UN environment programme

50 1972-2022

FROM POLLUTION TO SOLUTION

A GLOBAL ASSESSMENT OF MARINE LITTER AND PLASTIC POLLUTION

ISBN: 978-92-807-3881-0
Job number: DEP/2379/NA

Suggested citation
United Nations Environment Programme (2021). *From Pollution to Solution: A global assessment of marine litter and plastic pollution*. Nairobi.

Production
United Nations Environment Programme (UNEP)
Layout and figures: GRID-Arendal and Strategic Agenda
https://www.unep.org/

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A GLOBAL ASSESSMENT OF MARINE LITTER AND PLASTIC POLLUTION

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Background

The United Nations Environment Assembly continues to address the growing problem of marine litter, including plastics, and microplastics, through key resolutions adopted as follows: UNEP/EA.1/Res.6: Marine plastic debris and microplastics (2014); UNEP/EA.2/Res.11: Marine plastic litter and microplastics (2016); UNEP/EA.3/Res.7: Marine litter and microplastics (2017); UNEP/EA.4/Res.6: Marine plastic litter and microplastics (2019); and UNEP/EA.4/Res.9: Addressing single-use plastic products pollution (2019).

In 2016 the United Nations Environment Programme (UNEP) published a report, *Marine Plastic Debris and Microplastics – Global Lessons and Research to Inspire Action and Guide Policy Change*,[1] in response to UNEP/EA.1/Res1.6. The report focused on:

- identification of the key sources of marine plastic debris and microplastics;
- possible measures and best available techniques and environmental practices to prevent the accumulation and minimize the level of microplastics in the marine environment;
- recommendations for the most urgent actions;
- areas especially in need of more research, and other relevant priority areas.

At the fourth meeting of the United Nations Environment Assembly the Executive Director of UNEP, in resolution UNEP/EA.4/Res.6 paragraph 2, was requested to:

> "…immediately strengthen scientific and technological knowledge with regard to marine litter including marine plastic litter and microplastics, through the following activities:
>
> … (b) Compiling available scientific and other relevant data and information to prepare an assessment on sources, pathways and hazards of litter, including plastic litter and microplastics pollution, and its presence in rivers and oceans; scientific knowledge about adverse effects on ecosystems and potential adverse effects on human health; and environmentally sound technological innovations;"

Given that substantial new research has been conducted since the 2016 UNEP report, this assessment provides an update by highlighting new developments and building on the earlier report. This assessment is intended to inform discussions at the fifth session of the United Nations Environment Assembly (UNEA-5.2). In earlier drafts provided input to the Ad hoc open-ended expert group on marine litter and microplastics on the design of possible actions and the development of policy-relevant recommendations.

In 2019 the Executive Director of UNEP established a Scientific Advisory Committee on Marine Litter and Microplastics (SAC). The main objective of the SAC was to provide input and guidance during the preparation of this assessment. United Nations Member States, members of specialized agencies, and accredited major groups and stakeholders were invited to nominate experts to serve as members of the SAC. Once the SAC was established, the experts were invited to support the development of the assessment by providing scientific information, data, experiences, expert opinions, reviews and advice to ensure the highest scientific quality of its content. An in-person meeting of SAC members in February 2019, as well as a number of online working groups and other meetings, were organized by UNEP to guide and inform the implementation of paragraph 2 of UNEP/EA.4/Res.6, in particular the development of the assessment as requested in subparagraph 2(b).

The SAC members recommended that UNEP develop the assessment based on published evidence on the sources and drivers of marine litter, especially plastics and microplastics; the types and volumes of plastics found in waste streams entering the oceans and their pathways including transport between and within different compartments or zones in freshwater, soil, air and marine ecosystems, ingestion by animals and humans and uptake by plants and microorganisms; the hazards and impacts on oceans, marine ecosystems and human health; existing and new monitoring and observation programmes, including those involving citizen science; and examples of solutions, environmentally sound technologies and risk reduction measures. The assessment was to provide evidence to enable policymakers and the wider public to comprehend the magnitude and severity of the effects and risks associated with marine litter, especially plastics and microplastics; identify gaps in knowledge; raise awareness of solutions; and help stimulate global interventions to control and prevent marine plastic pollution and to safeguard human and ecological health.

In line with best practice in global assessment processes, the assessment is based on open access publications from the peer reviewed literature, intergovernmental and national reports, and, where relevant, stakeholder publications. The authors and SAC members also provided regional and national information on interdisciplinary and contextually relevant case studies.

Acknowledgements

Members of the Scientific Advisory Committee (Government and Major Groups and Stakeholder nominated experts):

Abid Abdeslam (National Laboratory for Pollution Monitoring, Morocco), Zahra Alavian (Department of Environment, Coasts, and Wetlands, Iran), Andres Hugo Arias (Argentinean Institute of Oceanography, Argentina), Thomas Backhaus (University of Gothenburg, Sweden), Bushra Hamid Ahmed Bashier (Higher Council of Environment - Khartoum, Sudan), Bhaguthsing Beerachee (Ministry of Environment, Mauritius), Lee Bell (International POPs Elimination Network, Sweden), Christian Sekomo Birame (University of Rwanda, Rwanda), Stephanie B. Borrelle (University of Toronto, Canada), Kalpana Chaudhari (Institute for Sustainable Development and Research, India), Benqlilou Chouaib (The National School of Engineering ENIM in Rabat, Morocco), Roxana Diaz (Grupo GEA, Peru), Thinley Dorji (National Environment Commission Secretariat, Bhutan), Mariatou Dumbuya (National Environment Agency, The Gambia), Sarah Dunlop (The Minderoo Foundation, Australia), Marcus Eriksen (5 Gyres Institute, Leap Lab, United States), Johanna Eriksson (The Swedish Agency for Marine and Water Management, Sweden), Trisia Angela Farrelly (Massey University, New Zealand), Andrew Forrest (The Minderoo Foundation, Australia), François Galgani (Institut Français de Recherche pour l'Exploitation de la Mer (IFREMER), France), Aziza Geleta (Ethiopian Embassy in Nairobi, Kenya), Samia Gharbi (Association Abel Granier, Tunisia), Olfat Hamdan (Ministry of Environment of Lebanon, Lebanon), Georg Hanke (European Commission - Joint Research Centre, Belgium), Mihail Otilia Hariclia (Ministry of Environment, Waters and Forests, Romania), Patricia Holm (University of Basel, Switzerland), Shahriar Hossain (Environment and Social Development Organization-ESDO, Bangladesh), Atsuhiko Isobe (Research Institute for Applied Mechanics, Kyushu University, Japan), Andrea A. Jacobs (Ministry of Agriculture, Fisheries and Barbuda Affairs, Antigua and Barbuda), Jenna Jambeck (University of Georgia, United States), Sarojini Jayasekara (Central Environment Authority, Sri Lanka), La Daana Kanhai (University of the West Indies, Trinidad and Tobago), Cheryl Rita Kaur (Maritime Institute of Malaysia, Malaysia), Ahmet Erkan Kideys (Middle East Technical University Institute of Marine Sciences, Turkey), Winnie Wing Yee Lau (The Pew Charitable Trusts, United States), Daoji Li (East China Normal University, China), Juan Pablo Lozoya (Regional Centre for the East Zone, University of the Republic, Uruguay), Jerome Sebadduka Lugumira (National Environment Management Authority, Uganda), Youna Lyons (National University of Singapore, Singapore), Thomas Maes, Milica Mandic (University of Montenegro, Montenegro), Mwita Marwa Mangora (Institute of Marine Sciences, Tanzania), Cristian Enrique Brito Martinez (Ministry of Environment, Chile), Collins Bruno Mboufack (Ministry of Environment of Cameroon, Cameroon), Miriam Mekki (Norwegian Environment Agency, Norway), Viana Martinez Roxana Melitza (Ministerio del Poder Popular para el Ecosocialismo, Venezuela), Maxine Monsanto (Ministry of Agriculture, Fisheries, Forestry, Environment, Sustainable Development and Immigration, Belize), Alethia Vazquez Morillas (Universidad Autonoma Metropolitana, Mexico), Eva S. Ocfemia (Manila Bay Environment, Philippines), Olga Pantos (Institute of Environmental Science and Research, New Zealand), Galia Pasternak (Independent Researcher, Israel), Chelsea M. Rochman (University of Toronto, Canada), Alex Jose Saer Saker (Ministry of Environment and Sustainable Development, Colombia), Salieu Kabba Sankoh (University of Sierra Leone and West Africa Regional Fisheries Programme, Sierra Leone), Suree Satapoomin (Department of Marine and Coastal Resources, Thailand), Outi Setala (Finnish Environment Institute (SYKE), Finland), Mohamed Lamine Sidibe (Ministry of Environment, Guinea), Faustine Sinzogan (Ministry of the Environment and Sustainable Development, Benin), Ruth Spencer (Marine Ecosystems Protected Areas (MEPA) Trust, Trinidad and Tobago), Jakob Strand (Aarhus University, Denmark), Hideshige Takada (Tokyo University of Agriculture and Technology, Japan), Neil Tangri (Global Alliance for Incinerator Alternatives, Philippines), Leonardo Trasande (New York University School of Medicine, United States), Pero Tutman (Institute for Oceanography and Fisheries, Croatia), Godson Cudjoe Voado (Environmental Protection Agency of Ghana, Ghana), Mengjiao Wang (Greenpeace International, The Netherlands), Karen Watson (Environmental Protection Agency, Guyana), Mohsina Zubair (Environmental Protection Agency Pakistan, Pakistan).

Overall lead author:

Jacqueline McGlade (University College London; and Strathmore University Business School, United Kingdom).

Lead authors:

Irene Samy Fahim (Nile University, Egypt), Dannielle Green (Anglia Ruskin University, United Kingdom), Philip Landrigan (Harvard School of Public Health and Mount Sinai School of Medicine, United States).

Contributing authors:

Anthony Andrady (North Carolina State University, United States), Monica Costa (Universidade Federal de Pernambuco, Brazil), Roland Geyer (University of California Santa Barbara, United States), Rachel Gomes (University of Nottingham, United Kingdom), Aileen Tan Shau Hwai (Universiti Sains Malaysia), Jenna Jambeck (University of Georgia, United States), Daoji Li (East China Normal University, China), Chelsea Rochman (University of Toronto, Canada), Peter Ryan (University of Cape Town, South Africa), Martin Thiel (Universidad Católica del Norte, Chile), Richard Thompson (University of Plymouth, United Kingdom), Kathy Townsend (University of the Sunshine Coast, Australia), Alexander Turra (University of São Paulo, Brazil).

Reviewers:

Tatsuya Abe, Stefano Aliani, Sandra Averous-Monnery, Michael Bank, Eva Bildberg, Anne Bowser, Bethanie Carney Almroth, Jennifer de France, Jost Dittkrist, Maria Deudero, Claudia Giacovelli, Kirsten Gilardi, Julie Goodhew, Natalie Harms, Nils Heuer, Andrea Hinwood, Kitakang Iya Joyce, Yasuhiko Kamakura, Peter Kershaw, Bart Koelmans, Arvind Kumar, Stephanie Laruelle, Carole Manceau, Nikolai Maximenko, Allan Meso, Llorenç Mila I Canals, Tapiwa Nxele, Kei Ohno, Sabine Pahl, Catalina Pizarro, Jordi Pon, Amelie Ritscher, Amparo Roda, Marta Ruiz, Aphrodite Smagadi, Emily Smail, Karsten Steinfatt, Anthony Talouli, Franco Teixeira de Mello, Elisa Tonda, Noam van der Hal, Erik van Sebille, Yegor Volovik, Jan Andries van Franeker, Karl Vrancken, Feng Wang, Ran Xie, Riccardo Zennaro, Weiwei Zhang, Shuang Zhu and Dror Zurel.

Production Team:

Leticia Carvalho, Carla Friedrich, Tessa Goverse, Heidi Savelli and Tabea Zwimpfer (UNEP).

Editor:

John Smith.

Abbreviations and acronyms

ABNJ	Areas beyond national jurisdiction
ALDFG	Abandoned, lost or otherwise discarded fishing gear
AMR	Antimicrobial resistance
ASTM	International standards organization, formerly the American Society for Testing and Materials
DFG	Derelict fishing gear
EFSA	European Food Safety Authority
FAD	Fish aggregation device
FAO	Food and Agriculture Organization of the United Nations
GESAMP	Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection
GHG	Greenhouse gas(es)
GPML	Global Partnership on Marine Litter
IMO	International Maritime Organization
ISO	International Organization for Standardization
POPs	Persistent organic pollutants
UNCLOS	United Nations Convention on the Law of the Sea
UNEA	United Nations Environment Assembly
UNEP	United Nations Environment Programme
WHO	World Health Organization
WTO	World Trade Organization

Common polymers:

ABS	acrylonitrile butadiene styrene
AC	acrylic
EP	epoxy resin (thermoset)
EPS	expanded polystyrene
HDPE	polyethylene high density
LDPE	polyethylene low density
LLDPE	polyethylene linear low density
PA	polyamide (nylon) 4, 6, 11, 66
PC	polycarbonate
PCL	polycaprolactone
PE	polyethylene
PET	polyethylene terephthalate
PGA	poly (glycolic acid)
PLA	poly (lactide)
PMMA	poly (methyl methacrylate)
PP	polypropylene
PS	polystyrene
PU	polyurethane (also abbreviated as PUR)
PVA	polyvinyl alcohol
PVC	polyvinyl chloride
SBR	styrene-butadiene rubber
TPU	thermoplastics polyurethane

Common chemical additives in plastics:

BFRs	brominated flame retardants
BPA	bisphenol A
BPF	bisphenol F
BPS	bisphenol S
DBP	dibutyl phthalate
DEP	diethyl phthalate
DEHP	di-(2-ethylhexyl)phthalate
FRs	flame retardants
HBCD	hexabromocyclododecane
NP	nonylphenol
NPE	nonylphenol ethoxylate
PBDEs	polybrominated diphenyl ethers (penta, octa and deca forms)
Phthalates	phthalate esters
TBBPA	tetrabromobisphenol

Common organic contaminants sorbed by plastics:

DDT	dichlorodiphenyltrichloroethane
HCHs	hexcyclohexane
PAHs	polycyclic aromatic hydrocarbons
PCBs	polychlorinated biphenyls

Glossary of terms and definitions

This glossary has been compiled by the lead authors of the report, drawing on glossaries and other resources available on the websites of various organizations, networks and projects.

Additives: Additives used in a manufacturing process include fillers, plasticizers, flame retardants, UV and thermal stabilizers, antimicrobial agents, colorants, residual monomers and catalysts trapped in plastic resins. They contain a number of hazardous chemicals, such as phthalates and polybrominated diphenyl ethers, and potentially toxic substances, i.e. hazardous substances that have to be leached- released-emitted before any toxicity is expressed (Andrady 2017; Hahladakis *et al.* 2018; Andrady and Rajapakse 2019; GESAMP 2020a). Many of the chemicals associated with plastics are listed as hazardous under the Stockholm Convention on Persistent Organic Pollutants and in national legislation or regulations, such as the United States Environmental Protection Agency (2014) priority pollutants list,97 because they are persistent, bioaccumulative and/or toxic (Gallo *et al.* 2018).

Atmospheric distillation: Plastic additives such as polychlorinated biphenyls (PCBs) and fluorinated compounds volatilize at equatorial and temperate latitudes, move poleward in the atmosphere, and precipitate to land and in water from the cold air of the far north, a phenomenon termed "atmospheric distillation" (Atlas and Giam 1981; Houde *et al.* 2011; Tekman *et al.* 2020). High concentrations of persistent pollutants are thus ingested by marine microorganisms in the circumpolar regions and accumulate in top predator fish species and marine mammals (Peng *et al.* 2020; Rubio *et al.* 2020), including fish and marine mammals traditionally consumed by coastal communities and indigenous peoples in the Arctic.

Bacterial biofilms: Surface-associated bacterial communities which are embedded within an exopolymeric substance matrix.

Baseline/reference: The state against which change is measured. In the context of pathways, the term "baseline scenarios" refers to current conditions. Baseline scenarios are not intended to be predictions of the future, but rather counterfactual constructions that can serve to highlight the level of emissions that would occur without further policy effort. Typically, baseline scenarios are then compared to mitigation scenarios that are constructed to meet different goals for greenhouse gas emissions, atmospheric concentrations or temperature change. The term "baseline scenario" is used interchangeably with "reference scenario" and "no policy scenario". In much of the literature the term is also synonymous with the term "business-as-usual (BAU) scenario", although this term has fallen out of favour because the idea of business-as-usual in century-long socioeconomic projections is hard to fathom.

Bathymetric lidar: A technique to capture near-shore water depth (bathymetry) as geospatial data relating to the coastline and (shallow) waters. It can be carried out from airborne platforms and satellites. It is a method potentially facilitating efficient and fast creation of hydrographic data. These measurements have been problematic historically since ships cannot operate close to the shore while collecting acoustic bathymetric soundings. Because highly dynamic coastal shorelines can be affected by erosion, wetland loss, hurricane impacts, sea level rise, urban development and population growth, consistent bathymetric data are important and are needed to better understand sensitive coastal land/water interfaces.

Bio-based plastics are polymers that are either biosourced, biodegradable or both. It is for this reason that the term "bioplastic" should never stand alone and why it is necessary to specify, each time this word is used, the plastic's origin (biosourced or not) and end of life (biodegradable or not).

Biodegradable: Means a material can be decomposed under the action of microorganisms such as bacteria, fungi, algae or earthworms. To be truly meaningful, the term must be linked to the end products, to a timescale that is compatible with a human scale, and to the conditions of biodegradation.

Biodegradable plastics: Polymers that undergo biodegradation under specified environmental conditions (a process in which the degradation results from the action of naturally occurring microorganisms such as bacteria, fungi, and algae) and above a specified degradation time as per accepted industry standards. Accepted industry standard specifications included, but were not limited to: ASTM D6400, ASTM D6868, ISO 17088 and EN 13432.

Biodegradation: The biological process that results in the formation of water, carbon dioxide (CO_2) and/or methane, energy and by-products (residues, new biomass). It is influenced by the physicochemical (temperature, humidity, pH) and microbiological variables (quantity and nature of microorganisms) of the environment in which it occurs.

Biogenic habitat: A habitat created by plants and animals. It may be the organism itself, such as a seagrass meadow or a bed of horse mussels, or arise from an organism's activities, such as the burrows created by crabs. Biogenic habitat-forming species also perform other important roles within the ecosystem and some are harvested in their own right. Examples of biogenic habitats include mangrove forests, seagrass meadows, mussel and oyster reefs, and kelp forests.

Biological endpoint: A direct marker of disease progression (e.g. disease symptoms or death) used to describe a health effect (or a probability of that health effect) resulting from exposure to a chemical.

Biopolymers: Natural polymers derived from renewable resources of plants or animals. They can be directly synthesized by plants or animals such as polysaccharides (starch, cellulose, chitosan, etc.), proteins (collagen, gelatin, casein, etc.) and lignins, or synthesized from biological resources such as vegetable oils (rape, soybean, sunflower, etc.). Other biopolymers, such as PHA, are produced by microorganisms (bacteria) through fermentation from sugars and starch.

Biosourced: Biosourced polymers are manufactured, in part or in whole, from renewable biological resources, most often vegetable. The sources of raw materials are very varied. We find everything related to biomass, organic matter, in particular starches, sugars and vegetable oils.

Carbon intensity: The amount of emissions of CO_2 released per unit of another variable such as gross domestic product, output energy use, transport or agricultural/forestry products.

Carbon price: The price of avoided or released CO_2 or CO_2-eq (CO_2 equivalent) emissions. This may refer to the rate of a carbon tax or the price of emission permits. In many models that are used to assess the economic costs of mitigation, carbon prices are used as a proxy to represent the level of effort in mitigation policies.

Carbon tax: A levy on the carbon content of fossil fuels. Because virtually all of the carbon in fossil fuels is ultimately emitted as CO_2, a carbon tax is equivalent to an emission tax on CO_2 emissions.

Co-benefits: The positive effects that a policy or measure aimed at one objective might have on other objectives, without yet evaluating the net effect on overall social welfare. Co-benefits are often subject to uncertainty and depend on, among others, local circumstances and implementation practices. Co-benefits are often referred to as ancillary benefits.

Compostable: In terms of polymers, those that are compostable are capable of being biodegraded at elevated temperatures in soil under specified conditions and time scales, usually only encountered in an industrial composter. For industrial composting, standards apply: ISO 17088, EN 13432, ASTM 6400. This is in contrast to domestic composting (see **Composting**).

Composting: An aerobic transformation process (i.e. in the presence of oxygen, unlike methanization which is an anaerobic reaction, i.e. without oxygen) of fermentable materials under controlled conditions. It helps obtain a stabilized fertilizing material, rich in humic compounds, called compost. It is accompanied by the release of heat and carbon dioxide. It is a process widely used, especially in agricultural environments, because compost helps amend soil by improving its structure and fertility.

Core indicators: Generally considered to be generic indicators which can be measured in all contexts by participating organizations, countries, and legal parties committed to collecting data on and reporting on specific issues. They are often combined with regional or site-specific indicators. Core indicators are often designed to answer key policy questions and support all phases of environmental policymaking, from designing policy frameworks to setting targets, and from policy monitoring and evaluation to communicating to policymakers and the public. They can be established for monitoring trends, measuring performance (e.g. reaching a target), measuring efficiency (e.g. whether the situation improving) and measuring policy effectiveness (i.e. whether the measures working). Examples of core indicator sets include those maintained by the European Environment Agency (2014) (for its 32 member countries), HELCOM (Ruiz and Stankiewicz 2019), and the Regional Seas Programme (UNEP 2014).

Degradation: The partial or complete breakdown of a polymer as a result of, for example, UV radiation, oxygen attack or biological attack. This implies alteration of properties, such as discolouration, surface cracking, and fragmentation. For biodegradation, see **Biodegradable**. In the context of polymer degradation, mineralization (see **Mineralization**) is the complete breakdown of a polymer as a result of combined abiotic and microbial activity into CO_2, water, methane, hydrogen, ammonia and other simple inorganic compounds. This is different from compostable (see **Compostable**) or oxo-degradable (see **Oxo-degradable**).

Downcycling: A form of recycling that involves reusing material in less demanding applications and accepting reduced performance of the material in terms of specifications such as hardness, tensile strength, or ductility. In its new application the downcycled material replaces a material of lower economic value than the original application.

Labelling (plastics recycling): Various systems of labelling the different types of plastics are in use. An example is the numbering system for recycling: 1. PET (or PETE) – polyethylene terephthalate, used in household containers such as carbonated beverage bottles, microwavable food trays, medicine bottles, hair combs and rope. If recycled, it is commonly found in carpet, fibrefill for coats and sleeping bags, cassette tapes and sails for boats. 2. HDPE – high-density polyethylene. Many packaging applications use this material as a moisture barrier and for its chemical resistance. Safer than PET, products made of HDPE will not transmit chemicals into food or drinks, so they are typically used in snack food packages and milk and margarine containers. HPDE is also used for shampoo, detergent and bleach bottles, motor oil containers and children's toys. When recycled, HDPE is used for plastic lumber, fencing or storage crates. Like PET, it is commonly found in lab bottles and larger chemical storage containers. 3. PVC – polyvinyl chloride, commonly used in plumbing pipes, floor coverings and buildings, but also found in synthetic leather products, shower curtains, car dashboards and cable and wire sheathing. Due to its ability to resist most chemicals and bacteria, it is found in blood bags and medical tubing. 4. LDPE – low-density polyethylene. A durable, flexible plastic known for its transparency and toughness, it is often used for sandwich bags, cling wrap, squeeze bottles, grocery bags and dry-cleaning bags. It is also popular in wire and cable applications because it has stable electrical qualities. It is not commonly recycled, but can be used in lumber, garbage cans and furniture. Like other PE products, LDPE is often used in lab bottles, particularly those with narrow necks such as wash and dropper bottles. 5. PP – polypropylene. A strong film with excellent chemical resistance, this is a popular synthetic for both solid and flexible packaging. It can handle higher temperatures, so is especially good for filling with hot liquids. It is used for food storage containers, ketchup bottles, diapers, prescription bottles and yoghurt containers, as well as plastic bottle caps. It is also found in automotive battery casings. If recycled, it is used in rakes, battery cables or ice scrapers. Due to its wide service temperature range, it is one of the most popular plastics used in laboratory test tubes, vials, bottles, jars, racks and microplates. 6. PS – (expanded) polystyrene used in either rigid or foam form. Because of its clarity, it is used for medical and laboratory specimen containers such as culture tubes, Petri dishes, pipettes, wells and microplates. It is popular in food packaging and plastic cutlery; as expandable foam, it can be easily shaped into disposable coffee cups, meat, fish and cheese trays and restaurant take-out boxes. It is also popular for packaging foam and "peanuts". When recycled, it is used for insulation, rulers and license plate frames. 7. OTHER – Covers all other plastics which are hardest to recycle. An example is Acrylonitrile Butadiene Styrene (ABS), known for its strength and rigidity and used in musical instruments, golf club heads, automotive trim and bumpers, luggage and small kitchen appliances.

Leaching: The washing out of soluble ions and compounds by water draining through soil.

Macroplastic: Anything plastic that can be easily seen. Some examples are plastic bags, water bottles and nets. While they still have a negative impact on the environment, they are less likely to enter the food chain because they are hard to ingest due to their size.

Marine debris: Considered as synonymous with marine litter in the Honolulu Strategy, where it is defined to include any anthropogenic, manufactured, or processed solid material (regardless of size) discarded, disposed of, or abandoned that ends up in the marine environment. It includes, but is not limited to, plastics, metals, glass, concrete and other construction materials, paper and cardboard, polystyrene, rubber, rope, textiles, timber and hazardous materials, such as munitions, asbestos and medical waste. In some instances, marine debris may also be a vessel for dangerous pollutants that are eventually released into the marine environment. Marine debris may result from activities on land or at sea. Marine debris is a complex cultural and multi-sectoral problem that exacts tremendous ecological, economic, and social costs around the globe. A distinction may be made between natural and artificial marine debris (Maximenko *et al.* 2019). *Natural disasters* can also greatly increase inputs of all kinds of natural and artificial debris. For example, the 2011 tsunami in Japan washed about 5 million metric tons of debris into the ocean within hours (Japan Ministry of the Environment 2012). Of this amount, 3.5 million metric tons sank on the shelf, severely damaging the benthic ecosystem and, together with the radioactive spill from the Fukushima nuclear plant, badly affecting the local fishing industry. The remaining 1.5 million tons (an amount close to a full-year input of land-based plastic debris for the entire North Pacific became flotsam and a fraction of this drifted to North America and Hawai'i. The composition of tsunami debris was very complex: according to Murray *et al.* (2018), counts of all categories of debris, monitored on beaches in the State of Washington (United States), increased in 2012 by a factor of 10 compared to pre-tsunami levels. See also: **Marine litter**.

Marine litter: As defined by UNEP (1995), marine litter is any persistent, manufactured or processed solid material discarded, disposed of or abandoned in the marine and coastal environment. Marine litter consists of items that have been made or used by people and deliberately discarded into the sea or rivers or on beaches; brought indirectly to the sea with rivers, sewage, storm water or winds; accidentally lost, including material lost at sea in bad weather (fishing gear, cargo); or deliberately left by people on beaches and shores. See also **Marine debris.**

Methanization: Methanization (or anaerobic digestion) is the natural biological process of degrading organic matter in the absence of oxygen (anaerobic). It occurs naturally in some sediments, marshes, rice paddies and landfills, as well as in the digestive tract of some animals such as termites or ruminants. Some of the organic matter is degraded to methane, while some is used by methanogenic microorganisms for their growth. The decomposition is not complete and leaves the "digestate" (partly comparable to compost), which requires composting in order to be stabilized. Methanization is also a technique used in "methanizers", where the process is accelerated and maintained to produce usable methane (biogas). Organic waste can thus provide energy.

Microplastics: Microplastics have been the focus of ongoing debate as to their size limit (Thompson 2015). Some authors take a broad view, including items less than 5 mm diameter (Arthur *et al.* 2009), whereas others restrict the term to items less than 2 mm, less than 1 mm or even less than 500 µm (Cole *et al.* 2011). Depending on the upper size limit, industrial pellets may or may not be included in the term. Microplastics are categorized as primary and secondary (see below). The proportion of primary microplastics in the environment is probably small compared with secondary microplastics, except in some areas of the Great Lakes in the United States (Eriksen *et al.* 2013), but it is a largely avoidable source of pollution. In this assessment, the definition of microplastics as particles less than 5 mm in diameter is used (Arthur *et al.* 2009).

Primary microplastics are purposefully manufactured to carry out a specific function (Cole *et al.* 2011). They include certain cosmetics, hand cleaners, air blast cleaning media, and plastic beads manufactured specifically for this purpose (e.g. abrasive particles, powders for injection moulding). Nurdles or pre-production pellets and resin beads are bulk transported between manufacturing sites. They are produced separately and melted down for use by plastics producers (plastics pellets), by manufacturers of household products (personal care products and cosmetics), for ship and building cleaning (abrasive powders), and in manufacturing (powders for injection moulds and 3D printing).

Secondary microplastics represent the results of wear and tear or fragmentation of larger objects, both during use and following loss to the environment (e.g. textile and rope fibres, weathering and fragmentation of larger litter items, vehicle tyre wear, paint flakes).

Mineralization: Mineralization in the context of polymer degradation is the complete breakdown of a polymer as a result of the combined abiotic and microbial activity, into CO_2, water, methane, hydrogen, ammonia, and other simple inorganic compounds.

Monitoring: The intent to measure the current status of an environment or to detect trends in space or time of environmental parameters. Monitoring should be performed systematically by harmonized sampling methods and a consistent data and metadata management procedure.

Nanoplastics: A subcategory of microplastics created by degradation. Due to the extremely small size of nanoplastics, they enter the food chain when ingested by unicellular and multicellular marine organisms. Nanoplastics also have a high surface area to volume ratio, making them more likely to absorb organic pollutants and other hazardous contaminants. The precise definition of nanoplastics is still under debate (Gigault *et al.* 2018); some authors use size, with some favouring less than 1µm (e.g. Pinto *et al.* (2016), while others use more than 1 µm (Rios and Balcer 2019). Gigault *et al.* (2018) define nanoplastics as "particles unintentionally produced (i.e. from the degradation and the manufacturing of the plastic objects) and presenting a colloidal behaviour, within the size range from 1 to 1,000 nm". No analogies or extrapolations can be made between nanoplastics and other "nanomaterials" due to the different production pathways and physical and chemical properties. Nanoplastics are highly polydisperse in physical properties and heterogeneous in composition (Gigault *et al.* 2016; Lambert and Wagner 2016; ter Halle *et al.* 2016). Indeed, because nanoplastics are produced unintentionally from the degradation of microscale plastic litter, it is highly probable that nanoplastics will form hetero-aggregates with other natural and/or anthropogenic materials (Hüffer *et al.* 2017); in this sense the colloidal behaviour of nanoplastics is relevant. Little is known about the adverse health effects of nanoplastics in organisms, including humans (Barria *et al.* 2020).

Natural (bio-) polymers: Polyamides which occur in proteins and form materials such as wool and silk. Bio-polymers are very large molecules with a long chain-like structure and a high molecular weight, produced by living organisms. They are very common in nature and form the building blocks of plant and animal tissue. *Cellulose* (C6H10O5)n is a polysaccharide (carbohydrate chains), and is considered the most abundant natural polymer on Earth, forming a key constituent of the cell walls of terrestrial plants. *Chitin* (C8H13O5N)n is a polymer of a derivative of glucose (N-acetylglucosamine) and is found in the exoskeleton of insects and crustaceans. *Lignin* (C31H34O11)n is a complex polymer of aromatic alcohols and forms another important component of cell walls in plants, providing strength and restricting the entry of water. *Cutin* is formed of a waxy polymer that covers the surface of plants.

Oxo-biodegradation: The oxo-biodegradation of plastics is degradation identified as resulting from oxidative and cell-mediated phenomena, either simultaneously or successively (CEN TC249/WG9).

Oxo-degradable: Oxo-degradable plastics (or "fragmentable", "oxo-fragmentable", or even "biofragmentable" or "oxo-biodegradable" plastics) are polymers of petrochemical origin containing pre-oxidants, such as mineral oxidizing additives, that promote their degradation into small pieces (until they become invisible to the naked eye). These plastics can fragment, under certain conditions (light, heat, etc.), but are not biodegradable according to current standards. In addition, these additives seem to contain heavy metals whose environmental effects are currently unknown. The new European Single-Use Plastic Products Directive, approved by the European Parliament on 27 March 2019, provides for the prohibition of these oxo-degradable plastics whatever their use.

Plastics: Defined as synthetic organic polymers with thermo-plastics or thermo-set properties (synthesized from hydrocarbon or biomass raw materials), elastomers (e.g. butyl rubber), material fibres, monofilament lines, coatings and ropes (GESAMP 2019). Many plastics are produced as a mixture of different polymers and various plasticizers, colorants, stabilizers and other additives. Most plastics can be divided into two main categories: thermoplastics (capable of being deformed by heating), which include polyethylene, polypropylene and polystyrene; and thermoset (non-deformable), which include polyurethane, paints and epoxy resins. About 15 per cent of total synthetic polymer production consists of fibres, such as polyester and acrylic. Another significant component of marine litter is semi-synthetic material, such as cellulose nitrate and rayon, made from biomass (UNEP 2018b).

Plastics debris and litter: These terms are often used interchangeably. There is no agreed or official text on the categorization of plastics debris or litter; the terminology used in this report thus follows that of GESAMP (2019).

Size: There has been an ongoing discussion about the definition of different sized plastics (Galgani *et al.* 2015). For example, Andrady (2011) argued the need for three size terms: mesoplastics (500 µm-5 mm), microplastics (50-500 µm) and nanoplastics (<50 µm), each with their own set of physical characteristics and biological impacts. In this assessment the size classes proposed by Lusher *et al.* (2017) are used: Mega >1 m; Macro 25 mm-1 m; Meso 5 mm-25 mm and Micro < 5 mm (see **Microplastics**). Plastic particles with a size of 1 µm or less are termed nanoplastics (see **Nanoplastics**).

Shape: There is currently no standardized scheme for the different shapes of plastics debris. The five shape categories used for marine litter are: 1) fragments or irregular shaped particles, crystals, fluff, powder, granules, shavings; 2) fibres/ filaments, microfibres, strands, and threads; 3) beads grains, spherical microbeads, microspheres; 4) films/sheets, and polystyrene, expanded polystyrene foams; 5) pellets resin pellets, nurdles, pre-production pellets, nibs (Lusher *et al.* 2017).

Colour: Colour is not regarded as a crucial parameter for categorization of plastics debris (GESAMP 2019: Hartmann *et al.* 2019).

Plastisphere: A term used to describe the habitats on microplastics which foster different microbial communities.

Polymer: Refers to a molecule of high molecular weight consisting of a repetitive sequence of a large number of simple molecules called monomers, which may or may not be the same. The number of monomer units constituting the macromolecule is called the "degree of polymerization". Polymers are generally polymolecular, i.e. they are composed of blends of molecules of different sizes. Sugars, starch and proteins are natural polymers synthesized by prokaryotes, plants, animals or bacteria; these are called "biopolymers".

Product light weighting: A process of creating lighter products through designs that require less material or substitute heavier material with lighter and/or less energy intensive materials. Lighter material alternatives, both in weight or volume, can generate substantial energy savings in the transport and building sectors.

Source: Any process, activity or mechanism that can lead to releases of litter and plastics into the environment.

Sustainable development: Development that meets the needs of the present without compromising the ability of future generations to meet their own needs.

Uncertainty: A cognitive state of incomplete knowledge that can result from a lack of information or from disagreement about what is known or even knowable. It may have many types of sources, from imprecision in the data to ambiguously defined concepts or terminology, or uncertain projections of human behaviour. Uncertainty can therefore be represented by quantitative measures (for example, a probability density function) or by qualitative statements (for example, reflecting the judgement of a team of experts)

Foreword

Decades of relentless use of plastics across our economies has led to a seemingly unstoppable flow of plastics into the environment including out into the deep oceans. Largely a result of unsustainable production and consumption patterns and inadequate waste management, the challenge of plastics is now being compounded by the COVID-19 pandemic. Large amounts of plastic waste from personal protective equipment and additional packaging are being discarded directly into the environment.

This assessment provides the strongest scientific argument to date for the urgency of acting, and for collective action to protect and restore our oceans. The assessment details the impacts of marine litter and plastic pollution – from the population level to the sub-cellular – revealing previously unknown aspects of the effects of microplastics on physiology as well as their ecotoxicological effects on ecosystems, wildlife and humans. Drawing on a comprehensive synthesis of the latest findings about the sources, pathways and fate of marine litter and plastic pollution, the assessment highlights the pervasiveness of plastics and microplastics, from the deepest abyssal environments to the most remote oceanic islands, and the extreme pressure being exerted on the planet.

The evidence presented in the assessment paints a comprehensive picture of how every stage in the life cycle of plastics is affecting the oceans, the main risks, and where gaps in our knowledge exist. A major concern is the fate of breakdown products, such as chemical additives and microplastics, many of which are known to be hazardous to both human and wildlife health as well as to ecosystems. There is evidence that microplastics appear in a range of seafoods after being ingested by many different marine organisms. While our understanding of the direct human health effects is still limited, the impacts of hazardous chemicals and microplastics on the physiology of marine organisms is abundantly clear.

The assessment also draws attention to plastics supply and demand, examining the absolute volumes of plastics produced, the slow growth of alternatives, the low levels of recycling, and poor waste management. It looks at critical market failures such as the low price of virgin fossil fuel feedstocks compared to that of recycled materials, disjointed efforts in regard to informal and formal plastic waste management, and lack of consensus on solutions. Importantly, the assessment offers guidance on what it will take to address marine litter and plastic pollution and provides examples of transformative actions and solutions that are fair and just.

The speed at which ocean plastic pollution is capturing public attention is encouraging. It is vital that we use this momentum to focus on opportunities across the life cycle of plastics and from source-to-sea for clean, healthy and resilient oceans, while at the same time contributing to vital Earth system processes, such as climate regulation, and to clean water, healthy ecosystems and biodiversity integrity.

Inger Andersen, Executive Director, UNEP

Key findings

1 The amount of marine litter and plastic pollution has been growing rapidly. Emissions of plastic waste into aquatic ecosystems are projected to nearly triple by 2040 without meaningful action.

The scale and rapidly increasing volume of marine litter and plastic pollution are putting the health of all the world's oceans and seas at risk. Plastics, including microplastics, are now ubiquitous. They are a marker of the Anthropocene, the current geological era, and are becoming part of the Earth's fossil record. Plastics have given their name to a new marine microbial habitat, the "plastisphere".

Despite current initiatives and efforts, the amount of plastics in the oceans has been estimated to be around 75-199 million tons. Estimates of annual global emissions from land-based sources vary according to the approaches used. Under a business-as-usual scenario and in the absence of necessary interventions, the amount of plastic waste entering aquatic ecosystems could nearly triple from some 9-14 million tons per year in 2016 to a projected 23-37 million tons per year by 2040. Using another approach, the amount is projected to approximately double from an estimated 19-23 million tons per year in 2016 to around 53 million tons per year by 2030.

2 Marine litter and plastics present a serious threat to all marine life, while also influencing the climate.

Plastics are the largest, most harmful and most persistent fraction of marine litter, accounting for at least 85 per cent of total marine waste. They cause lethal and sub-lethal effects in whales, seals, turtles, birds and fish as well as invertebrates such as bivalves, plankton, worms and corals. Their effects include entanglement, starvation, drowning, laceration of internal tissues, smothering and deprivation of oxygen and light, physiological stress, and toxicological harm.

Plastics can also alter global carbon cycling through their effect on plankton and primary production in marine, freshwater and terrestrial systems. Marine ecosystems, especially mangroves, seagrasses, corals and salt marshes, play a major role in sequestering carbon. The more damage we do to oceans and coastal areas, the harder it is for these ecosystems to both offset and remain resilient to climate change.

When plastics break down in the marine environment, they transfer microplastics, synthetic and cellulosic microfibres, toxic chemicals, metals and micropollutants into waters and sediments and eventually into marine food chains.

Microplastics act as vectors for pathogenic organisms harmful to humans, fish and aquaculture stocks. When microplastics are ingested, they can cause changes in gene and protein expression, inflammation, disruption of feeding behaviour, decreases in growth, changes in brain development, and reduced filtration and respiration rates. They can alter the reproductive success and survival of marine organisms and compromise the ability of keystone species and ecological "engineers" to build reefs or bioturbated sediments.

3 Human health and well-being are at risk.

Risks to human health and well-being arise from the open burning of plastic waste, ingestion of seafood contaminated with plastics, exposure to pathogenic bacteria transported on plastics, and leaching out of substances of concern to coastal waters. The release of chemicals associated with plastics through leaching into the marine environment is receiving increasing attention, as some of these chemicals are substances of concern or have endocrine disrupting properties.

Microplastics can enter the human body through inhalation and absorption via the skin and accumulate in organs including the placenta. Human uptake of microplastics via seafood is likely to pose serious threats to coastal and indigenous communities where marine species are the main source of food. The links between exposure to chemicals associated with plastics in the marine environment and human health are unclear. However, some of these chemicals are associated with serious health impacts, especially in women.

Marine plastics have a widespread effect on society and human well-being. They may deter people from visiting beaches and shorelines and enjoying the benefits of physical activity, social interaction, and general improvement of both physical and mental health. Mental health may be affected by the knowledge that charismatic marine animals such as sea turtles, whales, dolphins and many seabirds are at risk. These animals have cultural importance for some communities. Images and descriptions of whales and seabirds with their stomachs full of plastic fragments, which are prevalent in mainstream media, can provoke strong emotional impacts.

4 There are hidden costs for the global economy.

Marine litter and plastic pollution present serious threats to the livelihoods of coastal communities as well as to shipping and port operations. The economic costs of marine plastic pollution with respect to its impacts on tourism, fisheries and aquaculture, together with other costs such as those of clean-ups, are estimated to have been at least United States dollars (US$) 6-19 billion globally in 2018. It is projected that by 2040 plastic leakage into the oceans could represent a US$ 100 billion annual financial risk for businesses if governments require them to cover waste management costs at expected volumes and recyclability. By comparison, the global plastic market in 2020 has been estimated at around US$ 580 billion while the monetary value of losses of marine natural capital is estimated to be as high as US$ 2,500 billion per year.

5 Marine litter and plastics are threat multipliers.

The multiple and cascading risks posed by marine litter and plastics make them threat multipliers. They can act together with other stressors, such as climate change and overexploitation of marine resources, to cause far greater damage than if they occurred in isolation. Habitat alterations in key coastal ecosystems caused by the direct impacts of marine litter and plastics affect local food production and damage coastal structures, leading to wide-reaching and unpredictable consequences including loss of resilience to extreme events and climate change in coastal communities. The risks of marine litter and plastics therefore need to be assessed across the wider cumulative risks.

6 The main sources of marine litter and plastic pollution are land-based.

Approximately 7,000 million of the estimated 9,200 million tons of cumulative plastic production between 1950 and 2017 became plastic waste, three-quarters of which was discarded and placed in landfills, became part of uncontrolled and mismanaged waste streams, or was dumped or abandoned in the environment, including at sea. Microplastics can enter the oceans via the breakdown of larger plastic items, leachates from landfill sites, sludge from wastewater treatment systems, airborne particles (e.g. from wear and tear on tyres and other items containing plastic), run-off from agriculture, shipbreaking, and accidental cargo losses at sea. Extreme events such as floods, storms and tsunamis can deliver significant volumes of debris into the oceans from coastal areas and accumulations of litter on riverbanks, along shorelines and in estuaries. With global cumulative plastic production between 1950 and 2050 predicted to reach 34,000 million tons, it is urgent to reduce global plastic production and flows of plastic waste into the environment.

7 The movement and accumulation of marine litter and plastics occur over decades.

The movement of marine litter and plastics on- and offshore is controlled by ocean tides, currents, waves and winds, with floating plastics accumulating in the ocean gyres and sinking items concentrating in the deep sea, river deltas, mud belts and mangroves. There can be significant time intervals between losses on land and accumulation in offshore waters and deep-sea sediments. More than half the plastics found floating in some gyres were produced in the 1990s and earlier.

There are now a growing number of hotspots in which there is potential for long-term, large-scale risks to ecosystem functioning and human health. Major sources include the Mediterranean Sea, where large volumes of marine litter and plastic accumulate due its enclosed nature, presenting risks to millions of people; the Arctic Ocean because of potential damage to its pristine nature and harm to indigenous peoples and iconic species through ingestion of plastics in marine food chains; and the East and Southeast Asian region, where there are significant volumes of uncontrolled waste in proximity to very large human populations with a high dependency on the oceans.

8 Technological advances and the growth of citizen science activities are improving detection of marine litter and plastic pollution, but consistency of measurements remains a challenge.

There have been significant improvements in regard to effective and affordable global observational and surveying systems, as well as the protocols for detecting and quantifying litter and microplastics in physical and biotic samples. However, concerns remain among scientists about sampling biases in the determination of the absolute volumes of microplastics found in different habitats owing to high variability in physical and chemical characteristics and the need for greater consistency among different sampling and observation platforms and instruments. There are currently 15 major operational monitoring programmes linked to marine litter action co-ordination, data collection frameworks, and large-scale data repository and portal initiatives, but the data and information from them are largely unconnected. Alongside these programmes are indicator processes and baseline data collection activities, supported by a growing number of networks, citizen science projects and participatory processes worldwide.

9 Plastic recycling rates are less than 10 per cent and plastics-related greenhouse gas emissions are significant, but some solutions are emerging.

During the past four decades global plastic production has more than quadrupled, with the global plastic market valued at around US$ 580 billion in 2020. At the same time, the estimated global cost of municipal solid waste management is set to increase from US$ 38 billion in 2019 to US$ 61 billion in 2040 under a business-as-usual scenario. The level of greenhouse gas emissions associated with the production, use and disposal of conventional fossil fuel-based plastics is forecast to grow to approximately 2.1 gigatons of carbon dioxide equivalent (GtCO2e) by 2040, or 19 per cent of the global carbon budget. Using another approach, GHG emissions from plastics in 2015 were estimated to be 1.7 GtCO2e and projected to increase to approximately 6.5 GtCO2e by 2050, or 15 per cent of the global carbon budget.

A major problem is the low recycling rate of plastics, which is currently less than 10 per cent. Millions of tons of plastic waste are lost to the environment, or sometimes shipped thousands of kilometres to destinations where it is generally burned or dumped. The estimated annual loss in the value of plastic packaging waste during sorting and processing alone is US$ 80-120 billion. Plastics labelled as biodegradable present another problem, as they may take a number of years to degrade in the oceans and, as litter, can present the same risks as conventional plastics to individuals, biodiversity and ecosystem functioning.

A single-solution strategy will be inadequate to reduce the amount of plastics entering the oceans. Multiple synergistic system interventions are needed upstream and downstream of plastic production and use. Such interventions are already emerging. They include circularity policies, phasing out of unnecessary, avoidable and problematic products and polymers, fiscal instruments such as taxes, fees and charges, deposit-refund schemes, extended producer responsibility schemes, tradeable permits, removal of harmful subsidies, green chemistry innovations for safer alternative polymers and additives, initiatives to change consumer attitudes, and "closing the tap" in regard to virgin plastic production through new service models and ecodesign for product reuse.

10 Progress is being made at all levels, with a potential global instrument in sight.

A growing number of global, regional and national activities are helping to mobilize the global community in order to bring an end to marine litter and plastic pollution.

Cities, municipalities and large firms have been reducing waste flows to landfills; regulatory processes are expanding, driven by growing public pressure; and there has been an upsurge in local activism and local government actions including kerbside collections, plastics recycling and community clean-ups. However, the current situation is a mixture of widely varying business practices and national regulatory and voluntary arrangements.

There are already some international commitments to reduce marine litter and plastic pollution, especially from land-based sources, as well as several applicable international agreements and soft law instruments relating to trade in plastics or to reducing impacts on marine life. However, none of the international policies agreed since 2000 includes a global, binding, specific and measurable target limiting plastic pollution. This has led many governments, as well as business and civil society, to call for a global instrument on marine litter and plastic pollution.

Introduction

Since the 2016 UNEP report *Marine Plastic Debris and Microplastics – Global Lessons and Research to Inspire Action and Guide Policy Change*, substantial new research has been carried out and findings have been presented on marine litter, especially plastics. Research has focused mainly on estimating the volumes of plastics flowing into the oceans; the major sources of marine litter and plastic pollution; the pathways and fate of plastics within the oceans; the impacts of marine litter and plastic pollution, including microplastics and chemical leachates, on marine life, ecosystem functioning and planetary processes; the risks that microplastics pose to human health; and the types of policies, technologies and business solutions that may help to tackle the problems that marine litter and plastic pollution present to society and the economy. [2]

This assessment describes the far-reaching impacts of plastics in our oceans and across the planet. Plastics are a marker of the current geological era, the Anthropocene (Zalasiewicz *et al.* 2016). They have given their name to a new microbial habitat known as the plastisphere (Amaral-Zettler *et al.* 2020; see Glossary). Increased awareness of the negative impacts of microplastics on marine ecosystems and human health has led them to be referred to as a type of "Ocean PM2.5" akin to air pollution (i.e. particulate matter less than 2.5 micrometres [µm] in diameter) (Shu 2018). With cumulative global production of primary plastic between 1950 and 2017 estimated at 9,200 million metric tons and forecast to reach 34 billion metric tons by 2050 (Geyer 2020) (Figure i), the most urgent issues now to be addressed are how to reduce the volume of uncontrolled or mismanaged waste streams going into the oceans (Andrades *et al.* 2018) and how to increase the level of recycling. Of the 7 billion tons of plastic waste generated globally so far, less than 10 per cent has been recycled (Geyer 2020).

Today cumulative annual economic losses as a result of damage to maritime industries, including the costs of clean-ups, are estimated to total some US$ 6-19 billion (Deloitte 2019). Since this estimate does not include the costs of degradation of ecosystem goods and services due to marine litter (Beaumont *et al.* 2019), it is likely to significantly underestimate the total economic losses. The combination of cheap fossil fuel feedstocks and poor waste infrastructure and recycling has led to projections that by 2040 the expected mass of plastic leakage into the oceans could represent a US$ 100 billion annual financial risk for businesses if governments require them to cover waste management costs (The Pew Charitable Trusts and SYSTEMIQ 2020). These figures point to significant market failures and underline the need for urgent action.

The assessment sets out to address four key questions to help guide future actions:

- What can new research and evidence tell us about the environmental and human health impacts of marine litter and plastic pollution?
- What is the latest understanding of the sources, pathways, behaviour and fate of marine litter, especially plastics?
- What are the most effective field, laboratory and modelling approaches for monitoring and measurement of the sources, pathways, behaviour and fate of marine litter and plastic pollution?
- What ongoing responses and actions, environmental technologies and business solutions exist to tackle this urgent global problem?

Section 1 looks at the latest evidence concerning the impacts of exposure to marine litter and plastic pollution, including the effects of chemical leachates on marine life and humans. Ingestion, physical entanglement, smothering, and the transport of pathogens in biofilms are causeing a range of lethal and non-lethal effects in marine organisms, including physiological disturbances, disease, changes in gene expression, alterations of behaviour, and shifts in species assemblages and biodiversity. These in turn, have impacts on ecosystems, leading to a wide range of social and economic consequences such as loss of revenue from natural resources and damage to maritime industries and coastal infrastructure (Figure ii). At the same time, there are many significant knowledge gaps with respect to potential long-term climate effects, health impacts on marine organisms and humans (e.g. via consumption of seafood) and the full economic and social costs of the loss of ecosystem services.

Section 2 provides a recent estimate of the volume of plastics being produced. It also describes the major sources of marine litter and plastic pollution and the pathways along which litter, especially plastics and microplastics, flows into the marine environment and accumulates in different habitats. Nearly 85 per cent of plastic packaging waste goes to landfill or ends up as unregulated or uncollected waste, with a high likelihood of entering the oceans (Andrades *et al.* 2016). The volume of plastic waste from both land-based sources and sea-based activities continues to grow, while personal protective equipment and other plastic items such as those widely used and quickly disposed of during the COVID-19 pandemic are of increasing concern.

The pathways by which marine litter and plastic pollution enter and flow through the marine environment, and their distribution and fate, have been studied in much greater detail over the past five years than previously (Figure iii). However, the absolute volumes of plastics in different marine zones and habitats remain poorly known. This is mainly due to poor sampling coverage and the lack of standardized sampling protocols. Current global estimates have therefore been determined primarily through modelling based on proxies, such as population densities, rather than on direct measurements (Harris *et al.* 2021). In this section three regional hotspots are examined in detail: the

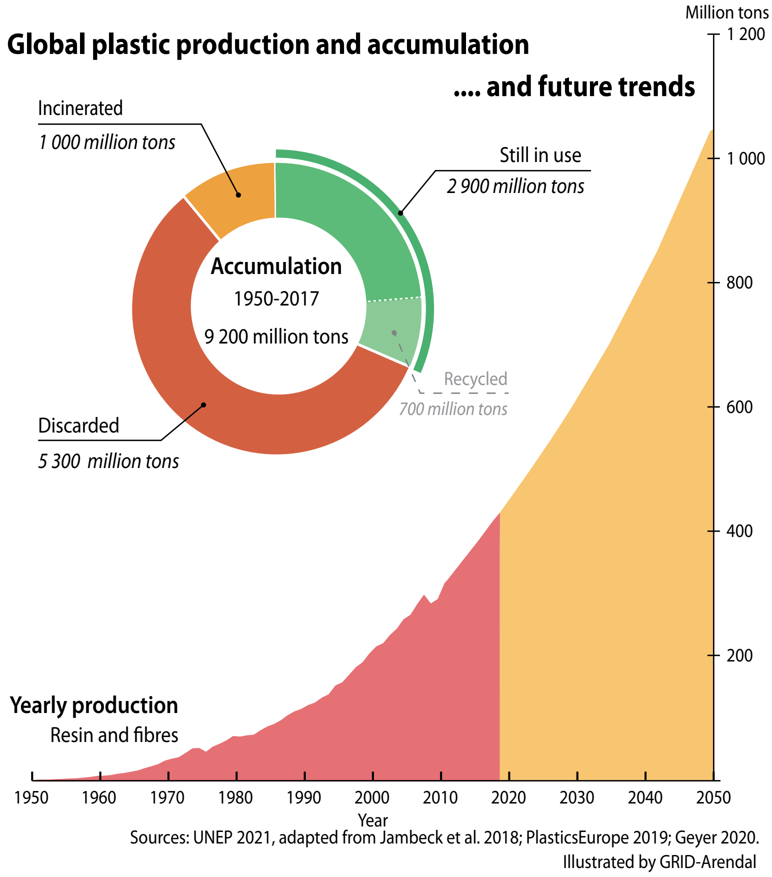

Figure i: Global plastic production, accumulation and future trends

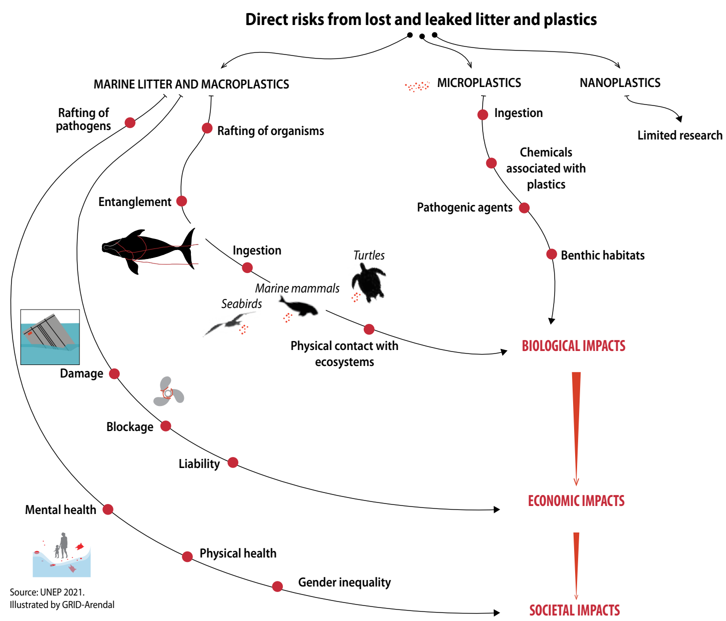

Figure ii: Direct risks and impacts of marine litter and plastics

Mediterranean Sea because of its enclosed nature and proximity to millions of people; the Arctic Ocean because of its pristine nature and impacts on indigenous peoples; and the East Asia and Association of Southeast Asian Nations (ASEAN) region because of its extensive coastline in proximity to very large populations which are highly dependent on the marine environment for survival but often have insufficient waste management systems.

Section 3 covers the latest improvements in and modifications to monitoring methods and surveys of litter and macroplastics in riverine, shoreline, coastal and offshore environments. In the past five years there have been significant efforts to develop effective global monitoring programmes. Currently there are 15 major operational monitoring programmes in different geographical ranges, linked to three types of activity: marine litter action coordination, data collection frameworks, and large-scale data repository and portal initiatives. To date, the data and information being collected remain largely unconnected and fragmented.

Digital technologies, satellites, aircraft and drones, combined with shipborne sensors, samplers and autonomous platforms (e.g. floats, gliders, benthic landers and crawlers), are opening up the possibility for affordable global monitoring programmes to track and determine the densities of marine litter and macroplastics from coastal areas out into the open ocean and into the hadal depths. Data from such platforms are especially important for determining volumes in sediments and riverine discharge over large areas, particularly when used with ground calibration. The main challenge now is their intercalibration, so that the data can be used for modelling and prediction of the distribution and quantities of marine litter and plastic pollution in different habitats.

There are still widespread concerns among scientists about the sampling biases of different field and laboratory techniques for identifying and determining the volume of microplastics in the environment. Intrinsic difficulties exist due to the high variability in the size, shape, colour, and degree of degradation of plastics. Without significant improvements in quality assurance and control protocols for sampling and analytical techniques, it will remain difficult to demonstrate the reliability and repeatability of published results.

Alongside large-scale monitoring programmes, there are indicator processes and baseline data collection activities at specific locations. The growing number of networks, citizen science projects and participatory processes involved in measuring and tackling marine litter and plastic pollution are yielding results that can support local decision-making. However, in most countries there is no consistent data collection approach suitable for national reporting.

Streamlining methodologies, data flows and indicator sets to establish baselines is now very important, especially in the case of transboundary waters. There is also a need to facilitate joint analyses, unified definitions, standards and formats, and well-developed infrastructures for data flow, storage and sharing.

Major sources and sinks of microplastics and marine litter

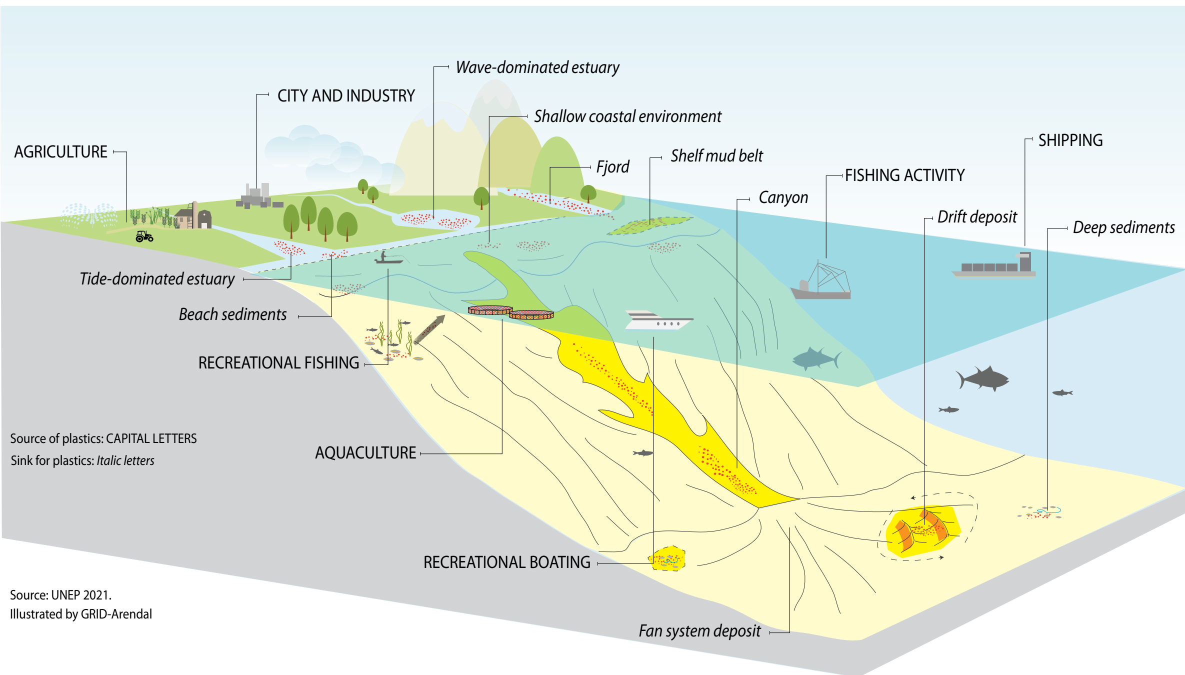

Figure iii: Major sources and sinks of microplastics and marine litter

In addition, Section 3 reviews current standards for biodegradability and traceability in plastics through labelling schemes. There are very few verification schemes for the manufacturing and processing of plastics or recyclates, and none that require listing constituent polymers or chemical additives in consumer products or provide traceability. Those schemes that do exist refer mainly to the recycling and biodegradability of plastics under controlled conditions. However, the specific conditions set out by standards-setting bodies, such as industrial composting requirements for biodegradation, may not be met outside highly regulated waste markets. Lack of information about recyclates is also a barrier to increasing recycling rates and the development of markets.

Section 4 looks at ongoing responses, actions and potential solutions to tackling marine litter, especially plastics. The growing number of global, regional, national and local responses and actions are raising awareness and, in some cases, helping to reduce flows of marine litter and plastic pollution into the marine environment. However, they are unevenly deployed and are fragmented. Connecting all the different responses and actions of governments, business and citizens is now critically important.

On the global scale there are numerous international agreements, conventions and organizations related to marine litter and pollution. They include the Basel, Rotterdam and Stockholm Conventions, the International Maritime Organization and the Conference of the Parties of the London Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter and its Protocol, and the Committee on Fisheries of the Food and Agriculture Organization of the United Nations. Parties to the Basel Convention are now required to control transboundary movements of the plastic waste covered under the procedures established by the Convention. The amendments to the Basel Convention concerning plastics do not imply a ban on the import, transit or export of plastic waste, but rather a clarification of when and how the Convention applies to such waste. World Trade Organization members can also take action to support international efforts to reduce and phase out plastic pollution, align trade policies, and advance dialogue and actions at ministerial level to strengthen the multilateral trading system in order to reduce plastic pollution.

Moreover, a range of legislative responses are showing success, with bans, taxes, improved waste operations, economic incentives, extended producer responsibility, regional conventions, marine litter action plans, education initiatives and public awareness campaigns being widely implemented. Shifts in public attitudes and greater levels of public concern, awareness and activism regarding the use, reuse or replacement of plastics with alternatives are also providing an impetus for action.

Many environmentally sound technologies and innovations are appearing in the plastics and waste sector (Ellen MacArthur Foundation 2020). A number of them are aimed at improving the labelling and traceability of plastic products throughout their life cycle, for example using blockchain technologies to help reduce the loss of materials along supply chains. Recently there has been a proliferation of business-led joint-industry initiatives and partnerships focusing on packaging, the waste hierarchy and circularity; the development of biodegradable plastics and alternative materials; and the application of ecodesign (UNEP 2019; Ellen MacArthur Foundation 2020; WWF, the Ellen MacArthur Foundation and BC [Boston Consulting] 2020; Ellen MacArthur Foundation 2021; IRP [International Research Panel] 2021). Some global brand companies have also put in place plans to retool and reconfigure their supply chains to align them with national-level schemes, shifting production away from fossil fuel-based plastics towards recycled materials.

The attention given to plastics has generated large amounts of research. However the experts involved in this assessment have identified a number of critical research areas that still need urgent attention. They include quantification of the volumes of different plastic fractions from key sources and their fate in different marine habitats; quantification of the damage and economic costs of marine litter and plastics to maritime industries and in terms of ecosystem and human health; improvements to recycling technologies and standards for plastic recyclates; the development of circularity and ecodesign for plastic products, including the use of alternative materials and biodegradability outside industrial conditions; improved risk assessment frameworks; and a deeper understanding of the impacts of marine litter and plastics on societal norms, attitudes and behaviour.

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SECTION 1

ENVIRONMENTAL, HEALTH, ECONOMIC AND SOCIAL IMPACTS AND RISKS

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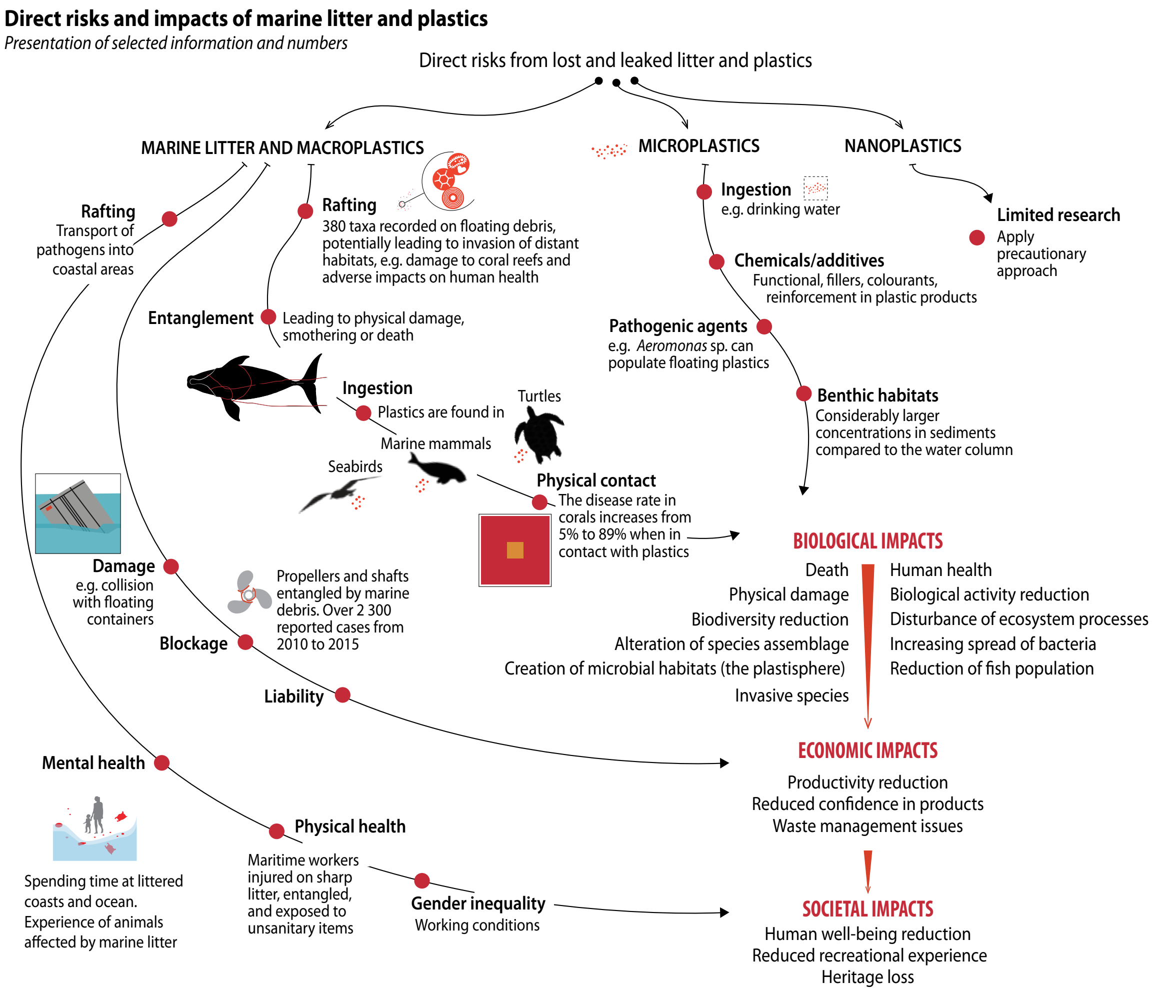
1.1 Evidence of biological and ecological impacts

Aquatic organisms are continuously exposed to litter and plastic pollution. The largest, most persistent fractions of marine litter are synthetic polymers and thermosets, known collectively as plastics; these account for at least 85 per cent of total marine waste (Law 2017; Agamuthu *et al.* 2019). In freshwater systems the threat of physical harm from litter and macroplastic debris remains relatively under-researched (Blettler *et al.* 2019). By contrast, there have been more than 100,000 marine studies on the lethal and non-lethal effects of litter and plastics at every level of the food web, including algae, zooplankton, crustacea and invertebrates, fish, birds, turtles and mammals (e.g. Boerger *et al.* 2010; Fossi *et al.* 2012; Fossi *et al.* 2014; Avio *et al.* 2015; Sbrana *et al.* 2017; Fossi *et al.* 2018; Lavers *et al.* 2018; Waite *et al.* 2018; Akhbarizadeh *et al.* 2019; Li *et al.* 2019; Zhu, C. *et al.* 2019; Zhu, L. *et al.* 2019).

The main effects observed come from entanglement, smothering, rafting of pathogenic organisms (Aliani and Molcard 2003; Woods *et al.* 2019), ingestion of plastic fragments (Anbumani and Kakkar 2018) and exposure to plastic-associated chemicals (Alimba and Faggio 2019). The impacts depend on the type, size and habitat (Rochman *et al.* 2016b; Lusher *et al.* 2017a; Bucci *et al.* 2019; Windsor *et al.* 2019) (Figure 1). In the following subsections the direct risks, impacts and effects of different sized plastics on marine life, human health, society and the economy are looked at in more detail.

1.1.1 Impacts of marine debris and macroplastics on marine life

Ongoing monitoring results continue to show that physical collisions with macro-sized plastics by marine mammals, fish, birds, reptiles and plants are a direct source of fatalities (Alomar and Deudero 2017; Alomar *et al.* 2017; Franco-Trecu *et al.* 2017; Reinert *et al.* 2017; Fossi *et al.* 2018; Thiel *et al.* 2018; Bucci *et al.* 2019; Woods *et al.* 2019). It is difficult to determine and quantify the causal links between mortality and ingestion of large plastic fragments, but there are growing numbers of investigations to better understand the origin of the plastics and the causes of death (Unger *et al.* 2016). What is widely reported is the



Sources: UNEP 2021, adapted from Aliani and Molcard 2003; Boerger et al. 2010; Fossi et al. 2012; Fossi et al. 2014; Goldstein et al. 2014; Avio et al. 2015; Kiessling et al. 2015; Santos et al. 2015; Rech et al. 2016; Rochman et al. 2016b; Hong et al. 2017b; Lusher et al. 2017a; Sbrana et al. 2017; Steer et al. 2017; Anbumani and Kakkar 2018; Godoy and Stockin 2018; Duncan et al. 2018a; Duncan et al. 2018b Fossi et al. 2018; Lamb et al. 2018; Lavers et al. 2018; Schymanski et al. 2018; Waite et al. 2018; Akhbarizadeh et al. 2019; Alimba and Faggio 2019; Battisti et al. 2019; Besseling et al. 2019; Bucci et al. 2019; Koelmans et al. 2019; Li et al. 2019; Windsor et al. 2019; Woods et al. 2019; Zhu, C. et al. 2019; Zhu, L. et al. 2019; Amaral-Zettler et al. 2020; Harris 2020; Landrigan et al. 2020; Prata et al. 2020a.
Illustrated by GRID-Arendal

Figure 1: Direct risks and impacts of marine litter and plastics

presence of plastic fragments in the guts and tissues of a wide range of marine species at all stages of their life cycle (Lusher *et al.* 2017a; Steer *et al.* 2017), including those directly vital to food provision such as fish and shellfish (Rochman *et al.* 2016a; Law 2017; Qiao *et al.* 2019; Rochman *et al.* 2019), either directly from the environment or via plastic-contaminated prey (Setälä *et al.* 2014; Prata *et al.* 2020a). Macroplastic debris has been found in the digestive system of aquatic organisms, including all marine turtle species sampled and nearly half of all surveyed seabird and marine mammal species (Poppi *et al.* 2012; Kühn *et al.* 2015; Provencher *et al.* 2015; Duncan *et al.* 2018a; Duncan *et al.* 2018b; Godoy and Stockin 2018; Verlis *et al.* 2018; Battisti *et al.* 2019).

Floating macroplastics remain a major concern in the conservation of sea turtles, as their visual feeding strategies mean that they select structures analogous to those of jellyfish, such as soft floating plastics. Their backward facing oesophageal papillae also inhibit regurgitation and facilitate particle accumulation in the gut (Schuyler *et al.* 2014; Vegter *et al.* 2014). Plastic bottle fragments, fishing lines and paint chips are commonly encountered in the guts of sea turtles (Wedemeyer-Strombel *et al.* 2015; Pham *et al.* 2017; Clukey *et al.* 2018). In Brazil 70 per cent of juvenile turtles analysed showed plastic ingestion (Santos *et al.* 2015). In the North Pacific Ocean and the Mediterranean Sea more than 80 per cent of turtles were shown to have ingested some form of debris (Wedemeyer-Strombel *et al.* 2015; Matiddi *et al.* 2017; Duncan *et al.* 2018a; Duncan *et al.* 2018b). In New Zealand 63 per cent of endangered green turtles, *Chelonia mydas*, had ingested synthetic debris (Godoy and Stockin 2018).

Non-lethal effects include lacerations from the particles as they translocate across the cell membrane into the circulatory, lymphatic, respiratory and/or other biological systems (Browne *et al.* 2008; Hämer *et al.* 2014; Brennecke *et al.* 2015; Landrigan *et al.* 2020; Vethaak and Legler 2021), suffocation and starvation (Wright *et al.* 2013a; Wright *et al.* 2013b; Adimey *et al.* 2014; Anbumani and Kakkar 2018; Sun *et al.* 2019), physiological disturbances (Au *et al.* 2015; Anbumani and Kakkar 2018), changes in gene expression (Rochman *et al.* 2014b) and alterations in behaviour (Green *et al.* 2017). Evidence of these effects has been documented for algae (e.g. Carson *et al.* 2013), zooplankton (e.g. Desforges *et al.* 2015), and consumers such as fish (e.g. Lusher *et al.* 2017a; McNeish *et al.* 2018; Arias *et al.* 2019), turtles (e.g. Duncan *et al.* 2018a; Duncan *et al.* 2018b), birds (e.g. Wilcox *et al.* 2015; Holland *et al.* 2016; Reynolds and Ryan 2018; Battisti *et al.* 2019), whales (e.g. Nelms *et al.* 2019a) and seals (e.g. Hallanger and Gabrielsen 2018; Donohue *et al.* 2019), although an absence of plastics in seal stomachs in the Arctic and Antarctic was reported by Bourdages *et al.* (2020) and Garcia-Garin *et al.* (2020), respectively.

1.1.2 Effects of microplastics in marine life

It has been recognized for more than a decade that microplastics can transfer a range of toxic chemicals, metals and micropollutants into open surface waters, where they can be ingested by a wide range of fauna (Arthur et al, 2009; Ashton *et al.* 2010; Mattsson *et al.* 2015; Haward 2018; Karlsson *et al.* 2018; UNEP 2018a). Over the past decade a broad range of laboratory and experimental studies have complemented field observations in the hope of achieving a better understanding of the effects of micro- and/or nanoplastics on different organisms (e.g. corals, birds, fish and mammals). However, monitoring microplastics remains challenging both in the environment and under laboratory conditions. Plastic particles are often naturally or experimentally co-contaminated with diverse chemical pollutants (Setälä *et al.* 2019; Jacob *et al.* 2020). It has therefore been concluded that more innovative, robust and scientifically sound experiments in the field are needed. For example, the nets commonly used to collect microplastics, and which have mesh sizes ranging from >500 µm to >200 µm, undersample microplastics and lead to lower estimates compared to studies using finer mesh nets of 0.45 µm. Previous reports on the quantities of microplastics in the marine environment are likely to have been underestimated (UNEP 2016a; Barrows *et al.* 2017; Barrows *et al.* 2018a; Green *et al.* 2018; Whitaker *et al.* 2019; Lindeque *et al.* 2020) and open to misinterpretation. Sample location is also important; for example, Ryan *et al.* (2019) collected microfibres (Box 1) in three ocean basins and found that fibre densities were 2.5 times greater at the sea surface than 5 metres subsurface. Properly estimating quantities of nanoplastics (Box 2) presents an even greater challenge.

Despite these difficulties, the body of literature looking at the effects of microplastics continues to increase (Lusher *et al.* 2017a; Anbumani *et al.* 2018; Arthur *et al.* 2019; Bradney *et al.* 2019; Maes *et al.* 2020; Peng, L. *et al.* 2020; Xu *et al.* 2020). However, the effects of microplastics and the causal mechanisms of harm in marine biota are unevenly studied and there are many discrepancies among reports (European Union 2019a; SAPEA [Science Advice for Policy from European Academies] 2019; de Ruijter *et al.* 2020; Lindeque *et al.* 2020; Peng, L. *et al.* 2020; Xu *et al.* 2020). Based on an assessment of more than 100 studies, de Ruijter *et al.* (2020) performed a weight of evidence analysis for causal mechanisms of harm in field and laboratory settings. They found that only three mechanisms could be considered to have been "demonstrated": inhibition of food assimilation and/or decreased nutritional value of food, internal physical damage, and external physical damage. The rest had to be discarded because of poor quality assurance/quality control of studies, or authors speculating rather than demonstrating mechanisms. Their recommendation is that risk assessment should address these mechanisms with higher priority.

Physically, just as with macroplastics, microplastics may lacerate the gut or cause an animal to feel full, and there is evidence (albeit using very high doses) that very small ingested microplastics may cross the gut lining and accumulate in tissues (Browne *et al.* 2008; Rosenkranz *et al.* 2009; Deng *et al.* 2017; Schür *et al.* 2019) where they can potentially have deleterious effects such as inflammation (Deng *et al.* 2017). Several effects appear to be exacerbated when organisms are exposed to plastics with sorbed contaminants (Lithner *et al.* 2012; Browne *et al.* 2013; Rochman *et al.* 2014a; Martínez-Gómez *et al.* 2017; Rochman *et al.* 2019). Variations in study findings are likely to

Box 1: Fibres and microfibres

The volume of cellulosic (natural and regenerated) and synthetic fibres entering the oceans every year has been estimated to range between 8,000 and 520,000 metric tons (Boucher and Friot 2017; Belzagui *et al.* 2019). A global compilation of datasets from 916 seawater samples collected in six ocean basins showed that although synthetic polymers currently account for two-thirds of global fibre production, oceanic fibres are mainly composed of natural polymers. Infrared characterization of ~2,000 fibres revealed that only 8.2 per cent of oceanic fibres are synthetic, with most being cellulosic (79.5 per cent) or of animal origin (12.3 per cent) (Suaria *et al.* 2020). This agrees with studies that report cellulosic fibres accounting for 60-80 per cent of all fibres in sea floor sediments (Woodall *et al.* 2015; Sanchez-Vidal *et al.* 2018), marine organisms (Remy *et al.* 2015; Taylor *et al.* 2016), wastewater (Primpke *et al.* 2019), freshwater (Dris *et al.* 2017; Miller *et al.* 2017), ice cores (Obbard *et al.* 2014) and airborne fibre populations (Dris *et al.* 2017; Stanton *et al.* 2019b).

Microfibres, the breakdown product of fibres, are ubiquitous in water, soil and air (Avio *et al.* 2017; Windsor *et al.* 2018; Zambrano *et al.* 2019). In the marine environment they are found suspended in the water column (Bagaev *et al.* 2017; Song *et al.* 2018), on the sea floor (Woodall *et al.* 2014; Sanchez-Vidal *et al.* 2018) and throughout marine ecosystems, where they are ingested by a wide range of biota (Taylor *et al.* 2016; Welden and Cowie 2016; Henry *et al.* 2019; Ronda *et al.* 2019; Suaria *et al.* 2020).

Synthetic microfibres are a distinct sub-category of the microplastics family, spanning a wide range of sizes (roughly 3 to 30 micrometres [μm] in width) and originating mainly from clothing and textiles as well as from uses in transportation such as wear and tear on tyres. They comprise various polymer materials, including synthetics, semi-synthetics and natural products (e.g. polyester, nylon, spandex, PLA-polylactic acid, cotton, hemp and silk). Gavigan *et al.* (2020) estimate that 5.6 metric tons of synthetic microfibres were emitted into the environment from clothes washing between 1950 and 2016, with a 12.9 per cent growth rate during the past decade. This figure is small compared to the total volume of plastics in the ocean, but is likely to be an underestimate given the poor understanding of the quantities involved in the emission pathways from clothing production, use and washing, along with emission and retention rates during washing, wastewater treatment and sludge management.

Microfibres were previously considered the most common type of microplastics found in samples (Browne *et al.* 2011; Rochman *et al.* 2015; Obbard 2018; Maximenko *et al.* 2019). However, in hundreds of studies cellulosic fibres (natural and regenerated) were included in the synthetic realm, inflating "microplastic" counts in both environmental matrices and

organisms (Wesch *et al.* 2016; Cesa *et al.* 2017). This error resulted either from the assumption that all coloured fibres were synthetic (Remy *et al.* 2015) or the assumption that man-made cellulosic fibres could be considered synthetic and included in microplastic counts because they are extruded and processed industrially (Obbard *et al.* 2014; Woodall *et al.* 2014). These errors were compounded by a previous large-scale investigation which reported that 69 per cent of marine fibres were synthetic (Barrows *et al.* 2018a); however, this study was based on the characterization of a small number of fibres using infrared techniques (Comnea-Stancu *et al.* 2017; Käppler *et al.* 2018). Visual and chemometric methods are being developed (Cai *et al.* 2019), but the presence of dyes, oxidation and microbial degradation can alter cellulose absorption bands, making it very difficult to distinguish natural and man-made cellulose, especially when dealing with environmentally degraded polymers (Stark 2019).

Different microfibres have different surface properties, making them variably capable of adsorbing materials from the surrounding environment and being modified by the addition of chemicals that convey specific properties such as UV protection, water repellence and colours. Because microfibres are denser than seawater, they are likely to accumulate on the ocean floor and slowly degrade over tens if not hundreds of years (Bejgarn *et al.* 2015; Andrady 2017) while being ingested by deep sea organisms (Taylor *et al.* 2016; Barrows *et al.* 2018a). All these factors complicate the assessment of their toxicity and health hazards (Botterell *et al.* 2019; Royer and Deheyn 2019).

be the consequence of different polymer types, associated chemicals, the study species, the doses used during testing, and the duration of the exposure (Brandon *et al.* 2016; Lenz and Labrenz 2018; Bucci *et al.* 2019).

Under laboratory conditions, microplastics have been shown to cause a variety of biological effects in crustaceans, molluscs and polychaetes (Anbumani and Kakkar 2018; Silva *et al.* 2020), including changes in gene and protein expression (Paul-Pont *et al.* 2016; Green *et al.* 2019), inflammation (von Moos 2012), disruption of feeding behaviour (Cole *et al.* 2015), decreases in growth (Au *et al.* 2015), decrease in reproductive success (Au *et al.* 2015; Sussarellu *et al.* 2016; Silva *et al.* 2020), changes in larval development (Nobre *et al.* 2015), reduced filtration and respiration rates (Paul-Pont *et al.* 2016), and decreased survival (Au *et al.* 2015; Cui *et al.* 2017). However, there are also studies in which no effects were detected (Hämer *et al.* 2014; Batel *et al.* 2016; Espinosa *et al.* 2018; Roman *et al.* 2019).

Jacob *et al.* (2020) reviewed a total of 782 direct markers of disease progression, known as biological endpoints (see Glossary), in 46 studies; nearly one-third of the markers were significantly affected by exposure to virgin microplastics. More effects were observed for small plastic particles ≤20 µm in size; for fish these effects included changes in behaviour, sensory and neuromuscular functions, activity and motion, shoaling,

Box 2: Nanoplastics

Nanoplastics are a sub-category of microplastics intentionally used in many products, for example textiles (Patra and Gouda 2013; Radetić 2013) and cosmetics. They are found throughout the oceans, including in large ocean gyres (ter Halle *et al.* 2016). The definition of nanoplastics is still under discussion (Gigault *et al.* 2018); some authors use ≤1 µm as the definition of size (da Costa *et al.* 2016) while others use 1 µm to 500 µm (Rios Mendoza and Balcer 2019) (see Glossary). Gigault *et al.* (2018) define nanoplastics as "particles unintentionally produced (i.e. from the degradation and manufacturing of plastic objects) and presenting a colloidal behaviour, within the size range from 1 to 1,000 nm [nanometres]".

Analogies or extrapolations between nanoplastics and other nanomaterials should be treated with caution due to different production pathways and physical and chemical properties. According to the International Organization for Standardization (ISO), a manufactured nanomaterial is "intentionally produced for commercial purposes to have specific properties or specific composition" (ISO/TS 80004e1:2015). Nanoplastics are highly polydisperse in physical properties and heterogeneous in composition (Gigault *et al.* 2016; Lambert and Wagner 2016; ter Halle *et al.* 2016). Because they are produced unintentionally from the degradation of microscale plastic litter (e.g. secondary nanoplastics from biodegradable microplastics, González-Pleiter *et al.* 2019), it is highly probable that nanoplastics will form heteromorphic aggregates with other natural and anthropogenic materials (Hüffer *et al.* 2017). In this sense the colloidal behaviour of nanoplastics is relevant.

The adverse health effects of nanoplastics in organisms, including humans, are largely unknown (Barría *et al.* 2020; Landrigan *et al.* 2020; UNEP 2020e). There is a significant mismatch between laboratory studies and environmental concentrations. For example, laboratory studies demonstrate the ability of microplastics to degrade into nanoplastics, but in the field scientists remain unable to quantify and characterize them. Some studies suggest that nanoplastics may be more hazardous than microplastics (Anbumani and Kakkar 2018; Bellingeri *et al.* 2019; Peng, L. *et al.* 2020; Rubio *et al.* 2020).This is because it is likely their small size will make them more likely to translocate beyond the gut and their high surface-to-volume ratio enables them to be efficient vectors for chemical contaminants. The available data show some evidence that nanoparticles, once ingested, can pass from the intestines into an animal's circulatory system and generate an immune response. In one laboratory experiment nanoparticles were able to pass into the food web, from algae to zooplankton and then to fish, where they entered the brain and incited behavioural disorder (Peng, L. *et al.* 2020).

There are longstanding data showing a high potential for bioaccumulation and biomagnification along parts of the marine food chain (Suedel *et al.* 1994; Akhbarizadeh *et al.* 2019; Saley 2019). However, the lack of standardized methodology for nanoplastics detection makes demonstrating that these processes occur a challenge. Studies have shown that different phyla react differently, so that it is difficult at this stage to predict the ecological risks of nanoplastics to the marine environment (Besseling *et al.* 2019).

More recently, the potential risk of nanoplastics in seafood has been raised. Compared to microplastics, nanoplastics have increased mobility in the tissues of living organisms and their larger surface to volume ratio increases the potential concentration of harmful chemicals they can adsorb. Nevertheless, as indicated in the recent review by Ferreira *et al.* (2019), the marine distribution and impact of plastic nanoparticles are relatively unknown. This presents an unknown risk to marine organisms, as well as to humans who consume seafood. As the most recent reviews stress, there is an urgent need for further research and experimental data to better understand the different processes and mechanisms that may affect marine life and human health (Rubio *et al.* 2020).

feeding, boldness and exploration, and vision. In contrast, aggressivity markers were not affected. Nervous system markers were specifically affected by small particles (24-45 µm in size), but the underlying mechanisms remain unclear. The review also highlighted the impacts of virgin microplastics on brain development and structure, and neurotoxicological indicators such as acetylcholinesterase activity, indicating that translocation of small plastic particles into the brain could directly initiate brain disorders. However, the accuracy of the methods used (e.g. fluorescence and brain dissection) was debated. In addition, the review found that particles >500 nanometres [nm] in size were very unlikely to pass the blood-brain barrier and that instead the underlying mechanisms responsible for their neurotoxicity came from altered immune responses and metabolism, culminating in impaired brain functions and behaviour. Jacob *et al.* (2020) concluded that the toxicity of virgin microplastics to fish should be more systematically evaluated, using rigorous laboratory-based methods, in order to obtain a better understanding of the underlying mechanisms of microplastic toxicity in marine organisms.

Microplastics have been demonstrated to sorb persistent organic pollutants (POPs) such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and dichloro-diphenyl-trichloroethane (DDT), as well as trace metals (e.g. copper or lead) (Anbumani and Kakkar 2018; Bradney *et al.* 2019; Camacho *et al.* 2019; Guo and Wang 2019; Fred-Ahmadu *et al.* 2020; Pozo *et al.* 2020). This has been shown experimentally with virgin plastic pellets in seawater (Bakir *et al.* 2016; Gallo *et al.* 2018; Guo and Wang 2019a). Natural sediments and organic matter also have the capacity to adsorb hydrophobic organic chemicals (Koelmans *et al.* 2016; Prata *et al.* 2020a).

Although the mechanisms of accumulation and concentration of chemicals, and how chemicals adsorb and desorb to and from plastics, are well researched, these concepts are not typically included in tests of microplastic biological effects, where many factors other than ingestion (Bakir *et al.* 2016; Herzke *et al.* 2016; Hermsen *et al.* 2018), such as the hydrophobicity of the pollutant, type of polymer, age of the plastic, water, temperature, pressure, presence of biofouling on the plastic surface, and salinity, all matter (Rochman 2015; Bakir *et al.* 2016; Anbumani and Kakker 2018; Peng, L. *et al.* 2020). In the case of nanoplastics, Koelmans *et al.* (2014, 2016) and Yu *et al.* (2019) concluded that their large surface area could lead to higher concentrations of organic toxic chemicals or heavy metals being retained compared to microplastics, which could lead to the risk of gradients building up within an organism's tissues (Boxes 2 and 3).

Primary or virgin microplastics may be considered harmful because of their potential to release chemicals into the environment (Ashton *et al.* 2010; Mattsson *et al.* 2015; Haward 2018; UNEP 2018a; Santana *et al.* 2020). Jacob *et al.* (2020) point out that it is their potential to act as vectors of contaminants that has been studied, rather than the toxicity of the microplastics themselves. UNEP (2019) notes that current evidence suggests ingestion of microplastics does not significantly enhance exposure/bioaccumulation of organic pollutants (including POPs) compared to other types of particles present in the environment or other exposure pathways (e.g. water, diet) in general. Jacob *et al.* (2020) recommend more in-depth research on the effects of size, concentration and charge, using environmentally relevant exposures, to determine toxicological tipping points and dose or threshold responses. However, they also conclude that even in worst case scenarios of plastic pollution, microplastics will likely remain a minor fraction of the other microparticles naturally present in water and sediment. Thus, researchers should consider using the same concentrations of natural particles as controls to address the induced effects of microplastics. They also recommend publishing positive, negative and neutral effects of virgin microplastics together to help prevent potential bias during meta-analysis and to enhance general understanding of the impacts and non-impacts of virgin microplastics on fish and other marine organisms.

1.1.3 Impacts on habitats, assemblages and ecosystem function

Plastic debris, whether flexible or rigid, can alter the structure and composition of macrofaunal, microfaunal and bacterial assemblages (e.g. Katsanevakis *et al.* 2007; Goldstein *et al.* 2012; Green *et al.* 2015; Carvalho-Souza *et al.* 2018; GESAMP 2019; Peng, G. *et al.* 2020). Flexible plastic items such as plastic bags also affect key ecosystem processes by blocking gas exchange and decreasing the flux of inorganic nutrients from sediment, thereby decreasing primary productivity (Green *et al.* 2015). Furthermore, abandoned, lost or otherwise discarded fishing gear (ALDFG) like nets, ropes, cages and nylon lines can damage key habitat-forming marine organisms such as corals and seagrasses through tissue abrasion and smothering (Ballesteros *et al.* 2018), sometimes significantly reducing their extent (Richards and Beger 2011; Carvalho-Souza *et al.* 2018).

Once onshore, debris and macroplastics can interact with marine biota. They can be ingested by a wide variety of organisms (e.g. van Cauwenberghe *et al.* 2015; Waite *et al.* 2018; Li *et al.* 2019, Peng, L. *et al.* 2019), provide additional substrates for organisms to disperse (e.g. Majer *et al.* 2012; Kirstein *et al.* 2016) and have wider ecological impacts. For example, the reduced ability of carbonate reefs and primary producers to absorb carbon due to uptake of microplastics can affect their functioning (Carvalho-Souza *et al.* 2018; GESAMP 2019) and potentially have a knock-on effect with respect to global warming (Center for International Environmental Law 2019). Lamb *et al.* (2018) demonstrated a link between macroplastic pollution and an increased likelihood of coral disease. The likelihood of disease in corals rose from 4 per cent to 89 per cent when they had been in contact with plastics, but the mechanism is unknown. Macroplastic debris can cause direct damage to the tissue of corals, opening them up to pathogenic agents such as ciliates (Sweet and Bythell 2015; Sweet and Brown 2016; Sweet and Séré 2016; Carvalho-Souza *et al.* 2018).

