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## **Impacts of Chemical Toxicity of Plastics Pollution to Aquatic Life**

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## **Notices**

This document provides a summary of the state of the science on the potential chemical toxicity of ingested plastic and associated chemicals on aquatic organisms and aquatic-dependent wildlife. While this document reflects EPA's assessment of the best available science for plastics pollution, it is not a regulation and does not impose legally binding requirements on EPA, states, tribes, or the regulated community, and might not apply to a particular situation based upon the circumstances. EPA may change this document in the future. This document has undergone contractor-led external peer review as well as a review process within the EPA. Final review by EPA's Office of Science and Technology, Health and Ecological Criteria Division has been completed and the document has been approved for publication.

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## Acronyms

<b>Acronym</b>	<b>Definition</b>
BBP	Benzylbutylphthalate
BPA	Bisphenol A
CHAP	Chronic Hazard Advisory Panel on Phthalates and Phthalate Alternatives
CPR	Continuous Plankton Recorder
DBP	Dibutylphthalate
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlorodiphenyltrichloroethane
DEHP	Diethylhexylphthalate
DEP	Diethylphthalate
DMP	Dimethylphthalate
GESAMP	Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection
IMDCC	Interagency Marine Debris Coordinating Committee
IPW	International Pellet Watch
JRC-IHCP	Joint Research Centre, Institute for Health and Consumer Protection (European Commission)
$K_{ow}$	Octanol-Water Partitioning Coefficient
$K_{pw}$	Polymer- Water Partitioning Coefficient
NOAA	National Oceanic and Atmospheric Administration
NP	Nonylphenol
NPE	Nonylphenol ethoxylate
OCP	Organochlorine Pesticide
PAH	Polycyclic aromatic hydrocarbon
PBDE	Polybrominated diphenyl ether
PBT	Persistent, Bioaccumulative, and Toxic
PCB	Polychlorinated biphenyl
PET	Polyethylene terephthalate
PFOA	Perfluorooctanoic acid
PFOS	Perfluorooctanesulfonic acid



<b>Acronym</b>	<b>Definition</b>
PTFE	Polytetrafluoroethylene
PUR	Polyurethane
PVC	Polyvinyl chloride
SCBD	Secretariat of the Convention on Biological Diversity
UNEP	United Nations Environmental Programme
U.S. EPA	United States Environmental Protection Agency

## Executive Summary

The purpose of this report is to synthesize the state of the science on the potential chemical toxicity of ingested plastic and associated chemicals on aquatic organisms and aquatic-dependent wildlife. The focus of this document is primarily on marine systems, with data provided on the Great Lakes and other freshwater systems, where available.

Since mass-production of plastics began in the 1940s and 1950s, the amount of plastic debris entering marine and freshwater ecosystems has increased by several orders of magnitude (Cole et al., 2011). However, recently the accumulation and potential impacts of plastic pollution has been recognized as an emerging environmental issue (GESAMP 2015; UNEP 2016). Recent estimates suggest that 4.8 to 12.7 million metric tons of plastic waste entered the global marine environment in 2010 (Jambeck et al., 2015). Plastics, including bags, packing materials, water bottles, and fishing line and their breakdown products are now found throughout marine ecosystems and in the Great Lakes and other freshwater ecosystems, including near urban and remote beaches, in the open ocean, in sediments, within the water column, and in Arctic Sea ice (Derraik, 2002; Law & Thompson 2014; Eerkes-Medrano et al., 2015). Plastic particles are generally the most abundant type of debris encountered in the marine environment with estimates suggesting that plastics comprise between 60% and 80% of total marine debris (Derraik, 2002).

Plastics found in the aquatic environment are generally categorized as *macroplastics* (i.e., items > 5 mm diameter, such as disposable cups, bottles, and shipping pallets) and *microplastics* (i.e., items < 5 mm diameter, such as microbeads and fishing line fragments) (Lambert et al., 2014). As a subcategory of microplastics, *nanoplastics* (<100 nm size range, as defined in Koelmans et al., 2015) are likely to occur from both primary and secondary sources; methods do not currently exist to detect nanoplastics in the natural aquatic environment (Koelmans et al., 2015). The abundance of microplastics (compared to macroplastics) in the marine environment is increasing, according to a state-of-the-science report conducted by Canada's Secretariat of the Convention on Biological Diversity (SCBD) (SCBD, 2012). Recent investigations have found the majority (90%) of plastic debris found in the pelagic environment (i.e., open ocean) is generally less than 5 mm in diameter (Eriksen et al., 2013; Browne et al., 2010; Thompson et al., 2004; Rochman et al., 2014b).

The harmful physical impacts (i.e., entanglement, smothering, or physical effects of ingestion) of plastic to aquatic invertebrates, fish, seabirds, sea turtles, and marine mammals has been well-documented (Kühn et al., 2015). In the aquatic environment, the ingestion of plastics also establishes a potential exposure pathway for other chemical contaminants including metals, and persistent, bioaccumulative, and toxic contaminants that may be sorbed from the water column to plastic or incorporated into the plastics during manufacture (Engler 2012). Given the potential for plastics to be a source of contaminants, from both the chemical constituents of the manufactured plastic itself and contaminants sorbed to plastics in the aquatic environment, there is growing concern about the toxicological impacts of chemicals associated with plastics on aquatic organisms, as well as, aquatic-dependent wildlife, such as seabirds (Teuten et al., 2009).

Because plastics have become pervasive in oceans, coasts, and inland watersheds, and there are concerns about the potential toxicological impacts of chemicals associated with plastics on aquatic organisms and aquatic-dependent wildlife, the United States Environmental Protection Agency's (EPA) Office of Water produced a state-of-the-science review that summarizes available scientific information on the effects of chemicals associated with plastic pollution and the potential impact of these chemical on aquatic life and aquatic-dependent wildlife. Furthermore, EPA's Office of Water is committed to reducing and preventing

aquatic debris from entering U.S. waterways and the ocean. The Agency works toward this goal through the Trash Free Waters Program, <https://www.epa.gov/trash-free-waters>, which builds upon the foundation of earlier Agency work through the Marine Debris Program. The Trash Free Waters program has been designed with a strong emphasis on working with government, business, and citizen stakeholders to reduce and prevent plastic trash and debris from entering both freshwater and coastal ecosystems. The EPA's Trash Free Waters Program is intended to be a catalyst for proactive trash prevention and reduction, using tools including research, public outreach and education, public/private partnerships, support for state and local trash prevention efforts, and engagement with international trash free ocean programs.

Scientific research including field studies (e.g., Yamashita et al, 2011; Lavers et al., 2014; Rochman et al., 2014b) and laboratory studies (e.g., Teuten et al., 2009; Besseling et al., 2013; Rochman et al., 2013a) suggests that several groups of aquatic or aquatic-dependent organisms (invertebrates, fish, and birds) can accumulate chemicals associated with plastics once ingested. Experimental studies investigating the effects of chemicals associated with plastics on invertebrates and fish indicate that there are negative sublethal effects on these organisms from chemicals associated with plastics as well as the plastic itself (e.g., Rochman et al., 2013a, 2014c; Avio et al., 2015). However, some bioaccumulation modeling approaches attempting to simulate environmentally realistic scenarios of exposure provide indirect evidence that the role of plastics in contributing to body burdens and effects of chemical pollutants may be relatively small compared with other exposure pathways, such as direct chemical exposure via water, sediment, or ingestion of contaminated prey (Koelmans et al., 2016; Bakir et al., 2016, Ziccardi et al., 2016). As identified in this report, there are significant opportunities for research to further our understanding of the actual impacts of the potential chemical toxicity of plastic ingestion throughout the food web.

Further research is needed to gain knowledge of the extent to which plastics transfer contaminants to organisms compared to other sources, as well as the toxicological impacts of plastic ingestion compared to other environmental stressors, in particular the following:

1. Studies to gain a better understanding of the fate of chemicals both sorbed to and in plastics under differing environment conditions and within an organism after ingestion.
2. Research on the relative role plastics play in chemical contaminant transfer to the tissues of organisms compared to other exposure pathways (aqueous dermal exposure and ingestion from natural prey).
3. Research is needed to understand the relative impacts of physical and chemical effects of ingested plastic particles on a wide range of organisms.
4. Research to determine whether the relatively high surface area of nanoplastics compared to microplastics and their potential to permeate membranes with increased retention time may increase their toxicological risk to organisms (Koelmans et al., 2015).

# *Introduction*

## **1 Introduction**

Modern plastics were first introduced in the early 20<sup>th</sup> century, with the production of polyvinyl chloride (PVC) and polyethylene starting in the 1920s and 1930s. Mass-production of plastics began in the 1940s and 1950s as part of the post-World War II increase in consumerism to include many of the plastics commonly used today (e.g., polypropylene, polystyrene, polyethylene terephthalate (PET), polyurethane (PUR), polycarbonate, and polytetrafluoroethylene (PTFE – Teflon<sup>®</sup>) coatings) (Barnes et al., 2009; Cole et al., 2011; PlasticsEurope, 2008; Andrady and Neal, 2009; Lambert et al., 2014). While uses in the 1920s and 1930s were more limited (e.g., a limited number of consumer goods, automobile parts, and military equipment (Freinkel, 2011)), plastics are used in the 21<sup>st</sup> century in a wide array of products, including packaging materials, water bottles, mobile telephones, computers, building insulation, medical devices, protective clothing, piping systems for drinking water, and artificial limbs and joints (PlasticsEurope, 2008). World plastic production has increased dramatically from an estimated 1.7 million tons in 1950 to 311 million tons in 2014 (PlasticsEurope, 2013 and 2015). Coinciding with an increase in production, the amount of plastics in the aquatic environment has been steadily increasing, and plastics and plastic particles (i.e., microplastics, items < 5 mm diameter) are now commonly found in freshwater and marine systems around the globe (Derraik, 2002; Eerkes-Medrano et al., 2015; Law & Thompson 2014; Wagner et al., 2014).

Recent estimates suggest that 4.8 to 12.7 million metric tons of plastic waste entered the global marine environment in 2010 (Jambeck et al., 2015). Areas of accumulation of plastic debris include enclosed basins, ocean gyres, and bottom sediments, including the deep ocean (Moore et al., 2001; Ryan et al., 2009; Collignon et al., 2012; Shimanaga & Yanagi, 2016). Plastics in the aquatic environment primarily originate from land-based sources such as littering, improper or ineffective solid waste management, and wind-blown debris, though plastic debris from fishing activities may be a key source in some areas (Galgani et al., 2000; Andrady, 2011; Lambert et al., 2014; Unger & Harrison, 2016). Plastic particles are generally the most abundant type of debris encountered in the marine environment, with estimates suggesting that 60% to 80% of marine debris is plastic, and that more than 90% of all floating debris particles is plastic (Derraik, 2002; Gordon, 2006).

Efforts to address the issues of debris (including plastics) in the marine environment in the U.S. began in the mid-1980s (NOAA, 2008). Since that time, numerous federal initiatives and committees have been formed to address the problem of debris in marine and freshwater aquatic environments. For example, the EPA's Office of Water has been committed to reducing and preventing trash, litter, and debris from entering U.S. waterways and the ocean through the current Trash Free Waters Program which builds upon the foundation of earlier Agency work through the Marine Debris Program. The Trash Free Waters program facilitates research, outreach, and education for state and local governments, the private sector, and the general public on ways to reduce and prevent plastic and other anthropogenic debris from entering the aquatic environment (U.S. EPA, 2016). The Interagency Marine Debris Coordinating Committee (IMDCC), formed in 2004, is a multi-agency committee with the National Oceanic and Atmospheric Administration (NOAA) as chair and EPA as vice-chair. The committee makes recommendations for research priorities, monitoring techniques, educational programs, and regulatory actions for reducing the sources and ecological impacts of debris (NOAA, 2008). In addition, NOAA's Marine Debris Program is authorized by Congress to work on debris through the Marine Debris Act, signed into law in 2006 and amended in 2012, and focuses on research, removal and prevention efforts including education and outreach such as an online podcast about the "garbage patch" in the Pacific Ocean (NOAA, 2014) and a series of online videos called "Trash Talk" (NOAA, 2015a).

