	NR	
		PVC	PVC			0.317	57.64	57.64	NR	
Huffer et al. 2018	Sorption of organic compounds by aged polystyrene microplastics	PS	PS	List of 21 compounds		0.859	125-250		DI	2,3
Fries and Zarfl, 2012	Sorption of polycyclic aromatic hydrocarbons (PAHs) to low and high density polyethylene (PE)	high-density polyethylene	PE	Mix of acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, and fluoroanthene		NR	2000	2000	DI	3
		low-density polyethylene	PE			NR	2800	2800	DI	
Dong et al. 2019	Sorption of Tonalide, Musk Xylene, Galaxolide, and Musk Ketone by microplastics of PE and PVC	polyethylene	PE	tonalide, musk xylene, galaxolide, musk ketone		NR	<150	150	DI to SS	3
		PVC	PVC			NR	<150	150	DI to SS	
Chen et al. 2019	Sorption of tri-n-butyl phosphate and tris(2-chloroethyl) phosphate on polyethylene and polyvinyl chloride microplastics in seawater	polyethylene	PE	tri-n-butyl phosphate		NR	1000 - 5000, 425 - 1000, 125 - 425, and 45 - 125		S	3
				tris(2-chloroethyl) phosphate		NR			S	
		PVC	PVC	tri-n-butyl phosphate		NR			S	
				tris(2-chloroethyl) phosphate		NR			S	
Xia et al. 2020	Strong influence of surfactants on virgin hydrophobic microplastics adsorbing ionic organic pollutants*	PVC	PVC	methylene blue		NR	200	200	SS	3
		polyethylene	PE			NR	200	200	SS	
		polypropylene	PP			NR	200	200	SS	
		polystyrene	PS			NR	200	200	SS	

BIOGRAPHY OF AUTHOR

Ashton McCormack Collins was born in Biddeford, Maine in 1997. He attended the Biddeford school system growing up, graduating from Biddeford high school in 2015. Ashton decided to attend Susquehanna University in Pennsylvania for his bachelor’s degree. At the university, he also was a part of the division III men’s tennis team. He graduated from the university in May of 2019 with a Bachelor’s of Science degree in earth and environmental science. After graduation, he got hired at Nelson Analytical in Kennebunk, Maine as an environmental analyst conducting water and soil testing for the commercial and residential communities of the southern Maine area. Realizing to progress in the environmental field, he decided to attend the University of Maine to pursue a master’s degree to advance his knowledge of the natural environment. He is a candidate for the Masters of Science degree in Civil Engineering from the University of Maine in August 2022.