Species assemblages can also be altered through the introduction of alien species that have been transported by plastics, and as a result of the loss of foundational species such

Box 3: Chemicals associated with marine litter and plastics

The chemicals found in plastics are either added during the production process (including additives such as flame retardants, plasticizers, antioxidants, UV stabilizers and pigments, a number of which may be substances of concern) or added unintentionally (e.g. when the composition of the input material is not known exactly or chemicals are accumulated from the environment) (Hong *et al.* 2017a; Groh *et al.* 2019; Guo and Wang 2019). Certain chemicals found in plastics may have been intentionally added to achieve desirable properties, but plastics can also include solvents or substances not intentionally added such as impurities from the packaging of pesticides or cleaning agents. For several decades it has been understood that plastics transfer chemicals to wildlife; they may be directly released from plastics when they reach the intestinal tissues of marine species, or leach into the marine environment as the plastics weather (Pettit *et al.* 1981). The rate of these transfers will depend upon factors such as the nature and strength of the bond between additive and polymer, pore diameter, the molecular weight of the additive, temperature, pressure and biofouling (De Frond *et al.* 2019).

Intentional additives in plastics, such as plasticizers and flame retardants, are relevant in elevated exposure cases. However, there have been many studies specifically focusing on those chemical additives used in plastics that exhibit endocrine disrupting properties and which may lead to a variety of health effects in wildlife and humans (UNEP/IPCP [International Panel on Chemical Pollution] 2016; Hermabessiere *et al.* 2017; M'Rabat *et al.* 2018; Flaws *et al.* 2020; UNEP 2020e). A wide range of chemicals in marine plastics collected from urban and remote beaches and open oceans were analysed and found to contain "non-persistent" additives such as alkyl phenols (i.e. nonylphenol, octylphenol and BPA) in concentrations ranging from nanograms per gram to micrograms per gram (Teuten *et al.* 2009; Hirai *et al.* 2011). Chemical additives with endocrine disrupting properties have also been recorded as prominent contaminants in marine species from areas where these types of chemicals are being used, for example in aquaculture operations (Hong *et al.* 2013) and where there is local production of textiles, polyurethane foams and toys (Chen *et al.* 2009; Darbra *et al.* 2011; Wang *et al.* 2017). Baini *et al.* (2017) reported the presence of seven different phthalate esters in samples of microplastics, plankton and blubber from different cetacean species taken in the same area; others are found in fish in some European waters (Rüdel *et al.* 2012).

Controlled laboratory experiments have demonstrated that ingested plastics can transfer sorbed and additive chemicals, including polycyclic aromatic hydrocarbons (PAHs), antimicrobials and halogenated flame retardants (HFRs), to marine worms (van Cauwenberghe *et al.* 2015), fish (Karlsson *et al.* 2017), amphipods (Chua *et al.* 2014) and plankton (Katija *et al.* 2017). Several studies have also tested for the transfer of chemicals from plastics into wildlife through laboratory experiments, modelling, and observational studies in natural settings. In field studies comparisons were made between abdominal fat tissue samples and plastics found in the gut of several seabird species (Tanaka *et al.* 2013; Hardesty *et al.* 2015; Tanaka *et al.* 2020); however, because the prey organisms, in which the chemicals were absent, had been collected several years after the seabirds the results could not be considered reliable (Tanaka *et al.* 2020).

More recent studies (Thaysen *et al.* 2020) show that the transfer of chemicals is context dependent, i.e. the likelihood of chemical transfer depends on several factors including the gut residence time, conditions (e.g. the presence of surfactants, pH, temperature) and the polymer type of the plastic (Gouin *et al.* 2011; Koelmans *et al.* 2014; Bakir *et al.* 2016; Rummel *et al.* 2016; Koelmans *et al.* 2019). Laboratory studies confirm that the contribution of chemical burdens by ingested plastics depends on the concentrations in the ingested plastics and the gut (Bakir *et al.* 2016; Herzke *et al.* 2016; Koelmans *et al.* 2016; Rummel *et al.* 2016; Anbumani and Kakkar 2018). UNEP (2020e) noted that ingestion is unlikely to increase the exposure to hydrophobic organic chemicals from adsorption because, overall, the flux of these chemicals from natural prey overwhelms the flux from ingested microplastics for organisms in most habitats.

In the case of highly contaminant-exposed animals ingesting all sizes of plastics with low concentrations of contaminants sorbed from the ambient environment, the concentration gradient is expected to be from the gut to plastics. This is referred to as depurating, whereby the ingested plastics in the gut sorb chemicals present in the organism, essentially "cleaning out" the animal (Rosenkranz *et al.* 2009; Koelmans *et al.* 2014; Herzke *et al.* 2016; Rummel *et al.* 2016; Thaysen *et al.* 2020). The opposite trend (i.e. transfer from plastics to the gut) is possible for chemicals with a higher concentration in the plastic, as occurs with polymeric additives or in the case of less contaminant-exposed animals ingesting highly contaminated plastics (Mohamed Nor and Koelmans 2019; Thaysen *et al.* 2020).

Finally, the chemical threat presented by certain alternative bio-based plastics (see Box 4) is similar to the threat presented by conventional plastics (Zimmermann *et al.* 2020).

as corals, cord grass, seagrasses and mangroves arising from contaminants of emerging concern, deprivation of oxygen and light, changes in reproductive output, and physical losses (Galloway and Lewis 2016; Rochman *et al.* 2016a; Sussarellu *et al.* 2016; Campbell *et al.* 2017; Anbumani and Kakkar 2018; Carvalho-Souza *et al.* 2018; Lamb *et al.* 2018; Paul-Pont *et al.* 2018; Bucci *et al.* 2019 UNEP 2019a; Kroon *et al.* 2020; Tanaka *et al.* 2020; Yu *et al.* 2020). Because plastic debris is lightweight and can float longer, it can disperse organisms further than other types of natural flotsam (Thiel and Gutow 2005; Bryant *et al.* 2016; Kirstein *et al.* 2016; Rech *et al.* 2016; Viršek *et al.* 2017; Lamb *et al.* 2018). Over 380 taxa, including microorganisms, seaweeds and invertebrates, have been found rafting on floating anthropogenic litter in the oceans (Kiessling *et al.* 2015), including pathogenic agents (Rech *et al.* 2016; Besseling *et al.* 2019). Goldstein *et al.* (2014) recorded the ciliate pathogen *Halofolliculina*, known to cause skeletal eroding band disease in corals, on floating plastic debris in the western Pacific and suggested that the spread of the disease to Caribbean and Hawaiian corals was due to rafting on the enormous quantities of litter reported in that area.

Other studies have highlighted the possibility that plastics can act as platforms for "chemical cocktails" of residual monomers, chemical additives and contaminants, such as heavy metals and persistent organic pollutants (POPs), sorbed from the surrounding environment (Rochman 2015 *et al.*; Turner 2016; Guo and Wang 2019; Yu *et al.* 2019).

When plastics fragment into microplastics and smaller sized particles (Corcoran 2021), they sink due to buoyancy loss and are deposited in different reservoirs (Ye and Andrady 1991), including on shorelines (McDermid and McMullen 2004) and the sea floor (e.g. Zhu, L. *et al.* 2019 and papers included), where they can impact different benthic communities. Microplastics possibly accumulate more in deep sedimentary habitats (Zhang *et al.* 2020) and within subsurface sediment layers (Näkki *et al.* 2017; Wang *et al.* 2019a). Once in the sediment, they can be ingested and reduced further in size (e.g. due to digestive grinding) and/or be transported to the sea floor upon egestion. There is also evidence of organisms forming microplastics through bioerosion (e.g. polychaetes in polystyrene debris [Jang *et al.* 2018] and sea urchins [Porter *et al.* 2019]).

The uptake and accumulation of microplastics throughout marine food chains is enhanced by the fact that organisms at lower trophic levels may "prey on" microplastics of a similar size as their natural prey by mistake (Ivar Do Sul and Costa 2014; Santana *et al.* 2017; Clukey *et al.* 2018; Choy *et al.* 2019; Peng, L. *et al.* 2020; Prata *et al.* 2020a; Rubio *et al.* 2020). They are also ingested indiscriminately by fish (Davison and Asch 2011; Chan *et al.* 2019) and filter feeders, accumulated and absorbed via the intestinal tract, and then translocated across trophic levels (e.g. Avery-Gomm *et al.* 2018; Renzi *et al.* 2018). Large filter feeders, including some whales, may devour microplastics directly or indirectly via contaminated organisms in seawater (e.g. humpback whales [*Megaptera novaeangliae*]) (Xiong *et al.* 2018; Besseling *et al.* 2019; Burkhardt-Holm and N'Guyen 2019).

Studies of ingested microplastics show that they generally stay within the digestive system, inside organs such as gills, intestines, stomachs and tubules (e.g. mussels [*Mytilus edulis*] and lugworms [*Arenicola marina*], Cauwenberghe *et al.* 2015; fish and prawns, Güven *et al.* 2017, Abbasi *et al.* 2018, Azevedo-Santos *et al.* 2019), where they are taken up by larger predators. In addition, microplastics can translocate and accumulate in other tissues, such as the haemolymph and circulatory system, after just a few days (Browne *et al.* 2008; Peng, L. *et al.* 2020). With seawaters becoming enriched by plastics, the likelihood of microplastics entering marine trophic webs where a wide variety of marine animals and humans will be exposed to them is increasing (Prata *et al.* 2020a).

Microplastics have been observed to compromise the ability of keystone species and ecological "engineers" in aquatic systems, such as corals and worms, to build reefs or to bioturbate sediments (Bradney *et al.* 2019; Renzi *et al.* 2019). For example, the attachment strength of blue mussels was halved after exposure to high-density polyethylene (HDPE) microplastics, potentially impacting their ability to form reefs (Green *et al.* 2019), and there was a reduced volume of sand overturned by lugworms exposed to microplastics (Green *et al.* 2016). In laboratory-based marine mesocosm studies the ingestion of microplastics reduces the health of lugworms in marine sediment by delivering harmful chemicals to them, including hydrocarbons, antimicrobials and flame retardants (Wright *et al.* 2013a; Wright *et al.* 2013b). Furthermore, the energy reserves of lugworms living in sediment contaminated with microplastic particles were reduced by up to 50 per cent due to reduced feeding activity, with adverse effects on their health. Lugworms perform vital ecosystem functions. They are a source of food for wader birds, fish and bait for fisheries. Lugworms provide another important ecosystem service by turning over huge volumes of sand, replenishing organic material and oxygenating the upper layers to keep the sediment healthy for other animals and microorganisms (including microscopic primary producers) to thrive in. A follow-up experiment by Green *et al.* (2016) found that lugworms exposed to either HDPE, PVC or polylactic acid (PLA) microplastics bioturbated less and that there was a corresponding decrease in the biomass of important primary producers in the sediment.

Mesocosm experiments in marine sedimentary habitats found that microplastics altered ecosystem functioning by decreasing the flux of inorganic nutrients (including ammonium and silicate) from the sediment and reduced the biomass of microscopic primary producers in the sediment (Green *et al.* 2017). There is also evidence that plastics can alter carbon cycling through their effect on primary production in marine, freshwater and terrestrial systems (Cole *et al.* 2016; Yokota *et al.* 2017; Porter *et al.* 2018; Boots *et al.* 2019; Prata *et al.* 2019a). Marine ecosystems such as mangroves, seagrasses, corals and salt marshes play a major role in carbon sequestration (McLeod *et al.* 2011; Herr and Landis 2016; Bindoff *et al.* 2019).

Microplastics, through their impacts on metabolic rates, reproductive success and survival of zooplankton, affect the carbon cycle in the ocean by altering the transfer of carbon to the deep sea (Cole *et al.* 2016; Wieczorek *et al.* 2019). Moreover, abiotic-biotic relationships can be affected by microplastics, for example through the microplastics causing temperature fluctuations on beaches where organisms such as sea turtle eggs occur and where sex is influenced by temperature (Carson *et al.* 2011; Beckwith and Guentes 2018). Reef-building corals in a remote coral reef atoll in the Maldivian archipelago were contaminated with phthalic acid esters, a class of microplastics-associated contaminants, possibly through ingestion of microplastics (Saliu *et al.* 2019). A laboratory experiment found that polyethylene microplastics (at 200 particles per litre for six months) can lead to a reduction in growth of some species of reef building corals (Reichert *et al.* 2019). Contamination of coral reefs could lead to deterioration of these vital biogenic habitats, but further research is needed.

Outdoor mesocosm experiments using naturally flowing seawater and intact sediment cores, simulating semi-field conditions, have also been used to assess the impacts of microplastics on invertebrate assemblages from three different sedimentary habitats (Green 2016; Green *et al.* 2017). In sandy habitats dominated by flat oysters the addition of either conventional (HDPE) or biodegradable (PLA) microplastics at high concentrations (80 µg per litre) caused a reduction in the number of species and in the overall abundance of organisms (Green 2016) (Box 4; Figure 2). Similarly, in a follow-up experiment, in muddy sediment dominated by flat oysters the addition of the same types of microplastics (25 µg per litre) resulted in a shift in community composition whereby opportunistic oligochaetes became dominant and predatory polychaetes declined (Green *et al.* 2017). Community level effects have also been found in an *in situ* experiment in freshwater sediments, where the diversity and abundance of macrofauna decreased in mesocosms with nano- or microplastics (Redondo-Hasselerharm *et al.* 2020). In addition, microplastics can have effects on the development of fungal communities with unique community composition and structure (Kettner *et al.* 2017).

Microplastics can act as habitats and may alter assemblages by fostering unique microbial communities. Factors driving the composition of the plastisphere are complex, mainly spatial and seasonal, but are also influenced by the polymer type, surface properties and size (Amaral-Zettler *et al.* 2015; Jacquin *et al.* 2019; Amaral-Zettler *et al.* 2020). Plastisphere communities studied in different polymer types floating in the North Pacific and North Atlantic reflected their biogeographic origins and, to a lesser extent, the plastic type (Amaral-Zettler *et al.* 2015, 2020). Similar conclusions were found for bacterial communities colonizing plastics along an environmental gradient. These communities are shaped firstly by freshwater and wastewater systems through to marine environmental conditions, and secondarily by the type of plastic (PS and PE) (Oberbeckmann *et al.* 2018). Inversely, another study based on a large number of microplastics sampled in the western Mediterranean Sea

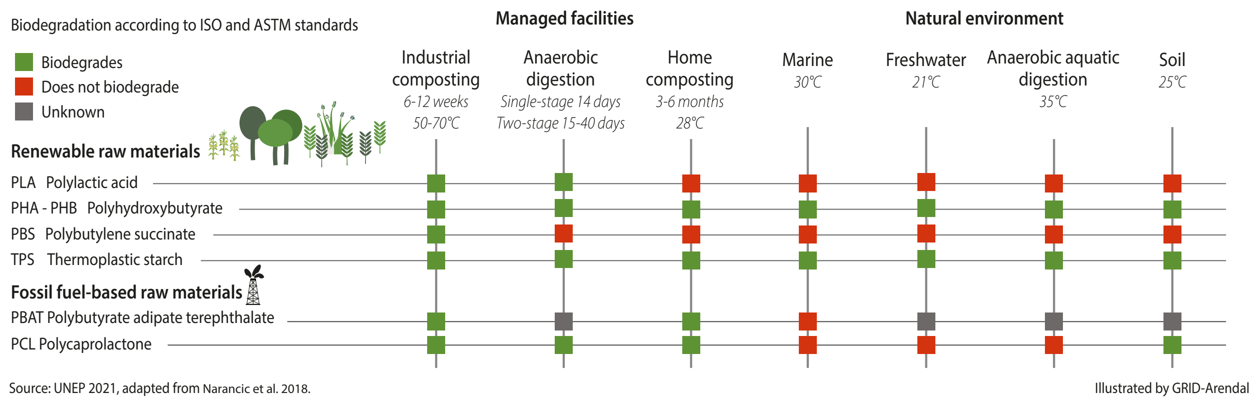


Figure 2: Bio-based plastics and their biodegradation

Box 4: Biological and ecological impacts of plastics labelled as biodegradable

Biodegradable and bio-based plastics have been presented as potential alternatives to fossil fuel-based plastics, for example as food packaging (Peelman *et al.* 2013). Although these two types currently account for only a small share of the market, their global production is predicted to grow rapidly as production costs decrease (European Bioplastics 2020) (see Figure 2). The persistence of bio-based and biodegradable plastics in aquatic habitats is uncertain, but for some time experiments have found that even after three years the majority of biodegradable plastics and blends failed to show any degradation in the marine environment or to meet International Organization for Standardization (ISO) and ASTM biodegradation standards (O'Brine and Thompson 2010; Alvarez-Zeferino *et al.* 2015; Narancic *et al.* 2018; UNEP 2018a; Napper and Thompson 2019). Unless specific and proper conditions for biodegradation are met (e.g. when industrial composting takes place), biodegradable plastics risk fragmenting into microplastic particles in much the same way as conventional plastics (Alvarez-Zeferino *et al.* 2015; Napper and Thompson 2019). In this sense biodegradable plastics do not advance sustainability considerations. Instead, the demand for more sustainable alternatives and non-petroleum-based materials should be encouraged (UNEP 2021a). The majority of bio-based and plant-based plastics also contain toxic chemicals and pose risks similar to those of conventional plastics, i.e. in terms of being carriers for pollutants and vectors for pathogenic organisms, with cellulose and starch-based products containing the greatest number of chemical features and inducing the strongest toxicity (Zimmermann *et al.* 2020).

There is evidence that, as litter, biodegradable plastics pose the same risks as conventional plastics to individuals, biodiversity and ecosystem functioning. For example, a field experiment comparing the impacts of conventional (HDPE) and biodegradable plastic bags found that the two types had exactly the same effect, reducing oxygen and light and decreasing the overall abundance of invertebrates and the biomass of primary producers beneath the bags as well as decreasing the flux of inorganic nutrients from the sediment (Green *et al.* 2015). In addition, several marine mesocosm experiments found that biodegradable microplastics (PLA) induced similar protein changes in mussels (Green *et al.* 2019), altered the feeding and metabolic rates of bivalves and lugworms (Green 2016; Green *et al.* 2016; Green *et al.* 2017), altered the diversity and abundance of infauna and the biomass of primary producers (Green 2016; Green *et al.* 2017), and decreased the release of inorganic nutrients from the sediment (Green *et al.* 2017).

Similarly, in terrestrial experiments microplastics composed of HDPE, PLA or synthetic clothing fibres had an effect on soil stability and decreased the germination and growth of plants, led to a lack of growth in annelids, and affected soil structure by reducing the formation of macroaggregates (Boots *et al.* 2019). In freshwater experiments, biodegradable polyhydroxybutyrate (PHB) and non-biodegradable polymethylmethacrylate microplastics both led to a decrease in biomass of the freshwater amphipod *Gammarus fossarum* (Straub *et al.* 2017). Although very little is known about the behaviour and breakdown of biodegradable microplastics in aquatic habitats, a recent study found that secondary nanoplastics released from polyhydroxybutyrate microplastics persist and have negative effects on freshwater organisms including water fleas, cyanobacteria and microalgae (González-Pleiter *et al.* 2019). Biodegradation tests are predominantly carried out in artificial environments that lack transferability to real conditions (Haider *et al.* 2018) and are unable to predict environmental impacts.

Very little is known about the effects of biodegradable plastic bag leaching (i.e. the transfer of chemicals from plastic into natural environments) on vegetation. Some plant species are highly sensitive to a variety of chemicals, and seedling growth is generally the most affected life history stage. In recent field studies (Balestri *et al.* 2017; Balestri *et al.* 2019; Balestri *et al.* 2020) the effects of conventional (HDPE) and compostable bags were tested when they were left in natural environments. The findings indicate that plastic bags labelled as meeting biodegradability and compostability standards do not meet those standards once they are discarded in natural environments.

For these reasons the general public should be adequately informed about the potential environmental impact of incorrect bag disposal through clearer labelling and information about the conditions under which biodegradability can occur (see Section 4). Simple, rapid standard phytotoxicity tests need to be applied to bag leachates.

showed no effect of geographical location (including coastal and open ocean samples) or plastic type (mainly PE, PP and PS) on the bacterial community composition.

Studies on the plastisphere are starting to give a better view of the microbial biofilm community on plastics in the oceans, but the complex network of influences is still the subject of ongoing debate (Sogin *et al.* 2006; Pedrós-Alió 2012; Zettler *et al.* 2013; Sauret *et al.* 2014; Amaral-Zettler *et al.* 2015; Dussud *et al.* 2018a; Dussud *et al.* 2018b; Wang *et al.* 2018). For example, some authors have reported that certain bacterial communities living on plastic, although they are rare in seawater, are made up of opportunistic species able to grow and become the "core species" living on plastics (McCormick *et al.* 2014; Dussud *et al.* 2018a). However, in a critical review of 66 studies which accounted for study quality (i.e. whether controls were included and interpreted properly) Wright *et al.* (2020) concluded that "research so far has not shown plastisphere communities to starkly differ from microbial communities on other inert surfaces".

At sea, plastics are almost immediately coated by an inorganic and organic conditioning film. The film is then rapidly colonized by microorganisms forming a biofilm on the surface, which is embedded within an exopolymeric substance matrix. These natural microorganismal assemblages act as a form of protection and offer metabolic cooperativity that can increase the possibility of gene transfer among cells. The composition of the biofilm depends on the polymer and its surface properties; some materials are very recalcitrant and inhibit the formation of biofilms, for example the stable aliphatic chains of polyethylene (PE), which dominates the composition of plastic waste on the sea surface (Auta *et al.* 2017; Morohoshi *et al.* 2018; Okshevsky *et al.* 2020). Weathered plastics may increase biofilm growth due to their increased surface area compared to non-weathered plastics (Rummel *et al.* 2017). Under different conditions various bacteria can degrade oxo-biodegradable and hydro-biodegradable plastics (Vázquez-Morillas *et al.* 2016; Dussud *et al.* 2018b; Eyheraguibel *et al.* 2018).

Pathogenic bacteria such as *Aeromonas salmonicida* and *Vibrio parahaemolyticus* have been found to colonize microplastic particles collected from the marine environment (Kirstein *et al.* 2016; Viršek *et al.* 2017). In laboratory studies plasmid transfer in bacterial assemblages has been found to be higher in communities that colonize microplastic particles compared to free-living communities (Arias-Andres *et al.* 2018). Horizontal transfer of genes encoding antimicrobial resistance (AMR) in microbes occurs faster within biofilms such as those developed on microplastics (Stewart and Costerson 2001; Goel *et al.* 2021); microbes in biofilms are also less susceptible to antibiotics than free-living cells in planktonic culture (Hall-Stoodley *et al.* 2004; Goel *et al.* 2021). Experimental work found that a greater rate of gene transfer occurred in bacteria on microplastics than in free-living bacteria, both in the water column (Arias-Andres *et al.* 2018) and in sediment (Huang *et al.* 2019). Data from the field support the hypothesized link between microplastics and AMR, with sampling in the North Pacific Gyre indicating that plastic debris (including both macroplastics and microplastics) is a reservoir of antimicrobial resistant microbes (Yang *et al.* 2019).

1.2 Potential risks to human health

1.2.1 Physical harm

The presence of litter can have direct consequences for physical and mental health. Visitors to beaches where there are large amounts of litter, as well as maritime workers, are susceptible to a range of injuries including cutting themselves on sharp debris, becoming entangled in nets, and exposure to unsanitary items (Santos *et al.* 2005; Campbell *et al.* 2019). Littered coastal areas have also been shown to be less beneficial to mood and mental well-being than unlittered ones, especially when the litter is post-consumer waste such as packaging (Wyles *et al.* 2015; Wyles *et al.* 2016).

1.2.2 Chemicals in marine plastics that pose risks to human health

Although the polymeric materials that comprise the core structure of marine plastics are biochemically inert, plastics contain chemical additives of small molecular size that are not bound to the polymeric materials and that may be harmful to human health (Landrigan *et al.* 2017; Takada and Karapanagioti 2019; Campanale *et al.* 2020; Landrigan *et al.* 2020; Prata *et al.* 2020a; Rubio *et al.* 2020; UNEP 2020e; Vethaak and Legler 2021). Once in the ocean, these additives can leach out of the plastic into the surrounding environment and enter the marine food chain (Andrady 2017; Peng, L. *et al.* 2020).

Some of the chemicals associated with plastics are recognized as mutagens and carcinogens (Landrigan *et al.* 2020; UNEP 2020e). Phthalates are produced in high volumes to be used as plasticizers, lubricants and solvents in a wide range of applications (e.g. in building and construction materials, medical and fragranced consumer products, and motor vehicles). Over 90 per cent of bisphenol A (BPA) is estimated to have been used as a monomer in the production of different polymers. Recent estimates show that in 2018 nearly 64 per cent of global BPA demand was for polycarbonates, nearly 30 per cent was for epoxy resins, and the rest was for other polymers such as phenoplast resins, phenolic resins, unsaturated polyesters and formaldehyde resins (Fischer *et al.* 2014; IHS Markit 2018). These polymers are commonly used in many everyday products across the globe. For example, polycarbonates are used in plastic bottles, food packaging materials, building and construction materials, optical media and electronics, and epoxy resins are used in marine and protective coatings, powder coatings, electronics, can and coil coatings and automotive materials, and as recyclates in roads and floorings (European Chemicals Agency 2017a,b). There is also a growing demand for black plastics in consumer products, which is being met by sourcing materials from the plastic housings of end-of-life waste electronic and electrical equipment (UNEP 2019e), thus potentially introducing restricted and hazardous substances into the recyclate (e.g. including brominated flame retardants, antimony, and the heavy metals cadmium, chromium, mercury and lead) (European Chemicals Agency 2018; Turner 2018; European Chemicals Agency 2019).

Other chemicals such as bisphenol A and phthalates, which are widely used in consumer products such as plastic bottles, are endocrine disruptors that can mimic, block or alter the actions of normal hormones, reduce human fertility and damage the nervous system (UNEP 2019e; Flaws *et al.* 2020; UNEP 2020e). Perfluorinated additives (PFAS compounds), widely used to create materials that repel water, are considered to be of concern, as are residual unreacted monomers and chemical catalysts that may be trapped in plastic resin during polymerization (UNEP 2020e). All these chemicals, as well as those that are adsorbed to plastic waste as it moves through the environment, can leach out of plastics so that people will potentially be exposed to them (Hahladakis *et al.* 2018; UNEP 2020e). Human biomonitoring studies and initiatives show that chemicals used in the manufacture of plastics, in water and wastewater treatment and in the food sector, or released from plastics during degradation, are widely present in human populations (WHO [World Health Organization] 2015; Mani *et al.* 2019; Wang *et al.* 2019b).

Additionally, there is growing evidence from studies in developed countries that human exposure to chemicals associated with the production, use and disposal of plastics is highly gendered (Lynn *et al.* 2017), although data from developing countries are limited. On a global scale there is a scarcity of gender disaggregated literature, for example on the number of workers in the plastic industry, their exposure to hazardous chemicals, and resulting health effects during specific plastic production processes and plastic waste management (i.e. recycling, incineration, open pit burning and combustion) (Lynn *et al.* 2017).

The potential health hazards of the polymers that are the structural backbone of marine plastics have been less well studied. Of particular concern are the microplastic and nanoplastic particles and microfibres formed when plastic waste enters the oceans and breaks down under the influence of weathering, mechanical abrasion and photodegradation. Manufactured microplastics are also of increasing concern. For example, synthetic microbeads (polystyrene spheres between 0.5 µm and 500 µm in diameter) are used, for example, in 3D printing, in human and veterinary medical products, and in cosmetics and personal care products such as toothpastes, abrasive scrubbers and sunscreens (Landrigan *et al.* 2020).

Plastic microparticles and microfibres in the marine environment can be absorbed by small organisms at the base of the food chain. They can then bioconcentrate and reach very high concentrations in top predator species. Microplastic and nanoplastic particles suspended in seawater are also ingested by filtering organisms such as oysters and mussels and can reach high concentrations in the tissues of these species, from whence they can potentially expose humans who eat seafood (Peng, L. *et al.* 2019; Kögel *et al.* 2020). These particles can affect assimilation efficiency (Blarer and Burkhardt-Holm 2016). In addition, marine microplastics and microscopic fibrous particles can become airborne through aerosolization and inhaled (Dris *et al.* 2016).

1.2.3 Potential human health effects

When considering any potential harm to humans from exposure to marine sources of microplastics and plastic associated chemicals, it is very important to recognize that humans are exposed to the same contaminants in their everyday lives (Figure 3a,b,c). The annual intake of microplastics by some humans has been estimated to range from 39,000 to 52,000 particles, depending on age and sex, rising to 74,000 to 121,000 particles when inhalation is considered; individuals who meet their recommended water intake only through bottled sources may be ingesting an additional 90,000 microplastics annually, compared to 4,000 in the case of those who consume only tap water (Cox *et al.* 2019). Any exposure from marine sources is thus most likely to be via ingestion of seafood rather than inhalation of microplastics suspended in the air or penetration of plastic nanoparticles through the skin, although such exposure may occur in the case of people handling beach waste (Dehaut *et al.* 2016; Adyel 2020; Kögel *et al.* 2020; Prata *et al.* 2020a). However, the overall exposure levels and health impacts remain uncertain (Wright and Kelly 2017; Koelmans *et al.* 2019; WHO 2019, Landrigan *et al.* 2020).

Human exposure to marine microplastics is primarily via ingestion of contaminated fish and shellfish when they are eaten whole, especially including the gut and liver (Landrigan *et al.* 2020). Generally only the flesh of large fish is eaten, but in some cultures the visceral organs of certain fish species are a sought-after delicacy (e.g. rabbitfish intestines, known as dayok in the Philippines, Bucol *et al.* 2020). Outside areas where fish and shellfish are the main sources of protein, exposure via this route may be limited compared to the inhalation and ingestion of microplastics via household dust (Bouwmeester *et al.* 2015; Catarino *et al.* 2018). Even remote coastal communities and indigenous peoples that rely heavily on marine mammals and fish species for food are likely to be exposed to microplastic particles and any toxic chemicals leaching from them via a phenomenon known as "atmospheric distillation" (see Glossary) (Atlas and Giam 1981; Houde *et al.* 2011; Tekman *et al.* 2020), creating a potential threat to their food security (European Environment Agency 2013; Hantoro *et al.* 2018; Danopoulos *et al.* 2020; Peng, L. *et al.* 2020; Rubio *et al.* 2020).

Exposure to microplastics in foodstuffs goes beyond seafood (Bouwmeester *et al.* 2015; Lusher *et al.* 2017a; Cox *et al.* 2019; International Pollutants Elimination Network 2019; Alexy *et al.* 2020; Conti *et al.* 2020). Other types of food containing microplastics include honey (40-660 items/kg honey), sugar (32 ± 7 items/kg sugar) (Liebezeit and Liebezeit 2013) and table salt (7-681 items/kg salt) (Yang *et al.* 2015; Karami *et al.* 2017; Lee *et al.* 2019). People can also be exposed to microplastic particles in drinking water (118 ± 88 particles/litre water) (Schymanski *et al.* 2018; Koelmans *et al.* 2019) and in foods such as bread, processed meat, dairy products (Kutralam-Muniasamy *et al.* 2020) and vegetables. Individuals who drink water only from bottled sources ingest more than 90,000 microplastic particles annually, compared to an annual intake of 4,000 particles ingested by those who drink tap water (Landrigan *et al.* 2020; Vethaak and Legler 2021). Much of the microplastics in foods may originate from plastic packaging materials, including plastic bottles.

Microplastics, particularly microfibres, are present in air (Dris *et al.* 2015a; Dris *et al.* 2016), especially indoors (Alzona *et al.* 1979). Indoor environments have been found to have microfibre concentrations ranging from 3-15 particles per cubic metre of air (Gasperi *et al.* 2015) to as high as 0.4-59.5 particles per cubic metre, while concentrations of 0.3-1.5 particles per cubic metre have been recorded in outdoor environments (Dris *et al.* 2017). Beyond these few studies, there is little information on levels of airborne plastic microparticles in households, workplaces or recreational parks. Preliminary investigations have found that airborne plastic microfibres in urban environments range between 200 µm and 600 µm in diameter (Dris *et al.* 2015a; Dris *et al.* 2016). They are respirable and small enough to penetrate deeply into the human lung, where plastic microfibres up to 250 µm in length have been detected (Pauly *et al.* 1998; Landrigan *et al.* 2020; Vethaak and Legler 2021).

As Landrigan *et al.* (2020) have shown, particle size impacts are critical in assessing potential human health impacts. The quality of studies is also vital when considering reliability. A systematic review of the quality of drinking water studies, and studies on source waters and microplastics, commissioned by WHO (Koelmans *et al.* 2019) shows that the vast majority of studies do not report on concentrations of smaller particles (including nanoparticles), which are most likely the sizes of concern for human exposure and health effects (Box 2). Koelmans *et al.* (2020) recently provided an approach for aligning different studies, and disseminating the results in common language, in order to assess the risks of microplastics as an environmental material (see Section 3.1.4).

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Biological gender differences such as body size, amount of fat tissue, reproductive organs, hormones, and other biological and physiological differences also have an impact on the effects and elimination of toxic substances in the body (Landrigan *et al.* 2020). Microplastics have now been detected in the placenta (Ragusa *et al.* 2021). Women's higher proportion of body fat provides a greater reservoir for bioaccumulating and lipophilic chemicals; for example, the United States Centers for Disease Control and Prevention reported that women, in comparison to men, had significantly higher levels of 10 of the 116 toxic chemicals tested, three of which were phthalates commonly found in health and beauty products (Lynn *et al.* 2017).

Gradually, as more is known about microplastic particles and fibres (Barboza and Gimenez 2015; Wright and Kelly 2017; European Union 2019a; Chang *et al.* 2020) it may be hypothesized that ingested or inhaled microplastics could harm human health through a variety of mechanisms, some of which will be relevant for marine plastics (Landrigan *et al.* 2020; Vethaak and Legler 2021). These mechanisms include physical presence (e.g. causing abrasion, blockages or cellular damage), chemical composition (chemical additives used in their production or ambient chemicals adsorbed from the surrounding environment), and acting as vectors for pathogenic bacteria such as *Vibrio* spp. (Kirstein *et al.* 2016) and antimicrobial resistant bacteria (Eckert *et al.* 2018). Molecular mechanisms through interactions with microplastic particles could also injure health as a result of oxidative stress, inflammatory reactions and metabolic disorders (Landrigan *et al.* 2020).

Information about the toxicity of microplastic particles

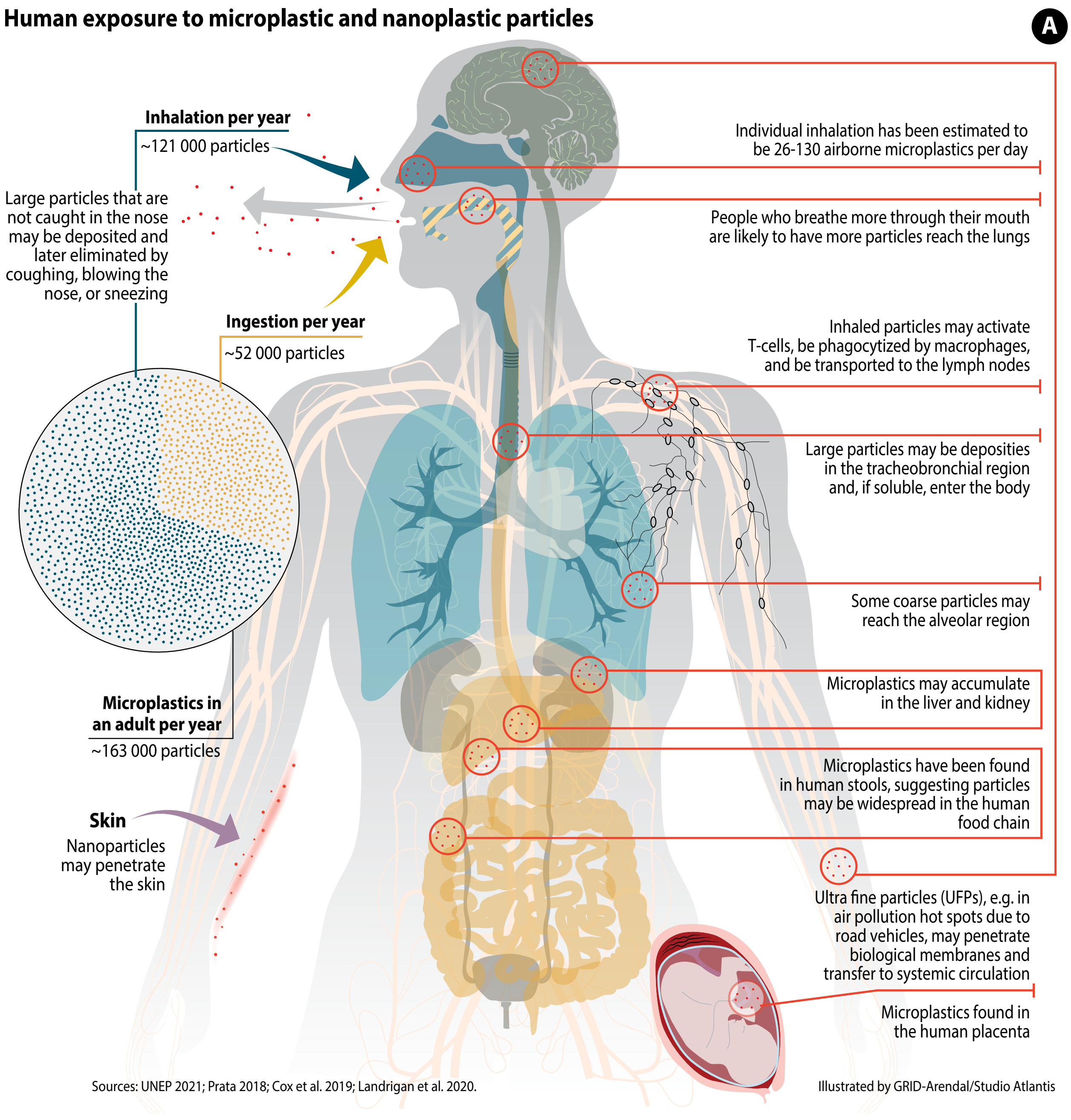

Figure 3a: Human exposure to microplastic and nanoplastic particles

and fibres is also beginning to emerge from two sources: toxicological studies of laboratory animals exposed to these materials, and occupational clinical and epidemiological studies.[3] Studies of occupationally exposed populations can be extremely informative because these groups are often exposed earlier than the general population and sustain relatively high levels of exposure. Since they consist of well-defined groups, occupational populations can also be readily followed and studied and gender aspects assessed (Lynn *et al.* 2017). Toxicological studies have reported that microplastics (5-20 µm in diameter) fed to rodents accumulate in the liver and kidney, causing inflammation and changes in metabolic profiles (Deng *et al.* 2017). Inhaled micro- and nanoplastic particles (1-20 nm in diameter) were reported to activate T-cells, leading to particles being transported to lymph nodes and creating a higher risk of cancers (Blank *et al.* 2013).

Occupational exposures to airborne microplastic fibres among workers in the textile and flocking industries have been associated with interstitial lung disease (Boag *et al.* 1999; Kern *et al.* 2000), cardiac and autoimmune disease, and lung cancer (Kern *et al.* 2011; Prata 2018). Some experts are concerned that the human health effects of microplastic fibres could be similar to those caused by exposure to asbestos (Kane *et al.* 2018).

Other health hazards associated with marine plastics can arise upstream depending on waste disposal methods, for example where there are informal waste management schemes (e.g. beach collection) and uncontrolled or incomplete combustion of the collected waste. Marine litter collected on beaches

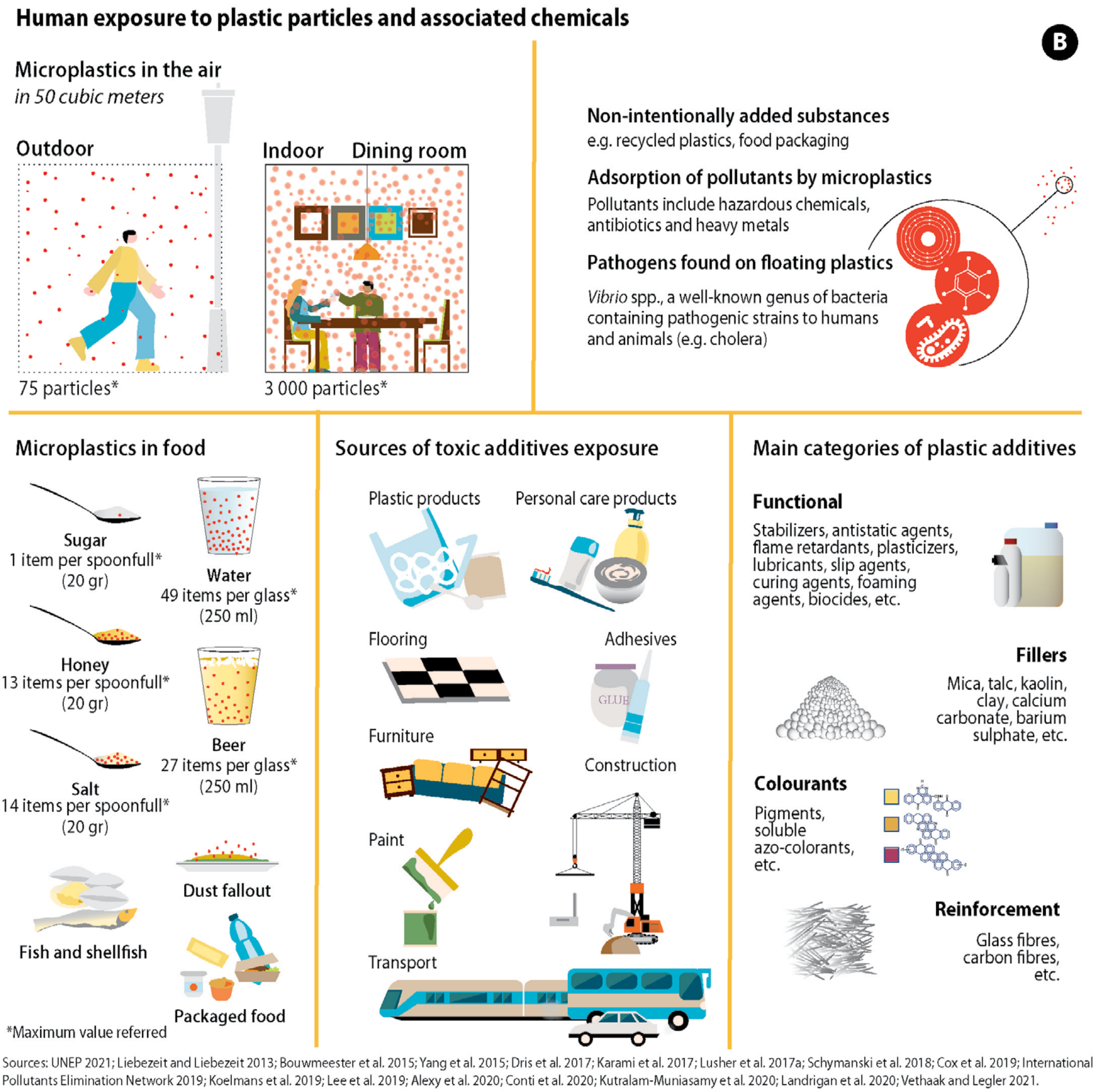

Figure 3b: Human exposure to plastic particles and associated chemicals

can endanger human health. When it is burned in open pits, people are exposed to the fumes, which can contain a variety of hazardous and carcinogenic materials including polycyclic aromatic hydrocarbons (PAHs), hydrochloric acid (from combustion of PVC plastics), dioxins and furans (from combustion of PVC plastics), lead (used as a plastic stabilizer), brominated flame retardants, and short-chain chlorinated paraffins (Zhang *et al.* 2017; Babayemi *et al.* 2019; UNEP 2019e). The significant increase in volumes of plastic waste arising from the disposal of personal protective equipment and other plastic items used during the COVID-19 pandemic are creating an additional hazard for local communities (Prata *et al.* 2020b). However, the need to solve this problem could trigger widespread shifts in the treatment of marine litter and plastic pollution (Adyel 2020; Canning-Clode *et al.* 2020).

Overall, there is still a poor understanding of the background levels of microplastic and microfibre contamination in an average household and whether these concentrations have the potential to cause harm to human health. The chronic toxic effect concentrations and underlying toxicological mechanisms by which microplastics elicit effects are still not well enough understood, although they have been looked at in six assessments with different scopes by national governments and intergovernmental institutions (UNEP 2020e). A precautionary approach in the management of plastics is thus still warranted (European Environment Agency 2013; European Food Safety Authority (EFSA) Safety Authority Panel on Contaminants in the Food Chain 2016; WHO 2019).

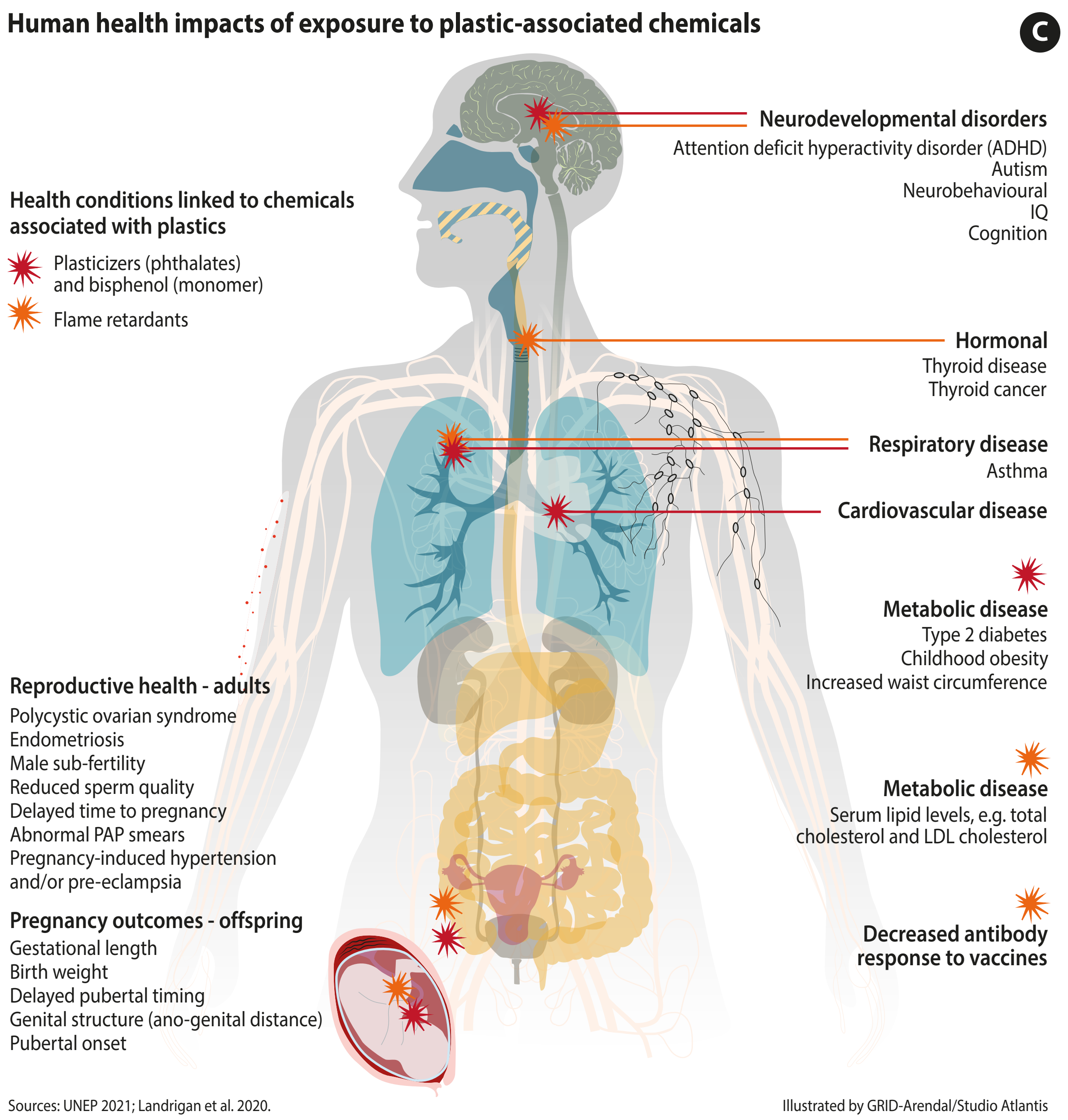

Figure 3c: Human health impacts of exposure to plastic-associated chemicals

1.3 Impacts of marine litter and plastic pollution on maritime industries

1.3.1 Impacts on fisheries and aquaculture

Marine plastics have the potential to reduce the efficiency and productivity of commercial fisheries and aquaculture operations through physical entanglement and damage (Mouat *et al.* 2010) and posing a direct risk to fish stocks and aquaculture (Lusher *et al.* 2017a).

The most important impact of macroplastic debris on fisheries occurs through ghost fishing by abandoned, lost or otherwise discarded fishing gear (ALDFG) (Richardson *et al.* 2019). The magnitude of losses to fisheries and other maritime industries remains uncertain, but in their interim report GESAMP Working Group 43 (GESAMP 2020b) finds that the contribution of sea-based activities and industries to the global burden of marine litter warrants concern largely because synthetic materials make up significant portions and components of the litter entering the world's oceans from fishing, aquaculture, shipping, ocean dumping, and other maritime activities and sources. Furthermore, certain types of sea-based marine litter, such as ALDFG, are known to impact marine resources, wildlife and habitats. A recent study suggests that commercial fishing off Norway alone contributes nearly 400 metric tons of plastics from lost fishing gear and parts per year to marine plastic waste (Deshpande *et al.* 2020).

Ghost fishing, so-called because abandoned nets and traps may continue to catch fish and shellfish, can cause significant levels of mortality to commercial stocks which, in many cases, are already under pressure. Gill nets and trammel nets are used worldwide, mostly by coastal and artisanal fisheries. They are the most problematic type of equipment in terms of quantities lost, as they are relatively non-selective with higher levels of by-catch of non-commercial species (e.g. Sullivan *et al.* 2019). Pots and certain types of long-line fisheries also present a threat to marine biodiversity when gear is lost or abandoned in coastal areas (Jeffrey *et al.* 2016; Sullivan *et al.* 2019). Few estimates of tonnage losses from ghost fishing have been published since the work of Webber and Parker (2012) and Scheld *et al.* (2016), but actions to remove this type of debris are important. For example, a long-term study by Sullivan *et al.* (2019) demonstrates that removing disused fishing gear in a large coastal estuary in the United States led to significant ecological and economic benefits, with future benefits anticipated from increased harvests. Actions to remove this type of debris from aquaculture are also important, but seem to be broadly missing in the recent global stock-taking exercise conducted as part of the work of the Ad hoc open-ended expert group (AHEG) on marine litter and microplastics (UNEP/AHEG/4/INF/6).

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Marine litter also has impacts on fisheries and aquaculture through the introduction of invasive alien species which can result in serious economic losses (e.g. Barnes 2002; Kiessling *et al.* 2015; Kirstein *et al.* 2016). Beck *et al.* (2018) showed that the loss of coastal flood protection services provided by reefs could lead to damages of up to US$ 272 billion globally.

Regarding fish stocks and aquaculture, the consumption of plastic and associated contaminants puts fish and shellfish stocks at risk of lethal and sublethal harm (e.g. through diminished reproductive success and growth), with the capacity for population-level impacts and economic losses (Sussarellu *et al.* 2016; Galloway *et al.* 2017; Peng, L. *et al.* 2020).

Overall, the published evidence suggests that the productivity, viability, profitability and safety of the fishing and aquaculture industries are highly vulnerable to the impacts of marine plastics, particularly when coupled with other factors including climate change and overfishing. A case study in Canada showed that farmed mussels had higher concentrations of microplastics compared with wild forms (Mathalon and Hill 2014). Globally marine aquaculture of fish and molluscs represents 17 per cent of the total volumes produced by aquaculture (FAO 2020); however, seafood from both mariculture and capture fisheries makes up 20 per cent of food intake by weight for 1.4 billion people (Golden *et al.* 2016). Such a high dependency on seafood for nutrition means the well-being of a significant proportion of the world's population is highly vulnerable to any changes in the quantity, quality and safety of this food source because of plastic pollution.

1.3.2 Impacts on tourism and heritage

Marine litter on beaches presents serious visual and aesthetic problems for tourists and others who visit beaches, especially in pristine areas, although this aspect has not been deeply researched. Litter has a substantial negative impact on recreational experiences and overall beach enjoyment, causing declines in coastal tourism and a corresponding loss of revenue (Munari *et al.* 2015; Pasternak *et al.* 2017; UNEP 2017a; Petrolia *et al.* 2019; Williams and Rangel-Buitrago 2019). People have reported their concerns about litter when visiting coastal areas (Penn *et al.* 2015; Krelling *et al.* 2017; Hartley *et al.* 2018a). Visitors spend less time at or avoid certain sites if they anticipate that those sites will be littered (e.g. Wyles *et al.* 2015; Kaminski *et al.* 2017; Krelling *et al.* 2017; Pasternak *et al.* 2017; Leggett *et al.* 2018; Qiang *et al.* 2020). Leggett *et al.* (2014) demonstrated that in Orange County, California (United States) marine debris had a significant impact on residents' beach choices; a 75 per cent reduction in marine debris at six popular beaches led to US$ 53.4 million in benefits to county residents during a three-month period.

1.3.3 Impacts on maritime shipping and port operations

Despite the potentially high risks to shipping posed by marine debris (Macfadyen *et al.* 2009), there are very few estimates of the total costs with respect to navigational systems and associated safety issues. Marine debris can present navigational hazards to ships at sea, for example due to entangled propellers and rudders, blocked water intakes, and collisions with floating objects. Entanglement of propellers can significantly reduce stability and maneuverability, with the potential to put crew and passengers in danger, particularly when weather conditions are bad. Injuries or deaths associated with marine debris could be accompanied by financial costs (Cho 2005; McIlgorm *et al.* 2011; Newman *et al.* 2015; UNEP 2016a).

Derelict fishing gear (DFG) can be a navigational hazard to commercial or recreational vessels (Jeffrey *et al.* 2016; Hong *et al.* 2017b). Economic costs result from necessary changes in navigation to avoid derelict gear, as well as damage to vessels and equipment. Costs may be significant in areas with heavy commercial or recreational traffic. Jeffrey *et al.* (2016) proposed using route planning models to quantify the costs of increased hazards in regard to navigational decision-making and/or vessel and equipment damage assessments. Fuel, labour and material/equipment costs related to vessel traffic patterns with and without the need to take DFG into account could then be evaluated. Hong *et al.* (2017b) studied ships belonging to the Republic of Korea Navy and found that propellers or shafts were entangled by DFG 2,386 times in six years (2010–2015), with each ship suffering at least one entanglement per year and requiring 135 hours of diver time. The costs for all vessels operating around the country's coasts were US$ 96.7 million per year.

Collisions with shipping containers are a recognized cause of damage, but they are not consistently reported unless they result from a catastrophic event such as loss of a vessel. The World Shipping Council (2020) estimated that in the period 2008–2019 an average of 1,382 containers were lost at sea each year, not counting those lost during catastrophic events, while an average of 1,582 containers were lost if these events were included.

Much ship-based waste is handled by port reception facilities (e.g. European Commission 2018c). The costs represent avoided costs of clean-up and of damage to ships by marine plastics in coastal areas. Most waste management plans drawn up by vessel operators entail discharging plastic with other waste products at port reception facilities for responsible land-based disposal. If plastic is to be recycled, it must be segregated before a vessel's arrival and properly handled once landed. Although this is regular practice in many commercial operations, other maritime sectors need to align with it to ensure that segregation is an integral part of the routine workflow. For example, at the Port of Rotterdam in the Netherlands plastics are specifically addressed in the waste management plan. The port collects plastic separately from other waste products, advocates waste prevention, and encourages vessels to limit the amount of plastics taken on board when replenishing (IMarEST [Institute of Marine Engineering, Science and Technology] 2019). Overall, the operational costs of port reception facilities are borne by the ships using a port.

1.4 Economic costs of marine litter and plastic pollution

The annual global economic costs of marine plastic pollution with respect to tourism, fisheries and aquaculture, together with other costs including clean-up activities, are estimated to be at least US$ 6-19 billion globally (Deloitte 2019). This estimate represents only a small percentage of the value of the global market for plastic products, estimated at around US$ 580 billion in 2020, (compared with an estimated US$ 502 billion in 2016) (Statista 2021a). However, the Deloitte (2019) estimate does not directly include impacts on human health or marine ecosystems. There is insufficient available research on these impacts. Lack of comprehensive figures for all costs appears to be a common problem (Newman *et al.* 2015; UNEP 2017a; Gattringer 2018).

Four types of economic costs need to be addressed: actual expenditures required to prevent or recover from damage caused by marine debris (e.g. for beach clean-ups, repair of vessels and fishing gear, and medical care following marine debris related accidents); losses of output or revenue owing to interactions with marine plastic pollution; losses of plastic material (as valuable material withdrawn from production); and welfare costs including human health impacts and loss of ecosystem services, among which are those ecosystem services related to aesthetic pleasure and recreation. The majority of published studies have focused on economic damage or direct losses at regional, national and local levels (e.g. Hall 2000; MacFadyen 2009; Mouat *et al.* 2010;

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McIlgorm *et al.* 2011; Jang *et al.* 2014; Newman *et al.* 2015; Krelling *et al.* 2017; Gattringer 2018; Leggett *et al.* 2018; Dalberg Advisors, WWF Mediterranean Marine Initiative 2019; Qiang *et al.* 2020; and sections below) and the price adjustments needed to internalize the social costs of plastics (e.g. Ferreira *et al.* 2007; Oosterhuis *et al.* 2014). Some studies have examined the non-market and intangible social and ecological costs of marine litter and plastic pollution; for example, in a study of a coastal fishing community on Thailand's Andaman Sea "increased garbage in the ocean" was ranked as the highest environmental stressor (Lynn *et al.* 2017). However there are too few studies to provide a robust estimate globally.

On the regional scale there are more studies looking at this issue. In the Mediterranean Sea, acknowledged to be one of the world's most affected seas (Eriksen *et al.* 2014; Cózar *et al.* 2015; UNEP/MAP 2015; Suaria *et al.* 2016; UNEP/MAP 2017; Campanale *et al.* 2019; Constantino *et al.* 2019; Dalberg Advisors, WWF Mediterranean Marine Initiative 2019; Fossi *et al.* 2020), there were annual losses of US$ 696 million in the three major sectors (fisheries and aquaculture, shipping and tourism), including US$ 150 million in the fisheries sector alone (Dalberg Advisors, WWF Mediterranean Marine Initiative 2019). These figures do not include losses due to reduced income or damage to ecosystem services caused by plastics.

In the Asia-Pacific Economic Cooperation (APEC) countries the estimated annual economic costs of marine litter in 2008 were US$ 1.26 billion (McIlgorm *et al.* 2008; McIlgorm *et al.* 2011), rising to US$ 10.8 billion in 2015 (Asia-Pacific Economic Cooperation 2017; McIlgorm *et al.* 2020). These figures for the Asia-Pacific region reflect increasing global plastic production. Statista (2021b) estimates that cumulative global production was 8.3 million metric tons in 2017 and will grow to 34 million metric tons in 2030. The world's maritime industries are also growing: as of 2019 the total value of annual seagoing shipping trade is reported to have been more than US$ 14 trillion (International Chamber of Shipping 2019).

For many countries economic data on the costs of damage caused by marine plastics do not exist (Janssen *et al.* 2014; Jambeck *et al.* 2018). However, avoided costs created by the informal waste picking sector are sometimes a useful indicator; in 2016 waste pickers were estimated to be responsible for collecting 55-64 per cent of plastics for recycling globally (Lau *et al.* 2020). This sector generally comprises small businesses and self-employed individuals who operate with low capital investment and little or no state regulation. Informal waste pickers generate huge savings for cities by reducing the volume of low-value waste that needs to be collected and taken to landfills (UNESCAP [United Nations Economic and Social Commission for Asia and the Pacific] 2019). For example, in India informal waste collection saves the Pune municipality an estimated US$ 10 million per year in labour costs, at statutory wage rates, and about US$ 2 million in reduced waste transportation and processing costs. In Lima (Peru), Cairo (Egypt), and Quezon City (the Philippines) informal waste pickers are estimated to contribute around US$ 15.9 million, US$ 13.7 million and US$ 3.9 million, respectively, to annual avoided waste collection and disposal costs.

Estimating the costs of damage to ecosystem functioning is also challenging, with limitations in the accuracy of previous analyses (e.g. Costanza *et al.* 1997; Börger *et al.* 2014; Costanza *et al.* 2014) having been pointed out (e.g. by Pendleton *et al.* 2016). Beaumont *et al.* (2019), who used De Groot *et al.* (2012) and Costanza *et al.* (2014) to combine economic values for different components of marine ecosystems with estimates of the impacts of marine plastic on ecosystem services based on the volume of plastics in the marine environment in 2011, estimated to be between 75 and 150 million metric tons (Jang *et al.* 2015; Ocean Conservancy and McKinsey Business Centre 2015). The outcome from Beaumont *et al.* (2019) was that each ton of plastic in the oceans leads to an annual cost, in terms of reduced marine natural capital, of between US$ 3,300 and US$ 33,000 or an overall yearly loss of US$ 500-2,500 billion. Analysing the loss of benefits that marine ecosystem services provide is an appropriate method for estimating the non-market, intangible costs of marine plastics, but before these costs can be applied globally it is clear that a more profound interdisciplinary approach is needed which better addresses the interdependencies between economic and ecological systems (Gattringer 2018).

Compared to the size of the global plastic market in 2020, estimated at around US$ 580 billion (Statista 2021a), the World Trade Organization reports that the value of global merchandise exports alone in 2020 was around US$ 17.65 trillion (compared to US$ 19.014 trillion in 2019 and 19.55 trillion in 2018, before the COVID-19 pandemic began) (World Trade Organization [WTO] 2021). The value of trade flows of plastics from raw materials to finished goods have recently been calculated to amount to about US$ 1 trillion (UNCTAD 2020). However, the price of virgin plastics does not reflect the full environmental, economic and social costs of disposing of them. Instead, these costs are passed on, for example to coastal communities and the maritime sectors. The Pew Charitable Trusts and SYSTEMIQ (2020), using a business-as-usual scenario for 2040, projected that 4 billion people are likely to be without organized waste collection services by that year and that businesses could face a US$ 100 billion annual financial risk if governments required them to cover waste management costs at expected volumes and recyclability. Figures such as these are indicative of widespread market failures and underline the need for a systems-wide, solutions-based approach that focuses on the challenges – technological (e.g. the scalability of different recycling technologies and substitute materials), economic (e.g. the relative cost of different solutions), environmental (e.g. greenhouse gas [GHG] emissions associated with different solutions) and social (e.g. equity and social justice for waste pickers) – that need to be met to prevent mismanaged plastic waste and the subsequent costs of environmental pollution entering the marine environment (Lau *et al.* 2020).

1.5 Social impacts of marine litter and plastic pollution

There is growing awareness worldwide that the marine environment is under threat from plastic pollution and overfishing (Wyles *et al.* 2016; Hartley *et al.* 2018b; Lotze *et al.* 2018). Awareness of other threats such as habitat alteration, climate change and biodiversity loss, although felt to be important, is not as great, perhaps due to their lower direct visibility, greater complexity or, as in the case of climate change, because they have not previously been perceived as a direct threat to the oceans (Reid *et al.* 2009; Lotze *et al.* 2018).

Changes in perception and awareness are important. There is extensive evidence that people experience well-being by knowing that marine animals exist and will continue to do so, even if they never experience them personally (Borger *et al.* 2014; Jobstvogt *et al.* 2014; Aanesen *et al.* 2015; Eagle *et al.* 2016). Charismatic marine animals, including turtles, whales, dolphins and many seabirds, have cultural and emotional importance. Images and descriptions of whales and seabirds whose stomachs are full of plastic fragments, which are prevalent in mainstream media,[4] can have a strong detrimental impact on people's emotions and sense of well-being (Lotze *et al.* 2018).

Litter is cited as a key reason visitors spend less time on beaches or avoid some sites altogether if they anticipate they will find litter there (Ballance *et al.* 2000; Tudor and Williams 2003; Kiessling *et al.* 2017; Hartley *et al.* 2018a). Not visiting beaches and shorelines can have health implications if it means there is a lack of opportunity to enjoy benefits such as physical activity, social interaction (e.g. strengthening of family bonds), and general improvement of physical and mental health (Ashbullby *et al.* 2013; Papathanasopoulou *et al.* 2016; Kiessling *et al.* 2017; White *et al.* 2020). On the other hand, the presence of litter is known to stimulate citizen programmes and beach clean-up activities (Brouwer *et al.* 2017; Hartley *et al.* 2018b).

Handling marine litter and plastics can have different impacts on particular groups (e.g. women, children, coastal communities, waste workers); moreover, when individuals collecting waste from beaches and coastal areas problems may arise if they are perceived to be competing with established municipal waste management systems (ILO [International Labour Organization] 2017; UNEP 2017a; ILO 2019). In hazardous working environments, where all types of waste workers may be exposed to fumes from waste burning and adequate occupational safety and health measures may be lacking, these workers are exposed to numerous health risks including exposure to hazardous chemicals associated with plastics (ILO 2017; ILO 2019; UNESCAP 2019; Velis and Cook 2021).

It has been proposed by van den Bergh and Botzen (2015) and others that the social costs of marine plastic should be included in solutions to the ways plastics are produced, used, reused and reprocessed, employing an approach similar to the "Social Cost of Carbon". Marine litter and plastic pollution can infringe on a number of human rights. They affect people in vulnerable conditions disproportionally, including those living in poverty, indigenous and coastal communities, and children, potentially aggravating existing environmental injustices (United Nations General Assembly 2021).

1.6 Risk framework for marine litter and plastic pollution

Because of the multiple and cascading risks that marine litter and plastic pollution pose to the oceans and society (Figure 1), they can act as threat multipliers (UNDRR [United Nations Office for Disaster Risk Reduction] 2019). Plastics, in particular, are stressors that may combine with other stressors (e.g. climate change, overexploitation of marine resources), resulting in far greater damage than when they are considered in isolation (Backhaus and Wagner 2019). For example, GHG emissions from the production, recycling and incineration of fossil fuel-based plastics account for 19 per cent of the total emissions budget allowable in 2040 if the world is to avoid significant climate change (The Pew Charitable Trusts and SYSTEMIQ 2020). Habitat alterations in key coastal ecosystems caused by the direct impacts of marine litter and plastic pollution not only affect local food production and coastal protection, but may lead to wide-reaching and unpredictable secondary societal consequences through impairment of ecosystem resilience and the potential of coastal communities to withstand extreme weather events and climate change (Galloway *et al.* 2017; Carvalho-Souza *et al.* 2018; Woods *et al.* 2019; GESAMP 2020a). Such considerations underscore the urgent need for a coherent approach to managing the risks of marine litter and plastic pollution (Colborn *et al.* 2011; UN General Assembly 2016; Hardesty and Wilcox 2017; Royer *et al.* 2018; Adam *et al.* 2019; Backhaus and Wagner 2019; UNDRR 2019; GESAMP 2020a; Peng, L. *et al.* 2020; Shen *et al.* 2020).

The Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) (2020a) suggested that no single approach to risk would be suitable for assessing the wide range of potential hazards and exposure routes associated with marine litter and microplastics which would take into account all the possible social, economic and environmental consequences. Instead, setting out a "risk assessment landscape" and adopting a tiered approach for addressing marine litter and plastic pollution has been proposed (Koelmans *et al.* 2017a; GESAMP 2020a). This approach reflects increasing experience with the development of tools to assess hazard and risk in a wide range of applications, for which relevant factors to be considered (including existing knowledge and urgency) vary, taking social considerations and potential public or environmental health risks into consideration. The objective of such a risk framework would be to deliver "fit for purpose" risk framework to ensure that non-priorities are set aside and to inform risk management (Koelmans *et al.* 2017a). Risk matrices can also provide a way to highlight where knowledge gaps exist and can aid problem formulation. The development of a risk appraisal procedure and common risk framework is thus a critical step going forward.


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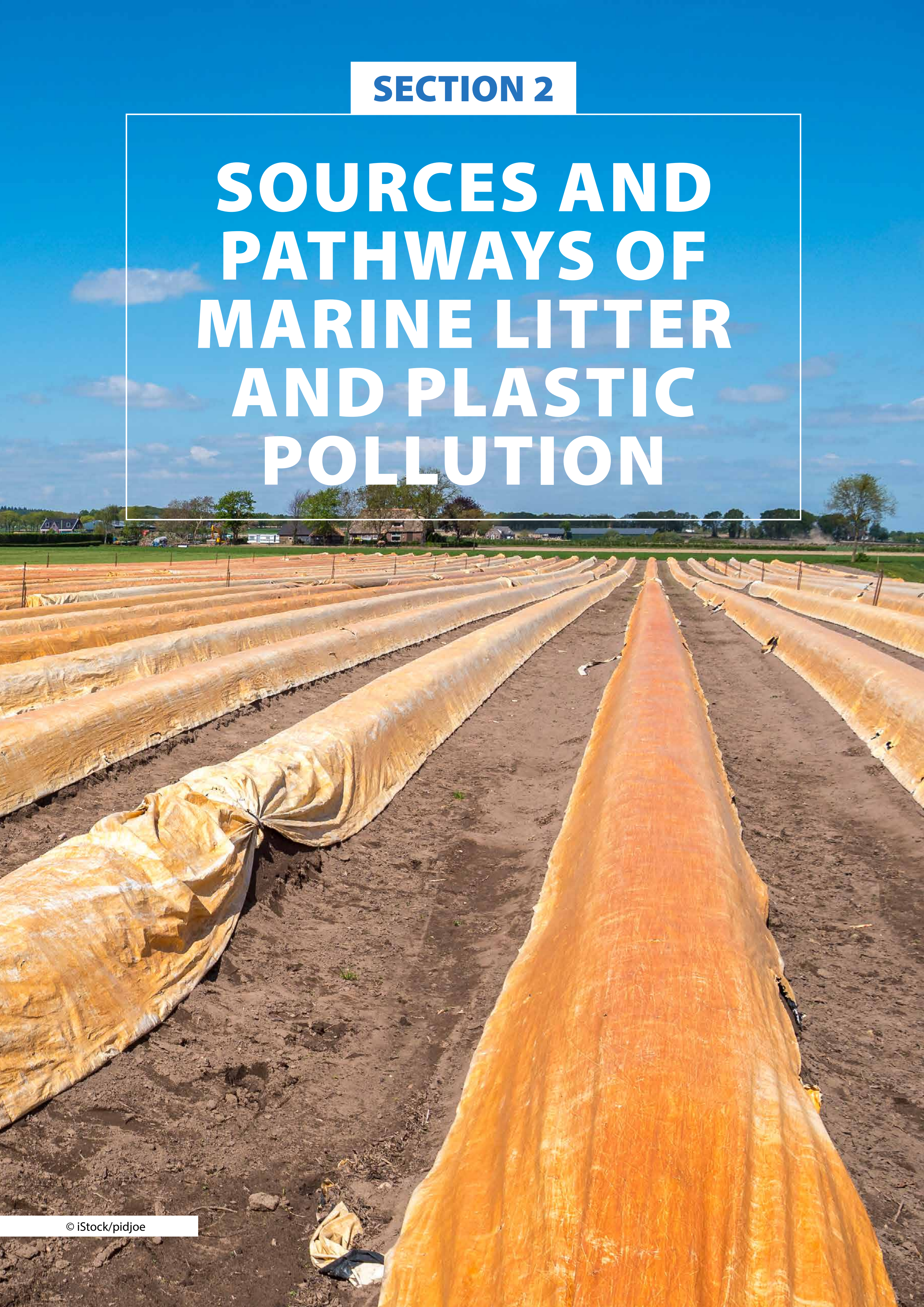
SECTION 2

SOURCES AND PATHWAYS OF MARINE LITTER AND PLASTIC POLLUTION



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2.1 Major sources of marine litter and plastic pollution

2.1.1 Land-based sources

The main drivers of marine litter and plastic pollution are the growing volumes of plastics being supplied to the global economy (IRP [International Resource Panel] 2019; Geyer 2020) and the waste and emissions arising from their use and disposal (Veiga *et al.* 2016; Lusher *et al.* 2017a; Piehl *et al.* 2018; Rochman *et al.* 2019).

On the supply side, total production of plastics in 2019 was 368 million metric tons. Due to the impacts of the COVID-19 pandemic, it is estimated that production in 2020 decreased by approximately 0.3 per cent (Malik *et al.* 2020; ICIS [Independent Commodity Intelligence Services] 2020; Statista 2021b). The chemical industry is likely to become more complex in the future, with GHG emissions, climate change, demographics and technology, for example, all having impacts (McKinsey and Company 2020). However, it was recently estimated that global production of primary plastic would increase to 1,100 million metric tons per year by 2050 if historic growth trends continue (Geyer 2020).

Of the global cumulative production of primary plastic between 1950 and 2017, estimated at 9,200 million metric tons, roughly 7,000 million metric tons became plastic waste; of this amount, 1,000 million metric tons were incinerated (14 per cent) and 5,300 million metric tons (76 per cent) were discarded, ending up in landfills or dumps or as a component of uncontrolled waste streams, and 2,900 million metric tons are still in use, including 700 million metric tons (8 per cent) that were recycled (IRP 2019; Geyer 2020). Across the plastics life cycle the largest losses to the environment occur during use and end-of-life, which account for approximately 36 per cent and 55 per cent, respectively (IRP 2019). Losses during plastic production account for only about 0.25 per cent of the total (Ryberg *et al.* 2019). Although some mismanaged plastic waste may be collected for reuse, or collected by street sweepers, citizens' groups and others, and reintroduced into landfills or dumps (Schneider *et al.* 2018) the amount of this waste is likely to be very small.

Marine litter and plastic pollution come mainly from land-based sources (UNEP 2018e; IRP 2019; van Truong *et al.* 2019) (Figure 4). These sources include agriculture (e.g. irrigation pipes, protective meshes, greenhouse covers, containers, fencing, pellets for the delivery of chemicals and fertilizers, seed coatings and mulching); building and construction (e.g. pipes, paints, flooring, roofing, insulants and sealants); transportation (e.g. abrasion of tyres, road surfaces and road markings); and a wide variety of personal care, pharmaceutical and healthcare products, including the personal protective equipment used during the COVID-19 pandemic (Adyel 2020). Approximately 36 per cent of all plastics produced are used in packaging, including single-use plastic products for food and beverage containers, approximately 85 per cent of which ends up in landfills or as unregulated waste and much of which

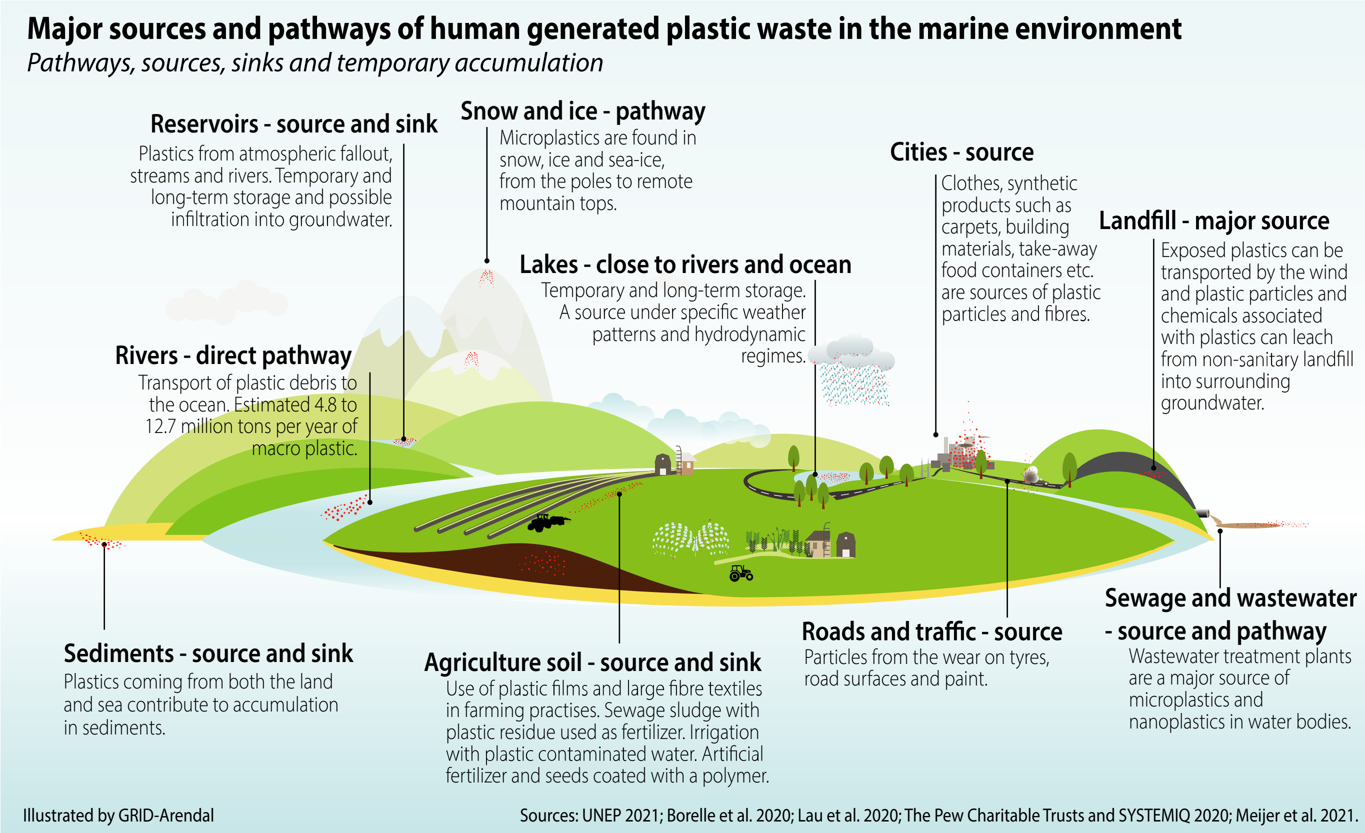


Figure 4: Major sources and pathways of human-generated plastic litter

will eventually enter the marine environment (Andrades *et al.* 2016). Despite changes in some countries' policies, the export of waste, including electronic waste, to countries with poor waste management infrastructure plays a major role in the generation of mismanaged waste and flows of litter and toxic chemicals into the oceans (Brooks *et al.* 2018; European Environment Agency 2019a; Awere *et al.* 2020) (see Section 4).

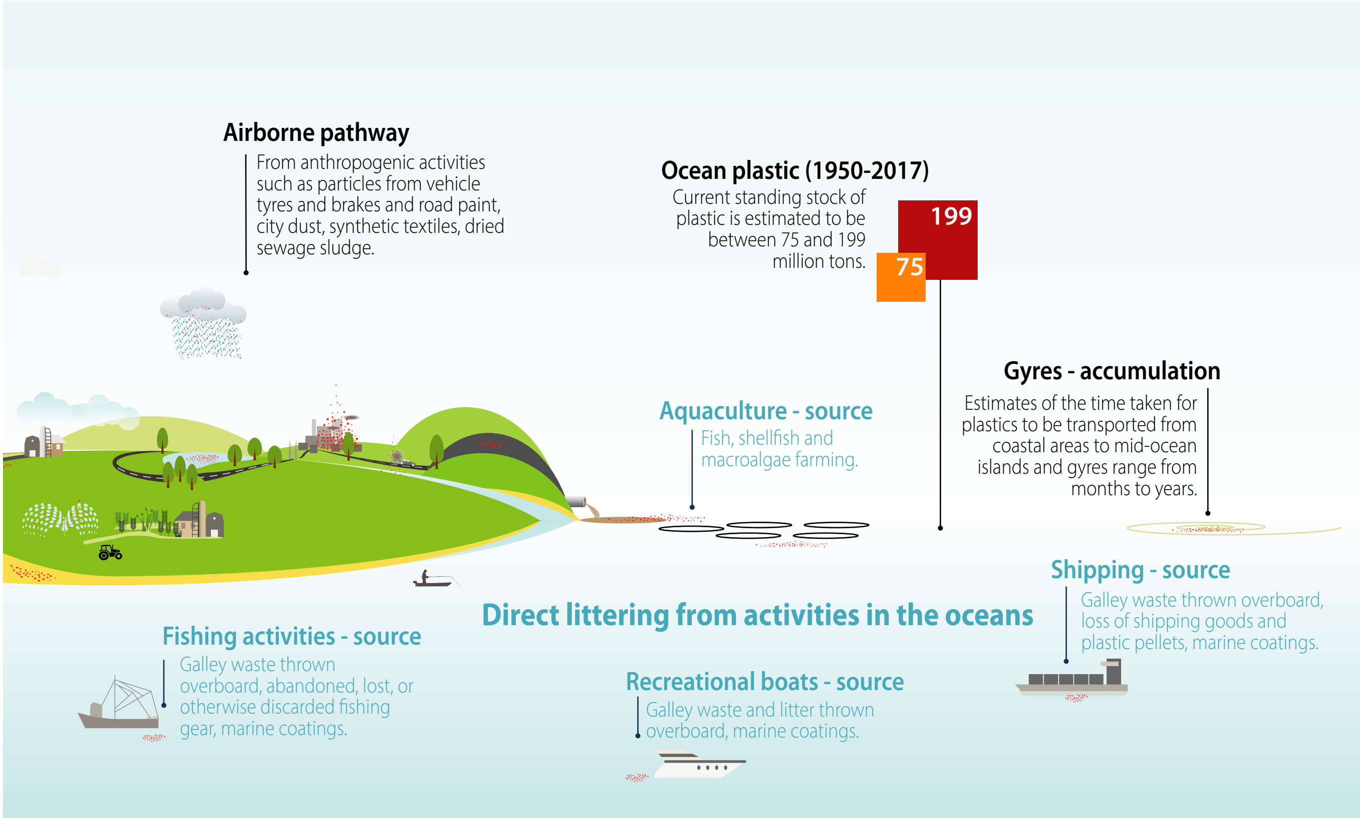
The volume of plastics in the oceans, which has been calculated by a number of researchers during the past five years or so, is estimated to be between 75 and 199 million metric tons (Jang *et al.* 2015; Ocean Conservancy and McKinsey Centre for Business and Environment 2015; Law 2017; IRP 2019; Lebreton *et al.* 2019; Borrelle *et al.* 2020; Lau *et al.* 2020; The Pew Charitable Trusts and SYSTEMIQ 2020).

Given the predicted increases in the production of plastics and in their future use, plastic litter and flows of plastics to marine environments from land-based sources are projected to continue to grow unless new governance and management structures are put in place. The types of structures and frameworks needed could include effective life cycle and end-of-life management strategies and corporate social responsibility (Borrelle *et al.* 2017; Haward 2018; Landon-Lane 2018; Borrelle *et al.* 2020; Maeland and Staupe-Delgado 2020; The Pew Charitable Trusts and SYSTEMIQ 2020) (Section 4).

There is as yet no consistent approach for estimating the volume of plastic waste flowing into the oceans. However, modelling analysis suggests that there are four main routes through which land-based primary macroplastic waste enters the oceans: uncollected waste directly dumped into water; uncollected waste dumped on land that makes its way to water; collected waste deposited in dumpsites that moves via land and air into water; and collected waste dumped directly into water by collection trucks (The Pew Charitable Trusts and SYSTEMIQ 2020). Based on these assumptions, The Pew Charitable Trusts and SYSTEMIQ (2020) estimate that 61 per cent of macroplastic leakage originates from uncollected waste, a share which could grow to 70 per cent by 2040 under a business-as-usual scenario as collection services fail to keep pace with macroplastic waste generation. Several calculations of annual flows of plastics into different aquatic ecosystems have been produced (Table 1).

Even with immediate and concerted action, Lau *et al.* (2020) estimate that 710 million metric tons of plastic waste would cumulatively enter aquatic and terrestrial ecosystems between 2016 and 2040. The scenarios of Lebreton and Andrady (2019) for the second half of the 21st century suggest that without a shift in waste generation and management, the mismanagement of waste from African (UNEP 2018c; UNEP Bamako 2020) and Asian watersheds would result in the release of millions of tons of litter and plastic waste into all the world's major terrestrial and aquatic ecosystems and eventually into the oceans.

Historical estimates of the volume and weight of plastics entering the ocean from land-based sources have generally relied on two indicators: waste generation per capita and the proportion of waste that is plastics. Based on per capita use of plastics and population density at a given location[5] and data on



country-specific waste generation and management (Hoornweg and Bhada-Tata 2012), Jambeck *et al.* (2015, 2018) and Lebreton and Andrady (2019) estimated that mismanaged plastic waste generated globally in 2010 amounted to 32 million metric tons and that the fraction of this waste consisting of plastics which reached the oceans from populations living within 50 km of the coastline amounted to between 4.8 million and 12.7 million metric tons. Thus, even in countries where there is low per capita use of plastics, waste volumes will build up if infrastructure is inadequate.

For the African continent, Jambeck *et al.* (2018) used the best available country-level data[6] to estimate that the total amount of mismanaged plastic waste was 4.4 million metric tons in 2010. They projected that this amount would increase to 10.5 million metric tons in 2025. Lebreton and Andrady (2019) updated figures for mismanaged and uncontrolled waste using country-level data on waste management, high resolution distribution, and long-term projections of populations and GDP growth. They estimated that 47 per cent of total annual municipal plastic waste generated globally (i.e. 60-90 million metric tons) was inadequately disposed of and likely to end up in the ocean.

With respect to microplastics (see Glossary), primary microplastics can be the result of leakage from production facilities and accidental losses of plastic pellets during transport (Karlsson *et al.* 2018) while secondary microplastics, produced when larger pieces of plastic break up or fragment, are found, for example, in leachates from landfill sites, biosludge from wastewater treatment plants, and agricultural run-off (Mason *et al.* 2016; Mahon *et al.* 2017; Li *et al.* 2018; Cowger *et al.* 2019; He *et al.* 2019; Sun *et al.* 2019) (Figure 5). Agricultural soils are now known to be sinks for microplastics as a result of intentional application of microplastic-coated seeds, chemicals and fertilizers (using new delivery technologies) intentional application of sewage sludge and effluents, plastic-coated seeds, chemicals and fertilizers (Nizzetto *et al.* 2016a; Nizzetto *et al.* 2016b; Piehl *et al.* 2018; Accinnelli *et al.* 2019; Corradini *et al.* 2019; Wang *et al.* 2019a; Wang *et al.* 2019b). These authors estimate that microplastic loadings to agricultural soils in Europe and North America represent a reservoir potentially larger than the marine environment. There is also evidence of microplastics uptake by edible plants (Conti *et al.* 2020; Li *et al.* 2020).

Estimated emissions of plastic waste (million metric tons per year)	Source-to-sea aspect	Projected emissions of plastic waste (million metric tons per year) under certain conditions	Approach used
19-23	Entered aquatic ecosystems in 2016	53 by 2030	Integrating expected population growth, annual waste generation per capita, the proportion of plastic in waste; incorporating an increase in plastic materials associated with predicted production increases, and the proportion of inadequately managed waste by country (Borelle et al. 2020)
9-14	Entered aquatic ecosystems in 2016	23-37 by 2040 (equivalent to 50 kg of plastic per metre of coastline worldwide)	Modelled stocks and flows of municipal solid waste and four sources of microplastics through the global plastic system, using five scenarios (2016–2040) and assuming no effective action is taken (Lau *et al.* 2020
0.8-2.7	Entered the oceans from global riverine systems in 2015	--	Based on >1,000 rivers, calibrated using field observations (Meijer *et al.* 2021)

Table 1: Estimates of global annual emissions of plastic waste (million metric tonnes) from land-based sources

2.1.2 Sea-based sources

Marine litter from sea-based activities arises from multiple sources (GESAMP 2015; GESAMP 2020b) (Figure 6). For example, all affordable, lightweight and durable maritime equipment is made of plastics. Major sea-based sources of plastics and microplastics include fisheries and aquaculture (e.g. sealants, storage boxes, packaging, buoys, ropes and lines, nets, various types of structures, and fishing gear such as fish aggregating devices or FADs) (FAO 2020); shipping and offshore operations (e.g. packaging, cargo, paints, end-of-life dismantling, ballast water); and ship-based tourism (e.g. packaging, personal goods). Ryan *et al.* (2019) observed that discarded plastic drinks bottles show the highest growth rate, increasing at 15 per cent per year compared with 7 per cent for other types of debris. Based on an analysis of bottle types and date of manufacture, they concluded that ships are responsible for most of the bottles floating in the central South Atlantic Ocean, in contravention of the International Convention for the Prevention of Pollution from Ships. This is consistent with the results of an internal investigation carried out by a fleet operator, which revealed that in one year crews on its 75 ships threw away more than 500,000 plastic drinks bottles (IMarEST 2019). [7]

Fisheries-related debris is the largest single category by volume found in beach litter. In Europe, based on numerous surveys, fisheries and aquaculture is estimated to contribute 39 per cent and 14 per cent of this debris, respectively; it consists of, for example, buoys, pots, feed sacks, gloves and boxes (Veiga *et al.* 2016; European Commission 2018a). The proportion of items on beaches from sea-based activities increases with stronger tides, suggesting that the share of litter in the water may be even higher (Unger and Harrison 2016). At sea 10 per cent of all floating debris is abandoned, lost or otherwise discarded fishing gear (ALDFG) (Stelfox *et al.* 2016); in the North Pacific Subtropical Gyre 46 per cent of this debris consists of fishing nets (Lebreton 2018). The contents of fishing nets in the western Atlantic and the Baltic Sea indicate that there are an equal number of items made of unnecessary, avoidable and problematic plastic polymers and ALDFG, whereas the majority of plastics found in Arctic waters come primarily from fishing (Veiga *et al.* 2016;

Vlachogianni *et al.* 2017; Fleet *et al.* 2021). In the area around Svalbard, Norway most of the marine litter analysed is associated with fishing-related activities (Nashoug 2017). The dominance of fishing-related objects is relatively unique to the northern parts of Norway, the Barents Sea region and the Arctic. In more southern areas household-related objects are the dominant type of plastics in marine litter (Nashoug 2017). Surveys in areas close to shore with high concentrations of fisheries and aquaculture show significant concentrations of plastics in the form of cages, long lines, poles, and other floating and fixed structures used for the culture of marine animals and plants.

There are no reliable estimates of the contribution of aquaculture to marine litter; however, aquaculture operators are likely to take considerable care to avoid losses. The material lost depends on culture systems, construction quality, vulnerability to damage, and management practices. It could include nets and cage structures (for marine fish cages), lines and floating raft structures (for seaweed systems), and poles, bags, lines and plastic sheeting (for mollusc farming). Degradation of polymer ropes also leads to plastics being released in sublittoral environments (Welden and Cowie 2017).

On beaches along the coastline of the Adriatic and Ionian Seas mussel nets were the seventh most frequent items found (Vlachogianni *et al.* 2017; Fleet *et al.* 2021), while in sea floor surveys litter from aquaculture accounted for 15 per cent of the items recorded (Spedicato *et al.* 2019). Statistics from the PRODCOM database[8] indicate that the contribution of fishing gear and aquaculture to waste and marine litter (netting and non-netting) in European waters is 11,000 metric tons per year (Unger and Harrison 2016; European Commission 2018a;

Ingeborg and Gabrielson 2018) compared to 15,604 metric tons per year from single-use plastics.

