Chapter 9. Micro and nanoplastics in the aquatic environment with special reference to synthetic fibers

By: A. Dick Vethaak[1,2] , C. Martínez-Gómez[3]

1 – Deltares, Marine and Coastal Systems, Boussinesqweg 1, 2629 HV Delft, the Netherlands
2 – Vrije Universiteit Amsterdam, Department of Environment and Health, De Boelelaan 1085, 1081 HV Amsterdam, the Netherlands
3 – Instituto Español de Oceanografía (IEO), Oceanographic Centre of Murcia, Varadero 1, PO BOX 22, 30740 San Pedro del Pinatar Murcia, Spain

9.1 Introduction

Plastics are polymers produced from synthetic solid petroleum- or bio-based materials and often mixed with chemical additives. The annual global production of these polymers has exponentially increased over the past decades, reaching 348 million tons in 2017, and is expected to double by 2050 [1, 2]. Such an increase is related with an important production of waste, of which a fraction will eventually end up in the environment due to poor waste management or inappropriate disposal. Plastics are insoluble in water and have (extremely) slow degradation rates after being discarded. Global leakage of persistent plastic waste flowing into rivers and waterways, and ultimately into the oceans, was estimated in the order of 10 million tonnes per year ($\mathrm{Mt\,y^{-1}}$) [3], resulting in a widespread occurrence and accumulation of microplastic pollution.

Plastic particles in the micrometer-size range have been detected worldwide in virtually all aquatic compartments, such as surface waters, water column, sea floor, coastlines, polar ice, rivers, small waterways and lakes, as well as in a wide range of species (e.g., [4–9]. This raises scientific and public concern on their possible impact on aquatic populations, food webs, and food production for humans. The debate is fuelled by the persistence of the plastics and a projected future increase in microplastic pollution levels [10, 11]. The term microplastics (MPs), whether originating from intentional production (primary microplastics) or fragmentation of larger plastics (secondary microplastics), was coined by Thompson and co-workers in 2004 to describe the widespread occurrence of microscopic plastic particles and fibers in the oceans [12]. Since then, microplastic research has attracted increasing attention in environmental sciences, interlinking them between the fields of macro-plastic pollution and the field of nanoparticle toxicology [13].

Microplastic debris is heterogeneous in nature with a large variety of shapes, from spherical particles to angular fragments or films and long fibers [14]. All plastic particles or fibers in the size range 1–5000 µm (in their largest dimension) are referred to as microplastics (MPs) and those in the size range < 1 µm (in their largest dimension) as nanoplastics (NPs) [14, 15]. Microplastic fibers are a specific type of microplastics defined as any artificial fibrous materials (e.g., acrylic, polyesters, nylon) of threadlike structure with a diameter less than 50 µm, length ranging from 1 to 5000 µm, and length to diameter ratio greater than 100 [16]. Some studies have used different morphological criteria to confirm a fibrous particle, for example "a length to diameter ratio of > 3" [17] or "a length substantially longer than its width" [14].

Field studies to date have mostly focused on plastic particles and fibers in the micro-size range. Due to analytical limitations, nano-sized plastic particles and fibers are not considered in field studies, but they have been commonly used in laboratory studies to understand the behavior and toxic effects of micro- and nanoplastics (MNPs) in aquatic organisms. The bulk of field and laboratory studies concerning environmental concentrations and effects to date have been performed for marine environments, but the impact of MNPs may be equally serious for freshwater and terrestrial environments. However, the occurrence, fate and ecological effects of this type of pollution are still poorly understood. In the past decade, a rapidly growing body of empirical research on aquatic microplastics has aimed at elucidating and understanding the ecological impact of MNPs to provide evidence to inform and support policy makers remediating plastic pollution (e.g., [4, 18–31].

This chapter aims to present a summary of what is currently known about MNP pollution in aquatic ecosystems. Special attention will be given to synthetic fibers, which are a predominant type of MNP pollution. Factors influencing MNP uptake and absorption, their bioaccumulation and ecological effects will be considered as well. Case studies on phytoplankton and bivalves will be highlighted. We conclude by discussing important knowledge gaps that need to be addressed in future work.