Recently the accumulation and potential impacts of plastic pollution has been recognized as an emerging environmental issue (GESAMP 2015; UNEP 2016). There is growing concern regarding the potential toxicological impacts caused by chemicals, in addition to the known physical impacts (i.e., entanglement, smothering, or physical effects of ingestion) of plastic debris on aquatic life and ecosystems (e.g., Moore et al., 2001; Arthur et al., 2009; Cole et al., 2011; Lavers et al., 2014). The purpose of this report is to synthesize the state of the science on the potential chemical toxicity of ingested plastic and associated chemicals on aquatic organisms and aquatic-dependent wildlife. The focus of this document is primarily on marine systems, with data provided on the Great Lakes and other freshwater systems where available. Recognizing that a variety of organic chemicals and metals found in the water column, sediments, and atmosphere may sorb to plastic debris, this paper focuses on the most prevalent and potentially hazardous contaminants currently known to be associated with plastics. This paper is organized as follows:

- Section 2 provides background information on the chemicals associated with plastics and their properties.
- Section 3 describes key sources of plastics and their modes of transport to and within the aquatic environment. This section also presents information on the general occurrence and abundance of plastic debris in the aquatic environment.
- Section 4 describes the bioaccumulation and toxicological effects that chemicals associated with plastics (i.e., their additives, as well as sorbed contaminants) may have on aquatic organisms and aquatic-dependent wildlife such as seabirds.
- Section 5 provides conclusions and recommendations for future research.

## **2 Background on the Chemicals Associated with Plastics**

Aside from the physical impacts plastics have on aquatic organisms (reviewed in Kühn et al., 2015), plastics may play a role in transporting chemicals that are associated with plastics within the aquatic environment into the food chain. Key contaminants detected in plastics in the aquatic environment include: phthalates, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), organochlorine pesticides (OCPs), polybrominated diphenyl ethers (PBDEs), alkylphenols, bisphenol A (BPA), and metals (e.g., cadmium, zinc, aluminum). Because plastic debris can persist in the aquatic environment for a long time, plastic may be a source of chemicals to the aquatic environment if the ingredients used in making the plastic (e.g., monomers and additives) leach into the surrounding waters and may be a sink for chemicals that may accumulate on plastics from the surrounding aquatic environment (e.g., persistent, bioaccumulative, and toxic (PBT) contaminants) (Engler 2012). These two categories of chemicals associated with plastics are described in more detail below.

### **2.1 Common Plasticizers and Polymer Additives**

While plastics are named for their primary monomeric ingredients, plastics typically contain additives that modify the properties of the pure polymers to increase pliability, resist ultraviolet radiation, reduce flammability or degradation, or impart other preferred physical characteristics to the finished product (Andrady and Neal, 2009; Lambert et al., 2014). Additives can leach from the plastic to the surrounding environment and as the plastic fragments and weathers, more chemicals are able to leach (Engler, 2012). Due to the large numbers of plasticizers and flame retardants associated with plastics, only the common and well-studied groups of these compounds are evaluated in this paper and discussed individually below.

Plasticizers are widely used to impart pliability to prevent shattering, with phthalates and BPA among the most common plasticizers. A variety of phthalates are used in plastics (See review by Staples et al. (1997) regarding the properties of 18 phthalates, most or all of which may be found in plastics). Some of the phthalates that were widely used in plastics in the past include dibutylphthalate (DBP), diethylhexylphthalate (DEHP), dimethylphthalate (DMP), and benzylbutylphthalate (BBP) (Oehlmann et al., 2009). Phthalates often comprise a substantial portion of polymeric materials; for example, phthalates are reported to be present at up to 50% of the mass of finished PVC plastics (Oehlmann et al., 2009). BPA is used as a monomer in the production of polycarbonate plastics and is also used as an antioxidant, inhibitor, and stabilizer in PVC and other plastics (Andrady and Neal, 2009; Oehlmann et al., 2009; Thompson et al., 2009; JRC-IHCP, 2010; Lambert et al., 2014). As a monomer in polycarbonates, BPA is a primary component in this type of plastic. Both phthalates and BPA have been reported to accumulate in organisms and to affect development and reproduction in a wide range of species, often at concentrations found in the environment (Oehlmann et al., 2009). As a result, some phthalates have been banned from use in the U.S. in children's toys and child care products as described in the Chronic Hazard Advisory Panel on Phthalates and Phthalate Alternatives (CHAP) report (CHAP, 2014), but these banned phthalates remain in use for other plastic applications.

Flame retardants, such as PBDEs, are commonly used in plastics (Ueno et al., 2004). PBDEs have been reported to comprise 5-30% of the polymers, resins, and similar substrates in which they are used (Darnerud et al., 2001). PBDEs are typically grouped by the number of bromine atoms they contain. There are 209 possible congeners of PBDEs (i.e., various substitution patterns of varying numbers of bromine atoms on the diphenyl ether structure); however, commercial products used as flame retardants primarily contain penta-, hepta-, octa-, and decabromodiphenyl ethers (Darnerud et al., 2001). Toxicological studies with both animals and humans have demonstrated that PBDEs are potential carcinogens, neurotoxins, and endocrine disruptors (Akortia et al., 2016). Primarily due to their extreme persistence in the environment and their potential human toxicity, EPA has phased out production of pentabromodiphenyl ethers and octabromodiphenyl ethers (U.S. EPA, 2006) and decabromodiphenyl ethers (U.S. EPA, 2010). Alkylphenols, such as nonylphenol (NP), are also commonly added to plastics. NP is a degradation product of nonylphenol ethoxylates (NPEs) which are produced in large volumes and used in a wide variety of industrial applications and consumer products (Engler, 2012; Soares et al., 2008). NP is associated with plastic debris as a result of both the manufacturing process and from re-adsorption of the chemical in the environment (Mato et al., 2001). NP is persistent in the aquatic environment, moderately bioaccumulative, and has been reported to be an endocrine disrupter that is toxic to aquatic organisms (Soares et al., 2008).

## **2.2 Properties of Plasticizers and Other Polymer Additives**

The octanol-water partitioning coefficient ( $\log K_{ow}$ ) represents the ratio of how much of a compound partitions to the organic solvent octanol relative to water in an octanol-water system, and is a useful parameter for assessing the potential biological partitioning of certain chemicals to lipid-rich tissues. Lipophilic/hydrophobic compounds have a greater affinity for octanol relative to water. Larger  $\log K_{ow}$  values thus indicate a greater affinity for non-polar materials (e.g., plastics and lipid membranes) and less of an affinity for water.

Water solubility and  $\log K_{ow}$  data for several plastic additives are summarized in Table 1, and those for several known PBT compounds are summarized in Table 2 in the next section. It should be noted that a review of physical property data in the literature reveals wide ranges of values for a single compound or mixture of compounds. The solubility and  $\log K_{ow}$  values included in Table 1 and 2 were selected as

generally representative of these compounds. While not intended as definitive values for water solubility and log K<sub>ow</sub>, the values in Table 1 and 2 are intended to highlight the general relationship between water solubility and log K<sub>ow</sub>. Also, reference temperatures are not available for most of the water solubilities listed in Table 1 and 2, which adds some uncertainty to how the data represent the relationship of water solubility and log K<sub>ow</sub> because this property is temperature-dependent.

Comparing the log K<sub>ow</sub> values in Table 1 to those for known PBT compounds in Table 2 (in particular, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and dichlorodiphenyltrichloroethane (DDT)) indicates that many polymer additives have log K<sub>ow</sub> values that are similar to those of known PBT compounds.

**Table 1. Selected physical properties of several plastic additives.**

Chemical	Water Solubility (µg/L) (Reference temperature noted where specified)	logK <sub>ow</sub> (dimensionless)	Reference
Dimethylphthalate	4,200,000	1.61	1
di-n-Butylphthalate	11,200	4.45	1
di-n-Octylphthalate	0.5	8.06	1
bis-(2-Ethylhexyl)phthalate	3	7.50	1
Bisphenol A	120,000 – 300,000*	3.40	2
Tetrabromodiphenyl ethers	Not reported	5.9-6.2	3
Pentabromodiphenyl ethers	0.0009 @20°C	6.5-7.0	3
Octabromodiphenyl ethers	Not reported	8.4-8.9	3
Decabromodiphenyl ether	Data considered unreliable	10	3
Perfluorooctanoic acid (PFOA)	9.50 x 10 <sup>6</sup> @ 25 °C	Not measurable	4
Perfluorooctane sulfonic acid (PFOS)	680,000	Not measurable	5

References:

1. Staples et al. (1997). Staples et al.-recommended values from ranges presented are used in Table 1.
2. Staples et al. (1998)
3. Darnerud et al. (2001)
4. ATSDR (2015)
5. ATSDR (2015), OECD (2002)

Notes:

\* The water solubility of BPA is reported to increase with increasing pH due to two ionizable hydroxyl groups (Staples et al., 1998)



## **2.3 Sorption of Chemicals to Plastics in the Aquatic Environment**

Plastics can accumulate chemical contaminants from the surrounding aquatic environment due to their physical properties, such as size, shape and surface area, and the chemical properties of the specific polymer (GESAMP, 2015). Adsorbed chemicals have been found on plastic debris samples globally and include metals as well as compounds categorized as PBT by the EPA, including PCBs, PAHs, and DDT (Mato et al., 2001; Teuten et al., 2007, 2009; Rochman et al., 2013c; Holmes et al., 2012; Engler, 2012). These chemicals resist degradation and persist in the aquatic environment. The longer a plastic particle remains in the aquatic environment, the more concentrated the contaminants can become as they accumulate on the particle surface over time (Carpenter et al., 1972; Mato et al., 2001; Lavers et al., 2014). However, the process of chemical contaminant sorption and desorption is dynamic and depends on various factors, the most important being the presence of a concentration gradient (Koelmans et al., 2013), but also other factors including temperature, pH, and salinity (Engler, 2012). In addition, fragmentation, degradation, and weathering, which occur over time in the aquatic environment, increase the surface area-to-volume ratio of the plastic (Bernstein and Woods, 2009). These processes can result in an increase in the relative concentration of sorbed contaminants and also allow for desorption of accumulated contaminants into the water (Bernstein and Woods, 2009; Engler, 2012). For example, tests conducted on PVC, polyethylene, and polypropylene indicate that exposure of the plastics to ultraviolet light, simulating weathering conditions, increases the sorptive capacity of these plastics relative to virgin plastics (Teuten, et al., 2009).

### **2.3.1 Hydrophobic Compounds in the Environment that May Sorb to Plastics**

Hydrophobic compounds (e.g., PCBs, PAHs, OCPs) tend to have low water solubility (Table 2) and partition from the water column to other matrices with similar hydrophobic properties (Engler, 2012). For this reason, when in the aquatic environment, they tend to partition to organic matrices or to plastic debris (Engler, 2012). The ability of hydrophobic compounds to partition to plastics is well documented; because of their sorptive capacity some plastics have been developed as passive sampling devices to assess the concentrations of hydrophobic compounds in different environmental media (Lohmann, 2012). In addition, a recent review by Ziccardi et al. (2016) compiled polymer-water partitioning coefficients ( $K_{pw}$ ) published for plastics and chemical combinations (see Table 1 in Ziccardi et al., 2016) and noted that previous studies have reported that  $K_{pw}$  values are correlated with  $K_{ow}$  values.