Other sea-based sources of marine litter include abandoned and end-of-life vessels and recreational boats, especially those made of fibreglass (IMO 2019). A major source of plastic contamination in some coastal areas is shipbreaking (Science for Environment

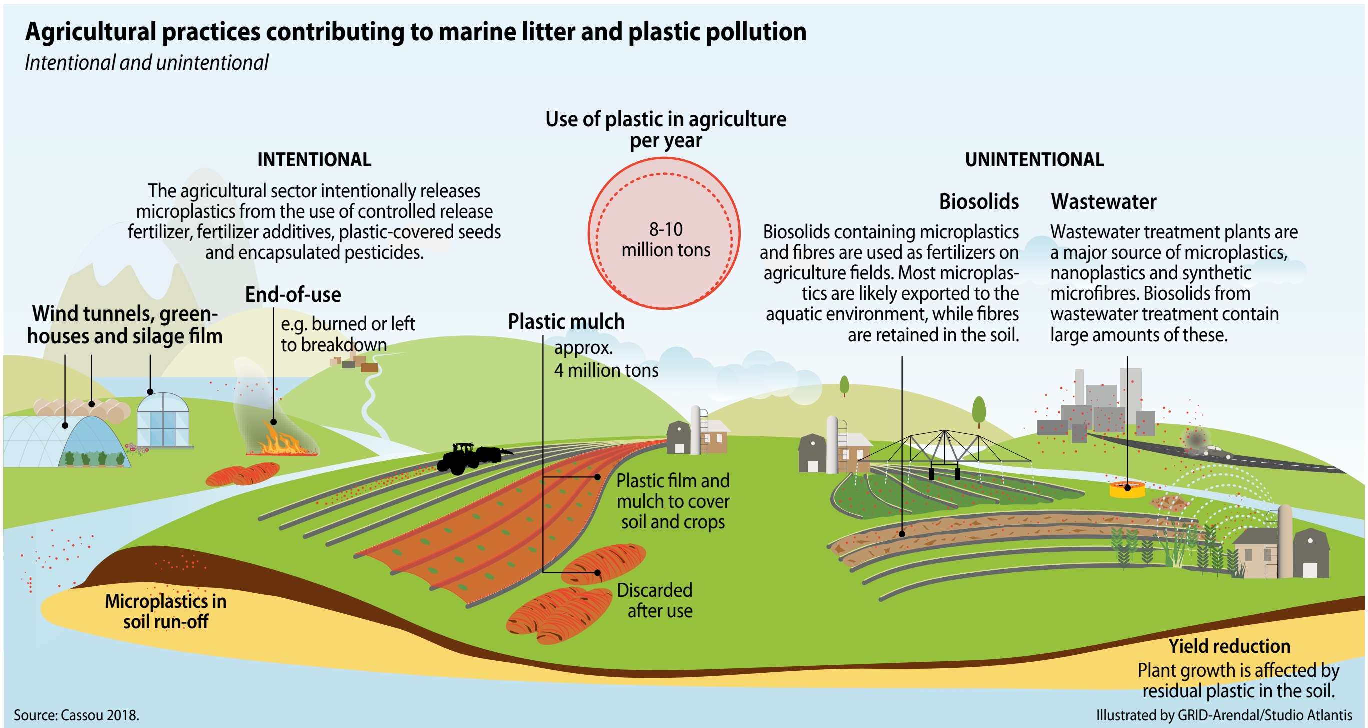


Figure 5: Agricultural practices contributing to marine litter and plastic pollution

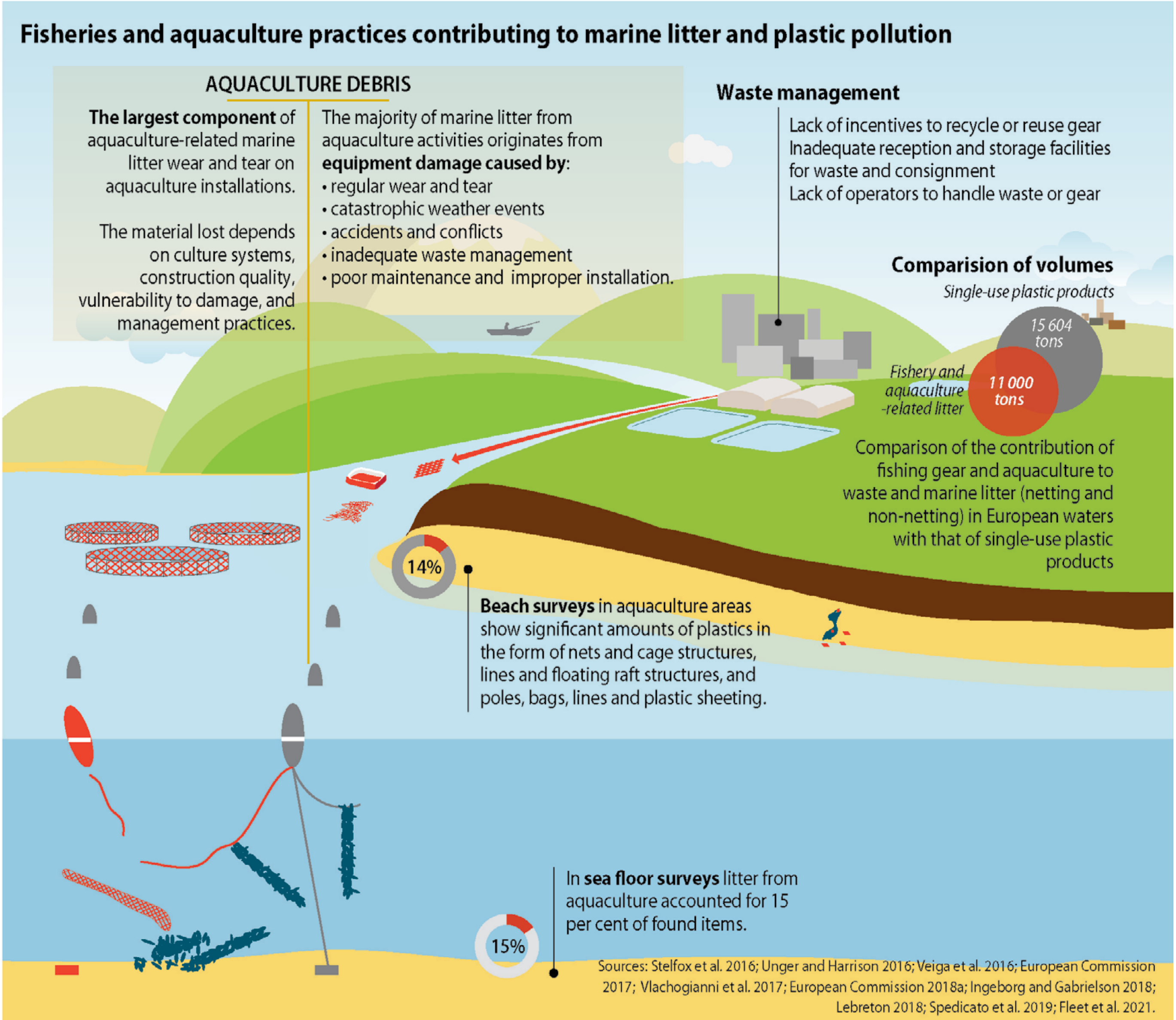


Figure 6a: Fisheries and aquaculture practices contributing to marine litter and plastic pollution

Policy 2016). In a study of the abundance of small pieces of plastic in a shipyard in India, the authors found on average 81 mg of small plastic fragments per kg of sediment, which they reported was the direct result of shipbreaking (Reddy *et al.* 2006). A yacht's average lifespan has been estimated to be 30 years, although in some instances it may stretch to 40-45 years. This lifespan has been increasing over time with the use of stronger materials such as fibre-reinforced polymers. It is thought that 1 to 2 per cent of the 6 million boats maintained in Europe (i.e. at least 80,000) reach their end-of-life each year. However, only around 2,000 are dismantled (European Commission 2017). A significant share of the remainder are abandoned, potentially ending up in the oceans and becoming marine litter.

Primary microplastics from sea-based sources can enter the oceans directly from accidental loss of cargo at sea and from illegal dumping of waste, as well as via paints and other materials such as sealants used in various industries. Secondary microplastics may arise from wear and tear on fishing gear, such as polypropylene ropes, and from aquaculture operations. Coastal and sea-based tourism is another source of plastic waste through intentional or accidental littering of shorelines (European Commission ARCADIS 2014). Once larger pieces of plastic are present in the marine environment, they may be broken down into secondary microplastics through mechanical, chemical or biological processes. In addition to microorganisms, biological breakdown includes the activity of marine organisms fragmenting items such as plastic bags (Hodgson *et al.* 2018), hard plastic trays, and polystyrene foam packaging (Jang *et al.* 2016) into microplastics. There is emerging evidence that some marine organisms, such as krill, can also reduce microplastic particles to nanoplastics through ingestion (Dawson *et al.* 2018). The interactions between fauna and microplastic production require further investigation, as they need to be considered when the movement of microplastics through ecosystems is modelled.

In a worldwide survey of seafloor litter, Canals *et al.* (2021) conclude that an important factor influencing the composition of benthic litter is the type of activities carried out in the vicinity. For example, analyses of sources in relation to benthic litter in the southern North Sea indicated the importance of ship-based litter compared to land-based litter, as is the case in the Mediterranean (Galimany *et al.* 2019). Deep-sea litter in the Indian Ocean is dominated by abandoned, lost or otherwise discarded fishing gear (ALDFG), whereas that in the Atlantic Ocean has been found to be a general mix of refuse (Woodall *et al.* 2015). The largest component of marine litter associated with fisheries and aquaculture is ALDFG and wear and tear on aquaculture installations. Examples of the reasons for discharging litter at sea include accidental and sometimes irretrievable loss of discarded fishing gear; the limited life span of some items used at sea; waste mismanagement (e.g. dumping at sea because of the high cost of waste handling in ports); inadequate facilities for waste handling at sea; inadequate reception and storage facilities for waste and consignment; lack of operators to handle waste or gear; and lack of incentives to recycle or reuse gear. In the revision of the EU Directive on port reception facilities (Ecorys 2017; European Commission 2018c; European Union 2019c) it was noted that up to 30 per cent of waste, including that from fishing vessels and recreational craft, which should be delivered to ports is not so delivered; instead, it potentially ends up being discharged at sea. There is no evidence that dumping of litter from ships at sea has decreased.

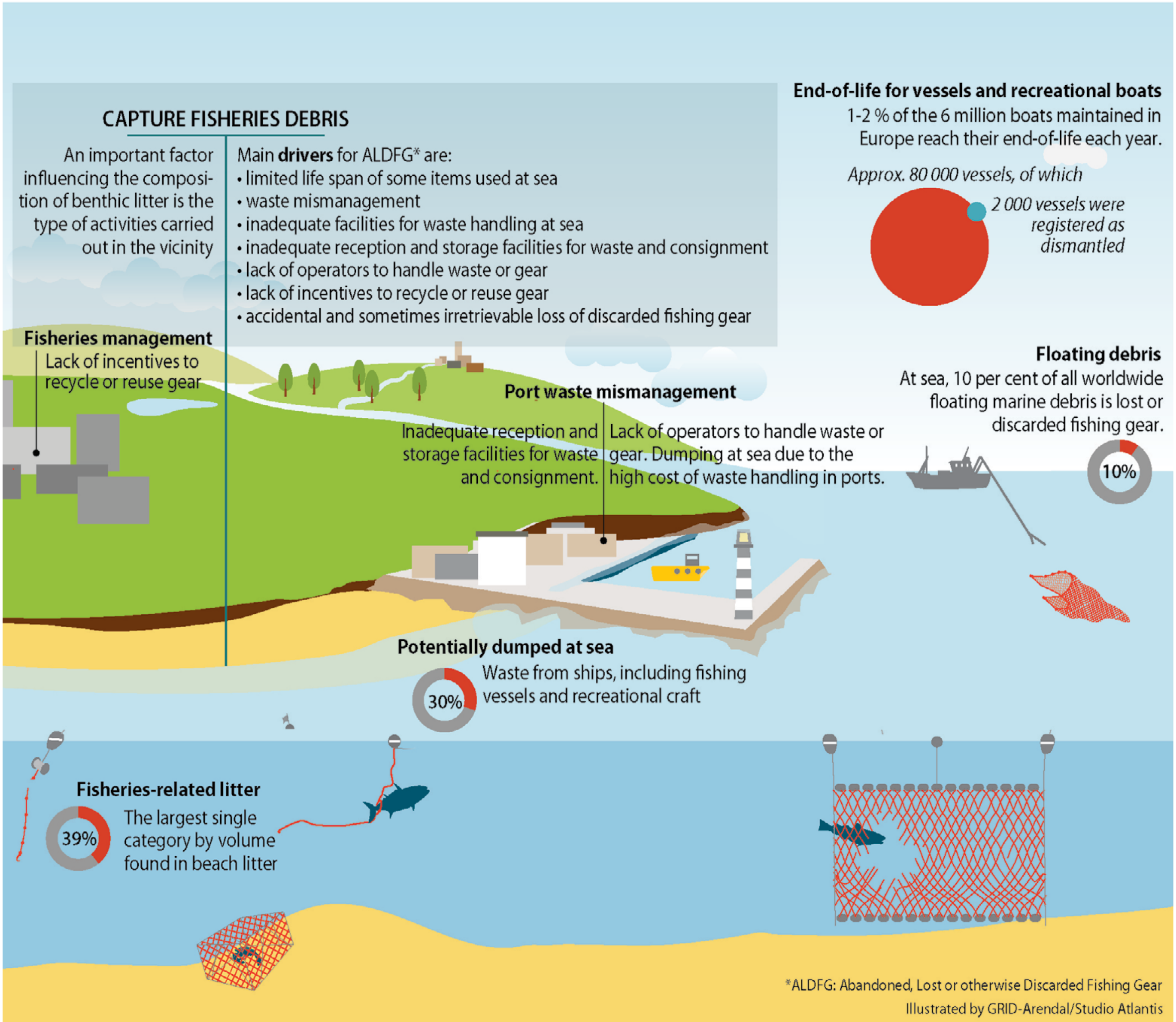


Figure 6b: Fisheries and aquaculture practices contributing to marine litter and plastic pollution

2.2 Major pathways of litter and plastic pollution

Marine litter and plastic pollution enter the marine environment along multiple pathways, including run-off over land, riverine flows, wastewater and greywater flows, airborne transport, and direct entry from ocean sources such as fisheries and maritime shipping (e.g. Alomar *et al.* 2016; Nizzetto *et al.* 2016a; Nizzetto *et al.* 2016b; Auta *et al.* 2017; Lebreton *et al.* 2017; Alimi *et al.* 2018; Horton and Dixon 2018; Best 2019; Akarsu *et al.* 2020; Chen *et al.* 2020; Birch *et al.* 2020; Peng, L. *et al.* 2020). Major events such as storms and tsunamis can also deliver significant volumes of plastic debris and microplastics via urban storm waters (Werbowski *et al.* 2021) and from damage to coastal infrastructure (NOAA 2015; Murray *et al.* 2018; GESAMP 2019). Some plastics, such as single-use items, which have been littered or washed down drains can also be transported via multiple pathways including wind, rivers and sewerage systems or enter the oceans directly. These pathways are closely connected. For example, similarities have been observed between the composition of riverine and beach litter, while an analysis of floating macrolitter from 52 rivers and on marine beaches found a significant overlap among 8,599 items (Gonzalez *et al.* 2016).

The movement of microplastics along different freshwater pathways into the marine environment is more difficult to monitor (e.g. Lebreton *et al.* 2017; Alimi *et al.* 2018; Redondo-Hasselerharm *et al.* 2020). Although the overwhelming majority of flows of microplastics come from land-based activities, sea-based activities such as fisheries, aquaculture and cruise ships also generate microplastics from discarded waste (Boucher and Friot 2017; Lebreton *et al.* 2017). The largest share of these particles is estimated to derive from the laundering and use of synthetic textiles and abrasion of tyres while driving (Boucher and Friot 2017). Using these results, recent modelling of four major sources of microplastics (tyres, plastic pellets, fibres and microplastics in personal care products) indicates that tyre dust from roadways and runways contributes 78 per cent of leakage by mass, plastic pellets 17 per cent, and textiles and personal care products 4 per cent combined (Lau *et al.* 2020). The microplastic sources analysed represent about 60 per cent of total leakage from land in high-income countries, where the per capita release rate is estimated to be three times higher than in middle- and low-income countries (The Pew Charitable Trusts and SYSTEMIQ 2020). Model-based analysis of the global release of microplastics into the marine environment showed that 44 per cent were from the road run-off pathway and 37 per cent from the wastewater pathway, 15 per cent were transported by wind, and 4 per cent were direct releases into the oceans.

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Absolute releases per region ranged from 134,000 to 281,000 metric tons per year, translating into a per capita release of microplastics globally of 110 to 750 grams per person per year.

The rates at which litter and plastic pollution move along the various transport pathways, or reside in different compartments of the marine environment, depend upon their chemical and physical properties such as buoyancy, surface properties and size (Box 5), as well as on oceanographic processes and meteorological conditions including storms (see the sections below). Lebreton *et al.* (2019) concluded that there is a significant time interval, in the order of several years to decades, between terrestrial emissions and accumulation in offshore waters, suggesting that the current generation of microplastics in the ocean is the result of aging and degradation of objects produced in the 1990s and earlier (Kedzierski *et al.* 2018).

2.2.1 Rivers and sedimentary pathways

Riverine waters and sediments are a major pathway for marine litter (van der Wal *et al.* 2013; Jambeck *et al.* 2015; Jambeck *et al.* 2018; Best 2019; van Calcar and van Emmerik 2019; van Emmerik *et al.* 2019; Borrelle *et al.* 2020; González-Fernández *et al.* 2021; Meijer *et al.* 2021). Earlier estimates of riverine inputs from mismanaged solid waste set the level at 4.8 and 12.7 million metric tons per year (Jambeck *et al.* 2015), while Schmidt *et al.* (2017) estimated that 95 per cent of marine plastic comes from just 10 rivers out of 57 river systems. Lebreton *et al.* (2018) in their model estimated that 67 per cent of all marine plastic comes from 20 rivers, mostly in Asia. Other published estimates have been linked to population centres (Eerkes-Medrano *et al.* 2015; Jambeck *et al.* 2015; Peters and Bratton 2016; Horton *et al.* 2017; Tibbetts *et al.* 2018). A European database of riverine floating macrolitter indicates that between 307 and 925 million litter items are released annually from Europe into the ocean, and that a major portion is routed through small-sized drainage basins (<100 km2) (González-Fernández *et al.* 2021).
A recent estimate, based on models and field observations, is that 80 per cent of plastic emissions to the oceans from riverine systems (i.e. 0.8-2.7 million metric tons) come from over 1,000 of the world's rivers (Meijer *et al.* 2021).

Riverine inputs arise from plastics mishandled during manufacture and use, which are found in wastewater treatment plant effluents (Horten *et al.* 2017a; Horten *et al.* 2017b; Alimi *et al.* 2018; Gavigan *et al.* 2020). Estimates of concentrations of plastics in freshwater and river sediments are similar to those of concentrations on marine shorelines, although variations occur in relation to proximity to urban or industrial sites or the presence of wastewater treatment plants (e.g. Browne *et al.* 2011; Klein *et al.* 2015; Alimi *et al.* 2018).

Predicting total emissions of litter, including plastic waste, from rivers is challenging, given the under-representation of litter and plastic pollution studies in freshwater environments and variability in monitoring techniques (Eerkes-Medrano *et al.* 2015; Blettler *et al.* 2018; van Emmerik *et al.* 2018; Blettler *et al.* 2019; Redondo-Hasselerharm *et al.* 2020). Many factors associated with river morphology, such as bottom type and curvature, can create internal river turbulences at different scales, wave action and mixing in the water column, while the presence of dams will also determine the behaviour of litter and microplastics in the river and its catchment area (Hoellein *et al.* 2014; Zhang *et al.* 2015). Most important is to recognize that sediments can act as sources, sinks and pathways (Mani and Burkhardt-Holm 2019). Pulsed or accidental releases have been identified as a primary source of peak loading events (Lechner *et al.* 2014). Periods of high flow can re-suspend particles within sediments and deposit them downstream. Stretches with settled flow are likely to show a pronounced stratification of plastic particles throughout the water column, whereas at lower flow rates more plastic is likely to be found either floating on the river surface or close to a riverbank. Flooding of catchment areas can also disperse microplastic contamination of riverbeds. Hurley *et al.* (2018) showed that flooding across catchments in the United Kingdom decreased plastic concentrations along riverbanks by 70 per cent.

Studies on the flows of litter in some larger rivers, such as the Chicago, Rhine-Main, Danube and Thames, underscore the extent and volumes of plastic litter that eventually accumulates in estuaries and along shorelines (Lechner *et al.* 2014; McCormick *et al.* 2014; Morritt *et al.* 2014; Klein *et al.* 2015; Alimi *et al.* 2018). However, riverine inputs of microplastics are more difficult to quantify as the majority of freshwater microplastic studies have been conducted only at a small number of sites and rarely over entire river catchments, and it is often the smaller drains that could be contributing more overall (Stanton *et al.* 2019a). Several studies are currently under way focusing on monitoring plastics in rivers in Southeast Asia and India through the CounterMEASURE Project.[9] UNEP has developed guidelines for the harmonization of methodologies for monitoring plastics in rivers and lakes (UNEP 2020b,c,d) to complement the GESAMP *Guidelines for the Monitoring and Assessment of Plastic Litter in the Ocean* (GESAMP 2019).

Wastewater treatment plant effluents are an important source of microplastics in riverine inputs to the ocean (Murphy *et al.* 2016; Mintenig *et al.* 2017; Talvitie *et al.* 2017; Ziajahromi *et al.* 2017; van Emmerik *et al.* 2019; Birch *et al.* 2020). The most abundant microplastic particles are synthetic fibres which come from the washing of synthetic textiles and are then concentrated in sewage sludge or discharged in wastewater treatment effluents (Browne *et al.* 2011; Gavigan *et al.* 2020). McCormick *et al.* (2014) found a 10-fold increase in plastic fibres downstream of a wastewater treatment plant in the Chicago River despite the fact that 95-99 per cent of plastics had been separated out into biosolids (Murphy *et al.* 2016; Talvitie *et al.* 2017). Personal care products are also a significant source of microplastics. Globally, about 1,500 metric tons of microplastics per year from personal care products are estimated to escape from wastewater treatment plants into aquatic environments (Sun *et al.* 2020). Overall efficiency in removing microfibres and microplastics depends on the individual wastewater treatment plant and hence very much on the local context. For example, in wastewater treatment plants with tertiary treatment as little

Box 5: Properties and processes affecting the transport and degradation of plastics in the marine environment

The different types of polymers used in plastics have a wide range of properties which affect their behaviour in different environments. These properties include density and buoyancy hydrophobic/hydrophilic properties, and propensity towards biofilm formation and biodegradability. In the marine environment one of the most important factors is the density of the plastic relative to that of seawater. The densities of common plastics range from 0.90 to 1.39 kg m^{-3}, compared to freshwater, which has a density of 1.0 for pure water, and seawater, which has a density ranging from 1.020 to 1.029 kg m^{-3}. Polyethylene, mainly LDPE, can have a density below 1 kg m^{-3}, so that it can float in fresh and marine water. This is one reason it is one of the types of plastic most commonly found when sampling surface waters. Polyethylene oxide (polyethylene glycol) (PEO) and polypropylene (PP) would be expected to float in freshwater, and expanded polystyrene (EPS) in seawater. Buoyancy is also affected by trapped air, water currents and turbulence, which explains why drinks bottles made of polyethylene terephthalate (PET) (1.34-1.39 kg m^{-3}) are commonly found both floating in coastal waters and deposited on the seabed. The buoyancy of plastic polymers can also be affected by the presence of biofilm on the surface (Napper and Thompson 2019). All of these considerations determine the depth profile of plastics in the ocean (Kooi *et al.* 2016).

Plastics tend to degrade and start to lose their original properties at a rate depending on the physical, chemical and biological conditions to which they are exposed. The degradation of plastics can be divided into six processes: thermal degradation, hydrolysis, mechanical/physical degradation, thermo-oxidative degradation, photodegradation and biodegradation (Mattsson *et al.* 2015). The main degradation processes at sea are hydrolysis, a bond-breaking reaction brought about by the addition of water that has been shown to contribute to the degradation of plastic marine debris; mechanical or physical degradation caused by waves and friction; thermo-oxidative degradation, a slow oxidative breakdown at moderate temperatures; photodegradation, brought about by sunlight but severely retarded in seawater; and biodegradation, whereby living organisms, usually microbes (such as bacteria), break down organic substances and alter the surface chemistry, which can then change due to oxidation and photodegradation.

Plastic degradation by exposure to UV light (photodegradation) results from the weakening and eventual breaking of covalent bonds within the structure of the plastic polymers, known as chain scission (Gewert *et al.* 2015). The chain scission can occur at any point within a polymer's structure, with the potential to cleave monomers from the inert polymer; some of these polymers may be hazardous, such as persistent organic and bioaccumulative pollutants which can themselves cause environmental harm (Lithner *et al.* 2011). Overall, degradation is generally slower in aquatic environments compared to land and may not even occur in environments with limited exposure such as in pelagic (surface waters and the water column) and benthic (sedimentary) environments (Webb *et al.* 2013). Biodegradable plastics that do not degrade fragment into microplastic particles in much the same way as conventional plastics (Napper and Thompson 2019).

Particle size is another important factor in regard to transport as well as detection (Barnes *et al.* 2009). Particles generally have a slow rate of degradation in seawater (Dussud *et al.* 2018a; Dussud *et al.* 2018b). Levels of microplastics in seawater and in freshwater were likely underestimated in the 2016 UNEP report, *Marine Plastic Debris and Microplastics – Global Lessons and Research to Inspire Action and Guide Policy Change* (UNEP 2016). Significant uncertainty remains as to the concentrations of nanoplastics in seawater.

as 0.1 per cent of incoming microplastics and microfibres can be released in the effluent water (Carr *et al.* 2016). However, a large fraction of microplastics can be trapped in biosolids. It is estimated that 520,000 metric tons per year of plastic waste is released in wastewater and greywater effluents in Europe alone (Horton *et al.* 2017a; Horten *et al.* 2017b). One key concern is pollution of soils by microplastics, for example through the application of biosolids and sludge to agricultural land, as this will clearly be a source of microplastics and associated chemicals in crops and run-off into rivers (Carr *et al.* 2016; Mahon *et al.* 2017; Hurley and Ho 2018; Li *et al.* 2018; Liu *et al.* 2018; SAPEA 2019). Soil-borne plastic litter, including agricultural and packaging films and microplastics in fertilizers, enters the marine environment through precipitation run-off and tidal washing (Horton *et al.* 2017a; Horton *et al.* 2017b; Ng *et al.* 2018). Depending on their polymer structure, these films start to degrade after eight to 12 months (Niaounakis *et al.* 2019).

2.2.2 Freshwater lakes, reservoirs, groundwater and drinking water supplies

Large lakes and reservoirs can act as temporary and long-term sinks of microplastics and as hotspots for plastic pollution (Eriksen *et al.* 2013; Hoellein *et al.* 2014; Driedger *et al.* 2015; Zhang *et al.* 2015; Zhang *et al.* 2016), as can smaller lakes and ponds and urban retention areas (Faure *et al.* 2015; Vaughan *et al.* 2017; Gilbreath *et al.* 2019). Making precise estimates of plastic loading in lakes is difficult because sampling is generally undertaken at the surface, while large concentrations of microplastics can also exist below the surface and in sediments (McCormick *et al.* 2014; Zhao *et al.* 2014; Imhof *et al.* 2018). Driedger *et al.* (2015) observed that surface water densities of plastics in certain areas of the Laurentian Great Lakes were as high as those reported for areas of litter accumulation within ocean gyres. Transport in lakes is driven not only by currents, similarly to rivers and streams, but also by wind patterns that can produce areas of seasonally high localized concentrations (Dris *et al.* 2015b). Hoffman and Hittinger (2017) estimated that 10,000 tons of plastics per year were introduced into the Great Lakes.

Far less is known about the processes and levels of the infiltration of microplastics into groundwater. Panno *et al.* (2019) reported on microplastics in karst groundwater systems (karst systems constitute one-quarter of the world's drinking water sources), which presented a median of 6.4 microfibres per litre. Studies of drinking water from a number of groundwater sources (Koelmans *et al.* 2019; WHO 2019) show the presence of microplastic particles, ranging from 0.0 to 6,292 particles per litre (Oßmann *et al.* 2018; Strand *et al.* 2018). However, questions have been raised regarding the methods used to quantify microplastic particles in these drinking water samples as there are no standard sampling extraction or identification methods for microplastic quantification (Koelmans *et al.* 2019; Stanton *et al.* 2019a; Stanton *et al.* 2019b). A number of studies have identified microplastic particles in bottled water (Mason *et al.* 2018; Welle 2018), showing low concentrations (e.g. 14 ± 14 particles per litre, Schymanski *et al.* 2018) and leading Welle (2018) to conclude that the reported amounts do not raise safety concerns. However, there are too few studies to obtain a comprehensive understanding of the fluxes of microplastics into the ocean from large reservoirs and drinking water supplies (Oßmann *et al.* 2018; Schymanski *et al.* 2018) or to properly inform human health risk assessments (Koelmans *et al.* 2019).

2.2.3 Atmospheric transport, including snow and ice

Long-range atmospheric transport of heavy metals and organic pollutants, such as polychlorobiphenyls, DDT, dieldrin and phthalate ester plasticizers, even to marine areas remote from industrial and human activity, has been recorded for several decades (e.g. Atlas and Giam 1981). Atmospheric transport of microplastics and airborne impacts are also critical for the marine environment (Evangeliou *et al.* 2020; Prata *et al.* 2021). Microplastics have even been observed in Arctic snow and sea ice (Bergmann *et al.* 2019). Microplastics in snow from ice floes in the Arctic (Obbard *et al.* 2014; Kanhai *et al.* 2018; Peeken *et al.* 2018; Bergmann *et al.* 2019; Kanhai *et al.* 2020), while lower in concentration than snow samples from the European Alps and urban areas, is still sufficient for atmospheric transport and deposition to be recognized as notable pathways for microplastics. Together the studies of Obbard *et al.* (2014), Peeken *et al.* (2018) and Kanhai *et al.* (2020) indicate that sea ice functions as a temporary sink, secondary source and transport medium for microplastics in the Arctic Ocean. The polymer composition in Arctic snow was dominated by varnish, rubber, polyethylene and polyamide. Sea ice can also act as a temporary sink for particles (Peeken *et al.* 2018), as well as a potential source of historic microplastic pollution released as sea ice melts (e.g. Obbard *et al.* 2014; Suaria *et al.* 2020).

Microplastics have been found in remote mountain catchments (Allen *et al.* 2019), in settled snow from different locations (Bergmann *et al.* 2019), and as atmospheric deposition in urban areas (Dris *et al.* 2016; WHO 2016; Cai *et al.* 2017; Bergmann *et al.* 2019; Klein and Fischer 2019; Stanton *et al.* 2019b). Microplastics are released to the air from numerous sources, including washing and wearing of synthetic textiles, abrasion of materials (e.g. tyres, building materials) and resuspension of microplastics on surfaces. Understanding the entrainment of microplastics into and their transport through the atmosphere is challenging given the variety of shapes, sizes and densities of microplastic particles. Different estimations are dependent on sampling methodologies, as well as on space use factors; particle properties, such as size and density, will influence their deposition on the respiratory system, with less dense and smaller particles reaching deeper in the lungs (Wright *et al.* 2019).

One of the first determinations of microplastics in the air refers to outdoor concentrations of 0.3-1.5 particles m^3 and indoor concentrations of 0.4-56.5 particles m^3 (33 per cent polymers), including inhalable sizes (Dris *et al.* 2017). Individual inhalation has been estimated to be 26-130 airborne microplastics per day (Prata 2018). Based on air sampling using a mannequin, it is expected that a male person with light activity inhales 272 microplastics per day (Vianello *et al.* 2019). However, earlier findings showed that particles with aerodynamic diameters <10 µm do not remain airborne for long, while the airborne residence times of particles with an aerodynamic diameter of 1-10 µm can be as low as 10-100 hours (Esmen and Corn 1971; Whelpdale 1974) and those of sea salt particles >50 µm can be even shorter (Athanasopoulou *et al.* 2008; Evangeliou *et al.* 2020).

Apart from issues relating to sample location and collection methods, the presence and residence times of airborne microplastics mean that atmospheric transport pathways to the marine environment need to be considered (Prata 2018; Prata *et al.* 2020).

2.2.4 Marine pathways

All the major factors affecting the transport of marine litter and plastic pollution (Figure 7) have been studied extensively (Ye and Andrady 1991; Kukulka *et al.* 2012; Chubarenko *et al.* 2016; Fazey and Ryan 2016; Pedrotti *et al.* 2016; Zhang 2017; Alimi *et al.* 2018; Chubarenko *et al.* 2018; Lebreton *et al.* 2018; Castro-Jiménez *et al.* 2019; Lebreton *et al.* 2019; Peng, G. *et al.* 2020; van Sebille *et al.* 2020).

The transport of litter and plastics in the marine environment is controlled by ocean currents, waves and winds. In the open

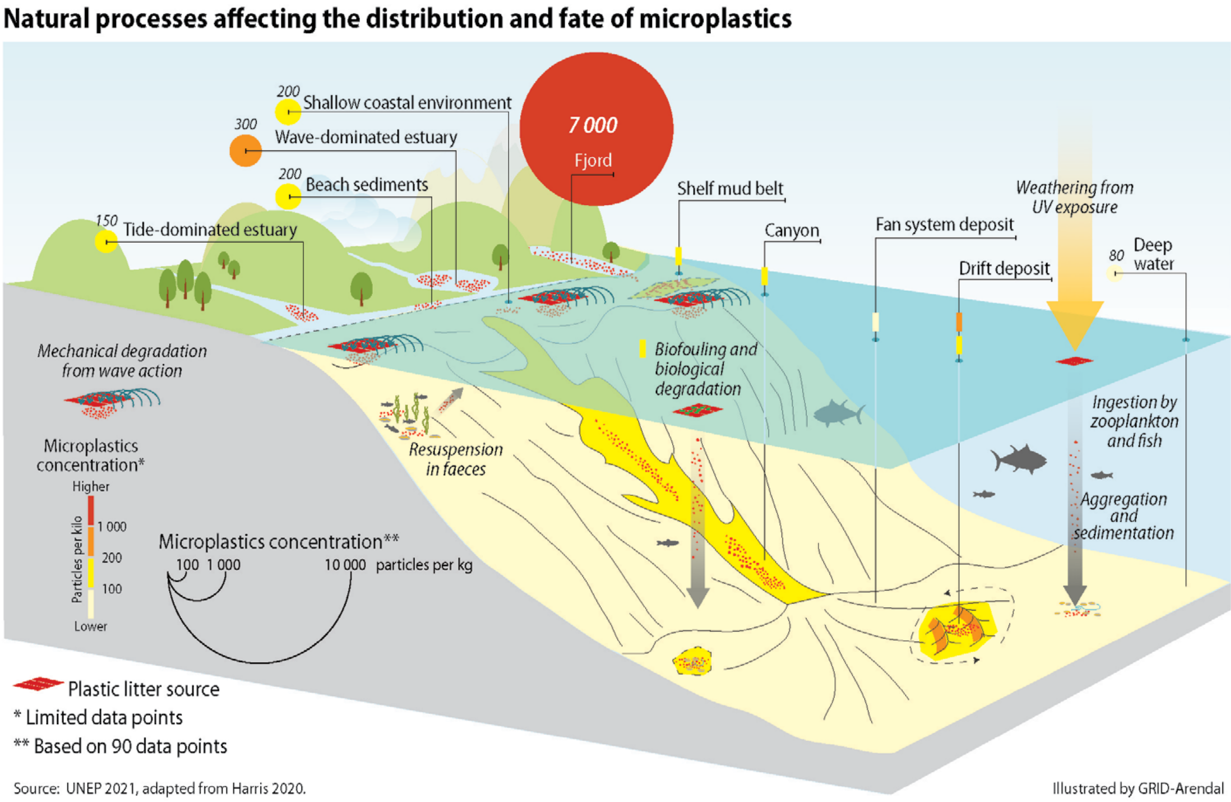

Figure 7: Natural processes affecting the distribution and fate of microplastics

ocean large-scale currents work together with eddies, as well as multi-scale convergent fronts, to move debris and litter around (Onink *et al.* 2019). In coastal areas tides are important; they interact with shoreline characteristics such as shape, morphology, coastal vegetation, bioturbation, and surface ice, terrain and slope to move debris and litter on and off beaches. Movement in the water depends on its chemical composition, surface charge, hydrophobicity, density, size and shape, which may be modified by biological interactions (e.g. biofouling and ingestion by marine animals).

There is, however, a debate concerning whether observations of marine plastics at sea, along pathways and into sinks (Lebreton *et al.* 2019; Ostle *et al.* 2019; Marimenko *et al.* 2019) are keeping pace with the rate of global plastic production (Goldstein *et al.* 2012; Geyer *et al.* 2017). A far better understanding of how these factors interact is needed in order to quantify and close the gap in the global inventory of marine plastics (van Sebille *et al.* 2020), which is overwhelmingly model-based (Galgani *et al.* 2021).

In a review of plastic fluxes, pathways and fate, van Sebille *et al.* (2020) conclude that discrepancies can arise because of the time delay between fluxes into the ocean and arrival in the regions where most measurements are taken (Lebreton *et al.* 2019). The differences in sinking and floating processes cause low-density particles to stay on the water surface and travel long distances; for example, foamed polystyrene particles cross the Baltic Sea (approximately 250 km) in two to three days with quite moderate winds of 10 m s -1, whereas heavier particles settle through 250 metres of water column in less than 18 hours (Chubarenko *et al.* 2016). However, there are many other physical processes that may account for the difference between estimates of plastic inputs and the pool of floating plastics at sea, including beaching, sedimentation and fragmentation to sizes that have not been measured (van Sebille *et al.* 2020). There is evidence that the size and composition of large debris changes with distance from major land-based sources (Ryan 2015), possibly as a result of these mechanisms. Biological processes (e.g. ingestion or settlement) may also aid the (horizontal and vertical) transport of plastics within the oceans.

To date, marine litter and plastic pollution have been studied and reported in all the major oceans, including the Pacific (e.g. Goldstein *et al.* 2013; Desforges *et al.* 2015; Lebreton *et al.* 2018; Choy *et al.* 2019), the Arctic and the seas around Antarctica (Obbard *et al.* 2014; Lusher *et al.* 2015; Amélineau *et al.* 2016; Isobe *et al.* 2017; Hallanger and Gabrielsen 2018; Kanhai *et al.* 2018; Kühn *et al.* 2018; Obbard 2018; Peeken *et al.* 2018; Kanhai *et al.* 2019; Mu *et al.* 2019; Kanhai *et al.* 2020; Tirelli *et al.* 2020), the Atlantic (e.g. Kanhai *et al.* 2017; Fossatti *et al.* 2020; Pabortsava and Lampitt 2020) and the Indian Ocean (e.g. Imhof *et al.* 2017; van der Mheen *et al.* 2019). Despite the number of studies, there are no standardized sampling protocols (e.g. units or abundance include numbers per unit area, per unit volume, per unit mass or per sampling site) (e.g. Ostle *et al.* 2019; Peng, G. *et al.* 2020). Generally, data coverage in most open ocean areas is sparse and the gaps are too large to allow direct derivation of the spatial patterns.

In a recent review of seabed litter, Canals *et al.* (2021) concluded that although plastic items on the seafloor can be assumed to be increasing continuously, they are the least investigated fraction of marine litter, which is not surprising since most of them lie in the least explored ecosystem. Consequently, there are significant knowledge gaps that need to be tackled and data needs for modelling, comparability and harmonization. Better understanding of seafloor macrolitter can inform international protection and conservation frameworks that prioritize efforts and measures to combat marine litter and its deleterious impacts (Canals *et al.* 2021).

Storms and other extreme events can result in coastal flooding, increased run-off, and the release of large volumes of debris into the ocean. They may also generate large volumes of marine litter. For example, the floating debris produced by the 2011 tsunami in Japan was comparable to the annual amount in the entire North Pacific and, a year later, showed up in the flux of debris on shorelines in the western United States with an order of magnitude increase in all categories (Murray *et al.* 2018). Climate change affects the distribution of microplastics due to changes in ocean circulation and surface winds (Welden and Lusher 2017). However, there are very few measurements of the rates of transfer of plastics between compartments or their accumulation in different media, making it extremely difficult to estimate the absolute impact of marine debris and plastics.

Beaches and coastal ecosystems

Shore deposition of plastics is part of an important transport pathway that has been studied since the 1970s, when surveys began to confirm that significant plastic loads were accumulating on the shores of even the remotest beaches (Lavers and Bond 2017). Marine debris and microplastics drift on surface currents and are repeatedly pushed ashore by tides and waves (Theocharis *et al.* 1999; International Pacific Research Center 2008). The stability of the shoreline, including the presence of sand dunes (McCormick and Hoellein 2016; Lee *et al.* 2017), the relative size of the substrate compared to the plastic debris, and the upward velocities of water (Hinata *et al.* 2017) affect this movement. As a result of abrasion and fragmentation on beaches, plastics continue to release toxic chemicals and heavy metals (e.g. Nakashima *et al.* 2016).

Lebreton *et al.* (2017) and Léon *et al.* (2018) have described how stranding, settling and resurfacing of plastic items in coastal environments play a major role in the transport of buoyant plastics in coastal waters and the transfer of organic pollutants from littoral plastic debris into the marine environment. Using a simple box model, they estimated that 46.7 to 126.4 million metric tons (approximately 66.8 per cent of the buoyant fraction released into surface waters since the 1950s) are stored along the world's shorelines. Approximately 32.3 per cent degrades into microplastics, 22.3 to 60.4 million metric tons of which remain on the shoreline while 0.29 to 0.80 million metric tons are transported out into the ocean.

Plastic debris generally becomes trapped on beaches and in coastal ecosystems such as mangrove forests and mudflats (Ivar do Sul *et al.* 2014; Garcés-Ordóñez *et al.* 2019; Martin *et al.* 2019; Riascos *et al.* 2019). Some types of plastic, such as acrylics,

sink whereas others that are more buoyant may remain in the surface layers for years (Lebreton *et al.* 2019). Microplastic concentrations in beach surface, body, and sands on underwater slopes have been observed to be of the same order of magnitude, suggesting that microplastics are repeatedly redistributed between the underwater and beach parts (Chubarenko *et al.* 2018). Observations that peak concentrations are related to stormy events and to places with stronger water dynamics and coarser sands are consistent with field studies from various locations (e.g. Turra *et al.* 2014). Storm events can also release buried microplastics, generating alternate states of source and sinks along the same beach across seasons (Rodríguez *et al.* 2020). These observations suggest that wave-driven marine waters which filter out microplastics and microfibres from uprush are the main source of microplastics on beaches, rather than beach sands or anthropogenic activities. The coherency of fragmentation across different marine environments also shifts the focus of the fragmentation process towards the material properties of synthetic particles themselves (Gigault *et al.* 2016; Chubarenko *et al.* 2018).

While beaches are characterized by a continuous turnover of depositional material and beach litter (Bowman *et al.* 1998; Browne *et al.* 2011), some coastal ecosystems, such as mangrove forests and mudflats, can accumulate large quantities of plastic debris and microplastics, especially during seasonal peak flows from rivers (Thiel *et al.* 2013; Lima *et al.* 2014; Ling *et al.* 2017; Lourenço *et al.* 2017; Naji *et al.* 2017; Garcés-Ordóñez *et al.* 2019; Martin *et al.* 2019; Riascos *et al.* 2019).

Mangrove forests cover about 132,000 km^2 along subtropical and tropical shores (Hamilton and Casey 2016). They occupy the intertidal fringe and develop a partially emerged root system, aerial and prop roots, forming an effective filter that dampens down wave energy and turbulence (Horstman *et al.* 2014; Norris *et al.* 2017) but also traps debris and plastics from land and riverine sources as well as from coastal waters. Plastics trapped by mangrove pneumatophores and prop roots may constitute a physical impediment, affecting both the tree itself and associated fauna, by preventing gas exchange and releasing harmful chemicals with the potential to alter important ecosystem services such as coastal protection, habitats and carbon sequestration (McLeod *et al.* 2011; Almahasheer *et al.* 2017; Martin *et al.* 2019). Residence times within mangrove forests are unknown. Studies show that retention capacities on beaches and shorelines depend upon the hydrodynamics of macroplastic objects and the depth of sediments (Ivar do Sul *et al.* 2014; Lourenço *et al.* 2017; Naji *et al.* 2017; Martin *et al.* 2019).

Items commonly found in beach debris include abandoned, lost or otherwise discarded fishing gear (ALDFG) and urban and industrial outflows (Duhec *et al.* 2015; Daniel *et al.* 2020). Plastic pellets and other microplastics are consistently found on beaches, indicating their high mobility (McCormick and Hoellein 2016; Fanini and Bozzeda 2018). In 2017 the International Coastal Cleanup (Bergevin 2018) reported the 10 most common items of plastic litter found on beaches around the world: cigarette butts (22 per cent), food wrappers (16 per cent), plastic bottles (14.5 per cent), plastic bottle caps (10 per cent), plastic grocery bags (7 per cent), other plastic bags (6.9 per cent), plastic straws and stirrers (5.9 per cent), plastic take-out food containers (5.8 per cent), plastic beverage lids (5.7 per cent) and foam takeout food containers (5.3 per cent). Similar percentages have been reported in Europe, where the main items were plastic drinks bottles, caps and lids (24 per cent) and cigarette butts (21.8 per cent) with the addition of cotton buds (13.5 per cent) and balloons and balloon sticks (2.7 per cent) (European Commission 2018a). Knowledge of the make-up of beach-collected marine litter is important in order to identify appropriate actions for clean-up and the reduction of sources (Schneider *et al.* 2018).

Ocean surface layer, gyres and remote islands

Since the 1970s marine plastics have been observed and increasingly monitored in the open oceans (Law *et al.* 2010; Kukulka *et al.* 2012; Cózar *et al.* 2014; Kukulka *et al.* 2015). Of the about 400 million metric tons of plastic produced annually, nearly two-thirds of the synthetic polymers created have a density lower than that of seawater and should thus be found in the ocean surface layer. Multidecadal observations of plastics in surface waters indicate the presence of plastics in all ocean basins and on many remote islands (e.g. Cózar *et al.* 2014; Law *et al.* 2014; Duhec *et al.* 2015; Díaz-Torres *et al.* 2017; Imhof *et al.* 2017; Lavers and Bond 2017; Collins and Hermes 2019; van der Mheen *et al.* 2019; Dunlop *et al.* 2020), but also the growth of large, concentrated accumulation areas within large-scale subtropical convergence zones, created by the wind-driven currents and circulation patterns (Maximenko *et al.* 2012; Ryan 2014; Lebreton *et al.* 2019; Wichmann *et al.* 2019).[10] In the Indian Ocean the garbage patch, located to the west of the basin, is very sensitive to different transport mechanisms and highly dispersive (van der Mheen *et al.* 2019). McAdam (2017) and Lebreton *et al.* (2019) estimated that the mass of plastic floating in the ocean surface ranged between 93,000 and 236,000 metric tons, two orders of magnitude lower than estimates of riverine inputs (e.g. Jambeck *et al.* 2015; van Sebille *et al.* 2015; Lebreton *et al.* 2017). The question of where the missing tons are located is addressed in a number of studies (e.g. van Sebille *et al.* 2020; Pabortsava and Lampitt 2020).

Nearly half the total mass of plastics in sub-tropical offshore waters consists of macroplastics. They are usually thick polyethylene and polypropylene plastic fragments older than 15 years, meaning that most of the microplastics in the mass of the oceans come from objects produced in the 1990s and earlier (Lebreton *et al.* 2019). Only certain types of plastics appear to have the capacity to endure long enough to eventually reach ocean gyres (Lebreton *et al.* 2019.) Estimates of the time it takes for plastics to be transported from coastal areas to mid-ocean islands and gyres range from months to years. Using recognizable litter items, van Sebille *et al.* (2019) estimated that it took just a few months for floating plastics to travel from the coasts of Peru, Ecuador and Colombia to the Galápagos region. Analysis of beach debris on the Ducie and Oeno Atolls and, 25 years later, on Henderson Island (all remote, uninhabited islands in the South Pacific) showed a 200- to 2,000-fold increase in

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the density of debris (Lavers and Bond 2017) and the highest density of plastics on the surface of a beach reported anywhere in the world. Asian and South American sources of plastic may reflect fishing activity in the surrounding waters. The high frequency of items which could be identified as having come from South America is probably the result of Henderson Island's position in the South Pacific Gyre. This current flows in an anticlockwise direction after travelling north along the coast of South America, transporting coastal waste to the island. A similar pattern is observed on remote islands off Chile and their adjacent waters (Lavers and Bond 2017). On Easter Island, for example, between 60 and 80 per cent of recognizable pieces of plastic come from industrial open ocean fisheries (Kiessling *et al.* 2017; Luna-Jorquera *et al.* 2019).

The water column and offshore marine sediments

Recent field survey results from Monterey Bay (California) show that microplastic particles readily flow between the epipelagic and mesopelagic water column and sea floor food webs, suggesting that the water column and sea floor sediments and food webs may be among the largest reservoirs of microplastics (Choy *et al.* 2019; Pabortsava and Lampitt 2020), for example in the Baltic Sea (Bagaev *et al.* 2017), in the Arctic Ocean (Kanhai *et al.* 2018) and off the west coast of Ireland (Courtene-Jones *et al.* 2018). Simulations from a mass balance model by Koelmans *et al.* (2017) suggest that of the plastics which have entered the oceans since1950, 99.8 per cent had settled below the ocean surface layer by 2016 with an additional 9.4 million metric tons per year settling thereafter. However, Lebreton *et al.* (2019) argue that the rapid degradation and sinking of more than 90 per cent of the plastics that enter the oceans each year does not reflect the age distribution of plastics at sea. They suggest that stranding, settling and resurfacing in coastal environments play a major role in the removal of buoyant plastics from the ocean surface in coastal waters. This conclusion coincides with age distributions from field surveys, which show that most of the macroplastic mass floating in coastal waters originates from objects less than five years old while older objects, such as thick polyethylene (PE) and polypropylene (PP) plastic fragments, are the most common type of plastic litter found in sub-tropical offshore waters (Lebreton *et al.* 2018).

Lebreton *et al.* (2019), who modelled the age of macroplastics in ocean gyres using whole ocean degradation rates, estimated that nearly half the total mass (47 per cent) was made up of macroplastics older than 15 years, suggesting that most of the microplastics in the mass of the oceans come from objects produced in the 1990s and earlier and that only certain types of plastics have the capacity to endure long enough to reach ocean gyres and accumulation zones (Lebreton *et al.* 2018). Plastic concentrations in the open ocean drop exponentially with depth (Reisser *et al.* 2015). Buoyant microplastics in the oceans eventually become submerged into deeper layers through loss of buoyancy or wind-induced turbulent mixing at the ocean surface. Wichmann *et al.* (2019) modelled the effects of different surface and near-surface currents on the global dispersal of floating plastics and showed that the gyre accumulations become more "leaky" in deeper layers, such that

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plastics disappeared in samples taken at about a 60 metre depth. At the same time, sub-surface currents transported significant amounts of microplastics from subtropical and subpolar regions to polar regions, providing a possible mechanism to explain why plastics are found in these remote areas (Cózar *et al.* 2017; Obbard 2018; Maximenko *et al.* 2019; Statista 2019). These results support observations of the fragmentation and typology of plastics indicating that aged debris is transferred from distant sources, for example via the poleward branch of the thermohaline circulation in the North Atlantic to the end of the conveyor belt in the Arctic Ocean, which then acts as a sink for plastic debris.

Microplastics fragment into undetectable sizes, sink due to buoyancy loss, and accumulate in different reservoirs (Ye and Andrady 1991). Benthic habitats can contain 103 to 104 microplastics per cubic metre versus 0.1 to 1 microplastics per cubic metre in water columns (Erni-Cassola *et al.* 2019). A recent review of the fate of microplastics in different sediments (Harris 2020) concluded that the median concentration of microplastic particles was highest in fjords at 7,000 particles per kilogram of dry sediment, followed by 300 in estuarine environments, 200 in beaches, 200 in shallow coastal environments, 50 on continental shelves, and 80 particles per kilogram in dry sediment from deep sea environments. Microplastics possibly accumulate more in deep sedimentary habitats (Zhang *et al.* 2020) and within subsurface sediment layers (Näkki *et al.* 2017; Wang *et al.* 2019a). Once deposited on shorelines (McDermid and McMullen 2004) and on the sea floor (e.g. Zhu, L. *et al.* 2019 and papers included), they can be ingested and reduced further in size (e.g. due to digestive grinding), and/or be transported to the sea floor upon egestion. There is also evidence of organisms forming microplastics through bioerosion (e.g. polychaetes in polystyrene debris [Jang *et al.* 2018] and sea urchins [Porter *et al.* 2019]).

Microplastic particles accumulate in beach sands and sediments in all parts of the world's oceans, including in Antarctica (e.g. Munari *et al.* 2017; Kanhai *et al.* 2019; Wang *et al.* 2019). Positive correlations have been established between microplastics in the water column and the sediments below, usually with a higher abundance (e.g. Zheng *et al.* 2019). Microplastics are now known to be present in the deepest parts of many ocean basins, including the Hadal trenches; this suggests that deep-sea sediments should be regarded as a potential sink for microplastics in the oceans (e.g. van Cauwenberg *et al.* 2013; Woodall *et al.* 2014; Fisher *et al.* 2015; Bergmann *et al.* 2017; Kanhai *et al.* 2019; Mu *et al.* 2019; Peng, G. *et al.* 2020). A series of 5,010 deep-sea surveys by remotely operated vehicles and submersibles of the Japan Agency for Marine-Earth Science and Technology (JAMSTEC) in the six oceanic regions in 1982–2015 located 3,425 man-made debris items. More than one-third of the debris consisted of macroplastics, of which nearly 90 per cent was single-use products (Chiba *et al.* 2018). In areas deeper than 6,000 metres these ratios increased to just over half and 92 per cent, respectively; the deepest item recorded was a plastic bag at 10,898 metres in the Mariana Trench. Deep-sea organisms were observed in 17 per cent of plastic debris images, and entanglement of plastic bags on chemosynthetic cold seep communities was shown (Chiba *et al.* 2018).

2.2.5 Regional variations and hotspots

Marine debris and plastics are accumulating in large volumes in many parts of the world. Three regional hotspots are of particular concern: the Mediterranean Sea because of its enclosed nature and proximity to millions of people; the Arctic Ocean because of its pristine nature and impacts on indigenous peoples; and the East Asia and ASEAN region because of its extensive coastline in proximity to very large populations with a high dependency on the marine environment for survival and, often, insufficient waste management systems.

The Mediterranean Sea

The Mediterranean region is the world's fourth largest producer of plastic goods. Residents and visitors generate 24 million metric tons of plastic waste per year. Tourism increases the amount of waste by up to one-third during the summer in some countries, and local waste management facilities are often overwhelmed. Through the transport pathways in the Mediterranean Sea, marine litter and plastic pollution affect the entire blue economy, with regional economic losses (especially those linked to tourism) estimated at US$ 700 million per year.

Every year 0.57 million metric tons of plastics enter Mediterranean waters, equivalent to dumping 33,800 plastic bottles into the sea every minute (Dalberg Advisors, WWF Mediterranean Marine Initiative 2019; Boucher and Bilard 2020). Marine litter, particularly floating plastics, has been found in the Mediterranean in quantities comparable to those in the five ocean gyres (the North Atlantic, South Atlantic, North Pacific, South Pacific and Indian Ocean Gyres). Studies based on global models have proposed that the Mediterranean is the world's sixth greatest accumulation zone for marine litter (Panti *et al.* 2015; van Sebille *et al.* 2015; Sauria *et al.* 2016; Guerranti *et al.* 2017; Baini *et al.* 2018; Consoli *et al.* 2018; Fossi *et al.* 2018). Marine litter and plastic pollution are a growing problem in the sub-basins of the Adriatic and Ionian Seas (Anastasopoulou *et al.* 2013; Munari *et al.* 2015; Arcangeli *et al.* 2017; Pellini *et al.* 2018; Zeri *et al.* 2018; Vlachogianni *et al.* 2018; Fortibuoni *et al.* 2019).

The structure of the circulation of the Mediterranean Sea means the heavy anthropogenic waste loads entering from its coastline are naturally retained inside its basin and eventually in the deep sea (Danovaro *et al.* 2020). Its circulation is characterized by an inward surface flow of waters from the Atlantic Ocean, but with no significant outward flow anywhere along its coastline. The return flow into the Atlantic, which occurs in the subsurface layer, hampers surface-floating items and prevents them being expelled, causing them to accumulate within it (Zambianchi *et al.* 2017). At the global ocean level, the Mediterranean's possible sink role for floating particles of global origin was originally shown by Lebreton *et al.* (2012). To date, there is no evidence of permanent litter accumulation areas being formed (Cózar *et al.* 2015; van Sebille *et al.* 2015; van Sebille *et al.* 2020), the main reason being the predominantly cyclonic circulation and high temporal variability of currents.

Using a range of models and historical data, Zambianchi *et al.* (2017) observed that litter including plastic waste accumulated in the southeastern portion of the Levantine basin and on the southern Mediterranean coasts. This is consistent with observations by Cózar *et al.* (2015), who showed that a clear zonation of debris in the Mediterranean is visible with a maxima in the southern portion of the basin, both in the western and eastern Mediterranean, and by Mansui *et al.* (2015), who identified the southern coastal strip of the eastern Mediterranean as an accumulation area or preferential beaching destination. Floating debris also appears in the Algero Provencal basin, the Sardinia Channel and south of the Balearic Islands, with further high concentration areas in the north associated with the Northern Current or with the northward propagation of Algerian eddies (Cózar *et al.* 2015). Observations in the Tyrrhenian Sea suggest greater abundance in the southern portion of the basin, characterized by very slow, basically stagnating dynamics (Guerramti *et al.* 2017), and in the Corsica Channel, the chokepoint for waters passing from the Tyrrhenian to the Ligurian Sea. Observations of the bottom distribution of debris indicate that it is largely influenced by the vertical detail of the Mediterranean circulation, with a long-term presence of litter in the southern Algerian basin and southeast off Crete, and in canyons and other areas influenced by strong sinking patterns such as the Gulf of Lyons. A Langrangian modelling analysis of the fluxes of plastic onto six selected coastlines of Mediterranean Marine Protected Areas showed that they were relatively low (0.4-3.6 kg [km day] in comparison with an average flux of 6.2 ± 0.8 kg [km day]) and calculated over the Mediterranean in the period 2013–2017 (Liubartseva *et al.* 2019).

The Arctic Ocean

The state of knowledge on marine litter, including microplastics, in the Arctic marine region primarily reflects the fact that information is more prevalent for areas where human activities are concentrated, including the Barents, Norwegian and Bering Seas, or for specific research topics (e.g. seabirds). Few data are available for the Central Arctic Ocean and the coastal areas around it in Siberia, Arctic Alaska, mainland Canada and the Canadian Arctic Archipelago (PAME [Protection of the Arctic Marine Environment Working Group of the Arctic Council] 2019). From the limited analysis of macrolitter (e.g. nets, floats and other debris) washed ashore on Arctic beaches or accumulating on the sea floor, most (50-100 per cent) can be attributed to fishing activities.

Microplastics have been increasingly reported throughout the pristine waters, beaches and sediments of the Arctic Ocean (Sundet *et al.* 2016; Hallanger and Gabrielsen 2018; Kanhai *et al.* 2018; Kanhai *et al.* 2019). Even a few years ago the abundance of microplastics in surface waters was of the same order of magnitude as that found in the North Pacific and North Atlantic (e.g. Lusher *et al.* 2014; Welden and Lusher 2017); van Sebille *et al.* (2012) identified a potential accumulation area in the Barents Sea by modelling global drifter data and concluded that this one was linked to the North Atlantic Gyre via advection. A median of

6,300 items per km^2 (plastic particles >0.5 mm, excluding fibres) was measured in the Greenland and Barents Seas. In addition, plastic particles were found in most (73 per cent) of the surface ice-free waters sampled in the circumpolar area (Cózar *et al.* 2017). Microplastics in surface and subsurface waters were nearly all microfibres, the breakdown products of larger plastic items such as fibres from shipping fishing gear, recreational and offshore industries, and washing of textiles (Sundet *et al.* 2016; Kanhai *et al.* 2018). The quantities of microplastics in the water column, although slightly lower than in surface waters in southwestern Svalbard (Obbard *et al.* 2014; Amelineau *et al.* 2016), doubled between 2005 and 2014, possibly as a result of changes in sea ice cover.

The distribution and accumulation of marine debris on the sea floor of the Barents Sea, the Norwegian Sea, and the Svalbard, Norway area are very uneven and are determined by hydrography, geomorphology, prevailing winds and human activity (Buhl-Mortensen and Buhl-Mortensen 2017). While no microplastics were found in sediment samples at a depth of 40-60 metres (Sundet *et al.* 2017), litter in trenches and canyons was more than 10 times as abundant as in sediments on the shelf (Buhl-Mortensen and Buhl-Mortensen 2017). West of Svalbard the Hausgarten deep-sea observatory station has operated regular video transects since 2002. These show that densities of debris almost doubled between 2002 and 2014 (Bergmann *et al.* 2015; Tekman *et al.* 2020) and are 20 to 40 times greater than offshore shelf sea levels in the Barents Sea (Buhl-Mortensen and Buhl-Mortensen 2017).

Substantial quantities of microplastic particles smaller than 25 µm, a size most other studies do not sample, have also been found in the deep sea, with densities exceeding those observed at the surface and subsurface (Ballent *et al.* 2013; Bergmann *et al.* 2015; Kanhai *et al.* 2019). Higher microplastic densities than in any other benthic regions investigated were also found at the Hausgarten deep-sea station (2,340-5,570 metres deep); these amounts were positively associated with chlorophyll α, suggesting that algae might play a role in downward transport to the deep sea in this area either as biofouling or as microplastics adsorbed to aggregating algae (Bergmann *et al.* 2015; Tekman *et al.* 2020).

Although there are no published data on entanglement, several seal species, such as harbour seals (*Phoca vitulina*) and bearded seals (*Erignathus barbatus*), have been observed with plastics wrapped around them and, in Svalbard, old fishing nets and abandoned, lost or otherwise discarded fishing gear (ALDFG) have been found entangling seabirds and Svalbard reindeers (*Rangifer tarandus platyrhynchus*) both dead and alive. In 2016, in gathered litter analysed for type and origin, plastics were found to be the most abundant material from fisheries-related and household items (Nashoug 2017).

Ingestion of plastics has been studied in several Arctic bird species that forage at sea, such as fulmars (Trevail *et al.* 2014), and in marine mammals (e.g. Donohue *et al.* 2019). The highest frequency was detected in little auks (*Alle alle*); microplastics were detected in all gular pouches (Amelineau *et al.* 2016), a matter of concern as this implies the chicks of little auks are exposed to microplastics. Plastics have also been found in stomach analyses of the Greenland shark (*Somniosus microcephalus*) from south Greenland (Nielsen *et al.* 2014; Morgana *et al.* 2018) and from Svalbard (Leclerc *et al.* 2012). However, no plastics were observed in Atlantic cod (*Gadus morhua*) from northern Norway, Varanger and the Lofoten area (Brate *et al.* 2016). Plastics, but no microplastics, have been reported in nearly all blue mussels (*Mytilus edulis*) and the Iceland cockle (*Clinocardium ciliatum*) (Sundet *et al.* 2016).

In addition to having some of the world's highest microplastic burdens, the Arctic, with its harsh living conditions, limited food web and monumental climate change now under way, is especially vulnerable to the effects of marine litter and plastics.

East Asia and Association of Southeast Asian Nations seas

Across the East Asia and ASEAN regions, including the East China Sea, the Sea of Japan and the South China Sea region, concerns about marine plastic pollution are growing (Isobe *et al.* 2015; Lyons *et al.* 2019; WWF 2020). In a 2010 estimate, six ASEAN member states were listed among the top 20 countries that mismanaged their waste, resulting in plastic leakage into the oceans (Jambeck *et al.* 2015). More recent figures indicate that Indonesia, the Philippines, Thailand and Viet Nam, together with China, are responsible for more than half the plastics entering the oceans (Ocean Conservancy 2015; Lebreton *et al.* 2017; UNEP 2017b).

Knowledge of the abundance, source and fate of marine debris in the region is increasing (South China, Cheung *et al.* 2016, Lin and Nakamura 2018; Republic of Korea, Lee *et al.* 2017, Lyons *et al.* 2019). Most of the 145 research papers published in the ASEAN region on marine plastic pollution appeared from 2017 onwards (Lyons *et al.* 2019). These papers identify deficiencies in waste management infrastructure as a major concern in the region and record the widespread presence of macro- and microplastic debris in high concentrations in all the water basins, as well as in marine life sampled or trapped in mangrove root systems and on coral reefs (Cai *et al.* 2017; Purba *et al.* 2019; Onda and Sharief 2021).

Peng, G. *et al.* (2019) used autonomous submarines to collect photo footage of debris in the tributary canyons of the Xisha Trough in the northern South China Sea. They found that the accumulation of debris at 1,800 metres was greater than in any other submarine canyon in the world. Most of the debris came from fisheries and navigational activities. It was brought by seasonal surface ocean currents, influenced by the geomorphology of submarine canyons. The waters of the northern South China Sea are home to a diverse array of benthic fauna, fish and corals which are likely to be harmed by plastic pollution.

SECTION 3

MONITORING METHODS, INDICATORS, STANDARDS AND PROGRAMMES

3.1 Developments in monitoring methods

3.1.1 Riverine sampling of litter and macroplastics

Given that rivers are a major pathway for litter including plastic waste entering the marine environment, there has been a growing emphasis on the monitoring of upstream and downstream sources and flows (UNEP 2020b,c,d; Borrelle *et al.* 2021; Meijer et al. 2021). Field and laboratory techniques for sampling microplastics in freshwater environments closely follow protocols for marine systems, although certain plastic polymers sink in freshwater because of its lower density compared to saltwater (Gonzalez *et al.* 2016; González-Fernández and Hanke 2017; GESAMP 2019; Karlsson *et al.* 2019; Forrest *et al.* 2020). In recent years studies have been carried out to improve the reliability of data on the flows of macrolitter and plastics in rivers and along riverbanks. These studies cover the use of visual observations and the collection of items, automated acquisition systems, inflow structures such as dams and weirs, inflow water, riverbeds and bottom nets, booms, floats, manta trawls, and nets and pumps (Picó and Barceló 2019).

Systematic surveys based on very simple visual protocols, such as observing rivers from bridges (González-Fernández and Hanke 2017) and debris sampling, have been shown to provide reliable estimates of the magnitude and seasonal dynamics of plastics transport (mass and number of items by size), spatial distribution across river width, and the plastic polymer composition of the most common items in rivers (van Calcar and van Emmerik 2019). Data from visual counting surveys can be used to compare the composition of plastics recovered from the sampling of rivers, sediment layers and ocean surface layers and to test hypotheses about the types of plastics that can be horizontally transported from rivers to oceans through currents, as well as the role of rivers in delivering buoyant plastics into the oceans. Scaling up visual observations can also be done using Earth observations (satellites and cameras) and unmanned aerial vehicles (Martin *et al.* 2018; Geraeds *et al.* 2019) and cameras (Kylili *et al.* 2019). However, because river flows can fluctuate significantly on an hourly, weekly, seasonal or multi-year basis, in situ monitoring in different parts of a river is required. Automated monitoring can support a multi-temporal sampling and monitoring approach.

Riparian areas and riverbanks are often transient sources of plastics due to water level variations and tides, which can lead to regular depositions as well as to depositions during extreme events. Sampling along dynamic banks does not yield consistent time series data. However, earlier studies have shown that it is possible to quantify time signals in plastics found in estuaries even though it remains difficult to model the effects of all the different processes and estimate the volumes transported into the ocean (Sadri and Thomson 2014; Dris *et al.* 2020).

3.1.2 Shoreline sampling of litter and macroplastics

Field methods for assessing the volumes of macrolitter along shorelines and beaches vary according to the indicator items being targeted. Indicator items do not provide information on the relative importance of different sources of litter for a given region, but can give an indication of the sources and a methodology to calculate trends in the inputs (Schulz *et al.* 2017; Schulz *et al.* 2019).

Other categorizations include those of the Regional Seas Conventions and Action Plans and national programmes such as the National Oceanographic and Atmospheric Administration Marine Debris Monitoring and Assessment Project (NOAA-MDMAP) in the United States (GESAMP 2019). At some locations where visual surveys are used to count items, Angelini *et al.* (2019) found that colour is an important determinant, with higher accuracy in blue plastic counts and undercounting of white and clear plastic. Comparing different categorizations is thus essential to reduce uncertainties among different estimates (van Calcar and van Emmerik 2019).

3.1.3 Methods of measuring interactions between litter and marine animals

Monitoring entanglement needs to be undertaken separately for different groups (marine mammals, birds, reptiles, fish and invertebrates). Observations can be recorded and analysed by ecosystem compartments (e.g. via stranding networks on coastlines, at the surface during oceanographic campaigns or through observer programmes, and on the seabed through scuba diving for shallow areas or use of submersibles, remotely operated vehicles and automated underwater vehicles for deeper waters (Galgani *et al.* 2018). Claro *et al.* (2019) reviewed the range of current entanglement methodologies and proposed several candidates as standard protocols. They identify long-term stranding and photo-identification networks (e.g. the United States National Marine Mammal Database) as the most appropriate monitoring tools. Generally, recording the impacts of entanglement and ingestion of marine litter is inhibited due to inconsistent record-keeping by different networks. There are also difficulties in determining what is abandoned, lost or otherwise discarded fishing gear (true debris) and what is active fishing gear. Animals can be caught incidentally by pieces of equipment that are not considered debris.

A number of ad hoc methods are used to assess sessile organisms and marine ecosystems such as "animal forests" (Galgani *et al.* 2018). These methods include monitoring of biodiversity in coral reef assemblages and kelp forests by divers using transect surveys, use of submersibles and remotely operated vehicles in deeper areas, invitations to the public to submit images of entanglement (Ryan 2018), and the addition of marine litter to

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routine surveys in long-term reef monitoring programmes (e.g. Reef Check) (Carvalho-Souza *et al.* 2018). In the case of the deep sea, organizing databases that compile photographic records of litter on the sea floor (Chiba *et al.* 2018) would also be very useful.

3.1.4 Sampling and detection of microplastics

There are intrinsic difficulties in determining and identifying microplastics and plastic microfibres in environmental samples due to their size and varied shape, colour and degree of degradation. Efforts to detect the presence of these particles have resulted in different methodologies (Löder 2015; Costa and Duarte 2017; Lusher *et al.* 2017b; Ryan *et al.* 2020) and have led to commentaries that there have been too many papers with too many variations (Borja and Elliott 2019). Various units of measurement and quantification have been used by researchers; for example, data are sometimes given per weight of sample, per volume of matrix or per sampling area, without information that would enable comparisons to be made (Besley *et al.* 2017; da Costa 2018).

In the meantime, Koelmans *et al.* (2020) have provided an approach using rescaling methods and probability density functions to improve the alignment of different microplastic studies by correcting for differences in size ranges used to report microplastic concentrations. They have also addressed the incompatibility of data used in current species sensitivity distributions (SSDs) caused by differences in the microplastic types in effect studies and nature. A combination of methods enabled them to correct for the diversity of microplastics, address results in a common language, and assess the risks of microplastics as an environmental material.

Aquatic sampling protocols

Microplastic pollution in freshwater ecosystems is being studied more widely (Blettler *et al.* 2018; van Emmerik and Schwartz 2019) and harmonized procedures are now being agreed (GESAMP 2019; UNEP 2020b,c,d). Several river systems in Europe, North America and China have been sampled extensively for microplastics at the surface and in the water column (Lechner *et al.* 2014; Mani *et al.* 2015; Zhang *et al.* 2015; Liedermann *et al.* 2018), in sediments (Su *et al.* 2016; Zhang *et al.* 2016; Wang *et al.* 2017; Mani *et al.* 2019) and along riverbanks (Imhof *et al.* 2013; Klein *et al.* 2015). However, the complexity of microplastics and the lack of harmonization in sampling methodology makes it difficult to compare results from earlier studies (Dris *et al.* 2015a; Dris *et al.* 2016; Dris *et al.* 2017; Dris *et al.* 2018; Koelmans *et al.* 2019; Kooi and Koelmans 2019).

Atmospheric sampling

There are currently no standard operating protocols for the analysis of airborne microplastics (Enyoh *et al.* 2019). In the atmosphere there may be thousands of microplastics that could be inhaled by humans and animals. These microplastics can come from the decomposition of synthetic materials such as those in electronic devices, packaging materials, tyres, clothing and furniture and are distributed both indoors and outdoors. A number of methodologies have been developed to collect airborne and deposited microplastic particles (Dris *et al.* 2016; Cai *et al.* 2017; Dris *et al.* 2017; Allen *et al.* 2019; Barbosa *et al.* 2019; Klein and Fischer 2019; Stanton *et al.* 2019a).

The collection of airborne particulate matter is mainly carried out using instruments that capture ambient aerosols (bulk samplers: glass or stainless-steel bottles containing funnels, deposition gauges, and other devices for source characterization, and other devices for source characterization, e.g. the passive sampler device) (Klein and Fischer 2019). In general, these sample collection systems are left where sample collection will not be affected by radiation or rainfall, and where height above ground, geographic coordinates and the sampled air volume are taken into consideration. Although air collection procedures are relatively simple, they are susceptible to contamination and the entrainment of other particles (Stanton *et al.* 2019b).

Evaluation of the presence of microplastics in air is still limited. The findings of some studies performed in Europe (Dris *et al.* 2016; Dris *et al.* 2017; Allen *et al.* 2019; Enyoh *et al.* 2019; Klein and Fischer 2019) and China (Cai *et al.* 2017) reported variations in the values of microplastic air deposition rates. The analysis of microplastics in air by Dris *et al.* (2017) showed that in the city of Paris, France the rate of deposition of microplastics and nanoplastics was much higher indoors than outdoors. Further work in mountainous and sparsely populated regions in France showed that the deposition rate outdoors was very similar to that in the city (Allen *et al.* 2019).

Biotic material sampling

Different methods are used to look at plastic ingestion: the naked eye, microscopic analysis, enzymatic methods and chemical digestion (e.g. OSPAR 2015). Markic *et al.* (2020) reviewed the literature for fish and found that chemical digestion is the most robust and has the highest detection rate, but can also cause damage to some polymers, and that enzymatic methods are known to be gentler although they are often laborious, expensive and time-consuming. Von Friesen *et al.* (2019) have developed a novel tissue digestion method for bivalves, using pancreatic enzymes and a pH buffer (Tris, or tris(hydroxymethyl) aminomethane) rather than potassium hydroxide. It is a candidate for a standardized digestion protocol, as it can provide high throughput with minimal handling, is low-cost, and does not impair correct identification of plastic polymers and textile fibre polymers.

Nelms *et al.* (2019b) have developed a novel and effective methodology pipeline to investigate dietary exposure of wild top predators (e.g. grey seals, *Halichoerus grypus*) to microplastics by combining scat-based molecular techniques with a microplastic isolation method. They use DNA meta-barcoding (a rapid method of genetic biodiversity assessment) to gather detailed information on prey composition from scats and investigate the potential relationship between diet and microplastics burden. The results show that such a non-invasive, data-rich approach can help reduce the costs and sample volumes required for analysis and could be used to underpin studies of the relationship between dietary composition and rates of microplastics ingestion in high trophic level species. Maes *et al.* (2020) have also developed a method for quantifying microplastics in marine mammals by looking at the spiral valves of porbeagle sharks (*Lamna nasus*).

In the case of birds, Provencher *et al.* (2017, 2018, 2019) observed that while they are the most studied megafauna group with regard to plastic ingestion, a lack of consistency in sample collection and processing impedes metaanalysis as well as large-scale comparisons of volumes of plastics consumed and impacts on other species. They have provided a set of recommendations on best practices in sample collection, processing and reporting, together with guidance on how carcasses, regurgitations and pellets should be handled and treated to prevent cross-contamination and methods to assess different size classes of microplastics. They propose that standardized techniques for removing sediment and biological materials can be used for other animal groups.

In a review of the quality criteria for sampling and analysis of microplastics in biota samples, Hermsen *et al.* (2018) set out 10 stages of the sampling and analysis process and protocols. They evaluated the reliability of each stage in 35 studies, assigning a score to each stage where: 2 = reliable without restrictions; 1 = somewhat reliable, but with restrictions; and 0 = unreliable. All the studies reviewed had at least one processing stage with a score of 0. The average overall score was 8.0 out of 20. This evaluation does not invalidate the different studies, but it suggests the need for significant improvement in how sampling and analysis is undertaken, with much greater emphasis placed on minimizing and accounting for sample contamination.

Sediment sampling protocols

Recent reviews of methods for sampling sediments conclude that the distribution of microplastics in sediments is highly varied (Hanvey *et al.* 2017; Prata *et al.* 2019b) and influenced by their properties, as well as by winds and currents. The results will depend upon the sampling area (e.g. high tide line, intertidal areas, transects) and depth. Collection of sediments on the tide line (the high accumulation area for microplastics) may result in overestimation. The collection of microplastics on beaches includes direct sampling with forceps, sieving, and collection of sediment samples. Collecting samples from the seabed requires a vessel and use of specialized equipment lowered to the seabed to collect the samples (e.g. grab sampler, box corer). Microplastics need to be separated from sediment samples in two separation steps: a reduction step using nets, followed by sieving; and a separation step, usually through filtration and/or density separation using sodium chloride. The protocols proposed by Hanvey *et al.* (2017) and Prata *et al.* (2019b) do not recommend using flow cytometry or electrostatic separation.

Esiukova *et al.* (2019) used modified United States National Oceanic and Atmospheric Administration (NOAA) methods and µ-Raman spectroscopy to obtain data suitable for a comparative analysis of sediments across the Baltic Sea.

Quality assurance and quality control protocols

Braun *et al.* (2018) have proposed a series of guidelines to ensure consistency and avoid aliasing results due to background levels of microplastics. These include: "plastic-free" or low-plastic working conditions during all analytical steps (sampling, preparation, detection); avoidance of standard plastic products in favour of alternatives made of metal, glass or silicone; frequent wiping down of laboratory workspaces; handling of samples in laminar flow boxes, especially during the preparation process for wet samples and during the determination of particle numbers; all glassware to be washed thoroughly, oven-dried and covered when not in use; microscopic traces of plastic to be reduced by heating the glassware in a burnout furnace (<600°C); filters and sieves to be inspected under a microscope prior to use; and personnel to wear natural (i.e. cotton) clothing and laboratory coats, as well as powder-free examination gloves throughout the experimental procedure. It is also recommended to undertake sterilization (via steam, radiation or chemicals) of dry samples from wastewater, greywater, sewage sludge and organic wastes.

Documentation and measurement of zero samples or blank value determination for the applied detection methods is essential, as contamination can occur during sampling, preparation and detection (contamination by airborne particles). Based on current knowledge, triple repetition is highly recommended for blank value determination in the particle counting process (including sample preparation) of each campaign. Defined reference materials and controls are also required for the comparison of all analytical procedures.

To improve the reliability and repeatability of results, published studies of various polymers under different conditions should document the sampling process, sample preparation and detection methods. Depending on the type of polymer, particle size and state of aging, there may be degradation and or fragmentation of larger particles. The studies recommend that in the presentation of results the following information is needed: number of microplastics per volume for sampled water bodies, or per total dry matter (kg) for sampled solids; microplastic mass per volume for sampled water bodies (µg per litre) or per total dry matter for sampled solids (mg per kg); plus a precise description and comprehensible documentation of the amount of the sampled environmental aliquot, the prepared laboratory sample and the analysed sample. Microplastics analysis should be presented in size classes; small particles that occur in higher quantities are grouped into narrower classification clusters than the larger particles, which are more relevant in terms of mass and classified into wider clusters. This can help ensure higher methodological feasibility (including filtration and detection limits) and better integration of particle quantities/masses in effects and impact analyses (i.e. for environmental assessments).

Identification of physical and chemical characteristics of microplastics

There are four basic detection approaches: microscopic, spectroscopic, thermal and chemical (Masura *et al.* 2015; Braun *et al.* 2018; GESAMP 2019). Microscopic methods are widely used in the case of microplastics that fall into the micron range (i.e. neuston net samples) to capture surface texture and structural information; however, the results are primarily a visual characterization only (Shim *et al.* 2017). Raman spectroscopy uses a laser beam, resulting in different frequencies of back-scattered light depending on the molecular structure and atoms present. This produces a unique spectrum for each polymer. There is a high risk of misinterpretation when real samples are measured using only imaging methods (e.g. light and electron microscope) and particle counting methods (e.g. light scattering, laser scattering). Therefore, measurements must be carried out with comparative and blank samples and only in combination with other chemical or chemical-physical analysis techniques.

In the case of thermoanalytical methods, the sample is pyrolyzed under inert conditions and specific decomposition products of the individual polymers are detected. Chemical methods are used to decompose the samples and detect specific fragments of polymers or elements such as molecular weight determination, chemical degradation and subsequent LC, staining with Nile Red and subsequent fluorescence detection, the application of TGA-FTIR/MS or TGA-GC-MS, and hyperspectral imaging methods (Braun *et al.* 2018). Staining with Nile Red results in discrimination of plastics from other particles, using fluorescence excitation and emission in the visible range, and can be implemented in situ (Maes *et al.* 2017b).

For the purification of microplastics from organic matter four major methods have now become routine, each of which has both pros and cons. Oxidative digestion is inexpensive, but the temperature needs to be controlled and several applications may be needed (Masura *et al.* 2015). Acid digestion is rapid, although it can attack some of the polymers (Claessens *et al.* 2013). Alkaline digestion is effective and causes minimal damage to most polymers, but it can attack cellulose acetate (Dehaut *et al.* 2016). Enzymatic digestion is effective and causes minimal damage to most polymers, although it is very time-consuming (Löder *et al.* 2017).

To improve the accuracy of the protocol, Prata *et al.* (2019b) recommend removing organic matter, using simple methods without affecting the structural or chemical integrity of polymers. They note that many studies do not include this step, thus increasing the likelihood of overestimating the density of microplastics in samples.

In converting the results of microplastic particle analyses into mass content, considerable errors are possible as the particles are often not uniformly spherical and the material density cannot be specified accurately enough due to undefined structures. In addition, the spherical diameter cannot be determined exactly, but enters the volume formula of a sphere with the third power (high fault levels possible). In the presentation and documentation of results it is therefore vital that the quantity of environmental aliquots analysed, and the process duration and hours of work per sample, are specified.

Automation

Many laboratory techniques are being semi-automated and can be applied in situ after the necessary engineering instrumentation is developed. For example, staining with Nile Red dye makes it possible to discriminate plastics from other particles using fluorescence excitation and emission in the visible range, which can be implemented in situ (Maes *et al.* 2017b). While chitin and some other organic matter are also stained, these particles may be discriminated using other means, e.g. by density or digestion. This technique has been semi-automated using image analysis software (Erni-Cassola *et al.* 2017). Full automation will require in situ filtration and image capture, and likely separation or digestion steps, which, although onerous, are not beyond the capabilities of in situ instrumentation (e.g. Scholin *et al.* 2017). Raman spectroscopy, previously applied in situ for other applications (Guo *et al.* 2017; Li *et al.* 2018) and for microplastics, can be operated in a spectral range with low absorption in water (e.g. 785 nm laser, Frère *et al.* 2016). Imaging and flow cytometry (e.g. Sgier *et al.* 2016) can also be used, as well as in situ flow and imaging cytometers (Lambert *et al.* 2017; Olson *et al.* 2017). However, a focused and significant effort is still required to turn these into mature sensor technologies that can become part of operational metrology of marine debris across a wide size range in the marine environment (Maximenko *et al.* 2019).

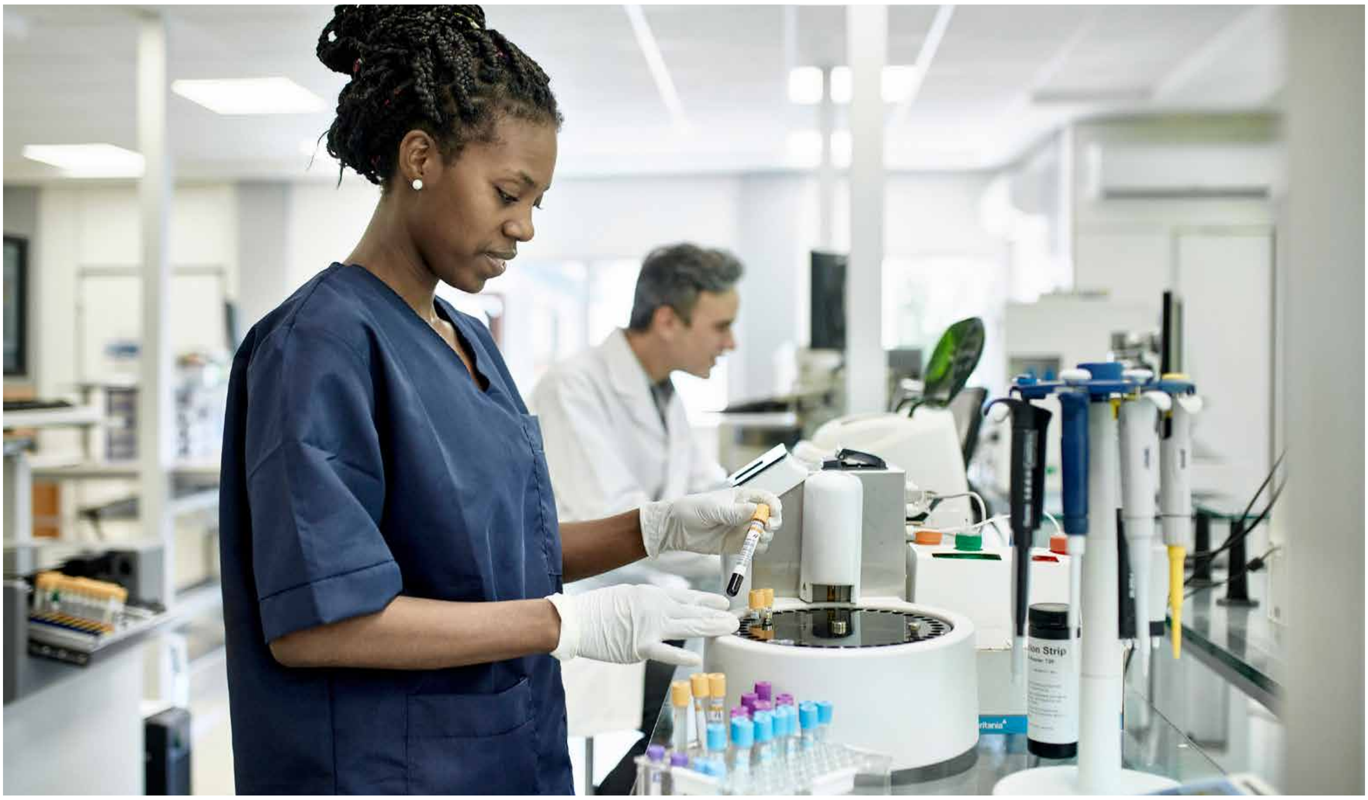
3.1.5 Toxicological assessment of micro- and nanoplastics

A review of current practices in toxicological studies of microplastics (Barbosa *et al.* 2019) underlines the enormous efforts made during the past five years to detect, characterize and quantify the fate and effect of microplastics and nanoplastics. Studies reviewed included cell viability, oxidative stress, cytotoxicity, uptake, changes in protein configuration, DNA damage, genotoxic effects, phagocytosis and gene folding. Despite the lack of a quality analysis, the results underline a number of problems with toxicological experimental methods, such as use of fluorescent microspheres in laboratory toxicity experiments compared to the many different forms in the environment. Overall, more research is needed on the uptake kinetics, accumulation and biodistribution of microplastics in biological systems (e.g. Saley *et al.* 2019; Wang *et al.* 2019a; Wang *et al.* 2019b).

3.1.6 Observing platforms, data and modelling technologies

Satellites, aircraft and drones

The latest generation of operational space missions is filling gaps in observations and proxy data streams for the regular monitoring of marine debris and macroplastics on riverbanks, in surface waters and along shorelines, although significant technological challenges remain (e.g. Garaba *et al.* 2018; Martínez-Vicente *et al.* 2019; Maximenko *et al.* 2019; van Sebille *et al.* 2020). While remote sensing is beginning to provide information about surface distributions of marine litter in the oceans, none of the currently orbiting instruments has been



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specifically designed to detect plastic marine litter. At the same time, the scope and capabilities of some existing and future missions overlap with the properties and dynamics of marine debris. Some of the same sensors can be used to monitor marine debris.

The European Union Copernicus Sentinel-2 provides free data with sufficient spatial and spectral resolution to detect the marine litter windrows (elongated accumulations commonly found on the sea surface, varying in length from tens of metres to over a kilometre), as well as individual objects as small as 5 m^2 on land. The Sentinel data can be integrated with those of the United States Landsat-8 satellite, which offers similar imagery, to provide an average revisit time of three days.

A second, operational system, Copernicus Sentinel-3, includes instrumentation for ocean-colour tracking and can be used to measure subtle spectral changes from ocean surface imaging. As part of the European Space Agency Optical Method for Marine Litter Detection (OptiMAL) programme, satellite data are combined with global observations and aerial ocean surveys using high spatial resolution cameras on aircraft, plus satellite bathymetric lidar, hyperspectral shortwave infrared imaging. The OptiMAL configuration is capable of detecting a range of litter from larger pieces to microplastics with diameters of less than 5 mm suspended in the upper layers of the ocean, as well as plastic debris found on shorelines.

Similarly, other missions will provide data on sub-mesoscale currents that will advance models simulating marine debris drift. For example, the PRISMA satellite, carrying a hyperspectral instrument operating in the 400-2,500 nm range, with spectral and spatial resolution of 12 nm and 30 metres, respectively, was launched in March 2019 by the Italian Space Agency. The Plankton, Aerosol, Cloud, ocean Ecosystem mission (PACE, United States National Aeronautics and Space Agency [NASA]) is expected to be launched in late 2022. It will include a hyperspectral Ocean Color Instrument (OCI), and the Environmental Mapping and Analysis Program (EnMAP, Germany DLR) will carry a hyperspectral "pushbroom" imager with high spectral (6.5-10 nm) and spatial (30 metre) resolutions.

Another important area in which remote sensing is likely to make a significant contribution is the measurement of riverine discharge using new remote sensing techniques (Gleason and Durand 2020). The Surface-Water Ocean Topography (SWOT) mission (Biancamaria *et al.* 2016), to be launched in September 2021, will revolutionize quantification of water levels in global rivers over 100 metres in width and enable monitoring of water discharges globally, offering new possibilities to quantify the flux of suspended sediment and nutrient fluxes in large rivers where ground calibration exists and provide bathymetric measurements.

Future planned and potential missions expected to give insights into sub-mesoscale dynamics include SKIM (Sea Surface KInematics Multiscale monitoring satellite mission, although not approved as an ESA mission; Ardhuin *et al.* 2019), SEASTAR, and WACM (Wind And Currents Mission).[11] Synthetic Aperture Radar (SAR) observations, embedded in these missions or used as complementing projects, will significantly enrich our knowledge of marine debris sources, sinks, patterns and pathways. Passive microwave radiometers may also be helpful in tracking marine litter; however, their capability is yet to be demonstrated.

Remote sensing will always require adequate calibration and validation, based on a combination of observations from in situ instruments and sampling. Ground-truthing of earth observation is best served when the presence of significant debris signals is detected from sensor data, ground-truthed, and the results fed back into the signal data for data extraction and calibration-validation (Garaba and Dierssen 2018). The calibration and validation of open ocean surface measurements also requires the influence of subsurface ocean processes such as the vertical shear of currents on debris drift to be included, using data, for example, from the various drifter and profiler arrays within the Global Ocean Observing System (GOOS)[12] (Moltmann *et al.* 2019). Satellites are important in order to cover the largest scales, but remote sensing technologies can also be used from suborbital platforms, aircraft, drones and ships and as portable devices to provide data on marine debris at the appropriate scales; in this sense they are an important element for the development of GOOS Regional Alliances (Moltmann *et al.* 2019).

The use of low-flying drones and small aircraft is also an important component of observing systems and can enhance monitoring with high spatial and temporal resolution data. A combination of drones and satellite imagery was recently experimented with to detect typical household items as floating plastic targets. Topouzelis *et al.* (2019) confirmed that floating plastics are seen from space as bright objects and demonstrated the benefits of using very high (around 0.02 metre) geo-spatial resolution imagery from drones to improve geo-referencing of Sentinel-2 data, resampled to 10 m resolution. In the MALINOR (Mapping marine litter in the Norwegian and Russian Arctic Seas) project[13] different methods for visual mapping of plastic litter in beach areas (i.e. satellite images, multicopter, and wing-drones) are being compared and automatic image management and machine learning used to quantify amounts of marine litter. The OPTIMAL project provided an evaluation of existing and planned sensor capabilities for detecting/quantifying marine litter and is feeding into the Scientific Committee on Ocean Research (SCOR) Working Group 153: Floating Litter and its Oceanic TranSport Analysis and Modelling (FLOTSAM).[14]

Ships

Ships have played an important role in the network of platforms used to collect data on marine debris. However, observations from ships are relatively sparse, vulnerable to weather conditions, and often sensitive to the type of ship and the expertise of the operator. Studies using the ship's log to calculate particle abundance report values per km^2 while studies using flow meter readings usually report concentration values per m^3, making comparisons generally impossible (Rivers *et al.* 2019).

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For example, Maes *et al.* (2017a) have suggested that the bow wave effect may cause less water to be filtered through the net than would be calculated using the ship's log, leading to an underestimation of plastic abundance. Others have shown that plastic abundances were lower when using flow meters, most probably because of the shorter distances compared to on-board instruments (Rivers *et al.* 2019). The general conclusion is that when flow meters are used the results of plastic abundance should be reported per km^2 and per m^3 (Rivers *et al.* 2019).

Ships also provide a platform from which a broad variety of sensors and samplers can be used for comprehensive study of the entire water column from the seabed to the surface. In view of the small number of research vessels, ships of opportunity[15] have great potential to considerably improve coverage by increasing the number of visual observations and using ship-borne autonomous systems (e.g. Ferrybox[16]).

Autonomous platforms

Floats, gliders (both sea gliders and wave gliders), and autonomous surface and underwater vehicles, equipped with sensors and cameras, are being deployed to survey marine litter and plastics throughout the water column, along the seabed and in remote areas, such as the hadal depths and under sea ice. For example, photographic surveys from 2013 to 2018 were used in a novel experiment to backdate sea floor macrolitter in the Mediterranean by using product codes and branding to identify items such as aluminium cans that date as far back as the 1980s (Cau 2019). It has been proposed that this technique could be used on fishing vessels to monitor seabed litter more consistently.

There are now multiple programmes, using a variety of Lagrangian platforms,[17] which form the backbone of the Global Ocean Observing System (Moltmann *et al.* 2019) and play an important role in tracking marine debris and understanding its pathways. There is currently an archive of more than 30,000 drifter years of historical trajectories obtained from these networks of thousands of active satellite-tracked drifting buoys reporting hourly (Maximenko *et al.* 2019). Satellite trackers, attached to large debris such as fishing nets or containers, facilitate retrieval of this debris from the ocean. There are also specialized Lagrangian tools developed to study the drift of debris and other pollution in focused regional projects. For example, Meyerjürgens *et al.* (2019) built compact surface drifters and used them on the southern North Sea shelf to study transport patterns in the nearshore zones and the beaching-refloating dynamics of marine debris. The expanding Animal Telemetry Network is also useful for marine debris data collection, as well as for monitoring the interaction of debris with marine life.