9.2 Sources, pathways and sinks

9.2.1 Major sources of MNPs

MNPs enter the aquatic environment from a variety of sources. The weathering and fragmentation of large plastic pieces, such as containers, packaging materials, bottles, and ropes are the most important source of secondary MNPs throughout the world ocean (e.g., [22, 32, 33]). It is estimated that between 1.15 and 2.41 million tons of both macro- and micro-plastic debris enter the ocean every year from rivers, with the top 20 polluting rivers, mostly located in Asia, accounting for 67 per cent of the global total [34]. Wastewater treatment plants are significant sources of microplastics in river catchments with concentrations reaching up to 125 particles per liter [35]. In addition to this, sea-based sources of plastic litter (beach littering, shipping and fisheries) and atmospheric transport and deposition of MNPs, may also be important pathways [16, 22, 36].

Not all MNPs originate from the breakdown of larger plastic items or particles. MNPs can also be generated by erosion and abrasion during the production, maintenance, and use of plastic products. In this case, tyre wear and tear during driving and shredding of synthetic fibers from textiles during washing are relevant and major sources [3, 37]. In their review on wear and tear of tyres, Kole and co-workers estimated that the per capita emission ranges from 0.23 to 4.7 $\mathrm{kg\,y^{-1}}$, with a global average of 0.81 $\mathrm{kg\,y^{-1}}$ [38]. These estimates indicate that car wear and tear, consisting largely of elongated particles of rubber polymers [39], significantly contribute to the flow of (micro)plastics into the aquatic environment with a relative contribution of 5–10% of total global oceanic plastic.

Some portion of MNPs are manufactured as micro or nano-sized particles and directly released to the environment, such as pellets, powders, or microbeads in personal care products (PCP) [22]. However, these primary particles are probably only a small fraction of the total amount of MNPs in the aquatic environment [3].

Well-known are plastic preproduction resin pellets, usually 1 to 5 mm in size and of variable composition. They are found in aquatic samples and on beaches all over the world and their presence is often linked to spillage during transportation or industrial effluents. Other major, but lesser known sources of plastic particles and fibers emitting to the environment are abrasion of polymeric paints from shipping, water works, city dust, and household dust [3, 40]. Microplastic pollution by smaller size micropellets has also been observed. For example, micro-debris detected in European river (Rhine) sediments close to polymer plants contained overall 18% PS pellets (100–200 µm) showing visual and spectroscopic resemblance to primary preproduction pellets/powder [41]. Another very recent example is the shipping accident of the MSC Zoe in 2019 during a storm in the North Sea. The loss of 342 containers of goods, several of which contained 5 mm plastic preproduction pellets (polystyrene and HPPE), resulted in a wide spread

microplastic contamination of Dutch and German islands in the Wadden Sea (a protected area under European Natura 2000 law).

9.2.2 Sources of fibrous MNPs

Synthetic microfibers in the aquatic environment originate from primary textile microfibers that are used in the textile and clothing industry, and the fragmentation of larger fabrics originates during textile production, use, and from discarded textiles [16, 42, 43]. Other sources of fibrous microplastics are fishing nets, ropes, and certain polymeric paints and coatings. Microfibers are easily shed from clothes and household textiles with loose structures, such as fleece sweaters, during daily use [44]. Boucher and Friot [3] estimated that approximately 35% of microplastics in the world's oceans arise from laundry of synthetic textiles. Some estimates are lower, but even a value of 20% for 2014 as reported by Eunomia [45] means that the equivalent to 0.19 million tons of textile microfibers from the production and normal use of synthetic textiles, particularly household washing of clothing, enters the marine environment alone annually (see Henry et al. [43]). Browne et al. [46] found that a single piece of clothing can produce $>$ 1900 synthetic microfibers per wash, while in a recent study by Napper and Thompson [44], it was estimated that the release of synthetic fibers from polyester, polyester-cotton blend and acrylic fabrics into waste water during each use of a washing machine could be as high as 700,000 microscopic fibers. Microplastics can be found in wastewater where approximately 35% of microplastics are thought to be fibers from synthetic clothes [47]. Although treated effluents only contain few microplastics per liter, the high volume of effluents constantly released constitutes a considerable source of microplastic contamination of aquatic systems. However, it is likely that globally over 80% of wastewater is released to the environment without treatment, resulting in an even much greater emission of synthetic fibers [48].