Table 2 summarizes water solubility and log  $K_{ow}$  values for several persistent and hydrophobic organic environmental contaminants known to sorb to plastics (Rochman et al, 2013a; Rochman et al, 2013b; Rochman et al, 2014a; Mato et al., 2001; Engler, 2012). Other properties affecting sorption and persistence may be of relevance (e.g., photolysis, hydrolysis and biodegradation); however, these other properties and their relationships to persistence of a chemical in the environment require further research.

**Table 2. Selected physical properties of several persistent, hydrophobic organic contaminants known to sorb to plastics.**

Chemical	Water Solubility (µg/L) @ 25°C unless noted	logK <sub>ow</sub> (dimensionless)	Reference
<b>Aroclor 1016 (PCBs)*</b>	225-250	5.6	1
<b>Aroclor 1242 (PCBs)</b>	240	5.6	1
<b>Aroclor 1248 (PCBs)</b>	54	6.2	1
<b>Aroclor 1254 (PCBs)</b>	12	Not reported	1
<b>Aroclor 1260 (PCBs)</b>	27	6.8	1
<b>Benzo(a)pyrene</b>	3.8	6.11	1
<b>Phenanthrene</b>	600 ±100 @22°C	4.55	1
<b>p,p'-Dichlorodiphenyltrichloroethane (DDT)</b>	3.1-3.4	6.83	1
<b>Nonylphenol (branched and straight-chain)</b>	4,600 @ pH 5.0 6,237 @ pH 7.0 11,897 @ pH 9.0	3.80-4.77	2

References:

1. U.S. EPA (2000)
2. U.S. EPA (2005a)

Notes:

\* Aroclors contain a four-digit numeric code that denotes the 12 carbons contained in the biphenyl molecule followed by two additional digits that reflect the average weight percentage of chlorine in the congeners in the aroclor group (the exception being aroclor 1016, whose congeners contain, on average, 41% chlorine) (ATSDR, 2000).

The quantity and rates at which persistent compounds sorb to plastics differ by many factors including plastic type and length of time in the aquatic environment. A long-term (12-month) study on contaminant sorption to plastics suggests that PCBs and PAHs sorbed more readily to high- and low-density polyethylene and polypropylene compared to PET and PVC (Rochman et al., 2013b). In addition, a six-day field experiment on polyethylene plastic resin pellets in marine water off the coast of Japan revealed a significant increase in PCB and dichlorodiphenyldichloroethylene (DDE) accumulation in the pellets over this short period of time via sorption from seawater (Mato et al., 2001).

The types and concentrations of hydrophobic chemicals detected in and on plastics collected from the aquatic environment can differ across geographic locations. For example, several PCB congeners, PAHs, DDT and its metabolites, PBDEs, and alkylphenols were detected at concentrations ranging from 1 to 10,000 ng/g in samples collected in the open ocean, as well as at remote and urban beaches (Hirai et al., 2011). PCBs were detected in all plastics sampled, with concentrations ranging from 1 to 436 ng/g, with generally higher concentrations found for polyethylene compared to polypropylene debris (Hirai et al., 2011). Levels of PCBs and PAHs were higher for plastic debris found on urban beaches compared to those found in the open ocean and remote beaches, as may be expected due to the impacts of local land-based sources of those contaminants (Hirai et al., 2011). In a study conducted along the Portuguese coastline, levels of PCBs and PAHs were orders of magnitude higher in plastic pellets collected near urban coastal areas compared to rural sites, suggesting that analysis of these contaminants in pellets should be conducted across regional scales to understand spatial trends in accumulation and source (Mizukawa et al., 2013). Measured concentrations of PCBs ranged from 10.5 to 307 ng/g-pellet, and PAH levels were

generally within the 100 to 300 ng/g-pellet range (Mizukawa et al., 2013). In plastic samples collected in the North Pacific Ocean gyre, PCBs, pesticides, and PAHs were prevalent and were measured in more than 50%, 40%, and nearly 80% of the samples, respectively (Rios et al., 2010). PCBs were produced in large quantities after the Second World War, and while PCB production and new uses have largely been phased out in developed countries, legacy uses and contamination remains an issue (Tanabe, 2002) and these hydrophobic compounds are still used in the U.S. and some developing countries. Their persistence and wide distribution in the environment add to the likelihood that marine ecosystems are the primary sink for PCBs (Tanabe, 2002). Another class of compounds, alkylphenols, which may be used as additives during the plastics manufacturing process are also often detected in wastewater discharges and can also sorb to plastics in the water column (Hirai et al., 2011).

To monitor the global pattern of hydrophobic organic contaminants found on beaches around the world, International Pellet Watch (IPW) was launched in 2005. Volunteers collect pellets along beaches and send them to Tokyo University of Agriculture and Technology for analysis. Results from analyses are shown for PCBs, DDT, and PAHs in Figure 1-3 respectively (reprinted with permission); methods are presented in Mato et al. (2001). In the IPW's results, concentrations of PCBs on beached plastic resin pellets were reported to be highest along the coasts of France (2,970 ng/g-pellet), followed by beaches in Los Angeles, California (602 ng/g-pellet), the shoreline of Lake Erie (502 ng/g-pellet), the Japanese coastline (453-499 ng/g-pellet), and Boston, Massachusetts (405 ng/g-pellet) (Figure 1). Total PCB values represent the sum of specific tetra, penta-, hexa-, hepta-, and nonachlorobiphenyl congeners (congeners 66, 101, 110, 149, 118, 105, 153, 138, 128, 187, 180, 170 and 206). This list includes two co-planar (i.e., dioxin-like) PCBs, congeners 105 and 118, and 11 non-co-planar PCBs. Levels of DDT and its metabolites were reported to be highest in beached pellets in Albania (1,061 ng/g-pellet), Brazil (777 ng/g-pellet), Vietnam (558 ng/g-pellet), and Japan (299 ng/g-pellet) (Figure 2). The higher levels of DDT found along Vietnam coastlines may be attributed to the continued use of DDT as a pesticide (Teuten et al., 2009). Concentrations of total PAHs were highest in beached pellets in Europe (24,364 ng/g-pellet and 15,608 ng/g-pellet reported on beaches in Portugal and the United Kingdom, respectively), and in New Zealand (13,278 ng/g-pellet) (Figure 3). Importantly, these results are based on analyses of pellets submitted by volunteers around the world. Therefore, the reported numbers of pellets across beaches is largely a function of how many pellets are submitted for analysis, and the ability to accurately reflect geographic contaminant trends based on the Tokyo University analyses is also closely tied to the extent of volunteer participation in a geographical region.

Hydrophobic compounds can readily sorb to plastic debris in the aquatic environment, but it is also important to compare concentrations on plastics to other environmental matrices. Studies have found hydrophobic compounds can accumulate on plastic debris at concentrations up to six orders of magnitude greater than the surrounding water (Ogata et al., 2009) and up to two orders of magnitude greater than sediment and suspended particles (Mato et al., 2001; Teuten et al., 2007). However, assuming equilibrium, a thermodynamically-based model predicted that with the current concentrations of plastic debris in the ocean, the fraction of hydrophobic compounds sorbed to the plastic is of limited importance relative to other exposure media, such as direct chemical exposure via water, sediment, or ingestion of contaminated prey (Gouin et al., 2011; Koelmans et al., 2016).

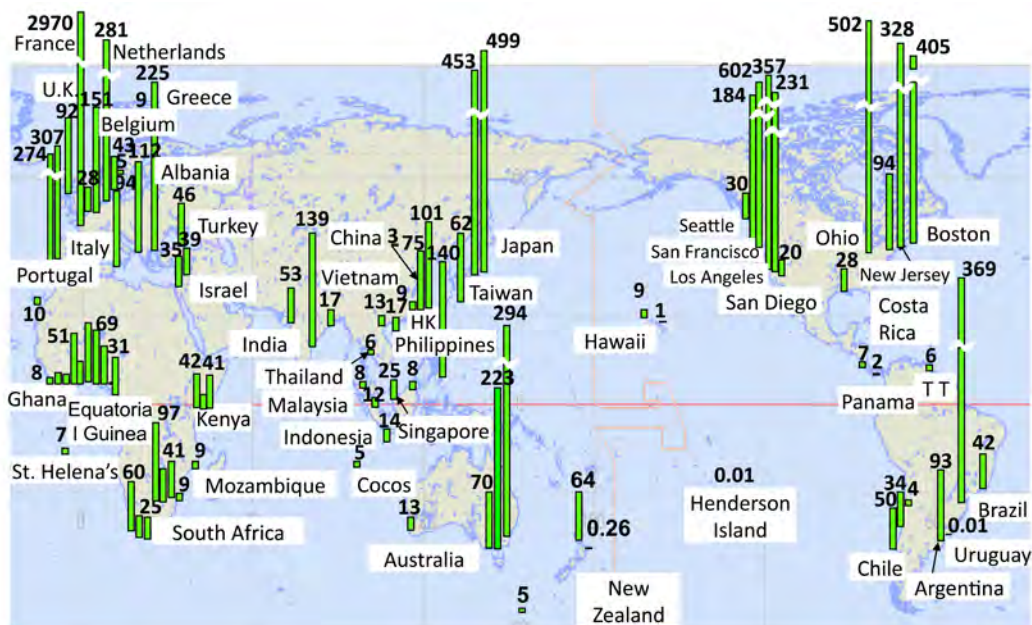


Figure 1. Concentrations of polychlorinated biphenyls (PCBs) in plastic resin pellets on beaches (ng/g – pellet). Reprinted with permission from Dr. Takada (IPW, <http://www.tuat.ac.jp/~gaia/ipw/en/map.html>).

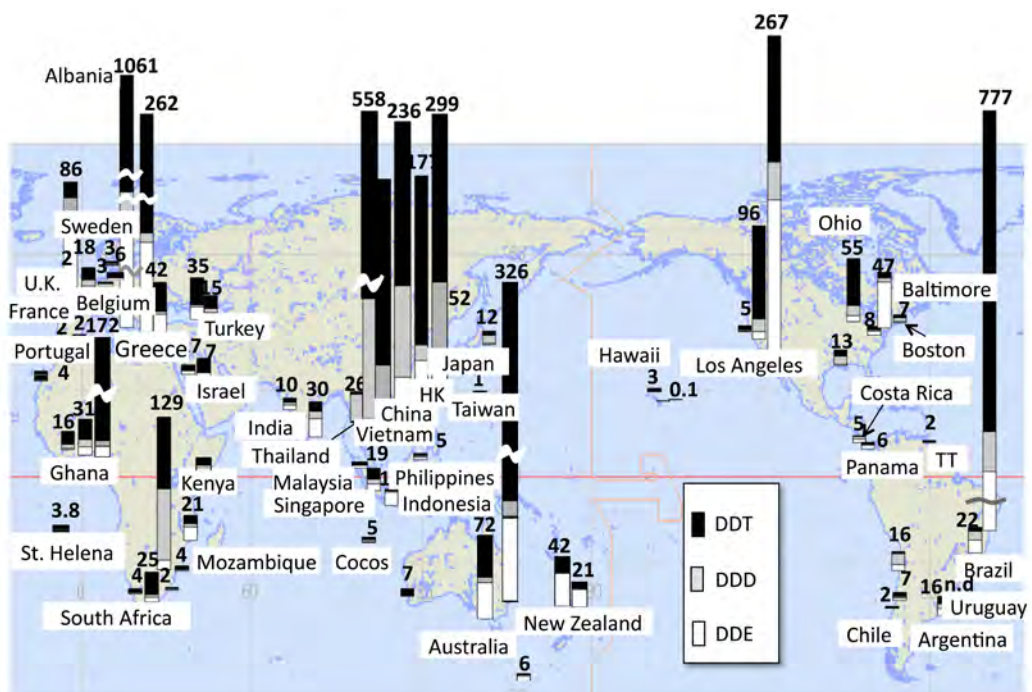


Figure 2. Concentrations of dichlorodiphenyltrichloroethane (DDT) and its metabolites (DDE: Dichlorodiphenyldichloroethylene, and DDD: Dichlorodipenyldichloroethane) in plastic resin pellets on beaches (ng/g – pellet). Reprinted with permission from Dr. Takada (IPW <http://www.tuat.ac.jp/~gaia/ipw/en/map.html>).