Fixed point observatories

Fixed point observatories are important and efficient platforms to monitor temporal variability and, in particular, long-term trends in environmental conditions such as deposition of marine debris and plastics and climate variability (Centurioni *et al.* 2019). Moored platforms have the great advantage of being able to carry sensors and samplers. At present there are about 120 open ocean observatories (OceanSITES[18]) and even greater numbers of coastal and shelf observatories that can be used for marine debris observations. Some of the observatories are cabled and can produce large volumes of real-time data. Their locations require close coordination with satellite observations and numerical models.

Benthic landers and crawlers

A range of devices have been developed which remain on the seabed for protracted periods of time. Some of them photograph the seabed repeatedly or can take time series sediment samples while crawling over the seabed. They are usually left on the seabed for long periods before they are recovered (e.g. Wally[19]). The crawlers can provide unique information about the arrival of debris on the seabed.

Sensors

Today the deployment of advanced sensors such as those described makes the division between direct and remote observations less distinct. Aerial surveys and remote technologies provide rich information on debris on ocean or shoreline surfaces. However, as the largest fraction of marine debris ends up in the ocean water column or buried in marine sediments (e.g. van Sebille *et al.* 2015; Koelmans *et al.* 2017b; Chubarenko *et al.* 2018; van Sebille *et al.* 2020) direct visual measurements remain critical for comprehensive monitoring and need to be integrated into future operational remote sensing operations (Garaba and Dierssen 2018; Goddijn-Murphy *et al.* 2018).

Sensors currently being used for detection and polymer identification of plastic debris are deployed on three types of platform. These are satellites: Sentinel-2/ MSI HR Multispectral 450 to 1,400 nm wave, TanDEM-X HR SAR instrument, and WorldView3 VHR Multispectral 400 to 2,365 nm, and PlanetScope VHR Multispectral 455 to 860 nm (Cubesat); airborne: SASI VHR Hyper 950 to 2,450 nm, APEX VHR Hyper 372 to 2,540 nm, AVIRIS-NG VHR Hyper 380 to 2,510 nm; and handheld: ASD FieldSpec Pro VHR Hyper 350 to 2,500 nm, Spectra vista corporation VHR Hyper 350 to 2,500 nm handheld, Spectral evolution VHR Hyper 350 to 2,500 nm.

Underwater deep cable systems are able to carry sensors which can be combined with sampling and post-retrieval analysis (Wang and Wang 2018). Other imaging systems developed for macrofauna or microorganism studies can also be used to study plastic debris. For example, a towed camera and human-assisted semi-automated image analysis (BIIGLE–Bio-Image Indexing and Graphical Labelling Environment-database) were used to track increases in debris between 2004 and 2014 in the eastern Fram Strait between Greenland and Svalbard, Norway (Tekman *et al.* 2017).

Modelling

Simulation models of surface currents, focused on Lagrangian methods and using satellite data, are being deployed routinely to identify areas where floating plastics are likely to concentrate in the open ocean (Zambianchi *et al.* 2017; Palatinus *et al.* 2019; Wichmann *et al.* 2019[20]). Advances in coupled global dispersal models with field observations of surface concentrations of plastic pollution and inputs from riverine sources have enabled researchers to go from estimates of the relative size of different accumulation zones (e.g. Lebreton *et al.* 2012) to estimates of global mass budget for positively buoyant macroplastic debris and to develop future projections (e.g. Koelmans *et al.* 2019; Lebreton *et al.* 2019; Lebreton *et al.* 2020; van Sebille *et al.* 2020).

However, predictive modelling of marine litter distribution is more challenging at the near shore since local geomorphological and hydrological features can affect spatial distribution (Galgani *et al.* 2015). Attempts to model litter deposition and accumulation on substrates have been limited to coastal areas and beaches (e.g. Critchell and Lambrechts 2016) and the results from predictive models for marine litter density on the seabed are often not in agreement with observed data.

In response, machine learning models are now successfully being used to predict distribution and quantities of marine litter on the sea floor. Franceschini *et al.* (2019) trained a set of Artificial Neural Network models to determine which environmental variables affect litter distribution and to predict quantities on the sea floor in the central Mediterranean. The results coincided with earlier field studies that showed sea floor debris occurring in large quantities in coastal canyons (Buhl-Mortensen and Buhl-Mortensen 2017) and point to these techniques, potential to identify potential hotspots.

Imaging and data analysis

High resolution cameras fixed on all the different types of platforms have been used to monitor marine debris nearshore. Analysis of high-resolution images requires advanced interpretation techniques to eliminate environmental perturbations from ocean bright targets (breaking waves, white caps, sea foam, surface-reflected glint), clouds and cloud shadow (e.g. Garaba and Dierssen 2018). In monitoring marine debris, True Color RGB images provide crucial complementary information about the apparent colour and shape of litter but do not provide information on the litter's physical and chemical composition. Machine learning, combined with hyperspectral information as demonstrated by Acuña- Ruz *et al.* (2018), will help increase the value of high-resolution visible imaging as an essential component of any future integrated marine litter observing programme to identify hot spots.

3.2 Monitoring programmes, indicators, data networks and platforms

3.2.1 Monitoring programmes

One of the clear conclusions from the review of all the different types of measurement and monitoring, going on today and historically, is that they remain fragmented and hard to compare and not easily subsumed within indicator processes. It is clear that there is an urgent need for further improvements in standardization, harmonization, and the interoperability of datasets and platforms if effective global monitoring programmes are to be implemented.

Various efforts are now being designed to address this issue. For example, in an evaluation of 174 studies Serra Gonçalves *et al.* (2019) found that 27 per cent reported marine debris densities in metrics that were not comparable; nearly 10 per cent failed to report basic parameters, such as the date of the sampling; and nearly 20 per cent failed to report the size of the collected debris. Maximenko *et al.* (2019) have proposed a design for an Integrated Marine Debris Observing System (IMDOS) to monitor and assess the risk posed by marine debris. The goal of IMDOS is to look at exposure and concentrations, and vulnerability or harm to the system, with the aim of accurately estimating the amount of debris in a region, and the fluxes in and out, and computing the risks of environmental impacts. The results can be combined with data from other observation systems to enable diagnoses and responses to be developed. Large gaps in knowledge still exist in regard to both freshwater sources and marine environments (Schmidt *et al.* 2017; Best 2019). Ideally, monitoring approaches should be harmonized, cover the whole size spectrum of plastics, and

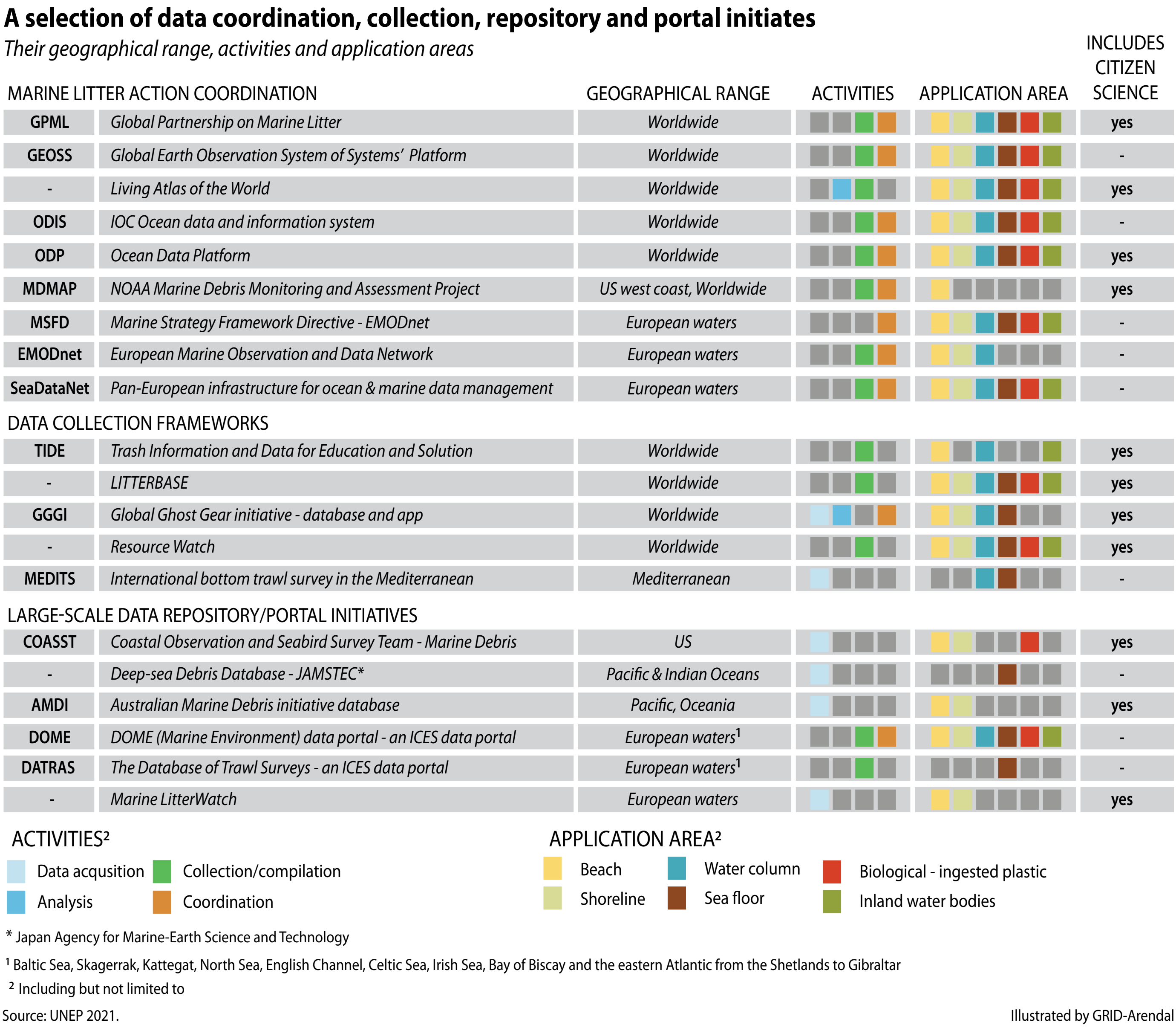

Figure 8: A selection of data coordination, collection, repository and portal initiatives

be designed to capture the spatial and temporal dynamics of the different fractions (Schmidt *et al.* 2017; GESAMP 2019; Maximenko *et al.* 2019).

For marine systems there are currently 15 major operational monitoring programmes (Maes *et al.* 2019). They cover both macroplastics and microplastics across all the marine compartments. For freshwater systems, new guidelines have been developed (UNEP 2020b,c,d) which build on the marine programmes (GESAMP 2019) and cover reservoirs and wastewater treatment plants. There are also a growing number of global platforms and databases supported through large non-governmental organizations (e.g. the International Coastal Cleanup, led by the Ocean Conservancy; Project AWARE, originating in the diving community; and the 5 Gyres Institute on Microplastics).[21]

Efforts are also under way to standardize and harmonize sample collection, analysis and reporting methods across both freshwater and marine systems (Isobe *et al.* 2019; Maximenko *et al.* 2019; Michida *et al.* 2019). The following sections review the different components of monitoring and observing systems and networks, including baseline data, indicators and information flows coming from direct observations using platforms, sensors and samplers, and remote sensing of marine debris using high spatial resolution imaging, optical spectro-radiometer techniques, and radar sensors (Figure 8).

3.2.2 Baseline data and indicators

One of the major needs for monitoring programmes, beyond the technical and operational requirements of government agencies, concerns the development of indicators for different policy measures. In a number of instances indicators have been developed with a view to future monitoring capabilities. For example, the measurement of floating plastics in the open ocean for Sustainable Development Goal (SDG) 14.1.1b[22] for floating plastic debris is being approached at two levels:

- Level 1: Globally available data from earth observations and modelling;
- Level 2: National data which will be collected from countries (through the relevant Regional Seas Programme, where applicable (i.e. for countries that are a member of a Regional Seas Programme).

The metadata for these two sources are being developed through the *Guidelines for the Monitoring and Assessment of Plastic Litter in the Ocean* (GESAMP 2019) with the aim of beginning data collection in 2021.

Indicators (i.e. specific, observable and measurable characteristics that can be used to show the changes or progress a programme is making in regard to achieving a specific outcome) and baseline data (i.e. information used to compare subsequent data collected) are also available for implementing a variety of regional and national policies (e.g. the Regional Seas Conventions and Action Plans, European Environment Agency 2019b). The main aim is to provide reliable information in order to set targets and baselines for policy decisions, such as baseline concentrations of beach litter (Hanke *et al.* 2019). Streamlining all the various indicator sets is also important, especially in the case of transboundary or cross-border issues such as marine litter, as it establishes a common understanding of priorities and monitoring. To date, the majority of indicators are focused on downstream processes and impacts, rather than on prevention measures or their effectiveness.

The establishment of baselines for different indicators and measures relies upon agreement on definitions and methods and the intercalibration, where necessary, of different methods (e.g. Maes *et al.* 2017a; Maes *et al.* 2018; GESAMP 2019). An example is BASEMAN[23] (Gerdts 2017), an interdisciplinary and international collaborative project that is bringing together experienced scientists from different disciplines and countries to undertake detailed comparisons and evaluations of all the analytical methods used for sampling, identification and quantification of microplastics in order to enable baseline measurements of the abundance and distribution of microplastics in the environment.

SDG indicator 14.1.1 is an index of coastal eutrophication and floating plastic debris intensity. Its goal is to prevent and significantly reduce marine pollution of all kinds by 2025, particularly from land-based activities, including plastic debris and nutrient pollution. Examples of the important role of regional activities in developing common agreement on definitions include:

- The OSPAR Commission has three indicators (beach litter, plastic particles in fulmars' stomachs, and seabed litter), with indicators for other biota and microplastics under development (OSPAR 2020);
- HELCOM, the Baltic Marine Environment Protection Commission or Helsinki Commission, is working on the development of three indicators for marine litter: beach litter, litter on the sea floor, and microlitter in the water column (HELCOM 2017, 2018);
- The Mediterranean Action Plan (Barcelona Convention for the Mediterranean) has indicators for marine litter, trends in the amount of litter washed ashore and/or deposited on coastlines, trends in the amount of litter in the water column (including microplastics on the sea floor), and trends in the amount of litter ingested by or entangling marine organisms, focusing on selected mammals, marine birds and marine turtles (UNEP/MAP 2015);
- The Northwest Pacific Action Plan has an ecological quality objective that marine litter does not adversely affect coastal and marine environments, as well as an indicator for marine plastic litter (Northwest Pacific Action Plan 2017).

Candidate parameters for specialized global monitoring programmes are also being proposed. For example, Brown and Takada (2017) have suggested a number of biomarkers including stable isotopes associated with bird populations as ways to monitor marine debris and pollution in the North Pacific. Tavares *et al.* (2016) have proposed monitoring debris in the nests of the brown booby (*Sula leucogaster*) as a potential indicator of the abundance of specific items in surrounding marine waters. The northern fulmar (*Fulmarus glacialis*) has been used to monitor plastic pollution in the North Atlantic seas (OSPAR) for several decades (Avery-Gomm *et al.* 2018). There are calls for seabird species in other ocean basins to be added as indicator species, such as the wedge tailed shearwater (*Ardenna pacifica*) in the pantropical regions.

3.2.3 Data networks and platforms

Efforts to create a global platform for marine debris data acquisition, streaming, quality control, and distribution to users are currently under way. The Global Partnership on Marine Litter (UNEP 2020a) (Box 6) is supporting the development of a digital platform[24] as the principal mechanism for linking existing marine litter information systems. Some of the challenges are that the systems vary according to their maturity, policy priorities, and the extent to which there is harmonization of formats and protocols for different compartments (beach and sea floor), particle size (macro- and microlitter) and geographic scales (e.g. regional, national). A number of marine litter data platforms assemble local observations for use in large-scale monitoring. These platforms are generally operated by national agencies (e.g. governmental agencies such as NOAA, CSIRO and SOA), regional bodies (e.g. OSPAR in the Northeast Atlantic), UNEP (e.g. UNEP/MAP in the Mediterranean and UNEP/NOWPAP in the Northwest Pacific), coordinated through joint efforts (such as the EU Marine Strategy Framework Directive, MSFD) or managed by non-governmental organizations (NGOs) which rely on coordinated crowdsourcing (e.g. the Trawlshare Program and the International Coastal Cleanup[25]) (GESAMP 2019). Some observations are collected on a regular basis, while others are opportunistic or acquired in the course of short-time projects, experiments or initiatives.

An example of an established European Union level data partnership and platform is the European Marine Observation and Data Network (EMODnet). EMODnet contributes to the large-scale collection and harmonization of environmental data in European seas, including the North Atlantic Ocean (OSPAR) and the Baltic (HELCOM),[26] the Mediterranean (UNEP/MAP) and the Black Sea (the Black Sea Commission), which are at different stages of development.[27] Currently the platform contains information from 518 beaches and 4,772 surveys in 29 countries (Maximenko *et al.* 2019). The aim is to provide reliable information in order to set targets and baselines for policy decisions. EMODnet Chemistry, one of seven thematic data portals, covers data on contaminants (hydrocarbons, metals, pesticides, radionuclides) and has recently been extended to cover marine debris with a focus on beach litter, sea floor litter (collected by fish trawl surveys) and microlitter (microplastics). Reporting of beach litter data in EMODnet uses OSPAR, MSFD, UNEP/MAP or UNEP-Marlin protocols.

For data on bottom trawl litter the ICES DATRAS database is combined with reports from some of the international bottom trawl surveys in the Mediterranean (MEDITS) (Cau *et al.* 2019).

Box 6: The Global Partnership on Marine Litter

The Global Partnership on Marine Litter (GPML) (UNEP 2020a) was launched at the United Nations Conference on Sustainable Development (Rio+20) in June 2012. The multi-stakeholder partnership provides a platform for cooperation and coordination; sharing ideas, knowledge and experiences; identifying gaps and emerging issues; and harnessing the expertise, resources and enthusiasm of all stakeholders (including the private sector, civil society, NGOs and regional bodies) working to reduce and prevent marine litter and plastic pollution from land- and sea-based sources. Specific objectives include reducing the leakage of plastics into the ocean, through improved design, the application of the 3Rs principle (reduce, reuse, recycle), encouraging closed-loop systems and more circular production cycles, maximization of resource efficiency, and minimization of waste generation.

One of the Partnership's flagship projects is a Massive Open Online Course (MOOC) on Marine Litter, which is now available in 10 languages.

Key to the GPML's evolution is the development of its Digital Platform. This one-stop-shop, mostly open-source platform compiles different resources including from innovative sources and integrates data from source-to-sea and throughout the plastic life cycle relevant to, for example, SDGs 6 (Clean Water and Sanitation), 11 (Sustainable Cities and Communities), 12 (Responsible Consumption and Production) and 14 (Life Below Water). The platform features accurate data and information on marine litter, plastic pollution and related topics, stakeholders, action plans, initiatives, technologies, events and training, policies, and technical and financing resources. In addition, the platform connects stakeholders through a "match-making" component in order to guide and coordinate action.

Contact: unep-gpmarinelitter@un.org

For floating marine microplastics the SeaDataNet metadata and data formats have been adapted to deal with the diversity of information from other European sources. Data on beach litter, floating microlitter and sea floor litter will be accessible through the dedicated discovery and access service in the EMODnet Chemistry portal. DOME is another data portal used by expert groups to manage chemical and biological data for regional marine evaluations. It includes quality assurance methods such as reporting of uncertainty in evaluating data. Above all, it will include data on microplastics. EMODnet also allows the inclusion of data from autonomous instruments and sensors. In addition, the observing network system will benefit from the use of platforms of opportunity such as ships, airplanes and coastal structures. Within the thematic portals there are plans to adopt consolidated data formats to address this heterogeneity (GESAMP 2019).

To make greater use of the growing numbers of initiatives and datasets, and to facilitate joint analyses, unified definitions, standards and formats and well-developed infrastructures for data flow and storage are now needed. For example, the use of categories such as mega-, meso-, macro-, micro- and nanoplastics needs to be based on clear size ranges (e.g. Frias and Nash 2019; GESAMP 2019) and accepted by all contributors and users of the combined monitoring and observing system. Examples of open data-sharing systems and platforms which are already available and could be linked to the proposed Global Partnership on Marine Litter (GPML) (UNEP 2020a) platform include the European Marine Observation and Data Network (EMODnet), the Copernicus Data Service for the Sentinel missions, Digital Earth Africa, and the HELCOM Map and Data Service.[28] The Global Partnership for Sustainable Development Data, established in 2015 to facilitate delivery of the Sustainable Development Goals, is also supported by a platform for the global network of data providers and users, which brings together data and information using the latest opportunities and technologies afforded by the data revolution.

3.3 Networks, citizen science and community initiatives

A recent analysis of global, regional and national marine debris networks[29] underlined their importance of tackling the issue of marine debris and plastics from different perspectives.

Adoption of a gendered approach in networks, citizen science projects and participatory processes is critical. It can encourage women's empowerment and participation and help ensure greater sustainability. An example of the positive impact this approach can have is the action in the South Pacific involving the Samoan Ministry of Natural Resources and Environment and that country's Ministry of Health and its Tourism Authority, in partnership with the Secretariat of the South Pacific Regional Environment Programme (SPREP) (2017) and UNEP through the GPML. In Samoa a range of community-led activities included participation by women's organizations in workshops on waste-craft, i.e. turning waste into saleable consumer items[30] Asari *et al.* (2019) emphasize the plastic waste management challenges in Pacific Island countries, where lifestyles are changing and people are increasingly concentrated in urban areas. Although marine plastic is a critical issue in this region, the authors point out that few data are available on its volumes and impacts. To gather information on plastic use and disposal they carried out a survey of Samoan households.

The growth of marine litter networks has helped to catalyse beach-cleaning activities worldwide. Despite their sometimes patchy geographical distribution and sparse timetables, beach clean-ups have shown great potential for the crowdsourcing of qualitative and quantitative marine debris data through coordinated surveys using approved protocols. Examples of coordinated surveys include the NOAA Marine Debris Tracker; the European Environment Agency Marine Litter Watch; the Ocean Conservancy TIDES; the JRC Floating Litter app; and the Marine Conservation Society surveys in the United Kingdom[31] (González-Fernández and Hanke 2017; Turrell 2019). These projects are important because, compared with more opportunistic platforms, they collect data that can easily be translated into number of pieces of plastic per square kilometre, which is required for reporting on indicator 14..1.b for Sustainable Development Goal (SDG) 14 (Conserve and sustainably use the oceans, seas and marine resources for sustainable development).[32]

During the past decade there has also been an upsurge in marine litter-related citizen science initiatives (e.g. Nelms *et al.* 2017). The UNEP stock-taking responses (UNEP/AHEG/4/INF/6) suggest that citizen science programmes effectively connect interested parties, as well as enable very specific data collection. Social and behavioural scientists note that behavioural change campaigns, which may include citizen science projects, change public motivation and awareness faster and more cost-effectively than, for example, policy tools (SAPEA 2019).

There have been successful citizen science initiatives on a range of aspects of marine litter, including monitoring litter on beaches and in rivers; tracking and analysing microbeads and plastic pellets in the environment; measuring the transport and deposition of marine litter; determining its composition and, specifically, the presence of different forms of microplastics; examining social aspects such as people's behaviour in reducing littering on coasts; analysing interactions with biota; and observing toxic effects (Hidalgo-Ruiz and Thiel 2015; Wyles *et al.* 2016; Zettler *et al.* 2017). For example, International Pellet Watch involved citizens from 17 countries in collecting plastic pellets from beaches, which were sent to Tokyo, Japan for laboratory analyses as inputs to mathematical models of dispersion (Heskett *et al.* 2012). The characteristics of pellets (small size, easy to recognize), as well as their worldwide distribution, make them appropriate for both citizen science and awareness-raising actions[33] (Yeo *et al.* 2015).

Permanent observatories are active internationally, and there are campaigns such as the Great Nurdle Hunt.[34] As well as recording the occurrence and amounts of stranded pellets, citizens monitor a standard set of environmental variables including those relevant to the weathering of pellets such as their colour, which is indicative of "new" and "old" pellets. Up to now, data have been compiled on the composition, origin and loads of pollutants adsorbed by pellets. Citizen science actions have been found to be of great help in tackling the temporal dimension and providing key information related to processes and dynamics, for example on the beaches of the Great Lakes region (Vincent *et al.* 2017). Barrows *et al.* (2018b) have shown that citizen science-based research on microplastic concentrations can be valuable in quantifying microplastic and microfibre abundance in a large, mixed land-use watershed.

Many citizen science initiatives and projects use mobile phone applications to gather, store and share data. Examples include[35] the 2minutebeachclean campaign, in which citizens monitor beach litter, clean it up and record the status on a mobile app; Beat the Microbead, in which the participants use a mobile application to scan barcodes of cosmetic products and check for the presence of microbeads; Coast Watch Microlitter, in which volunteers monitor visible microlitter and fill out a form via a mobile app or an online form to produce a microlitter map; Community Beach Clean (United Kingdom), in which participants monitor beach macrolitter and communities are brought together to clean up beaches; the International Coastal Cleanup, in which participants provide long-term global data on plastic through Clean Swell, a mobile app; RIMMEL (Europe), in which volunteers monitor visible macrolitter floating on rivers while standing on a bridge, or where rivers enter the ocean, and record the macrolitter seen during a specified amount of time using a mobile app to provide inputs for statistical models of riverine flows into marine environments; and The Plastic Tide, in which volunteers monitor plastic litter in drone photos by spotting and tagging the litter in online photographs

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and help to train algorithms to recognize plastics automatically. Other participatory processes have also been used to generate data on marine litter. Coleby and Grist (2019) describe a process of prioritized area mapping for multiple stakeholders using geospatial modelling to address marine plastic pollution in Hong Kong. They built map layers of the status of plastic waste, ports and shipping intensity, and ecological insecurity with stakeholders. They were then able to identify the regions of most concern and generate a Prioritized Area Map to characterize marine plastic waste linkages to land- and sea-based sources. Public participation in building datasets concerning the sources and impacts of marine debris has also been shown to be effective in quantifying the social factors that create and inhibit mitigation and the resolution of conflicts. In the Bay of Fundy in New Brunswick, Canada marine debris originates from the interaction of multiple industries within a small area, including aquaculture and inshore fisheries. Conflict between these two stakeholders contributes to both debris production and failure to mitigate. Gear entanglement creates debris that threatens transportation safety, wildlife and the local economy. Rehn *et al.* (2018) showed how Public Participation Geographic Information Systems mapping was used to assemble diverse data sets collected by different stakeholders and to stabilize a common view of what constituted debris, debris locations and threats during a three-year period.

A concern of policymakers in using data collected through citizen science initiatives has been the consistency and veracity of the information. However, it has been shown that as long as adequate background information is provided, most issues related to the reliability of data from citizen science activities can be solved (Garcia-Soto *et al.* 2017). In some instances protocols have been established that enable citizens to contribute directly to monitoring systems. For example, the European Environment Agency's Marine LitterWatch initiative engages with citizen scientists via a mobile phone app to build communities that gather and analyse litter from beaches in line with the Marine Strategy Framework Directive's Technical Group on Marine Litter.[36] Another successful initiative, through a partnership between UNEP, the Wilson Center's Earth Challenge 2020 and others, brings existing citizen science data together to better understand marine litter with the goal of supporting SDG reporting.[37]

It is important to note that in addition to gathering data, coastal clean-ups can remove significant quantities of waste and litter from local areas. In 2017 the International Coastal Cleanup, involving more than 0.5 million people worldwide, removed more than 8,000 metric tons of artificial debris. As Schneider *et al.* (2018) report, however, this can pose a serious threat if there are no facilities to treat the waste post-collection and the waste collected is burned in open pits on the beach. However, as Borelle at al. (2020) point out, even if all countries met their current commitments to reduce plastic waste by 2030, it is estimated that it would still require over 1 billion people to clean up just 40 per cent of annual plastic emissions.

3.4 Technical standards and traceability of plastic pollution

3.4.1 Ecolabelling schemes for beaches

Several eco-labelling schemes set standards for water quality, environmental management, health and safety, and public access to information (UNEP 2017a). The best known are the Blue Flag Programme, a voluntary scheme which has motivated clean-up efforts in countries across the world; the Quality Coast Awards and Seaside Awards, which are relevant for smaller coastal resorts; and the Green Coast Award, which is given to beaches which have a beach management plan and community engagement to meet standards in the EU Bathing Water Directive although they do not have the built infrastructure to achieve Blue Flag status. In Costa Rica, the Bandera Azul Ecológica award is given to communities engaged in protection, clean-up and maintenance efforts.

3.4.2 Technical standards and labelling of plastic products

There are only a few internationally established and acknowledged standards and certification and verification schemes for the manufacturing and processing of plastics. They cover aspects of biodegradability, recycling and degradation during the industrial composting process and in the marine environment (Harrison *et al.* 2018; UNEP 2018a; UNEP and Consumers International 2020). Examples are ISO 15279 Recovery and recycling of plastics waste; ISO 22526 Carbon and environmental footprint; ISO/CD 22722 Disintegration of plastics materials in marine habitats; and ISO 18830 Biodegradation test. Harrison *et al.* (2018) concluded in a review of the biodegradability of plastic bags that current international standards and regional test methods were insufficient to realistically predict the biodegradability of carrier bags in wastewater, inland waters (rivers, streams and lakes) and marine environments due to shortcomings in existing test procedures, the absence of relevant standards for the majority of unmanaged aquatic habitats, and a paucity of wider research on the biodegradation of plastic materials under real-world conditions.

Lack of information and evidence about the content and breakdown of different polymers, including biodegradable plastics, is of serious concern to many plastics, composting and waste management experts, as these products do not meet expectations and can lead to less effective waste disposal since they cannot be properly managed or contained (Plastic Industry Association 2018).

Industrial guidance is provided to support recycling. For example, the Association of Plastic Recyclers (2019) has issued a comprehensive laboratory-scale evaluation that can be used to assess the compatibility of polyethylene terephthalate (PET) packaging design features such as labels, closures, dispensers and attachments with common commercial-scale recycling processes. Product developers, as well as those who specify products, can use this protocol to maintain and improve the quality and productivity of PET recycling. However, it is only applicable to "see through or clear" PET articles.

In the case of biodegradability, the published standards have been certified by organizations such as DIN CERTCO in Germany, the Japanese BioPlastics Association, Vinçotte in Belgium, the Bureau de normalisation du Québec (BNQ) in Canada, the Australasian Bioplastics Association in Australia/New Zealand, and the Biodegradable Products Institute (BPI) in the United States. These organizations use test specifications to establish third-party, peer-reviewed programmes to confirm the end-of-life performance of bioplastic materials following the requirements of the standard specifications. In the development of new materials, new standards and certifications for end-of-life scenarios need to be established. One problem is that little of the testing information is made public. Unfortunately, unsubstantiated claims that go beyond standard specifications and certification by third parties concerning the rate, time and amount of biodegradation remain largely unchallenged.

Labelling of plastic products generally includes a recycling logo with a number inside it. This label was introduced by the Society of the Plastics Industry to provide a uniform system for identifying different polymer types (see Glossary: Labelling). However, public perception of this labelling system is that it is just about recycling; in light of the low levels of recycling and recovery of plastic food and drink packaging, the United Kingdom Select Committee (2019) recommended a change in labelling to a binary system of recyclable or not recyclable.[38]

Another aspect of labelling is the claim on some labels that plastics are recycled from the ocean.[39] The popular designation "made from ocean plastic" is popular with some consumers because of its emotional appeal, but it is not the ideal solution since the aim is to prevent plastic from entering the ocean in the first case. An established set of consistent terms to describe recycled plastics from ocean-related sources does not exist, which may be contributing to the confusion. For example, plastic recovered from the marine environment is often referred to as "ocean plastic" or "marine plastic", while plastic recovered from waterways or land within a certain distance of the ocean (many use 50 km, or 31 miles) is called "ocean-bound plastic" (Jambeck 2015). The term "beach plastic" is used to designate plastic specifically recovered from beaches.

Misconceptions about plastics' biodegradability are also common (UNEP 2015; Dilkes-Hoffman *et al.* 2019a). Labelling as "biodegradable" is understood by the public to mean the product will degrade no matter what the environmental conditions. Misleading labels such as these cause consumers to underestimate the impacts of plastic production in terms of GHG emissions and the impacts of disposal (Hartley *et al.* 2018a), as well as undermining solutions by encouraging rebound effects, the over-consumption of certain goods (known as the Jevons Paradox), and increased littering of "biodegradable" products (Giampetro and Mayumi 2018; Haider *et al.* 2018; Heidbreder *et al.* 2019). These key messages resonate with a recent report by UNEP and Consumers International (2020), which sets out five recommendations for how to improve labelling of plastic packaging and provides a mapping and assessment of existing labels, standards and claims for plastic packaging. A key conclusion is that the development of clear labelling standards is vital to help reduce the risks of plastic pollution and associated hazards in the marine environment.

3.4.3 Traceability and public access to information

The traceability of plastic products across their life cycle is essential to identify areas where interventions may be needed and bring about the adoption of circular approaches. In recent years citizen science and community-led activities and organizations have come together to address these problems; examples include the brand audit of Break Free from Plastic, which looks at tackling plastic pollution across the whole value chain from extraction to disposal,[40] and the Plastic Polluters Brand Audits of shorelines to identify major corporations whose products contribute to accumulations of plastic waste that are polluting inland waters and oceans. An example of information that can be derived from such surveys is the audit undertaken on Sable Island, Canada, in a partnership between the Sable Island Institute and Parks Canada.[41] Beach litter surveys have been carried out on the island since 1984. The data collected are used to identify trends in sources of marine litter and to assist the government, corporations and citizens in finding solutions

Traceability is also important for keeping track of the toxic chemicals added to plastics during production in order to help reduce the loss of materials and value and potentially achieve better environmental management of post-consumption waste. Delivering traceability has a long history in the food supply chain and the financial sector. It has become synonymous with the use of blockchain technologies. The plastic industry has recently begun to explore the use of these technologies to establish systems that will enable data exchanges among suppliers and producers and provide traceability and transparency across what is today a fragmented supply chain.[42] The use of blockchain technologies will also help make it easier for suppliers, processors, manufacturers, moulders and brand owners to choose traceable, sustainable and circular materials. In addition, it can incentivize suppliers and manufacturers to produce traceable, sustainable and circular materials and products and provide critical life cycle information for reverse logistics, including take-back of products, materials and components (Roos *et al.* 2019).

Such approaches are in line with the New Plastics Economy (Ellen MacArthur Foundation 2016), whose goal is a system in which plastic packaging never becomes waste but can re-enter the economy as a valuable biological or technical material. There are already examples in other supply chains, such as those for textiles, where blockchain and unique traceability tags are providing information to the consumer on the biobase of a feedstock and its ecological performance.[43]

The constituents of plastic products, such as additives, are not generally disclosed, which makes it difficult for consumers to determine the sustainability of products. UNEP and the International Trade Centre (ITC) have produced guidelines based on ten principles to provide product sustainability information more clearly and reliably to the consumer (UNEP and ITC 2017). It is widely recognized that traceability and public information systems are needed, supported, for example, by QR codes that allow consumers to find out about the properties of a traced object including any positive or negative effects with which it is associated and any certification standards. Consumers of plastic products also need to be aware of the institutional relations that activate and constrain such traceability systems, which could help them understand whether they can trust the information they receive. Certification and labelling schemes should provide clear guidance on which aspects of a product they are responsible for verifying or assuring. To date, the main such schemes for plastics have focused on recyclability. As more knowledge and research reveal post-consumption impacts, plastic traceability schemes will need to make consumers more aware of the full hazards and risks of products.

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SECTION 4

CHALLENGES, RESPONSES, INNOVATIONS, SOLUTIONS AND OPPORTUNITIES

4.1 The current industrial, social and governance landscape relating to marine litter and plastic pollution

Over the past four decades there has been a quadrupling of global plastics production (Geyer 2020). Demand continues to grow, with the size of the global plastic market in 2020 estimated to be around US$ 580 billion compared to an estimated US$ 502 billion in 2016 (Statista 2021a). At the same time, it is estimated that less than 10 per cent of the plastics ever produced have been recycled (Dauvergne 2018; Zheng and Suh 2019; Geyer 2020). One of the main reasons for current low recycling rates is lack of information about the constituents of plastic products, which can lead to loss of quality through the mixing of waste streams (Leslie *et al.* 2016). Ultimately, this causes millions of tons of plastic waste to be lost to the environment or shipped thousands of kilometres to destinations where it is generally burned or dumped in waterways (UNEP 2019b). Brooks *et al.* (2018) used commodity trade data for mass and value by region and income level to demonstrate the extent to which higher-income countries have exported plastic waste to lower-income countries in East Asia and the Pacific for decades.

Other challenges include the level of GHG emissions associated with the global life cycle of conventional fossil fuel-based plastics and the growing costs of managing plastic waste. The level of GHG emissions associated with the production, use and disposal of conventional fossil fuel-based plastics have been forecast to grow to approximately 2.1 gigatons of carbon dioxide equivalent (GtCO2e) by 2040, or 19 per cent of the global carbon budget (the total annual emissions budget allowable if global warming is to be limited to 1.5o Celcius) compared with some 3 per cent today. (The Pew Charitable Trusts and SYSTEMICS 2020). Using another approach, GHG emissions from plastics in 2015 have been estimated to be 1.7 GtCO2e and projected to increase to approximately 6.5 GtCO2e by 2050, or 15 per cent of the global carbon budget (Zheng and Suh 2019). The estimated global cost of municipal solid waste management is also set to grow from US$ 38 billion in 2019 to US$ 61 billion in 2040 under a business-as-usual scenario (Kaza *et al.* 2018). Even with increased taxes and government regulations, constraints on resources and reduced demand due to stockpiling (Business Research Company 2020), annual ocean plastic pollution is projected to triple by 2040 (The Pew Charitable Trusts and SYSTEMIC 2020).

As research on and knowledge about the diverse impacts of plastics increase (Lyons *et al.* 2019; Maes *et al.* 2019; Dauvergne 2020), concern on the part of the general public and governments is escalating (Avio *et al.* 2017; Borrelle *et al.* 2017; Maeland and Staupe-Delgado 2020). Many global, regional and national activities are helping mobilize the global community to bring an end to marine plastic pollution (UNEP 2018d). For example, municipalities and large firms have been reducing waste flows to landfill (Dauvergne 2018); regulatory processes are expanding, driven by growing evidence of the

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risks posed by plastics as well as by public pressure (Koelmans *et al.* 2017a; GESAMP 2020a); and there has been an upsurge in local activism, local government actions to increase curbside collections and recycling, community clean-ups, and public awareness campaigns (Schneider *et al.* 2018).

Successes at the local and national levels are being supported by policies and legal developments at the regional and international levels, for example through marine litter action plans developed within the framework of the UNEP Regional Seas Programme. Thirteen of the 18 entities in the Regional Seas Programme have adopted marine litter action plans, with another three regions currently drafting such plans (Section 4.2.3). A number of regional and national legislative efforts also aim to reduce marine litter directly (Black *et al.* 2019).[44]

In addition, there are international commitments by United Nations Member States to reduce marine pollution and litter, especially from land-based sources, as part of the 2030 Agenda for Sustainable Development Goal (SDG) 14 (Life Below Water) (UN General Assembly 2015; UNEA 2018). Numerous organizations within and outside the United Nations support these global efforts and are working on the development of legal mechanisms to this end. The international instruments and bodies involved include the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal, the Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade, and the Stockholm Convention on Persistent Organic Pollutants (POPs) (Chen 2015; UNEP/Stockholm Convention 2017; Raubenheimer and McIlgorm 2018), the International Maritime Organization (IMO), the Committee on Fisheries of the Food and Agriculture Organization of the United Nations (FAO), and the Conference of the Parties of the London Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter and its Protocol (Lyons 2019). Trade arrangements such as those covered by the Basel Convention and the World Trade Organization (WTO) also play a central role in the global plastics economy, which is why many governments are already taking trade-related measures to tackle plastic pollution (Birkbeck 2020; Borrelle *et al.* 2020; The Pew Charitable Trusts and SYSTEMIQ 2020).

The most important step required is to reduce the overall amount of plastic waste produced and its impacts (European Union 2019b). This means phasing out specific plastic products, introducing extended producer responsibility, and reshaping the established linear take-make-dispose economy to one in which material flows are part of closed-loop and resource-efficient circularity (Lieder and Rashid 2015; OECD 2016; Forrest *et al.* 2019; UNEP 2019b; Karasik *et al.* 2020; Raubenheimer and Uhro 2020). Many countries, including the European Union Member States and Japan, have set in motion action plans for circularity and, in some cases, zero pollution by implementing a waste hierarchy in which prevention, reuse and recycling are favoured over landfill (European Commission 2018b). The Ellen MacArthur Foundation has partnered with UNEP to launch the New Plastics Economy Global Commitment to inspire action by

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hundreds of key actors across the plastics value chain aimed at keeping plastics in the economy and out of the environment.

Concerted efforts at many levels will be needed to move towards circularity, linking business processes and social awareness with policies and consumer actions to significantly reduce the volume of fossil fuel-based plastics being produced, improving the design of products to reduce levels of waste and enhance decentralized recycling of materials (Joshi *et al.* 2019), eliminating unregulated plastic waste streams, and improving standards for the regulation of materials such as biodegradable plastics[45] (Dauvergne 2018; Carney Almroth and Eggert 2019; Forrest *et al.* 2019; Zheng and Suh 2019; Borrelle *et al.* 2020; Lau *et al.* 2020; The Pew Charitable Trusts and SYSTEMIQ 2020; UNEP and Consumers International 2020; WWF, the Ellen MacArthur Foundation and BCG 2020).

The Ellen MacArthur Foundation (2020) advocates that recycling needs to be proven to work "in practice and at scale", which is generally not the case due to lack of infrastructure and local facilities as well as the chemical complexity of the plastics used in many consumer products. There are also some who argue that an integrated economic and technical solution, catalysed through a voluntary industry-led contribution, is central to arrest plastic waste flows by making used plastics a valuable commodity, incentivizing their recovery, and accelerating the industrialization of polymer-to-polymer technologies (Forrest *et al.* 2019). However, as Borrelle *et al.* (2020) conclude, without significant reductions in plastic waste generation there is little prospect that the volumes of marine plastics will decline.

Some bottom-up actions are beginning to demonstrate that they can help reduce particular forms of marine plastic pollution (e.g. plastic grocery bags, products containing microbeads, plastic bottles) (Xanthos and Walker 2017; Dauvergne 2018; Schuyler *et al.* 2018). However, no single-solution strategy can reduce annual leakage of plastics to the ocean, even below 2016 levels, by 2040 (Borrelle *et al.* 2020; Lau *et al.* 2020). For example, an ambitious recycling strategy with a significant scale-up of collection, sorting and recycling infrastructure and design for recycling could reduce leakage in 2040, but only by 38 per cent relative to business-as-usual (i.e. 65 per cent above 2016 levels). Through the implementation of multiple synergistic system interventions both upstream and downstream, plastic pollution flows to the ocean could be reduced to an estimated 5 million metric tons per year by 2040, a reduction of 80 per cent relative to business-as-usual (The Pew Charitable Trusts and SYSTEMIQ 2020). However, the large volumes of waste from densely populated coastal areas, agricultural run-off, transport, wastewater treatment, greywater, waste export and fisheries are deflecting the full environmental and social costs into the global commons and contributing to strikingly high levels of marine pollution, especially in Asia and in many developing countries. The impacts of the large volumes of personal protective equipment and other plastic items generated during the COVID 19 pandemic are still to be determined (Adyel 2020), but even before the pandemic the volume of plastics flowing into the oceans was estimated to be on track to double between 2010 and 2025 (Dauvergne 2018).

Many industry and civil society initiatives are aiming to "turn off the tap" of plastic production (Birkbeck 2020; Borrelle *et al.* 2020; The Pew Charitable Trusts and SYSTEMIQ 2020). Reducing plastic production through elimination, expansion of consumer reuse options or new delivery models, implemented in conjunction with other strategies such as substitution, increased collection and recycling, and secure disposal of residual waste for a maximum reduction of plastic pollution flows offers the largest reduction of plastic pollution and can often represent a net savings in costs to consumers and producers while providing the best opportunity to reduce GHG emissions (The Pew Charitable Trusts and SYSTEMIQ 2020).

The current situation, rather than being specifically designed to meet the challenges of marine plastic pollution, is a mixture of widely varying business practices, increasing levels of plastic production, and very different national regulatory and voluntary arrangements. There is little policy coordination among states, and national and subnational policies are uneven, with loopholes, erratic implementation and inconsistent standards (Dauvergne 2018; Forrest *et al.* 2019; Birkbeck 2020).

As the pressures and complexities of tackling the plastics crisis mount up, including the need to address marine litter and plastic pollution in areas beyond national jurisdiction (ABNJ), discussions on global governance processes have intensified (Borrelle *et al.* 2017; Dauvergne 2018; Schneider *et al.* 2018; UNEA 2018; Forrest *et al.* 2019; UNEP 2019d; Maeland and Staupe-Delgado 2020). Analyses have shown that none of the international policies agreed since 2000 includes a global, binding, specific, measurable target limiting the extent of plastic pollution, leading governments, businesses,[46] and many in civil society to now call for a binding global treaty on plastic pollution (Muirhead and Porter 2019; Karasik *et al.* 2020; WWF, the Ellen MacArthur Foundation and BCG 2020). Such a global agreement would need to be consistent with ongoing legislative processes covering regulation, incentives and fiscal instruments, in order to reduce marine litter and plastic pollution and improve social, economic and environmental impacts along the plastic value chain; improve reporting and data sharing by industries and producers; enhance and harmonize standards and labelling; and ensure greater transparency in trade and subsidies (Raubenheimer and Urho 2020).

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4.2 Governance, legislation, coordination and cooperation

A number of international binding agreements/conventions, protocols, initiatives and cooperation processes, such as the Global Partnership on Marine Litter (GPML) (Box 6),[47] provide a foundation for a future global governance arrangement (UNEP 2016a; Raubenheimer and McIlgorm 2018; UNEP 2018e; UNEP 2020a). The timeline for marine litter and global initiatives, laws and policies since 1960 is shown in Figure 9.

4.2.1 International agreements and initiatives on marine pollution

The *United Nations Convention on the Law of the Sea (UNCLOS)* is the most fully encompassing international instrument on pollution from marine plastics. It is the legal framework governing all marine activities, and activities that may cause marine pollution, and establishes general principles and rules for global sea governance (e.g. see Farrelly *et al.* 2020). This Convention is the only binding framework that requires countries to adopt regulations to prevent, reduce and control pollution from both marine- and land-based sources which may enter the marine environment (UNEP 2018e). It encompasses States' requirements to prevent, reduce and control marine litter from shipping and fishing activities, among others.

The *International Convention for the Prevention of Pollution from Ships (MARPOL)* is the major International Maritime Organization (IMO) convention regulating accidental discharges of pollutants from ships (IMO 2016, 2019). It prohibits the disposal of any form of plastic from ships and requires all vessels, including fishing boats, to do their utmost to prevent the loss of plastic items overboard during operations.[48] Larger vessels are also required to develop garbage management plans and/or garbage record plans to ensure that ship-based pollution is minimized. In addition, there is an IMO Action Plan to address marine plastic

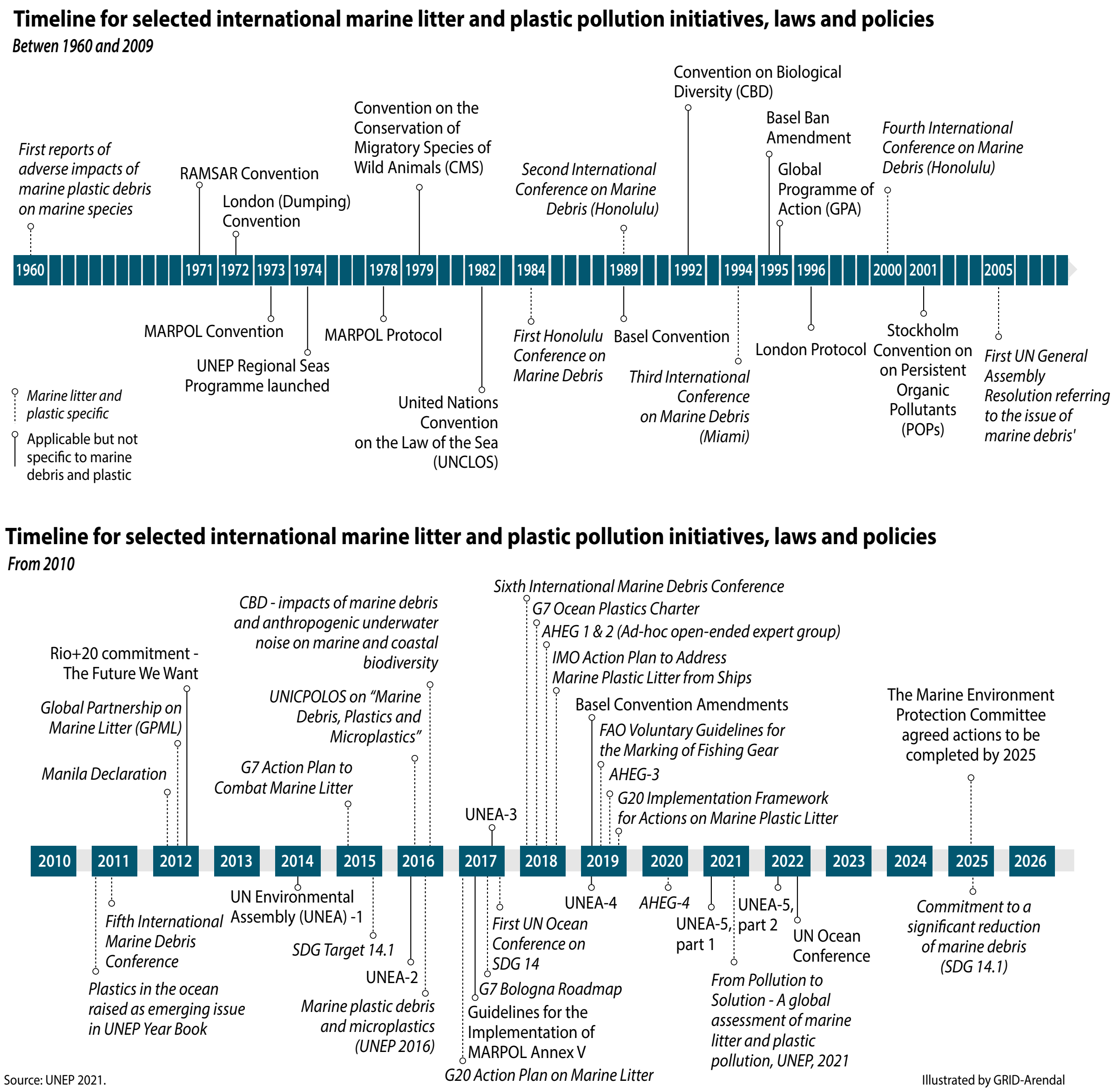

Figure 9: Timeline for global marine litter and plastic initiatives, law and policies

litter from ships.[49] *The London Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (LC)* and its *London Protocol (LP)* prevent Parties from dumping waste streams that contain plastic or similar synthetic materials into the marine environment (UNEP 2018e). Working groups established under the auspices of the IMO and the governing bodies of the London Convention and its Protocol, for which IMO discharges secretariat functions, are exploring ways to tighten mechanisms and further limit the discharge of macro- and microplastics from vessels and from waste streams authorized under the LC/LP.

The *Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal*[50] is the agreement with the greatest relevance to control of transboundary movements, environmentally sound management, and prevention and minimization of the generation of plastic waste (OECD [Organisation for Economic Co-operation and Development] 2009; Secretariat of the Basel Convention 2015; Raubenheimer and McIlgorm 2018) (Box 7). As stated in its preamble, the Parties to the Basel Convention are "mindful that the most effective way of protecting human health and the environment from the dangers posed by [hazardous and other] wastes is the reduction of their generation to a minimum in terms of quantity and/or hazard potential." (Secretariat of the Basel Convention 2015).[51]

In 2019, in Decision BC-14/12, the Conference of the Parties to the Basel Convention unanimously adopted the Plastic Waste Amendments, introducing new categories for plastic waste in Annexes II, VIII and IX to change the scope of plastic waste covered by the Convention. Since 1 January 2021, 186 States and one regional economic integration organization are bound by the amendments. This makes the Basel Convention the only global legally binding instrument that currently and specifically addresses plastic waste. Parties are now required to control transboundary movements of the plastic waste covered under the procedures established by the Convention. All plastic waste and mixtures of plastic waste generated by Parties to the Convention which are to be moved to another Party are subject to the prior informed consent (PIC) procedure, unless they are non-hazardous and destined for recycling in an environmentally sound manner and almost free from contamination and other types of waste. The Convention's provisions pertaining to environmentally sound management, as well as waste prevention and minimization, also apply to the listed types of plastic waste. Non-hazardous plastic wastes listed in Annex IX can be moved across Parties without any specific control under the Convention. The amendments as such do not imply a ban on the import, transit or export of plastic waste, but rather a clarification of when and how the Convention applies to such waste.

In 2002, in relation to plastics, the Conference of the Parties (COP-6) adopted technical guidelines for the identification and environmentally sound management of plastic wastes and for their disposal. At its 14th meeting, in Decision BC-14/13, the COP decided to update the technical guidelines on plastic wastes. A small intersessional working group has been established for this purpose and work is currently ongoing. The draft updated technical guidelines on the identification and environmentally sound management of plastic wastes and for their disposal were presented at the 12th meeting of the Open-ended Working Group of the Basel Convention.

The *Stockholm Convention on Persistent Organic Pollutants (POPs)*, which is binding on 184 Parties as of May 2021, has long controlled various POPs used as plastic additives with a view to their elimination or reduction. The Stockholm Convention requires Parties to prohibit, eliminate or restrict the production, use, import and export of listed intentionally produced POPs. It also requires Parties to reduce or eliminate releases of unintentionally produced POPs and has provisions on the management of stockpiles and wastes consisting of, containing or contaminated with POPs. This requirement is particularly relevant in the case of open burning of plastics which results in unintentionally produced POPs. Parties must ensure that stockpiles consisting of or containing chemicals listed in either Annex A or Annex B and wastes, including products and articles upon becoming wastes, consisting of, containing or contaminated with a chemical listed in Annex A, B or C, are managed in a way that is protective of human health and the environment.

The Stockholm Convention controls various POPs used as additives, flame retardants, water and oil repellents and

Box 7: The Basel Convention Partnership on Plastic Waste

In Section VI of Decision BC-14/13, the Basel Convention's Conference of the Parties welcomed the proposal to establish a Basel Convention Partnership on Plastic Waste and decided to establish a working group of the Partnership.[50] The goal of the Partnership is to improve and promote the environmentally sound management of plastic wastes at the global, regional and national levels and prevent and minimize their generation so as to reduce significantly and, in the long term, eliminate the discharge of plastic waste and microplastics into the environment, in particular the marine environment. Four project groups were established to support the implementation of its work plan: plastic waste prevention and minimization; plastic waste collection, recycling and other recovery, including financing and related markets; transboundary movements of plastic waste; and outreach, education and awareness-raising. Pilot projects are to be implemented under the Basel Convention Partnership on Plastic Waste to improve and promote the environmentally sound management of plastic waste and to prevent and minimize its generation.

plasticizers in plastics or in the manufacture of fluoropolymers. This requires Parties to eliminate their production and use, as well as their import and export (Secretariat of the Stockholm Convention 2020). International trade of Annex A POPs is only permitted for the purpose of "environmentally sound" disposal; however, this does not include recovery, recycling, reclamation, direct reuse or alternative uses of POPs. Wastes in this category may not be transported across international borders without taking into account the Basel Convention. These measures can be applied to littered plastic waste that sorbs toxins already present in the surrounding environment, in order to prevent the re-entry of banned POPs into the market (Raubenheimer and McIlgorm 2018).

In early 2021 the POPs Review Committee (POPRC), a subsidiary body responsible for reviewing POPs for listing in the Stockholm Convention, found that UV-328, an additive in plastic products, satisfies the screening criteria set out in Annex D, namely persistence, bioaccumulation, potential for long-range environmental transport, and adverse effects to human health and/or the environment. A decision by a future meeting of the COP could trigger its listing in the Annex to the Convention requiring Parties to take action towards its reduction or elimination.

A number of other international agreements are applicable to marine litter, including plastics. In 2016 the *Convention on Biological Diversity* adopted a decision[53] on the prevention of marine litter (Secretariat of the Convention on Biological Diversity 2016a), drawing on an earlier report (Secretariat of the Convention Biological Diversity 2016b) on the impacts of marine litter on marine and coastal biodiversity. This decision also provides a link to the work of the Ramsar Convention on Wetlands to protect migratory birds that depend on these critical habitats.[54]

Other bodies or legal agreements relating to the management and reduction of marine debris include the *Conference of the Parties to the Convention on Migratory Species of Wild Animals*, which adopted a resolution on the management of marine debris in 2014, and the FAO Code of Conduct for Responsible Fisheries, which sets out standards for fishing vessels to ensure that garbage is stored on-board and discharged effectively at port, and that the loss of fishing gear is minimized. The provisions of the Code of Conduct can de facto become binding through the application of other instruments such as UNCLOS or the *United Nations Fish Stocks Agreement (UNFSA)*, an implementing agreement which builds on UNCLOS. In addition, the disposal of fishing gear at sea is treated as disposal of garbage under MARPOL Annex V and is therefore forbidden.

As well as the amendments to the Basel Convention, trade policies are important in helping to reduce plastic pollution, for example by halting the export of plastic waste to countries without adequate waste infrastructure or putting in place import restrictions and bans on plastic waste (Birkbeck 2020). At the same time, however, governments support their plastic industries through import tariffs and subsidies (Birkbeck 2020).[55]

The World Trade Organization (WTO) has a unique role to play in regard to trade-related aspects. There are a number of concrete options, on which WTO members can take action, which would support international efforts to reduce and phase out plastic pollution and align trade policies with these objectives. They include increasing the transparency, data on and monitoring of plastic trade flows, supply chains and trade-related measures relevant to reducing plastic pollution and transforming the plastics economy; developing a shared understanding of the role of trade and trade policy in the global plastics economy, both upstream and downstream, and the development dimensions; promoting information-sharing and dialogue on trade-related policies, measures, innovations and best practices relevant to reducing plastic pollution and transforming the plastics economy; encouraging coherence between domestic and trade policies; reducing trade barriers and promoting technology transfer for goods and services that reduce plastic pollution, and promoting transformation of the plastics economy; encouraging voluntary trade-related targets and pledges to reduce the production, trade and use of unnecessary problematic plastics, including through the reduction of environmentally harmful subsidies; and using capacity building to support trade-related efforts by developing countries that help reduce plastic pollution, including through production/export of non-plastic substitutes/alternatives. The WTO and other settings, such as the Basel Convention, could also advance dialogue and action on these options at ministerial conferences and cooperate with other international organizations, intergovernmental processes and multi-stakeholder partnerships to strengthen the ability of the multilateral trading system to deliver on its core objective of sustainable development (Birkbeck 2020).

4.2.2 Soft law instruments

Various global strategies and soft law instruments support the reduction of marine litter and plastic pollution: the 2030 Agenda for Sustainable Development, the FAO Code of Conduct for Responsible Fisheries, the Global Programme of Action for the Protection of the Marine Environment from Land-based Activities (GPA) (UNEP/GPA 2020), the Strategic Approach to International Chemicals Management (SAICM) (which has adopted a Global Plan of Action), and the Honolulu Strategy (which provides a global framework for the prevention and management of marine debris) (UNEP 2018e). These non-binding agreements enable and encourage the putting in place of standards and activities across the life cycle of plastics and in some cases (e.g. the 2030 Agenda and the Honolulu Strategy) include targets and timelines.

The Strategic Approach to International Chemical Management (SAICM) and the Sustainable Development Goals also provide the basis for integrated and sustainable management of chemicals in relation to the oceans. Adopted by the First International Conference on Chemicals Management (ICCM1) on 6 February 2006 in Dubai, SAICM is a policy framework for the promotion of chemical safety around the world. Sound management of chemicals and waste is a specific target under Sustainable

Development Goal (SDG) 12 on Sustainable Consumption and Production and, with respect to oceans, chemicals and waste is also referred to under SDG 3 on Good Health and Well-being, SDG 6 on Clean Water and Sanitation, and SDG 14 on Life Below Water, which has a specific target to prevent and significantly reduce marine pollution of all kinds, particularly from land-based activities, including marine debris and nutrient pollution, by 2025. *The Sustainable Development Goals Report 2019* (United Nations 2019) reiterated that coastal areas worldwide remain affected by land-based pollutants, including sewage and nutrient run-off, leading to coastal eutrophication, degraded water quality, and impairment of coastal marine ecosystems. Analysis of the clean water indicator, a measurement of the degree of ocean pollution, shows that water quality challenges are widespread but are most acute in some equatorial zones, especially parts of Asia, Africa and Central America. The report concludes that while the SDG 14 targets are mostly aspirational rather than fully quantifiable (Cormier and Elliott 2017), analyses of trends from 2012 to 2019 show positive changes in nearly half the world's coastal regions. Further gains are considered possible, but they will require policy commitments at the country level to expand access to wastewater treatment, and reduce chemical and nutrient run-off from agricultural sources, as well as global commitments to reduce plastic debris.

The G20 adopted two declarations at its 2017 summit in Germany, one of which was the G20 Marine Litter Action Plan. A G20 Operational Framework was put forward which promotes several actions to reduce marine litter, including sustainable waste management, wastewater treatment, awareness-raising and increased stakeholder engagement. The Marine Litter Action Plan also establishes a voluntary Global Network of the Committed (GNC) to share knowledge and experience regarding the action plan. These measures were reaffirmed at the G20 summit in Japan in 2019 (G20 2019), including appropriate national actions for the prevention and significant reduction of discharges of plastic litter and microplastics to the oceans. There was also a call for other members of the international community to share, as a common global vision, the Osaka Blue Ocean Vision, which aims to reduce additional pollution from marine plastic litter to zero by 2050 through a comprehensive life cycle approach that includes reducing the discharge of mismanaged plastic litter through improved waste management and innovative solutions while recognizing the important role of plastics for society (IRP 2021). In addition, this meeting endorsed the G20 Implementation Framework for Actions on Marine Plastic Litter.

Other global initiatives include the Community of Ocean Action on Marine Pollution (UN Department of Economic and Social Affairs), which supports members' implementation of marine pollution-related voluntary commitments through exchange of knowledge and best practice, and the OECD's RE_CIRCLE project, which provides policy guidelines on resource efficiency and the transition to circularity.[56]

4.2.3 Regional instruments and key actions to improve waste management

While binding international instruments such as agreements, conventions, protocols and other initiatives provide a basis for action (Rochman *et al.* 2016b), they are constrained by their mandates. Far greater coordination and investments are needed to tackle the increasing volumes of plastic being produced which end up in the oceans. Lavers and Bond (2017), Löhr *et al.* (2017) and Raubenheimer and McIlgorm (2018) suggest that existing guidelines, such as the Basel Convention guidelines for "upstream alterations in product design", can be used to help reduce the quantity and the hazards of plastic waste. This is in line with the OECD (2016) guidelines on extended producer responsibility and supports the development of global industry guidelines that aim specifically to reduce both hazard and quantity of plastic waste.

Regional governance arrangements can potentially accelerate the uptake of legislative and industry initiatives. Some of the most important regional instruments are the Regional Seas Conventions and Action Plans[57] (refer to Annex I), a number of which include various measures to reduce marine litter, as well as monitoring and public awareness campaigns[58] (UNEP 2018d). Although ocean dumping is prohibited, not all countries are signatories to the international agreements, so that they are less effective than they could be in controlling marine litter and plastics. However, most Regional Seas instruments address industry pollution and emissions into water bodies through the duty to prevent pollution from point sources, and three Regional Seas Conventions have adopted protocols specific to the dumping of plastics from vessels.

In Africa some states have agreed under the Bamako Convention,[59] the regional instrument related to the Basel, Rotterdam and Stockholm Conventions, to strengthen the management of hazardous waste, including plastics and electronic waste (e-waste). They have also agreed to reinforce collaboration and create more synergies between the Bamako Convention and the global chemicals conventions (UNEP 2020b). The East African Community (EAC) Development Strategy outlines broad strategic goals for that region. It recognizes a lack of effective legislation, inadequate funds and services for municipal waste management, and the low priority given to solid waste management as major challenges facing member countries. Although this strategy does not have a recommended strategic intervention on waste management in general, it does include harmonization of policy interventions on the management of plastics and plastic waste and the establishment of an electronic waste management framework. Specific waste targets outlined under the EAC Development Strategy include a regional policy on the management of plastic and plastic waste in place and an EAC e-waste management framework. While this regional policy has yet to be fully developed, many countries have introduced total and partial bans on plastic products such as bags (UNEP 2018c).

In the South Pacific region the Waigani Convention, adopted in 2001, is the regional implementation of the international hazardous waste control regime and annexes of waste categories (the Basel, Rotterdam and Stockholm Conventions). The objective of this convention is to reduce and eliminate transboundary movements of hazardous and radioactive wastes, minimize the production of hazardous and toxic wastes in the Pacific region, and ensure that disposal of wastes in the Convention area is completed in an environmentally sound manner. The Waigani Convention includes each Party's Exclusive Economic Zone (200 nautical miles) rather than extending only to the outer boundary of each Party's territorial sea (12 nautical miles) as under the Basel Convention. It is also strongly related to the 1972 Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (the London Convention). There has been investment in improving waste management under the Waigani Convention through Global Environment Facility (GEF) funded work on reducing the unintentional release of persistent organic pollutants (UPOPs) and the European Union funded PacWaste projects (Secretariat of the South Pacific Regional Environment Programme 2017).

Within the Association of Southeast Asian Nations (ASEAN) there is a patchwork of governance arrangements, including general legislative frameworks for municipal solid waste (MSW); marine litter and anti-litter legislation; source reduction through material restriction; landfill regulations; waste to energy laws; some extended producer responsibility legislation; trade policies; green procurement; and recycled content policies. The Asia-Pacific Economic Cooperation (APEC) Virtual Working Group on marine debris, part of the APEC Chemical Dialogue and Oceans and Fisheries Working Group, is promoting innovative solutions to marine debris, especially through sustainable waste management. However, there is no integrating governance process to bring all these together.

Governments in East Asia, recognizing that the seas around their coastlines are among the world's most polluted, have put in place various mitigation initiatives to decrease plastic pollution, including government policies and waste management; education, media, monitoring and outreach campaigns by NGOs; and the development of alternative products and methods of production and recycling by inventors and businesses (Walther *et al.* 2020). Japan has established a legal system to promote the reduction and recycling of packaging and packaging waste, and has established national targets and a hierarchy of existing and potential interventions (Japan Ministry of Environment 2014).

Similarly, within the European Union there is a waste governance landscape comprising policy structures, regulations and standards at multiple administrative levels aimed at reducing and recovering materials over the resource life cycle. The overall approach of the European Union is demonstrated in its plastics strategy, circular economy action plan, and the Single-Use Plastics Directive (European Union 2019b; European Commission 2020). European Union Member States have also established targets to achieve a 90 per cent collection target for plastic bottles by 2029; plastic bottles must have at least 25 per cent recycled content by 2025 and 30 per cent by 2030.[60] Application of the "polluter pays principle" has also been strengthened by introducing extended producer responsibility (Arroyo *et al.* 2017).

The outcomes of the XXII Meeting of the Forum of Ministers of Environment for Latin America and the Caribbean in February 2021 included a pollution and waste agenda. Ministers highlighted the need to urgently address the issue of marine litter and microplastics and adopted a new Action Plan on regional cooperation for the management of chemicals and waste 2021–2024.[61]

4.2.4 National strategies and legislation

National-level arrangements addressing waste are very uneven (e.g. van Truong and Ping 2019; Ca 2020) and have led to situations in which waste is being distanced (both physically and psychologically) from millions of consumers by being sent to locations with poorly developed waste infrastructure (Waste Atlas 2014) or allowed to flow into the global commons (Dauvergne 2018; Birkbeck 2020). Analysis of a global policy inventory found that of the top 20 coastal countries producing mismanaged plastic waste from coastal land-based sources (based on estimates from 2010 in Jambeck *et al.* 2015), a number did not have a national policy document nor was there a reference to one in the literature reviewed (Karasik *et al.* 2020). However, increasing concerns about imports have led some receiving countries to put in place more rigorous inspection processes (O'Neill 2018), which are causing waste to be diverted to countries with less rigorous waste management standards (Dauvergne 2018).

There are a growing number of legal initiatives, including bans on certain single-use plastic products, plastic bags and microbead products (Xanthos and Walker 2017; Dauvergne 2018; Karasik *et al.* 2020). Analysis of the global policy inventory (Jambeck *et al.* 2015) indicates that the upward trend in the overall number of policies adopted at the national level to address plastic pollution in the last 20 years is largely due to new policies directed towards addressing pollution from plastic bags (Karasik *et al.* 2020). More than 60 countries now support bans on different types of plastic items (UNEP 2018d), and countries including Canada, the Netherlands, the United Kingdom and the United States have put in place legislation to ban the use of microbeads in cosmetics and personal care products (Xanthos and Walker 2017). In 2019 EU ministers agreed to ban by 2021 a long list of single-use plastic products based on surveys and monitoring of beaches and waterways (European Union 2019b).[62] Bans on specific items can be a step towards more comprehensive policies to reduce plastic production and replace plastic products with more sustainable alternatives. Costa Rica, for example, intends to become the first country in the world to ban all single-use plastic products by 2021. In Antigua and Barbuda the introduction of a ban on plastic bags has led to further measures forbidding the import of food plastic containers and utensils. In the United States a ban on Styrofoam (expanded polystyrene) containers in New York City, which had been challenged, was reinstated in 2017 and subsequently implemented in other cities and states,[63] based on evidence that it was not economically viable or environmentally effective to recycle these containers.

At the national level, levies have progressively been placed on plastic bag consumption. In Ireland plastic bags have been banned; in Germany, India, Thailand (2020) and 34 countries in Africa bans are also in place, although in some cases regulations have yet to be implemented (Babayemi *et al.* 2019).[64] In 2013 Mauritania became the first country in Africa to adopt a plastic bag ban, following the loss of up to 70 per cent of livestock due to plastic ingestion.[65] Much of what has been reported about the bans, and other levies and taxes aimed at reducing plastic carrier bag pollution, has focused on short-term effects; however, significant reductions in the consumption of plastic bags have consistently been measured within 24 months of the introduction of such instruments and typically within 12 months (Karasik *et al.* 2020). Unfortunately, for 50 per cent of regulations at national and local levels there are no monitoring systems or data to assess their effectiveness or impact; nearly one-third of the remainder registered dramatic drops in plastic bag consumption and pollution and one-fifth reported little to no impact, most probably due to lack of enforcement or affordable alternatives. There are also various loopholes and exceptions in many of these bans (e.g. concerning thickness), which reduces their effectiveness (UNEP 2018d).

A growing and important component of many national waste management policies is beach and coastal clean-up. In an analysis of the potential mitigation gains of removing plastics from the marine environment, De Frond *et al.* (2019) concluded that removing plastics not only reduced their overall volume, but also cut down on the amount of plastics entering important routes of exposure to the additives and other hazardous chemicals associated with them via wildlife ingestion. They also showed that in most jurisdictions plastic pollution prevention and clean-up are considered to be chemical pollution prevention and clean-up, and that shoreline clean-ups can remove legacy pollutants such as polychlorinated biphenyls (PCBs) from the environment. These findings are consistent with Sherman and van Sebille (2016), who considered coastal areas to be optimal locations for microplastics removal.

In areas where greater amounts of plastic debris concentrate, De Frond *et al.* (2019) calculated that plastic collection technologies could remove 31 per cent of total modelled microplastics mass by 2025 compared to 17 per cent in the North Pacific Gyre, where the plastics are also much older (Lebreton *et al.* 2019). Coastal areas contain relatively "young" particles with higher leachable content; thus, by removing plastic debris from beaches new plastic items are prevented from entering the oceans and exposing marine life to their leachable chemical content. For PCBs, De Frond *et al.* (2019) concluded that the clean-up of plastics and their associated chemicals is more efficient in terms of removing hazardous chemicals and pollutants on shorelines compared to ocean gyres because of the higher density of materials along beaches. For example, they estimated that beach collections removed approximately 85,000 times more PCBs than these gyres. However, Schneider *et al.* (2018) reported that out of 103 scientific studies on clean-up efforts, none mentioned the use of post-collection waste treatment pathways or options, or the impacts of these beach litter collection activities. A particular challenge for beach clean-

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ups is marine waste containing black plastic. Analyses show that black plastic is potentially non-compliant with regard to limits defined by the EU's Restriction of Hazardous Substances Directive (Shaw and Turner 2019). This also raises the issue that recycling electronic waste for maritime industrial uses, such as in transport, fisheries and aquaculture, is potentially an important pathway for the introduction of hazardous chemicals into the marine environment.

Recent data on global flows of electronic waste challenge conventional understanding of the trade in the associated plastic waste as a North to South flow (Lepawsky *et al.* 2017); databases of e-waste flows compiled by the authors revealed a complex web of trans-frontier movements,[66] with small groups of immigrants collecting or buying discarded electronics and shipping them to their countries of origin. This analysis shows that, under certain circumstances, when discarded electronics are refurbished and sold on informal workers may earn a living wage even where informal e-waste recycling operations are legally prohibited (O'Neill 2018). Given the multiple lives of electronic items which can be repurposed, and the plastics associated with them, Lepawsky *et al.* (2017) proposed an ethical framework and "worker scripts", based on various UN initiatives, that include worker safety, sufficient wages, and shared decision-making and profits (ILO 2017).

In addition to regulatory and legislative processes, a number of voluntary actions have been undertaken as part of national strategies (UNEP 2018d). Many have focused on single-use plastic products. They include public-private partnerships and voluntary agreements instead of bans (e.g. Austria), and voluntary reduction strategies which allow the population to change their consumption patterns and make it possible for affordable, eco-friendly alternatives to become available on the market. Examples include Fishing for Litter (KIMO, the international environmental organization for local authorities), which involves the fishing industry in collecting the waste caught in their nets; Zero Waste Cities,[67] which promotes a continuous effort to phase out waste (rather than burning or landfilling it) and to create systems that do not generate waste in the first place; and WRAP (2018), which works with government, local authorities and industry in the United Kingdom to support better recycling and innovation.[68]

4.2.5 Other types of financial and regulatory instruments

In the absence of any pricing policies for waste, industries and consumers behave as if the disposal of waste is free (Matheson 2019) although the collection and disposal of discarded goods consumes valuable resources such as labour, fuel and land. Some of these costs may be priced, but environmental costs such as those related to carbon and methane emissions are usually not priced at all while charges for improper disposal are often not enforced. Cheap and obscure prices for waste disposal have encouraged waste-intensive production and consumption patterns rather than recycling (IRP 2019). There are a range of fiscal instruments, requiring legislation, to enhance waste management and support circularity (OECD 2019). They include taxes, fees and charges, deposit-refund schemes, tradable permit schemes and subsidies.

Taxes increase the cost of polluting products or activities, thereby discouraging their consumption or production. In waste policy they are used to internalize the environmental costs of waste management and disposal, making more environmentally

harmful treatment methods costlier and creating incentives to use alternative approaches such as recovery, reuse and recycling and other actions higher up the waste hierarchy. Landfill and incineration taxes are good examples, but in some cases they have led to an increase in illegal dumping and open burning. In policies that support circularity, taxes may be used to discourage consumption of natural resources including biological resources, minerals and raw materials. Some countries have introduced a general retail tax for waste management purposes (e.g. the Jamaica Environmental Protection Levy). Specific excise fees can also be applied which internalize environmental and social costs. Sometimes referred to as advance disposal, recycling fees can be levied at the retail or production level; examples include fees on tyres and plastic bags (Xanthos and Walker 2017). Retail-led fees, although expensive to administer, are highly visible to consumers and can thus have an impact on behaviour (International Monetary Fund 2019). Taxes in individual jurisdictions can lead to regulatory arbitrage (a corporate practice of utilizing more favorable regulations in one jurisdiction to circumvent less favorable ones elsewhere) and the need for border taxes on plastics imports (Forrest *et al.* 2019). Nevertheless, there is serious interest in plastic production taxes around the world, especially when the funds can be hypothecated to improve waste infrastructure (Parts 2019; Walker *et al.* 2020).

Fees, levies and charges can be used to recover the costs of providing goods or services. Unlike taxes, use of fees and charges means the person paying gets something in return in proportion to the payment. In waste management this may include items such as municipal waste service charges or landfill gate fees. Waste management charges are generally applied locally to cover the costs of waste collection, and in some instances (e.g. landfill charges) the fees are hypothetically for improvements to waste management or the mitigation of impacts such as GHG emissions. Levies on waste disposal have the potential to complement product stewardship schemes. Levies increase the cost of waste disposal and make alternatives more attractive. In some countries funds from levies have been used to support product stewardship schemes (e.g. assisting with start-up costs). Funding support from waste levies can create disincentives for product stewardship by discouraging industry self-funding initiatives, as well as by discouraging industry ownership of the problem. Careful design of waste levies and the allocation of their revenue is critical to ensuring they support rather than work against product stewardship schemes (e.g. New Zealand Ministry of Environment 2019). For example, there are concerns that high waste levies incentivize illegal or improper waste disposal, including across borders (Interpol 2020).

Deposit refund/return systems place a surcharge on the price of a product likely to pollute the environment. In waste management this may include measures used to internalize the environmental costs of end-of-life products such as product levies, advanced recycling fees and extended producer responsibility measures. These systems are successfully used in many countries, e.g. in the Baltic countries, Denmark, Germany and Kenya (Balcers *et al.* 2019). Systems mandated by law, with clear stakeholders and role descriptions that guarantee equal treatment for all system participants (including producers, importers and retailers), are likely to be the most effective. It is also recommended that the deposit-return system cover a wide range of one-way and refillable beverage containers: return to the retailer has the clear advantage of being more convenient for consumers.

Subsidies can be used to encourage better waste management, waste reduction, and investments in improved waste management. They may take the form of direct subsidies or tax exemptions. However, a major barrier to realizing circularity is the extremely low direct cost of fossil fuel-based plastics caused by widespread subsidies (UNEP 2019d) and significant investment in fossil fuel-based chemical production (American Chemistry Council 2020; European Chemical Industry Council 2020). The low cost of fossil fuel-based plastics gives rise to perverse market price signals which put many technologies, such as those for recycling post-consumer plastics and resins (e.g. Ragaert *et al.* 2017; Rahimi and Garcia 2017), at a disadvantage (Forrest *et al.* 2019) and will considerably alter trade flows in coming years.

However, subsidies can be used in environmental policy to directly or indirectly reduce the use of something that has a proven negative effect on the environment, as in the case of GHG emissions arising from plastic production (Posen *et al.* 2017). Taking a life cycle approach to feedstock and energy substitution in the plastic industry opens up avenues to include climate mitigation (Zheng and Suh 2019), as well as substitutability, in the way subsidies are applied (UNEP 2019d). Linked to this are tradable permit schemes, which can be used to allocate emission or resource exploitation rights; such measures are used in waste policy, for example, in the United Kingdom's Landfill Allowance Trading Scheme.

4.2.6 Extended producer responsibility

Extended producer responsibility (EPR) is considered a cornerstone of waste policy, particularly in the Organisation for Economic Co-operation and Development (OECD) countries (Filho *et al.* 2019). According to the OECD's description, EPR aims to make producers responsible for the environmental impacts of their products all along the product chain from design to the post-consumer phase (OECD 2016; OECD 2019). If properly designed, EPR also alleviates the burden on public administrations of managing end-of-life products; in addition, it is important because it has been estimated that if no action is taken to reduce plastic production and consumption, businesses could face a US$ 100 billion annual financial risk by 2040 if governments require them to cover waste management costs at expected volumes and recyclability (The Pew Charitable Trusts and SYSTEMIQ 2020). EPR can incentivize waste prevention, reuse and recycling, for example by introducing clear labelling to help informal sorting processes (UNEP and Consumers International 2020; Walker *et al.* 2020). It can take many different forms and needs to be adjusted to local contexts to avoid inadequate transfer of technology (e.g. through taking into account informal workforce-based infrastructure and identifying sustainable and inclusive pathways to future-proof the livelihoods of waste pickers).

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EPR has been a main policy instrument in the EU, contributing financially to the ongoing collection and recycling of waste streams that contain plastic as well as encouraging the adoption of practices by companies, the education of consumers, and moving towards the more ambitious targets under the EU Green Deal (European Commission 2018b, 2019).[69] The EU has also adopted EPR for certain single-use items as well as fishing gear through directives that will be implemented in coming years (European Union 2019, 2020). Implementing EPR as a measure towards downstream waste management is being examined by the European Commission as a means of reducing marine litter through action on single-use plastic products and plastic fishing gear (European Commission 2018). This approach is also being developed in Southeast Asia and East Asia (ASEAN Framework of Action 2019 and COBSEA RAP MALI 2019).[70]

Under the German EPR scheme for packaging, companies pay a fee of around Euro 450 per ton; the EPR fees predominantly aim to cover the system's annual operating costs. France has also made efforts to use EPR for end-of-life boats under the London Convention. By applying this approach to the estimated 27.12 million metric tons of plastic packaging in China, Indonesia, Malaysia, the Philippines, Thailand and Viet Nam, use of EPR could raise a total of Euro 12.2 billion (WWF 2020). Although this is a very rough estimate, it gives an indication of the revenue-raising potential of EPR systems and their potential to help set up effective waste management infrastructure in Southeast Asia.

A major challenge in using EPR for marine litter is that it has become clear after a number of years that producer responsibility organizations managing the process do not assume the entire cost of managing the corresponding waste flows (Forrest *et al.* 2019), and therefore public administrations continue to sustain part of the costs that should be borne by producers and potentially included in the prices paid by consumers. Secondly, producer responsibility organizations do not sufficiently incentivize recyclability and ecodesign by individual producers; and thirdly, insufficient transparency makes it difficult for public administrations to assess compliance, among others (OECD 2016). Today EPR is limited to a small number of products (e.g. electric and electronic equipment, batteries and end-of-life vehicles).

For a legislative EPR framework to work effectively, it needs to ensure harmonization and transparency of implementation; comparability of procedures and fee levels; wider product coverage; improvements to separate collection and treatment of wastes; extension of deposit-refund schemes; and proper product design, especially circular design (e.g. design for reuse and recyclability) to improve the reuse and recycling levels of end-of-life products and the overall reduction of plastic waste to ensure the transition to sustainable circularity (Filho *et al.* 2019). Financial resources collected through EPR schemes can also do much to amplify these efforts. It is clear that the focus on low-cost collection solutions, for example collection of mixed packaging or collection of only the lowest-cost waste streams, is not sufficient to increase recycling of plastics. Instead, EPR should be seen as part of a wider policy mix of regulatory and economic instruments such as recycling targets, bans, product, material and waste taxes, pay-as-you-throw schemes, labelling, voluntary agreements, procurement policies, and information and awareness campaigns, bearing in mind that producers should not be double taxed through any combination of taxes and EPR.

Overall, EPR schemes work well if there is ecomodulation[71] rather than flat fees and where governments set national targets for waste collection, segregation and recycling, invest in national/

regional waste management infrastructure, and create a coherent and transparent EPR framework within national legislation. This framework needs to take account of local characteristics such as the role of the informal sector, monitor companies' plastics use and enforce EPR legislation. Companies could reduce the unnecessary use of plastics and transparently disclose the amount of plastic packaging they are putting on the market; take responsibility for products' end-of-life impacts, from the design and choice of materials to collection, sorting, recycling and disposal; and support the creation of EPR schemes and work with governments and other partners to improve waste management systems and raise consumer awareness. The key to the success of an EPR approach is investment and the incentivization of industry (Forrest *et al.* 2019).

4.2.7 Coastal zone management policies

Coastal zone management policies are important instruments for delivering waste abatement policies once litter has entered the environment, especially if they are implemented on a catchment-to-coast basis (Windsor *et al.* 2019). Littering on land and at sea is illegal behaviour which damages both terrestrial and marine life and has an enormous cost for society and the environment. Many governments invest significant resources in waste abatement infrastructure, policies, strategies and outreach programmes to intervene at different stages along the plastic waste pathway; however, these programmes are generally less targeted towards plastics once they have entered the environment.

Anti-litter campaigns educate the public and encourage people to improve their waste disposal behaviour, while community programmes such as the International Coastal Cleanup and other citizen science and community projects encourage members of local communities to be custodians of the environment by involving them in beach clean-up activities (Schneider *et al.* 2018; Willis *et al.* 2018). In their analysis of waste infrastructure by local councils in Australia, where litter clean-up costs were in excess of US$ 1 billion annually, Willis *et al.* (2018) showed that if the waste management proportion of the total council budget was less than 8 per cent, debris along a council's coast continued to increase. The most effective waste abatement policies were those that integrated recycling, litter prevention and programmes combatting illegal dumping.

Developing coastal zone management policies and strategies that address marine litter is a vital part of the legislative landscape, which is needed to help mitigate the problems of plastic pollution and the leaching of chemicals (Sherman and van Sebille 2016). Resource managers can optimize their efforts by focusing on waste abatement and undertaking waste recovery on beaches and close to shore, where greater quantities of chemicals and microplastics can be removed compared to the same sized area of open ocean. Coastal litter removal is also more cost-effective, as less time and resources are required to clean up beaches and shorelines than to clean up surface waters thousands of kilometres from land (De Frond *et al.* 2018).

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Marine Protected Areas (MPAs) are another important policy instrument to help reduce marine litter and plastics and their impacts on marine systems, and in recent years many large MPAs have been established around the world (Luna-Jorquera *et al.* 2019). The activities excluded or restricted in MPAs are mostly tourism and fisheries, but also mining and construction of harbours or offshore wind farms and dumping of solid materials (Lewis *et al.* 2017). However, there is growing evidence that floating marine debris from elsewhere is encroaching on MPAs, even in the open oceans, sometimes giving rise to high concentrations of microplastics being recorded (Barnes *et al.* 2018; Luna-Jorquera *et al.* 2019). In the Mediterranean an analysis of MPAs showed that they were sheltered from plastic pollution, but that this situation could change dramatically due to mismanagement of plastic litter within the MPAs themselves (Liubartseva *et al.* 2019).

4.2.8 Education and broader social policies and actions

Changing behaviour, perceptions and attitudes and raising awareness of marine litter and plastic pollution will require greater levels of ocean literacy, education and action in order to better understand the value of coastal environments and the damage caused by littering (Hartley *et al.* 2018a). A study by Hartley *et al.* (2018b) using a school pack designed for European educators and students aged 10-15 years[72] concluded that for educators, smart tools are needed to support what is already being taught within the national curricula and enable them to work together and share best practices and experience. Based on questionnaires and observations from over 6,000 pupils aged 11-13 years during 2012–2018, Kideys *et al.* (2018) reported that the marine environmental awareness training provided by the "I Know and Protect My Seas" (DTK) programme to Turkish schoolchildren proved to be very effective in changing littering behaviour as well as creating awareness and appreciation of biodiversity. Thiel *et al.* (2018) used a citizen science project to interest students in Chile, aged eight to 16, in the topic of microplastics in the marine environment. The students sampled, sorted and counted small plastic pieces on local beaches and entered the data on an interactive website. Afterwards, they reported that they had found the project interesting and fun and would be likely to participate in other environmental activities in the future. Hartley *et al.* (2015) implemented an environmental education activity with schoolchildren in the United Kingdom between seven and 18, and assessed their level of concern, understanding and self-reported behaviour regarding marine litter before and after engaging in this activity. After it, they were significantly more concerned about marine litter, had a better understanding of causes and negative impacts, and reported improved behaviour.

These studies support other research showing that knowledge acquisition is not sufficient to elicit behavioural change (Damerell *et al.* 2013; Geiger *et al.* 2019). More work is needed on engaging teachers, and on teaching methods that allow children opportunities to explore marine litter and plastic pollution through fieldwork, use of effective books (e.g. Stachowitsch 2020), and creating their own materials and responses rather than following a predetermined programme of education, in line with transformative pedagogy.

4.2.9 Social policies and communications actions

Nudges, norms, longevity of behavioural changes and behavioural economics are all important in influencing pro-environmental, behavioural change in populations (McGuire 2015; Krijnen *et al.* 2017; Geiger *et al.* 2019). To increase social awareness of marine litter and plastic pollution, and shift behaviour from use and throw away to reduce, reuse, repurpose and recycle, there needs to be greater understanding by and engagement of the public as well as clear communications strategies to ensure the success of government and industry initiatives (Dilkes-Hoffman *et al.* 2019b).

To date there has been little focus on documenting the general public's attitudes to marine litter and plastic pollution. Dilkes-Hoffman *et al.* (2019b) examined public beliefs and attitudes towards plastics in Australia, which provides insights on a global scale. Their survey results indicate that the public view plastics as a serious environmental issue. Plastics in the oceans had the highest mean rating for seriousness out of nine environmental issues, followed by two other issues relating to plastic waste production and disposal. There was an association of plastics with food packaging and convenience, but there was a more negative association with the use of plastics overall; 80 per cent of respondents indicated a desire to reduce plastics use, and the majority of respondents believed that paper and glass are more environmentally friendly packaging materials than plastics even though this is not always the case (Stanton *et al.* 2020; UNEP 2021b).

These and other studies suggest that the public is overly sensitive to plastics and potentially sees them as a bigger environmental issue than others such as climate change (Khan *et al.* 2019). However, analyses of outcomes show that many respondents did not translate their aspirations to reduce plastic use into action for a variety of reasons, for example due to habits, norms and situational constraints such as personal finance (whereby individuals can only afford to purchase items in small quantities in plastic intensive packaged amounts) or because responsibility for reducing the use of disposable plastic items was placed on industry and government. Social science research on risk awareness, consumer preferences, predictors of usage behaviour, and political and psychological intervention strategies (e.g. Heidbreder *et al.* 2019) shows that people appreciate and routinely use plastic despite a clear awareness of the associated problems. Unfortunately, what is missing from this research is a gendered perspective on norms and behaviours that would lead to a greater understanding of social actions that could be taken to reduce plastic pollution.

With respect to bio-based plastics, research indicates that there is an overall positive perception among the public (Dilkes-Hoffman *et al.* 2019a). Biodegradable plastics are generally seen as better for the environment than "normal plastics" and even "easily recyclable" plastics. However, the majority of respondents said they were unsure whether biodegradable plastics had negative environmental impacts, would like more plastic items to be biodegradable, and would dispose of bioplastic items in recycling bins. However, as highlighted by UNEP and Consumers International (2020), consumers often do not know how to dispose of biodegradable plastics correctly or facilities for handling biodegradable plastics do not yet exist. That report underlines the critical role governments and local councils play in driving the development of standards, labelling and waste management options for bioplastics and alternative materials. For example, France has passed a new law under which plastics are not allowed to be referred to as biodegradable unless they are home compostable; the European Union has indicated that it is also interested in taking action in this regard.

Building on people's perceptions of "bio-based" products (i.e. positive and negative associations, mixed feelings), Sijtsema *et al.* (2016) showed that this term is most often associated with positive environmental qualities such as "naturalness" and "environmentally friendly" but there are also negative environmental associations, linked especially to technological and health issues. This can cause uncertainty and mixed feelings and highlights both the complexity of, and a lack of familiarity with, the concept of bio-based plastics. Consumers have a holistic perception of bio-based products. They combine their perceptions of different aspects of the product in an evaluation of the whole product concept, including the product's origin, its usability, the production method, the proportion of bio-based materials used, price, packaging materials and appearance. The results illustrate the great variety of consumer perceptions, both cognitive and affective, of bio-based products and the care with which terms that are poorly understood by the general public should be introduced into policies. For example, some bio-based plastics may also contain problematic additives and substances that are not bio-based, which can make these products as problematic as products that are not bio-based for recycling/composting. This highlights the problems that can arise when poorly understood terms are increasingly being communicated to consumers on packaging and underlines the need for better education and more accessible information to tackle that problem.

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4.3 Business solutions and environmentally sound technologies and innovations

4.3.1 Identifying market failures and solutions in the globalized plastics industry

The growing quantities of discarded plastic waste are the outcome of multiple market failures linked to the low price of virgin feedstocks, the presence of subsidies, poor waste management, widespread use of plastic items, and throw-away behaviour (Borrelle *et al.* 2017; Law 2017; Dauvergne 2018; Borrelle *et al.* 2020; The Pew Charitable Trusts and SYSTEMIQ 2020). For example, losses which occur during sorting and reprocessing mean only 5 per cent of the value of materials is retained for subsequent use; this represents a loss of value with respect to packaging waste of between US$ 80-120 billion per year (Ellen MacArthur Foundation 2016).

During the past two decades plastic production has been shifting from North America and Europe towards Asia and, more recently, Africa; China, for example, accounts for nearly 30 per cent of global production of polyurethanes and thermoplastics (Geyer 2020). Plastics have substantially outpaced any other manufactured material in terms of production because of their low cost, durability, versatility, and resistance to degradation (Dauvergne 2018). Consumption of plastics is increasing, especially in emerging economies, where a three-fold increase has been forecast for the middle of the century[73] (Lebreton and Andrady 2019; American Chemistry Council 2020; Bond 2020; Borrelle *et al.* 2020; European Chemical Industry Council 2020; Geyer 2020; Lau *et al.* 2020; The Pew Charitable Trusts and SYSTEMIQ 2020). Globally, individuals discard on average more than 50 kg of plastic a year, although this amount is significantly lower in some developing countries such as India (Dauvergne 2018; Statista 2019). While the plastic supply side was negatively affected by COVID-19, with production in 2020 decreasing by approximately 0.3 per cent (Malik *et al.* 2020; Statista 2021a), large volumes of personal protective equipment (PPE) and other plastic items were consumed daily, adding significantly to the volumes of plastic waste on beaches and elsewhere (Adyel 2020). The World Health Organization requested a 40 per cent increase in disposable PPE production in view of monthly global consumption and waste of 129 billion face masks and 65 billion gloves; in the case of PPE use in the United States this would mean that an entire year's worth of medical waste would be generated in just two months (Adyel 2020).