9.2.3 Pathways and sinks

Relevant pathways of MNPs into the aquatic environment are through littering of larger plastic items, road and land runoff, storm water, flooding, soil amendments and irrigation, wind transfer, and atmospheric outfall [3, 26, 49, 50]. Sewage Treatment Plants (STPs) are considered important pathways of MNPs to surface waters [4], especially in parts of the world where sewage water is treated less rigorously. Plastic particles and fibers contained in personal care products (PCPs) and those originating from laundry, tyre wear, etc., will at least partly end up in the STP effluents [6, 35, 51], and thus get released into receiving surface waters. Atmospheric deposition of suspended atmospheric microplastics, especially textile fibers, has been recently recognized as an important input pathway [16]. Marine systems, especially sediments, still appear to be the ultimate sink for microplastics in the environment, but freshwater systems such as estuaries and lakes are also important recipients and reservoirs of microplastics pollution (e.g., [6, 52, 53]. Relatively high levels of MPs can be found in oceanic gyres and enclosed sea water regions, remote mountain lakes, industrial effluents, sites near factories, estuaries, seabed and urban sediments, filter feeding biota, and household dust [24, 52–60]. Generally, aquatic biota, such as filter and deposition feeders may represent relevant reservoirs, as they will temporarily retain plastic particles from other environmental compartments [61].

From the information above, it can be concluded that sources of aquatic MNPs are diverse. A major source of plastic fibers originates from wear, tear, and maintenance of textile and apparel. In addition to direct emissions into surface waters through effluents and raw untreated wastewater, atmospheric transport and deposition is likely to play a role in the distribution of MNPs, notably fibers.

9.3 Composition of aquatic micro- and nanoplastic debris

MNPs in the aquatic environment are highly complex cocktails of contaminants, including polymeric materials, chemical additives, residual monomers, ambient chemical substances that sorb to plastic, and proteins (ecocorona) [62]. Furthermore, the plastic debris serves as a novel substrate for microbial community structures (biofilms) on

their surfaces. The combinations of polymeric materials and associated chemical and biological contaminants are countless making each plastic particle or fiber having its own unique properties [63]. The polymeric, chemical, and biological components of plastic debris, in particular MNPs, are further described below.

9.3.1 Debris polymers

The distribution and abundance of MNPs is inseparably linked to macroplastic debris, largely via the degradation of bulk plastic into smaller particles down to the nanoscale [64]. Aquatic debris polymers, which are most commonly found, are mostly high production volume polymers (or mixtures thereof), such as polyethylene (as both high-density, HDPE, and low density, LDPE), polypropylene (PP), polystyrene (PS), polyvinylchloride (PVC), polyamide (nylon) (PA), polyurethane (PUR), polyethylene terephthalate (PET), polyester resin and glass fiber, and cellulose acetate (CA). Most debris-prone polymers contain a variety of chemical additives (chemicals which are used to polymerize, process or to modify end use properties of plastics), unpolymerized monomer residues and impurities, depending on polymer surface structure and affinity [23, 65, 66]. Less commonly reported debris polymers include polyvinyl alcohol (PVA), acrylic (AC), poly-methylmethacrylate (PMMA), alkyd (AKD), polyurethane (PU), styrene butadiene rubber (SBR) [41, 67, 68]. Polyamide 66 (nylon66), PVA, polyester, and PP are produced as synthetic monofilaments and used in fibrous materials in textiles and apparel [43], and for fishing lines and fishing nets [23]. Considering that overall 8000 different plastic grades are currently in the market, it is likely that the total number of debris polymers might add up to several hundreds (`https://www.campusplastics.com`).