**Figure 3. Concentration of total polycyclic aromatic hydrocarbons (PAHs) in plastic resin pellets on beaches (ng/g – pellet). Values with an asterisk (\*) are less than the limit of quantitation. Values with a double asterisk (\*\*) are a median of only one pool of 200 pellets as opposed to the median of five pools. Reprinted with permission from Dr. Takada (IPW <http://www.tuat.ac.jp/~gaia/ipw/en/map.html>).**

### 2.3.2 Metals

Metals can be added to plastics during the manufacturing process (Nakashima et al., 2012). However, metals are also found in the water column and sediments from a variety of other sources, and they can also sorb to plastic debris. While much research has been conducted on the sorption of persistent, hydrophobic compounds to plastics, the metal accumulation on plastics and potential impacts is a newer area of research (e.g., Ashton et al., 2010; Holmes et al., 2012; Nakashima et al., 2012; Rochman et al., 2014a). Metal ions, such as aluminum, antimony, cadmium, chromium, cobalt, copper, iron, lead, manganese, molybdenum, tin, uranium, and zinc, have also been shown to sorb to plastics in the aquatic environment (Ashton et al., 2010; Holmes et al., 2012; Rochman et al., 2014a; Turner, 2016; Turner & Lau, 2016). Similar to hydrophobic compounds, metals may sorb more readily to plastic particles as the particles become more degraded and porous in the aquatic environment (Ashton et al., 2010; Rochman et al., 2014a; Rochman et al., 2014b). For example, following a year-long study in San Diego Bay on the sorption of several metals on five different types of plastic, the authors reported that sorption of these metals increased with time and that similar concentrations of each metal were observed in the differing plastic types (Rochman et al., 2014a). In addition, concentrations of metals on plastics and in nearby sediment have been found to be similar (Holmes et al., 2012).

### **3 Sources and Transport of Plastics to the Aquatic Environment**

#### **3.1 Types, Quantities, and Abundance of Plastics in the Aquatic Environment**

Plastic debris is ubiquitous in the aquatic environment (Thompson et al., 2009). While plastics in the marine environment have received the most attention to date, investigations also indicate that plastics readily accumulate in freshwater environments (Eerkes-Mdrano et al., 2015; Baldwin et al. 2016). Plastics found in the aquatic environment are generally categorized as macroplastics (i.e., items > 5 mm diameter, such as disposable cups, bottles, and shipping pallets) and microplastics (i.e., items < 5 mm diameter, such as microbeads and fishing line fragments) (Lambert et al., 2014). Microplastics are further categorized into primary and secondary sources where primary sources include manufactured products and secondary sources result from the breakdown of macroplastics in the environment (GESAMP, 2015). As a subcategory of microplastics, nanoplastics (<100 nm size range) are likely to occur from both primary and secondary sources; methods do not currently exist to detect nanoplastics in the environment (Koelmans et al., 2015). The abundance of microplastics (compared to macroplastics) in the marine environment is increasing, according to a state-of-the-science report conducted by Canada's Secretariat of the Convention on Biological Diversity (SCBD) (SCBD, 2012). Recent investigations have found the majority (90%) of plastic debris found in the pelagic environment is generally less than 5 mm in diameter (Eriksen et al., 2013; Browne et al., 2010; Thompson et al., 2004; Rochman et al., 2014b). Examples of some of the common types of plastics typically found in the aquatic environment are shown in Figure 4.

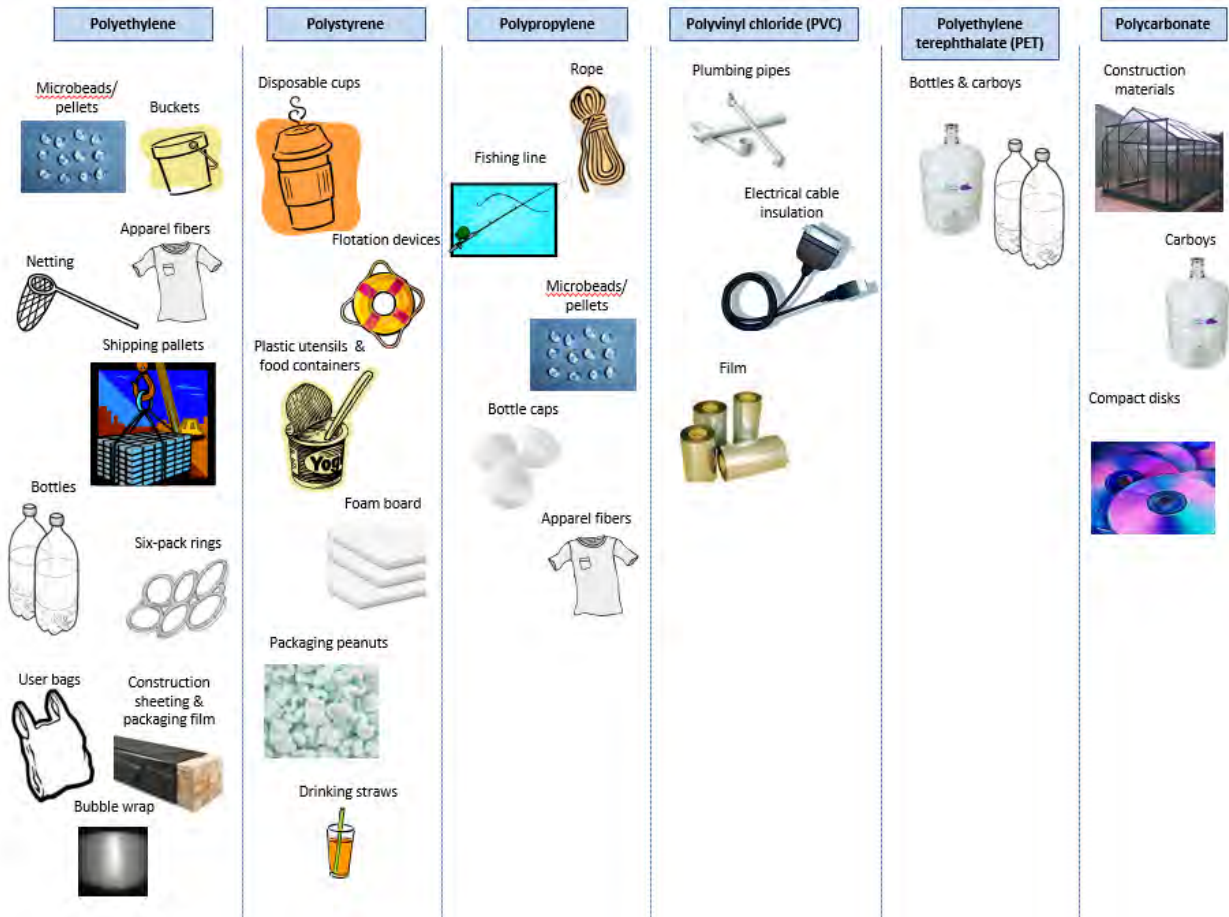


Figure 4. Types of plastics commonly found in the aquatic environment (Pruter, 1987; U.S. EPA, 1993; Andrady, 2011).

Plastic production continues to increase across the globe, and the volume of plastics entering the marine environment has increased as much as three to four orders of magnitude since its mass production began in the 1940s and 1950s (Thompson et al., 2004; Barnes et al., 2009; Ryan et al., 2009; Cole et al., 2011). The presence of plastic debris in the ocean has been reported since the early 1970s, although recent analyses of archived plankton samples collected around the United Kingdom from as early as the 1960s revealed various types of plastic including polyethylene, polyester, and polypropylene, among others (Thompson et al., 2004). Investigations conducted in US waters in the 1970s reported plastic debris in 62% of surface plankton samples collected in the Atlantic Ocean from Cape Cod to the Caribbean (Colton et al., 1974), as well as an abundance of polystyrene spherules (around one spherule per 0.03 cubic meters on average) along the coastal waters of New England in 1971 (Carpenter et al., 1972). In the past five decades, numerous studies have reported plastic debris in marine environments (see Table 3 in Lambert et al., 2014) and the Great Lakes (e.g., Eriksen et al., 2013). In more recent surveys, Barnes et al., 2009 noted that average size of plastic particles in the environment seems to be decreasing, correlated to breakdown of macroplastic debris and the increasing use of plastic scrubbers in cosmetics, personal hygiene products, and cleaning products, and with the abundance and global distribution of microplastic fragments increasing over the last few decades. Overall, approximately 60% to 80% of marine debris is estimated to be comprised of plastics (Derraik, 2002; Browne et al., 2010; SCBD, 2012), and because plastics are so frequently encountered in the ocean, the Continuous Plankton Recorder (CPR) Survey (the longest running plankton monitoring program in the North Sea and North Atlantic) added 'plastics' as its first non-biological entity to their program in 2004 (Richardson et al., 2006).

The research on plastics in the marine environment is more advanced than the freshwater environment (Wagner et al., 2014) despite the fact that a majority of marine plastics are thought to originate from land-based sources (see Section 3.2). Current studies suggest that plastics are also pervasive in freshwater environments (Eerkes-Medrano et al., 2015). For example, a recent survey of floating plastic debris in 29 Great Lakes tributaries found plastics in all 107 samples with 98% of the plastics considered microplastics. Most types of plastics (except fibers, the most frequently detected particle type) were positively correlated with urban-related watershed influences (Baldwin et al., 2016). In addition, some studies have found concentrations of microplastics in lakes and rivers similar to or higher than in oceanic gyres (e.g., Yonkos et al., 2014; Lechner et al., 2014; Mani et al., 2015). Freshwater environments may be similar to marine environments in terms of sources and transport mechanisms, prevalence, and potential for impacts to organisms, but the differences in waterbody size and the proximity to point sources may result in different plastic composition (Eerkes-Medrano et al., 2015).

Estimating the abundance of plastic debris in the aquatic environment is complicated by a variety of factors, including the vastness of the ocean compared to the often micro-sized plastic debris that may be floating along the surface, mixed within the water column, or incorporated in the sediment or beach sands (Ryan et al., 2009; Cole et al., 2011). Rates of degradation among plastic debris vary, with thicker types of plastic expected to persist in the aquatic environment for potentially hundreds of years (Barnes et al., 2009; Ryan et al., 2009). Ocean currents, weather patterns, and other transport mechanisms (see Section 3.3) alter the spatial distribution of plastic debris; therefore, a study conducted a decade ago may not reflect the abundance or types of plastics that would currently be observed in the same location. Also, the type of investigation and the methods employed (e.g., trawls vs. observational studies) can influence the study results. For example, studies investigating plastics along the ocean surface likely provide underestimates of the true abundance of plastic debris in the ocean, because the specific gravity of most plastics is higher than that of seawater, causing them to sink (Ryan et al., 2009; Andrady, 2011; Woodall et al., 2014). The buoyancy of plastic is dependent on the density of the material and presence of trapped air; after some time, floating plastic may become fouled with organisms that increase the density and cause it to sink (Andrady, 2011).