Single-use plastic products account for over one-third of the plastics produced every year, with 98 per cent manufactured from fossil fuels (Charles *et al.* 2021). Regional differences in production volumes (PlasticsEurope 2019; Statista 2021b) reflect both user demand and the price of fossil fuel feedstocks (Geyer 2020). For example, in the United States since 2010 there have been significant investments of more than US$ 200 billion in new plastic and chemical plants, stimulated by the low cost of raw materials, especially natural gas derived from fracking (American Chemistry Council 2020). A recent forecast of global plastic production is 1,100 million metric tons in 2050, not including fibres, a significant increase from current levels (Geyer 2020; Statista 2021b). Yet only 20 polymer producers produce more than half of all single-use waste generated (Charles *et al.* 2021), while 20 institutional asset managers hold over US$ 300 billion worth of shares in the parent companies of these polymer producers and 20 of the world's largest banks are estimated to have lent almost US$ 30 billion for the production of these polymers since 2011.[74]

The largest volumes of plastic waste are generated by the packaging, consumer and institutional products, and textile sectors. In 2017 the packaging sector accounted for 36 per cent of global plastic production and was responsible for 46 per cent of total plastic waste generated (Geyer 2020). The building and construction sector, which in 2017 accounted for 19.7 per cent of all global plastic production (resin, fibres and additives), generated only 4 per cent (14 million metric tons) of global plastic waste. Geyer (2020) calculated that 438 million metric tons were added to the in-use stock of plastics in 2017 while 328 million metric tons left it as waste; in other words, 110 million metric tons of plastics were added to the in-use stock. However, verifying these volumes is still very difficult due to lack of transparency and access to industry information (Zink and Geyer 2018; Zink *et al.* 2018). Geyer *et al.* (2017) estimated that 168 million metric tons of recyclable plastic waste were produced between 1988 and 2016 and that by 2050, if production continues along the same curve, 9,000 million metric tons of plastic waste will have been recycled, 12,000 million metric tons incinerated, and 12,000 million metric tons discarded to landfills or the natural environment, compared to 5,000 million metric tons today (Figure i). This represents an enormous reservoir of plastic waste (Geyer 2020).

Changing attitudes about the problems created by plastic waste are causing politicians and industries to exploit anti-plastic sentiments through environmental consumerism (UNEP and Consumers International 2020) and to consider ways of keeping the value of plastics in the market through feedstock substitution, expansion of consumer reuse options, and new delivery models to help avoid waste (Ellen Macarthur Foundation 2016; UNEP and International Trade Center 2017; ten Brink *et al.* 2018; Borrelle *et al.* 2020; Lau *et al.* 2020; The Pew Charitable Trusts and SYSTEMIC 2020; Ellen MacArthur Foundation 2021). Many global brand companies have already put in place plans to change their approaches to packaging use consistent with national-level recycling schemes, collection and recycling, and to make all packaging reusable, renewable or recyclable.[75]

There are now many initiatives involving the plastic industry, businesses, governments, international organizations and civil society (UNEP 2018d). Examples include the New Plastics

Economy Global Commitment led by the Ellen MacArthur Foundation (2018) in collaboration with UNEP, which unites more than 500 businesses, governments and other organizations through a common vision of circularity in regard to plastics; Marine Litter Solutions, the framework of the Global Plastics Alliance (an alliance of 74 plastic associations around the world), which supports over 355 projects aiming to prevent leakage of plastics into the environment; the Alliance to End Plastics, through which more than 80 member companies and partners aim to end plastic waste in the environment and address the plastic waste data gap through PRISM (Plastic Recovery Insight and Steering Model); the International Solid Waste Association Marine Task Force, which is helping to quantify leakage rates with the Plastic Pocket Calculator; Operation Clean Sweep, a voluntary programme that promotes proper pellet containment along the entire plastics value chain; Plastic Bank, which provides large-scale sustainable premiums in every recycling community around the world using blockchain technology to authenticate rewards; Plasticforchange, which makes it profitable for companies to transition away from virgin plastics and start sourcing recycled materials; NextWave, whose members commit to decreasing the volume of plastic and nylon waste before it enters the oceans and demonstrating the commercial viability and advantages of integrating ocean-bound plastics into their supply chains; the Ocean Recovery Alliance, which brings together new ways of thinking, technologies, creativity and collaborations (including the Plastics Disclosure Project and the Global Alert Platform) to improve the ocean environment; and Circulate Capital Ocean, an investment management fund which provides financing for small and medium-sized enterprises in developing countries including India, Indonesia and Thailand to set up recycling facilities for a wide range of plastics.[76]

Actions to curb the growth in plastic production will be crucial to achieving reductions in plastic pollution flows; under an ambitious scenario peak virgin plastic could be reached by 2027 and levels of virgin plastic production reduced by 11 per cent (±1 per cent) by 2040 relative to 2016 levels (Borrelle *et al.* 2020; Lau *et al.* 2020). Some initiatives are now focused on shifting production away from fossil fuel-based plastics. Sea to the Future, an industry producer responsibility organization-led contribution, exists at the resin production level. It aims to generate funds through investment in transformative technologies and to support environmental remediation, thus addressing the perverse market price signal that has prevented emerging technologies which can recycle used plastics into high-purity polymers from achieving global commercialization (Forrest *et al.* 2019). Think Beyond Plastics develops and commercializes bio-based materials that can replace fossil fuel-based plastics and assists in the development of manufacturing and the design of packing using these materials.[77]

4.3.2 Reuse and recycling

Current levels of plastic recycling, which have been estimated to be less than 10 per cent, fall well below global recycling rates for other commodities and resources such as paper (58 per cent), iron (70 per cent) and steel (98 per cent) (Dauvergne 2018; Geyer 2020). In 2017 industry figures for packaging indicated that 93 per cent of global plastic used was virgin, 7 per cent recycled (of which 98 per cent was downcycled), and only 2 per cent ended up in a closed loop (European Union Network for the Implementation and Enforcement of Environmental Law 2019). Problems with recycling plastics arise when waste streams are mixed or when the reuse of plastics is restricted, for example in the case of food packaging in the EU under Regulation EC No 282/2008. In this case the recycling process must be authorized and managed by an appropriate quality assurance system, guaranteeing the quality of the recycled materials (Schweitzer *et al.* 2018).

However, plastic recycling and reuse is gaining traction with the help of new technologies and legislative requirements. Most current commitments by governments and businesses are targeted towards specific plastic items or focused downstream towards increasing recycling and disposal. For example, the EU has put in place a requirement that manufacturers include a minimum of 30 per cent recycled plastic in PET bottles by 2030, and the Indonesian government has set a target to reduce marine plastic debris by 70 per cent by 2025. Joint industry initiatives to collect marine litter onshore and offshore also contribute to meeting the need to dispose of waste through recycling and reuse.

Examples of private-public partnerships around the world include the FlipFlopi company, which constructed a dhow out of flip flops found on the Kenyan coast; Waste Free Oceans, in partnership with Fapil, which uses plastics collected at sea to manufacture a range of household and cleaning products including brushes, mops, brooms and buckets for both domestic and professional purposes; Bureo, which manufactures skateboards, clothing and sunglasses from fish nets sourced from over 50 fisheries in South America and partners with numerous companies, such as Patagonia, to incorporate material made from recycled fishnets into their products; Norton Point, which makes sustainable sunglasses out of ocean plastics from the canals and coastlines of Haiti; Adidas and Parley, which teamed up to make high-performance sportswear that turns "the threat into a thread" using waste collected on coastlines, remote islands and coastal communities; Swaggr, which makes comfortable high-performance socks from recycled plastic bottles collected along coastlines; American Express, whose credit cards are manufactured primarily from ocean plastic; Tesco, a supermarket chain in the United Kingdom which has put reverse vending machines in its stores to collect plastic bottles and give customers money in exchange; Fair Harbor, which makes men's and women's swimwear from post-consumer recycled plastic bottles; and Bedford Technology, which takes both post-consumer and post-industrial HDPE recyclables and engineers them into structural and durable building materials that are a heavy-duty wood alternative.

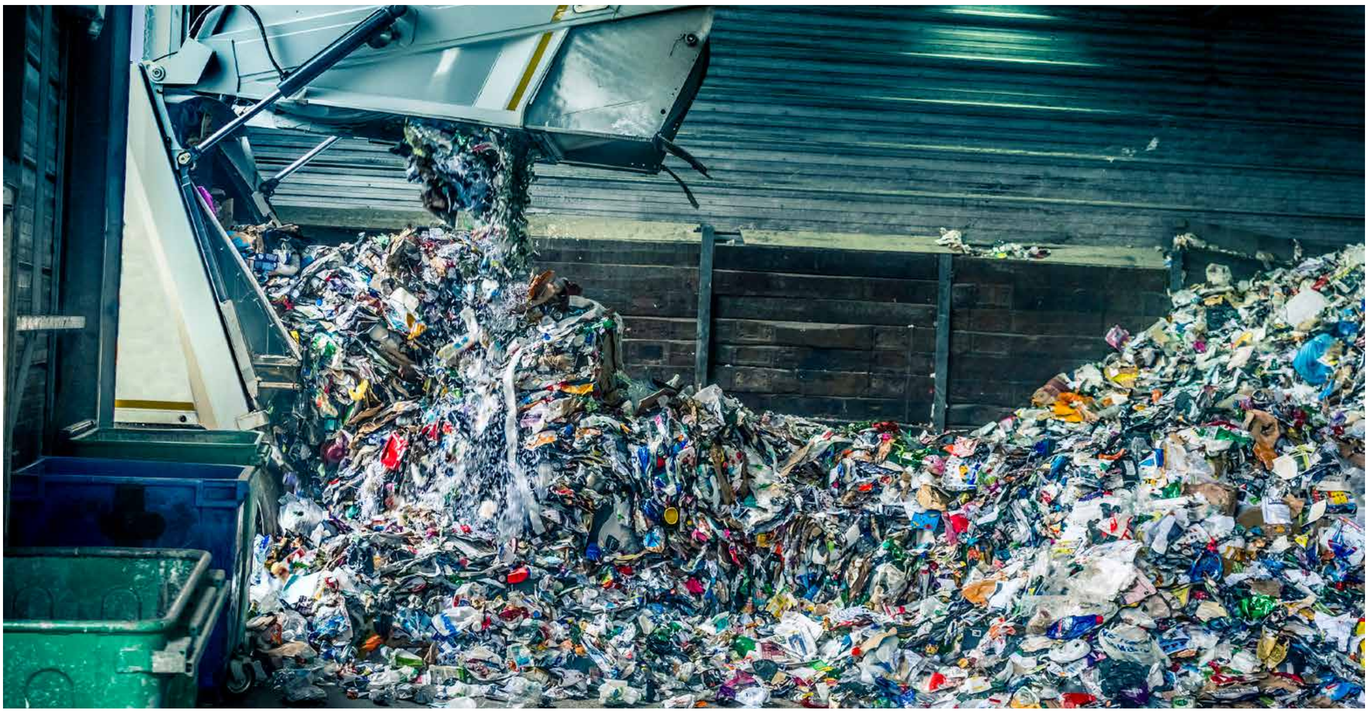
Plastic recycling is currently undertaken using mechanical and chemical processes. Mechanical recycling is used for non-fibre plastic, and increasingly for recycled polyester yarns. This process involves grinding up bottles into flakes, washing them, and then melting them back into new polyester chips. Chemical recycling, which combines various plastic-to-fuel and plastic-to-plastic technologies, turns plastic into liquids or gases, which can be used to make new plastic. Most recycled nylon comes from manufacturing waste (i.e. pre-consumer) and post-consumer waste such as fishing nets and carpets. However, in practice large volumes of these wastes are burned. Even if plastic-to-plastic chemical conversion is rapidly scaled up, it would address only 6 per cent of plastic waste in 2040 and currently has high energy requirements, with GHG emissions 110 per cent higher than mechanical recycling and 9 per cent higher than landfilling (The Pew Charitable Trusts and SYSTEMIQ 2020). Moreover, there are concerns about plastic-to-fuel processes because they perpetuate the burning of fossil fuels. On the other hand, plastic-to-plastic or "repolymerization" is technically challenging to scale up sufficiently to make it financially viable, although industrial examples are emerging.[78]

The production of hundreds of different plastic polymers and products also complicates the recycling potential of plastics (Geyer *et al.* 2016; Zink *et al.* 2018). For example, thermoplastics can be melted when heated, hardened when cooled, and reheated, reshaped and frozen repeatedly; thermosets such as polyurethane, vinyl ester and a range of resins undergo a chemical change when heated, meaning they cannot be re-melted and reformed. The many hundreds of additives can also alter the recycling potential of plastics and may restrict their reuse under the Stockholm Convention due to the likely release of hazardous chemicals into the environment (Hansen *et al.* 2013; Hahladakis *et al.* 2018; Secretariat of the Stockholm Convention 2020). Production data for additives are typically omitted from plastic production statistics, but there is some evidence to suggest that non-fibre plastics contain, on average, around 7 per cent additives by mass. In the case of additives such as phthalates, used as softening and anti-cracking agents, or flame retardants, there is a danger that recycling will release these hazardous chemicals (Hahladakis *et al.* 2018). There are also issues concerning non-intentionally added substances (NIAS), for example in recycled plastics, and where guidance for the risk assessment of these substances in food contact materials and articles is to be developed (Horodytska *et al.* 2020).

Plastics can eventually be destroyed thermally with or without energy recovery. A range of environmental and social concerns are associated with the conditions under which incineration is undertaken, especially when it is poorly managed. These concerns include GHG emissions, particulate matter, emissions of pollutants containing POPs, contamination by heavy metals (Li *et al.* 2017), social issues associated with the location of plants, as well as the need to continue to generate waste to keep incinerators working. The environmental and health impacts of waste incinerators strongly depend on the design, management and use of Best Available Techniques and Best Environmental Practices, along with countries' capacities to carry out effective supervision and monitoring. There are situations in which by-products are managed, but this requires intensive maintenance and management of infrastructure (Quina *et al.* 2018). For example, Stehel *et al.* (2019) analysed the use of separated waste streams, under the EU Waste Framework Directive, to replace fossil fuels in waste-to-energy operations for urban heating systems; the conclusion was that waste incineration needs to be done in a controlled manner, so that emissions and impacts can be controlled.

4.3.3 Development of alternative materials

Green and sustainable chemistry innovation can play an important role in advancing circularity, and provide significant



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improvements to plastics derived from fossil-fuel feed stocks, by designing molecules, materials and products that can be more easily recycled and up-cycled than those currently on the market (UNEP 2021a). This can be achieved, for example by eliminating chemicals of concern in products that currently prevent sound recovery and recycling (UNEP 2019b). For products that are intentionally released to the environment and have open-environmental applications (e.g. pesticides, cosmetics, biocides, or pharmaceuticals) green and sustainable chemistry innovation could help design molecules and materials that rapidly mineralize in the environment while retaining desired functions. The contribution of green chemistry to many end markets where plastics are currently used is significant, and innovations in transportation industry, the construction industry, food and packaging, and waste management need to take this into account. Even amid the Covid-19 crisis, the global green chemical market has been estimated at US$ 93.7 billion in 2020 and is projected to reach a revised size of US$ 167.1 billion by 2027, growing at a compound annual growth rate close to 10 per cent to reach US$ 77.4 billion by 2027.[79,80]

As yet only very small volumes of biosourced and bio-based plastics are being produced[81] (European Bioplastics 2020). In 2018, 2.11 million metric tons were produced, less than 1 per cent of the total volume of plastics produced. Of this amount, 43 per cent was biodegradable and 30 per cent was both biosourced and biodegradable (European Bioplastics 2020).[82] However, the market size for renewable energy from bio-based feedstocks is much larger than that for bio-based plastics (Posen *et al.* 2017). Thus, for advanced bio-based plastic pathways to take off, they must not only prove themselves technically and economically feasible.

A number of factors need to be considered in shifting towards more bio-based feedstocks (Posen *et al.* 2017). For example, there is heavy reliance on agriculture, with bio-based crops tending to score poorly on other environmental metrics such as ozone depletion, acidification, eutrophication, water use and food security (Spierling *et al.* 2018). In terms of energy substitution, there is no change in the final resin produced and bio-based polymers can substitute across the market without any changes to downstream production methods or product functionality. This is also the case for bioethylene-based plastics, but for renewable products such as polylactic acid (PLA) the potential for substitution is more limited.

While biodegradability may be an advantage for polylactic acid and some other bio-based plastics in terms of reducing the volumes of waste going to landfills, few cities and communities have the infrastructure required for composting under the correct conditions so many organizations using compostable biopolymers are likely to continue to send their waste to landfills (UNEP 2021b). This may present a major problem for bio-based as well as biodegradable plastics more generally (Napper and Thompson 2019) (Box 4). There is also significant confusion among consumers about recyclability and biodegradability, especially as descriptions such as "degradable", "oxo-degradable", "oxo-biodegradable" and "landfill degradable" have been used to promote products made with traditional fossil-fuel based plastics, supplemented with specific additives promoting degradability (UNEP and Consumers International 2020).

Overall, replacing one disposable product (e.g. made of plastic) with another disposable product made of a different material (e.g. paper, biodegradable plastic) is only likely to transfer the environmental burden and create other problems. Further, to avoid burden shifting between the environmental and the social dimension, it is important to shift the focus of manufacturers towards the production of more circular and sustainable commodities (UNEP 2021b).

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4.3.4 Building circularity for plastics

Building circularity in support of sustainable consumption and production objectives across the life cycle of plastics means going beyond the 3Rs (Reduce, Reuse and Recycle), to 5Rs with Recover and Redesign (Thompson *et al.* 2009), and further to 7Rs with Refuse and Rethink (Ivar do Sul and Costa 2014; Ivleva *et al.* 2018). Other patterns of Rs have also been designed for circularity, such as Receive, Recycle, Repair, Refill, Rent and Resell.[83] These are now being used to deliver new kinds of services, for example short-term loans of branded products can that be reused by different consumers include luxury fashion to furniture (e.g. from IKEA) and toys (e.g. Lego has a service called Netbricks for rental of its little plastic building blocks).

In some parts of the world movement towards circularity is already under way; for example, the European Union's Strategy for Plastics in the Circular Economy (European Commission 2018b) has set in motion a comprehensive set of initiatives, with business and governments responding to a challenge of serious public concern. These initiatives include increasing the uptake of recycled plastics and contributing to more sustainable use of plastics by implementing mandatory requirements for recycled content and waste reduction measures.

A new framework for using green chemistry to support the development of alternatives aimed at achieving greater circularity has been developed (UNEP 2021b), which fosters a vision of green and sustainable chemistry and emphasizes the potential for the global chemical industry to become fully aligned with the environmental, social and economic dimensions of sustainable development by creating greener and more sustainable chemistry innovations, while also addressing toxic and persistent legacies associated with past chemistries in order to minimize adverse impacts across the entire life cycle of chemicals and products. A key part of the framework is to keep processes as simple as possible, with a minimal number of steps, auxiliaries, energy, and unit operations, to improve the environmental performance of manufacturing materials.

Finally, an important part of building circularity for plastics is improving the traceability of products and their constituent parts. Green chemistry can provide innovative molecules that ensure traceability and can be used to create product digital passports (e.g. composition of products, components, and processes). These, coupled with blockchain technologies, can enable end-to-end traceability of supply chains (Cui *et al.* 2019). When a product failure occurs, or when the product is to be recycled, the molecules and digital passport can provide the information needed to identify the suppliers and or the constituent chemicals (UNEP 2021b). Blockchain technologies are revolutionizing supply chain operations and the tracing of plastics,[84] which will help improve supply chain quality control and protect the environment and human health as well as building consumer confidence.

4.3.5 Business engagement

Finding solutions to the marine litter and plastic litter crisis will require greater engagement of governments, and civil society with business and industry to bring about the necessary changes in policies and business practices (Uyarra and Borja 2016; Hartley *et al.* 2018b; Ashley *et al.* 2019). Multiple types of industries will need to change their business practices, including oil and gas extractors, producers of plastic resins, extruders and product manufacturers, automotive manufacturers, textile manufacturers, consumer product companies, consumer packaged goods companies, retailers, waste hauliers, land fillers, materials recovery operators, waste brokers and recyclers. Each of these industries has different and sometimes competing interests in the market, and each faces unique challenges with regard to addressing this crisis. The size of a business (small, national, regional, international) is also a significant factor in both identifying challenges and designing solutions. Some key challenges to improving business engagement include data sharing and transparency, financing, the regulatory environment, and access to research and development (Ocean Conservancy and McKinsey Center for Business and Environment 2015).

These issues were discussed by the members of the Scientific Advisory Committee on Marine Litter and Microplastics (SAC), and nominated stakeholders representing business and civil society, during the preparation of this assessment. Their conclusions are described in the set of recommendations below:

1) Data sharing and access to solutions

Producers and converters need to disclose more information about their products: for example, the amount of plastics produced annually, by type/resin code; resin type and additives used in plastic products or packaging; percentage of virgin feedstock in products; percentage of recycled plastics in products; specifics related to bio-based plastics, including source (e.g. if ethically sourced); and sustainability of feedstock, degradability and compostability. Preferably there should be buy-in to ecolabelling/certification schemes; international standards for all chemicals; full disclosure by the plastic industry, including any voluntary/mandatory EPS schemes they respond to; and reduction targets.

The packaging industry and retailers need to better inform the public about the use of plastics in their products.[85] For example, the volumes of plastics used, how much goes into different uses and the geographic distribution; whether products and polymers are going into markets that cannot manage their disposal; the amounts of single-use plastic (products and packaging) distributed into each local market (globally), by type and resin code; all additives used in any plastic products or packaging produced and used; resin type and additives used in plastic products or packaging; the percentage of virgin feedstock and recycled plastics in products; the specifics of any bio-based plastics used, including source (ethically sourced) and sustainability of feedstock, degradability and compostability; use of eco-labelling/certification schemes; participation in voluntary/mandatory extended producer responsibility schemes; plastic product and packaging reduction targets; refill and reuse rates, durability and lifespan and the right to repair. Retailers and the packaging industry drive production of plastics, as they set the criteria for the type of polymers they use in terms of marketing and branding. One solution is to develop industry guidelines on the different fates of plastics and alternatives.

Government and the waste brokers, recycling and landfill industries need to inform the public about the quantities of post-consumer plastic materials purchased annually by types and resin code, the amount of materials sold on to recyclers by type and resin code, the breakdown of quantities sold to recyclers offshore, including details related to market destination, and the quantities disposed of due to contamination of waste streams, poor quality and market conditions.[86] Important information on recyclability, reusability, and disposal instructions where a product is being used needs to be displayed on products, as well as the responsibilities of producers and consumers. This requires a consistent approach to labelling, especially about disposal, similar to the international standards for chemicals. Further information on the conditions under which a product is expected to (bio)degrade also need to be provided through a consistent labelling scheme.

2) Financing

Financing (e.g. though the UNEP Finance Initiative, Blue Bonds, plastic footprint and plastic offsetting, impact investment and plastic-specific EPR schemes),[87] could help the process of change by sharing information about their use within different industries, including insurance companies and retailers. Financial incentives from government to establish new economic entities and employment opportunities, based on innovative design of non-toxic materials using green chemistry solutions to replace the use of additives, such as in finishing textiles, were seen as vital (Holmquist *et al.* 2016; Gulzar *et al.* 2019)

There also needs to be more information about the full risks to different sectors of the use of plastics to better align with the different risk profiles of lenders, investors and insurance brokers, for example by communicating about the science of the carbon footprint of plastics.[88] Training courses for small and medium-sized enterprises on waste handling are also important to ensure that knowledge is spread throughout the supply chain upon which tendering and procurement decisions can be based.[89]

Some of the key challenges voiced by business representatives included making clear the financial risks of investing in the plastics/petrochemical industries; potential costs of inaction (i.e. not responding appropriately to public demand for environmentally friendly products and alternatives to plastics); being able to articulate the financial opportunities for investing in refill and reuse systems to replace single-use, alternative non-toxic materials for durable reuse applications; the ways of communicating the hazards that plastics present to their own consumers and workers; and the animal health and environmental hazards and potential damage to reputation and market share. There were also concerns about how to achieve full transparency and disclosure to end-users (consumers and governments) regarding the quantities of different materials being used, which additives are introduced into products. and how they are finally disposed of.

3) Challenges and enabling conditions

A key challenge from a business perspective is change in regulations and policies to prohibit the use of plastic materials that can cause harm to ecosystems and human health, and the need for the change to be mandatory (for example, all businesses would be obligated to meet single-use plastic product reduction targets). There is mounting evidence from various countries that voluntary-only approaches are neither environmentally effective nor economically efficient[90] and may lead to legislation that is incoherent and ineffective at enforcing reductions (Ma *et al.* 2020).

An important enabling factor would be to have industry, local government, central government and civil society working together on the same evidence so as to avoid mistaken assumptions, and to enable the best possible solutions to be developed with a holistic understanding of the challenges, risks, opportunities, drivers, and values of all stakeholders. A clear example of the effectiveness of this approach on a global scale has been the development of the COVID-19 vaccines. Another example, at a national level, is the multi-stakeholder working group which is co-designing the New Zealand National Container Deposit Scheme.[91]

Collaborative efforts are considered an important step in recognizing ongoing efforts and encouraging exchange of best practice, and sharing of solutions, including legislative instruments. For example, mandatory product stewardship has proven to be more effective than voluntary approaches as it prevents "free riders". However, some standards could begin as voluntary efforts and gradually inform a global standard; businesses that adopted voluntary standards early on would become front runners UNEP 2019b).

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4.4 Research and development

4.4.1 Progress on key topics and initiatives and priorities

In the 2016 UNEP report *Marine Plastic Debris and Microplastics – Global Lessons and Research to Inspire Action and Guide Policy Change* (UNEP 2016) a range of key research needs were identified, covering: the properties of plastics; sources and pathways of marine litter; distribution and fate, specifically the factors controlling degradation, including definitions and specifications of biodegradable products; monitoring, specifically the development and use of harmonized monitoring techniques, the development of automated technologies, and modelling to look at patterns of movement and deposition; quantification of the impacts of macroplastics on biota and potential risks of microplastics for food webs and human consumption; social impacts and drivers, including consumer perceptions and behavioural drivers; economic impacts and new forms of governance and decision-making; quantification of releases of debris and litter from fisheries and aquaculture; improved risk assessment; and improved assessments of the value of plastic, of reducing the use, and of recycling, elasticity of demand and different incentives.

As the present assessment shows, there has been a significant amount of research during the past years in many of the priority areas identified in UNEP (2016). These areas include impacts and risks to marine life, ecosystems and human health; the major land-based and sea-based sources of marine litter and plastic pollution; pathways into the marine environment and sinks; the enormous potential that many new technologies are providing for enhanced global monitoring of marine litter and plastic pollution; and the broad range of legislative, business and community initiatives that are now using research findings and new innovations to drive change and reduce the impacts of marine litter and plastic pollution.

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4.4.2 Overview of research activities and gaps

In a review of assessment research spanning 13 years and undertaken in 52 projects across Europe, Maes *et al.* (2019) concluded that marine litter research was "in its adolescence". They found that the most represented topics were policy, governance and management, and monitoring, and that risk assessment, the issue of plastic fragmentation and assessment tools were under-represented. Other topics included modelling, impact and effect, reduction and removal technologies and approaches, socioeconomics, bioaccumulation, education and outreach. They reported a geographic concentration of scientific capacity and thematic hotspots, with Western European countries having contributed most to marine litter research. Overall, the authors stressed the importance of European Union financial instruments (e.g. INTERREG, LIFE, Horizon 2020) in supporting large-scale environmental and nature conservation projects, as they helped to improve cooperation and harmonization over wide regions and to expand capacity building. The new Horizon Europe research programme aims to deliver solutions in five areas of research and innovation, including cancer, healthy oceans, climate-neutral cities, climate change, and healthy soil and food. These all touch upon the issue of plastics and circularity.

Marine plastic research is also growing within the 10 countries in the Association of Southeast Asian Nations (ASEAN). Most of the research is focused on monitoring and surveying of plastic in the marine environment and the impact of plastic on marine ecosystems (Lyons *et al.* 2019). However, the impact of marine plastic on human health and life has not attracted much attention. Countries have organized a series of regional forums and workshops to increase understanding of marine plastic pollution and to share and find solutions; however, most of the current activities remain focused on increasing understanding of where plastics occur and their direct impacts. Several other intergovernmental organizations are also promoting actions, plans and research projects in the Southeast Asia region. Among them, the Regional Seas Programme, COBSEA, is playing a leading role. Overall, countries in the region have recognized the importance of marine plastic pollution and that further research is needed.

From the literature used in the preparation of this assessment a number of specific research topics and gaps have been identified (Table 2). Carney Almroth and Eggert (2019), de Sá *et al.* (2018) and Maes *et al.* (2019) concluded that the current state of knowledge can provide a reasonable basis upon which to identify research priorities in general, and also to identify areas where there has been limited research and development funding despite policy and societal needs. Looking more broadly, they identified several areas which continue to require further research and investment: to enhance the solution space to address the issue of marine litter, including: the development of polymers, including bio-based, that are safer and more easily disposed of or recycled. Research should focus on polymer chemistry and recycling techniques, as well as on policies that restrict the use of compounds known to be toxic; research on and evaluation of environmental and health impacts of marine microplastics and nanoplastics, including the potential implications of new materials and new applications, because new uses introduce new risks, as well as gendered impacts of plastics and associated chemicals in food production, aquaculture, agriculture and food safety; policy research on effective measures to reduce microplastics, establish extended producer responsibility schemes and implement reinforcing fiscal instruments, and encourage ecodesign that stimulates the use of new materials and both recycling and reuse; and behavioural economics and education research on gender, nudges, norms and educational processes beyond knowledge acquisition to influence behavioural changes.

4.4.3 Future research priorities

Addressing the issues of marine litter and plastic pollution requires multidisciplinary, integrated research coupled with wide cooperation among academic researchers and professionals from different specialist areas and industry. This is particularly important in areas where uncertainties exist, such as the potential risks from plastic associated chemicals and microplastics (Burns and Boxall 2018), where there is a need for intercalibration of results using different methodologies and technical standards, and where more integrative approaches, such as nature-based solutions, life cycle approaches and circularity are required (Temmerman *et al.* 20013). Overall, there is a need for research to provide answers and inputs to policy analyses and risk assessments that are fit-for-purpose (Hurley and Nizzetto 2018; Besselling *et al.* 2019; Karn and Jenkinson 2019; Maeland and Staupe-Delgado 2020).

Based on inputs from the SAC members and the findings in this assessment, a number of systemic areas have been identified that would greatly benefit from deeper investigation over the next two to five years. These include cross-cutting issues such as gender and intersectionality (age, marginalized and vulnerable groups), especially in relation to exposure, health effects, attitudes to new innovative technologies and ocean literacy, where there has been virtually no research published in the peer reviewed literature,[92] plus the following:

- Evaluation of the full life cycle for key plastic products,[93] including environmental and health impacts of marine plastics, microplastics and nanoplastics, social and economic costs, loss of ecosystem services, the potential implications of new materials, gendered impacts of plastics and alternatives, and the risks and impacts of chemicals associated with plastics on food production, aquaculture, agriculture and food safety;
- Development of a risk framework, based on a full life cycle for marine litter and plastic pollution from source to sea, covering ecological, social, economic and health effects;
- Definition of health and toxicological criteria and testing needed to establish exposure of humans and wildlife to microplastics in aquatic environments;
- Implementation of open access platforms to enable global mass balance modelling of marine litter and plastic pollution and the fluxes and flows of plastics entering the marine environment from rivers, wastewater treatment plants, waste management, storm sewers through catastrophic events, and maritime sectors;
- Establishment of informatics and harmonized monitoring frameworks, including standard methodologies for sampling, laboratory testing and data collection to quantify the fluxes and flows of plastics into the marine environment, the distribution of plastics and microplastics and the toxicology of microplastics and additives in the environment emanating from plastic waste, to be able to measure the effectiveness and impacts of different interventions and mitigation efforts;
- Definition of core sets of indicators, from source to sea, across the Drivers Pressures State Impacts Response framework to monitor progress on the reduction of marine litter and microplastics;
- Green chemistry innovation to minimize the use of additives and develop alternative polymers and materials, including bio-based, based on a full-life-cycle approach and that are safer and more easily disposed of or recycled and develop pathways to switch to alternatives;
- Development of ecodesign principles across all major use sectors where plastics are used extensively and develop cost road maps;
- Development of waste and recycling technologies that enable mechanical and chemical recycling to be placed close to the sources of plastics production and consumption and technologies and which can help to avoid or reduce micro(nano)plastics leakage into the environment across the life cycle of plastic;
- Development of standards for plastic certification, traceability and labelling schemes for all plastics linked to consumer use, including biodegradability;
- Policy research on effective measures to reduce plastics including microplasticx, such as Extended Producer Responsibility schemes, reinforcing fiscal instruments, standards for plastic certification, traceability and labelling schemes for all plastics linked to consumer use, encouraging ecodesign and green chemistry to develop new materials;
- Assessment of social issues related to marine litter and plastics, including gender, consumer perceptions and social drivers, integrating a human rights-based approach that includes meaningful public participation and access to remedies;
- Development of literacy and educational programmes to raise awareness of the issue of marine litter and plastic pollution

and to help change human behaviours towards those that reduce mismanagement of plastic waste; and

- Behavioural economics and education research on nudges, norms and educational processes beyond knowledge acquisition to influence behavioural changes.

4.5 Conclusion

The findings presented in this assessment underline the step change that has taken place since the publication of the previous UNEP report assessment (UNEP 2016) on the issue of marine litter and plastic pollutions, *Marine Plastic Debris and Microplastics* – Global Lessons and Research to Inspire Action and Guide Policy Change (UNEP 2016). Virtually every major international organization and national government has published at least one report or briefing paper on the subject of plastics and their impacts on the environment, particulary the marine environment, and on society. As the United Nations Secretary-General has said,[94]

"We must remember: ... Everything is interlinked – the global commons and global well-being. That means we must act more broadly, more holistically, across many fronts, to secure the health of our planet on which all life depends. … we need much more ambition and greater commitment to deliver on measurable targets and means of implementation, particularly finance and monitoring mechanisms. In 2022, countries will hold the Ocean Conference to protect and advance the health of the world's marine environments … chemical and solid waste pollution – plastics in particular – must be reduced drastically; marine reserves must increase significantly; and coastal areas need greater protection … The blue economy offers remarkable potential. Already, goods and services from the ocean generate $2.5 trillion each year and contribute over 31 million direct full-time jobs – at least until the pandemic struck. We need urgent action on a global scale to reap these benefits but protect the world's seas and oceans from the many pressures they face. This is a moment of truth for people and planet alike. COVID and climate have brought us to a threshold. The door is open; the solutions are there. Now is the time to transform humankind's relationship with the natural world – and with each other. And we must do so together."

Research area	Examples
Monitoring and measurements	Volumes and characteristics of litter and microplastics in freshwater environments, including rivers, lakes and reservoirs
	Atmospheric concentrations of microplastics
Modelling and informatics	Global mass balance model estimates and scenarios
	Life cycle analysis of plastics production, reuse, recycling and disposal
	Blockchain technologies applications
Methods and indicators	Laboratory and field assays of micro(nano)plastics in all media
	Methodologies for sampling, laboratory testing and data collection to measure fluxes and flows of litter and plastics into the marine environment
	Specific applications of earth observations and remote sensing including satellites, drones, autonomous measurements
	Definition of core set of indicators from-source-to-sea across the DPSIR (Drivers, Pressures, State, Impact and Response) framework to monitor progress on the reduction of marine litter and microplastics
	Indicators and targets on retention and discharges from water treatment plants
	Indicators of impacts of litter and plastics on wildlife, including toxicology of microplastics and additives
	Indicators of social and economic impacts of exposure to marine litter and plastics
	Intercalibration of indicators across land-marine domains and integration of data flows
	Informatics and monitoring framework of fluxes and flows of litter and plastics entering the marine environment
Distribution and abundance	In situ sampling and estimation of deposition rates
Source and types	Integrated framework from-source-to-sea of categories of plastics and litter to enable life cycle analysis and assessment
Pathways	Measurements of time trajectories and degradation rates for different types of litter and plastic along different pathways, especially in sediments and coastal ecosystems
Spatial and temporal trends	Movement, concentration and sequestration estimates in different environments
Hotspots and accumulations	Identification of hotspots, determination of physical and chemical leaching processes and accumulation rates
Ecological, environmental and socioeconomic impacts and effects	Definition of good status/health relating to litter and plastics in freshwater and marine environments
	Ecosystem effects of marine litter and plastics
	Impacts of ingestion of plastics and litter on marine organisms
	Impacts of chemical additives and microplastics on physiology and epidemiology in marine organisms and in humans, through the food chain
Chemical additives and leachates	Chemical toxicity during manufacture; leaching rates from plastics, e.g. POPs in different environments; toxicological effects on marine organisms
	Sorption rates of chemicals by plastics in the field
Bioaccumulation and transfer	Evidence of bioaccumulation of micro(nano)plastics in marine organisms and mechanisms of transfer
Fragmentation rates and mechanisms	Environmental conditions and mechanisms affecting fragmentation
Risk and impact assessments	Integrated risk framework and impact criteria
Human health and food related issues	Exposure and uptake pathways; impacts from micro(nano)plastics including critical thresholds
Measures and solutions	Criteria and analysis of effectiveness of regulatory, fiscal and voluntary measures and instruments

Table 2: Research needs and gaps identified in this assessment

Research area	Examples
Material science, ecodesign, recycling and life cycles	Technologies to avoid or reduce microplastics entering the marine environment
	New chemistries and materials that provide "plastic" characteristics, e.g. flexibility, but with reduced post-consumer hazards and greater recyclability
	Alternatives to the most prevalent single-use plastic items and fishing gear found in litter, and costed road maps for the switch
	Ecodesign principles for plastic substitutes and costed road maps across sectors, with a particular focus on maritime industries such as fisheries, aquaculture, offshore operations, shipping and tourism
	Open access certification and traceability schemes for all plastics
	Improved labelling of polymers and resins for recyclability and degradation
	Technologies and costed road maps for sustainable bio-based plastic
Waste management technology and practices	Improvement to recycling including sorting and collection, e.g. artificial intelligence (AI), robotics and advanced sensors
	Technologies to detect, measure and remove substances of concern from plastics
	Technologies for recycling complex plastic waste, e.g. chemical recycling
Social and behavioural change	Design of market mechanisms to encourage fossil fuel free plastics
	Educational schemes to encourage turning knowledge into action
	Communication processes for moving to zero plastics emissions
	Comprehensive behavioural and community change programmes
Outreach and awareness	Effective communication with the general public to bring about behavioural change

Table 2: Research needs and gaps identified in this assessment *(continued)*

ANNEX I: REGIONAL ACTION PLANS ON MARINE LITTER[5]

Name	Organization/entity	Year	Link
Regional Action Plan on Marine Litter in the Arctic	Protection of the Arctic Marine Environment (PAME)	2021	https://digital.gpmarinelitter.org/action_plan/10017
Regional Action Plan for Marine Litter in the Baltic Sea	Helsinki Convention/Baltic Marine Environment Protection Commission (HELCOM)	2015	https://digital.gpmarinelitter.org/action_plan/197
Black Sea Marine Litter Regional Action Plan	Bucharest Convention/Commission the Protection of the Black Sea Against Pollution	2018	https://digital.gpmarinelitter.org/action_plan/194
Regional Action Plan on Marine Litter	Coordinating Body for the Seas of East Asia (COBSEA)	2019	https://digital.gpmarinelitter.org/action_plan/196
Regional Plan on Marine Litter Management in the Mediterranean	Convention for the Protection of the Mediterranean Sea Against Pollution (Barcelona Convention)/Mediterranean Action Plan	2013	https://digital.gpmarinelitter.org/action_plan/198
Regional Action Plan for Prevention and Management of Marine Litter in the North-East Atlantic	OSPAR Commission / Convention for the Protection of the Marine Environment of the North-East Atlantic	2014	https://digital.gpmarinelitter.org/action_plan/201
NOWPAP Regional Action Plan on Marine Litter	Northwest Pacific Action Plan (NOWPAP)	2008 (update expected 2021)	https://digital.gpmarinelitter.org/action_plan/200
Pacific Regional Action Plan – Marine Litter (2018-2025)	Noumea Convention/Secretariat of the Pacific Regional Environment Programme (SPREP)	2018	https://digital.gpmarinelitter.org/action_plan/205
Regional Action Plan for the Sustainable Management of Marine Litter in the Red Sea and Gulf of Aden	Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden (PERSGA)	2018	https://digital.gpmarinelitter.org/action_plan/203
Regional Marine Litter Action Plan for South Asia Seas Region	South Asia Co-operative Environment Programme (SACEP)	2019	https://digital.gpmarinelitter.org/action_plan/204
Basura Marina en la Region del Pacifico Sudeste	Permanent Commission for the South Pacific (CPPS)	2007	https://digital.gpmarinelitter.org/action_plan/238
Western Indian Ocean Regional Action Plan on Marine Litter	Nairobi Convention	2018	https://digital.gpmarinelitter.org/action_plan/199
Regional Action Plan on Marine Litter Management for the Wider Caribbean Region	Cartagena Convention – UNEP Caribbean Environment Programme (CEP)	2014	https://digital.gpmarinelitter.org/action_plan/195
ASEAN Regional Action Plan for Combating Marine Debris in the ASEAN Member States	Association of Southeast Asia Nations (ASEAN)	2021	https://digital.gpmarinelitter.org/action_plan/10008
G7 Action Plan to Combat Marine Litter	Group of 7	2015	https://digital.gpmarinelitter.org/action_plan/190
G20 Action Plan on Marine Litter	Group of 20	2017	https://digital.gpmarinelitter.org/action_plan/191
Action Plan to Address Marine Plastic Litter from Ships	International Maritime Organization (IMO)	2018	https://digital.gpmarinelitter.org/action_plan/237
APEC Roadmap on Marine Debris	Asia-Pacific Economic Cooperation (APEC)	2019	https://digital.gpmarinelitter.org/project/177

5 **Annex I:** The development of Draft Regional Action Plans on Marine Litter is under way in the Caspian, Northeast Pacific, and Western, Central and Southern Africa regions.

ENDNOTES
REFERENCES

Endnotes

1 https://wedocs.unep.org/handle/20.500.11822/7720
2 The 2016 UNEP report is available at https://wedocs.unep.org/handle/20.500.11822/7720. Using webofknowledge.com to search the scientific literature on marine litter, debris and plastics published in all languages shows that there has been a doubling in the annual publication rate during the past three years. Based on approximately 90,000 peer reviewed journal papers, it has been possible to identify areas where there is high research intensity, as well as where important knowledge gaps remain.
3 International Agency for Research on Cancer (IARC) monographs on the evaluation of cancer risks of chemicals: http://monographs.iarc.fr/; United States Centers for Disease Control and Prevention (CDC) National Biomonitoring Program: https://www.cdc.gov/biomonitoring/index.html
4 https://www.reuters.com/article/us-norway-whale-idUSKBN15I2EI
5 World Bank data on country-specific waste generation and management from What a Waste: A Global Review of Solid Waste Management (Hoornweg and Bhada-Tata 2012).
6 The average composition of municipal solid waste (MSW) in Sub-Saharan Africa is about 57 per cent organic, 9 per cent paper/cardboard, 13 per cent plastic, 4 per cent glass, 4 per cent metal and 13 per cent other materials (UNEP 2018c)
7 The sustainability of the use of plastics in agriculture, and options to improve their circularity, are addressed in depth in a forthcoming global assessment by the FAO.
8 Eurostat PRODuction COMmunautaire provides statistics on production, exports and imports of manufactured goods in the EU: https://ec.europa.eu/eurostat/statistics-explained/index.php/Industrial_production_statistics_introduced_-_PRODCOM
9 https://countermeasure.asia/
10 There is a visualization of the surface current distribution of plastic on the PlasticAdrift open platform: van Sebille (2019): http://www.plasticadrift.org/
11 https://www.esr.org/research/oscar/oscar-surface-currents/; http://marine.copernicus.eu/services- portfolio/access-to-products/; https://www.umr-lops.fr/en/Projects/Active-projects/SKIM; https://swot.jpl.nasa.gov/; https://www.frontiersin.org/articles/10.3389/fmars.2019.00457/full; https://mdc.coaps.fsu.edu/scatterometry/meeting/docs/2016/Tue_PM/B_winds_update.pdf
12 GOOS is the major international programme executed by the Intergovernmental Oceanographic Commission (IOC) of the UN Educational, Scientific and Cultural Organization (UNESCO), which combines the coordinated contributions of organizations and researchers worldwide. GOOS observations are coordinated by the Joint IOC-World Meteorological Organization (WMO) Technical Commission for Oceanography and Marine Meteorology. This commission was established by the IOC and the WMO to coordinate worldwide marine meteorological and oceanographic services and their supporting observational and data management services and capacity building programmes. See: https://www.goosocean.org
13 https://www.akvaplan.niva.no/mynewsdesk-articles/multicopter-drones-map-marine-litter-in-the-arctic/
14 https://www.pml.ac.uk/Research/Projects/OPTIMAL; http://scor-flotsam.it/index.html
15 https://imos.org.au/facilities/shipsofopportunity
16 https://www.ferrybox.com
17 Examples include international and global programmes such as the Argo Programme: http://www.argo.net; the NOAA/AOML XBT Network: https://www.aoml.noaa.gov/phod/goos/xbt_network/; the Global Drifter Program: NOAA hurricane gliders https://www.aoml.noaa.gov/phod/goos/gliders/index.php; PIRATA; South Atlantic Meridional Circulation (SAMOC)
18 http://www.oceansites.org
19 https://www.youtube.com/watch?v=DVKV5DagZic
20 www.plasticadrift.org
21 https://oceanconservancy.org/about/partnership/international-coastal-cleanup/ ; https://www.projectaware.org/; https://www.5gyres.org/
22 SDG target 14.1 contains a commitment by countries to "prevent and significantly reduce marine pollution of all kinds in particular from land-based activities, including marine debris (…) by 2025" (UN General Assembly 2015; UNEA 2018). The proposed metadata is given in *Understanding the State of the Ocean: A Global Manual on Measuring SDG 14.1.1, SDG 14.2.1, SDG14.5.1* (UNEP 2021c).
23 http://jpi-oceans.eu/baseman/workpackages
24 The Global Partnership on Marine Litter (GPML) was launched in June 2012 at Rio+20 in Brazil, under the Global Programme of Action for the Protection of the Marine Environment from Land-based Activities (GPA) (UNEP 2013), to protect human health and the global environment by the reduction and management of marine litter through a wide range of activities. The partnership comprises international agencies, governments, NGOs, academia, the private sector and civil society, which contribute in the form of financial support, in-kind contributions and/or technical expertise. The GPML is supporting the development of a digital multi-stakeholder platform that aims to integrate data and information sources; connect stakeholders; enhance cooperation and coordination; help to promote awareness of sources and pathways of marine litter and their fate and impacts; and support knowledge management, information sharing, policy development and monitoring of progress on implementation of SDG related indicators, action plans and voluntary commitments.
25 https://ospar.org/; http://web.unep.org/unepmap; https://www.5gyres.org/trawlshare-application
26 https://helcom.fi/baltic-sea-trends/data-maps/databases/
27 https://mcc.jrc.ec.europa.eu/main/dev.py?N=41andO=434andtitre_chap=TG%2520Marine%2520Litter; http://www.helcom.fi/; http://www.blacksea-commission.org/; http://www.emodnet-chemistry.eu/welcome; https://www.ices.dk/data/data-portals/Pages/DATRAS.aspx; https://www.ices.dk/data/data-portals/Pages/DOME.aspx
28 www.opendatacube.org/copy-of-aodn; www.data4sdgs.org/ARDC; http://maps.helcom.fi/website/mapservice/
29 The study looked at six networks in detail: the Australian Marine Debris Initiative https://www.tangaroablue.org; the German Round Table Marine Litter www.muell-im-meer.de; the Indonesian Waste Platform http://www.indonesianwaste.org/en/home; the Portuguese Marine Litter Association https://www.aplixomarinho.org; the African Marine Waste Network https://africanwastenetwork.org.za; and the Global Partnership on Marine Litter: https://www.gpmarinelitter.org/
30 https://sustainabledevelopment.un.org/partnership/?p=7471
31 https://marinedebris.noaa.gov/partnerships/marine-debris-tracker; https://oceanconservancy.org/trash-free-seas/

international-coastal-cleanup/; https://www.eea.europa.eu/themes/water/europes-seas-and-coasts/assessments/marine-litterwatch/briefing; https://www.mcsuk.org/beachwatch/; https://mcc.jrc.ec.europa.eu/main/dev.py?N=simpleandO=380andtitre_chap=%25C2%25A0andtitre_page=RIMMEL.

32 Indicator 14.1.1 is Index of coastal eutrophication and floating plastic debris density.

33 http://pelletwatch.org/

34 https://www.nurdlehunt.org.uk/

35 http://www.beatthemicrobead.org/; http://coastwatch.org/europe/microlitter/; https://www.sas.org.uk/our-work/beach-cleans/; https://www.ospar.org/work-areas/eiha/marine-litter; https://mcc.jrc.ec.europa.eu/main/dev.py?N=simple&O=394&titre_page=RIMMEL%2520observation%2520Network; https://www.zooniverse.org/projects/theplastictide/the-plastic-tide/classify

36 https://www.eea.europa.eu/themes/water/europes-seas-and-coasts/assessments/marine-litterwatch

37 https://www.wilsoncenter.org/article/citizen-science-and-data-integration-for-understanding-marine-litter

38 https://publications.parliament.uk/pa/cm201719/cmselect/cmenvfru/2080/208009.htm

39 https://www.oneplanetnetwork.org/sites/default/files/unep_ci_2020_can_i_recycle_this.pdf

40 Greenpeace Philippines: https://www.breakfreefromplastic.org/; https://drive.google.com/file/d/1dyAJfVEF0iNI5N0vlvkiF0Nj7FpQpuZf/view

41 https://sableislandinstitute.org/marine-litter-brand-audit-sable-island-september-2018/

42 https://www.bloomberg.com/news/articles/2021-03-18/even-garbage-is-using-blockchain-now

43 https://www.goodearthcotton.com/

44 https://maritim.go.id/portfolio/indonesias-plan-action-marine-plastic-debris-2017-2025/; http://asemconnectvietnam.gov.vn/default.aspx?ZID1=3&ID1=2&ID8=93138

45 Today nearly 98 per cent of plastics are made almost entirely from fossil fuels. Alternative bio-based plastics are made from organic waste material and crops such as maize (corn) and vegetable oil crops, potentially diverting land that could be used for food production and habitat protection. Biomass-based polymers also tend to be more expensive than those based on fossil fuels, reflecting widespread subsidies to the oil and gas industry (UNEP 2014).

46 https://www.plasticpollutiontreaty.org/

47 www.gpmarinelitter.org

48 MARPOL Annex V: see https://www.imo.org/en/OurWork/Environment/Pages/Garbage-Default.aspx

49 https://wwwcdn.imo.org/localresources/en/MediaCentre/HotTopics/Documents/IMO%20marine%20litter%20action%20plan%20MEPC%2073-19-Add-1.pdf

50 http://www.basel.int/tabid/6069/Default.aspx

51 http://www.basel.int/?tabid=8347

52 http://www.basel.int/?tabid=8347

53 https://www.cbd.int/doc/decisions/cop-13/cop-13-dec-10-en.pdf

54 https://www.ramsar.org/about-the-convention-on-wetlands-0

55 https://www.weforum.org/agenda/2020/01/wto-address-plastic-pollution/

56 http://mddb.apec.org/Documents/2015/OFWG/OFWG2/15_ofwg2_025.pdf; http://www.oecd.org/environment/waste/recircle.htm

57 Some of the Regional Seas Conventions and Action Plans have specific plans on marine litter. More details can be found at: Abidjan Convention http://abidjanconvention.org/; Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR), Antarctic Treaty https://www.ccamlr.org/en/data/data; Arctic Council, Ottawa Declaration https://arctic-council.org/en/resources/; Helsinki Commission (HELCOM), Helsinki Convention https://helcom.fi/action-areas/; Black Sea Commission, Bucharest Convention http://www.blacksea-commission.org/Inf.%20and%20Resources/Data%20Links/; Caspian Environment Programme, Tehran Convention https://www.unep.org/explore-topics/oceans-seas/what-we-do/working-regional-seas/regional-seas-programmes/caspian-sea; Oslo-Paris Convention (OSPAR) for the Protection of the Marine Environment of the North-East Atlantic https://www.ospar.org/; Antigua Convention https://www.unep.org/explore-topics/oceans-seas/what-we-do/working-regional-seas/regional-seas-programmes/north-east-0?_ga=2.256901323.1346881430.1623578768-437061112.1613130463; Coordinating Body on the Seas of East Asia (COBSEA) https://www.unep.org/cobsea/what-we-do, UN Environment Mediterranean Action Plan (UNEP-MAP), Barcelona Convention https://www.unep.org/unepmap/; Northwest Pacific Action Plan (NOWPAP) https://www.unep.org/nowpap/; Pacific Regional Environment Programme, Secretariat of the Pacific Regional Environment Programme (SPREP), Noumea Convention https://www.sprep.org/; Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden (PERSGA), Jeddah Convention http://www.persga.org/index.php; Regional organization for the Protection of the Marine Environment (ROMPE), Kuwait Convention http://ropme.org/home.clx; Hamilton Declaration http://www.sargassoseacommission.org/meet-the-commission/hamilton-declaration; South Asia Cooperative Environment Programme, South Asian Seas Action Plan http://www.sacep.org/; Permanent Commission for the South Pacific (CPPS), Lima Convention http://cpps-int.org/

58 An example of a marine litter plan is that of the Protection of the Arctic Marine Environment Working Group of the Arctic Council: https://www.pame.is/document-library/pame-reports-new/pame-ministerial-deliverables/2021-12th-arctic-council-ministerial-meeting-reykjavik-iceland/801-regional-action-plan-on-marine-litter-in-the-arctic/file

59 Established in 1991, the Bamako Convention on the Ban of the Import into Africa and the Control ofTransboundary Movement and Management of Hazardous Wastes within Africa came into force in 1998, in relation to Article 11 of the Basel Convention. Twenty-nine of 54 African countries have ratified the Convention.

60 https://www.europarl.europa.eu/news/en/press-room/20190321IPR32111/parliament-seals-ban-on-throwaway-plastics-by-2021

61 https://www.unenvironment.org/events/unep-event/xxii-forum-ministers-environment-latin-america-and-caribbean; https://wedocs.unep.org/bitstream/handle/20.500.11822/34801/APWMEN.pdf?sequence=4&isAllowed=y

62 Products included plates and cutlery (forks, knives, spoons and chopsticks), plastic straws, cotton buds made of plastic, plastic balloon sticks, oxo-degradable plastics and food containers, and expanded polystyrene cups.

63 "Maine First U.S. State to Ban Styrofoam Containers" (2019): https://www.ecowatch.com/maine-bans-styrofoam-2636014775.html

64 Based on inputs from UNEP (2018d), Deutsche Welle and the

Earth Policy Institute: https://www.greenpeace.org/africa/en/blogs/11156/34-plastic-bans-in-africa/
65 https://www.bbc.com/news/world-africa-20891539
66 www.reassemblingrubbish.xyz
67 https://zerowasteworld.org/
68 http://www.kimointernational.org/fishing-for-litter; https://zerowasteeurope.eu; http://www.wrap.org.uk.
69 Ireland has established EPR schemes for electronic waste, end-of-life vehicles, batteries, packaging, agricultural plastics and tyres and has plans to implement such a scheme for textile waste. France has EPR schemes for textile waste, establishing obligations for producers to take back 50 per cent of the volumes put on the market. Sweden considered a mandatory EPR system for textile products, but the final decision has been left to producer organizations. Several EU Member States have adopted EPR schemes for agricultural plastic waste, including plastic mulch films, row coverings, high and low tunnels and greenhouses. EPR is also being implemented for specific types of wastes containing plastics such as disposable plastic kitchenware (Belgium); pesticides, fertilizers, seed and plant packaging, furniture, office equipment and ink cartridges (France); medical and pharmaceutical packaging (Portugal); plastic foils and bulky plastics (Austria) (European Commission 2018b); packaging, electronics, PVC and microbeads (Australia); packaging and electronics (Brazil); packaging, electronics and vehicles (China); multilayer plastics and films (India); packaging and electronics (Japan); and plastic bags, crockery, bottles, homeware, stationary, carpets, textile electronics (Russian Federation) (UNEP 2018d).
70 https://www.gpmarinelitter.org/what-we-do/action-plans
71 Ecomodulation means providing clearer incentives for the adequate design of products rather than flat fees per ton, which result in light-weighting only, often leading to non-recyclable products such as most flexible plastic packaging. An example can be seen in France: https://www.sciencedirect.com/science/article/abs/pii/S0959652620357607
72 Developed within the MARLISCO project by the Mediterranean Information Office for Environment, Culture and Sustainable Development (MIO-ECSDE), the resource pack "Know, Feel, Act! to Stop Marine Litter: Lesson plans and activities for middle level students" is designed for educators working with young people aged 10-15 in formal or non-formal educational settings. It contains 17 learning activities that examine characteristics, sources, effects, and possible ways to tackle marine litter, addressing it from an environmental, societal, cultural and economic point of view. The online course trained educators to use the resource pack in their teaching and aimed at increasing their skills and confidence in doing so, with a strong focus on pedagogy rather than on marine litter facts. The pack is available in 15 languages (http://www.marlisco.eu/education.en.html)
73 https://phys.org/news/2020-09-oil-industry-risky-plastics.html
74 Kimman, C.D. and Saran, N. The Plastic Waste Makers Index, Minderoo Foundation: https://www.npr.org/2021/05/18/997937090/half-of-the-worlds-single-use-plastic-waste-is-from-just-20-companies-says-a-stu
75 West-Rosenthal, L.B. (2019). "22 big companies that are getting rid of plastic for good": https://www.rd.com/culture/companies-getting-rid-plastic/
76 http://www.marinelittersolutions.com; http://www.endplasticwaste.org; https://plasticpollution.leeds.ac.uk/; http://www.opcleansweep.eu; http://www.plasticbank.org; http://www.plasticsforchange.org; http://www.nextwaveplastics.org; http://www.oceanrecov.org; http://www.circulatecapital.com
77 https://www.thinkbeyondplastic.com/innovationcenter
78 For example, the Sekisui Chemical and Sumitomo Chemical companies plan to deploy a new technology in 2022 to manufacture polyolefins using waste as a raw material to support their circular economy initiatives (Bailey 2020). The technology enables gasification of combustible waste accumulated at waste disposal facilities into carbon monoxide and hydrogen, without the need for waste separation, and converts these gases into ethanol using a microbial catalyst, which obviates the need for heat or pressure.
79 https://member.reportlinker.com/#/search?query=green%20chemicals&type=report
80 https://www.businesswire.com/news/home/20200319005600/en/Green-Chemicals-Market-Demand-from-Emerging-Economies-to-Boost-Market-Growth-Technavio
81 Biosourced materials include agro-polymers such as polysaccharides (starches, ligno-cellulose, pectins, gums and chitosans) and animal and plant proteins and lipids (casein, whey, collagen, gelatin; soya, gluten); microorganisms; and biotechnology synthesis of polylactides. Biomass-based plastics come from sugar cane, starch, vegetable oil, etc. and minerals such as salt.
82 The bio-based plastics market is still driven by demand for bio-based polyethylene terephthalate (PET) (non- biodegradable) and biodegradable starch-based blends, followed by biosourced polyamide (PA) (non-biodegradable), polylactic acid (PLA) (compostable) and biosourced PE (non-biodegradable), with packaging accounting for 65 per cent of demand ahead of textiles, consumer goods, automobiles and transportation, or construction.
83 Fawkes, P. (2019). "Coming full circle: Sustainable retail in a post-recycling age": https://www.psfk.com/2019/12/sustainable-retail-circular-economy.html
84 www.circularise.com; www.eiravato.com
85 Example on plastic wrapping – Tesco: https://www.theguardian.com/business/2020/jan/24/tesco-to-stop-sale-of-plastic-wrapped-multipacks-in-stores
86 WRAP (2018) estimates that only one item in every 10 in the UK can be recycled.
87 The UNEP FI Principles for Responsible Banking provide the framework for a sustainable banking system and to help industry show how to make a positive contribution to society: https://www.unepfi.org/banking/bankingprinciples/; https://www.worldbank.org/en/news/press-release/2018/10/29/seychelles-launches-worlds-first-sovereign-blue-bond
88 Unwrapping the risks of plastic pollution to the insurance industry: https://wedocs.unep.org/handle/20.500.11822/30915 This study identifies how risks related to plastic pollution play out across insurance lines and asset classes in which insurers invest. It argues that insurers should take an active role in addressing the risks related to plastic pollution and in contributing to global efforts to reduce it. The plastics landscape series: A series aimed at equipping investors with the information they need to understand plastic as a systemic issue, providing a technical overview of plastic and the plastic market, and exploring common concepts. It also helps investors to identify where and how their portfolios might be exposed to plastic, enabling them to analyse relevant sectors and engage at the corporate and policy levels accordingly. https://www.unpri.org/esg-issues/environmental-issues/plastics
89 An example is the ABSA approach to encouraging learning in partnership with Strathmore University Business School in Kenya:

https://www.absa.africa/absaafrica/our-stories/our-voices/2020/three-ways-to-develop-your-supply-chain/

90 "The [New Zealand] government's apparent preference, reinforced by vocal sectors of society, for using voluntary measures to manage contentious resource management and environmental issues. Yet we use a more diverse policy mix, including economic instruments and regulation, to modify behaviour ranging from drinking and smoking to driving and dog control. The weight of evidence suggests that, where a significant shift in public behaviour is needed, voluntary measures are not enough" (New Zealand Parliamentary Commissioner for the Environment, 2006 p. 6). Additional mounting evidence that voluntary-only approaches are insufficient can be found (Auckland Council and WasteMINZ, 2017; CCME, 2014; EPR Canada, 2014; Zero Waste Europe and FPRCR, 2015): https://www.nzpsc.nz/wp-content/uploads/2018/09/NZPSC-open-submission-to-PCE-v-final-sept-2018.pdf

91 https://www.marlborough.govt.nz/services/recycling-and-resource-recovery/rubbish-and-recycling-projects/container-return-scheme/design-progress-to-date

92 http://www.oecd.org/dac/gender-development/1849277.pdf

93 The Life Cycle Initiative (hosted by UNEP) has developed a series of studies on this topic: https://www.lifecycleinitiative.org/activities/key-programme-areas/technical-policy-advice/single-use-plastic-products-studies/

94 https://www.un.org/sg/en/content/sg/speeches/2020-12-02/address-columbia-university-the-state-of-the-planet

References

Aanesen, M., Armstrong, C., Czajkowski, M., Falk-Petersen, J., Hanley, N. and Navrud, S. (2015). Willingness to pay for unfamiliar public goods: Preserving cold-water coral in Norway. *Ecological Economics* 112, 53-67. https://doi.org/10.1016/j.ecolecon.2015.02.007. Accessed 11 January 2021.

Abbasi, S., Soltani, N., Keshavarzi, B., Moore, F., Turner, A. and Hassanaghae, M. (2018). Microplastics in different tissues of fish and prawn from the Musa Estuary, Persian Gulf. *Chemosphere* 205 80-87. https://doi.org/10.1016/j.chemosphere.2018.04.076. Accessed 11 January 2021.

Accinnelli, C. Abbas, H.W., Shier, W.T., Vicari, A., Little, N.S. *et al.* (2019). Degradation of microplastic seed film-coating fragments in soil. *Chemosphere* 226 645-650. https://doi.org/10.1016/j.chemosphere.2019.03.161. Accessed 11 January 2021.

Acuña-Ruz, T., Uribe, D., Taylor, R., Amézquita, L., Guzmán, M.C., Merrill, J. *et al.* (2018). Anthropogenic marine debris over beaches: Spectral characterization for remote sensing applications. *Remote Sensing of Environment* 217, 309-322. https://doi.org/10.1016/j.rse.2018.08.008. Accessed 11 January 2021.

Adam, V., Yang, T. and Nowack, B. (2019). Toward an ecotoxicological risk assessment of microplastics: Comparison of available hazard and exposure data in freshwaters. *Environmental Toxicology and Chemistry* 38(2), 436-447. https://doi.org/10.1002/etc.4323. Accessed 11 January 2021.

Adimey, N., Hudak, C., Powell, J.R., Bassos-Hull, K., Foley, A., Farmer, N.A. *et al.* (2014). Fishery gear interactions from stranded bottlenose dolphins, Florida manatees and sea turtles in Florida, U.S.A. *Marine Pollution Bulletin* 81(1), 103-115. https://doi.org/10.1016/j.marpolbul.2014.02.008. Accessed 11 January 2021.

Adyel, T.M. (2020). Accumulation of plastic waste during COVID-19. *Science* 369(6509), 1314-1315. http://doi.org/10.1126/science.abd9925. Accessed 11 January 2021.

Agamuthu, P., Mehran, S.B., Norkhairah, A. and Norkhairiyah, A. (2019). Marine debris: A review of impacts and global initiatives. *Waste Management and Research* 37, 987-1002. https://doi.org/10.1177/0734242X19845041. Accessed 11 January 2021.

Akarsu, C., Kumbura, H., Gökdağb, K., Kıdeyş, A.E. and Sanchez-Vidal, A. (2020). Microplastics composition and load from three wastewater treatment plants discharging into Mersin Bay, north eastern Mediterranean Sea. *Marine Pollution Bulletin* 150, 110776. https://doi.org/10.1016/j.marpolbul.2019.110776. Accessed 11 January 2021.

Akhbarizadeh, R., Moore, F. and Keshavarzi, B. (2019). Investigating microplastics bioaccumulation and biomagnification in seafood from the Persian Gulf: A threat to human health? *Food Additives and Contaminants: Part A* 36(11), 1696-1708. http://doi.org/10.1080/19440049.2019.1649473. Accessed January 2021.

Alexy, P., Anklam, E., Emans, T., Furfari, A., Galgani, F., Hanke, G. *et al.* (2020). Managing the analytical challenges related to micro- and nanoplastics in the environment and food: Filling the knowledge gaps. *Food Additives and Contaminants: Part A* 37(1), 1-10. http://doi.org/10.1080/19440049.2019.1673905. Accessed 11 January 2021.

Aliani, S., and Molcard, A. (2003). Hitch-hiking on floating marine debris: Macrobenthic species in the western Mediterranean Sea. *Hydrobiologia* 503, 59-67. https://doi.org/10.1023/B:HYDR.0000008480.95045.26. Accessed 11 January 2021.

Alimba, C.G. and Faggio. C. (2019). Microplastics in the marine environment: Current trends in environmental pollution and mechanisms of toxicological profile. *Environmental Toxicology and Pharmacology* 68, 61-74. https://doi.org/10.1016/j.etap.2019.03.001. Accessed 11 January 2021.

Alimi, O.S., Budarz, J.F., Hernandez, M.L. and Tufenkji, N. (2018). Microplastics and nanoplastics in aquatic environments: Aggregation, deposition, and enhanced contaminant transport. *Environmental Science and Technology* 52, 1704-1724. https://pubs.acs.org/doi/abs/10.1021/acs.est.7b05559. Accessed 11 January 2021.

Allen, S., Allen, D., Phoenix, V.R., Le Roux, G., Jiménez, P.D., Simonneau, A. *et al.* (2019). Atmospheric transport and deposition of microplastics in a remote mountain catchment. *Nature Geoscience* 12(5), 339-344. https://doi.org/10.1038/s41561-019-0335-5. Accessed 11 January 2021.

Almahasheer, H., Serrano, O., Duarte, C.M., Arias-Ortiz, A., Masque, P. and Irigoien, X. (2017). Low Carbon sink capacity of Red Sea mangroves. *Scientific Reports* 7, 9700. https://doi.org/10.1038/s41598-017-10424-9. Accessed 11 January 2021.

Alomar, C. and Deudero. S. (2017). Evidence of microplastic ingestion in the shark *Galeus melastomus* Rafinesque, 1810 in the continental shelf off the western Mediterranean Sea. *Environmental Pollution* 223, 223-229. https://doi.org/10.1016/j.envpol.2017.01.015. Accessed 11 January 2021.

Alomar, C., Estarellas, F. and Deudero, S. (2016). Microplastics in the Mediterranean Sea: Deposition in coastal shallow sediments, spatial variation and preferential grain size. *Marine Environmental Research* 115, 1-10. https://www.researchgate.net/publication/291185927_Microplastics_in_the_Mediterranean_Sea_Deposition_in_coastal_shallow_sediments_spatial_variation_and_preferential_grain_size. Accessed 11 January 2021.

Alomar, C., Sureda, A., Capó, X., Guijarro, B., Tejada, S. and Deudero, S. (2017). Microplastic ingestion by *Mullus surmuletus* Linnaeus, 1758 fish and its potential for causing oxidative stress. *Environmental Research* 159, 135-142. https://doi.org/10.1016/j.envres.2017.07.043. Accessed 11 January 2021.

Alvarez-Zeferino, J.C., Beltrán-Villavicencio, M. and Vázquez-Morillas, A. (2015). Degradation of plastics in seawater in laboratory. *Open Journal of Polymer Chemistry* 5 (4), 55-62. http://dx.doi.org/10.4236/ojpchem.2015.54007. Accessed 11 January 2021.

Alzona, J., Cohen, B.L., Rudolph, H., Jow, H.N. and Frohliger, J.O. (1979). Indoor-outdoor relationships for airborne particulate matter of outdoor origin. Atmospheric *Environment* 13(1), 5-60. https://doi.org/10.1016/0004-6981(79)90244-0. Accessed 11 January 2021.

Amaral-Zettler, L.A., Zettler, E.R., Slikas, B., Boyd, G.D., Melvin, D.W., Morrall, C.E. *et al.* (2015). The biogeography of the plastisphere: Implications for policy. *Frontiers in Ecology and the Environment* 13(10), 541-546. https://doi.org/10.1890/150017. Accessed 11 January 2021.

Amaral-Zettler, L.A., Zettler, E.R., and Mincer, T.J. (2020). Ecology of the plastisphere. *Nature Reviews in Microbiology* 18, 139-151. https://doi.org/10.1038/s41579-019-0308-0. Accessed 11 January 2021.

Amélineau, F., Bonnet, D., Heitz, O., Mortreux, V., Harding, A.M.A., Karnovsky, N. *et al.* (2016). Microplastic pollution in the Greenland Sea: Background levels and selective contamination of planktivorous diving seabirds. *Environmental Pollution* 219, 1131-1139. https://doi.org/10.1016/j.envpol.2016.09.017. Accessed 11 January 2021.

American Chemistry Council (2020). *2020 Guide to the Business of Chemistry.* American Chemistry Council. Washington D.C. https://www.americanchemistry.com/2020-Guide-to-the-Business-of-Chemistry.pdf. Accessed 11 January 2021.

Anastasopoulou, A., Mytilineou, C., Smith, C.J. and Papadopoulou, K.N. (2013). Plastic debris ingested by deep-water fish of the Ionian Sea (Eastern Mediterranean). *Deep Sea Research Part I: Oceanographic Research Papers* 74, 11-13. https://doi.org/10.1016/j.dsr.2012.12.008. Accessed 11 January 2021.

Anbumani, S. and Kakkar, P. (2018). Ecotoxicological effects of microplastics on biota: A review. *Environmental Science and Pollution Research* 25, 14373-14396. https://doi.org/10.1007/s11356-018-1999-x. Accessed 11 January 2021.

Andrades, R., Martins, A.S., Fardim, L.M., Ferreira, J.S. and Santos, R.G. (2016). Origin of marine debris is related to disposable packs of ultra-processed food. *Marine Pollution Bulletin* 109(1), 192-195. https://doi.org/10.1016/j.marpolbul.2016.05.083. Accessed 11 January 2021.

Andrady, A.L. (2011). Microplastics in the marine environment. *Marine Pollution Bulletin* 62, 1596-1605. https://doi.org/10.1016/j.marpolbul.2011.05.030. Accessed 11 January 2021.

Andrady, A. (2017). The plastic in microplastics: A review. *Marine Pollution Bulletin* 119(1), 12-22. https://doi.org/10.1016/j.marpolbul.2017.01.082. Accessed 11 January 2021.

Andrady, A. and Rajapakse, N. (2019). Additives and chemicals in plastics. In *Hazardous Chemicals Associated with Plastics in the Marine Environment. Handbook in Environmental Chemistry* 78. Takada, H. and Karapangioti, H.K. (eds.). Springer International Publishing. 1-17. https://www.springer.com/gp/book/9783319955667. Accessed 11 January 2021.

Angelini, Z., Kinner, N., Thibault, J., Ramsey, P. and Fuld, K. (2019). Marine debris visual identification assessment. *Marine Pollution Bulletin* 142, 69-75. https://doi.org/10.1016/j.marpolbul.2019.02.044. Accessed 11 January 2021.

Arcangeli, A., Campana, I., Angeletti, D., Atzori, F., Azzolin, M., Carosso, L. *et al.* (2017). Amount, composition, and spatial distribution of floating macro litter along fixed trans-border transects in the Mediterranean basin. *Marine Pollution Bulletin* 129(2), 545-554. https://doi.org/10.1016/j.marpolbul.2017.10.028. Accessed 11 January 2021.

Ardhuin, F., Aksenov,Y., Benetazzo, A., Bertino, L., Brandt, P., Caubet, E., Chapron, B., Collard, F., Cravatte, S. *et al.* (2018). Measuring currents, ice drift, and waves from space: The Sea Surface KInematics Multiscale monitoring (SKIM) concept. *Ocean Science* 14, 337-354. https://doi.org/10.5194/os-14-337-2018. Accessed 30 November 2020.

Arias, A.H., Ronda, A.C., Oliva, A.L. and Marcovecchio, J.E. (2019). Evidence of microplastic ingestion by fish from the Bahía Blanca estuary in Argentina, South America. *Bulletin of Environmental Contamination and Toxicology* 102(6), 750-756. https://doi.org/10.1007/s00128-019-02604-2. Accessed 11 January 2021.

Arias-Andres, M., Klümper, U., Rojas-Jimenez, K. and Grossart, H.P. (2018). Microplastics pollution increases gene exchange in aquatic ecosystems. *Environmental Pollution* 237, 253-261. https://doi.org/10.1016/j.envpol.2018.02.058. Accessed 11 January 2021.

Arroyo Schnell, A., Klein, N., Gómez Girón, E. and Sousa, J. (2017). *National Marine Plastic Litter Policies in EU Member States: An Overview.* Brussels: International Union for Conservation of Nature and Nature Resources (IUCN). https://portals.iucn.org/library/sites/library/files/documents/2017-052.pdf. Accessed 11 January 2021.

Arthur, C., Baker, J., Bamford, H., Barnea, N., Lohmann, R., McElwee, K. *et al.* (2009). Summary of the international research workshop on the occurrence, effects, and fate of microplastics marine debris. In *Proceedings of the International Research Workshop of the Occurrence, Effects, and Fate of Microplastics Marine Debris, 9-11 September 2009*. Arthur, C., Baker, J. and Bamford, H. (eds.). Silver Spring, MD: United States National Oceanic and Atmospheric Administration. 7-17. https://marinedebris.noaa.gov/proceedings-international-research-workshop-microplastic-marine-debris. Accessed 11 January 2021.

Asari, M., Tsuchimura, M., Sakai, S., Tsukiji, M. and Sagapolutele, F. (2019). Analysis of mismanaged plastic waste in Samoa to suggest proper waste management in Pacific island countries. *Waste Management and Research*, 37, 1207-1216. https://journals.sagepub.com/doi/10.1177/0734242X19867391. Accessed 20 June 2021.

Ashbullby, K.J., Pahl, S., Webley, P. and White, M.P. (2013). The beach as a setting for families' health promotion: A qualitative study with parents and children living in coastal regions in Southwest England. *Health and Place* 23, 138-147. https://doi.org/10.1016/j.healthplace.2013.06.005. Accessed 11 January 2021.

Ashley, M., Pahl, S., Glegg, G. and Fletcher, S. (2019). A change of mind: Applying social and behavioural research methods to the assessment of the effectiveness of ocean literacy initiatives. *Frontiers in Marine Science* 6, 228. https://doi.org/10.3389/fmars.2019.00288. Accessed 11 January 2021.

Ashton, K., Holmes, L. and Turner, A. (2010). Association of metals with plastic production pellets in the marine environment. *Marine Pollution Bulletin* 60(11), 2050-2055. https://doi.org/10.1016/j.marpolbul.2010.07.014. Accessed 11 January 2021.

Asia-Pacific Economic Cooperation (APEC) (2017). *Capacity Building for Marine Debris Prevention and Management in the APEC Region.* Singapore: Asia-Pacific Economic Cooperation Secretariat. https://www.apec.org/Publications/2017/12/Capacity-Building-for-Marine-Debris-Prevention-and- Management-in-the-APEC-Region. Accessed 11 January 2021.

Association of Plastic Recyclers (2019). APR enhances PET Critical Guidance and Test Documents. https://plasticsrecycling.org/images/Press_Releases/APR-PET-Protocol-Updates-Release-April_2019-Final.pdf. Accessed 11 January 2021.

Athanasopoulou, E., Tombrou, M., Pandis, S.N. and Russell, A.G. (2008). The role of sea-salt emissions and heterogeneous chemistry in the air quality of polluted coastal areas. *Atmospheric Chemistry and Physics* 8, 3807-3841. https://doi.org/10.5194/acp-8-5755-2008. Accessed 11 January 2021.

Atlas, E. and Giam, C.S. (1981). Global transport of organic pollutants: Ambient concentrations in the remote marine atmosphere. *Science* 211(4478), 1653-165. http://doi.org/10.1126/science.211.4478.163. Accessed 11 January 2021.

Au, S.Y., Bruce, T.F., Bridges, W.C. and Klaine, S.J. (2015). Responses of *Hyalella azteca* to acute and chronic microplastic exposures. *Environmental Toxicology and Chemistry* 34(11), 2564-2572. https://doi.org/10.1002/etc.3093. Accessed 11 January 2021.

Auta, H.S., Emenike, C.U. and Fauziah, S.H. (2017). Distribution and importance of microplastics in the marine environment: A review of the sources, fate, effects, and potential solutions. *Environment International* 102, 165-176. http://doi.org/10.1016/j.envint.2017.02.013. Accessed 11 January 2021.

Avery-Gomm, S., Provencher, J.F., Liboiron, M., Poon, F.E. and Smith, P.A. (2018). Plastic pollution in the Labrador Sea: An assessment using the seabird northern fulmar *Fulmarus glacialis* as a biological monitoring species. *Marine Pollution Bulletin* 127, 817-822. https://doi.org/10.1016/j.marpolbul.2017.10.001. Accessed 11 January 2021.

Avio, C.G., Gorbi, S., Milan, M., Benedetti, M., Fattorini, D., d'Errico, G. *et al.* (2015). Pollutants bioavailability and toxicological risk from microplastics to marine mussels. *Environmental Pollution* 198, 211-222. https://doi.org/10.1016/j.envpol.2014.12.021. Accessed 11 January 2021.

Avio, C.G., Gorbi, S. and Regoli, F. (2017). Plastics and microplastics in the oceans: From emerging pollutants to emerged threat. *Marine Environmental Research* 126, 2-11. https://doi.org/10.1016/j.marenvres.2016.05.012. Accessed 11 January 2021.

Awere, E., Obeng, P.A., Bonoli, A. and Obeng, P.A. (2020). E-waste recycling and public exposure to organic compounds in developing countries: A review of recycling practices and toxicity levels in Ghana. *Environmental Technology Reviews* 9(1), 1-19. http://doi.org/10.1080/21622515.2020.1714749. Accessed 11 January 2021.

Azevedo-Santos, V.M., Gonçalves, G.R.L., Manoel, P.S., Andrade, M.C., Lima, F.P. and Pelicice, F.M. (2019). Plastic ingestion by fish: A global assessment. *Environmental Pollution* 255(1), 112994. https://doi.org/10.1016/j.envpol.2019.112994. Accessed 11 January 2021.

Babayemi, J.O., Nnorom, I.C., Osibanjo, O. and Weber, R. (2019). Ensuring sustainability in plastics use in Africa: Consumption, waste generation and projections. *Environmental Sciences Europe* 31, 60. https://doi.org/10.1186/s12302-019-0254-5. Accessed 11 January 2021.

Backhaus, T. and Wagner, M. (2019). Microplastics in the environment: Much ado about nothing? A debate. *Global Challenges* 4(6), 1900022. https://doi.org/10.1002/gch2.201900022. Accessed 11 January 2021.

Bagaev, A., Mizyuk, A., Khatmullina, L., Isachenko, I., and Chubarenko, I. (2017). Anthropogenic fibres in the Baltic Sea water column: Field data, laboratory and numerical testing of their motion. *Science of the Total Environment,* 599, 560-571. https://doi.org/10.1016/j.scitotenv.2017.04.185. Accessed 11 January 2021.

Bailey, M.P. (2020). Sekisui Chemical forms JV to commercialize waste-to-ethanol technology. *Chemical Engineering,* 22 April 2020. https://www.chemengonline.com/sekisui-chemical-forms-jv-to-commercialize-waste-to-ethanol-technology/. Accessed 25 May 2021.

Baini, M., Martellini, T., Cincinelli, A., Campani, T., Minutoli, R., Panti, C. *et al.* (2017). First detection of seven phthalate esters (PAEs) as plastic tracers in superficial neustonic/planktonic samples and cetacean blubber. *Analytic Methods* 9, 1512-1520. http://doi.org/10.1039/c6ay02674e. Accessed 11 January 2021.

Baini, M., Fossi, M.C., Galli, M., Caliani, I., Campani, T., Finoia, M.G. *et al.* (2018). Abundance and characterization of microplastics in the coastal waters of Tuscany (Italy): The application of the MSFD monitoring protocol in the Mediterranean Sea. *Marine Pollution Bulletin* 133, 543-552. https://doi.org/10.1016/j.marpolbul.2018.06.016. Accessed 11 January 2021.

Bakir, A., O'Connor, I.A., Rowland, S.J., Hendriks, A.J. and Thompson, R.C. (2016). Relative importance of microplastics as a pathway for the transfer of hydrophobic organic chemicals to marine life. *Environmental Pollution* 219, 56-65. https://doi.org/10.1016/j.envpol.2016.09.046. Accessed 11 January 2021.

Balcers, O., Brizga, J., Moora, H., and Raal, R. (2019). *Deposit Return Systems for Beverage Containers in the Baltic States, Riga. Green Liberty.* https://www.researchgate.net/publication/332242306_Deposit_Return_Systems_for_Beverage_Containers_in_the_Baltic_States_Riga_Green_Liberty. Accessed 25 May 2021.

Balestri, E., Menicagli, V., Vallerini, F. and Lardicci, C. (2017). Biodegradable plastic bags on the seafloor: A future threat for seagrass meadows? *Science of The Total Environment* 605-606, 755-763. https://doi.org/10.1016/j.scitotenv.2017.06.249. Accessed 11 January 2021.

Balestri, E., Mienicagli, V., Ligorni, V., Fulignati, S., Galletti, A.M.R. and Lardicci, C. (2019). Phytotoxicity assessment of conventional and biodegradable plastic bags using seed germination test. *Ecological Indicators* 102, 569-580. https://doi.org/10.1016/j.ecolind.2019.03.005. Accessed 11 January 2021.

Balestri, E., Menicagli, V., Ligorini, V., Fulignati, S., Galletti, A.M.R. and Lardicci, C. (2020). Reply to "Letter to Editor regarding the article "Evaluation of the phytotoxicity of conventional and biodegradable plastic bags using seed germination tests" by Balestri *et al.* (2019) published on Ecological Indicators 102 (2019): 569-580". *Ecological Indicators* 110, 105876. https://doi.org/10.1016/j.ecolind.2019.105876. Accessed 11 January 2021.

Ballance, A., Ryan, P. and Turpie, J. (2000). How much is a clean beach worth? The impact of litter on beach users in the Cape Peninsula, South Africa. *South African Journal of Science* 96(5), 210. https://www.researchgate.net/publication/279579359_How_much_is_a_clean_beach_worth_The_impact_of_litter_on_beach_users_in_the_Cape_Peninsula_South_Africa. Accessed 11 January 2021.

Ballent, A., Pando, S., Purser, A., Juliano, M.F. and Thomas, L. (2013). Modelled transport of benthic marine microplastic pollution in the Nazaré Canyon. *Biogeosciences* 10, 7957-7970. https://doi.org/10.5194/bg-10-7957-2013. Accessed 11 January 2021.

Ballesteros, L.V., Matthews, J.L. and Hoeksema, B.W. (2018). Pollution and coral damage caused by derelict fishing gear on coral reefs around Koh Tao, Gulf of Thailand. *Marine Pollution Bulletin* 135, 1107-1116. https://doi.org/10.1016/j.marpolbul.2018.08.033. Accessed 11 January 2021.

Barbosa Jr., F., Adeyemi, J.A., Bocato, M.Z., Comas, A. and Campiglia, A. (2020). A critical viewpoint on current issues, limitations, and future research needs on micro- and nanoplastic studies: From the detection to the toxicological assessment. *Environmental Research* 182, 109089. https://doi.org/10.1016/j.envres.2019.109089. Accessed 11 January 2021.

Barboza, L.G.A. and Gimenez, B.C.G. (2015). Microplastic in the marine environment: Current trends and future perspectives. *Marine Pollution Bulletin* 97(1-2), 5-12. https://doi.org/10.1016/j.marpolbul.2015.06.008. Accessed 11 January 2021.

Barnes, D.K. (2002). Biodiversity: Invasions by marine life on plastic debris. *Nature* 416, 808-809. https://doi.org/10.1038/416808a. Accessed 11 January 2021.

Barnes, D.K., Galgani, F., Thompson, R.C. and Barlaz, M. (2009). Accumulation and fragmentation of plastic debris in global environments. *Philosophical Transactions of the Royal Society B: Biological Sciences* 364(1526), 1985-1998. http://doi.org/10.1098/rstb.2008.0205. Accessed 11 January 2021.

Barnes, D.K.A., Morley, S.A., Bell, J., Brewin, P., Brigden, K., Collins, M. *et al.* (2018). Marine plastics threaten giant Atlantic Marine Protected Areas. *Current Biology* 28(19), R1137-R1138. https://doi.org/10.1016/j.cub.2018.08.064. Accessed 11 January 2021.

Barría, C., Brandts, I., Torta, L., Oliveira, M. and Telesa, M. (2020). Effect of nanoplastics on fish health and performance: A review. *Marine Pollution Bulletin* 151, 110791. https://doi.org/10.1016/j.marpolbul.2019.110791. Accessed 11 January 2021.

Barrows, A.P.W., Neumann, C.A., Berger, M.L. and Shaw S.D. (2017). Grab vs. neuston tow net: A microplastic sampling performance comparison and possible advances in the field. *Analytical Methods* 9, 1446-1453. http://doi.org/10.1039/C6AY02387H. Accessed 11 January 2021.

Barrows, A.P.W., Cathey, S.E. and Petersen, C.W. (2018a). Marine environment microfibre contamination: Global patterns and the diversity of microparticle origins. *Environmental Pollution* 237, 275-284. https://doi.org/10.1016/j.envpol.2018.02.062. Accessed 11 January 2021.

Barrows, A.P.W., Christiansen, K. S., Bode, E.T. and Hoellein, T.J. (2018b). A watershed-scale, citizen science approach to quantifying microplastics concentration in a mixed land use river. Water Research 147, 382-392. https://doi.org/10.1016/j.watres.2018.10.013. Accessed 11 January 2021.

Batel, A., Linti, F., Scherer, M., Erdinger, L. and Braunbeck, T. (2016). Transfer of benzo[a]pyrene from microplastics to *Artemia nauplii* and further to zebrafish via a trophic food web experiment: CYP1A induction and visual tracking of persistent organic pollutants. *Environmental Toxicology and Chemistry* 35(7), 1656-1666. http://doi.org/10.1002/etc.3361. Accessed 11 January 2021.

Battisti, C., Staffieri, E., Poeta, G., Sorace, A., Luiselli, L. and Amori, G. (2019). Interactions between anthropogenic litter and birds: A global review with a 'black-list' of species. *Marine Pollution Bulletin* 138, 93-114. https://doi.org/10.1016/j.marpolbul.2018.11.017. Accessed 11 January 2021.

Beaumont, N.J., Aanesen, M., Austen, M.C., Börger, T., Clark, J.R., Cole, M. *et al.* (2019). Global ecological, social and economic impacts of marine plastic. *Marine Pollution Bulletin* 142, 189-195. https://doi.org/10.1016/j.marpolbul.2019.03.022. Accessed 11 January 2021.

Beck, M.W., Losada, I.J., Menéndez, P., Reguero, B.G., Díaz-Simal, P. and Fernández, F. (2018). The global flood protection savings provided by coral reefs. *Nature Communications* 9, 2186. https://doi.org/10.1038/s41467-018-04568-z. Accessed 11 January 2021.

Beckwith, V.K., and Fuentes, M.M. (2018). Microplastic at nesting grounds used by the northern Gulf of Mexico loggerhead recovery unit. *Marine Pollution Bulletin*, 131, 32–37. https://doi.org/10.1016/j.marpolbul.2018.04.001 Accessed June 9 2021

Bejgarn, S., MacLeod, M., Bogdal, C. and Breitholz, M. (2015). Toxicity of leachate from weathering plastics: An exploratory screening study with *Nitocra spinipes*. *Chemosphere* 132, 114-119. https://doi.org/10.1016/j.chemosphere.2015.03.010. Accessed 11 January 2021.

Bellingeri, A., Bergami E., Grassi, G,. Faleri, C., Redondo-Hasselerham, P., Koelmans, A.A. *et al.* (2019). Combined effects of nanoplastics and copper on the freshwater alga *Raphidocelis subcapitata*. *Aquatic Toxicology* 210, 179-187. https://doi.org/10.1016/j.aquatox.2019.02.022. Accessed 11 January 2021.

Belzagui, F., Crespi, M., Álvarez, A., Gutiérrez-Bouzán, C. and Vilaseca,M. (2019). Microplastics' emissions: Microfibres' detachment from textile garments. *Environmental Pollution* 248, 1028-1035. https://doi.org/10.1016/j.envpol.2019.02.059. Accessed 11 January 2021.

Bergevin, K. (2018). The 10 types of litter most commonly found on beaches around the world. WorldAtlas, 19 September. https://www.worldatlas.com/articles/the-10-types-of-litter-most-commonly-found-on-beaches-around-the-world.html. Accessed 11 January 2021.

Bergmann, M., Gutow, L. and Klages, E. (eds.) (2015). *Marine Anthropogenic Litter*. Cham: Springer Open Access. https://link.springer.com/book/10.1007/978-3-319-16510-3. Accessed 11 January 2021.

Bergmann, M., Mützel, S., Primpke, S., Tekman, M.B., Trachsel, J. and Gerdts, G. (2019). White and wonderful? Microplastics prevail in snow from the Alps to the Arctic. *Science Advances* 5(8), eaax1157. https://doi.org/10.1126/sciadv.aax1157. Accessed 11 January 2021.

Besley, A., Vijver, M.G., Behrens, P. and Bosker, T. (2017). A standardized method for sampling and extraction methods for quantifying microplastics in beach sand. *Marine Pollution Bulletin* 114(1), 77-83. https://doi.org/10.1016/j.marpolbul.2016.08.055. Accessed 11 January 2021.

Besseling, E., Redondo-Hasselerharm, P., Foekema, E.M. and Koelmans, A.A. (2019). Quantifying ecological risks of aquatic micro-and nanoplastic. *Critical Reviews in Environmental Science and Technology* 49(1), 32-80. https://doi.org/10.1080/10643389.2018.1531688. Accessed 11 January 2021.

Best, J. (2019). Anthropogenic stresses on the world's big rivers. *Nature Geoscience* 12(1),7-21. https://doi.org/10.1038/s41561-018-0262-x. Accessed 11 January 2021.

Biancamaria, S., Lettenmaier, D.P. and Pavelsky, T.M. (2016). The SWOT Mission and its capabilities for land hydrology. *Surveys in Geophysics* 37, 307-337. https://doi.org/10.1007/s10712-015-9346-y. Accessed 11 January 2021.

Bindoff, N.L., Cheung, W.W.L., Kairo, J.G., Arístegui, J., Guinder, V.A. *et al.* (2019). Changing ocean, marine ecosystems, and dependent communities. In *IPCC Special Report on the Ocean and Cryosphere in a Changing Climate*. Pörtner, H.-O., Roberts, D.C., Masson-Delmotte, V., Zhai, P., Tignor, M. *et al.* (eds.). Geneva: Intergovernmental Panel on Climate Change. https://www.ipcc.ch/site/assets/uploads/sites/3/2019/11/09_SROCC_Ch05_FINAL-1.pdf. Accessed 12 October 2021.

Birch, Q.T., Potter, P.M., Pinto, P.X., Dionysiou, D.D. and Al-Abed, S.R. (2020). Sources, transport, measurement and impact of nano and microplastics in urban watersheds. *Reviews in Environmental Science and Bio/Technology* 19, 275-336. https://doi.org/10.1007/s11157-020-09529-x. Accessed 11 January 2021.