9.3.2 Chemical additives

Besides synthetic polymers or mixtures thereof, a wide range of chemicals are present in plastic as a result of manufacturing specifications. Chemical additives in polymeric and plastic packaging materials include monomers (e.g., styrene), intermediates, solvents, surfactants, plasticizers (bisphenol-A and phthalates), stabilizers, biocides, flame retardants, accelerators, and colorants [69]. Also, the polymerization process may leave trace quantities of residual monomer or low-molecular-mass polymers in the plastic, especially in primary MNPs. Furthermore, upon aging/weathering, polymer degradation products are likely to be formed, for example, as result of UV-exposure. Many of these substances are not exclusively used by the plastic industry and have applications elsewhere as well, for example in textiles and food technology industry [70]. A considerable number of plastic additives have been listed as chemicals of very high concern because of their endocrine disrupting or carcinogenic properties and other toxicological hazards [71, 72]. Once in the water, hazardous additives might rapidly leach out of the plastics to the surroundings and into the food chain, as it degrades. As such, debris plastics, especially macro-sized objects, significantly contribute to chemical contamination of aquatic systems [73–75].

9.3.3 Adsorption of chemical contaminants

MNPs have a hydrophobic nature (i.e., low polarity) with relatively large ratio of surface to volume with a significant ability to highly accumulate a wide range of chemicals [76] either directly from the environment or as a result of biofouling. Under laboratory conditions, PE, PVC, PP, and PS display high sorption capacity for polycyclic aromatic hydrocarbons (PAHs), dichlorodiphenyltrichloroethane (DDT), hexachlorocyclohexanes, chlorinated benzenes, musks, pharmaceuticals, and personal care products [77–82]. Many of the contaminants adsorbed from the surrounding water have toxic, endocrine disrupting and/or immunomodulating properties [23, 83]. Examples are metals [84] and hydrophobic contaminants such as PCBs, PAHs and PBDEs [23, 85] and emerging contaminants such as PFAS [86] and pharmaceuticals [87]. Compared to microplastics, nanoplastics exhibit a large surface area with increased sorption capacity and are able to bind increased levels of organic contaminants [88] and metals [89]. The contaminants present in MNPs may migrate from the particles into the organism by equilibrium partitioning [88] and can cause chemical toxicity [71, 76]. Contaminant transfer is bi-directional and can either increase or decrease contaminant body burden

depending on polymer type, environmental conditions, and chemical fugacity/transport gradients [88, 90]. The above studies show that MNPs could be significant vectors of both organic and inorganic substances, potentially affecting bioavailability and chemical toxicity.

9.3.4 Eco-corona, biofilm and biofouling

In addition to polymers and chemical additives and adsorbed chemicals, MNPs can bind organic matter, nutrients, and also macromolecules (e.g., lipids, proteins, polysaccharides) and may serve as substrates for microbial biofilms, forming a so called eco-corona around the MNPs [62]. Subsequently, MNPs can be colonized by a distinct fouling community of aquatic microorganisms and small invertebrates, such as diatoms, bacteria, cyanobacteria, fungi, unicellular algae, ciliates, dinoflagellates, bryozoans, crustaceans, worms, and marine insect eggs [91–93].

Attracted by nutrients and depending on the polymer type, MNPs develop unique microbial community structures compared to their surrounding environment [93]. Main bacterial colonies found on MPs appear to be established by their adherence properties, such as hydrophobicity and surface roughness, rather than by the type of polymer itself [94], although species able to degrade the specific polymeric material or chemical additives usually also occur in the biofilm communities [95–97]. Thus, additional to weathering processes, some bacteria may also act as plastic degraders and accelerate MP degradation processes [98], although biodegradation in aquatic habitats is recognized to be low [97]. On the other hand, biofilm micro-organisms may produce exudates acting as infochemicals of food occurrence, potentially increasing the ingestion rate of MNPs [62].

An increasing number of studies on (micro)plastic-microbiological research indicate that aquatic (micro)plastics represent a distinct environmental niche that can potentially harbor and disperse pathogenic bacteria [92, 99–103], antibiotic resistant micro-organisms [104, 105], toxic dinoflagellates [106], toxic cyanobacteria [97], thus posing novel health risks to fish and other aquatic organisms and potentially humans as well [92, 99, 100, 103–105].The long decomposition time coupled with lower density of MNPs favor their dispersal in aquatic systems compared to non-plastic particles [107].