In addition, current collection methods often use nets with a mesh size of ~330  $\mu\text{m}$ , which is larger than many microplastic particles (Andrady, 2015). Also, open ocean sampling necessitates large sample sizes to allow for statistical power to detect temporal and spatial changes in abundance (Ryan et al., 2009). Reports from beach and shoreline clean-ups may also provide underestimates of the abundance of plastics because many small microplastic fragments are not easily discernable by the naked eye and can be the size of a grain of sand or planktonic organism (Ryan et al., 2009; Andrady, 2011; Cole et al., 2011).

## **3.2 Key Sources of Plastics**

### **3.2.1 Land-Based Sources**

Plastic debris originates primarily from land-based activities that include intentional and accidental disposal (Lambert et al., 2014; Figure 5). Plastics can enter the environment through many pathways including littering of bags, bottles, and other plastic items, particularly at large public gatherings and events, or from coastal tourism (Gregory, 2009). Plastic debris from uncovered trash receptacles or inadequately lined or covered landfills can serve as sources (Rayne, 2008; Teuten et al., 2009). Sewage treatment facilities generally do not have the equipment needed to adequately screen for wastewater discharges of microplastic debris such as polyethylene beads in exfoliating scrubs, abrasive plastic fragments in cleaning agents, or acrylic shed from clothing during washing (Thompson et al., 2004; Zubris and Richards, 2005; Fendall and Sewell, 2009). Therefore, wastewater effluent discharges serve as sources of plastics to surface waters. A recent study by Browne et al. (2011) found that a single piece of clothing can produce more than 1,900 plastic fibers during one wash, which may not be completely removed during wastewater treatment and screening. Another example of a land-based source is air blasting technologies, where microplastic scrubbers such as acrylic, polyester, or melamine are blasted at boat hulls, engines, and machinery to remove rust or paint, and can contribute plastics to the environment through wastewater effluent discharges (Derraik, 2002; Cole et al., 2011). Air blasting of boat hulls is performed at marinas that can be in close proximity to water and the microplastic scrubbers may enter the water through stormwater runoff as well. In addition, low-density polyethylene film fragments on farmlands from hay sleeves and silage bags, bunker silo covers, silage wrap, and films used on greenhouses or as weed barriers can be windblown or degrade into the soil (Xu et al., 2006). Lastly, plastic pellets, which are the 1-5 mm diameter raw material melted to create plastic products, can be discharged or accidentally spilled into the environment throughout their lifecycle, including during their production, transport, and waste disposal (Pruter, 1987; U.S. EPA, 1993; Thompson et al., 2004; Ryan et al., 2009).

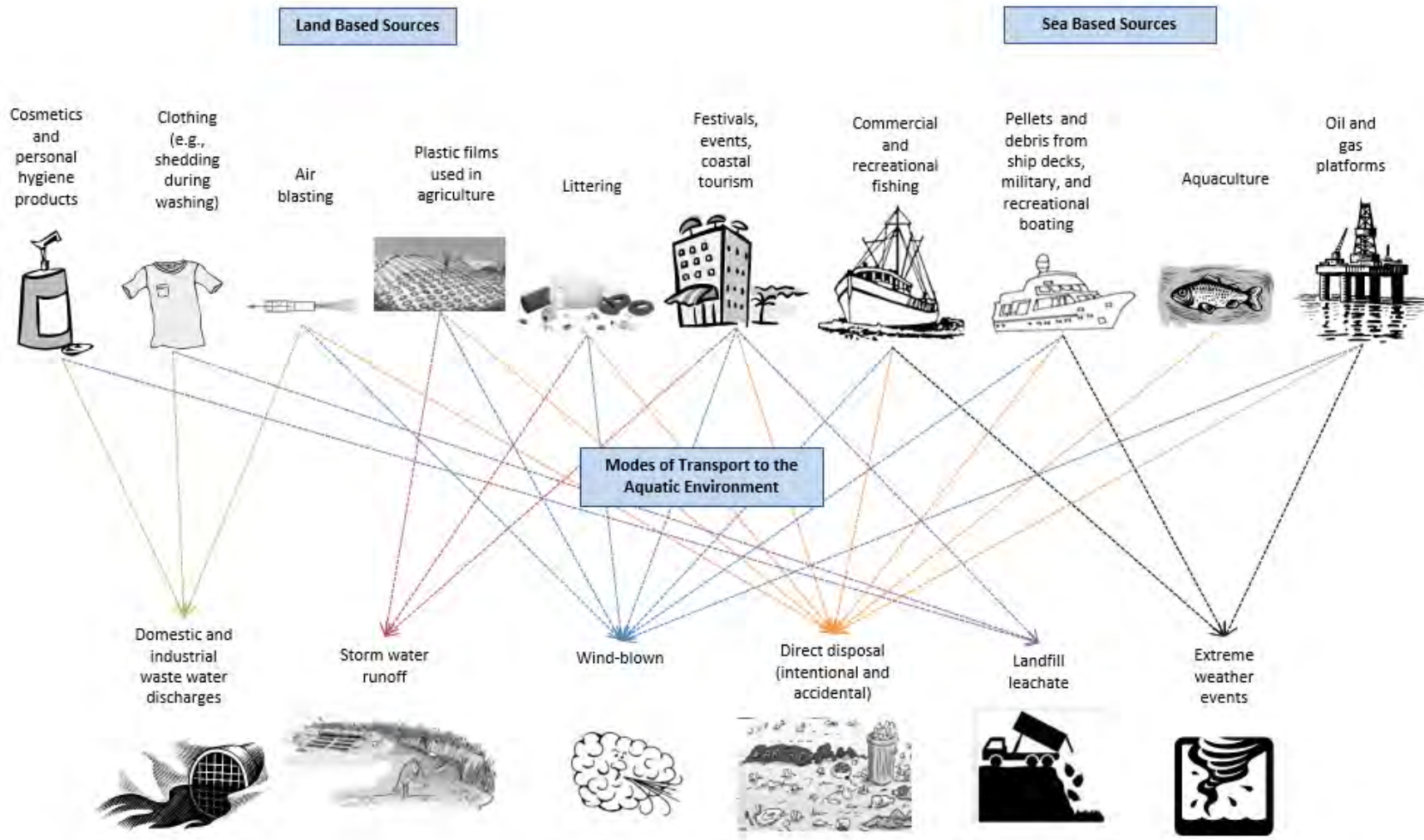


Figure 5. Key sources of plastics and their modes of transport to the aquatic environment. (U.S. EPA, 1993; Rayne, 2008; Gregory, 2009; Cole et al., 2011; Lambert et al., 2014).

### **3.2.2 Sea-Based Sources**

Sea-based sources of plastics include commercial and recreational fishing vessels, barges, recreational boaters, military activities, oil and gas platforms, and aquaculture farms (Pruter, 1987; Cole et al., 2011; Lambert et al., 2014) (Figure 5). Fishing gear, such as nylon netting, buoys, and plastic monofilament line are used in fishing operations. Contributions of plastics to the sea from marine fishing vessels have increased from an estimated 23,000 tons during the 1970s (Pruter, 1987) to an estimated 6.5 million tons during the 1990s (Derraik, 2002). Pre-production resin polyethylene and polypropylene pellets have been used on ship decks to facilitate the movement of large objects, and can be washed from the deck accidentally or due to improper storage or handling (Tharpes, 1989; U.S. EPA, 1993). Plastic debris from ship decks can be windblown or lost/discarded accidentally or during improper loading and unloading of equipment or because of inadequate storage (Lambert et al., 2014). General litter from recreational boaters is also a source of plastics, such as bags and bottles. Industrial plastic netting and tubing used in ocean-based aquaculture operations can directly enter the marine environment via improper disposal (Cole et al., 2011). While a number of studies have investigated incidences of plastic ingestion by terrestrial and aquatic life (e.g., Lattin et al., 2004; van Franeker et al., 2011) (see also Section 4), minimal research has been conducted on plastic ingestion as a mode of transport of plastics to and within the aquatic environment, particularly by migrating birds and ocean-dwelling animals. However, plastic debris can be transported when seabirds regurgitate indigestible materials (i.e., a bolus) (U.S. EPA 2014a).

### **3.2.3 Relative Importance of Plastic Sources**

Primary sources of plastics are land-based, contributing an estimated 80% of the total plastic debris to the environment (Andrady, 2011) with littering and improper waste disposal as the major route for plastics to enter the environment (Galgani et al., 2000; Lambert et al., 2014). Recently, Jambeck et al. (2015) estimated that 1.7 to 4.6% (4.8 to 12.7 million metric tons) of the total plastic waste generated in coastal countries in 2010 entered the ocean. Fishing-related activities can also be a key source in certain areas (Galgani et al., 2000; Andrady, 2011; Lambert et al., 2014; Unger & Harrison, 2016). The proportion of plastics originating from ships tends to increase with distance from shore, with much of the plastic debris eventually ending up in ocean gyres (see Section 3.4) and bottom sediments (Moore et al., 2001; Ryan et al., 2009). An investigation of debris along the Southern California Bight found that the majority of anthropogenic debris was from ocean vessels and fishing activities (Moore and Allen, 2000), which may account for approximately 18% of all marine plastic debris found globally (Andrady, 2011). A recent oceanic survey conducted by NOAA on the abundance and characteristics of plastics in plankton samples collected in the Southeast Bering Sea and off the coast of southern California found that the majority of plastic particles were product fragments (secondary microplastics) typically less than 2.5 mm in size (Doyle et al., 2011). As noted by the authors, the prevalence of product fragments may help demonstrate the persistence of plastics in the marine environment (Doyle et al., 2011). Of the particles that were greater than 5 mm in size, the majority were fishing net and line fibers (Doyle et al., 2011). A key factor that influences the quantity of plastics in the aquatic environment is the proximity of the source, in particular urban areas, to shorelines as evidenced by the prevalence of plastics in samples collected in Lake Erie, which is the most populated of the three Great Lakes (Eriksen et al., 2013), and the Mediterranean Sea (Barnes et al., 2009; Pasquini et al., 2016) (see also discussion in Sections 3.4).

### **3.3 Transport Mechanisms**

Once released to the environment, plastics and other debris can be transported to freshwater and marine environments via streams and rivers, stormwater and wastewater discharges, littering and disposal along shorelines, wind, and weather events. The distances plastics travel in the environment vary because plastics vary in size, weight, density, and shape. Lightweight plastic products and particles (e.g., bags, films, clothing fibers, pellets and plastic bottles) are more easily transported greater distances by wind, weather events, stormwater, effluent discharges, and inputs from freshwater systems, compared to more dense and larger plastic items. A plastic litter survey conducted by the Algalita Marine Research Foundation reported up to 81 grams per cubic meter of small plastic debris in stormwater discharges during rain events in southern California (Ryan et al., 2009). Wastewater effluent discharges are an important source and transport mechanism of synthetic clothing fibers. One study reported an average of one fiber of polyester, acrylic, or polyamide per liter of effluent from two Australian wastewater treatment plants (Browne et al., 2011; Dris et al., 2016). Airborne plastic fibers released from residential and commercial clothes dryers have not been characterized for the amount and types of plastic particles released.

Both natural and human activities influence the lateral and vertical distribution of plastics within the marine environment. Ocean currents carry plastic debris to accumulation areas such as enclosed basins (Collignon et al., 2012) and ocean gyres (Barnes et al., 2009). The distribution of plastics near river outflows can mimic the patterns of sediment deposition, distributing loads of plastic debris to estuaries and open water areas (Galgani et al., 1996; Williams and Simmons, 1997; Baldwin et al., 2016; Sutton et al. 2016; Wessel et al., 2016). Fouling organisms such as algae, biofilms, and invertebrates often colonize plastic particles, causing the debris to sink to the seafloor and mix with the bottom sediment (Derraik, 2002; Barnes et al., 2009; Cole et al., 2011). In addition, organisms that colonize plastic debris may limit degradation by photolysis because they prevent the plastic surface from being exposed to ultraviolet light (Barnes et al., 2009).