Birkbeck, C.D. (2020). *Strengthening International Cooperation to Tackle Plastic Pollution: Options for the WTO. Global Governance Brief No. 01*. Graduate Institute Geneva, Global Governance Centre. https://static1.squarespace.com/static/5b0520e5d274cbfd845e8c55/t/5e25683a556e15498ad1e73f/1579509842688/Plastic_Trade_WTO_Final.pdf. Accessed 11 January 2021.

Black, J.E., Kopke, K. and O'Mahony, C. (2019). A trip upstream to mitigate marine plastic pollution – a perspective focused on the MSFD and WFD. *Frontiers in Marine Science* 6, 1-6. https://doi.org/10.3389/fmars.2019.00689. Accessed 11 January 2021.

Blank, F., Stumbles, P.A., Seydoux, E., Holt, P.G., Fink, A., Rothen-Rutishauser, B. *et al.* (2013). Size-dependent uptake of particles by pulmonary antigen-presenting cell populations and trafficking to regional lymph nodes. *American Journal of Respiratory Cell Molecular Biology* 49, 67-77. https://doi.org/10.1165/rcmb.2012-0387OC. Accessed 11 January 2021.

Blarer, P. and Burkhardt-Holm, P. (2016). Microplastics affect assimilation efficiency in freshwater amphipod *Gammarus fossarum*. *Environmental Science and Pollution Research* 23, 23522-23532. https://doi.org/10.1007/s11356-016-7584-2. Accessed 11 January 2021.

Blettler, M.C., Abrial, E., Khan, F.R., Sivri, N. and Espinola, L.A. (2018). Freshwater plastics pollution: Recognizing research biases and identifying knowledge gaps. *Water Research* 143, 416-424. https://doi.org/10.1016/j.watres.2018.06.015. Accessed 11 January 2021.

Blettler, M.C. and Wantzen, K.M. (2019). Threats underestimated in freshwater plastic pollution: Mini-review. *Water, Air and Soil Pollution* 230, 174. https://doi.org/10.1007/s11270-019-4220-z. Accessed 11 January 2021.

Boag, A.H., Colby, T.V., Fraire, A. E., Kuhn, C., Roggli, V.L., Travis, W.D. *et al.* (1999). The pathology of interstitial lung disease in nylon flock workers. *American Journal of Surgical Pathology* 23(12), 1539. http://doi.org/10.1097/00000478-199912000-00012. Accessed 11 January 2021.

Börger, C.M., Lattin, G.L., Moore, S.L. and Moore, C.J. (2010). Plastic ingestion by planktivorous fishes in the North Pacific Central gyre. *Marine Pollution Bulletin* 60(12), 2275-2278. https://doi.org/10.1016/j.marpolbul.2010.08.007. Accessed 11 January 2021.

Bond, K., Benham, H., Vaughan, E. and Chou, L. (2020). *The Future's Not in Plastics. Why plastics demand won't rescue the oil sector. Carbon Tracker Initiative. Analyst Report*. https://carbontracker.org/reports/the-futures-not-in-plastics/. Accessed 11 January 2021.

Boots, B., Russell, C.W. and Green, D.S. (2019). Effects of microplastics in soil ecosystems: Above and below ground. *Environmental Science and Technology* 53, 11496-11506. https://doi.org/10.1021/acs.est.9b03304. Accessed 11 January 2021.

Börger, T., Hattam, C., Burdon, D., Atkins, J.P. and Austen, M.C. (2014). Valuing conservation benefits of an offshore marine protected area. *Ecological Economics* 108, 229-241. https://doi.org/10.1016/j.ecolecon.2014.10.006. Accessed 11 January 2021.

Borja, A.M. and Elliott, J. (2019). So when will we have enough papers on microplastics and ocean litter? *Marine Pollution Bulletin* 146, 312-316. https://doi.org/10.1016/j.marpolbul.2019.05.069. Accessed 11 January 2021.

Borrelle, S.B., Rochman, C., Liboironc, M., Bond, A.L., Lusher, A., Bradshaw, H. *et al.* (2017). Why we need an international agreement on marine plastic pollution. *Proceedings of the National Academy of Sciences* 114(38), 9994-9997. https://doi.org/10.1073/pnas.1714450114. Accessed 11 January 2021.

Borrelle, S.B., Ringma, J., Law, K.L., Monnahan, C.C., Lebreton, L., McGiverb, A. *et al.* (2020). Predicted growth in plastic waste exceeds efforts to mitigate plastic pollution. *Science* 369(6510), 1515-1518. https://doi.org/10.1126/science.aba3656. Accessed 11 January 2021.

Botterell, Z.L.R., Beaumont, N., Dorrington, T., Steinke, M., Thompson, R.C. and Lindeque, P.K. (2019). Bioavailability and effects of microplastics on marine zooplankton: A review. *Environmental Pollution* 245, 98-110. https://doi.org/10.1016/j.envpol.2018.10.065. Accessed 11 January 2021.

Boucher, J. and Bilard, G. (2020). *The Mediterranean: Mare plasticum*. Gland, Switzerland: IUCN. https://portals.iucn.org/library/sites/library/files/documents/2020-030-En.pdf. Accessed 30 June 2021

Boucher, J. and Friot, D. (2017). *Primary Microplastics in the Oceans: A Global Evaluation of Sources*. Gland, Switzerland: International Union for Conservation of Nature and Natural Resources (IUCN). https://portals.iucn.org/library/sites/library/files/documents/2017-002-En.pdf. Accessed 11 January 2021.

Bourdages, M.P.T., Provencher, J.F., Sudlovenick, E., Ferguson, S.H., Young, B.G., Murphy, M.J.J. *et al.* (2020). No plastics detected in seal (*Phocidae*) stomachs harvested in the eastern Canadian Arctic. *Marine Pollution Bulletin* 150, 110772. https://doi.org/10.1016/j.marpolbul.2019.110772. Accessed 11 January 2021.

Bouwmeester, H., Hollman, P.C.H. and Peters, R.J.B (2015). Potential health impact of environmentally released micro- and nanoplastics in the human food production chain: Experiences from nanotoxicology. *Environmental Science and Technology* 49(15), 8932-8947. http://doi.org/10.1021/acs.est.5b01090. Accessed 11 January 2021.

Bowman, D., Manor-Samsonov, N. and Golik, A. (1998). Dynamics of litter pollution on Israeli Mediterranean beaches: A budgetary, litter flux approach. *Journal of Coastal Research* 14(2), 418-432. https://www.jstor.org/stable/4298796. Accessed 11 January 2021.

Bradney, L., Wijesekara, H., Palansooriya, K.N., Obadamudalige, N., Bolan, N.S., Ok, Y.S. *et al.* (2019). Particulate plastics as a vector for toxic trace-element uptake by aquatic and terrestrial organisms and human health risk. *Environment International* 131, 104937. https://doi.org/10.1016/j.envint.2019.104937. Accessed 11 January 2021.

Brandon, J., Goldstein, M. and Ohman, M.D. (2016). Long-term aging and degradation of microplastic particles: Comparing in situ oceanic and experimental weathering patterns. *Marine Pollution Bulletin* 110(1), 299-308. https://doi.org/10.1016/j.marpolbul.2016.06.048. Accessed 11 January 2021.

Brate, I.L.N., Eidsvoll, D.P., Steindal, C.C. and Thomas, K.V. (2016). Plastic ingestion by Atlantic cod (*Gadus morhua*) from the Norwegian coast. *Marine Pollution Bulletin* 112(102), 105-110. https://doi.org/10.1016/j.marpolbul.2016.08.034. Accessed 11 January 2021.

Braun, U., Jekel, M., Gerdts, G., Ivleva, N.P. and Reiber, J. (2018). *Microplastics Analytics. Sampling, Preparation and Detection Methods.* Discussion Paper within the scope of the research of the Bundesministerium für Bildung und Forschung. Plastics in the Environment: Sources, Sinks, Solutions. Berlin. https://www.ecologic.eu/sites/files/publication/2018/discussion_paper_mp_analytics_en.pdf. Accessed 11 January 2021.

Brennecke, D., Ferreira, E.C., Costa, T.M., Appel, D., da Gama, B.A. and Lenz, M. (2015). Ingested microplastics (>100 µm) are translocated to organs of the tropical fiddler crab *Uca rapax*. *Marine Pollution Bulletin* 96(1-2), 491-495. https://doi.org/10.1016/j.marpolbul.2015.05.001. Accessed 11 January 2021.

Brooks, A.L., Wang, S. and Jambeck, J.R. (2018). The Chinese import ban and its impact on global plastic waste trade. *Science Advances* 4(6), eaat0131. http://doi.org/10.1126/sciadv.aat0131. Accessed 11 January 2021.

Brouwer, R., Hadzhiyska, D., Ioakeimidis, C. and Ouderdorp, H. (2017). The social costs of marine litter along European coasts. *Ocean and Coastal Management* 138, 38-49. https://doi.org/10.1016/j.ocecoaman.2017.01.011. Accessed 11 January 2021.

Brown, T.M. and Takada, H. (2017). Indicators of marine pollution in the North Pacific. *Archives of Environmental Contamination and Toxicology* 73, 171-175. https://doi.org/10.1007/s00244-017-0424-7. Accessed 11 January 2021.

Browne, M.A., Dissanayake, A., Galloway, T.S., Lowe, D.M. and Thompson, R.C. (2008). Ingested microscopic plastic translocates to the circulatory system of the mussel, *Mytilus edulis* (L.). *Environmental Science and Technology* 42(13), 5026-5031. https://doi.org/10.1021/es800249a. Accessed 11 January 2021.

Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T. *et al.* (2011). Accumulation of microplastics on shorelines worldwide: Sources and sinks. *Environmental Science and Technology* 45(21), 9175-9179. https://doi.org/10.1021/es201811s. Accessed 11 January 2021.

Browne, M.A., Niven, S.J., Galloway, T.S., Rowland, S.J. and Thompson, R.C. (2013). Microplastic moves pollutants and additives to worms, reducing functions linked to health and biodiversity. *Current Biology* 23(23), 2388-2392. https://doi.org/10.1016/j.cub.2013.10.012. Accessed 11 January 2021.

Bryant, J.A., Clemente, T.M., Viviani, D.A., Fong, A.A., Thomas, K.A., Kemp, P. *et al.* (2016). Diversity and activity of communities inhabiting plastics debris in the North Pacific Gyre. *mSystems* 1, 00024-16. http://doi.org/10.1128/mSystems.00024-16. Accessed 11 January 2021.

Bucci, K., Tulio, M. and Rochman, C.M. (2019). What is known and unknown about the effects of plastic pollution: A meta-analysis and systematic review. *Ecological Applications* 30(2), e02044. https://doi.org/10.1002/eap.2044. Accessed 11 January 2021.

Bucol, L., Romanos, E., Cabcaban, S., Siplon, L.M., Madrid, G.C., Bucol, A. and Polidoro, B. (2020). Microplastics in marine sediments and rabbitfish (*Siganus fuscescens*) from selected coastal areas of Negros Oriental, Philippines. *Marine Pollution Bulletin* 150, 110685. https://doi.org/10.1016/j.marpolbul.2019.110685. Accessed 11 January 2021.

Buhl-Mortensen, L. and Buhl-Mortensen, P. (2017). Marine litter in the Nordic Seas: Distribution composition and abundance. *Marine Pollution Bulletin* 125, 260-270. https://doi.org/10.1016/j.marpolbul.2017.08.048. Accessed 11 January 2021.

Burkhardt-Holm, P. and N'Guyen, A. (2019). Ingestion of microplastics by fish and other prey organisms of cetaceans, exemplified for two large baleen whale species. *Marine Pollution Bulletin* 144, 224-234. https://doi.org/10.1016/j.marpolbul.2019.04.068. Accessed 11 January 2021.

Burns, E.E. and Boxall, A.B.A. (2018). Microplastics in the aquatic environment: Evidence for or against adverse impacts and major knowledge gaps. *Environmental Toxicology and Chemistry* 37, 2776-2796. https://doi.org/10.1002/etc.4268. Accessed 11 January 2021.

Business Research Company (2020). Plastics and Rubber Products Manufacturing Global Market Report 2020. https://www.thebusinessresearchcompany.com/report/plastics-and-rubber-products-manufacturing-global-market-report. Accessed 11 January 2021.

Ca, V.T. (2020). A regional ocean governance framework for the integrated management of the environment and biological resources in the South China Sea. In *Building a Normative Order in the South China Sea. Evolving Disputes, Expanding Options*. Tran, T.T., Welfiled, J.B. and Le T.T. (eds.). Cheltenham, U.K.: Edward Elgar. 196-210. https://ideas.repec.org/h/elg/eechap/17679_10.html. https://ideas.repec.org/h/elg/eechap/17679_10.html. Accessed 11 January 2021.

Cai, H., Du, F., Li, L., Li, B., Li, J. and Shi, H. (2019). A practical approach based on FT-IR spectroscopy for identification of semi-synthetic and natural celluloses in microplastic investigation. *Science of The Total Environment* 669, 692-701. https://doi.org/10.1016/j.scitotenv.2019.03.124. Accessed 12 January 2021.

Cai, L., Wang, J., Peng, J., Tan, Z., Zhan, Z., Tan, X. *et al.* (2017). Characteristics of microplastics in the atmospheric fallout from Dongguan city, China: Preliminary research and first evidence. *Environmental Science and Pollution Research* 24(32), 24928-24935. https://doi.org/10.1007/s11356-019-06979-x. Accessed 12 January 2021.

Camacho, M., Herrera, A., Gómez, M., Acosta-Dacal, A., Martínez, I., Henríquez-Hernández, L.A. *et al.*(2019). Organic pollutants in marine plastic debris from Canary Islands beaches. *Science of The Total Environment* 662, 22-31. https://doi.org/10.1016/j.scitotenv.2018.12.422. Accessed 12 January 2021.

Campanale, C., Suaria, G., Bagnuolo, G., Baini, M., Galli, M., de Rysky, E. *et al.* (2019). Visual observations of floating macro litter around Italy (Mediterranean Sea). *Mediterranean Marine Science* 20, 271-281. https://doi.org/10.12681/mms.19054. Accessed 12 January 2021.

Campanale, C., Massarelli, C., Savino, I., Locaputo, V. and Uricchio, V.F. (2020). A detailed review study on potential effects of microplastics and additives of concern on human health. *International Journal of Environmental Research and Public Health* 17, 1212. http://doi.org/10.3390/ijerph17041212. Accessed 12 January 2021.

Campbell, M., King, S., Heppenstall, L.D., Gool, E., Martin, R., and Hewitt, C.L. (2017). Aquaculture and urban marine structures facilitate native and non-indigenous species transfer through generation and accumulation of marine debris. *Marine Pollution Bulletin* 123(1), 304-312. https://doi.org/10.1016/j.marpolbul.2017.08.040. Accessed 12 January 2021.

Campbell, M.L, Peters, L., McMains, C., Rodrigues de Campos, M.C., Sargisson, R.J. *et al.* (2019). Are our beaches safe? Quantifying the human health impact of anthropogenic beach litter on people in New Zealand. *Science of The Total Environment* 651, 2400-2409. https://doi.org/10.1016/j.scitotenv.2018.10.137. Accessed 12 January 2021.

Canals, M., Pham, C.K., Bergmann, M., Gutow, L., Hank, G. *et al.* (2021). The quest for seafloor macrolitter: A critical review of background knowledge, current methods and future prospects. *Environmental Research Letters,* 16, 023001. https://iopscience.iop.org/article/10.1088/1748-9326/abc6d4. Accessed 16 February 2021.

Canning-Clode, J., Sepúlveda, P., Almeida, S. and Monteiro, J. (2020). Will COVID-19 containment and treatment measures drive shifts in marine litter pollution? *Frontiers in Marine Science*, 25 August. https://doi.org/10.3389/fmars.2020.00691. Accessed 12 January 2021.

Carney Almroth, B. and Eggert, H. (2019). Marine plastics pollution: Sources, impacts and policy issues. *Review of Environmental Economics and Policy* 13, 317-26. https://doi.org/10.1093/reep/rez012. Accessed 12 January 2021.

Carr, S.A., Liu, J. and Tesoro, A.G. (2016). Transport and fate of microplastics particles in wastewater treatment plants. *Water Research* 91, 174182. https://doi.org/10.1016/j.watres.2016.01.002. Accessed 12 January 2021.

Carson, H.S., Colbert, S.L., Kaylor, M.J. and McDermid, K.J. (2011). Small plastics debris changes water movement and heat transfer through beach sediments. *Marine Pollution Bulletin* 62(8), 1708-1713. https://doi.org/10.1016/j.marpolbul.2011.05.032. Accessed 12 January 2021.

Carson, H.S., Nerheim, M.S., Carroll, K.A. and Eriksen, M. (2013). The plastic-associated microorganisms of the North Pacific gyre. *Marine Pollution Bulletin* 75(1-2), 126-132. https://doi.org/10.1016/j.marpolbul.2013.07.054. Accessed 12 January 2021.

Carvalho-Souzaa, G.F., Llope, M., Tinôco, M.S., Medeiros, D.V., Maia-Nogueira, R. and Sampaio, C.L.S. (2018). Marine litter disrupts ecological processes in reef systems. *Marine Pollution Bulletin* 133, 464-471. https://doi.org/10.1016/j.marpolbul.2018.05.049. Accessed 12 January 2021.

Cassou, E. (2018). *Agricultural Pollution: Plastics.* Washington, D.C.: World Bank, https://openknowledge.worldbank.org/handle/10986/29505. Accesssed 12 January 2021.

Castro-Jiménez, J., González-Fernández, D., Fornier, M., Schmidt, N. and Sempere, R. (2019). Macro-litter in surface waters from the Rhone River: Plastics pollution and loading to the NW Mediterranean Sea. *Marine Pollution Bulletin* 146, 60-66. https://doi.org/10.1016/j.marpolbul.2019.05.067. Accessed 12 January 2021.

Catarino, A.I., Macchia, V., Sanderson, W.G., Thompson, R.C. and Henry, T.B. (2018). Low levels of microplastics (MP) in wild mussels indicate that MP ingestion by humans is minimal compared to exposure via household fibres fallout during a meal. *Environmental Pollution* 237, 675e684. https://doi.org/10.1016/j.envpol.2018.02.069. Accessed 12 January 2021.

Cau, A., Bellodi, A., Moccia, D., Mulas, A., Porcu, C., Pusceddu, A. *et al.* (2019). Shelf-life and labels: A cheap dating tool for seafloor macro litter? Insights from MEDITS surveys in Sardinian sea. *Marine Pollution Bulletin* 14, 430-433. https://doi.org/10.1016/j.marpolbul.2019.03.004. Accessed 12 January 2021.

Center for International Environmental Law (2019). *Plastics and Climate. The Hidden Costs of a Plastic Planet.* https://www.ciel.org/wp-content/uploads/2019/05/Plastic-and-Climate-FINAL-2019.pdf. Accessed 12 January 2021.

Centurioni, L., Chen, Z., Lumpkin, R., Braasch, L., Brassington, G., Chao, Y. *et al.* (2019). Multidisciplinary global *in situ* observations of essential climate and ocean variables at the air-sea interface in support of climate variability and change studies and to improve weather forecasting, pollution, hazard and maritime safety assessments. *Frontiers in Marine Science*, 30 August. https://doi.org/10.3389/fmars.2019.00419. Accessed 12 January 2021.

Cesa, F.S., Turra, A. and Baruque-Ramos, J. (2017). Synthetic fibres as microplastics in the marine environment: A review from textile perspective with a focus on domestic washings. *Science of The Total Environment* 598, 1116-1129. https://doi.org/10.1016/j.scitotenv.2017.04.172. Accessed 12 January 2021.

Chan, H.S.H., Dingle, C. and Not, C. (2019). Evidence for non-selective ingestion of microplastic in demersal fish. *Marine Pollution Bulletin* 149, 110523. https://doi.org/10.1016/j.marpolbul.2019.110523. Accessed 12 January 2021.

Chang, X.R., Xue, Y.Y., Li, J.Y., Zou, L.Y. and Tang, M. (2020). Potential health impact of environmental micro- and nanoplastics pollution. *Journal of Applied Toxicology* 40, 4-15. https://doi.org/10.1002/jat.3915. Accessed 12 January 2021.

Charles, D., Kimman, L. and Saran, N. (2021). *The Plastic Waste Makers Index*. Minderoo Foundation. https://www.minderoo.org/plastic-waste-makers-index/. Accessed 25 May 2021.

Chen, C.-L. (2015). Regulation and management of marine litter. In *Marine Anthropogenic Litter*. Bergmann, M., Gutow, L. and Klages, E. (eds.). Springer Open. 395-428. https://link.springer.com/content/pdf/bfm%3A978-3-319-16510-3%2F1.pdf. Accessed 12 January 2021.

Chen, G., Feng, Q. and Wang, J. (2020). Mini-review of microplastics in the atmosphere and their risks to humans. *Science of The Total Environment* 703, 135504. https://doi.org/10.1016/j.scitotenv.2019.135504. Accessed 12 January 2021.

Cheung, P.K., Cheung, L.T.O. and Fok, L. (2016). Seasonal variation in the abundance of marine plastic debris in the estuary of a subtropical macro-scale drainage basin in South China. *Science of The Total Environment* 562, 658-665. https://doi.org/10.1016/j.scitotenv.2016.04.048. Accessed 12 January 2021.

Chiba, S., Saito, H., Fletcher, R., Yogi, T., Kayo, M., Miyagi, S. *et al.* (2018). Human footprint in the abyss: 30 year records of deep-sea plastic debris. *Marine Policy* 96, 204-212. https://doi.org/10.1016/j.marpol.2018.03.022. Accessed 12 January 2021.

Cho, D.O. (2005). Challenges to marine debris management in Korea. *Coastal Management* 33(4), 389-409. https://doi.org/10.1080/08920750500217559. Accessed 12 January 2021.

Choy, C.A., Robison, B.H., Gagne, T.O., Erwin, B. and Firi, E. (2019). The vertical distribution and biological transport of marine microplastics across the epipelagic and mesopelagic water column. *Scientific Reports* 9(1), 7843. https://doi.org/10.1038/s41598-019-44117-2. Accessed 12 January 2021.

Chua, E., Shimeta, J., Nugegoda, D., Morrison, P.D. and Clarke, B.O. (2014). Assimilation of polybrominated diphenyl ethers from microplastics by the marine amphipod, *Allorchestes compressa*. *Environmental Science and Technology* 48(14), 8127-8134. https://doi.org/10.1021/es405717z. Accessed 12 January 2021.

Chubarenko, I., Bagaev, A., Zobkov, M. and Esiukova, E. (2016). On some physical and dynamical properties of microplastic particles in marine environments. *Marine Pollution Bulletin* 108(1-2), 105-112. https://doi.org/10.1016/j.marpolbul.2016.04.048. Accessed 12 January 2021.

Chubarenko, I.P., Esiukova, E.E., Bagaev, A.V., Bagaeva, M.A. and Grave, A.N. (2018). Three- dimensional distribution of anthropogenic microparticles in the body of sandy beaches. *Science of The Total Environment* 628-629, 1340-1351. https://doi.org/10.1016/j.scitotenv.2018.02.167. Accessed 12 January 2021.

Claessens, M., van Cauwenberghe, L., Vandegehuchte, M.B. and Janssen, C.R. (2013). New techniques for the detection of microplastics in sediments and field collected organisms. *Marine Pollution Bulletin* 70, 227-233. https://doi.org/10.1016/j.marpolbul.2013.03.009. Accessed 12 January 2021.

Claro, F., Fossi, M.C., Ioakeimidis, C., Baini, M., Lusher, A.L. McFee, W. *et al.* (2019). Tools and constraints in monitoring interactions between marine litter and megafauna: Insights from case studies around the world. *Marine Pollution Bulletin* 141, 147-160. https://doi.org/10.1016/j.marpolbul.2019.01.018. Accessed 12 January 2021.

Clukey, K.E., Lepczyk, C.A., Balazs, G.H., Work, T.M., Li, Q.X., Bachman. M.J. *et al.* (2018). Persistent organic pollutants in fat of three species of Pacific pelagic longline caught sea turtles: Accumulation in relation to ingested plastic marine debris. *Science of The Total Environment* 610-611, 402-411. https://doi.org/10.1016/j.scitotenv.2017.07.242. Accessed 12 January 2021.

Colborn, T., Kwiatkowski, C., Schultz, K. and Bachran, M. (2011). Natural gas operations from a public health perspective. *Human and Ecological Risk Assessment: An International Journal* 17(5), 1039-1056. https://doi.org/10.1080/10807039.2011.605662. Accessed 12 January 2021.

Cole, M., Lindeque, P., Halsband, C. and Galloway, T. S. (2011). Microplastics as contaminants in the marine environment: A review. *Marine Pollution Bulletin*, 62, 2588–2597. https://doi.org/10.1016/j.marpolbul.2011.09.025. Accessed 7 June 2021.

Cole, M., Lindeque, P., Fileman, E., Halsband, C. and Galloway, T.S. (2015). The impact of polystyrene microplastics on feeding, function and fecundity in the marine copepod *Calanus helgolandicus*. *Environmental Science and Technology* 49(2), 1130-1137. https://doi.org/10.1021/es504525u. Accessed 12 January 2021.

Cole, M., Lindeque, P.K., Fileman, E., Clark, J., Lewis, C., Halsband, C. *et al.* (2016). Microplastics alter the properties and sinking rates of zooplankton faecal pellets. *Environmental Science and Technology* 50(6), 3239-3246. https://doi.org/10.1021/acs.est.5b05905. Accessed 12 January 2021.

Coleby, A.M. and Grist, E.P.M. (2019). Prioritized area mapping for multiple stakeholders through geospatial modelling: A focus on marine plastics pollution in Hong Kong. *Ocean and Coastal Management* 171, 131-141. https://doi.org/10.1016/j.ocecoaman.2018.12.021. Accessed 12 January 2021.

Collins, C. and Hermes, J.C. (2019). Modelling the accumulation and transport of floating marine microplastics around South Africa. *Marine Pollution Bulletin* 139, 46-58. https://doi.org/10.1016/j.marpolbul.2018.12.028. Accessed 12 January 2021.

Comnea-Stancu, I.R., Wieland, K., Ramer, G., Schwaighofer, A. and Lendl, B. (2017). On the identification of rayon/viscose as a major fraction of microplastics in the marine environment: Discrimination between natural and manmade cellulosic fibres using Fourier transform infrared spectroscopy. *Applied Spectroscopy* 71, 939-950. https://doi.org/10.1177/0003702816660725. Accessed 12 January 2021.

Consoli, P., Andaloro, F., Altobelli, C., Battaglia, P., Campagnuolo, S., Canese, S. *et al.* (2018). Marine litter in an EBSA (Ecologically of Biologically Significant Area) of the central Mediterranean Sea: Abundance, composition, impact on benthic species and basis for monitoring entanglement. *Environmental Pollution* 236, 405-415. https://doi.org/10.1016/j.envpol.2018.01.097. Accessed 12 January 2021.

Constantino, E., Martins, I., Sierra, J.M.S. and Bessa, F. (2019). Abundance and composition of floating marine macro litter on the eastern sector of the Mediterranean Sea. Marine *Pollution Bulletin* 138, 260-265. https://doi.org/10.1016/j.marpolbul.2018.11.008. Accessed 12 January 2021.

Conti, G.O., Ferrante, M., Banni, M., Favara, C., Nicolosi, I., Cristaldi, A., Fiore, M. and Zuccarello, P. (2020). Micro-and nano-plastics in edible fruit and vegetables. The first diet risks assessment for the general population. E*nvironmental Research* 187, 109677. https://doi.org/10.1016/j.envres.2020.109677. Accessed 12 January 2021.

Convention on the Conservation of Migratory Species of Wild Animals (2014). Adopted at COP11: Res.11.30: Management of Marine Debris. https://www.cms.int/en/document/management-marine-debris-3. Accessed 12 January 2021.

Corcoran, P.L. (2021). Degradation of microplastics in the environment. In *Handbook of Microplastics in the Environment.* Rocha-Santos, T., Costa, M. and Mouneyrac, C. (eds.). Cham: Springer. 1-12. https://doi.org/10.1007/978-3-030-10618-8_10-1. Accessed 12 January 2021.

Cormier, R. and Elliott, M. (2017). SMART marine goals, targets and management – is SDG 14 operational or aspirational, is '*Life Below Water*' sinking or swimming? *Marine Pollution Bulletin* 123, 28-33. https://doi.org/10.1016/j.marpolbul.2017.07.060. Accessed 12 January 2021.

Corradini, F., Pablo Meza, P., Eguiluz, R., Casado, F., Huerta-Lwanga, E. and Geissen, V. (2019). Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. *Science of The Total Environment* 671, 411-420. https://doi.org/10.1016/j.scitotenv.2019.03.368. Accessed 12 January 2021.

Costa, M.F. and Duarte, A.C. (2017). Microplastics sampling and sample handling. In *Comparative Analytical Chemistry* 75. Rocha-Santos, T.A.P. and Duarte, A.C. (eds.). Elsevier. 25-47. https://doi.org/10.1016/bs.coac.2016.11.002. Accessed 12 January 2021.

Costanza, R., d'Arge, R., de Grooy, R., Farber, S., Grasso, M., Hannon, B. *et al.* (1997). The value of the world's ecosystem services and natural capital. *Nature* 387, 253-260. https://doi.org/10.1038/387253a0. Accessed 12 January 2021.

Costanza, R., de Groot, R., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S. *et al.* (2014). Changes in the global value of ecosystem services. *Global Environmental Change* 26, 152-158. https://doi.org/10.1016/j.gloenvcha.2014.04.002. Accessed 12 January 2021.

Courtene-Jones, W., Quinn, B., Ewins, C., Gary, S.F. and Narayanaswamy, B.E. (2018). Consistent microplastic ingestion by deep-sea invertebrates over the last four decades (1976-2015), a study from the North East Atlantic. *Environmental Pollution* 244, 503-512. https://doi.org/10.1016/j.envpol.2018.10.090. Accessed 12 January 2021.

Cowger, W., Gray, A.B. and Schult, R.C. (2019). Anthropogenic litter cleanups in Iowa riparian areas reveal the importance of near-stream and watershed scale land use. *Environmental Pollution* 250, 981-989. http://doi.org/10.1016/j.envpol.2019.04.052. Accessed 12 January 2021.

Cox, K., Covernton, A., Davies, H., Dower, J., Juanes, F. and Dudas, S. (2019). Human consumption of microplastics. *Environmental Science and Technology* 53(12), 7068-7074. https://doi.org/10.1021/acs.est.9b01517. Accessed 12 January 2021.

Cózar, A., Echevarría, F., González-Gordillo, J.I., Irigoien, X., Úbeda, B., Hernández-León, S. *et al.* (2014). Plastic debris in the open ocean. *Proceedings of the National Academy of Sciences* 111(28), 10239-10244. https://doi.org/10.1073/pnas.1314705111. Accessed 12 January 2021.

Cózar, A., Sanz-Martin, M., Marti, E., González-Gordillo, J.I., Ubeda, B., Gálvez, J.A., Irigoen, X. and Duarte, C.M. (2015). Plastic accumulation in the Mediterranean Sea. *PLoS ONE* 10(4), e0121762. https://doi.org/10.1371/journal.pone.0121762. Accessed 12 January 2021.

Cózar, A., Marti, E., Duarte, C.M., Garcia-de-Lomas, J., van Sebille, E., Ballatore, T.J. *et al.* (2017). The Arctic Ocean as a dead end for floating plastics in the North Atlantic branch of the thermohaline circulation. *Scientific Advances* 3(4), e1600582. http://doi.org/10.1126/sciadv.1600582. Accessed 12 January 2021.

Critchell, K. and Lambrechts, J. (2016). Modelling accumulation of marine plastics in the coastal zone; what are the dominant physical processes? *Estuarine, Coastal and Shelf Science* 171, 111-122. https://doi.org/10.1016/j.ecss.2016.01.036. Accessed 12 January 2021.

Cui, R., Kim, S.W. and An, Y.J. (2017). Polystyrene nanoplastics inhibit reproduction and induce abnormal embryonic development in the freshwater crustacean *Daphnia galeata*. *Scientific Reports* 7(1), 1-10. https://doi.org/10.1038/s41598-017-12299-2. Accessed 12 January 2021.

Cui, Y., Hu, M. and Liu, J. (2019). Values of traceability in supply chains. *SSRN Electronic Journal*. http://doi.org/10.2139/ssrn.3291661. Accessed 12 January 2021.

da Costa, J. (2018). Micro- and nanoplastics in the environment: Research and policymaking. *Current Opinions in Environmental Science and Health* 1,12-16. https://doi.org/10.1016/j.coesh.2017.11.002. Accessed 12 January 2021.

da Costa, J., Santos, P.S.M., Duarte, A.C. and Rocha-Santos, T. (2016). (Nano)plastics in the environment – sources, fates and effects. *Science of The Total Environment* 566-567, 15-26. https://doi.org/10.1016/j.scitotenv.2016.05.041. Accessed 12 January 2021.

Damerell, P., Howe, C. and Gulland, E.J. (2013). Child-oriented environmental education influences adult knowledge and household behaviour. *Environmental Research Letters* 8, 15016-15022. http://dx.doi.org/10.1088/1748-9326/8/1/015016. Accessed 12 January 2021.

Daniel, D.B., Thomas, S.N. and Thomson, K.T. (2020). Assessment of fishing-related plastic debris along the beaches in Kerala Coast, India. *Marine Pollution Bulletin* 150, 110696. https://doi.org/10.1016/j.marpolbul.2019.110696. Accessed 12 January 2021.

Danovaro, R., Fanelli, E., Canals, M., Ciuffardi, T., Fabri, M.-C., Taviani, M. *et al.* and IDEM Consortium (2020). Towards a marine strategy for the deep Mediterranean Sea: Analysis of current ecological status. *Marine Policy* 112, 103781. https://doi.org/10.1016/j.marpol.2019.103781. Accessed 12 January 2021.

Dalberg Advisors, WWF Mediterranean Marine Initiative (2019). *Stop the Flood of Plastic: How Mediterranean Countries Can Save Their Sea*. WWF-World Wide Fund for Nature. http://awsassets.panda.org/downloads/a4_plastics_reg_low.pdf. Accessed 11 January 2021.

Danopoulos, E., Jenner, L.C., Twiddy, M., and Rotchell, J.M. (2020). Microplastic contamination of seafood intended for human consumption: a systematic review and meta-analysis. *Environmental Health Perspective,* 128, 126002. https://doi.org/10.1289/EHP7171. Accessed 9 June 2021.

Darbra, R., Dan, J.G., Casal, J., Àgueda, A., Capri, E., Fait, G. *et al.* (2011). Additives in the textile industry. In *Global Risk-Based Management of Chemical Additives I: Production, Usage and Environmental Occurrence*. Bilitewski, B., Darbra, R.M. and Barcelo, D. (eds.). *The Handbook of Environmental Chemistry vol.* 18. Berlin, Heidelberg: Springer 83-107. https://doi.org/10.1007/698_2011_101 . Accessed 11 January 2021.

Dauvergne, P. (2018). Why is the global governance of plastic failing the oceans? *Global Environmental Change* 51, 22-31. https://doi.org/10.1016/j.gloenvcha.2018.05.002. Accessed 11 January 2021.

Davison, P. and Asch, R.G. (2011). *Plastic Ingestion by Mesopelagic Fishes in the North Pacific Subtropical Gyre.* Marine Ecology Progress Series 432, 173-180. https://doi.org/10.3354/meps09142. Accessed 11 January 2021.

Dawson, A.L., Kawaguchi, S., King, C.K., Townsend, K.A., King, R. Huston, W.M. *et al.* (2018). Turning microplastics into nanoplastics through digestive fragmentation by Antarctic krill. *NatureCommunications* 9, 1001. https://www.nature.com/articles/s41467-018-03465-9.pdf. Accessed 11 January 2021.

De Frond, H.L., van Sebille, E., Parnis, J.M., Diamond, M.L., Mallos, N., Kingsbury, T. *et al.* (2018). Estimating the mass of chemicals associated with ocean plastic pollution to inform mitigation efforts. *Integrated Environmental Assessment Management* 15, 596-606. https://doi.org/10.1002/ieam.4147. Accessed 11 January 2021.

De Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M. *et al.* (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services* 1, 50-61. https://doi.org/10.1016/j.ecoser.2012.07.005. Accessed 30 November 2020.

Dehaut, A., Cassone, A.L., Frere, L., Hermabessiere, L., Himber, C., Rinnert *et al.* (2016). Microplastics in seafood: Benchmark protocol for their extraction and characterization. *Environmental Pollution* 215, 223-233. https://doi.org/10.1016/j.envpol.2016.05.018. Accessed 11 January 2021.

Deloitte (2019). *The Price Tag of Plastic Pollution: An Economic Assessment of River Plastic*. https://www2.deloitte.com/content/dam/Deloitte/nl/Documents/strategy-analytics-and-ma/deloitte-nl-strategy-analytics-and-ma-the-price-tag-of-plastic-pollution.pdf. Accessed 12 February 2021.

Deng, Y., Zhang, Y., Lemos, B. and Ren, H. (2017). Tissue accumulation of microplastics in mice and biomarker responses suggest widespread health risks of exposure. *Scientific Reports* 7, 46687. https://doi.org/10.1038/srep46687. Accessed 11 January 2021.

de Ruijter, V.N., Redondo-Hasselerharm, P.E., Gouin, T., and Koelmans, A.A. (2020). Quality criteria for microplastic effect studies in the context of risk assessment: A critical review. *Environmental Scienc and Technology* 54(19), 11692-11705. https://pubs.acs.org/doi/10.1021/acs.est.0c03057. Accessed 11 January 2021.

de Sá, L.C., Oliveira, M., Ribeiro, F., Rocha, T.L. and Futter, M.N. (2018). Studies of the effects of microplastics on aquatic organisms: What do we know and where should we focus our efforts in the future? *Science of The Total Environment* 645, 1029-1039. https://doi.org/10.1016/j.scitotenv.2018.07. 207. Accessed 11 January 2021.

Desforges, J P., Galbraith, M. and Ross, P.S. (2015). Ingestion of microplastics by zooplankton in the Northeast Pacific Ocean. *Archives of Environmental Contamination and Toxicology* 69, 320-330. https://doi.org/10.1007/s00244-015-0172-5. Accessed 11 January 2021.

Deshpande, P.C., Philis, G., Brattebø and Fet, A.M. (2020). Using material flow analysis (MFA) to generate the evidence on plastic waste management from commercial fishing gears in Norway. *Resources, Conservation and Recycling: X* 5,100024. https://doi.org/10.1016/j.rcrx.2019.100024. Accessed 11 January 2021.

Díaz-Torres, E.R., Ortega-Ortiz, C.D., Silva-Iñiguez, L., Nene-Preciado, A. and Torres Orozco, E. (2017). Floating marine debris in waters of the Mexican Central Pacific. *Marine Pollution Bulletin* 115 (1-2), 225-232. https://doi.org/10.1016/j.marpolbul.2016.11.065. Accessed 11 January 2021.

Diez, S.M., Patil, P.G., Morton, J., Rodriguez, D.J., Vanzella, A., Robin, D.V. *et al.* (2019). *Marine Pollution in the Caribbean: Not a Minute to Waste.* Washington, D.C.: World Bank Group. https://documents.worldbank.org/en/publication/documents-reports/documentdetail/482391554225185720/marine-pollution-in-the-caribbean-not-a-minute-to-waste. Accessed 12 January 2021.

Dilkes-Hoffman, L.S., Ashworth, P, Laycock, B., Pratt, S. and Lant, P. (2019a). Public attitudes towards bioplastics – knowledge, perception and end-of-life management. *Resources Conservation and Recycling* 151, 104479. https://doi.org/10.1016/j.resconrec.2019.104479. Accessed 12 January 2021.

Dilkes-Hoffman, L.S., Pratt, S., Laycock, B., Ashworth, P. and Lant, P.A. (2019b). Public attitudes towards plastics. *Resources, Conservation and Recycling* 147, 227-235. https://doi.org/10.1016/j.resconrec.2019.05.005. Accessed 12 January 2021.

Donohue, M.J., Masura, J., Gelatt, T., Ream, R., Baker, J.D., Faulhaber, K. *et al.* (2019). Evaluating exposure of northern fur seals, *Callorhinus ursinus*, to microplastic pollution through faecal analysis. *Marine Pollution Bulletin* 138, 213-221. https://doi.org/10.1016/j.marpolbul.2018.11.036. Accessed 12 January 2021.

Driedger, A.G.J., Dürr, H.H., Mitchell, K. and Van Cappellen, P. (2015). Plastic debris in the Laurentian Great Lakes: A review. *Journal of Great Lakes Research* 41 (1), 9-19. https://doi.org/10.1016/j.jglr.2014.12.020. Accessed 12 January 2021.

Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N. and Tassin, B. (2015a). Microplastic contamination in an urban area: A case study in Greater Paris. *Environmental Chemistry* 12(5), 592-599. https://doi.org/10.1071/EN14167. Accessed 12 January 2021.

Dris, R., Imhof, H., Sanchez, W., Gasperi, J., Galgani, F., Tassin, B. *et al.* (2015b). Beyond the ocean: Contamination of freshwater ecosystems with (micro-) plastics particles. *Environmental Chemistry* 12(5), 539-550. https://doi.org/10.1071/EN14172. Accessed 12 January 2021.

Dris, R., Gasperi, J., Saad, M., Mirande-Bret, C. and Tassin, B. (2016). Synthetic fibres in atmospheric fallout: A source of microplastics in the environment? *Marine Pollution Bulletin* 104(1-2), 290-293. https://doi.org/10.1016/j.marpolbul.2016.01.006. Accessed 12 January 2021.

Dris, R., Gasperi, J., Mirande, C., Mandin, C., Guerrouache, M., Langlois, V. *et al.* (2017). A first overview of textile fibres, including MPs, in indoor and outdoor environments. *Environmental Pollution* 221, 453-458. https://doi.org/10.1016/j.envpol.2016.12.013. Accessed 12 January 2021.

Dris, R., Gasperi, J., Rocher, V. and Tassin, B. (2018). Synthetic and non-synthetic anthropogenic fibres in a river under the impact of Paris Megacity: Sampling methodological aspects and flux estimations. *Science of The Total Environment* 618, 157-164. https://doi.org/10.1016/j.scitotenv.2017.11.009. Accessed 12 January 2021.

Dris, R., Tramoy, R., Alligant, S., Gasperi, J. and Tassin, B. (2020) Plastic debris flowing from rivers to oceans: the role of the estuaries as a comple and poorly understood key interface. In *Handbook on Microplastics in the Envrionment*. Rocha-Santos, T., Coasts, M., and Mouneyrac, C. (eds). Cham: Springer. 1-28 https://doi.org/10.1007/978-3-030-10618-8_3-1. Access 16 January 2021.

Duhec, A.V., Jeanne, R.F., Maximenko, N. and Hafner, J. (2015). Composition and potential origin of marine debris stranded in the Western Indian Ocean on remote Alphonse Island, Seychelles. *Marine Pollution Bulletin* 96(1-2), 76-86. https://doi.org/10.1016/j.marpolbul.2015.05.042. Accessed 12 January 2021.

Duncan, E.M, Broderick, A.C., Fuller, W.J., Galloway, T.S., Godfrey, M.H., Hamann, M. *et al.* (2018a). Microplastic ingestion ubiquitous in marine turtles. *Global Change Biology* 25, 744-752. https://doi.org/10.1111/gcb.14519. Accessed 12 January 2021.

Duncan, E.M., Arrowsmith, J., Bain, C., Broderick, A.C., Lee, J., Metcalfe, K. *et al.* (2018b). The true depth of the Mediterranean plastic problem: Extreme microplastic pollution on marine turtle nesting beaches in Cyprus. *Marine Pollution Bulletin* 136, 334-340. https://doi.org/10.1016/j.marpolbul.2018.09.019. Accessed 12 January 2021.

Dunlop, S.W., Dunlop, B.J. and Brown, M. (2020). Plastic pollution in paradise: Daily accumulation rates of marine litter on Cousine Island, Seychelles. *Marine Pollution Bulletin* 151, 110803. https://doi.org/10.1016/j.marpolbul.2019.110803. Accessed 12 January 2021.

Dussud, C., Meistertzheim, A.L., Conan, P., Pujo-Pay, M., George, M., Fabre, P. *et al.* (2018a). Evidence of niche partitioning among bacteria living on plastics, organic particles and surrounding seawaters. *Environmental Pollution* 236, 807-816. https://doi.org/10.1016/j.envpol.2017.12.027. Accessed 12 January 2021.

Dussud, C., Hudec, C., George, M., Fabre, P., Higgs, O., Bruzuad, S. *et al.* (2018b). Colonization of non-biodegradable and biodegradable plastics by marine microorganisms. *Frontiers in Microbiology* 9, 1571. https://doi.org/10.3389/fmicb.2018.01571. Accessed 12 January 2021.

Eagle, L., Hamann, M. and Low, D.R. (2016). The role of social marketing, marine turtles and sustainable tourism in reducing plastic pollution. *Marine Pollution Bulletin* 107(1), 324-332. https://doi.org/10.1016/j.marpolbul.2016.03.040. Accessed 12 January 2021.

Eckert, E.M., Di Cesare, A., Kettner, M.T., Arias-Andres, M., Fontaneto, D., Grossart, H.-P. *et al.* (2018). Microplastics increase impact of treated wastewater on freshwater microbial community. *Environmental Pollution* 234, 495-502. https://doi.org/10.1016/J.ENVPOL.2017.11.070. Accessed 12 January 2021.

Ecorys (2017). Supporting study for an impact assessment for the revision of Directive 2000/59/EC on Port Reception Facilities. https://ec.europa.eu/transport/sites/default/files/2017-06-support-study-ia-prf-dir.pdf. Accessed 12 January 2021.

Eerkes-Medrano, D., Thompson, R.C. and Aldridge, D.C. (2015). Microplastics in freshwater systems: A review of the emerging threats, identification of knowledge gaps and prioritisation of research needs. *Water Research* 75, 63-82. https://doi.org/10.1016/j.watres.2015.02.012. Accessed 12 January 2021.

Ellen MacArthur Foundation (2016). *The New Plastics Economy: Rethinking the Future of Plastics and Catalysing Action*. https://ellenmacarthurfoundation.org/the-new-plastics-economy-rethinking-the-future-of-plastics-and-catalysing. Accessed 12 January 2021.

Ellen MacArthur Foundation (2020). *The Global Commitment Progress Report 2020*. https://www.newplasticseconomy.org/assets/doc/Global-Commitment-2020-Progress-Report.pdf. Accessed 12 January 2021.

Ellen MacArthur Foundation (2021). *Upstream Innovation. A Guide to Packaging Solutions.* https://www.ellenmacarthurfoundation.org/publications/upstream-innovation. Accessed 13 July 2021.

Enyoh, C.E., Verla, A.W., Verla, E.N., Ibe, F.C. and Amaobi, C.E. (2019). Airborne microplastics: A review study on method for analysis, occurrence, movement and risks. *Environmental Monitoring and Assessment* 191, 668. https://doi.org/10.1007/s10661-019-7842-0. Accessed 12 January 2021.

Eriksen, M., Mason, S., Wilson, S., Box, C., Zellers, A., Edwards, W. *et al.* (2013). Microplastics pollution in the surface waters of the Laurentian Great Lakes. *Marine Pollution Bulletin* 77(1-2), 177-182. https://doi.org/10.1016/j.marpolbul.2013.10.007. Accessed 12 January 2021.

Eriksen, M., Lebreton, L.C., Carson, H.S., Thiel, M., Moore, C.J., Borerro, J.C. *et al.* (2014). Plastic pollution in the world's oceans: More than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. *PLoS ONE* 9(12), e111913. https://doi.org/10.1371/journal.pone.0111913. Accessed 12 January 2021.

Erni-Cassola, G., Zadjelovic, V., Gibson, M.I. and Christie-Oleza, J.A. (2019). Distribution of plastic polymer types in the marine environment; A meta-analysis. *Journal of Hazardous* Materials 369, 691-698. https://doi.org/10.1016/j.jhazmat.2019.02.067. Accessed 12 January 2021.

Esiukova, E., Zobkov, M. and Chubarenko, I. (2019). Data on microplastic contamination of the Baltic Sea bottom sediment samples in 2015-2016. *Data in Brief* 28, 104887. https://doi.org/10.1016/j.dib.2019.104887. Accessed 12 January 2021.

Esmen, N.A. and Corn, M. (1971). Residence time of particles in urban air. *Atmospheric Environment* (1967) 5(8), 571-578. https://doi.org/10.1016/0004-6981(71)90113-2. Accessed 12 January 2021.

Espinosa, C., García Beltrán, J.M., Esteban, M.A. and Cuesta, A. (2018). *In vitro* effects of virgin microplastics on fish head-kidney leucocyte activities. *Environmental Pollution* 235, 30-38. https://doi.org/10.1016/j.envpol.2017.12.054. Accessed 12 January 2021.

European Bioplastics (2020). Bioplastics market data. https://www.european-bioplastics.org/market/. Accessed 16 January 2021.

European Chemical Industry Council (2020). *2020 Facts and Figures of the European Chemical Industry.* https://cefic.org/app/uploads/2019/01/The-European-Chemical-Industry-Facts-And-Figures-2020.pdf. Accessed 12 January 2021.

European Chemicals Agency (2017a). *Annex XV Report: An Evaluation of the Possible Health Risks of Recycled Rubber Granules Used As Infill In Synthetic Turf Sports Fields.* https://echa.europa.eu/documents/10162/13563/annex-xv_report_rubber_granules_en.pdf/dbcb4ee6-1c65-af35-7a18-f6ac1ac29fe4. Accessed 12 January 2021.

European Chemicals Agency (2017b). Glyphosate Not Classified as a Carcinogen by ECHA – All News – ECHA. Accessed 4/5/20. https://echa.europa.eu/-/glyphosate-not-classified-as-a-carcinogen-by-echa. Accessed 20/6/2021.

European Chemicals Agency (2018). Guidance in a Nutshell on requirements for substances in articles. Version 3.0 – December 2017 https://echa.europa.eu/documents/10162/23036412/nutshell_guidance_articles2_en.pdf/1e13dcce-b46b-43cb-904e-6c4675613e9d. Accessed 12 January 2021.

European Chemicals Agency (2019). *Annex XV Registration Report: Proposal for a Restriction. Substance Name(s): intentionally added microplastics*. https://echa.europa.eu/documents/10162/05bd96e3-b969-0a7c-c6d0-441182893720. Accessed 12 January 2021.

European Commission (2017). *Nautical Tourism*. Commission Staff Working Document. Brussels, 30.3.2017 SWD(2017) 126 final https://ec.europa.eu/oceans-and-fisheries/system/files/2021-03/swd-2017-126_en.pdf. Accessed 31 January 2021.

European Commission (2018a). *Reducing Marine Litter: Action on single-use plastics and fishing gear. Accompanying the document Proposal for a Directive of the European Parliament and of the Council on the reduction of the impact of certain plastics products on the environment. Commission Staff Working Document Impact Assessment 28.5.2018 SWD(2018) 254 final.* PART 1/3. Brussels. https://eur-lex.europa.eu/resource.html?uri=cellar:4d0542a2-6256-11e8-ab9c-01aa75ed71a1.0001.02/DOC_1&format=PDF. Accessed 25 May 2021.

European Commission (2018b). Single-use plastics: New EU rules to reduce marine litter. https://ec.europa.eu/commission/presscorner/detail/en/MEMO_18_3909. Accessed 12 January 2021.

European Commission (2018c). Proposal for a Directive of the European Parliament and of the Council on port reception facilities for the delivery of waste from ships, repealing Directive 2000/59/EC and amending Directive 2009/16/EC. 16.1.2018 COM (2018) 33 SWD 22 Final. https://eur-lex.europa.eu/legal-content/EN/TXT/DOC/?uri=CELEX:52018SC0021&from=EN. Accessed 15 February 2021.

European Commission (2019). The European Green Deal. Brussels, 11.12.2019 COM(2019) 640 final. https://ec.europa.eu/info/sites/default/files/european-green-deal-communication_en.pdf. Accessed 20 June 2021.

European Commission (2020). Circular economy action plan. https://ec.europa.eu/environment/strategy/circular-economy-action-plan_en. Accessed 19 August 2020.

European Commission ARCADIS (2014). *Final Report: Marine Litter Study to support the establishment of an initial quantitative headline reduction target - SFRA0025. European Commission DG Environment, Project Number BE0113.000668/final version.* https://ec.europa.eu/environment/marine/good-environmental-status/descriptor-10/pdf/final_report.pdf. Accessed 12 January 2021.

European Food Safety Authority Panel on Contaminants in the Food Chain (2016). Presence of microplastics and nanoplastics in food, with particular focus on seafood. *EFSA Journal* 14(6), e04501. https://doi.org/10.2903/j.efsa.2016.4501. Accessed 12 January 2021.

European Union (2019a). *Environmental and Health Risks of Microplastic Pollution. Group of Chief Scientific Advisors Scientific Opinion 6/2019 (Supported by SAPEA Evidence Review Report No. 4).* Scientific Advice Mechanism (SAM). https://doi.org/10.2777/65378. Accessed 12 January 2021.

European Union (2019b). *Directive (EU) 2019/904 of the European Parliament and of the Council of 5 June 2019 on the reduction of the impact of certain plastic products on the environment. Official Journal of the European Union* L 155/1. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32019L0904. Accessed 12 January 2021.

European Union (2019c). *Directive (EU) 2019/883 of the European Parliament and of the Council of 17 April 2019 on port reception facilities for the delivery of waste from ships, amending Directive 2010/65/EU and repealing Directive 2000/59/EC* L151/116. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32019L0883&from=IT. Accessed 15 February 2021.

European Union Network for the Implementation and Enforcement of Environmental Law (2019). All about plastic waste. IMPEL Waste and TFS Conference, Bucharest, October 2019. https://www.impel.eu/impel-waste-tfs-conference-all-about-plastic-waste-bucharest-october-2019/. Accessed 12 January 2021.

Evangeliou, N., Grythe, H., Klimont, Z. Heyes, C., Eckhardt,S., Lopez-Aparicio, S. and Stohl, A. (2020). Atmospheric transport is a major pathway of microplastics to remote regions. *Nature Communications* 11, 3381. https://doi.org/10.1038/s41467-020-17201-9. Accessed 12 January 2021.

Eyheraguibel, B., Leremboure, M., Traikia, M., Sancelme, M., Bonhomme, S., Fromageot, D. *et al.* (2018). Environmental scenarii for the degradation of oxo-polymers. *Chemosphere* 198, 182-190. https://doi.org/10.1016/j.chemosphere.2018.01.153. Accessed 12 January 2021.

Fanini, L. and Bozzeda, F. (2018). Dynamics of plastic resin pellets deposition on a microtidal sandy beach: Informative variables and potential integration into sandy beach studies. *Ecological Indicators* 89, 309-316. https://doi.org/10.1016/j.ecolind.2018.02.027. Accessed 12 January 2021.

FAO (Food and Agriculture Organization of the United Nations) (2020). *The State of World Fisheries and Aquaculture 2020*. Rome. http://www.fao.org/state-of-fisheries-aquaculture. Accessed 12 January 2021.

Farrelly, T., Borelle, S. and Fuller, S. (2020). *Plastic Pollution Prevention in Pacific Island Countries: Gap Analysis of Current Legislation, Policies and Plans*. Environmental Investigation Agency. https://reports.eia-international.org/wp-content/uploads/sites/6/2020/09/Plastic-Prevention-Gap-Analysis-2020.pdf. Accessed 12 January 2021.

Faure, F., Demars, C., Wieser, O., Kunz, M. and de Alencastro, L.F. (2015). Plastics pollution in Swiss surface waters: Nature and concentrations, interaction with pollutants. *Environmental Chemistry* 12(5), 582-591. https://doi.org/10.1071/EN14218. Accessed 12 January 2021.

Fazey, F.M. and Ryan, P.G. (2016). Biofouling on buoyant marine plastics: An experimental study into the effect of size on surface longevity. *Environmental Pollution* 210, 354-360. https://doi.org/10.1016/j.envpol.2016.01.026. Accessed 12 January 2021.

Ferreira, I., Venânciob, C., Lopesb, I. and Oliveira, M. (2019). Nanoplastics and marine organisms: What has been studied? *Environmental Toxicology and Pharmacy* 67, 1-7. https://doi.org/10.1016/j.etap.2019.01.006. Accessed 12 January 2021.

Ferreira, S., Convery, F. and McDonnell, S. (2007). The most popular tax in Europe? Lessons from the Irish plastic bags levy. *Environmental and Resource Economics* 38, 1-11. https://doi.org/10.1007/s10640-006-9059-2. Accessed 12 January 2021.

Filho, W.L., Saari, U., Fedoruk, M., Iital, A., Moora, H., Klöga, M. *et al.* (2019). An overview of the problems posed by plastic products and the role of extended producer responsibility in Europe. *Journal of Cleaner Production* 214, 550-558. https://doi.org/10.1016/j.jclepro.2018.12.256. Accessed 12 January 2021.

Fischer, B., Milunov, M., Floredo, Y., Hofbauer, P. and Joas, A. (2014). *Identification of Relevant Emission Pathways to the Environment and Quantification of Environmental Exposure for Bisphenol A.* Project No. (FKZ) 360 01 063, Report No. (UBA-FB) 001933/E. Dessau-Roßlau, Germany: Federal Environment Agency. (UmweltBundesamt). https://www.umweltbundesamt.de/en/publikationen/identification-of-relevant-emission-pathways-to-the. Accessed 20 June 2021.

Flaws, J., Damdimopolou, P., Patisaul, H.B., Gore, A., Raetzman, L., and Vandenberg, L.N. (2020). *Plastics, EDCs and Health. A Guide for Public Interest Organisations and Policy-makers on Endocrine Disrupting Chemicals and Plastics*. Endocrine Society and IPEN. https://www.endocrine.org/-/media/endocrine/files/topics/edc_guide_2020_v1_6chqennew-version.pdf. Accessed 25 May 2021.

Fleet, D., Vlachogianni, T. and Hanke, G. (2021). *A Joint List of Litter Categories for Marine Macrolitter Monitoring*. EUR 30348 EN; JRC121708. Luxembourg: Publications Office of the European Union. https://doi.org/10.2760/127473. Accessed 17 August 2021.

Forrest, A., Giacovazzi, L., Dunlop. S., Reisser, J., Tickler, D., Jamieson, A. *et al.* (2019). Eliminating plastic pollution: How a voluntary contribution from industry will drive the circular plastics economy. *Frontiers in Marine Science* 6, 627. https://doi.org/10.3389/fmars.2019.00627. Accessed 12 January 2021.

Forrest, S.A., Bourdages, M.P.T. and Vermaire, J.C. (2020). Microplastics in freshwater ecosystems. In *Handbook of Microplastics in the Environment*. Rocha-Santos, T., Costa, M., and Mouneyrac, C. (eds.). Cham: Springer. 1-19. https://doi.org/10.1007/978-3-030-10618-8_2-1. Accessed 12 January 2021.

Fortibuoni, T., Ronchi, F., Macic, V., Mandic, M., Mazziotto, C., Peterlin, M. *et al.* (2019). A harmonized and coordinated assessment of the abundance and composition of seafloor litter in the Adriatic-Ionian macroregion (Mediterranean Sea). *Marine Pollution Bulletin* 139, 412-426. https://doi.org/10.1016/j.marpolbul.2019.01.017. Accessed 12 January 2021.

Fossatti, M., Teixeira de Mello, F. Lozoya, J.P. (2020). Mesoplastics and large microplastics along a use gradient on the Uruguay Atlantic coast: Types, sources, fates, and chemical loads. *Science of The Total Environment* 721, 137734. https://doi.org/10.1016/j.scitotenv.2020.137734. Accessed 13 January 2021.

Fossi, M.C., Panti, C., Guerranti, C., Coppola, D., Giannetti, M., Marsili, L. *et al.* (2012). Are baleen whales exposed to the threat of microplastics? A case study of the Mediterranean fin whale (*Balaenoptera physalus*). *Marine Pollution Bulletin* 64(11), 2374-2379. https://doi.org/10.1016/j.marpolbul.2012.08.013. Accessed 12 January 2021.

Fossi, M.C., Coppola, D., Baini, M., Giannetti, M., Guerranti, C., Marsili, L. *et al.* (2014). Large filter feeding marine organisms as indicators of microplastics in the pelagic environment: The case studies of the Mediterranean basking shark (*Cetorhinus maximus*) and fin whale (*Balaenoptera physalus*). *Marine Environmental Research* 100, 17-24. https://doi.org/10.1016/j.marenvres.2014.02.002. Accessed 12 January 2021.

Fossi, M.C., Panti, C., Baini, M. and Lavers, J.L. (2018). A review of plastic-associated pressures: Cetaceans of the Mediterranean Sea and Eastern Australian Shearwaters as case studies. *Frontiers in Marine Science* 5, 173. https://doi.org/10.3389/fmars.2018.00173. Accessed 12 January 2021.

Fossi, M.C., Vlachogianni, T., Galgani, F., Innocenti, F.D., Zampetti, G. and Leone, G. (2020). Assessing and mitigating the harmful effects of plastic pollution: The collective multi-stakeholder driven Euro- Mediterranean response. *Ocean and Coastal Management* 184, 105005. https://doi.org/10.1016/j.ocecoaman.2019.105005. Accessed 12 January 2021.

Franceschini, S., Mattei, F., D'Andrea, L., Nardi, A. Di, Fiorentino, F., Garofalo, G. *et al.* (2019). Rummaging through the bin: Modelling marine litter distribution using Artificial Neural Networks. *Marine Pollution Bulletin* 149, 110580. https://doi.org/10.1016/j.marpolbul.2019.110580. Accessed 12 January 2021.

Franco-Trecu, V., Drago, M., Katz, H., Machin, E. and Marin, Y. (2017). With the noose around the neck: Marine debris entangling otariid species. *Environmental Pollution* 220 (Part B), 985-989. https://doi.org/10.1016/j.envpol.2016.11.057. Accessed 12 January 2021.

Fred-Ahmadu, O.H., Bhagwat, G., Oluyoye, I., Benson, N.U., Ayejuyo, O.O and Palanisami, T. (2020). Interaction of chemical contaminants with microplastics: Principles and perspectives. *Science of The Total Environment* 706, 135978. https://doi.org/10.1016/j.scitotenv.2019.135978. Accessed 12 January 2021.

Frère, L., Paul-Pont, I., Moreau, J., Soudant, P., Lambert, C., Huvet, A. *et al.* (2016). A semi-automated Raman micro-spectroscopy method for morphological and chemical characterizations of microplastic litter. *Marine Pollution Bulletin* 113(1-2), 461-468. https://doi.org/10.1016/j.marpolbul.2016.10.051. Accessed 12 January 2021.

Frias, J.P.G.L. and Nash, R. (2019). Microplastics: Finding a consensus on the definition. *Marine Pollution Bulletin* 138, 145-147. https://doi.org/10.1016/j.marpolbul.2018.11.022. Accessed 12 January 2021.

G20 (2019). *G20 Workshop on Scientific Knowledge and Innovative Solutions for Marine Plastic Litter: Supporting the G20 Implementation Framework for Actions on Marine Plastic Litter. 8 October 2019, Tokyo, Japan: Meeting Report*. Luxembourg: Publications Office of the European Union. https://op.europa.eu/en/publication-detail/-/publication/2371d5f9-22d6-11ea-af81-01aa75ed71a1/language-en/format-PDF. Accessed 12 January 2021.

Galgani, F., Hanke, G. and Maes, T. (2015). Global distribution, composition and abundance of marine litter. In *Marine Anthropogenic Litter*. Bergmann, M., Gutow, L and Klages, M. (eds.). Cham: Springer. 29-56. https://link.springer.com/content/pdf/bfm%3A978-3-319-16510-3%2F1.pdf. Accessed 12 January 2021.

Galgani, F., Pham, C.K., Claro, F. and Consoli, P. (2018). Marine animal forests as useful indicators of entanglement by marine litter. *Marine Pollution Bulletin* 135, 735-738. https://doi.org/10.1016/j.marpolbul.2018.08.004. Accessed 12 January 2021.

Galgani, F., Brien, A. So., Weis, J., Ioakemedidis, C., Schuyler, Q., Makarenko, I. *et al.* (2021). Are litter, plastic and microplastic quantities increasing in the ocean? *Microplastics and Nanoplastics* 1, 2. https://doi.org/10.1186/s43591-020-00002-8. Accessed 17 August 2021.

Galimany, E., Marco-Herrero, E., Soto, S., Recasens, L., Lombarte, A., Lleonart, J. *et al.* (2019). Benthic marine litter in shallow fishing grounds in the NW Mediterranean Sea. *Waste Management* 95, 620-627. https://doi.org/10.1016/j.wasman.2019.07.004. Accessed 12 January 2021.

Gallo, F., Fossi, C., Weber, R., Santillo, D., Sousa, J., Ingram, I. *et al.* (2018). Marine litter plastics and microplastics and their toxic chemicals components: The need for urgent preventive measures. *Environmental Sciences Europe* 30, 13. https://doi.org/10.1186/s12302-018-0139-z. Accessed 12 January 2021.

Galloway, T.S. and Lewis C.N. (2016). Marine microplastics spell big problems for future generations. *Proceedings of the National Academy of Sciences* 113, 2331-2333. https://doi.org/10.1073/pnas.1600715113. Accessed 12 January 2021.

Galloway, T.S., Cole, M. and Lewis, C. (2017). Interactions of microplastic debris throughout the marine ecosystem. *Nature Ecology and Evolution* 1(5), 1-8. https://doi.org/10.1038/s41559-017-0116. Accessed 12 January 2021.

Garaba, S.P. and Dierssen, H.M. (2018). An airborne remote sensing case study of synthetic hydrocarbon detection using short wave infrared absorption features identified from marine- harvested macro- and microplastics. *Remote Sensing of Environment* 205, 224-235. https://doi.org/10.1016/j.rse.2017.11.023. Accessed 12 January 2021.

Garaba, S.P., Aitken, J., Slat, B., Dierssen, H.M., Lebreton, L., Zielinski, O. and Reisser, J. (2018). Sensing ocean plastics with an airborne hyperspectral shortwave infrared imager. *Environmental Science and Technology* 52(20), 11699-11707. https://doi.org/10.1021/acs.est.8b02855. Accessed 12 January 2021.

Garcés-Ordóñeza, O., Castillo-Olayab, V.A., Granados-Briceñoc, A.F., Blandón Garcíaa, L.M., Espinosa Díaz, L.F. (2019). Marine litter and microplastic pollution on mangrove soils of the Ciénaga Grande de Santa Marta, Colombian Caribbean. *Marine Pollution Bulletin* 145, 455-462. https://doi.org/10.1016/j.marpolbul.2019.06.058. Accessed 12 January 2021.

Garcia-Garin, O., Garcia-Cuebas, I., Drago, M., Rita, D., Parga, M., Gazo, M. and Cardona, L. (2020). No evidence of microplastics in Antarctic fur seal scats from a hotspot of human activity in Western Antarctic. *Science of The Total Environment* 737, 140210. https://doi.org/10.1016/j.scitotenv.2020.140210. Accessed 12 January 2021.

Garcia-Soto, C., van der Meeren, G.I., Busch, J.A., Delany, J., Domegan, C., Dubsky, K. *et al.* (2017). *Advancing Citizen Science for Coastal and Ocean Research*. French, V., Kellett, P., Delany, J. and McDonough, N. (eds.). Position Paper 23. Ostend, Belgium: European Marine Board. https://www.marineboard.eu/sites/marineboard.eu/files/public/publication/EMB_PP23_Citizen_Science_web.pdf. Accessed 12 January 2021.

Gasperi, J., Dris, R., Miranda-Bret, C. Mandin, C., Langlois, V. and Tassin, B. (2015). First overview of microplastics in indoor and outdoor air. 15th EuChemMS International Conference on Chemistry and the Environment. https://www.researchgate.net/publication/281657363_First_overview_of_microplastics_in_indoor_and_outdoor_air. Accessed 12 January 2021.

Gattringer, C.W. (2018). A revisited conceptualization of plastic pollution accumulation in marine environments: Insights from a social ecological economics perspective. *Marine Pollution Bulletin* 96, 221-226. https://doi.org/10.1016/j.marpol.2017.11.036. Accessed 12 January 2021.

Gavigan, J., Kefela, T., Macadam-Somer, I., Suh, S. and Geyer R. (2020). Synthetic microfiber emissions to land rival those to waterbodies and are growing. *PLoS ONE* 15(9), e0237839. https://doi.org/10.1371/journal.pone.0237839. Accessed 12 January 2021.

Geiger, S.M., Geiger, M. and Wilhelm, O. (2019) Environment-specific vs. general knowledge and their role in pro-environmental behavior. *Frontiers in Psychology* 10, 718. https://doi.org/10.3389/fpsyg.2019.00718. Accessed 12 January 2021.

Geraeds, M., van Emmerik, T., de Vries, R. and Ab Razak, M.S. (2019). Riverine plastic litter monitoring using unmanned aerial vehicles (UAVs). *Remote Sensing* 11,2045. https://doi.org/10.3390/rs11172045. Accessed 12 January 2021.

Gerdts, G. (2017). Defining the BASElines and standards for microplastics ANalyses in European waters (BASEMAN.). 2nd JPI Oceans Conference, 26 October 2017, Lisbon. https://repository.oceanbestpractices.org/handle/11329/1205. Accessed 11 January 2021.

GESAMP (Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection) (2015). *Sources, Fate and Effects of Microplastics in the Marine Environment: A Global Assessment*. Kershaw, P.J. (ed.). IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP. https://ec.europa.eu/environment/marine/good-environmental-status/descriptor-10/pdf/GESAMP_microplastics%20full%20study.pdf. Accessed 11 January 2021.

GESAMP (2019). *Guidelines for the Monitoring and Assessment of Plastics Litter in the Ocean*. IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP. http://www.gesamp.org/publications/guidelines-for-the-monitoring-and-assessment-of-plastic-litter-in-the-ocean. Accessed 11 January 2021.

GESAMP (2020a). *Proceedings of the GESAMP International Workshop on Assessing the Risks Associated with Plastics and Microplastics in the Marine Environment*. Kershaw, P.J., Carney Almroth, B., Villarrubia- Gómez, P., Koelmans, A.A. and Gouin, T. (eds.). IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/ UNEP/UNDP/ISA.http://www.gesamp.org/publications/gesamp-international-workshop-on-assessing-the-risks-associated-with-plastics-and-microplastics-in-the-marine-environment. Accessed 11 January 2021.

GESAMP (2020b). *Sea-based Sources of Marine Litter – A Review of Current Knowledge and Assessment of Data Gaps. Second Interim Report of GESAMO Working Group 43. June 2020*. Rome: Food and Agriculture Organization of the United Nations. http://www.fao.org/3/cb0724en/cb0724en.pdf. Accessed 11 January 2021.

Gewert, B., Plassmann, M.M. and Macleod, M. (2015). Pathways for degradation of plastic polymers floating in the marine environment. *Environmental Science: Processes and* Impacts 17, 1513-1521. https://doi.org/10.1039/c5em00207a. Accessed 11 January 2021.

Geyer, R. (2020). Production, use and fate of synthetic polymers in plastic waste and recycling. In *Plastic Waste and Recycling: Environmental Impact, Societal Issues, Prevention, and Solutions*. Letcher, T.M. (ed.). Cambridge, MA: Academic Press.13-32. https://www.sciencedirect.com/science/article/pii/B9780128178805000025?via%3Dihub. Accessed 11 January 2021.

Geyer, R., Kuczenski, B., Zink, T. and Henderson, A. (2016). Common misconceptions about recycling. *Journal of Industrial Ecology* 20(5), 1010-1017. https://doi.org/10.1111/jiec.12355 . Accessed 11 January 2021.

Geyer, R., Jambeck, J.R. and Law, K.L. (2017). Production, use and fate of all plastics ever made. *Science Advances* 3(7), e1700782. https://doi.org/10.1126/sciadv.1700782. Accessed 12 January 2021.

Giampietro, M. and Mayumi, K. (2018). Unravelling the complexity of the Jevons' Paradox: The link between innovation, efficiency and sustainability. *Frontiers in Energy Research* 6, 26. https://doi.org/10.3389/fenrg.2018.00026. Accessed 12 January 2021.

Gigault, J., Pedrono, B., Maxit, B. and ter Halle, A. (2016). Marine plastic litter: The unanalyzed nano-fraction. *Environmental Science: Nano* 3, 346-350. https://doi.org/10.1039/C6EN00008H. Accessed 12 January 2021.

Gigault, J., ter Halle, A., Baudrimont, M., Pascal, P.-Y., Gauffre, F., Phi, T-L. *et al.* (2018). Current opinion: What is a nanoplastic? *Environmental Pollution* 235, 1030-1034. https://doi.org/10.1016/j.envpol.2018.01.024. Accessed 12 January 2021.

Gilbreath, A., McKee, L., Shimabuku, I., Lin, D., Werbowski, L.M., Zhu, X. *et al.* (2019). Multiyear water quality performance and mass accumulation of PCBs, mercury, methylmercury, copper and microplastics in a bioretention rain garden. *Journal of Sustainable Water in the Built Environment* 5(4), 04019004. https://doi.org/10.1061/JSWBAY.0000883. Accessed 12 January 2021.

Gleason, C.J. and Durand, M.T. (2020). Remote sensing of river discharge: A review and a framing for the discipline. *Remote Sensing* 12, 1107. https://doi.org/10.3390/rs12071107. Accessed 12 January 2021.

Goddijn-Murphy, L., Steef, L., van Sebille, E., James, N.A. and Gibb, S. (2018). Concept for a hyperspectral remote sensing algorithm for floating marine macro plastics. *Marine Pollution Bulletin* 126, 255-262. https://doi.org/10.1016/j.marpolbul.2017.11.011. Accessed 12 January 2021.

Godoy, D. and Stockin, K. (2018). Anthropogenic impacts on green turtles *Chelonia mydas* in New Zealand. *Endangered Species Research* 37, 1-9. https://doi.org/10.3354/esr00908. Accessed 12 January 2021.

Goel, N., Fatima, S.W., Kumar, S., Sinha, R. and Khare, S.K. (2021). Antimicrobial resistance in biofilms: exploring marine actinobacteira as a potential source of antibiotics and biofilm inhibitors. *Biotechnology Reports,* 30, e00613 https://doi.org/10.1016/j.btre.2021.e00613. Accessed 8 June 2021.

Golden, C.D., Allison, E.H., Cheung, W.W.L., Dey, M.M., Halpern. B.S., McCauley, D.J. *et al.* (2016). Fall in fish catch threatens human health. *Nature* 534, 317-320. https://doi.org/10.1038/534317a. Accessed 12 January 2021.

Goldstein, C.M., Rosenberg, M. and Cheng, L. (2012). Increased oceanic microplastics debris enhances oviposition in an endemic pelagic insect. *Biology Letters* 8, 817-820. https://doi.org/10.1098/rsbl.2012.0298. Accessed 12 January 2021.

Goldstein, M.C., Titmus, A. and Ford, M. (2013). Scales of spatial heterogeneity of plastic marine debris in the Northeast Pacific Ocean. *PLoS ONE* 8(11), e80020. https://doi.org/10.1371/journal.pone.0080020. Accessed 12 January 2021.

Goldstein, M.C., Carson, H.S. and Eriksen, M. (2014). Relationship of diversity and habitat area in North Pacific plastic-associated rafting communities. *Marine Biology* 161, 1441-1453. https://doi.org/10.1007/s00227-014-2432-8. Accessed 12 January 2021.

González, D., Hanke, G., Tweehuysen, G., Bellert, B., Holzhauer, M., Palatinus, A. *et al.* (2016). *Riverine Litter Monitoring – Options and Recommendations.* MSFD GES TG Marine Litter Thematic Report. JRC Technical Reports. EUR 28307. https://doi.org/10.2788/461233. Accessed 12 January 2021.

González-Fernández, D. and Hanke, G. (2017). Toward a harmonized approach for monitoring of riverine floating macro litter inputs to the marine environment. *Frontiers in Marine Science* 4, 86. https://doi.org/10.3389/fmars.2017.00086. Accessed 12 January 2021.

González-Fernández, D., Cózar, A., Hanke, G., Viejo, J. et al. (2021). Floating macrolitter leaked from Europe into the ocean. *Nature Sustainability* 4, 474–483. https://doi.org/10.1038/s41893-021-00722-6 Accessed 13 October 2021.

González-Pleiter, M., Tamayo-Belda, M., Pulido-Reyes, G., Amariei, G., Leganés, F., Rosal, R. and Fernández-Piñas, F. (2019). Secondary nanoplastics released from a biodegradable microplastic severely impact freshwater environments. *Environmental Science: Nano* 6, 1382-1392. https://doi.org/10.1039/C8EN01427B. Accessed 12 January 2021.

Gouin, T., Roche, N., Lohmann, R. and Hodges, G. (2011). A thermodynamic approach for assessing the environmental exposure of chemicals absorbed to microplastic. *Environmental Science and Technology* 45(4), 1466-1472. https://doi.org/10.1021/es1032025. Accessed 12 January 2021.

GPML (Global Partnership for Marine Litter) (2021). GPML Digital Platform. https://digital.gpmarinelitter.org/. Accessed 13 July 2021.

Green, D.S. (2016). Effects of microplastics on European flat oysters, *Ostrea edulis* and their associated benthic communities. *Environmental Pollution* 216, 95-103. https://doi.org/10.1016/j.envpol.2016.05.043. Accessed 12 January 2021.

Green, D.S., Boots, B., Blockley, D.J., Rocha, C. and Thompson, R. (2015). Impacts of discarded plastic bags on marine assemblages and ecosystem functioning. *Environmental Science and Technology* 49(9), 5380-5389. https://doi.org/10.1021/acs.est.5b00277. Accessed 12 January 2021.

Green, D.S., Boots, B., Sigwart, J., Jiang, S. and Rocha, C. (2016). Effects of conventional and biodegradable microplastics on a marine ecosystem engineer (*Arenicola Marinamarina*) and sediment nutrient cycling. *Environmental Pollution* 208, 426-434. https://doi.org/10.1016/j.envpol.2015.10.010. Accessed 12 January 2021.

Green, D.S., Boots, B., O'Connor, N.E. and Thompson, R. (2017). Microplastics affect the ecological functioning of an important biogenic habitat. *Environmental Science and Technology* 51(1), 68-77. https://doi.org/10.1021/acs.est.6b04496. Accessed 12 January 2021.

Green, D., Kregting, L. and Boots, B. (2018). A comparison of sampling methods for seawater microplastics and a first report of the microplastics litter in coastal waters of Ascension and Falkland Islands. *Marine Pollution Bulletin* 137, 695-701. https://doi.org/10.1016/j.marpolbul.2018.11.004. Accessed 12 January 2021.

Green, D.S., Colgan, T.J., Thompson, R.C. and Carolan, J.C. (2019). Exposure to microplastics reduces attachment strength and alters the haemolymph proteome of blue mussels (*Mytilus edulis*). *Environmental Pollution* 246, 423-434. https://doi.org/10.1016/j.envpol.2018.12.017. Accessed 12 January 2021.

Greenpeace (2020). *Biodegradable platsics: Breaking Down the Facts. Production,Composition and Environmental Impact.* Greenpeace East Asia. https://www.greenpeace.org/static/planet4-eastasia-stateless/84075f56-biodegradable-plastics-report.pdf. Accessed 20 January 2021.

Groh, K.J., Backhaus, T., Carney-Almroth, B., Gueke, B., Inostroza, P.A., Lennquist, A. *et al.* (2019). Overview of known plastic packaging-associated chemicals and their hazards. *Science of The Total Environment* 651, 3253-3268. https://doi.org/10.1016/j.scitotenv.2018.10.015. Accessed 12 January 2021.

Guerranti, C., Cannas, S., Scopetani, C, Fastelli, P., Cincimelli, A. and Renzi., M. (2017). Plastic litter in aquatic environments of Maremma Regional Park (Tyrrhenian Sea, Italy): Contribution by the Ombrone river and levels in marine sediments. *Marine Pollution Bulletin* 117, 366-370. https://doi.org/10.1016/j.marpolbul.2017.02.021. Accessed 12 January 2021.

Gulzar, T., Farooq, T., Kiran, S., Ahmad, I. and Hameed, A. (2019). Green chemistry in the wet processing of textiles. In *The Impact and Prospects of Green Chemistry for Textile Technology*. Shahid-ul-Islam and Butola, B.S. (eds.). Woodhead (Elsevier). 1-20. https://doi.org/10.1016/C2017-0-01957-2. Accessed 12 January 2021.

Guo, J.J., Zhang, F., Liu, C.H., Li, Y, and Zheng, R. E. (2017). Raman-fluorescence spectroscopy for underwater in-situ application. *Spectroscopy and Spectral Analysis* 37, 3099-3102. https://www.researchgate.net/publication/323666800_Raman-Fluorescence_Spectroscopy_for_Underwater_in-situ_Application. Accessed 17 August 2021.

Guo, X. and Wang, J. (2019). The chemical behaviours of microplastics in marine environment: A review. *Marine Pollution Bulletin* 142, 1-14. https://doi.org/10.1016/j.marpolbul.2019.03.019. Accessed 12 January 2021.

Guven, O., Gökdağ, K., Jovanović, B. and Kideys, A.E. (2017). Microplastic litter composition of the Turkish territorial waters of the Mediterranean Sea, and its occurrence in the gastrointestinal tract of fish. *Environmental Pollution* 223, 286-294. https://doi.org/10.1016/j.envpol.2017.01.025. Accessed 12 January 2021.

Haider, T.P, Völker, C., Kramm, J., Landfester, K. and Wurm, F.R. (2018). Plastics of the future? The impact of biodegradable polymers on the environment and on society. *Angewandte Chemie International Edition* 58, 50-62. https://doi.org/10.1002/anie.201805766. Accessed 12 January 2021.

Hahladakis, J.N., Velis, C.A., Weber, R., Iacovidou, E. and Purnella. P. (2018). An overview of chemical additives present in plastics: Migration, release, fate and environmental impact during their use, disposal and recycling. *Journal of Hazardous Materials* 344, 179-199. https://doi.org/10.1016/j.jhazmat.2017.10.014. Accessed 12 January 2021.

Hall, K. (2000). *Impacts of Marine Debris and Oil: Economic and Social Costs to Coastal Communities*. Lerwick, Shetland, United Kingdom: Kommunenes Internasjonale Miljøorganisasjon (KIMO). https://www.kimointernational.org/wp/wp-content/uploads/2017/09/KIMO_Impacts-of-Marine-Debris-and-Oil_Karen_Hall_2000.pdf. Accessed 12 January 2021.

Hall-Stoodley, L., Costerton, J.W. and Stoodley, P. (2004). Bacterial biofilms: From the Natural environment to infectious diseases. *Nature Reviews Microbiology* 2, 95-108. https://doi.org/10.1038/nrmicro821. Accessed 12 January 2021.

Hallanger, I.G. and Gabrielsen, G.W. (2018). *Plastics in the European Arctic*. Brief Report No. 045, Norwegian Polar Institute. http://www.synturf.org/images/NPI_Report_-_Kortrapport45.pdf. Accessed 12 January 2021.

Hämer, J., Gutow, L., Köhler, A., Saborowski, R., Hämer, J., Gutow, L. *et al.* (2014). Fate of microplastics in the marine isopod *Idotea emarginata*. *Environmental Science and Technology* 48(22), 13451-13458. https://doi.org/10.1021/es501385y. Accessed 12 January 2021.

Hamilton, S.E. and Casey, D. (2016). Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21). *Global Ecology and Biogeography* 25(6), 729-738. https://doi.org/10.1111/geb.12449. Accessed 12 January 2021.

Hanke, G., Walvoort, D., Van Loon, W., Addamo, A.M., Brosich, A., del Mar Chaves Montero, M. *et al.* (2019). *EU Marine Beach Litter Baselines: Analysis of a Pan-European 2012-2016 Beach Litter Dataset*. EUR 30022. Luxembourg: Publications Office of the European Union. https://doi.org/10.2760/16903. Accessed 12 January 2021.

Hansen, E., Nilsson, N.H., Lithner, D. and Lassen, C. (2013). *Hazardous Substances in Plastic Materials*. COWI and the Danish Technological Institute on behalf of the Norwegian Climate and Pollution Agency, Oslo. https://www.byggemiljo.no/wp-content/uploads/2014/10/72_ta3017.pdf. Accessed 12 January 2021.

Hanvey, J.S., Lewis, P.J., Lavers, J.L., Crosbie, N.D., Pozo, K. and Clarke, B.O. (2017). A review of analytical methods for quantifying microplastics in sediments. *Analytical Methods* 9, 1369-1383. https://pubs.rsc.org/en/content/articlelanding/2017/ay/c6ay02707e. Accessed 12 January 2021.

Hantoro, I., Löhr, A.J., Van Belleghem, F.G.A.J., Widianarko, B. and Ragas, A.M.J. (2019). Microplastics in coastal areas and seafood: implications for food safety. *Food Additives and Contaminants: Part A*,5, 36: 674-711. https://doi.org/10.1080/19440049.2019.1585581. Accessed 9 June 2021.

Hardesty, B.D. and Wilcox, C. (2017). A risk framework for tackling marine debris. *Analytical Methods* 9, 1429. https://pubs.rsc.org/en/content/articlepdf/2017/ay/c6ay02934e. Accessed 20 June 2021.

Hardesty, B.D., Holdsworth, D., Revill, A.T. and Wilcox, C. (2015). A biochemical approach for identifying plastics exposure in live wildlife. *Methods in Ecology and Evolution* 6, 92-98. https://doi.org/10.1111/2041-210X.12277. Accessed 12 February 2021.

Harris, P.T. (2020). The fate of microplastic in marine sedimentary environments: A review and synthesis. *Marine Pollution Bulletin* 158, 111398. https://doi.org/10.1016/j.marpolbul.2020.111398. Accessed 12 February 2021.

Harris, P.T., Tamelander, J., Lyons, Y., Neo. M.L. and Maes, T. (2021). Taking a mass-balance approach to assess marine plastics in the South China Sea. *Marine Pollution Bulletin* 171, 112 708.
https://doi.org/10.1016/j.marpolbul.2021.112708. Accessed 20 June 2021.
Harrison, J.P., Boardman, C., O'Callaghan, K., Delort, A.M. and Song, J. (2018). Biodegradability standards for carrier bags and plastics films in aquatic environments: A critical review. *Royal Society Open Science* 5, 171792. https://doi.org/10.1098/rsos.171792. Accessed 12 January 2021.

Hartley, B.L., Thompson, R.C. and Pahl, S. (2015). Marine litter education boosts children's understanding and self-reported actions. *Marine Pollution Bulletin* 90(1-2), 209-217. https://doi.org/10.1016/j.marpolbul.2014.10.049. Accessed 12 January 2021.

Hartley, B.L., Pahl, S., Veiga, J., Vlachogianni, T., Vasconcelos, L., Maes, T. *et al.* (2018a). Exploring public views on marine litter in Europe: Perceived causes, consequences and pathways to change. *Marine Pollution Bulletin* 133, 945-955. https://doi.org/10.1016/j.marpolbul.2018.05.061. Accessed 12 January 2021.

Hartley, B.L., Pahl, S., Holland, M., Alampei, I., Veiga, J. and Thompson, R.C. (2018b). Turning the tide on trash: Empowering European educators and school students to tackle marine litter. *Marine Policy* 96, 227-234. https://doi.org/10.1016/j.marpol.2018.02.002. Accessed 12 January 2021.

Haward, M. (2018). Plastic pollution of the world's seas and oceans as a contemporary challenge in ocean governance. *Nature Communications* 9, 667. http://doi.org/10.1038/s41467-018-03104-3. Accessed 12 January 2021.

He, P., Chen, L., Shao, L., Zhang, H. and Lu, F. (2019). Municipal solid waste (MSW) landfill: A source of microplastic? – Evidence of microplastics in landfill leachate. *Water Research* 159, 38-45. https://doi.org/10.1016/j.watres.2019.04.060. Accessed 12 January 2021.

Heidbreder, L.M., Bablok, I., Drews, S. and Menzel, C. (2019) Tackling the plastic problem: A review on perceptions, behaviours and interventions. *Science of The Total Environment* 668, 1077-1093. https://doi.org/10.1016/j.scitotenv.2019.02.437. Accessed 12 January 2021.

HELCOM (2017). *Measuring Progress for the Same Targets in the Baltic Sea.* The Baltic Marine Environment Protection Commission. http://www.helcom.fi/Lists/Publications/BSEP150.pdf. Accessed 12 January 2021.

HELCOM (2018). *HELCOM Guidelines for Monitoring Beach Litter*. The Baltic Marine Environment Protection Commission. https://helcom.fi/wp-content/uploads/2021/03/HELCOM-guidelines-for-monitoring-beach-litter.pdf. Accessed 12 January 2021.

Henry, B., Laitala, K. and Grimstad Klepp, I. (2019). Microfibres from apparel and home textiles: Prospects for including microplastics in environmental sustainability assessment. *Science of The Total Environment* 652, 483-494. https://doi.org/10.1016/j.scitotenv.2018.10.166. Accessed 12 January 2021.

Hermabessiere, L., Dehaut, A., Paul-Pont, I., Lacroix, C., Jezequel, R., Soudant, P. *et al.* (2017). Occurrence and effects of plastic additives on marine environments and organisms: A review. *Chemosphere* 182, 781-793. https://doi.org/10.1016/j.chemosphere.2017.05.096. Accessed 12 January 2021.

Hermsen, E., Mintenig, S.M., Besseling, E. and Koelmans, A.A. (2018). Quality criteria for the analysis of microplastic in biota samples: A critical review. *Environmental Science and Technology* 52(18), 10230-10240. https://doi.org/10.1021/acs.est.8b01611. Accessed 12 January 2021.

Herr, D. and Landis, E. (2016). *Coastal Blue Carbon Ecosystems*. Opportunities for Nationally Determined Contributions. *Policy Brief*. International Union for Conservation of Nature and the Nature Conservancy. https://portals.iucn.org/library/sites/library/files/documents/Rep-2016-026-En.pdf. Accessed 12 October 2021.

Herzke, D., Anker-Nilssen, T., Nøst, T.H., Götsch, A., Christensen-Dalsgaard, S., Langset, M. *et al.* (2016). Negligible impact of ingested microplastics on tissue concentrations of persistent organic pollutants in northern fulmars off coastal Norway. *Environmental Science and Technology* 50(4), 1924-1933. http://dx.doi.org/10.1021/acs.est.5b04663. Accessed 12 January 2021.

Heskett, M., Takada, H., Yamashita, R., Yuyama, M., Ito, M., Geok, Y.B. *et al.* (2012). Measurement of persistent organic pollutants (POPs) in plastic resin pellets from remote islands: Toward establishment of background concentrations for International Pellet Watch. *Marine Pollution Bulletin* 6, 445-8. https://doi.org/10.1016/j.marpolbul.2011.11.004. Accessed 12 January 2021.

Hidalgo-Ruiz, V. and Thiel, M. (2015). The contribution of citizen scientists to the monitoring of marine litter. In *Marine Anthropogenic Litter*. Bergmann, M., Gutow, L. and Klages, E. (eds.). Cham: Springer. 429-447. https://www.springer.com/gp/book/9783319165097. Accessed 12 January 2021.

Hinata, H., Moria, K., Ohnoa, K., Miyaoa, Y. and Kataoka, T. (2017). An estimation of the average residence times and onshore-offshore diffusivities of beached microplastics based on the population decay of tagged meso- and macrolitter. *Marine Pollution Bulletin* 122(102), 17-26. https://doi.org/10.1016/j.marpolbul.2017.05.012. Accessed 12 January 2021.

Hirai, H., Takada, H., Ogata, Y., Yamashita, R., Mizukawa, K., Saha. M. *et al.* (2011). Organic micropollutants in marine plastic debris from the open ocean and remote and urban beaches. *Marine Pollution Bulletin* 62, 1683-1692. https://doi.org/10.1016/j.marpolbul.2011.06.004. Accessed 12 January 2021.

Hodgson, D.J., Bréchon, A.L. and Thompson, R.C. (2018). Ingestion and fragmentation of plastic carrier bags by the amphipod *Orchetis gammaerellus*: Effects of plastic type and fouling load. Ma*rine Pollution Bulletin* 127, 154-159. https://doi.org/10.1016/j.marpolbul.2017.11.057. Accessed 12 January 2021.

Hoellein, T., Rojas, M., Pink, A., Gasior, J. and Kelly, J. (2014). Anthropogenic litter in urban freshwater ecosystems: Distribution and microbial interactions. *PLoS One* 9 (6), e98485. https://doi.org/10.1371/journal.pone.0098485. Accessed 12 January 2021.

Hoffman, M.J. and Hittinger, E. (2017). Inventory and transport of plastic debris in the Laurentian Great Lakes. *Marine Pollution Bulletin* 115, 273-281. https://doi.org/10.1016/j.marpolbul.2016.11.061. Accessed 12 January 2021.

Holland, E.R., Mallory, M.L. and Shutler, D. (2016). Plastics and other anthropogenic debris in freshwater birds from Canada. *Science of The Total Environment* 571, 251-258. https://doi.org/10.1016/j.scitotenv.2016.07.158. Accessed 12 January 2021.

Holmquist, H., Schellenberger, S., van Der Veen, I., Peters, G., Leonards, P. and Cousins, I.T. (2016). Properties, performance and associated hazards of state-of-the-art durable water repellent (DWR) chemistry for textile finishing. Environment International 91, 251-264. https://doi.org/10.1016/j.envint.2016.02 035. Accessed 12 January 2021.

Hong, S., Lee, J., Jang, Y.C., Kim, Y.J., Kim, H.J., Han, D. *et al.* (2013). Impacts of marine debris on wild animals in the coastal area of Korea. *Marine Pollution Bulletin* 66(1-2), 117-124. https://doi.org/10.1016/j.marpolbul.2012.10.022. Accessed 12 January 2021.