It is becoming clear that eco-corona and biofilm formation and subsequent biofouling and plastic biodegradation may change the identity and density of MNPs, and could influence their behavior, fate, bioavailability, and effects [62, 108–112]. Overall, these interactions between MNPs and microbial ecology are far from being understood and require further research.

9.4 Factors that control degradation and fate of polymeric material

Physical and chemical processes, such as exposure to UV light (photodegradation), oxidation, leaching of additives, biofilm growth, and mechanical abrasion (with rocky shores and sandy beaches), initiate plastic degradation and fragmentation, consequently releasing numerous plastic particles from the surface of larger items or fragments [66, 113, 114]. The fragmentation process primarily depends on the type of plastic, the concentrations of antioxidants and stabilizers used as additives [115] and the physical-chemical environmental conditions including contact by biota [116]. Eco-corona and biofilm formation on plastic particles surfaces could prevent the plastic from photo-catalysis, either directly or via decreased buoyancy, thereby promoting microplastic sedimentation. Over time, the polymeric material itself degrades very slowly and consequently, MNPs will persist for decades and even centuries [113, 115, 117]. Decreasing particle size as a consequence of fragmentation will further result in a higher particle abundance and increase encounter probability with biota [23]. Further, nanoplastics can be generated from microplastics through digestive fragmentation by organisms such as Antarctic krill [118]. These effects have potential consequences as the number of particles that are sufficiently small for translocation into the body, but it introduces a new role for biota in the fate of plastic and nutrient cycling and biogeochemical processes. In fact, the likelihood that marine and freshwater biota has not been exposed to MNPs from any source is very low.

The fate of microplastics in the aquatic environment primarily depends on polymer density, particle size and shape which influences buoyancy, and residence time (biofilm development and the potential interaction with detritus, living organisms, and inorganic matter) [18, 119]. Further, the fate of MNPs in the water column is highly affected by the aggregation of nano and microplastic with suspended solids [120]. The densities of the most encountered plastic materials, such as polyethylene, polypropylene, nylon, and polyester typically range between 0.9 and 1.4 $\mathrm{g\,cm^{-3}}$. Polymers less dense than (sea)water (e.g., PE and PP) will tend to float on or within the water surface layers, while those with higher density (e.g., PVC, polyester, acrylics, and polyamide) will sink. Settling/resuspension behavior of MPs is highly dependent on the particle shape (i.e., flat fragments may stay motionless and flexible threads and fibers can be easily captured by the flow) [121]. The transport and distribution of fibrous MNPs are intrinsically differently due to the low sinking rate, delayed settling and low re-suspension threshold compared to non-fibrous plastic particles [122]. Biofilm formation and the settlement of organisms on the plastic surface increase the weight of particles, thus increase their sinking velocity [123–125], with small-sized particles losing buoyancy much faster than larger ones, depending on the shape and especially specific density [126]. Other studies have shown that MPs flocculate and adhere to sticky exo-polymers [119] or be incorporated into faecal pellets [127], influencing their buoyancy and position in the water column. On the other hand, evidence indicates that aggregates in marine waters are involved in removing MPs from the ocean surface and facilitating their transfer to marine food webs [128].

9.5 Physical and chemical quantification and characterization of MNPs

9.5.1 Analysis of microplastics in aquatic matrices

There is currently no ideal analytic method to quantify and chemically characterize plastic particles in aquatic samples ($<$ 5 mm). The major issues are the complexity of environmental samples – to adequately remove the organic fraction (e.g., biological tissues) – and the lower detection limit for particle size. Often one or more extraction and cleaning steps are needed to isolate the microplastics from the surrounding matrix prior to analysis. These extractions and cleaning techniques include visual sorting, density gradient-based separation, filtration, acid and alkaline cleaning, and enzymatic destruction [14, 129–131].

Microplastic measurement results rely strongly on good sampling and sample purification. Therefore, the combination of a separating and qualifying method has been widely used [132]. Visual sorting can lead to large error values (e.g., [67, 133, 134]) in determining the concentration of MPs. Other techniques reportedly generate much higher recovery and identification rates for MPs, although these rates generally decrease as the particle size decreases [135].