The net transport of plastic to and within the ocean (and the Great Lakes) can be exacerbated by extreme weather events such as floods, tsunamis, hurricanes, and tornados (Barnes et al., 2009). For example, a 9.0 magnitude earthquake off the coast of a highly urbanized region in Japan in 2011 caused a tsunami that transported an estimated 5 million tons of debris, including plastics, into the marine environment (NOAA, 2015b). Based on ocean current data, the debris is expected to be transported through the North Pacific Current and California Current before looping back towards the Hawaiian Islands and eventually accumulating in the North Pacific Gyre (refer to Section 3.4 and Figure 6) (Bagulayan et al., 2012). There have been reports of fishing nets and floats, as well as other tsunami debris along the coast of Alaska, British Columbia, Washington, Oregon, and Hawaii (Bagulayan et al., 2012; NOAA, 2013; Kubota, 2014). While not reviewed in this paper, plastic debris can be a vehicle for invasive species that can colonize particles and be transported great distances, and such concerns have already been raised with the tsunami debris that has recently washed ashore in North America (Bagulayan et al., 2012; NOAA, 2013). For further information on the colonization and transport of non-native species and pathogens via plastic debris, refer to Barnes (2002), Gregory (2009), SCBD (2012), and Keswani et al. (2016).

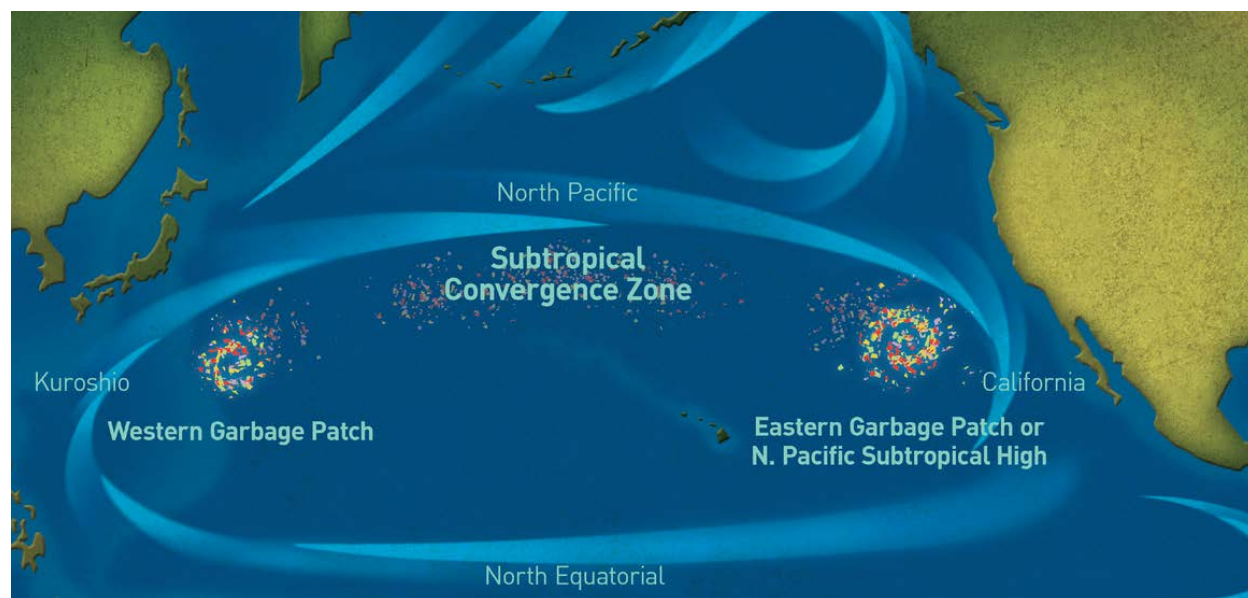
### **3.4 General Occurrence and Accumulation in the Aquatic Environment**

Plastics are ubiquitous in the aquatic environment and particles may be found along shorelines, floating at the water surface, mixed within the water column, or settled within the sediments and submerged aquatic vegetation (Colton et al., 1974; Thompson et al., 2004; Barnes et al., 2009; Ryan et al., 2009; Van Cauwenberghe et al., 2013; Eriksen et al., 2014). Plastic debris also accumulates in Arctic Sea ice where 1

to 7 pieces of plastic/ft<sup>3</sup> ice were found in ice cores (Obbard et al., 2014; Lusher et al., 2015). Plastic particles have been reported in the ocean near urban areas, offshore of remote islands, in the Great Lakes, and even in Antarctica (see reviews by Cole et al., (2011) and Lambert et al., (2014)). Sediments are an ultimate sink of plastic debris; microplastics were recently found in surface sediment samples collected in three of four remote deep sea areas in the Atlantic and Mediterranean Sea (Van Cauwenberghe et al., 2013). Higher concentrations of microplastics in sediment may exist near harbors and densely populated areas (Besseling et al., 2013), although further research is needed in this area. Reported occurrence of plastics in the aquatic environment is a function of the geographical areas studied. As evidenced by a review of the literature on the impacts of marine debris on aquatic life and seabirds, the majority of research to date has been conducted on the east and west coasts of North America, Europe, and Australia, while fewer studies have been conducted in Asia, Africa, the Arctic, and Antarctic (SCBD, 2012). However, a recent study reports an estimate of the total number of plastic particles and their weight floating in the world's oceans across all five sub-tropical gyres, coastal Australia, Bay of Bengal and the Mediterranean Sea (Eriksen et al., 2014).

The occurrence and abundance of plastics at sea is dependent on a variety of factors, including: the density and buoyancy of the plastic particle; currents and water flow; wind and weather conditions; oceanic geographical characteristics (e.g., shallow vs. deep areas; confined areas vs. open ocean); the presence of large rivers; proximity to urban areas as well as industrial and wastewater treatment discharges; proximity to trade routes and shipping channels; and the prevalence of fishing activities (Moore and Allen, 2000; Barnes et al., 2009; Eriksen et al., 2013). Plastic debris can accumulate in bay areas that have relatively low water circulation compared to the open sea (Hess et al., 1999; Stefatos et al., 1999; Collignon et al., 2012), although ocean gyres are an exception to this generalization as discussed further in this section. Proximity of urbanized areas influence the abundance of plastic found in an area: for example, approximately 90% of the plastics encountered during expeditions in the Great Lakes were found in Lake Erie, which is the most densely populated of the three lakes sampled (Eriksen et al., 2013). Large rivers typically carry plastic debris farther into the ocean, thereby leading to generally lower concentrations of plastic debris along continental shelves (Moore and Allen, 2000). Deep, confined marine areas, such as coastal canyons, have been reported to have high densities of plastics in the sediments, particularly in the Northwest Mediterranean Sea, Celtic Sea, North Sea, Baltic Sea, Adriatic Sea, and East-Corsica (Galagani et al., 2000). Sites in the Mediterranean tend to have a higher density of plastic debris compared to other seas due to densely populated coastlines, limited water circulation, and shipping trade routes (Barnes et al., 2009; Pasquini et al., 2016). Based on current knowledge, macroplastics generally tend to accumulate at higher densities in semi-enclosed, deep, and confined marine areas (particularly near urban centers), as well as at oceanic fronts in the Northern Hemisphere; lower densities of macroplastics are generally encountered in remote marine areas, along the bottom of continental shelves, and in the Southern Ocean (Barnes et al., 2009). In addition, gyres, high-pressure systems with rotating ocean currents, facilitate the accumulation, concentration, and retention of marine debris, including plastics (Moore et al., 2001; Moore, 2008; Barnes et al., 2009). Eriksen et al. (2014) found surface plastics in all ocean regions, converging in gyres in both the northern hemisphere and southern hemisphere. Although coastal population density is much lower in the southern hemisphere, the total amount of plastics was within the same range for oceans in both northern and southern hemispheres. The authors also modeled a loss of microplastics from the sea surface and estimated plastic pollution on the sea surface is 0.1% of the world annual production (Eriksen et al., 2014).





**Figure 6. Illustration of the North Pacific Ocean central gyre (NOAA, 2014).**

As a case study, Tern Island, part of the Hawaiian Islands National Wildlife Refuge, has been a subject of ecological concern given the significant quantity of marine plastics in the area (Friedlander et al., 2009) and the known release of hazardous substances associated with past activities on the Island (U.S. EPA, 2014a). Tern Island and the surrounding French Frigate Shoals is designated as critical habitat for the endangered Hawaiian monk seal (*Monachus schauinslandi*) and the threatened Hawaiian green sea turtle (*Chelonia mydas*) and a number of other endemic species. The Hawaiian Monk Seal, the nation's most endangered marine mammal, is in precipitous decline with the species facing possible extinction. Among the stressors potentially impacting the survival of the Monk Seals, are high levels of PCBs found in their blood and blubber. Tern Island/French Frigate Shoals is located in the North Pacific Subtropical Convergence Zone where a high concentration of marine debris has been observed to accumulate (shown in Figure 6), and studies have documented plastics in all 18 seabird species found nesting on Tern (U.S. EPA, 2014a). In September 2014, the EPA completed a Preliminary Assessment of the impacts of marine debris, including plastics, on the threatened and endangered species on and around Tern Island (U.S. EPA, 2014a). PCBs and lead have been detected in sediment, soils, ground water, surface water, and biological samples collected on Tern Island (U.S. EPA, 2014a). Landfilled military waste (e.g., scrap metal, cable, batteries, and electronic equipment) is recognized as a key source of these hazardous chemicals on Tern Island, and the Preliminary Assessment specifically identifies plastic debris as a potential exposure pathway for PCBs and other chemicals (U.S. EPA, 2014b). EPA postulates that ingested plastic particles may be acting as a transport mechanism to carry chemical contaminants such as PCBs into the food chain. Trends in macro marine debris were analyzed on Tern Island from 1990-2006. During that 16-year timeframe, over 52,000 marine debris items were deposited on the beaches of Tern Island, 71% of which were plastics (U.S. EPA, 2014a).

## **4 Toxicological Impacts of Chemicals Associated with Plastics on Aquatic Organisms and Aquatic-Dependent Wildlife**

The adverse physical impacts to organisms from plastic debris in the aquatic environments, including ingestion, entanglement, and smothering, have been well documented (reviewed by Kühn et al., 2015). Ingestion of plastic is less visible than entanglement, but may lead to direct mortality or indirect mortality due to poor nutrition or dehydration (Browne et al., 2015; Kühn et al., 2015). However, the ingestion of plastics also establishes an exposure pathway between chemical additives or chemicals sorbed to plastics and the organism which has ingested this plastic. This section reviews the literature in terms of: (1) how aquatic organisms and aquatic-dependent wildlife may be exposed to plastics and associated contaminants (Section 4.1) and (2) the potential for chemicals associated with plastics to bioaccumulate, biomagnify, and cause toxic effects in aquatic life (Section 4.2).