Hong, S.H., Shim, W.J. and Hong, L. (2017a). Methods of analysing chemicals associated with microplastics: A review. *Analytical Methods* 9, 1361. https://doi.org/10.1039/c6ay02971j. Accessed 12 January 2021

Hong, S., Lee, J. and Lim, S. (2017b). Navigational threats by derelict fishing gear to navy ships in the Korean Seas. *Marine Pollution Bulletin* 119(2), 100-105. https://doi.org/10.1016/j.marpolbul.2017.04.006. Accessed 12 January 2021.

Hoornweg, D. and Bhada-Tata, P. (2012). *What a Waste: A Global Review of Solid Waste Management.*Urban Development Series. World Bank Urban Development and Local Government Unit of the Sustainable Development Network. Washington, D.C.: World Bank. https://openknowledge.worldbank.org/handle/10986/17388. Accessed 12 January 2021

Horodytska, O., Cabanes, A. and Fullana, A. (2020). Non-intentionally added substances (NIAS) in recycled plastics. *Chemosphere* 251, 126373. https://doi.org/10.1016/j.chemosphere.2020.126373. Accessed 12 January 2021.

Horstman, E.M., Dohmen-Janssen, C.M., Narra, P.M.F., van den Berg, N.J.F., Siemerink, M. and Hulscher, S.J.M.H. (2014). Wave attenuation in mangroves: A quantitative approach to field observations. *Coastal Engineering* 94, 47e62. https://doi.org/10.1016/j.coastaleng.2014.08.005. Accessed 12 January 2021.

Horton, A.A. and Dixon, S.J. (2018). Microplastics: An introduction to environmental transport processes. *WIREs Water* 5(2), e1268. https://doi.org/10.1002/wat2.1268. Accessed 12 January 2021.

Horton, A.A., Svendsen, C., Williams, R.J., Spurgeon, D.J. and Lahive, E. (2017a). Large microplastics particles in sediments of tributaries of the River Thames, UK – Abundance, sources and methods for effective quantification. *Marine Pollution Bulletin* 114(1), 218-226. https://doi.org/10.1016/j.marpolbul.2016.09.004. Accessed 12 January 2021.

Horton, A.A., Walton, A., Spurgeon, D.J., Lahive, E. and Svendsen, C. (2017b). Microplastics in freshwater and terrestrial environments: Evaluating the current understanding to identify the knowledge gaps and future research priorities. *Science of The Total Environment* 586, 127-141. https://doi.org/10.1016/j.scitotenv.2017.01.190. Accessed 12 January 2021.

Houde, M., De Silva, A.O., Muir, D.C.G and Letcher, R.J. (2011). Monitoring of perfluorinated compounds in aquatic biota: An updated review. *Environmental Science and Technology* 45(19), 7962- 7973. https://doi.org/10.1021/es104326w. Accessed 12 January 2021.

Huang, F.Y., Yang, K., Zhang, Z.X., Su, J.Q., Zhu, Y.G. and Zhang, X. (2019). Effects of microplastics on antibiotic resistance genes in estuarine sediments. PMID 40(5), 2234-2239 [in Chinese]. https://doi.org/10.13227/j.hjkx.201810108; English abstract at https://pubmed.ncbi.nlm.nih.gov/31087861/. Accessed 12 January 2021.

Hüffer, T., Praetorius, A., Wagner, S., der Kammer, F.V. and Hofmann, T. (2017). Microplastic exposure assessment in aquatic environments: Learning from similarities and differences to engineered nanoparticles. *Environmental Science and Technology* 51(5), 2499-2407. https://doi.org/10.1021/acs.est.6b04054. Accessed 12 January 2021.

Hurley, R.R. and Nizzetto, L. (2018). Fate and occurrence of micro(nano) plastics in soils: Knowledge gaps and possible risks. *Current Opinion in Environmental Science and Health* 1, 6-11. https://doi.org/10.1016/j.coesh.2017.10 006. Accessed 12 January 2021

Hurley, R., Woodward, J. and Rothwell, J.J. (2018). Microplastics contamination of river beds significantly reduced by catchment-wide flooding. *Nature Geoscience* 11(4), 251-257. https://doi.org/10.1038/s41561-018-0080-1. Accessed 12 January 2021.

ICIS (Independent Commodity Intelligence Services) (2020). Post corona virus, what will change? https://icis.com/explore/resources/news/2020/04/30/10502603/post-corona-what-will-change. Accessed 13 July 2021.

IHS Markit (2018). Bisphenol A: Chemical Economics Handbook. https://ihsmarkit.com/products/bisphenol-chemical-economics-handbook.html. Accessed 20 June 2021.

ILO (International Labour Organization) (2017). *Cooperation among Workers in the Informal Economy: A Focus on Home-based Workers and Waste Pickers. A Joint ILO and WIEGO Initiative*. Geneva. https://www.ilo.org/global/topics/cooperatives/publications/WCMS_567507/lang--en/index.htm. Accessed 12 January 2021.

ILO (2019). *Waste Pickers' Cooperatives and Social and Solidarity Economy Organizations.* Cooperatives and the World of Work Series No. 12. Geneva. https://www.ilo.org/wcmsp5/groups/public/---ed_emp/---emp_ent/---coop/documents/publication/wcms_715845.pdf. Accessed 12 January 2021

IMarEST (Institute of Marine Engineering Science and Technology) (2019). *Steering towards an Industry Level Response to Marine Plastic Pollution: Roundtable Summary Report*. London. https://www.imarest.org/policy-news/thought-leadership/1039-marine-plastics/file. Accessed 12 January 2021.

Imhof, H.K., Ivleva, N.P., Schmid, J., Niessner, R. and Laforsch, C. (2013). Contamination of beach sediments of a subalpine lake with microplastic particles. *Current Biology* 23(19), R867-R868. https://doi.org/10.1016/j.cub.2013.09.001. Accessed 12 January 2021.

Imhof, H.K., Sigl, R., Brauer, E., Feyl, S., Giesemann, P., Klink, S. *et al.* (2017). Spatial and temporal variation of macro-, meso- and microplastic abundance on a remote coral island of the Maldives, Indian Ocean. *Marine Pollution Bulletin* 116, 340-347. https://doi.org/10.1016/j.marpolbul.2017.01 010. Accessed 12 January 2021.

Imhof, H.K., Wiesheu, A.C., Anger, P.M., Niessner, R., Ivleva, N.P. and Laforsch, C. (2018). Variation in plastic abundance at different lake beach zones. A case study. *Science of The Total Environment* 613, 530-537. https://doi.org/10.1016/j.scitotenv.2017.08 300. Accessed 12 January 2021.

IMO (International Maritime Organization) (2016). *Review of the Current State of Knowledge Regarding Marine Litter in Wastes Dumped at Sea under the London Convention and Protocol - Final Report* (LC 38/16). London. https://wwwcdn.imo.org/localresources/en/OurWork/Environment/Documents/Marine%20litter%20review%20for%20publication%20April%202016_final_ebook_version.pdf. Accessed 12 January 2021.

IMO (2019). *End-of-Life Management of Fibre Reinforced Plastic Vessels: Alternatives to At Sea Disposal*. London. https://wwwcdn.imo.org/localresources/en/OurWork/Environment/Documents/Fibre%20Reinforced%20Plastics%20final%20report.pdf. Accessed 12 January 2021.

International Chamber of Shipping (2021). Shipping and world trade. https://www.ics-shipping.org/shipping-fact/shipping-and-world-trade-driving-prosperity/. Accessed 10 September 2021.

International Monetary Fund (2019). *Disposal is Not Free: Fiscal Instruments to Internalize the Environmental Costs of Solid Waste.* https://www.imf.org/en/Publications/WP/Issues/2019/12/20/Disposal-is-Not-Free-Fiscal-Instruments-to-Internalize-the-Environmental-Costs-of-Solid-Waste-48854. Accessed 25 May 2021.

International Pacific Research Center (2008). Tracking ocean debris. *IPRC Climate* 8(2), 14-16. http://iprc.soest.hawaii.edu/newsletters/newsletter_sections/iprc_climate_vol8_2/tracking_ocean_debris.pdf. Accessed 12 January 2021.

International Pollutants Elimination Network (2019). *Plastic Waste Poisons Indonesia's Food Chain.* https://ipen.org/documents/plastic-waste-poisons-indonesia-food-chain. Accessed 25 May 2021.

Interpol (2020). *Emerging Criminal Trends in the Global Waste Market since January 2018. Interpol Strategic Analysis Report*. https://www.interpol.int/en/News-and-Events/News/2020/INTERPOL-report-alerts-to-sharp-rise-in-plastic-waste-crime. Accessed 12 January 2021.

IRP (International Resource Panel) (2019). *Global Resources Outlook 2019: Natural Resources for the Future We Want.* Oberle, B., Bringezu, S., Hatfield-Dodds, S., Hellweg, S., Schandl, H., Clement, J., and Cabernard, L., Che, N., Chen, D., Droz-Georget *et al.* A Report of the International Resource Panel. Nairobi, Kenya: United Nations Environment Programme. https://www.resourcepanel.org/reports/global-resources-outlook. Accessed 15 June 2021.

IRP (2021). IRP (2021). *Policy Options to Eliminate Additional Marine Plastic Litter by 2050 Under the G20 Osaka Blue Ocean Vision.* Nairobi: UNEP. https://www.resourcepanel.org/sites/default/files/documents/document/media/policy_options_to_eliminate_additional_marine_plastic_litter.pdf

Isobe, A., Uchida, K., Tokai, T. and Iwasaki, S. (2015). East Asian seas: A hot spot of pelagic microplastics. *Marine Pollution Bulletin* 101, 618-623. https://doi.org/10.1016/j.marpolbul.2015.10.042. Accessed 12 January 2021.

Isobe, A., Uchiyama-Matsumoto, K., Uchida, K. and Tokai, T. (2017). Microplastics in the Southern Ocean. *Marine Pollution Bulletin* 114(1), 623-626. https://doi.org/10.1016/j.marpolbul.2016.09.037. Accessed 12 January 2021.

Isobe, A., Buenaventura, N.T. Chastain, S., Chavanich, S., Cózar, A., DeLorenzo *et al.* (2019). An interlaboratory comparison experiment to quantify the abundance of microplastics in standard sample bottles. *Marine Pollution Bulletin* 146, 831-837. https://doi.org/10.1016/j.marpolbul.2019.07.033. Accessed 12 January 2021.

Ivar do Sul, J.A. and Costa, M.F. (2014). The present and future of microplastic pollution in the marine environment. *Environmental Pollution* 185, 352-364. https://doi.org/10.1016/j.envpol.2013.10.036. Accessed 12 January 2021.

Ivar do Sul, J.A., Costa, M.F., Silva-Cavalcanti, J.S. and Araújo, M.C.B. (2014). Plastics debris retention and exportation by a mangrove forest patch. *Marine Pollution Bulletin* 78(1-2), 252-257. https://doi.org/10.1016/j.marpolbul.2013.11.011. Accessed 12 January 2021.

Ivleva, N.P., Wiesheu, A. and Reinhard, N. (2018). Microplastic in aquatic ecosystems. *Angewandte Chemie International Edition* 56, 1720-1739. https://doi.org/10.1002/anie.201606957. Accessed 12 January 2021.

Jacob, H., Besson, M., Swarzenski, P.W., Lecchini, D. and Metian, M. (2020). Effects of virgin micro- and nanoplastics on fish: Trends, meta-analysis, and perspectives. *Environmental Science and Technology* 54(8), 4733-4745. https://dx.doi.org/10.1021/acs.est.9b05995. Accessed 12 January 2021.

Jacquin, J., Cheng, J., Odobel, C., Pandin, C., Conan, P., Pujo-Pay, M. *et al.* (2019). Microbial ecotoxicology of marine plastic debris: A review on colonization and biodegradation by the "plastisphere". *Frontiers in Microbiology,* 25 April. https://doi.org/10.3389/fmicb.2019.00865. Accessed 12 January 2021.

Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A. *et al.* (2015). Plastic waste inputs from land into the ocean. *Science* 347(6223), 768-771.
https://doi.org/10.1126/science.1260352. Accessed 12 January 2021.

Jambeck, J., Hardesty, B.D., Brooks, A.L., Friend, T., Teleki, K., Fabres, J. *et al.* (2018). Challenges and emerging solutions to the land-based plastic waste issue in Africa. *Marine Policy* 96, 256-263. https://doi.org/10.1016/j.marpol.2017.10.041. Accessed 12 January 2021.

Jang, M., Shim, W.J., Han, G.M., Rani, M., Song, Y.K. and Hong, S.H. (2016). Styrofoam debris as a source of hazardous additives for marine organisms. *Environmental Science and Technology* 50(10), 4951-4960. https://doi.org/10.1021/acs.est.5b05485. Accessed 12 January 2021.

Jang, M., Shim, W.J., Han, G.M., Song, Y.K. and Hong, S.H. (2018). Formation of microplastics by polychaetes (*Marphysa sanguinea*) inhabiting expanded polystyrene marine debris. *Marine Pollution Bulletin* 131, 365-369. https://doi.org/10.1016/j.marpolbul.2018.04.017. Accessed 12 January 2021.

Jang, Y.C., Hong, S., Lee, J., Lee, M.J. and Shim, W.J. (2014). Estimation of lost tourism revenue in Geoje island from the 2011 marine debris pollution event in South Korea. *Marine Pollution Bulletin* 81, 49-54. https://doi.org/10.1016/j.marpolbul.2014.02.021. Accessed 12 January 2021.

Jang, Y.C., Lee, J., Hong, S., Choi, H.W., Shim, W.J. and Hong, S.Y. (2015). Estimating the global inflow and stock of plastic marine debris using material flow analysis. *Journal of the Korean Society for Marine Environment and Energy* 18, 263-273. https://doi.org/10.7846/JKOSMEE.2015.18.4.263. Accessed 12 January 2021.

Janssen, C., de Rycke, M. and van Cauwenberghe, L. (2014). *Marine Pollution along the East Africa Coast: Problems and Challenges. International Workshop – Sustainable Use of Coastal and Marine Resources in Kenya: From Research to Societal Benefits.* Laboratory of Environmental Toxicology and Aquatic Ecology, Environmental Toxicology Unit Lab (GhenToxLab), University of Ghent, Belgium. http://www.vliz.be/kenya/sites/vliz.be.kenya/files/public/KMFRIdocuments/Colin%20Janssen.pdf. Accessed 12 January 2021.

Japan Ministry of the Environment (2012). Estimated total amount of debris washed out by the Great East Japan Earthquake. https://www.env.go.jp/en/focus/docs/files/20120901-57.pdf. Accessed 12 January 2021.

Japan Ministry of the Environment (2014). *History and Current State of Waste Management in Japan*. https://www.env.go.jp/en/recycle/smcs/attach/hcswm.pdf. Accessed 12 January 2021.

Jeffrey, C.F., Havens, K.J., Slacum, H.W., Bilkovic, D.M., Zaveta, D., Scheld, A.M. *et al.* (2016*). Assessing Ecological and Economic Effects of Derelict Fishing Gear: A Guiding Framework.* Virginia Institute of Marine Science, William and Mary. http://doi.org/10.21220/V50W23. Accessed 12 January 2021.

Jobstvogt, N., Hanley, N., Hynes, S., Kenter, J. and Witte, U. (2014). Twenty thousand sterling under the sea: Estimating the value of protecting deep-sea biodiversity. *Ecological Economics* 97, 10-19. https://doi.org/10.1016/j.ecolecon.2013.10.019. Accessed 12 January 2021.

Joshi, C., Seay, J. and Banadda, N. (2019). A perspective on locally managed decentralized circular economy for water plastic in developing countries. *Environmental Programmes in Sustainable Energy* 38, 3-11. https://doi.org/10.1002/ep.13086. Accessed 12 January 2021.

Kaminski, A., Bell, K.P., Noblet, C.L. and Evans, K.S. (2017). An economic analysis of coastal beach safety information-seeking behavior. *Agricultural Resource Economics and Review* 46, 365-387. https://doi.org/10.1017/age.2017.17. Accessed 12 January 2021.

Kandziora, J.H., van Toulon, N., Sobralb, P., Taylor, H.L., Ribbink, A.J., Jambeck, J.R. *et al.* (2018). The important role of marine debris networks to prevent and reduce ocean plastic pollution. *Marine Pollution Bulletin* 141, 657-662. https://doi.org/10.1016/j.marpolbul.2019.01.034. Accessed 12 January 2021.

Kane, A.B., Hurt, R.H. and Gao, H. (2018). The asbestos-carbon nanotube analogy: An update. Toxicology *Applications and Pharmacy* 361, 68-80. https://doi.org/10.1016/j.taap.2018.06.027. Accessed 12 January 2021.

Kanhai, L.D.K., Officer, R., Lyashevska, O., Thompson, R.C. and O'Connor, I. (2017). Microplastic abundance, distribution and composition along a latitudinal gradient in the Atlantic Ocean. *Marine Pollution Bulletin* 115(1-2), 307-314. https://doi.org/10.1016/j.marpolbul.2016.12.025. Accessed 12 January 2021.

Kanhai, L.D.K., Gårdfeldt, K., Lyashevska, O., Hesselhöv, Thompson, R.C. and O'Conner, I. (2018). Microplastics in sub-surface waters of the Arctic Central Basin. *Marine Pollution Bulletin* 130, 8-18. https://doi.org/10.1016/j.marpolbul.2018.03.011. Accessed 12 January 2021.

Kanhai, L.D.K., Johansson, C., Frias, J.P.G.L., Gårdfeldt, K., Thompson, R.C. and O'Connor, I. (2019). Deep sea sediments of the Arctic Central Basin: A potential sink for microplastics. *Deep-Sea Research I: Oceanography Research Papers* 145, 137-142. https://doi.org/10.1016/j.dsr.2019.03.003. Accessed 12 January 2021.

Kanhai, L.D.K., Gardfeldt, K., Krumpen, T., Thompson, R.C and O'Connor, I. (2020). Microplastics in sea ice and seawater beneath ice floes from the Arctic Ocean. *Scientific Reports* 10, 5004. https://doi.org/10.1038/s41598-020-61948-6. Accessed 12 January 2021.

Käppler, A., Fischer, M., Scholz-Böttcher, B.M., Oberbeckmann, S., Labrenz, M., Fischer, D. *et al.* (2018). Comparison of μ-ATR-FTIR spectroscopy and py-GCMS as identification tools for microplastic particles and fibres isolated from river sediments. *Analytical and Bioanalytical Chemistry* 410, 5313- 5327. https://doi.org/10.1007/s00216-018-1185-5. Accessed 12 January 2021.

Karami, A., Golieskardi, A., Choo, C.K., Larat, V., Galloway, T.S. and Salamatinia, B. (2017). The presence of microplastics in commercial salts from different countries. *Scientific Reports* 7, 46173. https://doi.org/10.1038/srep46173. Accessed 12 January 2021.

Karasik, R., Vegh, T., Diana, Z., Bering, J., Caldas, J., Pickle, A., Rittschof, D. and Virdin, J. (2020*). 20 Years of Government Responses to the Global Plastic Pollution Problem*. Nicholas Institute for Environmental Policy Solutions, Duke University, Durham, North Carolina, United States. https://nicholasinstitute.duke.edu/publications/20-years-government-responses-global-plastic-pollution-problem. Accessed 12 January 2021.

Karlsson, T.M., Vethaaka, A.D., Carney Almroth, B., Ariesee, F., van Velzena, M., Hassellöv, M. *et al.* (2017). Screening for microplastics in sediment, water, marine invertebrates and fish: Method development and microplastic accumulation. *Marine Pollution Bulletin* 122(1-2), 403-408. https://doi.org/10.1016/j.marpolbul.2017.06.081. Accessed 12 January 2021.

Karlsson, T.M., Arneborg, L. Bronström, G., Carney Almroth, B., Gipperth, L. and Hassellöv, M. (2018). The unaccountability case of plastic pellet pollution. *Marine Pollution Bulletin* 129, 52-60. https://doi.org/10.1016/j.marpolbul.2018.01.041. Accessed 12 January 2021.

Karlsson, T.M., Kärman, A., Rotander, A. and Hassellöv, M. (2019). Comparison between manta trawl and in situ pump filtration methods, and guidance for visual identification of microplastics in surface waters. *Environmental Science and Pollution Research* 27, 5559-5571. https://doi.org/10.1007/s11356-019-07274-5. Accessed 12 January 2021.

Karn, S. and Jenkinson, I.R. (2019). Plastics and macroplastic: A major risk factor to the soil, water and marine environment. *Current Biotechnology* 8(1), 64-74. https://doi.org/10.2174/2211550108999190717091621. Accessed 12 January 2021.

Katija, K., Choy, C.A., Sherlock, R.E., Sherman, A.D. and Robison, B.H. (2017). From the surface to the seafloor: How giant larvaceans transport microplastics into the deep sea. *Science Advances* 3(8), e1700715. https://doi.org/10.1126/sciadv.1700715. Accessed 12 January 2021.

Katsanevakis, S., Verriopoulos, G., Nicolaidou, A. and Thessalou-Legaki, M. (2007). Effect of marine litter on the benthic megafauna of coastal soft bottoms: A manipulative field experiment. *Marine Pollution Bulletin* 54(6), 771-778. https://doi.org/10.1016/j.marpolbul.2006.12.016. Accessed 12 January 2021.

Kaza, S.L.C., Yao, P., Bhada-Tata, P. and Van Woerden, F. (2018). *What a Waste 2.0 : A Global Snapshot of Solid Waste Management to 2050*. Urban Development Series. Washington, D.C.: World BankGroup. https://openknowledge.worldbank.org/handle/10986/30317. Accessed 12 January 2021.

Kedzierski, M., d'Almeida, M., Magueresse, A., Le Grand, A., Duval, H., César, G. *et al.* (2018). Threat of plastic ageing in marine environments. Adsorption/desorption of micropollutants. *Marine Pollution Bulletin* 127, 684-694. https://doi.org/10.1016/j.marpolbul.2017.12.059. Accessed 12 January 2021.

Kern, D.G., Crausman, R.S. and Clapp, R.W. (2011). A retrospective cohort study of lung cancer incidence in nylon flock workers, 1998-2008. *International Journal of Occupational and Environmental Health* 17, 345-352. https://doi.org/10.1179/107735211799041814. Accessed 12 January 2021.

Kern, D.G., Kuhn, C. III., Ely, W., Pransky, G.S., Mello, C.J., Fraire, A.E. *et al.* (2000). Flock worker's lung: Broadening the spectrum of clinicopathology, narrowing the spectrum of suspected etiologies. *Chest* 117, 251-259. https://doi.org/10.1378/chest.117.1.251. Accessed 12 January 2021.

Kettner, M. T., Rojas-Jimenez, K., Oberbeckmann, S., Labrenz, M. and Grossart, H.-P. (2017). Microplastics alter composition of fungal communities in aquatic ecosystems. *Environmental Microbiology* 19(11), 4447-4459. https://doi.org/10.1111/1462-2920.13891. Accessed 12 January 2021.

Khan, F., Ahmed, W. and Najmi, A. (2019). Understanding consumers' behavior intentions towards dealing with the plastic waste: Perspective of a developing country. *Resources, Conservation and Recycling* 142, 49-58. https://doi.org/10.1016/j.resconrec.2018.11.020. Accessed 12 January 2021.

Kideys A.E., Halisdemir, B., Kideys, A., Gucu, A.C. Gazihan, A. *et al.* (2018). A University–Municipality partnership: Marine Environmental Awareness training (k12) in Turkey. Abstract published in the EMSEA (European Marine Science Educators Association) Conference Book, 2-5 October 2018, Newcastle-Upon-Tyne, UK, pp 3. https://conferences.ncl.ac.uk/emsea2018/. Accessed 15 February 2021.

Kiessling, T., Gutow, L. and Thiel, M. (2015). Marine litter as habitat and dispersal vector. In *Marine Anthropogenic Litter*. Bergmann, M., Gutow, L. and Klages, E. (eds.). Cham: Springer. 141-184. https://link.springer.com/chapter/10.1007/978-3-319-16510-3_6#citeas. Accessed 12 January 2021.

Kiessling, T., Salas, S., Mutafoglu, K. and Thiel, M. (2017). Who cares about dirty beaches? Evaluating environmental awareness and action on coastal litter in Chile. *Ocean and Coastal Management* 137, 82-95. https://doi.org/10.1016/j.ocecoaman.2016.11.029. Accessed 12 January 2021.

Kirstein, I.V., Kirmizi, S., Wichels, A., Garin-Fernandez, A., Erler, R., Martin, L. *et al.* (2016). Dangerous hitchhikers? Evidence for potentially pathogenic *Vibrio* spp. on microplastics particles. *Marine Environmental Research* 120, 1-8. https://doi.org/10.1016/j.marenvres.2016.07.004. Accessed 12 January 2021.

Klein, M. and Fischer, E.K. (2019). Microplastics abundance in atmospheric deposition within the Metropolitan area of Hamburg, Germany. *Science of the Total Environment* 685, 96-103. https://doi.org/10.1016/j.scitotenv.2019.05.405. Accessed 12 January 2021.

Klein, S., Worch, E. and Knepper, T.P. (2015). Occurrence and spatial distribution of microplastics in river shore sediments of the Rhine-Main area in Germany. *Environmental Science and Technology* 49(10), 6070-6076. https://doi.org/10.1021/acs.est.5b00492. Accessed 12 January 2021.

Koelmans, A.A., Besseling, E. and Foekema, E.L. (2014). Leaching of plastics additives to marine organisms. *Environmental Pollution* 187, 49-54. https://doi.org/10.1016/j.envpol.2013.12.013. Accessed 12 January 2021.

Koelmans, A.A., Bakir, A., Burton, G.A. and Janssen C.R. (2016). Microplastic as a vector for chemicals in the aquatic environment: Critical review and model-supported reinterpretation of empirical studies. *Environmental Science and Technology* 50(7), 3315-3326. https://doi.org/10.1021/acs.est.5b06069. Accessed 12 January 2021.

Koelmans, A.A., Besseling, E., Foekema, E., Kooi, M., Mintenig, S., Ossendorp, B.C. *et al.* (2017a). Risks of plastic debris: Unravelling fact, opinion, perception and belief. *Environmental Science and Technology* 51(20), 11513-11519. https://doi.org/10.1021/acs.est.7b02219. Accessed 12 January 2021.

Koelmans, A.A., Kooi, M. Law, K. and van Sebille, E. (2017b). All is not lost: Deriving a top-down mass budget of plastic at sea. *Environmental Research Letters* 12, 114028. https://iopscience.iop.org/article/10.1088/1748-9326/aa9500. Accessed 12 January 2021.

Koelmans, A.A., Mohamed Nor, N.H., Hermsen, E., Kooi, M., Mintenig, S.M. and De France, J. (2019). Microplastics in freshwaters and drinking water: Critical review and assessment of data quality. *Water Research* 155, 410-422. https://doi.org/10.1016/j.watres.2019.02.054. Accessed 12 January 2021.

Koelmans, A.A., Redondo-Hasselerharm, P.E., Nor, N.H.M. and Kooi, M. (2020). Solving the nonalignment of methods and approaches used in microplastic research to consistently characterize risk. *Environmental Science and Technology* 54 (19), 12307-12315. https://doi.org/10.1021/acs.est.0c02982. Accessed 12 January 2021.

Kögel T., Refosco A. and Maage A. (2020). Surveillance of seafood for microplastics. In *Handbook of Microplastics in the Environment*. Rocha-Santos, T., Costa, M. and Mouneyrac, C. (eds.). Cham: Springer. 1-34. https://doi.org/10.1007/978-3-030-10618-8_28-1. Accessed 12 January 2021.

Kooi, M. and Koelmans, A.A. (2019). Simplifying microplastic via continuous probability distributions for size, shape, and density. *Environmental Science and Technology Letters* 6(9), 551-557. https://doi.org/10.1021/acs.estlett.9b00379. Accessed 12 January 2021

Kooi, M., Reisser, J., Slat, B., Ferrari, F.F., Schmid, M.S., Cunsolo, S. *et al.* (2016). The effect of particle properties on the depth profile of buoyant plastics in the ocean. *Scientific Reports* 6, 33882. https://doi.org/10.1038/srep33882. Accessed 12 January 2021.

Krelling, A.P., Williams, A.T and Turra, A. (2017). Differences in perception and reaction of tourist groups to beach marine debris that can influence a loss of tourism revenue in coastal areas. *Marine Policy* 85, 87-99. https://doi.org/10.1016/j.marpol.2017.08.021. Accessed 12 January 2021.

Krijnen, J.M.T., Tannenbaum, D. and Fox, C.R. (2017). Choice architecture 2.0: Behavioral policy as an implicit social interaction. *Behavioral Science & Policy* 3(2), 1-18. https://behavioralscientist.org/choice-architecture-2-0-how-people-interpret-and-make-sense-of-nudges/. Accessed 25 May 2021.

Kroon, F.J., Berry, K.L.B., Brinkman, D.L. Kookana, R., Leusch, F.D.L., Melvin, S.D. *et al.* (2020). Sources, presence and potential effects of contaminants of emerging concern in the marine environments of the Great Barrier Reef and Torres Strait, Australia. *Science of The Total Environment* 719, 135140. https://doi.org/10.1016/j.scitotenv.2019.135140. Accessed 12 January 2021

Kühn, S., Bravo Rebolledo, E.L. and van Franeker, J.A. (2015). Deleterious effects of litter on marine life. In *Marine Anthropogenic Litter*. Bergmann, M., Gutow, L. and Klages, E. (eds.). Cham: Springer. 75-116. https://link.springer.com/chapter/10.1007/978-3-319-16510-3_4. Accessed 12 January 2021.

Kühn, S., Schaafsma, F.L., van Werven, B., Flores, H., Bergmann, M., Egelkraut-Holtus, M. *et al.* (2018). Plastic ingestion by juvenile polar cod (*Boreogadus saida*) in the Arctic Ocean. *Polar Biology* 41, 1269-1278. https://doi.org/10.1007/s00300-018-2283-8. Accessed 12 January 2021.

Kukulka, T., Proskurowski, G., Moret-Ferguson, S., Meyer, D. W. and Law, K.L. (2012). The effect of wind mixing on the vertical distribution of buoyant plastics debris. *Geophysical Research Letters* 39(7). https://doi.org/10.1029/2012GL051116. Accessed 13 January 2021.

Kukulka, T. and Brunner, K. (2015). Passive buoyant tracers in the ocean surface boundary layer: 1. Influence of equilibrium wind-waves on vertical distributions. *Journal of Geophysical Research Oceans* 120, 3837-3858. https://doi.org/10.1002/2014JC010487. Accessed 13 January 2021.

Kutralam-Muniasamy, G., Perez-Guevara, F., Elizalde-Martinez, I. and Shruti, V.C. (2020). Branded milks – are they immune from microplastics contamination? *Science of The Total Environment* 714, 136823. https://doi.org/10.1016/j.scitotenv.2020.136823
Accessed 13 January 2021.

Kylili, K., Kyriakides, I., Artusi, A. and Hadjistassou, C. (2019). Identifying floating plastics marine debris using a deep learning approach. *Environmental Science and Pollution Research* 26(17), 17091-17099. https://doi.org/10.1007/s11356-019-05148-4. Accessed 13 January 2021.

Lamb, J.B., Willis, B.L., Fiorenza, E.A., Couch, C.S., Howard, R., Rader, D.N. *et al.* (2018). Plastic waste associated with disease on coral reefs. *Science* 359(6374), 460-462. https://doi.org/10.1126/science.aar3320. Accessed 13 January 2021.

Lambert, B.S., Olson, R.J. and Sosik, H.M. (2017). A fluorescence-activated cell sorting subsystem for the imaging flowcytobot. *Limnology and Oceanography Methods* 15(1), 94-102. https://doi.org/10.1002/lom3.10145. Accessed 13 January 2021.

Lambert, S. and Wagner, M. (2016). Characterisation of nanoplastics during the degradation of polystyrene. *Chemosphere* 145, 265-268. https://doi.org/10.1016/j.chemosphere.2015.11.078. Accessed 13 January 2021.

Landon-Lane, M. (2018). Corporate social responsibility in marine plastic debris governance. *Marine Pollution Bulletin* 127, 310-319. https://doi.org/10.1016/j.marpolbul.2017.11.054. Accessed 13 January 2021.

Landrigan, P.J., Fuller, R., Acosta, N.J.R., Adeyi, O., Arnold, R., Basu, N. *et al.* (2017). The *Lancet* Commission on pollution and health. *The Lancet* 391(10119), P462-P512. http://dx.doi.org/10.1016/S0140-6736(17)32345-0. Accessed 13 January 2021.

Landrigan, P.J., Stegeman, J., Fleming, L., Allemand, D., Anderson, D., Backer, L. *et al.* (2020) Human health and ocean pollution. *Annals of Global Health* 86(1) 151, 1-64. https://doi.org/10.5334/aogh.2831. Accessed 13 January 2021.
Lau, W.Y., Shiran, Y., Bailey, R.M., Cook, E., Stutchey, M.R., Koskella, J. *et al.* (2020). Evaluating scenarios toward zero plastic pollution. *Science* 369(6510), 1455-1461. https://doi.org/10.1126/science.aba9475; and Lau, W. and Palardy, J. (2020). Plastic pollution, rampant worldwide, could be cut by 80% in 20 years. https://www.pewtrusts.org/en/research-and-analysis/articles/2020/10/08/plastic-pollution-rampant-worldwide-could-be-cut-by-80-percent-in-20-years. Accessed 13 January 2021.

Lavers, J.L. and Bond, A.L. (2017). Exceptional and rapid accumulation of anthropogenic debris on one of the world's most remote and pristine islands. *Proceedings of the National Academy of Sciences* 114(23), 6052-6055. https://doi.org/10.1073/pnas.1619818114. Accessed 13 January 2021.

Lavers, J.L., Hutton, I. and Bond, A.L. (2018). Ingestion of marine debris by Wedge-tailed Shearwaters (*Ardenna pacifica*) on Lord Howe Island, Australia during 2005-2018. *Marine Pollution Bulletin* 133, 616-621. https://doi.org/10.1016/j.marpolbul.2018.06.023. Accessed 13 January 2021.

Law, K.L.L. (2017). Plastics in the marine environment. *Annual Review of Marine Science* 9, 205-29. https://doi.org/10.1146/annurev-marine-010816-060409. Accessed 13 January 2021.

Law, K.L.L., Morét-Ferguson, S., Maximenko, N.A., Proskurowski, G., Peacock, E.E., Hafner, J. *et al.* (2010). Plastic accumulation in the North Atlantic Subtropical Gyre. *Science* 329(5996), 1185-1188. https://doi.org/10.1126/science.1192321. Accessed 13 January 2021.

Law, K.L., Morét-Ferguson, S.E., Goodwin, D.S., Zettler, E.R., DeForce, E., Kukulka, T. *et al.* (2014). Distribution of surface plastic debris in the eastern Pacific Ocean from an 11-year data set. *Environmental Science and Technology* 48(9), 4732-38. https://doi.org/10.1021/es4053076. Accessed 13 January 2021.

Lebreton, L. and Andrady, A. (2019). Future scenarios of global plastic waste generation and disposal. *Palgrave Communications* 5, 6. https://doi.org/10.1057/s41599-018-0212-7. Accessed 13 January 2021.

Lebreton, L.C., Greer, S.D. and Borrero, J.C. (2012). Numerical modelling of floating debris in the world's oceans. *Marine Pollution Bulletin* 64, 653-661. https://doi.org/10.1016/j.marpolbul.2011.10.027. Accessed 13 January 2021.

Lebreton, L.C., van der Zwet, J., Damsteeg, J.W., Slat, B., Andrady, A. and Reisser, J. (2017). River plastic emissions to the world's oceans. *Nature Communications* 8, 5611. https://doi.org/10.1038/ncomms15611. Accessed 13 January 2021.

Lebreton, L., Slat, B., Ferrari, F., Sainte-Rose, B., Aitken, J., Marthouse, R. *et al.* (2018). Evidence that the Great Pacific Garbage Patch is rapidly accumulating plastic. *Scientific Reports* 8, 4666 https://doi.org/10.1038/s41598-018-22939-w. Accessed 13 January 2021.

Lebreton, L., Egger, M. and Slat, B. (2019) A global mass budget for positively buoyant macroplastic debris in the ocean. *Scientific Reports* 9, 12922 [also see Lebreton, L., Egger, M. and Slat, B. (2020). Author correction: A global mass budget for positively buoyant macroplastic debris in the ocean. *Scientific Reports* 10, 1841, listed below]. https://doi.org/10.1038/s41598-019-49413-5. Accessed 13 January 2021.

Lebreton, L., Egger, M. and Slat, B. (2020). Author correction: A global mass budget for positively buoyant microplastic debris in the ocean in the ocean. *Scientific Reports* 10, 1841. https://doi.org/10.1038/s41598-020-58755-4. Accessed 13 January 2021.

Lechner, A., Keckeis, H., Lumesberger-Loisl, F., Zens, B., Krusch, R., Tritthart, M. *et al.* (2014). The Danube so colourful: A potpourri of plastics litter outnumbers fish larvae in Europe's second largest river. *Environmental Pollution* 188, 177-181. https://doi.org/10.1016/j.envpol.2014.02.006. Accessed 13 January 2021.

Leclerc, L.-M.E., Lydersen, C., Haug, T., Bachmann, L., Fisk, A.T. and Kovacs K.M. (2012). A missing piece in the Arctic food web puzzle? Stomach contents of Greenland sharks sampled in Svalbard, Norway. *Polar Biology* 35, 1197-1208. https://doi.org/10.1007/s00300-012-1166-7. Accessed 13 January 2021.

Lee, H., Kunz, A., Shim, W.J. and Walther, B.A. (2019). Microplastic contamination of table salts from Taiwan, including a global review. *Scientific Reports* 9, 10145. https://doi.org/10.1038/s41598-019-46417-z. https://doi.org/10.1038/s41598-019-46417-z. Accessed 13 January 2021.

Lee, J., Lee, J., Hong, S., Hong, S.H., Shimb, W.J and Eo, S. (2017). Characteristics of meso-sized plastic marine debris on 20 beaches in Korea. *Marine Pollution Bulletin* 123(1-2), 92-96. https://doi.org/10.1016/j.marpolbul.2017.09.020. Accessed 13 January 2021.

Leggett, C., Scherer, N., Curry, M. and Bailey, R. (2014). *Final Report: Assessing the Economic Benefits of Reductions in Marine Debris: A Pilot Study of Beach Recreation in Orange County, California*. Prepared for the Marine Debris Division, United States National Oceanic and Atmospheric Administration (NOAA). https://marinedebris.noaa.gov/sites/default/files/publications-files/MarineDebrisEconomicStudy_0.pdf. Accessed 13 January 2021.

Leggett, C., Schere, N., Haab, T.C., Bailey, R., Landrum, J.P. and Domanski, A. (2018). Assessing the economic benefits of reductions in marine debris at southern California beaches: A random utility travel cost model. *Marine Resource Economics* 33(2), 133-153. https://doi.org/10.1086/697152. Accessed 13 January 2021.

Lenz, R. and Labrenz, M. (2018). Small microplastics sampling in water: Development of an encapsulated filtration device. *Water* 10(8), 1055. https://doi.org/10.3390/w10081055. Accessed 13 January 2021.

Léon, V., Garcia, I., González, E., Samper, R., Fernández- González, V. and Muniategui-Lorenzo, S. (2018). Potential transfer of organic pollutants from littoral plastics debris to the marine environment. *Environmental Pollution* 236, 442-453. https://doi.org/10.1016/j.envpol.2018.01.114. Accessed 13 January 2021.

Lepawsky, J., Araujo, E., Davis, J.-M. and Kahzat, R. (2017). Best of two worlds? Towards ethical electronics repair, reuse, repurposing and recycling. *Geoforum* 81, 87-99. https://doi.org/10.1016/j.geoforum.2017.02.007. Accessed 13 January 2021.

Leslie, H.A., Leonards, P.E.G., Brandsma, S.H., J. de Boer, and Jonkers, N. (2016) Propelling plastics into the circular economy – weeding out the toxics first. *Environment International* 94, 230-234. https://www.sciencedirect.com/science/article/pii/S0160412016301854. Accessed 25 May 2021.

Lewis, N., Day, J.C., Wilhelm, A., Wagner, D., Gaymer, C.F., Parks, J. *et al.* (2017). *Large-scale Marine Protected Areas: Guidelines for Design and Management. Prepared by Big Ocean and the IUCN WCPA Large-Scale MPA Task Force.* Best Practice Protected Area Guidelines Series No. 26. Gland, Switzerland: International Union for Conservation of Nature. https://portals.iucn.org/library/sites/library/files/documents/PAG-026.pdf. Accessed 13 January 2021.

Li, J., Lusher, A.L., Rotchell, J.M., Deudero, S., Turra, A., Bråte, I.L.N. *et al.* (2019). Using mussel as a global bioindicator of coastal microplastic pollution. *Environmental Pollution* 2(44), 522-533. https://doi.org/10.1016/j.envpol.2018.10.032. Accessed 13 January 2021.

Li, L., Lu, Y., Li, R., Zhou, Q., Peijnenburg, W.J., Yin, N. *et al.* (2020). Effective uptake of sub micrometre plastics by crop plants via a crack-entry mode. *Nature Sustainability* 3, 929-937. https://doi.org/10.1038/s41893-020-0567-9. Accessed 13 January 2021.

Li, L.F., Zhang, X., Luan, Z.D., Du, Z.F., Xi, S.C., Wang, B. *et al.* (2018). In situ quantitative raman detection of dissolved carbon dioxide and sulfate in deepsea high-temperature hydrothermal vent fluids. *Geochemical Geophysical Geosystems* 19, 7445. https://doi.org/10.1029/2018GC007445. Accessed 20 June 2021.

Li, X., Chen, L., Mei, Q., Dong, B., Dai, X., Ding, G. *et al.* (2018). Microplastics in sewage sludge from the wastewater treatment plants in China. 142, 75-85. https://doi.org/10.1016/j.watres.2018.05.034. Accessed 13 January 2021.

Li, Y., Zhang, H., Shao, L.-M. and He, P.-J. (2017). Tracing source and migration of Pb during waste incineration using stable Pb isotopes. *Journal of Hazardous Materials* 327, 28-34. https://doi.org/10.1016/j.jhazmat.2016.12.029. Accessed 13 January 2021.

Liebezeit, G. and Liebezeit, E. (2013). Non-pollen particulates in honey and sugar. *Food Additives and Contaminants* 30(12), 2136-12124. https://doi.org/10.1080/19440049.2013.843025. Accessed 13 January 2021.

Lieder, M. and Rashid, A. (2015) Towards circular economy implementation: a comprehensive review in context of manufacturing industry. *Journal of Cleaner Production* 115, 36-51.
https://doi.org/10.1016/j.jclepro.2015.12.042. Accessed 20 June 202.1

Liedermann, M., Gmeiner, P., Pessenlehner, S., Haimann, M., Hohenblum, P. and Habersack, H. (2018). A methodology for measuring microplastics transport in large or medium rivers. *Water* 10(4), 414. https://doi.org/10.3390/w10040414. Accessed 13 January 2021.

Lima, A.R.A., Costa, M.F. and Barletta, M. (2014). Distribution patterns of microplastics within the plankton of a tropical estuary. *Environmental Research* 132, 146-155. https://doi.org/10.1016/j.envres.2014.03.031. Accessed 13 January 2021.

Lin, C. and Nakamura, S. (2018). Approaches to solving China's marine plastic pollution and CO_2 emission problems. *Economic Systems Research* 31, 143-157. https://doi.org/10.1080/09535314.2018.1486808. Accessed 13 January 2021.

Lindeque, P.K., Cole, M., Coppock, R.L., Lewis, C.N., Miller, R.Z., Watts, A.J.R. *et al.* (2020). Are we underestimating microplastic abundance in the marine environment? A comparison of microplastic capture with nets of different mesh-size. *Environmental Pollution* 265, Part A, 114721. https://doi.org/10.1016/j.envpol.2020.114721. Accessed 13 January 2021.

Ling, S.D., Sinclair, M., Levi, C.J., Reeves, S.E. and Edgar, G.J. (2017). Ubiquity of microplastics in coastal seafloor sediments. Mar*ine Pollution Bulletin* 121(1-2), 104-110. https://doi.org/10.1016/j.marpolbul.2017.05.038. Accessed 13 January 2021.

Lithner, D., Larsson, A. and Dave, G. (2011). Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. *Science of The Total Environment* 409(18), 3309-3324. https://doi.org/10.1016/j.scitotenv.2011.04.038. Accessed 13 January 2021.

Lithner, D., Nordensvan, I. and Dave, G. (2012). Comparative acute toxicity of leachates from plastic products made of polypropylene, polyethylene, PVC, acrylonitrile-butadiene-styrene, and epoxy to *Daphnia magna*. *Environmental Science and Pollution Research* 19(5), 1763-1772. https://doi.org/10.1007/s11356-011-0663-5. Accessed 13 January 2021.

Liu, M., Lu, S., Song, Y., Lei. L., Hu, J., Lu, W. *et al.* (2018). Microplastic and mesoplastic pollution in farmland soils in suburbs of Shanghai, China. Enviro*nmental Pollution* 242, Part A, 855-862 https://doi.org/10.1016/j.envpol.2018.07.051. Accessed 13 January 2021.

Liubartseva, S., Coppini, G. and Lecci, R. (2019). Are Mediterranean Marine Protected Areas sheltered from plastic pollution? *Marine Pollution Bulletin* 140, 579-587. https://doi.org/10.1016/j.marpolbul.2019.01.022. Accessed 13 January 2021.

Löder, M.G. and Gerdts, G. (2015). Methodology used for the detection and identification of microplastics – a critical appraisal. In *Marine Anthropogenic Litter*. Bergmann, M., Gutow, L. and Klages, E. (eds.). Cham: Springer. 201-227. https://doi.org/10.1007/978-3-319-16510-3_8. Accessed 13 January 2021.

Löder, M.G.J., Imhof, H.K., Ladehoff, M., Loschel, L.A., Lorenz, C., Mintenig, S. *et al.* (2017). Enzymatic purification of microplastics in environmental samples. *Environmental Science and Technology* 51(24), 14283-14292. https://doi.org/10.1021/acs.est.7b03055. Accessed 13 January 2021.

Löhr, A., Savelli, H., Beunen. R., Kalz, M, Ragas, A. and F. Van Belleghem (2017). Solutions for global marine litter pollution. *Current Opinion in Environmental Sustainability* 28, 90-99. https://doi.org/10.1016/j.cosust.2017.08.009. Accessed 13 January 2021.

Lotze, H.K., Guest, H., O'Leary, J., Tuda, A. and Wallace, D. (2018). Public perception of marine threats and protection from around the world. *Ocean and Coastal Management* 152, 14-22. https://doi.org/10.1016/j.ocecoaman.2017.11 004. Accessed 13 January 2021.

Lourenço, P.M., Serra-Gonçalves, C., Ferreira, J.L., Catry, T. and Granadeiro, P. (2017). Plastic and other microfibres in sediments, macroinvertebrates and shorebirds from three intertidal wetlands of southern Europe and west Africa. *Environmental Pollution* 23, Part 1, 123e133. https://doi.org/10.1016/j.envpol.2017.07.103. Accessed 13 January 2021.

Luna-Jorquera, G., Thiel, M., Portflitt-Toro, M. and Dewitte, B. (2019). Marine protected areas invaded by floating anthropogenic litter: An example from the South Pacific. *Aquatic Conservation Marine and Freshwater Ecosystems* 29 (S2), 245-259. https://doi.org/10.1002/aqc.3095.https://doi.org/10.1002/aqc.3095. Accessed 13 January 2021.

Lusher, A.L., Burke, A., O'Connor, I. and Officer, R. (2014). Microplastic pollution in the Northeast Atlantic Ocean: Validated and opportunistic sampling. *Marine Pollution Bulletin* 88(1-2), 325-333. https://doi.org/10.1016/j.marpolbul.2014.08.023. Accessed 13 January 2021.

Lusher, A. L., Tirelli, V., O'Connor, I. and Officer, R. (2015). Microplastics in Arctic polar waters: The first reported values of particles in surface and sub-surface samples. *Scientific Reports* 5, 1-9. https://doi.org/10.1038/srep14947. Accessed 13 January 2021.

Lusher, A.L., Hollman, P.C.H. and Mendoza-Hill, J.J. (2017a). *Microplastics in Fisheries and Aquaculture: Status of Knowledge on Their Occurrence and Implications for Aquatic Organisms and Food Safety*. FAO Fisheries and Aquaculture Technical Paper No. 615. Rome. http://www.fao.org/3/a-i7677e.pdf. Accessed 13 January 2021.

Lusher, A.L., Welden, N.A., Sobral, P. and Cole, M. (2017b). Sampling, isolating and identifying microplastics ingested by fish and invertebrates. *Analytical Methods* 9, 1346. https://doi.org/10.1039/C6AY02415G. Accessed 13 January 2021.

Lynn, H., Rech, S. and Samwel-Mantingh, M. (2017). *Plastics, Gender and the Environment: Findings of a Literature Study on the Lifecycle of Plastics and its Impacts on Women and Men, from Production to Litter*. Women Engage for a Common Future (WECF), The Netherlands, France and Germany. https://www.wecf.org/wp-content/uploads/2018/11/PlasticsgenderandtheenvironmentHighRes-min.pdf. Accessed 13 January 2021.

Lyons, Y., Su, T.L. and Meo, M.L. (2019). A *Review of Research on Marine Plastics in Southeast Asia. Who Does What?* National University of Singapore, British High Commission Singapore, UK Science & Innovation Network. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/813009/A_review_of_research_on_marine_plastics_in_Southeast_Asia_-_Who_does_what.pdf. Accessed 13 January 2021.

Ma, X., Park, C. and Moultrie, J. (2020). Factors for eliminating plastic in packaging: The European FMCG experts' view. *Journal of Cleaner Production* 256, 120492. https://doi.org/10.1016/j.jclepro.2020.120492. Accessed 13 January 2021.

Macfadyen, G., Huntington, T. and Cappell, R. (2009). *Abandoned, Lost or Otherwise Discarded Fishing Gear*. UNEP Regional Seas Reports and Studies No.185; FAO Fisheries and Aquaculture Technical Paper No. 523. Rome. http://www.fao.org/3/i0620e/i0620e00.htm. Accessed 13 January 2021.

Maeland. C.E. and Staupe-Delgado, R. (2020). Can the global problem of marine litter be considered a crisis? *Risks, Hazards and Crisis in Public Policy* 11, 87-104. https://doi.org/10.1002/rhc3.12180. Accessed 13 January 2021.

Maes, T., Van der Meulen, M.D., Devriese, L.I., Leslie, H.A., Huvet, A., Frère, L. *et al.* (2017a). Microplastics baseline surveys at the water surface and in sediments of the North-East Atlantic. *Frontiers in Marine Science* 4, 135. https://doi.org/10.3389/fmars.2017.00135. Accessed 13 January 2021.

Maes, T., Jessop, R., Wellner, N., Haupt, K. & Mayes, A. G. (2017b). A rapid-screening approach to detect and quantify microplastics based on fluorescent tagging with Nile Red. *Science Report* 7, 44501. https://doi:10.1038/srep44501. Accessed 15 February 2021.

Maes, T., Barry, J., Leslie, H.A., Vethaak, A.D., Nicolaus, E.E.M., Law, R.J. *et al.* (2018). Below the surface: Twenty-five years of seafloor litter monitoring in coastal seas of North West Europe (1992- 2017). *Science of The Total Environment* 630, 790-798. https://doi.org/10.1016/j.scitotenv.2018.02. 245. Accessed 13 January 2021.

Maes, T., Perry, J., Alliji, K., Clarke, C. and Birchenough, A.N.R. (2019). Shades of grey: Marine litter research developments in Europe. *Marine Pollution Bulletin* 146, 274-281. https://doi.org/10.1016/j.marpolbul.2019.06.019. Accessed 13 January 2021.

Maes, T., Barry, J., Stenton, C., Roberts, E., Hicks, R., Bignell, J. *et al.* (2020). The world is your oyster: Low-dose, long-term microplastic exposure of juvenile oysters. *Heliyon Research Article* 6 E03103. https://doi.org/10.1016/j.heliyon.2019.e03103. Accessed 15 February 2021.

Mahon, A.M., O'Connell, B., Healy, M.G., O'Connor, I., Officer, R., Nash, R. *et al.* (2017). Microplastics in sewage sludge: Effects of treatment. *Environmental Science and Technology* 51(2), 810-818. https://doi.org/10.1021/acs.est.6b04048. Accessed 13 January 2021.

Majer, A.P., Vedolin, M.C. and Turra, A. (2012). Plastic pellets as oviposition site and means of dispersal for the ocean-skater insect *Halobates*. *Marine Pollution Bulletin* 64(6), 1143-1147. https://doi.org/10.1016/j.marpolbul.2012.03.029. Accessed 13 January 2021.

Malik, D., Manchanda, P., Simons, T.J. and Wallach, J. (2020). The impact of COVID-19 on the global petrochemical industry, McKinsey and Company, 28 October. https://www.mckinsey.com/industries/chemicals/our-insights/the-impact-of-covid-19-on-the-global-petrochemical-industry. Accessed 10 February 2020.

Mani, T., Hauk, A., Walter, U. and Burkhardt-Holm, P. (2015). Microplastics profile along the Rhine River. *Scientific Reports* 5, 1-7. https://doi.org/10.1038/srep17988 . Accessed 13 January 2021.

Mani, T., Primpke, S., Lorenz, C., Gerdts, G. and Burkhardt-Holm, P. (2019). Microplastic pollution in benthic mid-stream sediments of the Rhine River. *Environmental Science and Technology* 53(10), 6053-6062. https://doi.org/10.1021/acs.est.9b01363. Accessed 13 January 2021.

Mansui, J., Molcard, A. and Ourmieres, A. (2015). Modelling the transport and accumulation of floating marine debris in the Mediterranean basin. *Marine Pollution Bulletin* 91(1), 249-257. https://doi.org/10.1016/j.marpolbul.2014.11.037. Accessed 13 January 2021.

Markic, A., Gaertner, J.C., Gaertner-Mazouni, N. and Koelmans, A.A. (2020). Plastic ingestion by marine fish in the wild. *Critical Reviews in Environmental Science and Technology* 50(7), 67-697. https://doi.org/10.1080/10643389.2019.1631990. Accessed 13 January 2021.

Martin, C., Almahasheer, H. and Duarte, C.M. (2019). Mangrove forests as traps for marine litter. *Environmental Pollution* 247, 499-508. https://doi.org/10.1016/j.envpol.2019.01.067. Accessed 13 January 2021.

Martin, C., Parkes, S., Zhang, Q., Zhang, X., McCabe, M. and Duarte, C.M. (2018). Use of unmanned aerial vehicles for efficient beach litter monitoring. *Marine Pollution Bulletin* 131, Part A, 662-673. https://doi.org/10.1016/j.marpolbul.2018.04.045. Accessed 13 January 2021.

Martínez-Gómez, C., León, V.M., Calles, S., Gomáriz-Olcina, M. and Vethaak, A.D. (2017). The adverse effects of virgin microplastics on the fertilization and larval development of sea urchins. *Marine Environmental Research* 130, 69-76. https://doi.org/10.1016/j.marenvres.2017.06.016. Accessed 13 January 2021.

Martínez-Vicente, V., Clark, J.R. Corradi, P., Aliani, S., Arias, M., Bochow, M. *et al.* (2019). Measuring marine plastic debris from space: Initial assessment of observation requirements. *Remote Sensing* 11, 2443. https://doi.org/10.3390/rs11202443. Accessed 13 January 2021.

Maryland General Assembly (2020). Environment - Compostable, Degradable, and Biodegradable Plastic Products - Labeling. Annapolis, MD, USA. https://mgaleg.maryland.gov/mgawebsite/legislation/details/hb1349?ys=2017rs. Accessed 22 September 2020.

Mason, S.A., Garneau, D., Sutton, R., Chu, Y., Ehmann, K., Barnes, J. *et al.* (2016). Microplastics pollution is widely detected in US municipal wastewater treatment plant effluent. *Environmental Pollution* 218, 1045-1054. https://doi.org/10.1016/j.envpol.2016.08.056. Accessed 13 January 2021.

Mason, S.A., Welch, V.G. and Neratko, J. (2018). Synthetic polymer contamination in bottled water. *Frontiers in Chemistry* 6, 407. https://doi.org/10.3389/fchem.2018.00407. Accessed 13 January 2021.

Masura, J., Baker, J., Foster, G. and Arthur, C. (2015). *Laboratory Methods for the Analysis of Microplastics in the Ma*rine Environment: *Recommendations for Quantifying Synthetic Particles in Waters and Sediments*. United States National Oceanic and Atmospheric Administration, Marine Debris Program. NOAA Technical Memorandum NOS-OR&R-48. https://repository.oceanbestpractices.net/bitstream/handle/11329/1076/noaa_microplastics_methods_manual.pdf?sequence=1&isAllowed=y. Accessed 13 January 2021.

Mathalon, A. and Hill, P. (2014). Microplastic fibres in the intertidal ecosystem surrounding Halifax Harbor, Nova Scotia. *Marine Pollution Bulletin* 81(1),69-79.
https://doi.org/10.1016/j.marpolbul.2014.02.018. Accessed 13 January 2021.

Matheson, T. (2019). *Disposal is Not Free: Fiscal Instruments to Internalize the Environmental Costs of Solid Waste*. International Monetary Fund Working Paper 19/283. https://www.imf.org/en/Publications/WP/Issues/2019/12/20/Disposal-is-Not-Free-Fiscal-Instruments-to-Internalize-the-Environmental-Costs-of-Solid-Waste-48854. Accessed 13 January 2021.

Matiddi, M., Hoschsheid, S., Camedda, A., Baini, M., Cocumelli, C., Serena, P. *et al.* (2017). Loggerhead sea turtles (*Caretta caretta*): A target species for monitoring litter ingested by marine organisms in the Mediterranean Sea. *Environmental Pollution* 230, 199-209. https://doi.org/10.1016/j.envpol.2017.06.054. Accessed 13 January 2021.

Mattsson, K., Hansson, L.-A. and Cedervalla, T. (2015). Nano-plastics in the aquatic environment. *Environmental Sciences: Processes and Impacts* 17, 1712. https://doi.org/10.1039/c5em00227C. Accessed 13 January 2021.

Maximenko, N., Hafner, J. and Niller, P. (2012). Pathways of marine debris derived from trajectories of Lagrangian drifters. *Marine Pollution Bulletin* 65(1-3), 51-62. http://dx.doi.org/10.1016/j.marpolbul.2011.04.016. Accessed 13 January 2021.

Maximenko, N., Corradi, P., Law, K.L., Van Sebille, E., Garaba, S.P., Lampitt, R.S. *et al.* (2019). Toward the Integrated Marine Debris Observing System. *Frontiers in Marine Science* 6, 447. https://doi.org/10.3389/fmars.2019.00447. Accessed 13 January 2021.

McAdam, R. (2017). Plastic in the ocean: How much is out there? *Significance* 14(5), 24-27. https://doi.org/10.1111/j.1740-9713.2017.01072.x. Accessed 13 January 2021.

McCormick, A., Hoellein, T., Mason, S., Schluep, J. and Kelly, J. (2014). Microplastic is an abundant and distinct microbial habitat in an urban river. *Environmental Science and Technology* 48(20), 11863-11871. https://doi.org/10.1021/es503610r. Accessed 13 January 2021.

McCormick, A.R. and Hoellein, T.J. (2016). Anthropogenic litter is abundant, diverse and mobile in urban rivers: Insights from cross-ecosystem analyses using ecosystem and community ecology tools. *Limnology and Oceanography* 61(5), 1718-1734. https://doi.org/10.1002/lno.10328. Accessed 13 January 2021.

McDermid, K.J. and McMullen, T.L. (2004). Quantitative analysis of small-plastic debris on beaches in the Hawaiian archipelago. *Marine Pollution Bulletin* 48(7-8), 790-794.
https://doi.org/10.1016/j.marpolbul.2003.10.017. Accessed 13 January 2021.

McGuire, N.M. (2015). Environmental education and behavioural change: An identity-based environmental education model. *International Journal of Environmental Science Education* 10, 695-715. https://files.eric.ed.gov/fulltext/EJ1081842.pdf. Accessed 13 January 2021.

McIlgorm, A., Campbell H.F. and Rule M.J. (2008). *Understanding the Economic Benefits and Costs of Controlling Marine Debris in the APEC Region* (MRC 02/2007). A report to the Asia-Pacific Economic Cooperation Marine Resource Conservation Working Group by the National Marine Science Centre (University of New England and Southern Cross University), Coffs Harbour, NSW, Australia, December. https://www.apec.org/Publications/2009/04/Understanding-the-Economic-Benefits-and-Costs-of-Controlling-Marine-Debris-In-the-APEC-Region. Accessed 27 July 2021.

McIlgorm, A., Campbell, H.F. and Rule, M.J. (2011). The economic cost and control of marine debris damage in the Asia-Pacific region. *Ocean and Coastal Management* 54(9), 643-651. https://doi.org/10.1016/j.ocecoaman.2011.05.007. Accessed 13 January 2021.

McIlgorm, A., Raubenheimer, K. and McIlgorm, D.E. (2020). *Update of 2009 APEC Report on Economic Costs of Marine Debris to APEC Economies*. Report to the APEC Oceans and Fisheries Working Group by the Australian National Centre for Ocean Resources and Security (ANCORS), University of Wollongong, Australia. https://www.apec.org/Publications/2020/03/Update-of-2009-APEC-Report-on-Economic-Costs-of-Marine-Debris-to-APEC-Economies. Accessed 13 January 2021.

McKinsey and Company (2020). *The State of the Chemical Industry – It is getting more complex*. https://www.mckinsey.com/industries/chemicals/our-insights/the-state-of-the-chemical-industry-it-is-getting-more-complex. Accessed 10 February 2020.

McLeod, E., Chmura, G.L., Bouillon, S., Salm, R., Björk, M., Duarte, C. M. *et al.* (2011). A blueprint for blue carbon: Toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Frontiers in Ecology and the Environment* 9(10), 552-560. https://doi.org/10.1890/110004. Accessed 15 February 2021.

McNeish, R.E., Kim, L.H., Barrett, H.A., Mason, S.A, Kelly, J.J. and Hoellein, T.J. (2018). Microplastic in riverine fish is connected to species traits. *Scientific Reports* 8(1), 11639. https://doi.org/10.1038/s41598-018-29980-9. Accessed 13 January 2021.

Meijer, J.J., van Emmerik, T., van der Ent, R. Schmidt, C. and Lebreton, L. (2021). More than 1000 rivers account for 80% of global riverine plastic emissions into the ocean. *Science Advances* 7(18), eaaz5803. https://doi.org/10.1126/sciadv.aaz5803. Accessed 30 May 2021.

Meyerjürgens, J., Badewien, T.H., Garaba, S.P., Wolff, J.O. and Zielinski, O. (2019). A state-of-the-art compact surface drifter reveals pathways of floating marine litter in the German bight. *Frontiers in Marine Science* 6, 58. https://doi.org/10.3389/fmars.2019.00058. Accessed 13 January 2021.

Michida, Y., Chavanich, S., Cózar Cabañas, A., Hagmann, P., Hinata, H., Isobe, A. *et al.* (2020). *Guidelines for Harmonizing Ocean Surface Microplastic Monitoring Methods.* Version 1.1, June 2020. Ministry of the Environment of Japan. https://www.env.go.jp/en/water/marine_litter/guidelines/guidelines.pdf. Accessed 13 January 2021.

Miller, R.Z., Watts, A.J., Winslow, B.O., Galloway, T.S. and Barrows, A.P.W. (2017). Mountains to the sea: River study of plastic and non-plastic microfibre pollution in the northeast USA. *Marine Pollution Bulletin* 124(1), 245-251. https://doi.org/10.1016/j.marpolbul.2017.07.028. Accessed 13 January 2021.

Mintenig, S.M., Int-Veen, I., Löder, M.G., Primpke, S. and Gerdts, G. (2017). Identification of microplastics in effluents of wastewater treatment plants using focal plane array-based micro-Fourier- transform infrared imaging. *Water Research* 108, 365-372. https://doi.org/10.1016/j.watres.2016.11.015. Accessed 13 January 2021.

Mohamed Nor, N.H. and Koelmans, A.A. (2019). Transfer of PCBs from microplastics under simulated conditions is biphasic and reversible. *Environmental Science and Technology* 53(4), 1874-1883. https://doi.org/10.1021/acs.est.8b05143. Accessed 13 January 2021.

Moltmann, T., Turton, J., Zhang, H.-M., Nolan, G., Gouldman, C., Griesbauer, L. *et al.* (2019). A Global Ocean Observing System (GOOS), delivered through enhanced collaboration across regions, communities, and new technologies. *Frontiers in Marine Science* 6, 291. https://doi.org/10.3389/fmars.2019.00291. Accessed 13 January 2021.

Morgana, S., Ghigliotti, L., Estévez-Calvar, N., Stifanese, R., Wieckzorek, A., Doyle, T. *et al.* (2018). Microplastics in the Arctic: A case study with sub-surface water and fish samples off Northeast Greenland. *Environmental Pollution* 242, Part B, 1078-1086. https://doi.org/10.1016/j.envpol.2018.08.001. Accessed 13 January 2021.

Morohoshi, T., Ogata, K., Okura, T. and Sato, S. (2018). Molecular characterization of the bacterial community in biofilms for degradation of poly(3-hydroxybutyrate-co-3-hydroxyhexanoate) films in seawater. *Microbes and Environment* 33(1), 19-25. https://doi.org/10.1264/jsme2.ME17052. Accessed 13 January 2021.

Morritt, D., Stefanoudis, P.V., Pearce, D., Crimmen, O.A. and Clark, P.F. (2014). Plastic in the Thames: A river runs through it. *Marine Pollution Bulletin* 78(1-2), 196-200. https://doi.org/10.1016/j.marpolbul.2013.10.035. Accessed 13 January 2021.

Mouat, J., Lozano, R.L. and Bateson, H. (2010). *Economic Impacts of Marine Litter*. KIMO (Kommunernes International Miljøorganisation/ Local Authorities International Environmental Organisation). http://www.kimointernational.org/wp/wp-content/uploads/2017/09/KIMO_Economic-Impacts-of-Marine-Litter.pdf. Accessed 13 January 2021.

M'Rabat, C., Pringault, O., Zmerli-Triki, H., Héla, B.G., Couet, D. and Kéfi-Daly Yahia, O. (2018). Impact of two plastic-derived chemicals, the Bisphenol A and the di-2-ethylhexyl phthalate, exposure on the marine toxic dinoflagellate *Alexandrium pacificum*. *Marine Pollution Bulletin* 126, 241-249. https://doi.org/10.1016/j.marpolbul.2017.10.090. Accessed 13 January 2021.

Mu, J. Zhang, S., Qu, L., Jin, F., Fang, C., Ma, A. *et al.* (2019). Microplastics abundance and characteristics in surface waters from the Northwest Pacific, the Bering Sea, and the Chukchi Sea. *Marine Pollution Bulletin* 143, 58-65, https://doi.org/10.1016/j.marpolbul.2019.04.023. Accessed 13 January 2021.

Muirhead, J. and Porter, T. (2019). Traceability in global governance. *Global Networks* 19(3), 423-443. https://doi.org/10.1111/glob.12237. Accessed 13 January 2021.

Munari, C., Corbau, C., Simeoni, U. and Mistri, M., (2015). Marine litter on Mediterranean shores: Analysis of composition, spatial distribution and sources in north-western Adriatic beaches. *Waste Management* 49, 483-490. https://doi.org/10.1016/j.wasman.2015.12.010. Accessed 13 January 2021.

Munari, C., Infantini, V., Scoponi, M., Rastelli, E., Coridesi, C. and Mistri, M. (2017). Microplastics in the sediments of Terra Nova Bay (Ross Sea, Antarctica). *Marine Pollution Bulletin* 122(1-2), 161-165. https://doi.org/10.1016/j.marpolbul.2017.06.039. Accessed 13 January 2021.

Muncke, J., Andersson, A.M., Backhaus, T., Boucher, J.M., Almroth, B.C., Castillo, A.C. *et al.* (2020). Impacts of food contact chemicals on human health: A consensus statement. *Environmental Health* 19, 25. https://doi.org/10.1186/s12940-020-0572-5. Accessed 13 January 2021.

Murphy, F., Ewins, C., Carbonnier, F. and Quinn, B. (2016). Wastewater Treatment Works (WwTW) as a source of microplastics in the aquatic environment. *Environmental Science and Technology* 50(11), 5800-5808. https://doi.org/10.1021/acs.est.5b05416. Accessed 13 January 2021.

Murray, C.C., Maximenko, N. and Lippiatt, S. (2018). The influx of marine debris from the great Japan Tsunami of 2011 to North America shorelines. *Marine Pollution Bulletin* 132, 26-32. https://doi.org/10.1016/j.marpolbul.2018.01.004. Accessed 13 January 2021.

Naji, A., Esmaili, Z. and Khan, F.R. (2017). Plastic debris and microplastics along the beaches of the strait of Hormuz, Persian Gulf. *Marine Pollution Bulletin* 114(2), 1057e1062. https://doi.org/10.1016/j.marpolbul.2016.11.032. Accessed 13 January 2021.

Nakashima, E., Isobe, A., Kako, S., Itai, T., Takahashi, S. and Guo, X. (2016). The potential of oceanic transport and onshore leaching of additive-derived lead by marine macro-plastic debris. *Marine Pollution Bulletin* 107, 333-339. https://doi.org/10.1016/j.marpolbul.2016.03.038. Accessed 13 January 2021.

Näkki P., Setälä, O. and Lehtiniemi, M. (2017). Bioturbation transports secondary microplastics to deeper layers in soft marine sediments of the northern Baltic Sea. *Marine Pollution Bulletin* 119(1), 255-261. https://doi.org/10.1016/j.marpolbul.2017.03.065. Accessed 13 January 2021.

Napper, I.E. and Thompson, R.C. (2019). Environmental deterioration of biodegradable, oxo biodegradable, compostable, and conventional plastics carrier bags in the sea, soil, and open-air over a 3-year period. *Environmental Science and Technology* 53(9), 4775-4783. https://doi.org/10.1021/acs.est.8b06984. Accessed 13 January 2021.

Narancic, T., Verstichel, S., Chaganto, S.R., Morales-Gamez, L., Kenny, S.T., De Wilde, B. *et al.* (2018). Biodegradable plastic blends create new possibilities for end-of-life management of plastics but they are not a panacea for plastic pollution. *Environmental Science and Technology* 52(18),10441-10452. https://doi.org/10.1021/acs.est.8b02963. Accessed 13 January 2021.

Nashoug, B.F. (2017). *"Sources of Marine Litter" – Workshop Report from WP 1.2 in the MARP[3] project.* Svalbard, 4-6 September 2016. SALT Report 1017. Lofoten, Norway. https://pame.is/document-library/desktop-study-on-marine-litter-library/marine-litter-sources/577-nashoug-2017-sources-of-marine-litter-worksh/file. Accessed 13 January 2021.

Nelms, S.E., Coombes, C., Foster, L.C., Galloway, T.S., Godley, B.J., Lindeque, P.K. *et al.* (2017). Marine anthropogenic litter on British beaches: A 10-year nationwide assessment using citizen science data. *Science of The Total Environment* 579, 1399-1409. https://doi.org/10.1016/j.scitotenv.2016.11.137. Accessed 13 January 2021.

Nelms, S.E., Barnett, J., Brownlow, A., Davison, N.J., Deaville, R., Galloway, T.S. *et al.* (2019a). Microplastics in marine mammals stranded around the British coast: Ubiquitous but transitory? *Scientific Reports* 9(1), 1075. https://doi.org/10.1038/s41598-018-37428-3. Accessed 13 January 2021.

Nelms, S.E., Parry, H.E., Bennett, K.A., Galloway, T.S., Godley, B.J., Santillo, D. *et al.* (2019b). What goes in, must come out: Combining scat-based molecular diet analysis and quantification of ingested microplastics in a marine top predator. *Methods in Ecological Evolution* 10(10), 1712-1722. https://doi.org/10.1111/2041-210X.13271. Accessed 13 January 2021.

New Zealand Ministry for the Environment (2019). Reducing waste: a more effective landfill levy. https://environment.govt.nz/publications/reducing-waste-a-more-effective-landfill-levy-consultation-document/. Accessed 25 May 2021.

Newman, S., Watkins, E., Farmer, A., ten Brink, P. and Schweitzer, J.P. (2015). The economics of marine litter. In *Marine Anthropogenic Litter*. Bergmann, M., Gutow, L. and Klages, E. (eds.). Cham: Springer Open Access. 367-394. https://link.springer.com/chapter/10.1007/978-3-319-16510-3_14. Accessed 13 January 2021.

Ng, E-L., Lwanga, E.H., Edridge, S.M., Johnston, P., Hu, H-W., Geissen, V. *et al.* (2018). An overview of microplastic and nanoplastic pollution in agroecosystems. *Science of The Total Environment* 627, 1377- 1388. https://doi.org/10.1016/j.scitotenv.2018.01.341. Accessed 13 January 2021.

Niaounoukis, M., Kontou, E., Pispas, S., Kafetzi, M. and Giaouzi, D. (2019). Aging of packaging films in the marine environment. *Polymer Engineering Science* 59(issue s2), E432-441. https://doi.org/10.1002/pen.25079. Accessed 13 January 2021.

Nielsen J., Hedeholm R.B., Simon, M. and Steffensen J.F. (2014). Distribution and feeding ecology of the Greenland shark (*Somniosus microcephalus*) in Greenland waters. *Polar Biology* 37, 37-46. https://doi.org/10.1007/s00300-013-1408-3. Accessed 13 January 2021.

Nizzetto, L., Futter, M. and Langaas, S. (2016a). Are agricultural soils dumps for microplastics of urban origin? *Environmental Science and Technology* 50(20), 10777-10779. https://doi.org/10.1021/acs.est.6b04140. Accessed 13 January 2021.

Nizzetto, L., Bussi, G., Futter, M.N., Butterfield, D. and Whitehead, P.G. (2016b). A theoretical assessment of microplastic transport in river catchments and their retention by soils and river sediments. *Environmental Science: Processes and Impacts* 18(8), 1050-1059. https://doi.org/10.1039/C6EM00206D. Accessed 13 January 2021.

Njeru, J. (2006). The urban political ecology of plastic bag waste problem in Nairobi, Kenya. *Geoforum* 37(6), 1046-1058. https://doi.org/10.1016/j.geoforum.2006.03.003. Accessed 13 January 2021.

NOAA (United States National Oceanographic and Atmospheric Administration) (2015). *Detecting Japan Tsunami Marine Debris at Sea: A Synthesis of Efforts and Lessons Learned*. NOAA Marine Debris Program, United States Department of Commerce, Technical Memorandum NOS-OR&R-51. https://marinedebris.noaa.gov/sites/default/files/JTMD_Detection_Report.pdf. Accessed 20 November 2020.

Nobre, C.R., Santana, M.F.M., Maluf, A., Cortez, F.S., Cesar, A., Pereira, C.D.S. *et al.* (2015). Assessment of microplastic toxicity to embryonic development of the sea urchin *Lytechinus variegatus* (Echinodermata: Echinoidea). *Marine Pollution Bulletin* 92(1-2), 99-104. https://doi.org/10.1016/j.marpolbul.2014.12.050. Accessed 13 January 2021.

Norris, B.K., Mullarney, J.C., Bryan, K.R. and Henderson, S.M. (2017). The effect of pneumatophore density on turbulence: A field study in a Sonneratia-dominated mangrove forest, Vietnam. *Continental Shelf Research* 147, 114e127. https://doi.org/10.1016/j.csr.2017.06.002. Accessed 13 January 2021.

Northwest Pacific Action Plan (2017). *NOWPAP Medium-term Strategy 2018-2023*. https://wedocs.unep.org/handle/20.500.11822/27258. Accessed 13 January 2021.

Obbard, R.W. (2018). Microplastics in polar regions: The role of long range transport. *Current Opinion in Environmental Science and Health* 1, 24-29. https://doi.org/10.1016/j.coesh.2017.10.004. Accessed 13 January 2021.

Obbard, R.W., Sadri, S., Wong, Y.Q., Khitun, A.A., Baker, I. and Thompson, R.C. (2014). Global warming releases microplastics legacy frozen in Arctic Sea ice. *Earth's Future* 2(6), 315-320. https://doi.org/10.1002/2014EF000240. Accessed 13 January 2021.

Oberbeckmann, S., Osborn, A.M. and Duhaime, M.B. (2016). Microbes on a bottle: substrate, season and geography influence community composition of microbes colonizing marine plastic debris. *PLoS One* 11, e0159289. https://doi.org/10.1371/journal.pone.0159289. Accessed 23 May 2020.

O'Brine, T. and Thompson, R.C. (2010). Degradation of plastic carrier bags in the marine environment. Mar*ine Pollution Bulletin* 60, 2279-2283. https://doi.org/10.1016/j.marpolbul.2010.08.005. Accessed 13 January 2021.

Oßmann, B.E., Sarau, G., Holtmannspötter, H., Pischetsrieder, M., Christiansen, S.H. and Dicke, W. (2018). Small-sized microplastics and pigmented particles in bottled mineral water. *Water Research* 141, 307-316. https://doi.org/10.1016/j.watres.2018.05.027. Accessed 13 January 2021.

Ocean Conservancy and McKinsey Center for Business and Environment (2015). *Stemming the Tide; Land-based Strategies for a Plastic-free Ocean*. https://www.mckinsey.com/business-functions/sustainability/our-insights/stemming-the-tide-land-based-strategies-for-a-plastic-free-ocean. Accessed 13 January 2021.

OECD (Organisation for Economic Co-operation and Development) (2009). *Guidance Manual for the Control of Transboundary Movements of Recoverable Wastes*. https://www.oecd.org/env/waste/guidance-manual-control-transboundary-movements-recoverable-wastes.pdf. Accessed 13 January 2021.

OECD (2016). *Extended Producer Responsibility: Updated Guidance for Efficient Waste Management*. https://doi.org/10.1787/9789264256385-en. Accessed 13 January 2021.

Okshevsky, M., Gautier, E., Farner, J.M., Schreiber, L. and Tufenkji, N. (2020). Biofilm formation by marine bacteria is impacted by concentration and surface functionalization of polystyrene nanoparticles in a species-specific manner. *Environmental Microbiology Reports* 12(2), 203-213. https://doi.org/10.1111/1758-2229.12824. Accessed 13 January 2021.

Olson, R.J., Shalapyonok, A., Kalb, D.J., Graves, S.W. and Sosik, H.M. (2017). Imaging flowcytobot modified for high throughput by in-line acoustic focusing of sample particles. *Limnology and Oceanography Methods* 15, 867-874. https://doi.org/10.1002/lom3.10205. Accessed 13 January 2021.

Onda, D.F., and Sharief, K.M. (2021). Identifcation of microorganisms related to microplastics. *Handbook of Microplastics in the Environment*. T. Rocha-Santos *et al.* (eds.). https://doi.org/10.1007/978-3-030-10618-8_40-1. Accessed 20 June 2021.

O'Neill, K. (2018). The new global political economy of waste. In *A Research Agenda for Global Environmental Politics.* Dauvergne, P. and Alger, J. (eds.). Cheltenham, UK: Edward Elgar. 87-100. https://www.e-elgar.com/shop/gbp/a-research-agenda-for-global-environmental-politics-9781789902181.html. Accessed 13 January 2021.

Onink, V., Wichmann, D., Delandmeter, P. and van Sebille, E. (2019). The role of Ekman currents, geostrophy, and Stokes drift in the accumulation of floating microplastic. *Journal of Geophysical Research: Oceans* 124, 1474-1490. https://doi.org/10.1029/2018JC014547. Accessed 13 January 2021.

Oosterhius, F., Papyrakis, E. and Boteler, B. (2014). Economic Instrument and marine litter control. *Ocean and Coastal* Management 102, 47-54. https://doi.org/10.1016/j.ocecoaman.2014.08.005. Accessed 13 January 2021.

OSPAR (2015). OSPAR request on development of a common monitoring protocol for plastics particles in fish stomachs and selected shellfish on the basis of existing fish disease surveys. http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2015/Special_Requests/OSPAR_PLAST_advice.pdf. Accessed 13 January 2021.

OSPAR (2020). Monitoring and assessing marine litter: Marine litter indicator assessments. https://www.ospar.org/work-areas/eiha/marine-litter/assessment-of-marine-litter. Accessed 13 January 2021.

Ostle, C., Thompson, R.C., Broughton, D., Gregory, L., Wootton, M. and Johns, D.G. (2019). The rise in ocean plastics evidence from a 60-year time series. *Nature Communications* 10, 1622. http://doi.org/10.1038/s41467-019-09506-1. Accessed 13 January 2021.

Pabortsava, K. and Lampitt. R.S. (2020). High concentrations of plastic hidden beneath the surface of the Atlantic Ocean**.** *Nature Communications* 11, 4073. https://doi.org/10.1038/s41467-020-17932-9. Accessed 13 January 2021.

Palatinus, A., Kovač Viršek, M., Robič, U., Grego, M., Bajt, O., Šiljić, J. *et al.* (2019). Marine litter in the Croatian part of the middle Adriatic Sea: Simultaneous assessment of floating and seabed macro and micro litter abundance and composition. *Marine Pollution Bulletin* 139, 427-439. https://doi.org/10.1016/j.marpolbul.2018.12.038. Accessed 13 January 2021.

PAME (Protection of the Arctic Marine Environment Working Group of the Arctic Council) (2019). *Desktop Study on Marine Litter including Microplastics in the Arctic.* http://hdl.handle.net/11374/2389. Accessed 13 January 2021.

Panno, S.V., Kelly, W.R., Scott, J., Zheng, W., McNeish, R.E., Holm, N. *et al.* (2019). Microplastics contamination in Karst groundwater systems. *Groundwater* 5, 189-196. https://doi.org/10.1111/gwat.12862. Accessed 13 January 2021.

Panti, C., Giannetti, M., Baini, M., Rubegni, F., Minutoli, R. and Fossi, M.C. (2015). Occurrence, relative abundance and spatial distribution of microplastics and zooplankton NW of Sardinia in the Pelagos sanctuary protected area, Mediterranean Sea. *Environmental Chemistry* 12, 618. https://doi.org/10.1071/EN14234. Accessed 13 January 2021.

Papathanasopoulou, I., White, M.P., Hattam, C., Lannin, A., Harvey, A. and Spencer, A., (2016). Valuing the health benefits of physical activities in the marine environment and their importance for marine spatial planning. *Marine Policy* 63, 144-152. https://doi.org/10.1016/j.marpol.2015.10.009. Accessed 13 January 2021.

Parts, C. (2019). Waste not want not: Chinese recyclable waste restrictions, their global impact, and potential U.S. responses. *Chicago Journal of International Law* 20(1), article 8. https://chicagounbound.uchicago.edu/cjil/vol20/iss1/8. Accessed 13 January 2021.

Pasternak, G., Zviely, D. and Ribic, C.A. (2017). Sources, composition and spatial distribution of marine litter along the Mediterranean coast of Israel. *Marine Pollution Bulletin* 114, 1036-1045. https://doi.org/10.1016/j.marpolbul.2016.11.023. Accessed 13 January 2021.

Patra, J.K. and Gouda, S. (2013). Application of nanotechnology in textile engineering: An overview. *Journal of Engineering and Technology Research* 5(5), 104-111. https://academicjournals.org/article/article1379503776_Patra%20and%20Sgouda.pdf. Accessed 13 January 2021.

Paul-Pont, I., Lacroix, C., Fernández, C.G., Hégaret, H., Lambert, C., Le Goïc, N. *et al.* (2016). Exposure of marine mussels *Mytilus* spp. to polystyrene microplastics: toxicity and influence on fluoranthene bioaccumulation. *Environmental Pollution* 216, 724-737. https://doi.org/10.1016/j.envpol.2016.06.039. Accessed 13 January 2021.

Paul-Pont. I., Tallec, K., Gonzalez-Fernandez, C., Lambert, C., Vincent, D., Mazurais, D. *et al.* (2018). Constraints and priorities for conducting experimental exposures of marine organism microplastics. *Frontiers in Marine Science* 5, 252. https://doi.org/10.3389/fmars.2018.00252. Accessed 13 January 2021.

Pauly, J.L., Stegmeier, S.J., Allaart, H.A., Cheney, R.T., Zhang, P.J., Mayer, A.G. *et al.* (1998). Inhaled cellulosic and plastic fibres found in human lung tissue. *Cancer, Epidemiology, Biomarkers and Prevention* 7, 419-428. https://cebp.aacrjournals.org/content/cebp/7/5/419.full.pdf. Accessed 13 January 2021.

Pedrós-Alió, C. (2012). The rare bacterial biosphere. *Annual Review of Marine Science* 4, 449–466. https://doi.org/10.1146/annurev-marine-120710-100948. Accessed 23 May 2020.

Pedrotti, M.L., Petit, S., Elineau, A., Bruzaud, S., Crebassa, J.-C., Dumontet, B. *et al.* (2016). Changes in the floating plastics pollution of the Mediterranean Sea in relation to the distance to land. *PLoS ONE* 11(8), e0161581. https://doi.org/10.1371/journal.pone.0161581. Accessed 13 January 2021.

Peeken, I., Primpke, S., Beyer, B., Gütermann, J., Katlein, C., Krumpen, T. *et al.* (2018). Arctic sea ice is an important temporal sink and means of transport for microplastic. *Nature Communications* 9(1):1-12. https://doi.org/10.1038/s41467-018-03825-5. Accessed 13 January 2021.

Peelman, N., Ragaert, P., De Meulenaer, B., Adons, D., Peeters, R., Cardon, L., Van Impe, F. and Devlieghere, F. (2013). Application of bioplastics for food packaging. *Trends in Food Science and Technology* 32, 128-141. https://doi.org/10.1016/j.tifs.2013.06.003. Accessed 13 January 2021

Pellini, G., Gomiero, A., Fortibuoni, T., Ferrà, C., Grati, F., Tassetti, N. *et al.* (2018). Characterization of microplastic litter in the gastrointestinal tract of *Solea solea* from the Adriatic Sea. *Environmental Pollution* 234, 943-952. https://doi.org/10.1016/j.envpol.2017.12.038. Accessed 13 January 2021.

Pendleton, L.H., Thebaud, O., Mongruel, R.C. and Levrel, H. (2016). Has the value of global marine and coastal ecosystem services changed? *Marine Policy* 64, 156-158. https://doi.org/10.1016/j.marpol.2015.11.018. Accessed 13 January 2021.

Peng, G., Bellerby, R., Zhang, F., Sun, X. and Li, D. (2020). The ocean's ultimate trashcan: Hadal trenches as major depositories for plastics pollution. Water Research 168, 15121. https://doi.org/10.1016/j.watres.2019.115121. Accessed 13 January 2021.

Peng, L., Du, D., Qi, H., Lan, C.Q., Yu, H. and Ge, C. (2020). Micro- and nano-plastics in marine environment: Source, distribution and threats – a review. *Science of The Total Environment* 698, 134254. https://doi.org/10.1016/j.scitotenv.2019.134254. Accessed 13 January 2021.

Peng, X., Dasgupta, S., Zhong, G., Du, M., Xu, H., Chen, M. *et al.* (2019). Large debris dumps in the northern South China Sea. *Marine Pollution Bulletin* 142, 164-168. https://doi.org/10.1016/j.marpolbul.2019.03.041. Accessed 13 January 2021.
Penn, J. (2015). Values for recreational beach quality in Oahu, Hawaii. *Marine Resource Economics* 31, 47-62. https://doi.org/10.1086/683795. Accessed 13 January 2021.

Peters, C.A. and Bratton, S.P. (2016). Urbanization is a major influence on microplastics ingestion by sunfish in the Brazos River Basin, Central Texas, USA. *Environmental Pollution* 210, 380-387. https://doi.org/10.1016/j.envpol.2016.01.018. Accessed 13 January 2021.

Petrolia, D.P., Penn, J., Quainoo, R., Caffey, R.H. and Fannin, J.M. (2019). Know the beach: Values of beach condition information. *Marine Resource Economics* 34, 331-359. https://doi.org/10.1086/706248. Accessed 13 January 2021.

Pettit, T.N., Grant, G.S. and Whittow, G.C. (1981). Ingestion of plastics by Laysan albatross. *The Auk* 98, 839-841. http://www.jstor.org/stable/4085908. Accessed 13 January 2021.

Pham, C., Rodríguez, Y., Dauphin, A., Carriço, R., Frias, J.P.G.L., Vandeperre, F. *et al.* (2017). Plastics ingestion in oceanic-stage loggerhead sea turtles (*Caretta caretta*) off the North Atlantic subtropical gyre. *Marine Pollution Bulletin* 121(1-2), 222-229. https://doi.org/10.1016/j.marpolbul.2017.06.008. Accessed 13 January 2021.

Picó, Y. and Barceló, D. (2019). Analysis and prevention of microplastics pollution in water: Current perspectives and future directions. *ACS Omega* 4, 4, 6709-6719. https://doi.org/10.1021/acsomega.9b00222. Accessed 13 January 2021.

Piehl, S., Leibner, A., Loder, M.G., Dris, R., Bogner, C. and Laforsch, C. (2018). Identification and quantification of macro- and microplastics on an agricultural farmland. *Scientific Reports* 8, 17950. https://doi.org/10.1038/s41598-018-36172-y. Accessed 13 January 2021.

PlasticsEurope (2019). *Plastics – The Facts 2019*. An Analysis of European Plastics *Production, Demand and Waste Data* https://www.plasticseurope.org/application/files/9715/7129/9584/FINAL_web_version_Plastics_th. Accessed 13 January 2021.

Plastics Industry Association (2018). *Position Paper on Degradable Additives*. https://www.plasticsindustry.org/sites/default/files/2018%20PLASTICS%20-%20Position%20Paper%20on%20Degradable%20Additives.pdf. Accessed 13 January 2021.

Poppi, L., Zaccaroni, A., Pasotto, D. and Mazzariol, S. (2012). Post-mortem investigations on a leatherback turtle *Dermochelys coriacea* stranded along the Northern Adriatic coastline. *Diseases of Aquatic Organisms* 100, 71-76. https://doi.org/10.3354/dao02479. Accessed 13 January 2021.

Porter, A., Lyons, B.P., Galloway, T.S. and Lewis, C. (2018). Role of marine snows in microplastics fate and bioavailability. *Environmental Science and Technology* 52(12), 7111-7119. https://doi.org/10.1021/acs.est.8b01000. Accessed 13 January 2021.

Porter, A., Smith, K.E. and Lewis, C. (2019). The sea urchin *Paracentrotus lividus* as a bioeroder of plastic. *Science of The Total Environment* 693, 133621. https://doi.org/10.1016/j.scitotenv.2019.133621. Accessed 13 January 2021.

Posen, I.D., Jramillo, P., Landis, A.E. and Griffin, W.M. (2017). Greenhouse gas mitigation for U.S. plastics production: Energy first, feedstocks later. *Environmental Research Letters* 12, 034024. https://iopscience.iop.org/article/10.1088/1748-9326/aa60a7/meta. Accessed 13 January 2021.

Pozo, K., Urbina, W., Gómez, V., Torres, M., Nuñez, D., Přibylová, P. *et al.* (2020). Persistent organic pollutants sorbed in plastic resin pellet – "Nurdles" from coastal areas of Central Chile. *Marine Pollution Bulletin* 151, 110786. https://doi.org/10.1016/j.marpolbul.2019.110786. Accessed 13 January 2021.

Prata, J.C. (2018). Airborne microplastics: Consequences to human health? *Environmental Pollution* 234, 115-126. https://doi.org/10.1016/j.envpol.2017.11.043. Accessed 13 January 2021.

Prata, J.C., da Costa, J.P., Lopes, I., Duarte, A.C. and Rocha-Santos, T. (2019a). Effects of microplastics on microalgae populations: A critical review. *Science of The Total Environment* 665, 400-405. https://doi.org/10.1016/j.scitotenv.2019.02.132. Accessed 13 January 2021.

Prata, J.C., da Costa, J.P. Duarte, A.C. and Rocha-Santos, R. (2019b). Methods for sampling and detection of microplastics in water and sediment: A critical review. *TrAC Trends in Analytical Chemistry* 110, 150-159. https://doi.org/10.1016/j.trac.2018.10.029. Accessed 13 January 2021.

Prata, J.C., da Costa, J.P. Lopes, I., Duarte, A.C. and Rocha-Santos, T. (2020a). Environmental exposure to microplastics: An overview on possible human health effects. *Science of The Total Environment* 702, 13445. https://doi.org/10.1016/j.scitotenv.2019.134455. Accessed 13 January 2021.

Prata, J.C., Silva, A.L.P., Walker, T.R., Duarte, A.C., and Rocha-Santos, T. (2020b). COVID-19 pandemic repercussions on the use and management of plastics. *Environmental Science and Technology* 54(13), 7760-7765. https://doi.org/10.1021/acs.est.0c02178. Accessed 13 January 2021.

Prata J.C., Castro J.L., da Costa J.P., Cerqueira M., Duarte A.C., and Rocha-Santos T. (2021). Airborne microplastics. In *Handbook of Microplastics in the Environment*. Rocha-Santos, T., Costa, M. and Mouneyrac, C. (eds.). Cham:

Springer. 1-25. https://doi.org/10.1007/978-3-030-10618-8_37-2. Accessed 13 January 2021.

Primpke, S., Dias, P.A. and Gerdts, G. (2019). Automated identification and quantification of microfibres and microplastics. *Analytical Methods* 11, 2138-2147. https://doi.org/10.1039/C9AY00126C. Accessed 13 January 2021.