Analysis techniques commonly used for the identification of (micro)plastics include Fourier transform infrared microscopy (FTIR), Raman spectroscopy, scanning electron microscopy with energy dispersive x-ray spectroscopy (SEM-EDS), and pyrolysis-based gas chromatography mass spectrometry (GC-MS) [129, 130, 132, 133]. The spectroscopic FTIR and Raman techniques are currently best suited for the detection of plastic particles larger than about 10 and 1 µm respectively, depending on the equipment used [67]. Stimulated Raman Scattering (SRS) microscopy is a powerful spectroscopic imaging technique based on the synchronized action of two pulsed laser beams with a specific photon energy difference. Compared to conventional Raman spectroscopy, it allows a thousand-fold faster mapping of, for instance, particles on a filter surface. A multispectral SRS-approach of this method was recently developed for MPs in the low-micrometer size range from harbour sediment, although this technique needs improvement to be used for small sized particles in a high throughput mode [136].

The advantages and limitations of the various methods are discussed in detail elsewhere [14, 132, 137]. A major limitation in the analysis of MPs in complex environmental (organic rich and weathered plastics) matrices is the lack of standard operating procedures for detecting and quantifying microplastics. Clearly, there is a need for harmonized protocols for sampling, analysis, and identification of microplastic particles and fibers. Further research is required to determine whether any of the currently available analytical techniques can be developed to the extent where they can be routinely used for MNP exposure assessment the entire range of sizes. Most suitable methods for analyzing

microplastics reported in aquatic matrices have been reviewed and recommended by GESAMP [14].

Field studies on microplastic abundance reported in the literature contain mostly particle counts and less often quantitative data on polymer types. Environmental concentrations are usually expressed in particle number per unit (volume, surface area, mass) and rarely in mass per unit. The physico-chemical heterogeneity (i.e., in polymer type, size, shape) of MPs found in field samples implies that the conversion between particle and mass-based concentrations is fraught with imprecision. On the other hand, mass-based concentrations are commonly reported in laboratory studies using test MNPs, exhibiting more homogenous characteristics. It would be preferable, when possible, to record both number and mass (e.g., using pyrolysis GC-MS) to facilitate comparisons of quantities between micro-sized (high mass) and nano-sized (high numbers) debris. From a toxicological perspective, particle numerical concentrations are preferred, because these represent the number of MNPs biota are exposed to. For further discussion, see Skåre et al. [31].

9.5.2 Analysis of microplastic fibers and nanoplastics

Fibers may even be more difficult to count and qualify than non-fibrous particles and some recent reviews have questioned the effectiveness of microfiber identification [138–140]. There are several issues with the accurate identification of synthetic microfibers. Natural (e.g., cotton, flax, wool, linen, silk, and hair) and artificial (regenerated) cellulosic microfibers (e.g., viscose (or rayon), bamboo and acetate fibers) have similar morphological properties, but are not plastic [141]. For example, the difficulty of separating natural and synthetic microfibers following visual inspections may result in misidentification of synthetic microfibers to natural fibers [140] or misidentification of natural fiber (e.g., cotton, hemp) to manmade cellulosic fibers (e.g., viscose) [138]. Spectroscopic analysis is essential to ascertain the plastic nature and differentiate the different types of fibers. On the other hand, fibers are often too thin and their abundance in water samples taken with bigger net mesh sizes is therefore often likely to be underestimated [41]. A simple and effective step before spectroscopic analysis seems to be the use of Nile Red (9-diethylamino-5H-benzo[a]phenoxazine-5-one), offering the possibility of selective identification of plastics, based on surface polarity characteristics of plastic particles, including small MPs (< 100 µm) [142, 143]. For recommendations on how to accurately differentiate types of fibers, the reader is referred to the work of Woodall et al. [144] and Comnea-Stancu et al. [138].

Due to methodological challenges, the smaller sized MNPs, such as sub-micron or nano-sized plastics, are yet to be detected in environmental field samples or in tissues of field-collected organisms [14, 145–147]. Advanced separation techniques, such as asymmetric flow-field-flow fractionation (AF4), might also be applicable to nano-sized plastic particles and fibers. Similarly, electron microscopy-based analytical techniques might also be helpful in visualizing nano-sized particles and fibers, although this technique probably would have to be combined with other analytical techniques, such as pyrolysis GC-MS, for sub-micron-sized particle identification. Possible routes for developing adequate sampling preparation and analytical techniques for the detection of the yet unanalyzed MNPs could come from the field of engineered nanoparticles and fiber toxicology.