### **4.1 Routes of Exposure**

As plastic production grows, more enters the aquatic environment as debris. Several recent reviews of the literature on the lethal and sublethal interaction of marine debris and aquatic life and aquatic-dependent wildlife have found that at least 693 species are impacted by debris in the aquatic environment (e.g., entanglement or ingestion; Gall & Thompson, 2015; Kühn et al., 2015; SCBD, 2012), with 92% of all encounters occurring between individual organisms and debris reported to be with plastic (Gall & Thompson, 2015). Reports of plastic ingestion have been made for 13,110 individuals from 208 species and reports of entanglement have been made for 30,896 individuals from 243 species (Gall & Thompson, 2015). The most commonly reported plastic items causing impacts to organisms included rope and netting (24%); fragments (20%)<sup>1</sup>; packaging (17%) including plastic bags; fishing debris (16%); and microplastics (11%) (SCBD, 2012). It is possible that the higher incidence of impacts from rope and netting compared to other plastic types is related to a higher number of studies reporting on entanglement as opposed to ingestion impacts (see Ivar do Sul and Costa, (2014) and Lavers et al. (2014)) as the detection of ingestion is typically less obvious and requires a necropsy to confirm (Gall & Thompson, 2015). However, in all species groups, the number of species reported to have ingested marine debris (much of which is plastic) has increased since 1997 (Table 3) (Gall & Thompson, 2015; Kühn et al., 2015; SCBD, 2012). Importantly, our understanding of the types of species that have been impacted by ingested debris reflects only a small fraction of the biodiversity of aquatic life, and therefore does not reflect the true extent of plastic debris ingestion.

The processes by which organisms may be exposed to the chemicals associated with plastics and associated contaminants include direct ingestion (i.e., if the animal mistakes the plastic as prey or food or inadvertently consumes plastics while feeding), indirect ingestion (i.e., consumption of prey that ingested the plastic), and dermal exposure (GESAMP, 2015). In addition, the size of the plastic pellet or fragment strongly affects the rate at which sorbed chemicals may be subsequently desorbed into organisms after ingestion. The smaller the particle or fragment, the larger the relative surface area-to-volume ratio, and the greater relative adsorption and desorption opportunity (Teuten et al., 2009). Additionally, the

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<sup>1</sup> SCBD (2012) did not specify how “fragments” were defined and how they were differentiated from microplastics.

retention time and exposure to the organism’s digestive fluids will affect the extent and rate of desorption.

Aquatic organisms and aquatic-dependent wildlife can be selective in the types, forms, colors, and sizes of plastics they ingest depending on their foraging technique and diet (Derraik, 2002). For example, Carpenter et al. (1972) was one of the first studies to report plastic ingested by fish, and found only white plastic spherules in the digestive tracts of various fish off the coast of southern New England. In a study of loggerhead sea turtles (*Caretta caretta*) in the Mediterranean Sea in the late 1980s, the turtles appeared to only ingest white plastic particles, including PVC and extruded polystyrene (Gramentz, 1988). A 14-year survey of Western North Atlantic seabirds found that the seabirds studied were selective in the shapes and colors of plastics they ingested as well (Moser and Lee, 1992). Rochman et al. (2015) were the first to assess fish and shellfish from markets in Indonesia and California, USA for the presence of anthropogenic debris for aquatic life sold for human consumption. Anthropogenic debris was observed by visual interpretation in 28% of fish in Indonesia, and 25% of fish from California, as well as 33% of shellfish evaluated overall. All debris in fish from Indonesia were plastic fragments, whereas anthropogenic debris recovered from fish in California markets were primarily fibers.

**Table 3. The number of species with records of ingestion as documented in the reviews by Laist (1997) and Gall and Thompson (2015).**

(Modified version of Table 1 in Gall and Thompson, 2015).

Species Group	Total # of Known Species	# of Species with Ingestion Records		% Increase in the # of Species with Ingestion Records (1997-2015)
		Laist (1997)	Gall & Thompson (2015)	
Fish	16,754	33 (0.20%)	50 (0.30%)	52%
Seabirds	312	111 (36%)	122 (39%)	10%
Sea turtles	7	6 (86%)	6 (86%)	0%
Marine Mammals	115	26 (23%)	30 (26%)	15%

## 4.2 Bioaccumulation and toxicological effects of chemicals associated with plastics

Numerous aquatic organisms ingest plastics (see Section 4.1) and because they can act as a source and sink for chemicals incorporated into plastics as part of the manufacturing process and chemicals that adsorb onto their surface from the aquatic environment (Rochman, 2015), there is the potential for these chemicals to be transferred to organisms (Teuten et al., 2007, 2009). However, whether organisms can bioaccumulate chemicals from ingested plastic and the relative importance of this exposure route compared to other pathways has been a recent topic of discussion (Koelmans et al., 2016). Plastic is a concern as a vector for chemicals if it increases exposure levels, and if that exposure brings the chemical’s body burden to higher levels than effect thresholds; whether plastics play a role as vectors depends on the gradient between the plastic and the organism (Koelmans, 2015; Bakir et al., 2016; Ziccardi et al., 2016). The following sections describe (1) empirical field and laboratory research on bioaccumulation of chemicals associated with plastics, as well as recent modelling studies to address this issue, and (2) studies focusing on the toxicological effects of chemicals associated with plastics.



## 4.2.1 Bioaccumulation

### Field Studies

Several studies provide evidence from the field that concentrations of chemicals are positively correlated with ingested plastic densities or chemical concentrations. Fossi et al. (2014) suggested that the presence of chemicals, specifically phthalates and organochlorines, in basking sharks (*Cetorhinus maximus*) and fin whales (*Balaenoptera physalus*) might be evidence of microplastic ingestion. There have been a number of field studies on birds. For example, PCBs found in the abdominal adipose tissues of dead great shearwaters (*Puffinus gravis*) in Australia were correlated with plastic loads of ingested particles (Ryan et al., 1988). Yamashita et al. (2011) investigated plastic ingestion in 12 short-tailed shearwaters (*Puffinus tenuirostris*) in the North Pacific Ocean and found total PCB concentrations and higher-chlorinated congeners in abdominal adipose tissue was not correlated with the mass of ingested plastic, but was positively correlated with lower-chlorinated congeners. Since the natural prey of the shearwaters concentrate higher-chlorinated congeners, but plastics have been found to retain lower-chlorinated congeners, the authors conclude that the PCBs are likely to have been accumulated from plastic. Tanaka et al. (2013) also studied short-tailed shearwaters from the North Pacific Gyre and reported that three in 12 had detectable PBDE congeners (BDE209 and BDE183) in their fatty tissues. These PBDE congeners were not found in the prey of short-tailed shearwaters but were found in the plastic particles within the bird's digestive tracts, suggesting that ingested plastic was the source of PBDEs. Lavers et al. (2014) found high concentrations of chromium and silver in fledgling flesh-footed shearwaters (*Puffinus carneipes*), and the level for both metals was positively correlated with the mass of plastic ingested by the birds.

Fish have also been found to ingest plastic and have elevated tissue concentrations of chemicals associated with plastics. Gassel et al. (2013) investigated plastic ingestion and tissue concentrations of persistent organic pollutants and nonionic surfactants in juvenile yellowtail (*Seriola lalandi*) from the North Pacific Central Gyre. They found evidence of synthetic debris in 10% of the fish sampled and nonylphenol was found in one-third of their samples. Nonylphenol was previously measured in plastic from the gyre and because it is associated with anthropogenic sources, the authors concluded that long-range transport was unlikely and exposure to the additive is most likely from plastic debris. Rochman et al. (2014b) found a significant relationship between the density of plastic debris and the concentration of higher-brominated PBDE congeners typical of those found in plastic in myctophid fish (lanternfish) tissues in the same region of the South Atlantic Ocean. However, the challenge with field studies is that it is difficult to definitively link bioaccumulation of chemicals to the ingestion of plastics versus other uptake pathways (Koelmans et al., 2016).

### Laboratory Studies

Some experimental studies provide evidence that chemicals can be transferred from plastics to aquatic organisms. In the laboratory, invertebrates have been found to accumulate both chemicals associated with plastic manufacturing and chemicals adsorbed to the plastic. Besseling et al. (2013) found that concentrations of PCBs in tissue increased by a factor of 1.1 to 1.5 in lugworms (*Arenicola marina*) exposed to polystyrene microplastics mixed with PCB-contaminated sediment compared to contaminated sediment alone. However, there was a decrease in bioaccumulation with increasing plastic concentration that the authors attributed to the physical effects of plastic ingestion. Browne et al. (2013) found that both additive chemicals and chemicals sorbed from the surrounding environment desorb from PVC and can accumulate in lugworms after ingestion. PBDEs were found to bioaccumulate in amphipods (*Allorchestes compressa*) exposed in the presence and absence of polyethylene particles (Chua et al.,

2014). PBDE concentrations were measured in amphipod tissues after a 72-hr exposure indicating transfer to tissues. However, the authors observed that the presence of microplastics reduced the uptake of PBDE compared to controls, suggesting the PBDEs were retained within the microplastic and were less bioavailable than unadsorbed PBDEs. A study conducted with fish (Japanese medaka; *Oryzias latipes*) fed polyethylene pellets at 10% of their diet reported that mean concentrations of hydrophobic organic compounds in body tissues were 1.2 to 2.4 times greater than those in control fish that were not fed the pellets (Rochman et al., 2013a). In a field experiment, Teuten et al. (2009) fed eight 40-day old streaked shearwater (*Calonectris leucomelas*) chicks plastic resin pellets contaminated with PCBs mixed with natural fish prey. Within one week, PCB concentrations, especially lower-chlorinated congeners which were relatively enriched in the plastic compared with the fish, increased in preen gland oil, suggesting that plastic ingestion may facilitate PCB accumulation. Other researchers have found that polystyrene beads amended with fluoranthene did not directly increase the amount of fluoranthene in mussels, however the beads themselves caused cellular and tissue damage (Paul-Pont et al., 2016). While these studies do confirm the potential for chemicals associated with plastics to bioaccumulate in aquatic organisms once ingested, most of these studies were not environmentally realistic in that a large chemical concentration gradient existed favoring transfer by using unexposed organisms and feeding them high levels of plastics or plastics with high concentrations of chemicals (Koelmans et al., 2016). Indeed, some experimental studies have shown plastic ingestion may reduce bioaccumulation by sorbing chemicals to ingested plastic (Teuten et al., 2007; Gouin et al., 2011; Koelmans et al., 2013; Chua et al., 2014).

### **Modeling Studies**

The relative contributions of plastic- and trophic-derived contaminants can be difficult to quantify in both field observational studies and experimental studies (Bakir et al., 2014; Holmes et al. 2012; Ryan et al., 1988), and therefore several researchers have applied equilibrium-partitioning modeling to provide indirect evidence of the transfer of chemicals associated with plastics into organisms compared to other exposure pathways. In general, experimental studies and modeling studies both predict an increase in bioaccumulation of up to a factor of two to three if plastic is the only source of uptake. However, considering more environmentally relevant situations, such as accounting for all potential routes of exposure, the role of plastics may be relatively small compared with other pathways (Koelmans, 2015). Food-web models developed by Gouin et al. (2011) predicted reductions in bioaccumulation for PBT chemicals with log  $k_{ow}$  values between 5.5 and 6.5 in piscivorous fish with polyethylene as 10% of their diet due to the high adsorption affinity of the chemicals for plastic. Koelmans et al. (2013) developed a bioaccumulation model for hydrophobic organics which was evaluated using recent laboratory research on the lugworm and accumulation of PCBs from polystyrene microplastic (Besseling et al., 2013) using several environmentally realistic scenarios. The model results indicated a negligible contribution of plastic to the accumulation of PCBs compared to other exposure pathways. In a follow up study using the same model, Koelmans et al. (2014a) investigated leaching of the plastic additives nonylphenol and BPA after plastic ingestion to lugworms and cod (*Gadus morhua*). The authors argue that additives may be more relevant than adsorbed chemicals because the plastic itself would be the source, particularly for larger organisms with longer gut retention times. However, the model results suggest that bioaccumulation of nonylphenol and BPA through plastic ingestion by lugworms and cod would be negligible. Recently, Herzke et al. (2016) investigated the bioaccumulation of hydrophobic organic chemicals (PCBs, PBDEs, and DDTs) in northern fulmars (*Fulmarus glacialis*) off the coast of Norway. Using fugacity calculations and bioaccumulation modeling, the authors conclude that plastic is more likely to act as a passive sampler than as a vector of these chemicals and ingested plastic reflects the chemical profiles of ingested natural prey. The modeling studies, using invertebrates, fish, and seabirds, demonstrate that while plastics have a high capacity to sorb chemicals, the desorption of the chemical once ingested may not be a major

factor because chemicals bioaccumulated from natural prey may overwhelm the chemical dose from the plastic (Koelmans et al., 2016; Ziccardi et al., 2016).