Provencher, J.F., Bond, A.L. and Mallory, M.L. (2015). Marine birds and plastic debris in Canada: A national synthesis and a way forward. *Environmental Reviews* 23, 1-13. https://doi.org/10.1139/er- 2014-0039. Accessed 13 January 2021.

Provencher, J.F., Bond, A.L., Avery-Gomm, S., Borrelle, S., Rebolledo, E.L.B., Hammer, S. *et al.* (2017). Quantifying ingested debris in marine megafauna: A review and recommendations for standardization. *Analytical Methods* 9, 1454. https://doi.org/10.1039/c6ay02419j. Accessed 13 January 2021.

Provencher, J.F., Vermaire, J.C., Avery-Gomm, S., Braune, B.M. and Mallory, M.L. (2018). Garbage in guano? Microplastic debris found in faecal precursors of seabirds known to ingest plastics. *Science of The Total Environment* 644, 1477-1484. https://doi.org/10.1016/j.scitotenv.2018.07101. Accessed 13 January 2021.

Provencher, J.F., Borrelle, S.B., Bond, A.L., Lavers, J.L., van Franeker, J.A., Kühn, S. *et al.* (2019). Recommended best practices for plastics and litter ingestion studies in marine birds: Collection, processing, and reporting. *Facets* 4, 111-130. https://doi.org/10.1139/facets-2018-0043. Accessed 13 January 2021.

Purba, N.P., Handyman, D.I.W., Pribadi, T.D., Syakti, A.D., Pranowo, W.S., Harvey, A., and Ihsan, Y. (2019) Marine debris in Indonesia: A review of research and status. *Marine Pollution Bulletin,* 146, 1340144. https://doi.org/10.1016/j.marpolbul.2019.05.057. Accessed 20 June 2021.

Qiang, M., Shen, M. and Xie, H. (2020). Loss of tourism revenue induced by coastal environmental pollution: A length-of-stay perspective. *Journal of Sustainable Tourism,* 28(4): 550-567. https://doi.org/10.1080/09669582.2019.1684931. Accessed 13 January 2021.

Qiao, R., Deng, Y., Zhang S., Wolosker, M.B., Zhu, Q., Ren, H. *et al.* (2019). Accumulation of different shapes of microplastics initiates intestinal injury and gut microbiota dysbiosis in the gut of zebrafish. *Chemosphere* 236, 124334. https://doi.org/10.1016/j.chemosphere.2019.07.065. Accessed 13 January 2021.

Quina, M.J., Bontemp, E., Bogush, A., Schlumberger, S., Weibel, G., Braga, R. *et al.* (2018). Technologies for the management of MSW incineration ashes from gas cleaning: New perspectives on recovery of secondary raw materials and circular economy. *Science of The Total Environment* 635, 526-542. https://doi.org/10.1016/j.scitotenv.2018.04.150. https://doi.org/10.1016/j.scitotenv.2018.04.150. Accessed 13 January 2021

Radetić, M. (2013). Functionalization of textile materials with TiO2 nanoparticles. *Journal of Photochemistry and Photobiology C: Photochemistry Reviews* 16, 62-76. https://doi.org/10.1016/j.jphotochemrev.2013.04.002. Accessed 13 January 2021.

Ragaert, K., Delva, L., and Van Geem, K. (2017). Mechanical and chemical recycling of solid plastic waste. *Waste Management* 69, 24-58. https://pubmed.ncbi.nlm.nih.gov/28823699/. Accessed 25 May 2021.

Ragusa, A., Svelato, A., Santacroce, C., Catalano, P., Notarstefano, V., Carneval, O. *et al.* (2021) Plasticenta: First evidence of microplastics in human placenta. *Environment International* 146, 106274. https://doi.org/10.1016/j.envint.2020.106274. Accessed 25 May 2021.

Rahimi, A. and García, J.M. (2017). Chemical recycling of waste plastics for new materials production. *Nature Reviews in Chemistry* 1, 46. https://www.researchgate.net/publication/317835495_Chemical_recycling_of_waste_plastics_for_new_materials_production. Accessed 25 May 2021.

Raubenheimer, K. and McIlgorm, A. (2018). Can the Basel and Stockholm conventions provide a global framework to reduce the impact of marine plastics litter? *Marine Policy* 96, 285-290. https://doi.org/10.1016/j.marpol.2018.01.013. Accessed 13 January 2021.

Raubenheimer, K. and Uhro, N. (2020). Rethinking global governance of plastics – the role of industry. *Marine Policy,* 113, 103802. https://doi.org/10.1016/j.marpol.2019.103802. Accessed 13 January 2021.

Rech, S., Borrell, Y. and García-Vazquez, E. (2016). Marine litter as a vector for non-native species: What we need to know. *Marine Pollution Bulletin* 113(1-2), 40-43. https://doi.org/10.1016/j.marpolbul.2016.08.032. Accessed 13 January 2021.

Reddy, M. S., Shaik Basha, Adimurthy, S. and Ramachandraiah, G. (2006). Description of the small plastics fragments in marine sediments along the Alang-Sosiya ship-breaking yard, India. *Estuarine, Coastal and Shelf Science* 68(3-4), 656-660. https://doi.org/10.1016/j.ecss.2006.03.018. Accessed 20 June 2021.

Redondo-Hasselerharm, P.E., Gort, G., Peeters, E.T.H.M. and Koelmans, A.A. (2020). Nano- and microplastics affect the composition of freshwater benthic communities in the long term. *Science Advances* 6(5), eaay4054. http://doi.org/10.1126/sciadv.aay4054. Accessed 13 January 2021.

Rehn, A.C., Barnett, A.J. and Wiber, M.G. (2018). Stabilizing risk using public participatory GIS: A case study on mitigating marine debris in the Bay of Fundy, Southwest New Brunswick, Canada. *Marine Policy* 96, 264-269. https://doi.org/10.1016/j.marpol.2017.11.033. Accessed 13 January 2021.

Reichert, J., Arnold, A.L., Hoogenboom, M.O., Schubert, P. and Wilke, T. (2019). Impacts of microplastics on growth and health of hermatypic corals are species-specific. *Environmental Pollution* 254, Part B, 113074. https://doi.org/10.1016/j.envpol.2019.113074. Accessed 13 January 2021.

Reid, P. C., Fischer, A.C., Lewis-Brown, E., Meredith, M.P., Sparrow, M *et al.* (2009). Impacts of the oceans on climate change. *Advances in Marine Biology* 56, 1-150. https://doi.org/10.1016/S0065-2881(09)56001-4. Accessed 15 February 2021.

Reinert, T.R., Spellman A.C. and Bassett, B.L. (2017). Entanglement in and ingestion of fishing gear and other marine debris by Florida manatees, 1993 to 2012. *Endangered Species Research* 32, 415-427. https://doi.org/10.3354/esr00816. Accessed 13 January 2021.

Reisser, J., Slat, B., Noble, K., du Plessis, K., Epp, M., Proiett, M. *et al.* (2015). The vertical distribution of buoyant plastics at sea: An observational study in the North Atlantic Gyre. *Biogeosciences* 12:1249- 1256. https://doi.org/10.5194/bg-12-1249-2015. Accessed 13 January 2021.

Remy, F., Collard, F., Gilbert, B., Compoére, P., Eppe, G. and Lepoint, G. (2015). When microplastic is not plastic: The ingestion of artificial cellulose fibres by macrofauna living in seagrass macrophytodetritus. *Environmental Science and Technology* 49(18), 11158-11166. https://doi.org/10.1021/acs.est.5b02005. Accessed 13 January 2021

Renzi, M., Blaškovic, A., Bernardi, G. and Russo, G.F. (2018). Plastic litter transfer from sediments towards marine trophic webs: A case study on holothurians. *Marine Pollution Bulletin*, 135, 376-385. https://doi.org/10.1016/j.marpolbul.2018.07.038. Accessed 13 January 2021.

Renzi, M., Grazioli, E. and Blašković, A. (2019). Effects of different microplastic types and surfactant-microplastic mixtures under fasting and feeding conditions: A case study on *Daphnia magna*. *Bulletin of Environmental Contamination and Toxicology* 103(3), 367-373. https://doi.org/10.1007/s00128-019-02678-y. Accessed 13 January 2021.

Reynolds, C. and Ryan, P.G. (2018). Micro-plastic ingestion by waterbirds from contaminated wetlands in South Africa. *Marine Pollution Bulletin* 126, 330-333. https://doi.org/10.1016/j.marpolbul.2017.11.021. Accessed 13 January 2021.

Riascos, J.M., Valencia, N., Peña, E.J. and Cantera, J.R. (2019). Inhabiting the technosphere: The encroachment of anthropogenic marine litter in Neotropical mangrove forests and its use as habitat by macrobenthic biota. *Marine Pollution Bulletin* 142, 559-568. https://doi.org/10.1016/j.marpolbul.2019.04.010. Accessed 13 January 2021.

Richards, Z.T. and Beger, M. (2011). A quantification of the standing stock of macro-debris in Majuro lagoon and its effect on hard coral communities. *Marine Pollution Bulletin* 62(8), 1693-1701. https://doi.org/10.1016/j.marpolbul.2011.06 003. Accessed 13 January 2021.

Richardson, K., Asmutis-Silvia, R., Drinkwin, J., Gilardi, K.V.K., Giskes, I., Jones, G. *et al.* (2019). Building evidence around ghost gear: Global trends and analysis for sustainable solutions at scale. *Marine Pollution Bulletin* 138, 222-229. https://doi.org/10.1016/j.marpolbul.2018.11.031. Accessed 13 January 2021.

Rios Mendoza, L.M. and Balcer, M. (2019). Microplastics in freshwater environments: A review of quantification assessment. *TrAC Trends in Analytical Chemistry* 113, 402-408. https://doi.org/10.1016/j.trac.2018.10 020. Accessed 13 January 2021.

Rivers, M.L., Gwinnett, C. and Woodall, L. (2019). Quantification is more than counting: Actions required to accurately quantify and report isolated marine microplastics. *Marine Pollution Bulletin* 139, 100-104. https://doi.org/10.1016/j.marpolbul.2018.12 024. Accessed 13 January 2021.

Rochman, C.M., Hentschel, B.T. and Teh, S.J. (2014a). Long-term sorption of metals is similar among plastic types: Implications for plastic debris in aquatic environments. *PLoS ONE* 9(1), e85433. https://doi.org/10.1371/journal.pone.0085433. Accessed 13 January 2021.

Rochman, C.M., Kurobe, T., Flores, I. and Teh, S.J. (2014b). Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment. *Science of The Total Environment* 493, 656-661. https://doi.org/10.1016/j.scitotenv.2014.06.051. Accessed 13 January 2021.

Rochman, C.M., Tahir, A., Williams, S.L., Baxa, D.V., Lam, R., Miller, J.T. *et al.* (2015) Anthropogenic debris in seafood: Plastic debris and fibres from textiles in fish and bivalves sold for human consumption. *Scientific Reports* 5, 14340. https://doi.org/10.1038/srep14340. Accessed 13 January 2021.

Rochman, C.M., Browne, M.A., Underwood, A.J., van Franeker, J.A., Thompson, R.C. and Amaral- Zettler, L.A. (2016a). The ecological impacts of marine debris: Unraveling the demonstrated evidence from what is perceived. *Ecology* 97(2), 302-312. https://doi.org/10.1890/14-2070.1. Accessed 13 January 2021.

Rochman, C.M., Cook, A.M. and Koelmansk, A.A. (2016b). Plastic debris and policy: Using current scientific understanding to invoke positive change. *Environmental Toxicology and Chemistry* 35(7), 1617-1626. https://doi.org/10.1002/etc.3408. Accessed 13 January 2021.

Rochman, C.M., Brookson, C., Bikker, J., Djuric, N., Earn, A., Bucci, K. *et al.* (2019). Rethinking microplastics as a diverse contaminant suite. *Environmental Toxicology and Chemistry* 38, 703-711. https://doi.org/10.1002/etc.4371. Accessed 13 January 2021.

Rodrigues, D., Antunes, A., Otero, V., Sobral, P. and Costa, M.H. (2020). Distribution patterns of microplastics in seawater surface at a Portuguese estuary and marine park. *Frontiers in Environmental Science* 8, 582217. https://doi.org/10.3389/fenvs.2020.582217. Accessed January 2021

Roman, L., Lowenstine, L., Parsley, L.M., Wilcox, C., Hardesty, B.D., Gilardi, K. *et al.* (2019). Is plastic ingestion in birds as toxic as we think? Insights from a plastic feeding experiment. *Science of The Total Environment* 665, 660-667. https://doi.org/10.1016/j.scitotenv.2019.02.184. Accessed 13 January 2021.

Ronda, A.C., Arias, A.H., Oliva, A.L. and Marcovecchio, J.E. (2019). Synthetic microfibres in marine sediments and surface seawater from the Argentinean continental shelf and a Marine Protected Area. *Marine Pollution Bulletin* 149, 110618. https://doi.org/10.1016/j.marpolbul.2019.110618. Accessed 13 January 2021.

Roos, S., Jönsson, C., Posner, S., Arvidsson, R. and Svanström, M. (2019). An inventory framework for inclusion of textile chemicals in life cycle assessment. *International Journal of Life Cycle Assessment* 24(5), 838-847. https://doi.org/10.1007/s11367-018-1537-6. Accessed 13 January 2021.

Rosenkranz, P., Chaudhry, Q., Stone, V. and Fernandes, T.F. (2009). A comparison of nanoparticle and fine particle uptake by Daphnia magna. *Environmental Toxicology and Chemistry* 28(10), 2142-2149. https://doi.org/10.1897/08-559.1. Accessed 13 January 2021.

Royer, S.-J. and Deheyn, D.D. (2019). The technological challenges of dealing with plastics in the environment. *Marine Technology Society Journal* 53, 13-20. https://assets.theoceancleanup.com/app/uploads/2019/12/The_Technological_Challenges_of_Dealing-with-Plastic-in-the-Environment.pdf. Accessed 13 January 2021.

Royer, S.-J., Ferrón, S., Wilson, S.T. and Karl, D.M. (2018). Production of methane and ethylene from plastic in the environment. *PLoS ONE*, 13(8), e0200574. https://doi.org/10.1371/journal.pone.0200574. Accessed 13 January 2021.

Rubio, L., Marcos, R. and Hernández, A. (2020). Potential adverse health effects of ingested micro- and nanoplastics on humans. Lessons learned from *in vivo* and *in vitro* mammalian models. *Journal of Toxicology and Environmental Health (Part B)* 23(2), 51-68. https://doi.org/10.1080/10937404.2019.1700598. Accessed 13 January 2021.

Rüdel, H., Müller, J., Quack, M. and Klein, R. (2012). Monitoring of hexabromocyclododecane diastereomers in fish from European freshwaters and estuaries. *Environmental Science and Pollution Research* 19, 772-783. https://doi.org/10.1007/s11356-011-0604-3. Accessed 13 January 2021.

Ruiz, M. and Stankiewicz, M. (2019). Five years since the 2013 HELCOM Ministerial Declaration. In *The Handbook of Environmental Chemistry*. Berlin, Heidelberg: Springer. 1-26. https://link.springer.com/chapter/10.1007/698_2019_378. Accessed 13 January 2021.

Rummel, C.D., Löder, M.G.J., Fricke, N.F., Lang, T., Griebeler, E-M., Janke, M. *et al.* (2016). Plastic ingestion by pelagic and demersal fish from the North Sea and Baltic Sea. *Marine Pollution Bulletin* 102, 134-141. https://doi.org/10.1016/j.marpolbul.2015.11.043. Accessed 13 January 2021.

Rummel, C.D., Jahnke, A., Gorokhova, E., Kühnel, D. and Schmitt-Jansen, M. (2017). Impacts of biofilm formation on the fate and potential effects of microplastics in the aquatic environment. *Environmental Science and Technology Letters* 4(7), 258-267. https://doi.org/10.1021/acs.estlett.7b00164. Accessed 13 January 2021.

Ryan, P.G. (2014). Litter survey detects the South Atlantic 'garbage patch'. *Marine Pollution Bulletin* 79, 220-224. https://doi.org/10.1016/j.marpolbul.2013.12.010. Accessed 13 January 2021.

Ryan, P.G. (2015). A brief history of marine litter research. In *Marine Anthropogenic Litter*. Bergmann, M., Gutow, L. and Klages. M (eds.). Cham: Springer. 1-25. https://doi.org/10.1007/978-3-319-16510-3_1. Accessed 13 January 2021.
Ryan, P.G (2018). Entanglement of birds in plastics and other synthetic materials. *Marine Pollution Bulletin* 135, 159-164. https://doi.org/10.1016/j.marpolbul.2018.06.057. Accessed 13 January 2021.

Ryan, P.G., Dilley, B.J., Ronconi, R.A. and Connan, M. (2019). Rapid increase in Asian bottles in the South Atlantic Ocean indicates major debris inputs from ships. *Proceedings of the National Academy of Sciences* 116 (42), 20892-20897. https://doi.org/10.1073/pnas.1909816116. Accessed 13 January 2021.

Ryan, P.G., Suaria, G., Perolda, V., Pierucci, A., Bornman, T.G. and Aliani, S. (2020). Sampling microfibres at the sea surface: The effects of mesh size, sample volume and water depth. *Environmental Pollution* 258, 113413. https://doi.org/10.1016/j.envpol.2019.113413. Accessed 13 January 2021

Ryberg, M.W., Hauschild, M.Z., Wang, F., Averous-Monnery, S., and Laurent, A. (2019): Global environmental losses of plastics across their value chains. *Resources, Conservation and Recycling* 151, 104459. https://orbit.dtu.dk/en/publications/global-environmental-losses-of-plastics-across-their-value-chains. Accessed 20 June 2021.

Sadri, S.S. and Thompson, R.C. (2014). On the quantity and composition of floating plastics debris entering and leaving the Tamar Estuary, Southwest England. *Marine Pollution Bulletin* 81(1), 55-60. https://doi.org/10.1016/j.marpolbul.2014.02.020. Accessed 13 January 2021.

Saley, A.M., Smart, A.C., Bezerra, M.F., Burnham, T.L.U., Capece, L.R., Lima, L.F.O. *et al.* (2019). Microplastic accumulation and biomagnification in a coastal marine research situated in a sparsely populated area. *Marine Pollution Bulletin* 146, 54-59. https://doi.org/10.1016/j.marpolbul.2019.05.065. Accessed 13 January 2021.

Saliu, F., Montano, S., Leioni, B., Lasagni, M. and Galli, P. (2019). Microplastics as a threat to coral reef environments: Detection of phthalate esters in neuston and scleractinian corals from the Faafu Atoll, Maldives. *Marine Pollution Bulletin* 142, 234-241. https://doi.org/10.1016/j.marpolbul.2019.03.043. Accessed 13 January 2021.

Sanchez-Vidal, A., Thompson, R.C., Canals, M., and de Haan, W.P. (2018). The imprint of microfibres in southern European deep seas. *PLoS ONE* 13, e0207033. https://doi.org/10.1371/journal.pone.0207033. Accessed 13 January 2021

Santana, M.F.M., Moreira, F.T. and Turra, A. (2017). Trophic transference of microplastics under a low exposure scenario: Insights on the likelihood of particles cascading along marine food-webs. *Marine Pollution Bulletin* 121(1-2), 154-59. https://doi.org/10.1016/j.marpolbul.2017.05.061. Accessed 13 January 2021.

Santana, M.F.M. and Turra, A. (2020). Toxicity of microplastics in the marine environment. In *A Handbook of Environmental Toxicology: Human Disorders and Ecotoxicology*. D'Mello, J.P.F. (ed.). London: CAB International. 436-453. https://www.cabi.org/environmentalimpact/ebook/20193493760. Accessed 13 January 2021.

Santos, I.R., Friedrich, A.C., Wallner-Kersanach, M. and Fillmann, G. (2005). Influence of socio- economic characteristics of beach users on litter generation. *Ocean and Coastal Management* 48(9-10), 742e752. https://doi.org/10.1016/j.ocecoaman.2005.08.006. Accessed 13 January 2021.

Santos, R.G., Andrades, R., Boldrini, M.A. and Martins, A.S. (2015). Debris ingestion by juvenile marine turtles: An underestimated problem. *Marine Pollution Bulletin* 93(1-2), 37-43. https://doi.org/10.1016/j.marpolbul.2015.02.022. Accessed 13 January 2021.

SAPEA (Science Advice for Policy by European Academies) (2019). *A Scientific Perspective on Microplastics in Nature and Society*. https://doi.org/10.26356/microplastics. Accessed 23 May 2020.

Sauret, C., Severin, T., Vétion, G., Guigue, C., Goutx, M., Pujo-Pay, M. *et al.* (2014). 'Rare biosphere' bacteria as key phenanthrene degraders in coastal seawaters. *Environmental Pollution* 194, 246-253. https://doi.org/10.1016/j.envpol.2014.07.024. Accessed 23 May 2020.

Sbrana, A., Valente, T., Scacco, U., Bianchi, J., Silvestri, J., Palazzo, L. *et al.* (2017). Spatial variability and influence of biological parameters on microplastic ingestion by *Boops boops* (L.) along the Italian coasts (Western Mediterranean Sea). *Environmental Pollution* 263 (Part A), 114429. https://doi.org/10.1016/j.envpol.2020.114429. Accessed 13 January 2021.

Scheld, A.M., Bilkovic, D.M. and Havens, K.J., (2016). The dilemma of derelict gear. *Scientific Reports* 6, 19671. https://doi.org/10.1038/srep19671. Accessed 13 January 2021.

Schmidt, C., Krauth, T. and Wagner, S. (2017). Export of plastics debris by rivers into the sea. *Environmental Science and Technology* 51(21), 12246-12253. https://doi.org/10.1021/acs.est.7b02368. Accessed 13 January 2021.

Schneider, F., Parsons, S., Clift, S., Stolte, A. and McManus, M.C. (2018). Collected marine litter – A growing waste challenge. *Marine Pollution Bulletin* 128, 162-174. https://doi.org/10.1016/j.marpolbul.2018.01.011. Accessed 13 January 2021.

Scholin, C.A., Birch, J., Jensen, S., Marin, R., Massion, E., Pargett, D. *et al.* (2017). The quest to develop ecogenomic sensors: A 25-year history of the Environmental Sample Processor (ESP) as a case study. *Oceanography* 30, 427. https://doi.org/10.5670/oceanog.2017.427. Accessed 13 January 2021.

Schulz, M., van Loon, W., Fleet, D.M., Baggelaar, P. and der Meulene, E. (2017). OSPAR standard method and software for statistical analysis of beach litter data. *Marine Pollution Bulletin* 122(1-2), 166-175. https://doi.org/10.1016/j.marpolbul.2017.06.045. Accessed 13 January 2021.

Schulz, M., Walvoort, D.J.J., Barry, J., Fleet, D.M. and van Loon, W.G.M. (2019). Baseline and power analyses for the assessment of beach litter reductions in the European OSPAR region. *Environmental Pollution* 248, 555-564. https://doi.org/10.1016/j.envpol.2019.02.030. Accessed 13 January 2021.

Schür, C., Rist, S., Baun, A., Mayer, P., Hartmann, N.B. and Wagner, M. (2019). When fluorescence is not a particle: The tissue translocation of microplastics in *Daphnia magna* seems an artefact. *Environmental Toxicology and Chemistry* 38 (7), 1495-1503. https://doi.org/10.1002/etc.4436. Accessed 13 January 2021.

Schuyler, Q.A., Wilcox, C., Townsend, K., Hardesty, B.D. and Marshall, N.J. (2014). Mistaken identity? Visual similarities of marine debris to natural prey items of sea turtles. *BMC Ecology* 14, 14. https://doi.org/10.1186/1472-6785-14-14. Accessed 13 January 2021.

Schuyler, Q.A., Hardesty, B.D., Lawson, T.J., Opie, K. and Wilcox, C. (2018). Economic incentives reduce plastic inputs to the ocean. *Marine Policy* 96, 250-255. https://doi.org/10.1016/j.marpol.2018.02.009. Accessed 13 January 2021.

Schweitzer, J.-P., Gionfra, S., Pantzar, M., Mottershead, D., Watkins, E., Petsinaris, F. *et al.* (2018). *Unwrapped: How Throwaway Plastic is Failing to Solve Europe's Food Waste Problem (And What We Need to Do Instead). A Study by Zero Waste Europe and Friends of the Europe for the Rethink Plastic Alliance.* Brussels: Institute for European Environmental Policy. https://www.foeeurope.org/sites/default/files/materials_and_waste/2018/unwrapped_-_throwaway_plastic_failing_to_solve_europes_food_waste_problem.pdf. Accessed 13 January 2021.

Schymanski, D., Goldbeck, C., Humpf, A.U. and Fürst, P. (2018). Analysis of microplastics in water by micro-Raman spectroscopy: Release of plastic particles from different packaging into mineral water. *Water Research* 129, 154-162. https://doi.org/10.1016/j.watres.2017.11.011. Accessed 13 January 2021.

Science for Environment Policy (2016). *Ship Recycling: Reducing Human and Environmental Impacts.* Thematic Issue 55. Issue produced for the European Commission DG Environment by the Science Communication Unit, UWE, Bristol, UK. http://ec.europa.eu/science-environment-policy https://ec.europa.eu/environment/integration/research/newsalert/pdf/ship_recycling_reducing_human_and_environmental_impacts_55si_en.pdf. Accessed 20 June 2021.

Secretariat of the Basel Convention (2015). *Technical Guidelines on Transboundary Movements of Electrical and Electronic Waste and Used Electrical and Electronic Equipment, in Particular Regarding the Distinction between Waste and Non-waste under the Basel Convention* (UNEP/CHW.12/5/Add.1/Rev.1), http://www.basel.int/Portals/4/download.aspx?d=UNEP-CHW.12-. Accessed 13 January 2021.

Secretariat of the Convention on Biological Diversity (2016a). *Decision Adopted by The Conference of the Parties to The Convention On Biological Diversity XIII/10. Addressing Impacts of Marine Debris and Anthropogenic Underwater Noise on Marine and Coastal Biodiversity* (CBD/COP/DEC/XIII/10). https://www.cbd.int/doc/decisions/cop-13/cop-13-dec-10-en.pdf. Accessed 13 January 2021.

Secretariat of the Convention on Biological Diversity (2016b). *Marine Debris: Understanding, Preventing and Mitigating the Significant Adverse Impacts on Marine and Coastal Biodiversity.* Technical Series No. 83. https://www.cbd.int/doc/publications/cbd-ts-83-en.pdf. Accessed 13 January 2021.

Secretariat of the Stockholm Convention (2020). All POPs listed in the Stockholm Convention. http://chm.pops.int/TheConvention/ThePOPs/ListingofPOPs/tabid/2509/Default.aspx. Accessed 13 January 2021.

Secretariat of the South Pacific Environment Programme (SPREP) (2017). *Report on the Status of Actions Taken on the Implementation of the 2016-2017 Business Plan since COP-8.* https://www.sprep.org/attachments/2017SM28/Waigani%20Convention/WC%20COP-9%20-%20Agenda%204.3.Att.1%20-%20Project%20related%20activities.pdf. Accessed 13 January 2021.

Serra-Gonçalves, A.C., Lavers, J.L. and Bond, A.L. (2019). Global review of beach debris monitoring and future recommendations. *Environmental Science and Technology* 53(21), 12158-12167. https://doi.org/10.1021/acs.est.9b01424. Accessed 13 January 2021.

Setälä, O., Fleming-Lehtinen, V. and Lehtiniemi, M. (2014). Ingestion and transfer of microplastics in the planktonic food web. *Environmental Pollution* 185, 77-83. https://doi.org/10.1016/j.envpol.2013.10 013. Accessed 13 January 2021.

Setälä, O., Granberg, M., Hassellöv, M., Karlsson, T., Lehtiniemi, M., Mattsson, K. *et al.* (2019). *Monitoring of Microplastics in the Marine Environment: Changing Directions towards Quality Controlled Tailored Solutions.* Nordic Council of Ministers. http://norden.diva-portal.org/smash/get/diva2:1373249/FULLTEXT02.pdf. Accessed 13 January 2021.

Sgier, L., Freimann, R., Zupanic, A. and Kroll, A. (2016). Flow cytometry combined with viSNE for the analysis of microbial biofilms and detection of microplastics. *Nature Communications* 7, 11587. http://doi.org/10.1038/ncomms11587. Accessed 13 January 2021.

Shaw, E. and Turner, A. (2019). Recycled electronic plastic and marine litter. *Science of The Total Environment* 694, 133644. https://doi.org/10.1016/j.scitotenv.2019.133644. Accessed 13 January 2021.

Shen, M., Huang, W., Chen, M., Song, B., Zeng, G. and Zhang, Y. (2020). (Micro)plastic crisis: Un-ignorable contribution to global greenhouse gas emissions and climate change. *Journal of Cleaner Production* 254, 120138. https://doi.org/10.1016/J.JCLEPRO.2020.120138. Accessed 13 January 2021.

Sherman, P. and van Sebille, E. (2016). Modeling marine surface microplastic transport to assess optimal removal locations. *Environmental Research Letters* 11, 014006. https://doi.org/10.1088/1748-9326/11/1/014006. Accessed 13 January 2021.

Shevealy, S., Courtney, K. and Parks, J.E. (2012). *The Honolulu Strategy: A Global Framework for Prevention and Management of Marine Debris*. UNEP and United States National Oceanic and Atmospheric Administration (NOAA). https://wedocs.unep.org/bitstream/handle/20.500.11822/10670/Honolulu%20strategy.pdf?sequence=1&isAllowed=y. Accessed 13 January 2021.

Shim, W.J., Hong, S.H. and Eo, S. (2017). Identification methods in microplastic analysis: A review. *Analytical Methods* 9, 1384-1391. https://doi.org/10.1039/C6AY02558G. Accessed 13 January 2021.

Shu, C. (2018). The Ocean PM2.5. *Land Resource* 23-27 [in Chinese].

Sijtsema, S.J., Onwezen, M., Reinders, M.J., Dagevos, H., Partanen, A. and Meeusen, M. (2016). Consumer perceptions of bio-based – an exploratory study in 5 European countries. *NJAS-Wageningen Journal of Life Sciences* 77, 61-69. https://doi.org/10.1016/j.njas.2016.03.007. Accessed 13 January 2021.

Silva, M.S.S., Oliveira, M., Lopéz, D., Martins, M., Figueira, E. and Pires, A. (2020). Do nanoplastics impact the ability of the polychaeta Hediste diversicolor to regenerate? *Ecological Indicators* 110, 105921. https://doi.org/10.1016/j.ecolind.2019.105921. Accessed 13 January 2021.

Sogin, M. L., Morrison, H. G., Huber, J. A., Welch, D. M., Huse, S. M., Neal, P. R. *et al.* (2006). Microbial diversity in the deep sea and the underexplored "rare biosphere". *Proceedings of the National Academy of Sciences* 103, 12115-12120. https://doi.org/10.1073/pnas.0605127103. Accessed 23 May 2020

Song, Y.K., Hong, S. H., Eo, S., Jang, M., Han, G. M., Isobe, A., and Shim, W. J. (2018). Horizontal and vertical distribution of microplastics in Korean coastal waters. *Environmental Science and Technology* 52(21), 12188-12197. https://doi.org/10.1021/acs.est.8b04032. Accessed 13 January 2021.

Spedicato, M.T., Zupa, W., Carbonara, P., Fiorentino, F., Follesa, M.C., Galgani, F. *et al.* (2019). Spatial distribution of marine macro-litter on the seafloor in the northern Mediterranean Sea: The MEDITS initiative. *Scientia Marina* 83, 267. https://doi.org/10.3989/scimar.04987.14a. Accessed 13 January 2021.

Spierling, S., Knüpffer, E., Behsen, H., Mudersbach, M., Krieg, H., Springer, S. *et al.* (2018). Bio-based plastics – a review of environmental, social and economic impact assessments. *Journal of Cleaner Production* 185, 476-491. https://doi.org/10.1016/j.jclepro.2018.03.014. Accessed 13 January 2021.

Stachowitsch, M. (2020). *The Beachcomber's Guide to Marine Debris*. Springer Nature. https://www.springer.com/gp/book/9783319907277. Accessed 13 January 2021.

Stanton, T., Johnson, M., Nathanail, P., Gomes, R.L., Needham, T. and Burson, A. (2019a). Exploring the efficacy of Nile red in microplastics quantification: A costaining approach. *Environmental Science and Technology Letters* 6(10), 606-611. https://doi.org/10.1021/acs.estlett.9b00499. Accessed 13 January 2021.

Stanton, T., Johnson, M., Nathanail, P., MacNaughtan, W. and Gomes, R.L. (2019b). Freshwater and airborne textile fibre populations are dominated by 'natural', not microplastic, fibres. *Science of The Total Environment* 666, 377-389. https://doi.org/10.1016/j.scitotenv.2019.02.278. Accessed 13 January 2021.

Stanton, T., Kay, P., Johnson M., Ka Shun Chan, F., Gomes, R.L., Hughes, J. *et al.* (2020). It's the product not the polymer: Rethinking plastic pollution. *WIREs Water* 8(1), e1490. https://dx.doi.org/10.1002/wat2.1490. Accessed 13 January 2021.

Stark, M. (2019). Letter to the editor regarding "Are We Speaking the Same Language? Recommendations for a definition and categorization framework for plastic debris". *Environmental Science and Technology* 53(9), 4677-4677. https://doi.org/10.1021/acs.est.9b01360. Accessed 13 January 2021.

Statista (2019). Global generation of plastic MSW by region 2018 (published by Tiseo, I. 24 June 2019). https://www.statista.com/statistics/1016756/generated-plastic-municipal-solid-waste-globally-by-region/. Accessed 11 January 2021

Statista (2021a). Global plastic market size 2016-2028 (published by Tiseo, I. 24 June 2021). https://www.statista.com/statistics/1060583/global-market-value-of-plastic/. Accessed 12 September 2021.

Statista (2021b). Cumulative plastic production volume worldwide from 1950 to 2050. Published by Tiseo, I. 27 January 2020. https://www.statista.com/statistics/1019758/plastics-production-volume-worldwide/. Accessed 11 February 2021.

Steer, M., Cole, M., Thompson, R.C. and Lindeque, P.K. (2017). Microplastic ingestion in fish larvae in the western English Channel. *Environmental Pollution,* 226, 250-259. https://doi.org/10.1016/j.envpol.2017.03 062. Accessed 13 January 2021.

Stehel, V., Dvořák, J., Wittlingerová, Z. and Petruželková, A. (2019). Economic contradictions of the waste-to-energy concept and emissions reduction plan (case study, Czech Republic). *Energy Sources, Part A: Recovery, Utilization, and Environmental Effects* 41(13),1622-1629. https://doi.org/10.1080/15567036.2018.1549137. Accessed 13 January 2021.

Stelfox, M., Hudgins, J. and Sweet, M. (2016). A review of ghost gear entanglement amongst marine mammals, reptiles and elasmobranchs. *Marine Pollution Bulletin* 111(102), 6-17. https://doi.org/10.1016/j.marpolbul.2016.06.034. Accessed 13 January 2021.

Stewart, P.S. and Costerson, J.W. (2001). Antibiotic resistance of bacteria in biofilms. *The Lancet* 358, 135-38. https://doi.org/10.1016/S0140-6736(01)05321-1. Accessed 13 January 2021.

Strand, J., Feld, L., Murphy, F., Mackevica, A. and Hartmann, N.B. (2018). *Analysis of Microplastic Particles in Danish Drinking Water*. Aarhus University, DCE – Danish Centre for Environment and Energy. Scientific Report No. 291. http://dce2.au.dk/pub/SR291.pdf. Accessed 13 January 2021.

Straub, S., Hirsch, P.E. and Burkhardt-Holm, P. (2017). Biodegradable and petroleum-based microplastics do not differ in their ingestion and excretion but in their biological effects in a freshwater invertebrate *Gammarus fossarum*. *International Journal of Environmental Research and Public Health* 14(7), 774. https://doi.org/10.3390/ijerph14070774. Accessed 13 January 2021.

Su, L., Xue, Y., Li, L., Yang, D., Kolandhasamy, P., Li, D. *et al.* (2016). Microplastics in Taihu Lake. China. *Environmental Pollution* 216, 711-719. https://doi.org/10.1016/j.envpol.2016.06.036. Accessed 13 January 2021.

Suaria, G., Avio, C.G., Mineo, A., Lattin, G.L., Magaldi, M.G., Belmonte, G. *et al.* (2016). The Mediterranean Plastic Soup: Synthetic polymers in Mediterranean surface waters. *Scientific Reports* 6, 37551. https://doi.org/10.1038/srep37551. Accessed 13 January 2021.

Suaria, G., Achtypi, A., Perold, V., Lee, J.R., Peirucci, A., Bornmans, T.G., Aliani, S. and Ryan, P.G. (2020). Microfibers in oceanic surface waters: A global characterization. *Science Advances* 6(23), eaay8493. http://doi.org/10.1126/sciadv.aay8493. Accessed 17 August 2021.

Suedel, B., Boraczek, J., Peddicord, R., Clifford, P. and Dillon, T. (1994). Trophic transfer and biomagnification potential of contaminants in aquatic ecosystems. In *Reviews of Environmental Contamination and Toxicology (Continuation of Residue Reviews)* vol. 136. Ware, G.W. (ed.). New York: Springer. 21-89. https://doi.org/10.1007/978-1-4612-2656-7_2. Accessed 13 January 2021.

Sullivan, M., Evet, S., Straub, P., Reding, M., Robinson, N., Zimmermann, E. *et al.* (2019). Identification, recovery, and impact of ghost fishing gear in the Mullica river-great bay estuary (New Jersey, USA): Stakeholder-driven restoration for smaller-scale systems. *Marine Pollution Bulletin* 138, 37-48. https://doi.org/10.1016/j.marpolbul.2018.10.058. Accessed 13 January 2021.

Sun, J., Dai, X., Wang, Q., van Loosdrecht, M.C. and Ni, B.J. (2019). Microplastics in wastewater treatment plants: Detection, occurrence and removal. *Water Research* 152, 21-37. https://doi.org/10.1016/j.watres.2018.12.050. Accessed 13 January 2021.

Sun, Q., Ren, S.-Y. and Ni, H.-G. (2020). Incidence of microplastics in personal care products: An appreciable part of plastic pollution. *Science of The Total Environment* 742, 140218. https://doi.org/10.1016/j.scitotenv.2020.140218. Accessed 13 January 2021.

Sundet, J.H., Herzke D. and Jenssen, M. (2016). *Svalvards Miljøvernfond. Forekomst og kilder i mikroplastikk i sediment, og konsekvenser for bunnlevende fisk og evertebrater på Svalbard*. RIS-prosjekt nr. 10495. https://pame.is/document-library/desktop-study-on-marine-litter-library/additional-documents/annexes-literature-from-the-desktop-study/table-2-4-abundance-of-microplastics-observed-in-sediments. Accessed 13 January 2021.

Sussarellu, R., Suquet, M., Thomas, Y., Lambert, C., Fabioux, C., Pernet, M.E.J. *et al.* (2016). Oyster reproduction is affected by exposure to polystyrene microplastics. *Proceedings of the National Academy of Sciences* 113(9), 2430-2435. http://doi.org/10.1073/pnas.1519019113. Accessed 13 January 2021.
Sweet, M.J. and Brown, B.E. (2016). Coral responses to anthropogenic stress in the twenty-first century: An ecophysiological perspective. In *Oceanography and Marine Biology: An Annual Review vol.* 54. Hughes, R.N., Hughes, D.J., Smith, I.P. and Dale, A.C. (eds.). CRC Press. 271-314. https://www.routledge.com/Oceanography-and-Marine-Biology-An-Annual-Review-Volume-54/Hughes-Hughes-Smith-Dale/p/book/9781498747981. Accessed 13 January 2021.

Sweet, M.J. and Bythell, J. (2015). White syndrome in *Acropora muricata*: Nonspecific bacterial infection and ciliate histophagy. *Molecular Ecology* 24(5), 1150-1159. https://doi.org/10.1111/mec.13097. Accessed 13 January 2021.

Sweet, M.J. and Séré, M.G. (2016). Ciliate communities consistently associated with coral diseases. *Journal of Sea Research* 113, 119-131. https://doi.org/10.1016/j.seares.2015.06.008. Accessed 13 January 2021.

Takada, H. and Karapangioti, H.K. (eds.) (2019). *Hazardous Chemicals Associated with Plastics in the Marine Environment. Handbook of Environmental Chemistry vol* 78. Springer Nature Switzerland. https://doi.org/10.1007/978-3-319-95568-1. Accessed 13 January 2021.

Talvitie, J., Mikola, A., Koistinen, A. and Setälä, O. (2017). Solutions to microplastics pollution – Removal of microplastics from wastewater effluent with advanced wastewater treatment technologies. *Water Research* 123, 401-407. https://doi.org/10.1016/j.watres.2017.07.005. Accessed 13 January 2021.

Tanaka, K., Takada, H., Yamashita, R., Mizukawa, K., Fukuwaka, M. and Watanuki, Y. (2013). Accumulation of plastic-derived chemicals in tissues of seabirds ingesting marine plastic. *Marine Pollution Bulletin* 69(1-2), 219-222. https://doi.org/10.1016/j.marpolbul.2012.12.010. Accessed 13 January 2021.

Tanaka, K., Watanuki, Y., Takada, H., Ishizuka, M., Yamashita, R., Kazama, M. *et al.* (2020). *In vivo* accumulation of plastic-derived chemicals into seabird tissues. *Current Biology* 30(4), 723-728. https://doi.org/10.1016/j.cub.2019.12.037. Accessed 13 January 2021.

Tavares, D.C., da Costa, L.L., Rangel, D.F., de Moura, J.F., Zalmon, I.R. and Siciliano, S. (2016). Nests of the brown booby (*Sula leucogaster*) as a potential indicator of tropical ocean pollution by marine debris. *Ecological Indicators* 70, 10-14. https://doi.org/10.1016/j.ecolind.2016.06.005. Accessed 13 January 2021.

Taylor, M.L., Gwinnett, C., Robinson, L.F. and Woodall, L.C. (2016). Plastic microfibre ingestion by deep-sea organisms. *Scientific Reports* 6, 33997. https://doi.org/10.1038/srep33997. Accessed 13 January 2021.

Tekman, M.B., Krumpen, T. and Bergmann, M. (2017). Marine litter on deep Arctic seafloor continues to increase and spreads to the North at the HAUSGARTEN observatory. *Deep Sea Research Part I: Oceanographic Research Papers* 120, 88-99. https://doi.org/10.1016/j.dsr.2016.12.011. Accessed 13 January 2021.

Tekman, M.B., Wekerle, C., Lorenz, C., Primpke, S., Hasemann, C., Gerdts, G. *et al.* (2020). Tying up loose ends of microplastic pollution in the Arctic: Distribution from the sea surface through the water column to deep-sea sediments at the HAUSGARTEN Observatory. *Environmental Science and Technology* 54(7), 4079-4090. https://dx.doi.org/10.1021/acs.est.9b06981. Accessed 13 January 2021.

Temmerman, S., Meire, P., Bouma, T., Herman, P.M.J., Ysebaert, T and De Vriend, H.J. (2013). Ecosystem-based coastal defence in the face of global change. *Nature* 504, 79-83. https://doi.org/10.1038/nature12859. Accessed 15 February 2021.

ten Brink, P., Schweitzer, J-P., Watkins, E., Janssens, C., De Smet, M., Leslie, H. *et al.* (2018). *Circular Economy Measures to Keep Plastics and their Value in the Economy, Avoid Waste and Reduce Marine Litter*. Economics Discussion Papers 2018-3. Kiel Institute for the World Economy. http://www.economics-ejournal.org/economics/discussionpapers/2018-3/. Accessed 13 January 2021.

ter Halle, A., Ladirat, L., Gendre, X., Goudounèche, D., Pusineri, C., Routaboul, C. *et al.* (2016). Understanding the fragmentation pattern of marine plastic debris. *Environmental Science and Technology* 50 (11), 5668-5675. https://doi.org/10.1021/acs.est.6b00594. Accessed 13 January 2021.

Teuten, E.L., Saquing, J.M., Knappe, S.D.R.U., Barlaz, M.A., Jonsson, S., Bjoern, A. *et al.* (2009). Transport and release of chemicals from plastics to the environment and to wildlife. *Philosophical Transactions of the Royal Society B: Biological Sciences* 364, 2027-2045. https://doi.org/10.1098/rstb.2008.0284. Accessed 13 January 2021.

Thaysen, C., Sorais, M., Verreault, J., Diamond, M.L., and Rochman, C.M. (2020). Bidirectional transfer of halogenated flame retardants between the gastrointestinal tract and ingested plastics in urban- adapted ring-billed gulls. *Science of The Total Environment* 730, 138887. https://doi.org/10.1016/j.scitotenv.2020.138887. Accessed 13 January 2021.

Theocharis, A., Balopoulos, E., Kioroglou, S., Kontoyiannis, H. and Iona, A. (1999). A synthesis of the circulation and hydrography of the South Aegean Sea and the Straits of the Cretan Arc (March 1994- January 1995). *Progress in Oceanography* 44(4), 469-509. https://doi.org/10.1016/S0079-6611(99)00041-5. Accessed 13 January 2021.

The Pew Charitable Trusts and SYSTEMIQ (2020). *Breaking the Plastics Wave: A Comprehensive Assessment of Pathways towards Stopping Ocean Plastic Pollution.* https://www.oneplanetnetwork.org/resource/breaking-plastic-wave-comprehensive-assessment-pathways-towards-stopping-ocean-plastic. Accessed 13 January 2021.

Thiel, M. and Gutow, L. (2005). The ecology of rafting in the marine environment. II. The rafting organisms and community. *Oceanography and Marine Biology: An Annual Review* 43, 279-418. https://epic.awi.de/id/eprint/11613/1/Thi2005a.pdf. Accessed 13 January 2021.

Thiel, M., Hinojosa, I.A., Miranda, L., Pantoja, J., Rivadeneira, M.M. and Vasquez, N. (2013). Anthropogenic marine debris in the coastal environment: A multiyear comparison between coastal waters and local beaches. *Marine Pollution Bulletin* 71(1-2), 307-316. https://doi.org/10.1016/j.marpolbul.2013.01.005. Accessed 13 January 2021.

Thiel, M., Luna-Jorquera, G., Álvarez-Varas, R., Gallardo, C., Hinojosa, I.A., Luna, N. *et al.* (2018). Impacts of marine plastic pollution from continental coasts to subtropical gyres – fish, seabirds, and other vertebrates in the SE Pacific. *Frontiers in Marine Science* 5, 238. https://doi.org/10.3389/fmars.2018.00238. Accessed 13 January 2021.

Thompson, R.C., Moore, C.J., vom Saal, F.S. and Swan, S.H. (2009). Plastics, the environment and human health: Current consensus and future trends. *Philosophical Transactions of the Royal Society B Biological Sciences* 364, 2153-2166. https://doi.org/10.1098/rstb.2009.0053. Accessed 13 January 2021.

Tibbetts, J., Krause, S., Lynch, I. and Sambrook Smith, G. (2018). Abundance, distribution and drivers of microplastics contamination in urban river environments. *Water* 10(11), 1597-1611. https://doi.org/10.3390/w10111597. Accessed 13 January 2021.

Tirelli, V., Suaria, G., and Lusher, A.L. (2020). Microplastics in polar samples. In *Handbook of Microplastics in the Environment*. Rocha-Santos, T., Costa, M. and Mouneyrac, C. (eds,). Cham: Springer. 1-42. https://doi.org/10.1007/978-3-030-10618-8_4-1. Accessed 13 January 2021.

Topouzelis, K., Papakonstantinou, A., and Garaba, S. P. (2019). Detection of floating plastics from satellite and unmanned aerial systems (plastic litter project 2018). *International Journal of . Applied Earth Observation and Geoinformatics*, 79: 175–183. https://doi.org/10.1016/j.jag.2019.03.011

Trevail, A., Gabrielsen, G.W., Kühn, S., Bock, A. and van Franeker, J.A. (2014). *Plastic ingestion by Northern Fulmars,* Fulmarus glacialis, *in Svalbard and Iceland, and relationships between plastic ingestion and contaminant uptake.* Kort rapport/Brief report. 029. Norwegian Polar Institute, Tromsø, Norway. https://www.researchgate.net/publication/283398736_Plastic_Ingestion_by_Northern_Fulmars_Fulmarus_glacialis_in_Svalbard_and_Iceland_and_Relationships_between_Plastic_Ingestion_and_Contaminant_Uptake. Accessed 13 January 2021.

Tudor, D.T. and Williams, A.T. (2003). Public perception and opinion of visible beach aesthetic pollution: The utilisation of photography. *Journal of Coastal Research* 19, 1104-1115. https://www.jstor.org/stable/4299252. Accessed 13 January 2021.

Turner, A. (2016). Heavy metals, metalloids and other hazardous elements in marine plastic litter. *Marine Pollution Bulletin* 111(1-2), 136-142. https://doi.org/10.1016/j.marpolbul.2016.07.020. Accessed 13 January 2021.

Turner, A. (2018). Black plastics: Linear and circular economies, hazardous additives and marine pollution. *Environment International* 117, 308-318. https://doi.org/10.1016/j.envint.2018.04.036. Accessed 20 December 2020.

Turra, A., Manzano, A.B., Dias, R.J.S., Mahiques, M.M., Barbosa, L. *et al.* (2014). Three dimensional distribution of plastic pellets in sandy beaches: Shifting paradigms. *Scientific Reports* 4, 4435. https://doi.org/10.1038/srep04435. Accessed 13 January 2021.

Turrell, W. (2019). Spatial distribution of foreshore litter on the northwest European continental shelf. *Marine Pollution Bulletin* 142, 583-594. https://doi.org/10.1016/j.marpolbul.2019.04.009. Accessed13 January 2021.

UNCTAD (United Nations Conference on Trade and Development) (2020). Global trade in plastics: insights from the first life-cycle trade database -UNCTAD Research Paper No. 53. https://unctad.org/fr/node/32014. Accessed 20 October 2021.

Unger, A. and Harrison, N. (2016). Fisheries as a source of marine debris on beaches in the United Kingdom. *Marine Pollution Bulletin* 107(1), 52-58. https://doi.org/10.1016/j.marpolbul.2016.04.024. Accessed 11 January 2021.

UN (United Nations) (2020). *Sustainable Development Goals Report 2020.* New York. https://unstats.un.org/sdgs/report/2019/The-Sustainable-Development-Goals-Report-2019.pdf. Accessed 11 January 2021.

UNDRR (United Nations Office for Disaster Risk Reduction) (2019). *Global Assessment Report on Disaster Risk Reduction 2019.* Distillation and full report. Geneva. https://gar.undrr.org/report-2019. Accessed 11 January 2021.

UNEA (United Nations Environment Assembly) (2018). *Combating Marine Plastic Litter and Microplastics: An Assessment of the Effectiveness of Relevant International, Regional and Subregional Governance Strategies and Approaches – Summary for Policy Makers*. UNEP/AHEG/2018/1/INF/3. Nairobi. https://wedocs.unep.org/bitstream/handle/20.500.11822/31035/k1800347inf5.pdf. Accessed 14 January 2021.

UNEP (United Nations Environment Programme) (2014). *Valuing Plastic: The Business Case for Measuring, Managing and Disclosing Plastic Use in the Consumer Goods Industry.* Nairobi. https://wedocs.unep.org/handle/20.500.11822/9238. Accessed 14 January 2021.

UNEP (2015). *Biodegradable Plastics and Marine Litter: Misconceptions, Concerns and Impacts on Marine Environments*. Nairobi. https://wedocs.unep.org/20.500.11822/7468. Accessed 14 January 2021.

UNEP (2016). *Marine Plastic Debris and Microplastics: Global Lessons and Research to Inspire and Guide Policy Change*. Nairobi. https://wedocs.unep.org/handle/20.500.11822/7720. Accessed 14 January 2021.

UNEP (2017a). Marine Litter Socio-Economic Study. Nairobi. https://wedocs.unep.org/20.500.11822/26014. Accessed 14 January 2021.

UNEP (2017b). *Waste Management in ASEAN Countries: Summary Report*. Bangkok. https://stg-wedocs.unep.org/bitstream/handle/20.500.11822/21134/waste_mgt_asean_summary.pdf. Accessed 14 January 2021.

UNEP (2018a). *Exploring the Potential for Adopting Alternative Materials to Reduce Marine Plastic Litter.* Nairobi. https://stg-wedocs.unep.org/bitstream/handle/20.500.11822/21134/waste_mgt_asean_summary.pdf. Accessed 14 January 2021.

UNEP (2018b). *Single-use Plastics: A Roadmap for Sustainability.* Nairobi. https://www.unenvironment.org/resources/report/single-use-plastics-roadmap-sustainability. Accessed 14 January 2021.

UNEP (2018c). *Africa Waste Management Outlook*. Nairobi. https://wedocs.unep.org/handle/20.500.11822/25514. Accessed 14 January 2021.

UNEP (2018d). *Addressing Marine Plastics: A Systemic Approach – Recommendations for Action.* Notten, P. (author). Nairobi. https://www.unenvironment.org/resources/report/addressing-marine-plastics-systemic-approach-recommendations-actions. Accessed 14 January 2021.

UNEP (2018e). *Mapping of Global Plastics Value Chain and Plastics Losses to the Environment: With a Particular Focus on Marine Environment*. Nairobi. https://wedocs.unep.org/handle/20.500.11822/26745. Accessed 16/6/2021

UNEP (2019a). *Plastics and Shallow Water Coral Reefs: Synthesis of the Science for Policy-makers.* Sweet, M., Stelfox, M. and Lamb, J. (authors). Nairobi. https://stg-wedocs.unep.org/handle/20.500.11822/27646. Accessed 14 January 2021.

UNEP (2019b). *The Role of Packaging Regulations and Standards in Driving the Circular Economy.* Nairobi. http://sos2019.sea-circular.org/wp-content/uploads/2019/11/FINAL_THE-ROLE-OF-PACKAGING-REGULATIONS-AND-STANDARDS-IN-DRIVING-THE-CIRCULAR-ECONOMY.pdf. Accessed 14 January 2021.

UNEP (2019c). *Lessons from a Decade of Emissions Gap Assessments*. Christensen, J. and Ohloff, A. (authors). Nairobi.
https://wedocs.unep.org/bitstream/handle/20.500.11822/30022/EGR10.pdf?sequence=1&isAllowed=y. Accessed 14 January 2021.

UNEP (2019d). *Measuring Fossil Fuel Subsidies in the Context of the Sustainable Development Goals*. Nairobi. https://wedocs.unep.org/bitstream/handle/20.500.11822/28111/FossilFuel.pdf?sequence=1&isAllowed=y. Accessed 14 January 2021.

UNEP (2019e) *Global Chemicals Outlook II. From Legacies to Innovative Solutions: Implementing the 2030 Agenda for Sustainable Development.* Nairobi. https://www.unep.org/explore-topics/chemicals-waste/what-we-do/policy-and-governance/global-chemicals-outlook. Accessed 20 November 2020.

UNEP (2020a). Global Partnership on Marine Litter. https://www.unenvironment.org/explore-topics/oceans-seas/what-we-do/addressing-land-based-pollution/global-partnership-marine. Accessed 20 November 2020.

UNEP (2020b). *Monitoring Plastics in Rivers and Lakes: Guidelines for the Harmonization of Methodologies.* Nairobi. https://wedocs.unep.org/bitstream/handle/20.500.11822/35405/MPRL.pdf. Accessed 6 May 2021.

UNEP (2020c). *Water Pollution by Plastics and Microplastics: A Review of Technical Solutions from Source to Sea*. Nairobi. https://www.unep.org/resources/report/water-pollution-plastics-and-microplastics-review-technical-solutions-source-sea. Accessed 6 May 2021.

UNEP (2020d). *Catalogue of Technologies to Address the Risks of Contamination of Water Bodies with Plastics and Microplastics.* Nairobi. https://wedocs.unep.org/handle/20.500.11822/34423. Accessed 6 May 2021.

UNEP (2020e). *An Assessment Report on Issues of Concern: Chemicals and Waste Issues Posing Risks to Human Health and the Environment.* Nairobi. https://wedocs.unep.org/bitstream/handle/20.500.11822/33807/ARIC.pdf?sequence=1&isAllowed=y. Accessed 7 June 2021.

UNEP (2021a). *Green and Sustainable Chemistry: Framework Manual.* Nairobi. https://wedocs.unep.org/handle/20.500.11822/34338. Accessed 20 June 2021.

UNEP (2021b). *Single-Use Plastic Products and Their Alternatives: Recommendations from Life Cycle Assessments.* https://wedocs.unep.org/xmlui/bitstream/handle/20.500.11822/31932/SUPB.pdf?sequence=1&isAllowed=y. Accessed 18 June 2021.

UNEP (2021c). *Understanding the State of the Ocean: A Global Manual on Measuring SDG 14.1.1, SDG 14.2.1 and SDG 14.5.1. https://wedocs.unep.org/handle/20.500.11822/35086;jsessionid=BD2C69D18A6CBD1A14C24C40A895133D*. Accessed 18 June 2021.

UNEP/GPA (Global Programme of Action for the Protection of the Marine Environment from Land-based Activities) (2020). Governing the Global Programme of Action. https://www.unenvironment.org/explore--topics/oceans-seas/what-we-do/addressing-land-based-pollution/governing-global-programmetopics/oceans-seas/what-we-do/addressing-land-based-pollution/governing-global-programme.
Accessed 14 January 2021.

UNEP/IPCP (International Panel on Chemical Pollution) (2016). *Overview Report I: A Compilation of Lists of Chemicals Recognised as Endocrine Disrupting Chemicals (EDCs) or Suggested as Potential EDCs*. Geneva. https://wedocs.unep.org/handle/20.500.11822/12218. Accessed 14 June 2021.

UNEP/MAP (Mediterranean Action Plan) (2015). *Marine Litter Assessment in the Mediterranean.* Athens. https://wedocs.unep.org/handle/20.500.11822/7098. Accessed 14 June 2021.

UNEP/MAP (2017). *Integrated Monitoring and Assessment Programme of the Mediterranean Sea and Coast and Related Assessment Criteria*. Athens.
https://wedocs.unep.org/bitstream/handle/20.500.11822/17012/imap_2017_eng.pdf?sequence=5andisAllowed=y. Accessed 14 June 2021.

UNEP/Stockholm Convention (2017). *The 16 New POPs. An Introduction to the Chemicals Added to the Stockholm Convention as Persistent Organic Pollutants by the Conferences of the Parties.* Geneva. http://www.pops.int/TheConvention/ThePOPs/TheNewPOPs/tabid/2511/Default.aspx.
Accessed 14 January 2021.

UNEP and Consumers International (2020). *Can I Recycle This? A Global Mapping and Assessment of Standards, Labels and Claims on Plastic Packaging*. https://www.oneplanetnetwork.org/resource/can-i-recycle-global-mapping-and-assessment-standards-labels-and-claims-plastic-packaging. Accessed 14 January 2021.

UNEP and the International Trade Centre (2017). *Guidelines for Providing Product Sustainability Information: Global Guidance on Making Effective Environmental, Social and Economic claims, to Empower and Enable Consumer Choice*. Geneva. https://www.oneplanetnetwork.org/resource/guidelines-providing-product-sustainability-information. Accessed 14 January 2021.

UNESCAP (United Nations Economic and Social Commission for Asia and the Pacific) (2019). *Closing the Loop: Regional Policy Guide. Innovative Partnerships with Informal Workers to Recover Plastic Waste, in an Inclusive Circular Economy Approach*. https://www.unescap.org/resources/closing-loop-regional-policy-guide. Accessed 11 January 2021.

UN General Assembly (2015). *Transforming our World: The 2030 Agenda for Sustainable Development.* A/RES/70/1.
https://sdgs.un.org/sites/default/files/publications/21252030%20Agenda%20for%20Sustainable%20Development%20web.pdf. Accessed 11 January 2021.

UN General Assembly (2016). *Report of the Open-ended Intergovernmental Expert Working Group on Indicators and Terminology Relating to Disaster Risk Reduction.* United Nations General Assembly Seventy-first session, 1 December 2016. A/71/644. https://reliefweb.int/sites/reliefweb.int/files/resources/50683_oiewgreportenglish.pdf. Accessed 11 January 2021.

UN General Assembly (2021). *Report of the Special Rapporteur on the implications for human rights of the environmentally sound management and disposal of hazardous substances and wastes, Marcos Orellana: The stages of the plastics cycle and their impacts on human rights*. United Nations General Assembly Seventy-sixth session, 22 July 2021. A/76/207. https://undocs.org/A/76/207. Accessed 18 October 2021.

United States Environmental Protection Agency (2014). *Priority Pollutant List. Report 40 CFR Part 423. Appendix A.* https://www.epa.gov/sites/production/files/2015-09/documents/priority-pollutant-list-epa.pdf.
Accessed 30 April 2019.

Uyarra, M.C. and Borja, A. (2016). Ocean literacy: A 'new' socio-ecological concept for a sustainable use of the seas. *Marine Pollution Bulletin* 104, 1-2. https://doi.org/10.1016/j.marpolbul.2016.02.060. Accessed 14 January 2021.

van Calcar, C.J. and van Emmerik, T.H.M. (2019). Abundance of plastic debris across European and Asian rivers. *Environmental Research Letters* 14, 124051. https://iopscience.iop.org/article/10.1088/1748-9326/ab5468/meta. Accessed 12 January 2021.

van Cauwenberghe, L., Claessens, M., Vandegehuchte, M.B. and Janssen, C.R. (2015). Microplastics are taken up by mussels (*Mytilus edulis*) and lugworms (*Arenicola marina*) living in natural habitats. *Environmental Pollution* 199, 10-17. https://doi.org/10.1016/j.envpol.2015.01.008. Accessed 12 January 2021.

van den Bergh, J. and Botzen, W. (2015). Monetary valuation of the social cost of CO_2 emissions: A critical survey. *Ecological Economics* 114, 33-46. https://doi.org/10.1016/j.ecolecon.2015.03.015. Accessed 12 January 2021.

van der Mheen, M., Pattiaratchi, C. and van Sebille, E. (2019). Role of Indian Ocean dynamics on accumulation of buoyant debris. *Journal of Geophysical Research: Oceans* 124, 2571–2590. https://doi.org/10.1029/2018JC014806. Accessed 12 January 2021.

van der Wal, M., van der Meule, M., Roex, E., Wolthuis, Y., Tweehuysen, G. and Vethaak, D. (2013). *Summary report: Plastics litter in Rhine, Meuse and Scheldt, contribution to plastics litter in the North Sea*. Project 1205955, Deltares, Delft, the Netherlands. https://kenniswijzerzwerfafval.nl/document/summary-report-plastic-litter-rhine-meuse-and-scheldt-contribution-plastic-litter-north-sea. Accessed 12 January 2021.

van Emmerik, T., Kieu-Le, T.-C., Loozen, M., van Oeveren, K., Strady, E., Bui, X-T. *et al.* (2018) A methodology to characterize riverine macroplastic emission into the ocean. *Frontiers in Marine Science* 5, 372. https://doi.org/10.3389/fmars.2018.00372. Accessed 12 January 2021.

van Emmerik, T., Loozen, M., van Oeveren, K., Buschman, F. and Prinsen, G. (2019). Riverine plastic emission from Jakarta into the ocean. *Environmental Research Letters* 14(8), 084033. https://doi.org/10.1088/1748-9326/ab30e8. Accessed 12 January 2021.

van Emmerik, T. and Schwarz, A. (2019). Plastic debris in rivers. *WIREs Water* 7(1), e1398. https://doi.org/10.1002/wat2.1398. Accessed 12 January 2021.

van Sebille, E., England, M.H. and Froyland, G. (2012). Origin, dynamics and evolution of ocean garbage patches from observed surface drifters. *Environmental Research Letters* 7, 044040. https://doi.org/10.1088/1748-9326/7/4/044040. Accessed 12 January 2021.

van Sebille, E., Wilcox, C., Lebreton, L., Maximenko, N., Hardesty, B.D., van Franeker *et al.* (2015). A global inventory of small floating plastic debris. *Environmental Research Letters* 10, 124006. https://doi.org/10.1088/1748-9326/10/12/124006. Accessed 12 January 2021.

van Sebille, E., Delandmeter, P., Schofield, J., Hardesty, B.D., Jones, J. and Donnelly, A. (2019). Basin- scale sources and pathways of microplastic that ends up in the Galápagos Archipelago. *Ocean Science* 15, 1341-1349. https://doi.org/10.5194/os-15-1341-2019. Accessed 12 January 2021.

van Sebille, E., Aliani, S. , Law, K.L. , Maximenko, N., Alsina, J.M., Bagaev, A. *et al.* (2020). The physical oceanography of the transport of floating marine debris. *Environmental Research Letters* 15, 023003. https://doi.org/10.1088/1748-9326/ab6d7d. Accessed 12 January 2021

van Truong, N. and Ping, C.B. (2019). Plastic marine debris: Sources, impacts and management, *International Journal of Environmental Studies* 76(6), 953-973. https://doi.org/10.1080/00207233.2019.1662211. Accessed 12 January 2021.

Vaughan, R., Turner, S.D. and Rose, N. (2017). Microplastics in the sediments of a UK urban lake. *Environmental Pollution* 229, 10-18. https://doi.org/10.1016/j.envpol.2017.05.057. Accessed 12 January 2021.

Vázquez-Morillas, A., Beltrán-Villavicencio, M., Alvarez-Zeferino, J.C., Osada-Velázquez, M.H., Moreno, A., Martínez, L. *et al.* (2016). Biodegradation and ecotoxicity of polyethylene films containing pro-oxidant additive. *Journal of Polymers and the Environment* 24, 221-229. https://doi.org/10.1007/s10924-016-0765-8. Accessed 12 January 2021.

Vegter, A.C., Barletta, M., Beck, C., Borrero, J., Burton, H., Campbell, M.L. *et al.* (2014). Global research priorities to mitigate plastic pollution impacts on marine wildlife. *Endangered Species Research* 25(3), 225-247. https://doi.org/10.3354/esr00623. Accessed 12 January 2021.

Veiga, J.M., Fleet, D., Kinsey, S., Nilsson, P., Vlachogianni, T., Werner, S. *et al.* (2016). *Identifying Sources of Marine Litter*. MSFD GES TG Marine Litter Thematic Report; JRC Technical Report; EUR 28309. https://doi.org/10.2788/018068. Accessed 12 January 2021.

Velis, C.A. and Cook, E. (2021). Mismanagement of plastic waste through open burning with emphasis on the global south: A systematic review of risks to occupational and public health. *Environmental Science and Technology*, 55, 11, 7186-7207. https://doi.org/10.1021/acs.est.0c08536. Accessed 13 July 2021.

Verlis, K.M., Campbell, M.L. and Wilson, S.P. (2018). Seabirds and plastics don't mix: Examining the differences in marine plastic ingestion in wedge-tailed shearwater chicks at near-shore and offshore locations. *Marine Pollution Bulletin* 135, 852-861. https://doi.org/10.1016/j.marpolbul.2018.08.016. Accessed 12 January 2021.

Vethaak, A.D. and Legler, J. (2021). Microplastics and human health. *Science* 371, 672-674. https://doi.org/10.1126/science.abe5041. Accessed 15 February 2021.

Vianello, A., Jensen, R.L., Liu, L. and Vollertsen, J. (2019). Simulating human exposure to indoor airborne microplastics using a Breathing Thermal Manikin. *Science Reports,* 9: 8670 https://doi.org/10.1038/s41598-019-45054-w. Accessed 30 November 2020.

Villarubia-Gómez, P., Cornell, S.E. and Fabres, J. (2018). Marine plastic pollution as a planetary boundary threat – the drifting piece in the sustainable puzzle. *Marine Policy* 96, 213-220. https://doi.org/10.1016/j.marpol.2017.11.035. Accessed 12 January 2021.

Vincent, A., Drag, N., Lyandres, O., Neville, S. and Hoellein, T. (2017). Citizen science datasets reveal drivers of spatial and temporal variation for anthropogenic litter on Great Lakes beaches. *Science of The Total Environment* 15, 577:105-112. https://doi.org/10.1016/j.scitotenv.2016.10.113. Accessed 12 January 2021.

Viršek, M.K., Lovšin, M.N., Koren, Š., Kržan, A. and Peterlin, M. (2017). Microplastics as a vector for the transport of the bacterial fish pathogen species Aeromonas salmonicida. *Marine Pollution Bulletin* 125(1-2), 301-309. https://doi.org/10.1016/j.marpolbul.2017.08.024. Accessed 14 January 2021.

Vlachogianni, T., Anastasopoulou, A., Fortibouni, T., Ronchi, F. and Zeri, C. (2017). *Marine Litter Assessment in the Adriatic and Ionian seas*. IPA-Adriatic DeFishGear Project, MIO-ECSDE, HCMR and ISPRA. https://mio-ecsde.org/project/5054/. Accessed 12 January 2021.

Vlachogianni, T., Fortibuoni, T., Ronchi, F., Zeri, C., Mazziotti, C., Tutman, P. *et al.* (2018). Marine litter on the beaches of the Adriatic and Ionian Seas: An assessment of their abundance, composition and sources. *Marine Pollution Bulletin* 131, 745-756. https://doi.org/10.1016/j.marpolbul.2018.05.006. Accessed 12 January 2021.

von Friesen, L.W., Granberg, M.E., Hassellöv, M., Gabrielsen, G.W. and Magnusson, G. (2019). An efficient and gentle enzymatic digestion protocol for the extraction of microplastics from bivalve tissue. *Marine Pollution Bulletin* 142, 129-134. https://doi.org/10.1016/j.marpolbul.2019.03.016. Accessed 14 January 2021.

von Moos, N., Burkhardt-Holm, P. and Köhler, A. (2012). Uptake and effects of microplastics on cells and tissue of the blue mussel *Mytilus edulis* L. after an experimental exposure. *Environmental Science and Technology* 46(20), 11327-11335. https://doi.org/10.1021/es302332w. Accessed 14 January 2021.

Waite, H.R., Donnelly, M.J. and Walters, L.J. (2018). Quantity and types of microplastics in the organic tissues of the eastern oyster *Crassostrea virginica* and Atlantic mud crab *Panopeus herbstii* from a Florida estuary. *Marine Pollution Bulletin* 129(1), 179-185. https://doi.org/10.1016/j.marpolbul.2018.02.026. Accessed 14 January 2021.

Walker, T., Gramlich, D. and Dumont-Bergeron, A. (2020). The case for a plastic tax: A review of its benefits and disadvantages within a circular economy. In *Sustainability. Business and Society* 360 *vol.* 4. Wasieleski, D.M. and Weber, J. (eds.). Emerald Publishing Limited. 185-211. https://www.emerald.com/insight/content/doi/10.1108/S2514-175920200000004010/full/html. Accessed 14 January 2021.

Walther, B.A., Kusui, T., Yen, N., Hu, C.S. and Lee, H. (2020). Plastic pollution in East Asia: Macroplastics and microplastics in the aquatic environment and mitigation efforts by various actors. In *The Handbook of Environmental Chemistry*. Berlin, Heidelberg: Springer. https://link.springer.com/chapter/10.1007%2F698_2020_508. Accessed 14 January 2021.

Wang, J. (2018). High levels of microplastic pollution in the sediments and benthic organisms of the South Yellow Sea, China. *Science of The Total Environment* 651, Part 2, 1661-1669. https://doi.org/10.1016/j.scitotenv.2018.10.007. Accessed 14 January 2021.

Wang, J., Peng, J., Tan, Z., Gao, Y., Zhan, Z., Chen, Q. *et al.* (2017). Microplastics in the surface sediments from the Beijing River littoral zone: Composition, abundance, surface textures and interaction with heavy metals. *Chemosphere* 171, 248-258. https://doi.org/10.1016/j.chemosphere.2016.12.074 . Accessed 14 January 2021.

Wang, J., Liu, X., Li, Y., Powell, T., Wang, X., Wang, G. *et al.* (2019a). Microplastics as contaminants in the soil environment: A mini-review. *Science of The Total Environment* 691 848-857. https://doi.org/10.1016/j.scitotenv.2019.07.209. Accessed 14 January 2021.

Wang, J., Coffin, S., Sun, C., Schlenk, D. and Gan, J. (2019b). Negligible effects of microplastics on animal fitness and HOC bioaccumulation in earthworm *Eisenia fetida* in soil. *Environmental Pollution* 249, 776-784. https://doi.org/10.1016/j.envpol.2019.03.102. Accessed 14 January 2021.

Wang, S., Lydon, K.A., White, E.M., Grubbs III, J.B., Lipp, E.K., Locklin, J. *et al.* (2018). Biodegradation of poly(3-hydroxybutyrate-co-3-hydroxyhexanoate) plastic under anaerobic sludge and aerobic seawater conditions: Gas evolution and microbial diversity. *Environmental Science and Technology* 52(10), 5700-5709. https://doi.org/10.1021/acs.est.7b06688. Accessed 14 January 2021.

Wang, W. and Wang, J. (2018). Investigation of microplastics in aquatic environments, An overview of the methods used, from field sampling to laboratory analysis. *TrAC Trends in Analytical Chemistry* 108, 26. https://doi.org/10.1016/j.trac.2018.08.026. Accessed 14 January 2021.

Wang, Y., Zhu, H. and Kannan, K. (2019). A Review of Biomonitoring of Phthalate Exposures. Toxics 7(2), 21. https://doi.org/10.3390/toxics7020021 Accessed 8 June 2021.

Waste Atlas (2014). The World's 50 Biggest Dump Sites. 2014 Report. http://atlas.d-waste.com/. Accessed 14 January 2021.

Webb, H.K., Arnott, J., Crawford, R.J. and Ivanova, E.P. (2013). Plastic degradation and its environmental implications with special reference to poly (ethylene terephthalate). *Polymers* 5, 1-18. https://doi.org/10.3390/polym5010001. Accessed 14 January 2021.

Webber, D.N. and Parker, S.J. (2012). Estimating unaccounted fishing mortality in the Ross Sea region and Amundsen Sea (CCAMLR subareas 88.1 and 88.2) bottom longline fisheries targeting Antarctic toothfish. *CCAMLR Science* 19, 17-30. https://www.ccamlr.org/en/publications/science_journal/ccamlr-science-volume-19/17–30. Accessed 14 January 2021.

Wedemeyer-Strombel, K.R., Balazs, G.H., Johnson, J.B., Peterson, T.D., Wicksten, M.K. and Plotkin, P.T. (2015). High frequency of occurrence of anthropogenic debris ingestion by sea turtles in the North Pacific Ocean. *Marine Biology* 162, 2079-2091. https://doi.org/10.1007/s00227-015-2738-1. Accessed 14 January 2021.
Welden, N.A. and Cowie, P.R. (2017). Degradation of common polymer ropes in a sublittoral marine environment. *Marine Pollution* Bulletin 118 (1-2), 248-253. https://doi.org/10.1016/j.marpolbul.2017.02.072. Accessed 14 January 2021.

Welden, N.A. and Lusher, A.L. (2017). Impacts of changing ocean circulation on the distribution of marine microplastic litter. *Integrated Environmental Assessment and Management* 13 (3), 483-487. https://doi.org/10.1002/ieam.1911. Accessed 14 January 2021.

Welle, F. (2018). Microplastic in bottled natural mineral water – literature review and considerations on exposure and risk assessment. *Food Additives and Contaminants* 35, 2482-2492. https://doi.org/10.1080/19440049.2018.1543957. Accessed 14 January 2021.

Werbowski, L.M., Gilbreath, A.N., Munno, K., Zhu, X., Grbic,J., Wu, T., Sutton, R., Sedlak, M.D. Deshpande, A.D., and Rochman, C.M. (2021). Urban stormwater runoff: a major pathway for anthropogenic particles, black rubbery fragments, and other types of microplastics to urban receiving waters. *American Chemical Society Environmental Science &Technology Water* 1 (6), 1420-1428 https://doi.org/10.1021/acsestwater.1c00017. Accessed 23 June 2021.

Wesch, C., Barthel, A.-K., Braun, U., Klein, R., and Paulus, M., (2016). No microplastics in benthic eelpout (*Zoarces viviparus*): An urgent need for spectroscopic analyses in microplastic detection. *Environmental Research* 148, 36-38. https://doi.org/10.1016/j.envres.2016.03.017. Accessed 14 January 2021.

Whelpdale, D.M. (1974). Particulate residence times. *Water, Air and Soil Pollution* 3, 293-300. https://doi.org/10.1080/19440049.2018.1543957. Accessed 14 January 2021.

Whitaker, J., Garza, T.N. and Janosik, A.M. (2019). Sampling with Niskin bottles and microfiltration reveals a high prevalence of microfibres. *Limnologica – Ecology and Management of Inland Waters* 78, 125711. http://doi.org/10.1016/j.limno.2019.125711. Accessed 14 January 2021.

White, M.P., Elliott, L.R., Gascon, M., Roberts, B. and Fleming, L.E. (2020). Blue space, health and well-being: a narrative overview and synthesis of potential benefits. Environ*mental Research* 191, 110169- 110169. https://doi.org/10.1016/j.envres.2020.110169. Accessed 14 January 2021.

WHO (World Health Organization) (2016). *Ambient Air Pollution: A Global Assessment of Exposure and Burden of Disease*. Geneva. https://apps.who.int/iris/bitstream/handle/10665/250141/9789241511353-eng.pdf?sequence=1. Accessed 14 January 2021.

WHO (2019). *Microplastics in Drinking-water.* Geneva. https://apps.who.int/iris/bitstream/handle/10665/326499/9789241516198-eng.pdf?ua=1. Accessed 14 January 2021.

Wichmann, D., Delandmeter, P. and van Sebille, E. (2019). Influence of near-surface current on the global dispersal of marine microplastic. *JGR Oceans* 124(8), 6086-6096. https://doi.org/10.1029/2019JC015328. Accessed 14 January 2021.

Wieczorek, A., Croot, P., Lombard, L., Sheahan, J. and Doyle, T. (2019). Microplastic ingestion by gelatinous zooplankton may lower efficiency of the biological pump. *Environmental Science and Technology* 53(9), 5387–5395. https://doi.org/10.1021/acs.est.8b07174. Accessed 14 January 2021.

Wilcox, C., van Sebille, E., and Hardesty, B.D. (2015). Threat of plastic pollution to seabirds is global, pervasive, and increasing. *Proceedings of the National Academy of Sciences* 38, 11899-11904. http://doi.org/10.1073/pnas.1502108112. Accessed 14 January 2021.

Williams, A.T. and Rangel-Buitrago, N. (2019). Marine litter: Solutions for a major environmental problem. *Journal of Coastal Research* 35(3), 648-663. https://doi.org/10.2112/JCOASTRES-D-18-00096.1. Accessed 14 January 2021.

Willis, K., Maureaud, C., Wilcox, C. and Hardesty, B.D. (2018). How successful are waste abatement campaigns and government policies at reducing plastic waste into the marine environment? *Marine Policy* 96, 243-249. https://doi.org/10.1016/j.marpol.2017.11.037. Accessed 14 January 2021.
Windsor, F.M., Durance, I., Horton, A.A., Thompson, R.C., Tyler, C.R. and Ormerod, S.J. (2018). A catchment-scale perspective of plastic pollution. *Global Change Biology* 25, 1207-1221. https://doi.org/10.1111/gcb.14572. Accessed 14 January 2021.

Windsor, F.M., Tilley, R.M., Tyler, C.R. and Omerod, S.J. (2019). Microplastic ingestion by riverine macroinvertebrates. *Science of The Total Environment* 646, 68-74. https://doi.org/10.1016/j.scitotenv.2018.07.271. Accessed 14 January 2021.

Woodall, L.C., Sanchez-Vidal, A., Canals, M., Paterson, G.L., Coppock, R., Sleight, V. *et al.* (2014). The deep sea is a major sink for microplastic debris. *Royal Society Open Science* 1, 140317. https://doi.org/10.1098/rsos.140317. Accessed 14 January 2021.

Woodall, L.C., Robinson, L.F., Narayanaswamy, B.E. and Paterson, G.L.J. (2015). Deep-sea litter: A comparison of seamounts, banks and a ridge in the Atlantic and Indian Oceans reveals both environmental and anthropogenic factors impact accumulation and composition. *Frontiers in Marine Science*, 2 February. https://doi.org/10.3389/fmars.2015.00003. Accessed 14 January 2021.

Woods, J.S., Rødder, G. and Verones, F. (2019). An effect factor approach for quantifying the entanglement impact on marine species of macroplastic debris within the life cycle impact assessment. *Ecological Indicators* 99, 61-66. https://doi.org/10.1016/j.ecolind.2018.12.018. Accessed 14 January 2021.

World Economic Forum (2020). *Radically Reducing Plastic Pollution in Indonesia: A Multistakeholder Action Plan. National Plastic Action Partnership.* https://globalplasticaction.org/wp-content/uploads/NPAP-Indonesia-Multistakeholder-Action-Plan_April-2020.pdf. Accessed 14 January 2021.

World Shipping Council (2020). Containers lost at sea 2020 update. Accessed 14 January 2021. https://test.iims.org.uk/wp-content/uploads/2020/11/World-Shipping-Council-containers-lost-at-sea-2020.pdf. Accessed 10 September 2021.

World Trade Organization (2021). International trade statistics. https://data.wto.org. Accessed 10 September 2021.

WRAP (2018). A roadmap to 2025 – The UK Plastics Pact. https://www.wrap.org.uk/content/the-uk-plastics-pact-roadmap-2025. Accessed 14 January 2021.

Wright, R.J., Erni-Cassola, G., Zadjelovic, V., Latva, M., and Christie-Oleza, J.A. (2020). Marine plastic debris: A new surface for colonization. *Environmental Science and Technology* 54(19), 11657-11672. https://pubs.acs.org/doi/10.1021/acs.est.0c02305, Accessed 14 January 2021.

Wright, S.L., Rowe, D., Thompson, R.C. and Galloway, T.S. (2013a). Microplastic ingestion decreases energy reserves in marine worms. *Current Biology* 23, R1031-R1033. https://doi.org/10.1016/j.cub.2013.10.068. Accessed 14 January 2021.

Wright, S.L., Thompson, R.C. and Galloway, T.S. (2013b). The physical impacts of microplastics on marine organisms: A review. *Environmental Pollution* 178, 483-492. https://doi.org/10.1016/j.envpol.2013.02.031. Accessed 14 January 2021.

Wright, S.L., and Kelly, F.J. (2017). Plastic and human health: A micro issue? *Environmental Science and Technology* 51(12), 6634-6647. https://doi.org/10.1021/acs.est.7b00423. Accessed 14 January 2021.

Wright, S.L., Levermore, J.M. and Kelly, F.J. (2019). Raman spectral imaging for the detection of inhalable microplastics in ambient particulate matter samples. *Environmental Science and Technology* 53(15), 8947-8956. https://doi.org/10.1021/acs.est.8b06663. Accessed 14 January 2021.

WWF (2020). *Plastic Packaging in Southeast Asia and China.* https://www.greengrowthknowledge.org/sites/default/files/downloads/resource/Plastic_Packaging_in_SE_Asia_and_China2020_WWF%20-%20Copy.pdf. Accessed 14 January 2021.

WWF, the Ellen MacArthur Foundation and BCG (Boston Consulting Group) (2020). *The Business Case for a UN Treaty on Plastic Pollution*. https://f.hubspotusercontent20.net/hubfs/4783129/Plastics/UN%20treaty%20plastic%20poll%20report%20a4_single_pages_v15-web-prerelease-3mb.pdf. Accessed 13 July 2021.

Wyles, K.J., Pahl, S., Thomas, K., and Thompson, R.C. (2015). Factors that can undermine the psychological benefits of coastal environments: Exploring the effect of tidal state, presence, and type of litter. *Environment and Behavior* 48(9), 1095-1126. https://journals.sagepub.com/doi/full/10.1177/0013916515592177. Accessed 14 January 2021.

Wyles, K.J., Pahl, S., Holland, M., and Thompson, R.C. (2016). Can beach cleans do more than clean-up litter? Comparing beach cleans to other coastal activities *Environment and Behavior* 49(5), 509-535. https://doi.org/10.1177/0013916516649412. Accessed 14 January 2021.
Xanthos, D. and Walker, T.R. (2017). International policies to reduce plastic marine pollution from single-use plastics (plastic bags and microbeads): A review. *Marine Pollution Bulletin* 18(1-2), 17-26. https://doi.org/10.1016/j.marpolbul.2017.02.048. Accessed 14 January 2021.

Xia, Z. (2020). Study on the migration of bisphenols from plastic food contact materials to food simulants. International Conference on Energy, Chemical and Materials Science (ECMS 2019), 6-8 December 2019, Malacca, Malaysia. *Materials Science and Engineering* 738, 012024. https://iopscience.iop.org/article/10.1088/1757-899X/738/1/012024. Accessed 14 January 2021.

Xiong, X., Chen, X., Zhang, K., Mei, Z., Hao, Y., Zheng, J. *et al.* (2018). Microplastics in the intestinal tracts of East Asian finless porpoises (*Neophocaena asiaeorientalis sunameri*) from Yellow Sea and Bohai Sea of China. *Marine Pollution Bulletin* 136, 55-60. https://doi.org/10.1016/j.marpolbul.2018.09.006. Accessed 14 January 2021.

Xu, S., Ma., J., Ji, R., Pan, K. and Miao, A-J. (2020). Microplastics in aquatic environments: occurrence, accumulation and biological effects. *Science of The Total Environment* 703, 134699. https://doi.org/10.1016/j.scitotenv.2019.134699. Accessed 14 January 2021.

Yang, D., Shi, H., Li, L., Li, J., Jabeen, K. and Kolandhaasamy, P. (2015). Microplastic pollution in table salts from China. *Environmental Science and Technology* 49(22), 13622-13627. https://doi.org/10.1021/acs.est.5b03163. Accessed 14 January 2021.

Yang, Y., Liu, G., Song, W., Ye, C., Lin, H., Li, Z. *et al.* (2019). Plastics in the marine environment are reservoirs for antibiotic and metal resistance genes. *Environment International* 123. 79-86. https://doi.org/10.1016/j.envint.2018.11.061. Accessed 14 January 2021.

Ye, S. and Andrady, A.L. (1991). Fouling of floating plastic debris under Biscayne Bay exposure conditions. *Marine Pollution Bulletin* 22 (12), 608-613. https://doi.org/10.1016/0025-326x(91)90249-r. Accessed 14 January 2021.

Yeo, B.G., Takada, H., Taylor, H., Ito, M., Hosoda, J., Allinson, M. *et al.* (2015). POPs monitoring in Australia and New Zealand using plastic resin pellets, and International Pellet Watch as a tool for education and raising public awareness on plastic debris and POPs. *Marine Pollution Bulletin* 101(1), 137-145. http://doi.org/10.1016/j.marpolbul.2015.11.006. Accessed 14 January 2021.

Yokota, K., Waterfield, H., Hastings, C., Davidson, E., Kwietniewski, E. and Wells, B. (2017). Finding the missing piece of the aquatic plastics pollution puzzle: Interaction between primary producers and microplastics. *Limnology and Oceanography Letters* 2, 91-104. https://doi.org/10.1002/lol2.10040. Accessed 14 January 2021.

Yu, F., Sun, Y., Yang, M. and Ma, J. (2019). Adsorption mechanism and effect of moisture contents on ciprofloxacin removal by three-dimensional porous graphene hydrogel. *Journal of Hazardous Materials* 374, 195-202. https://doi.org/10.1016/j.jhazmat.2019.04.021. Accessed 14 January 2021.

Yu, H., Fan, P., Hou, J., Dang, Q., Cui, D., Xi, B. and Tan, W. (2020). Inhibitory effect of microplastics on soil extracellular enzymatic activities by changing soil properties and direct adsorption: An investigation at the aggregate-fraction level. *Environmental Pollution* 267, 115544. https://doi.org/10.1016/j.envpol.2020.115544. Accessed 14 January 2021.

Zalasiewicz, J., Waters, C.N., Ivar do Sul, J.A., Corcoran, P.L., Barnosky, A.D., Cearreta, A. *et al.* (2016). The geological cycle of plastics and their use as a stratigraphic indicator of the Anthropocene. *Anthropocene* 13, 4-17. https://doi.org/10.1016/j.ancene.2016.01.002. Accessed 14 January 2021.

Zambianchi, E., Trani, M. and Falco, P. (2017). Lagrangian transport of marine litter in the Mediterranean Sea. *Frontiers in Environmental Science*, 1 February. https://doi.org/10.3389/fenvs.2017.00005. Accessed 14 January 2021.

Zambrano, M.C., Pawlak, J.J., Daystar, J., Ankeny, M., Cheng, J.J. and Venditti, R.A. (2019). Microfibres generated from the laundering of cotton, rayon and polyester based fabrics and their aquatic biodegradation. *Marine Pollution Bulletin* 142, 394-407. https://doi.org/10.1016/j.marpolbul.2019.02.062. Accessed 14 January 2021.

Zeri, C., Adampoulou, A., Varezić, D.B., Fortibuoni, T., Viršek, M.K., Kržan, A. *et al.* (2018). Floating plastics in Adriatic waters (Mediterranean Sea): From the macro- to the micro-scale. *Marine Pollution Bulletin* 136, 341-350. https://doi.org/10.1016/j.marpolbul.2018.09.016. Accessed 14 January 2021.

Zettler, E.R., Mincer, T.J. and Amaral-Zettler, L.A. (2013). Life in the "Plastisphere": Microbial communities on plastic marine debris. *Environmental Science and Technology* 47(13), 7137-7146. https://doi.org/10.1021/es401288x. Accessed 14 January 2021.

Zettler, E.R., Takada, H., Monteleone, B., Mallos, N., Eriksen, M. and Amaral-Zettler, L.A. (2017). Incorporating citizen science to study plastics in the environment. *Analytical Methods* 9, 1392-1403. http://doi.org/10.1039/C6AY02716D. Accessed 14 January 2021.

Zhang, D., Liu, X., Huang, W., Li, J., Wang, C., Zhang, D. *et al.* (2020). Microplastic pollution in deep-sea sediments and organisms of the western Pacific Ocean. *Environmental Pollution* 259, 113948. https://doi.org/10.1016/j.envpol.2020.113948. Accessed 14 January 2021.

Zhang, H. (2017). Transport of microplastics in coastal seas. *Estuarine, Coastal and Shelf Science* 199, 74-86. https://doi.org/10.1016/j.ecss.2017.09.032. Accessed 14 January 2021.

Zhang, K., Gong, W., Lu, J., Xiong, X. and Wu, C. (2015). Accumulation of floating microplastics behind the Three Gorges Dam. *Environmental Pollution* 204,117-23. https://doi.org/10.1016/j.envpol.2015.04. 023. Accessed 14 January 2021

Zhang, K., Su, J., Xiong, X., Wu, X., Wu, C. and Liu, J. (2016). Microplastic pollution of lakeshore sediments from remote lakes in Tibet plateau, China. *Environmental Pollution* 219, 450-455. https://doi.org/10.1016/j.envpol.2016.05.048. Accessed 14 January 2021.

Zhang, M., Buekens, A. and Li, X. (2017). Open burning as a source of dioxins. Critical Reviews in Environmental Science and Technology 47(8), 543-620. https://doi.org/10.1080/10643389.2017.1320154. Accessed 20 November 2020.

Zhao, S., Zhu, L., Wang, T. and Li, D. (2014). Suspended microplastics in the surface water of the Yangtze Estuary System, China: First observations on occurrence, distribution. *Marine Pollution Bulletin* 86(1-2), 562-568. http://doi.org/10.1016/j.marpolbul.2014.06.032. Accessed 14 January 2021.

Zheng, J. and Suh, S. (2019). Strategies to reduce the global carbon footprint of plastics. *Nature Climate Change* 9, 374-378. http://doi.org/10.1038/s41558-019-0459-z. Accessed 14 January 2021.

Zheng, Y., Li, J., Cao, W., Liu, X., Jiang, F., Ding, J. *et al.* (2019). Distribution characteristics of microplastics in the seawater and sediment: A case study in Jiaozhou Bay, China. *Science of The Total Environment* 674, 27-35. https://doi.org/10.1016/j.scitotenv.2019.04.008. Accessed 14 January 2021.

Zhu, C., Li, D., Sun, Y., Zheng, X., Peng, X., Zheng, K. *et al.* (2019). Plastic debris in marine birds from an island located in the South China Sea. *Marine Pollution Bulletin* 149, 110566. https://doi.org/10.1016/j.marpolbul.2019.110566. Accessed 13 January 2021.

Zhu, L., Wang, H., Chen, B., Suna, Z. and Xi, Q.B. (2019). Microplastic ingestion in deep-sea fish from the South China Sea. *Science of The Total Environment* 677, 493-501. https://doi.org/10.1016/j.scitotenv.2019.04.380. Accessed 14 January 2021.

Ziajahromi, S., Neale, P.A., Rintoul, L. and Leusch, F.D. (2017). Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics. *Water Research* 112, 93-99. https://doi.org/10.1016/j.watres.2017.01.042. Accessed 14 January 2021.

Zimmermann, L., Dombrowski, A., Völker, C. and Wagner, M. (2020). Are bioplastics and plant-based materials safer than conventional plastics? In vitro toxicity and chemical composition. *Environment International* 145, 106066. https://doi.org/10.1016/j.envint.2020.106066. Accessed 13 January 2021.

Zink, T. and Geyer, R. (2018) Material recycling and the myth of landfill diversion. Journal of Industrial Ecology 23, 541-548. https://doi.org/10.1111/jiec.12808. Accessed 13 January 2021.

Zink, T., Geyer, R. and Startz, R. (2018). Toward estimating displaced primary production from recycling. *Journal of Industrial Ecology* 22, 314-326. https://doi.org/10.1111/jiec.12557. Accessed 14 January 2021.


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