9.5.3 Uncertainties in aquatic MNP measurements

The reliability of some early microplastic results is questioned [134]. This is because cross-contamination is likely to occur at each stage of sampling, handling, and analysing microplastics samples, as demonstrated by various studies (e.g., [134, 144]). In most instances, investigations pinpoint that microplastics were most likely acquired from worker clothing, unwashed lab equipment, and general contamination from airborne microplastics in the laboratory. Recommendations on avoiding contamination and applying quality criteria for the analysis of microplastics in biota, including the use of procedural blank analysis and clean air devices, have been described by [134].

9.6 Occurrence of microplastics in aquatic systems

9.6.1 Microplastics in abiotic matrices

Concentrations reported in water vary from a few particles up to thousands of particles m^{-3} [26, 148]. Microplastic concentrations in suspended matter and sediment can reach several thousand particles kg^{-1} of dry sediment [148–151]. In a study in Dutch coastal and inland waters, microplastic analysis (particle sizes between 10–5000 µm) in major rivers Rhine and Meuse and urban water in and around Amsterdam in 2014, showed concentrations between 100 and 3600 particles kg^{-1} of dry sediment along the Dutch North Sea coast and between 68 and 10,500 MPs kg^{-1} in urban canal sediments. Amsterdam canal water samples contained microplastic concentrations (48–187 MPs l^{-1}) similar to those observed in treated wastewater effluent in the area. These data show that MPs do settle in urban waters, meaning that at least a part of the MPs is retained in freshwater systems. On the other hand, microplastic concentrations in riverine suspended particulate matter appearing to be high (1400–4900 MPs kg^{-1} dry weight (dw)) [6], substantiating previous studies that rivers are a key factor for microplastics to move towards the ocean [26, 152].

In aquatic samples, PE, PP and PS are the most frequently found polymers [24], but also a range of other polymer types have been occasionally identified, for example PET, nylon, polyether urethane, styrene acrylate, alkyd, styrene butadiene rubber, and polymethyl methacrylate [41, 136, 153]. Haave et al. [60] identified twenty different polymer types, at concentrations from 12,000 to 200,000 MPs kg^{-1} dw. Over 95% of the MPs were smaller than 100 µm and commonly consisted of polyurethane acrylate resins, while the larger MPs consisted mostly of polyamide fibers.

Several studies indicate fibers to be a most common type [23]. However, this may be partly due to cross contamination of samples as discussed above. Burns et al. [27] reported that the most abundant shape type in water and sediment is fibers (48.5%), followed by fragments (31%), spherical beads (6.5%), films (5.5%), and foam (3.5%). However, till date, published microplastic data for aquatic matrices are only partly complete and as previously explained, limited to the upper end of the particle size spectrum. In addition to this, published field and monitoring studies often do not mention any specific aspect ratio to distinguish fibrous from non-fibrous particles (e.g., irregular shaped thin fragments or films). As microplastic measurements can be fraught with uncertainties, the representativeness and reliability of obtained data can be compromised. Moreover, existing microplastic data collected from aquatic systems may vary considerably largely because different methods and level of QA have been applied [67], hampering useful comparison. Generally, methods that can sample or detect smaller particles, such as mesh size effects on sample composition, will result in higher concentrations [65, 154]. These studies indicate the significance of the unanalyzed MNP fraction and the need for improved methods for MNP sampling and analysis. To illustrate, microplastic measurements in marine and coastal surface waters have often been limited to manta trawl or neuston samples using nets with > 0.3 mm mesh, underestimating the actual concentrations of microplastics that are likely to accumulate at the sea surface.