#### **4.2.2 Toxicological Effects**

Currently, the research investigating harmful toxicological effects of chemicals associated with plastics are limited to several experimental studies and may not adequately reflect environmental concentrations of either the chemicals associated with the plastics or realistic exposure scenarios via various routes (Gall & Thompson, 2015; Koelmans et al., 2014). However, there are some studies that indicate that additional research is needed to clarify the scope of the potential issue. For example, Lavers et al. (2014) found that body condition is negatively influenced by the amount of ingested plastic in flesh-footed shearwaters, and that the shearwater contaminant load (chromium and silver) was positively related to the amount of ingested plastic.

There have been several laboratory toxicology tests using invertebrates. The toxicity of chemicals desorbed from 32 plastic products to waterfleas (*Daphnia*) was performed via 24-hr batched and 3-day diffusion tests (Lithner et al., 2009). The study revealed that *Daphnia* were most affected by the silver within a compact recordable disc, followed by leachate from plastics made of plasticized PVC (e.g., artificial leather, a bath tub toy, and a table cloth), and polyurethane (e.g., floor coating, a child's handbag, and artificial leather). *Daphnia* 48 hour EC<sub>50</sub> values ranged from 5 - 80 g plastic material/L for nine of the tested plastic products; leachate from the remaining plastic products did not demonstrate toxicity to *Daphnia*. Browne et al. (2013) found that nonylphenol, phenanthrene, and PBDE-47 sorbed to PVC can transfer to the tissues of uncontaminated lugworms following ingestion. Their research found non-significant trends to suggest that the lugworms may have accumulated enough pollutants by ingesting the microplastic to show potential patterns of reduced feeding (PBDE), compromised immunity (nonylphenol), and reduced antioxidant capacity (PVC) (Browne et al., 2013). Another laboratory study involving the lugworm investigated the effect of microplastic uptake on weight loss and the potential for bioconcentration of PCBs via bioassays. According to Besseling et al. (2013), higher concentrations of polystyrene microplastics within sediment were positively correlated with weight loss in *A. marina*. The authors cite other research indicating that microplastics have been found in marine sediments at concentrations as high as 81 mg/kg, which is approximately three orders of magnitude below the author's observed effects concentration (Besseling et al., 2013).

Avio et al. (2015) conducted a 7-day study where polyethylene and polystyrene microplastics with and without adsorbed pyrene were fed to mussels (*Mytilus gallaprovincialis*) as their sole diet. Pyrene accumulation was found in the gills and digestive glands of the mussels at concentrations greater than those measured on the contaminated microplastics. Biochemical and cellular biomarkers were assessed and all microplastic treatment groups were found to alter immunological responses, lysosomal membrane stability, peroxisomal proliferation, antioxidant response, neurotoxic effects, and genotoxicity. The study found that most effects were not influenced by the type of polymer or contamination except for genotoxic responses where an increased frequency of micronuclei was found after exposure to pyrene-contaminated polystyrene. The authors suggest that, in the short-term, energy resources in the mussels may have been directed towards the physical impacts rather than the chemical impacts of the microplastics, but recognized that pyrene-contaminated plastics could be a potential risk for the condition of the mussels with long-term, chronic exposure. Another mussel study (Paul-Pont et al., 2016) showed that microplastics amended with fluoranthene did not change fluoranthene bioaccumulation in marine

mussels, but that microplastic exposure alone increased hemocyte mortality, changed oxidative and energetic processes in mussels, and combined microplastic and fluoranthene exposure led to highest tissue alterations and anti-oxidant marker levels.

Exposure to PCBs and PBDEs has been linked to toxicological impacts in fish, and the transfer of these toxicants to fish via plastic ingestion has been documented (Rochman et al., 2013a). In a unique study, Rochman et al. (2013a) deployed pieces of low-density polyethylene in San Diego Bay, California. After three months, the levels of total PBDEs, PAHs, and PCBs on the polyethylene were measured at 1.4, 4, and 15 times greater than measured amounts on virgin low-density polyethylene, demonstrating that these contaminants sorbed to the plastics from the sea water. The marine-exposed and virgin plastics were ground and fed to uncontaminated Japanese medaka as 10% of their diet via the water column for two months to simulate likely exposure in the wild. The study found that PBDEs and some PCBs bioaccumulated in the fish and induced liver toxicity (i.e., glycogen depletion, fatty vacuolation, and single cell necrosis), but the authors concluded that the observed toxicity resulted from both the sorbed contaminants and the plastic material itself (Rochman et al., 2013a). In a follow-up study using the same fish, Rochman et al. (2014c) investigated whether the chemicals associated with plastics produced endocrine-disrupting effects in the fish. They found altered gene expression in male fish exposed to the marine-deployed plastic (down-regulation of choriogenin) and in female fish exposed to both the marine-deployed plastic (down-regulation of vitellogenin, choriogenin, and the estrogen receptor) and virgin plastic (down-regulation of choriogenin). The changes in gene expression observed in females fed the virgin plastic suggests that the chemicals within the plastic may induce endocrine-disrupting effects.

While biomarkers of exposure and physiological signs of stress have been observed in test organisms after ingestion of chemicals associated with plastics, few studies (with the exception of Browne et al., 2013) have found effects on endpoints which are expected to directly affect populations such as growth, survival, and reproduction (Ziccardi et al., 2016). In addition, in two of these studies (Browne et al., 2013 and Rochman et al., 2013c), plastics alone induced adverse effects, but the effects were greater when exposure to both the plastic and the chemical sorbed to the plastic was assessed. In light of the modeling studies discussed above, the physical impact associated with exposure to plastic may be as concerning as the potential chemical impacts (Koelmans et al. 2014).

## **5 Conclusions and Research Needs**

Plastic particles are ubiquitous in the aquatic environment and are routinely found along beaches, in sediment, within the water column, and at the water surface (Thompson et al., 2009). Plastic debris can be found in freshwater and marine environments ranging from coastlines near densely populated areas to the remote open ocean and along remote island shorelines (Cole et al., 2011; Lambert et al., 2014; Eerkes-Medrano et al., 2015). The very characteristics that make plastics useful (e.g., durability and longevity) allow plastics to persist in the aquatic environment (Engler, 2012). Most plastics (by particle number) encountered in the aquatic environment are microplastics, which can be roughly the size of a grain of sand or a planktonic organism (Browne et al., 2008; Eriksen et al., 2014; Ivar do Sul and Costa, 2014). Methods to extract, isolate and identify these microplastics exist, but standardization to enumerate and identify these very small particles need to be standardized. Numerous research studies demonstrate that plastics are ingested (either directly or via prey) by aquatic invertebrates, fish, seabirds, sea turtles, and marine mammals (Laist, 1997; SCBD, 2012; Goldstein and Goodwin, 2013; Setälä et al., 2014; Kühn et al., 2015).

Plastics in aquatic systems contain chemicals originating from the plastic material, chemicals added during the manufacturing process, as well as organic chemicals, metals, and other contaminants sorbed from the water column (Mato et al., 2001; Hirai et al., 2011; Lithner et al., 2011; Rochman et al., 2013). Given that many of these chemicals have been found to have harmful effects once in the aquatic environment, the potential toxicological impacts of these chemicals associated with plastic once ingested by aquatic organisms and aquatic-dependent wildlife is an area of concern (Mato et al., 2001; Oehlmann et al., 2009). However, aquatic organisms and seabirds face a multitude of environmental stressors and attributing toxicological impacts directly to the ingestion of plastics and associated contaminants is challenging because organisms are exposed to metals, and organic chemicals including PBT contaminants from wastewater discharges, atmospheric deposition, and other sources in addition to plastics. The extent to which plastics are a relative source of metals and other chemicals to aquatic organisms and aquatic-dependent wildlife is a recent area of study.

There is evidence that aquatic organisms and aquatic-dependent wildlife accumulate chemicals from ingested plastics. Field studies have observed correlations of plastic densities or chemical concentrations in plastic with chemical concentrations in organisms and laboratory experiments document transfer of chemicals from plastic to organisms when there is a concentration gradient favoring transfer (i.e., high concentrations of chemicals on plastics and/or unexposed experimental organisms; see Section 4.2.1). Limited modeling approaches have been used to attempt to mimic environmentally realistic scenarios, and the models generally show a small to negligible contribution of plastic to the bioaccumulation of associated chemicals to aquatic organisms and seabirds, relative to other sources (reviewed in Koelmans et al., 2016). Some recent reviews suggest that, the bioaccumulation of chemicals associated with plastics is most likely overwhelmed by uptake through other pathways; however, this does not imply that plastics do not have negative effects on aquatic organisms (GESAMP, 2015; Koelmans et al., 2016). A limited number of toxicological studies have been performed, mainly in the laboratory, investigating the effects of chemicals associated with plastics. Similar to the laboratory bioaccumulation studies, many of the toxicological studies were conducted under environmentally unrealistic situations that favored accumulation of the chemicals from the plastic. While negative sublethal effects were observed in treatments with chemically contaminated plastics and effects were often greater than treatments with the plastic alone, adverse effects were also demonstrated in organisms exposed to the plastic alone (see Section 4.2.2).

Based on the review of the literature performed to support the development of this document, a number of research needs have been identified:

1. Studies have demonstrated that plastic debris can contain or sorb chemicals of concern, but chemical exchange kinetics under conditions of weathering, degradation, and biofilm formation are poorly understood (Koelmans et al., 2015). Further research to gain a better understanding of the fate of chemicals both sorbed to and in plastics under differing environment conditions and within an organism after ingestion is needed.
2. Laboratory experiments and modeling approaches confirm that chemicals can transfer from plastic to organisms, but because organisms in the environment can accumulate the same classes of chemicals from other sources, further research on the relative role plastics play in chemical contaminant transfer to the tissues of organisms compared to other exposure pathways (aqueous dermal exposure and ingestion from natural prey) is needed.

3. Distinguishing the chemical from physical effects of plastics in the field is a challenge. Laboratory experiments have shown toxicological effects from both the chemicals associated with plastics and the plastic itself. Further research is needed to understand the relative impacts of physical and chemical effects of ingested plastic particles.
4. There is relatively little known about nanoplastics compared to other plastic size classes, and they are difficult to study due to the lack of detection methods (Koelmans et al., 2015). However, research is needed because the relatively high surface area of nanoplastics may result in higher concentrations per unit weight than microplastics. Nanoplastics may also have additional impacts and potentially long retention times if these particles are able to cross tissue and cellular membranes, potentially increasing their risk (Koelmans et al., 2015).

In addition to highlighting these research needs, EPA's Trash Free Waters program is a catalyst for the development of innovative strategies to keep trash from entering U.S. rivers, lakes, and coastal waters, thereby reducing the country's contribution to the ever-increasing volume of trash (particularly plastic trash) in the world's oceans. Trash Free Waters program activities support trash prevention efforts by many public and private stakeholders. Given the land-based origins of the trash problem, the program has placed a strong emphasis on helping states, municipalities, and businesses work together to define more effective ways to reduce litter, prevent trash entry into water, and minimize packaging waste. The Trash Free Waters program uses an inclusive approach of multi-stakeholder consultation, strategic planning, innovative pilot initiatives, and public/private collaboration, providing a forum for all constituencies to contribute solutions to this non-point source challenge.



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