Concentrations of MPs in marine and coastal sediments that reach from zero to several thousands of particles kg^{-1} (dw), show generally higher concentrations at more inland sites. For example, the average amount found across the SW North Sea and Channel region in 2014 ranged between 200 and approximately 700 particles kg^{-1} (dw), with levels up to 3,146 particles kg^{-1} in sediment in harbor areas (particle sizes between 10 and 5000 µm). In North Sea sediments, mainly fibers and spheres were found, whereas at the sea surface, fragments were dominant. Several studies have shown that the concentrations of microplastics at the sea surface are lower and more variable than in sediments, indicating that most microplastics including fibers eventually do sink to the seabed supporting the hypothesis that the seabed may act as an important long-term sink [36, 142].

Gago et al. [155] reviewed the presence of microfibers in marine water and sediments and concluded that their occurrence is worldwide and represent a dominant fraction of the total microplastics, in some samples up to 100%. Most common colors were blue (seawater and sediments), transparent and black (seawater), and black (sediments); most common polymers were Polypropylene (water and sediments), Polyethylene (water), Polyester (water and sediments), HD/LD polyethylene and cellophane (only in waters), and rayon (sediments). Concentrations ranged from

0 to 459681 items km^{-2} in sea water and from 12 to 160 items kg^{-1} (dw) in sediments, with blue and polypropylene microfibers as the most abundant microfiber type.

Several studies indicate that microfibers have reached the deep waters of the Mediterranean Sea (3500-meter-deep) and Atlantic (up to 4844-meter-deep) [156], and Indian and West Pacific Ocean (5768 meters deep) [157], where they can be found in similar concentrations as in coastal sediments [158], most likely via the supply of organic detritus "marine snow" containing these particles [55]. Long range transport of synthetic fibers from southern European coastal waters to open deep sea was investigated by Sanchez-Vidal et al. [33]. The measured abundances and composition of microfibers in sediments suggested a downslope transport from coastal seas via submarine canyons to deep open sea floor. They found abundances of 10–70 microfibers in 50 ml of sediment, including both natural and regenerated cellulose, and synthetic fibers; the plastic microfibers consisted of polyester, acrylic, polyamide, polyethylene, and polypropylene.

It can be concluded that microplastics, notably fibers, and also fragments, films and pellets, or granules, are found widespread in all abiotic compartments (surface water, water column, suspended matter, sediments) of marine and freshwater systems. Microplastic concentrations reported for fresh water systems and estuaries are mostly largely similar to those reported for marine and coastal environments, depending on the location, while in some deep-sea sediments microplastics have already reached concentrations similar to those reported for intertidal and shallow sub-tidal sediments [55, 57].

9.6.2 Microplastics in aquatic biota

The occurrence of MNPs in marine biota and commercial marine species have been well documented. Numerous studies show that MPs are ingested by an increasing number of marine species, including invertebrates (protists, small and large crustaceans, mollusks, annelids, echinoderms, cnidarian, and tunicates), pelagic and demersal fish [159], turtles [160] and large top predating marine fish, mammals [161], and penguins [162]. For more comprehensive reviews, the reader is referred to Kershaw et al., Wesch et al., Lusher et al., O'Connor et al. [23, 140, 163, 164].

Taylor et al. [158] provided convincing evidence that microplastics are present within deep-water organisms, and thus have become infiltrated into deep ocean ecosystems. Screening for microplastics and applying major quality control measures, they found 15 rather different plastic fibers ingested or internalized by diverse organisms with different feeding mechanisms. The fibers were made of modified acrylic, polypropylene, viscose, polyester, and acrylic. An increasing number of studies also reported MP consumption by freshwater species such as duck mussel [165], tadpoles [166], wild gudgeons [167], fish eating birds such and cormorants [168]. Overall, reported microplastic data indicate that almost all aquatic species and phyla analyzed to date show plastic particles in their digestive system or tissues. Further, it clearly shows that MNPs have infiltrated aquatic food chains through various pathways, ingestion, and trophic transfer [169].

The concentrations of microplastics in biota generally depend on the habitat and feeding modes [140, 163], but one study on fresh water invertebrates found MPs ubiquitous across different taxa and sites independent of feeding guild and biological traits [170]. Only few studies did not encounter any microplastics in a particular fish or invertebrate species [41, 158, 171].

Table 9.1.: Percentage of fibrous and other and not specified shapes in measurements of marine shellfish and fish (based on data from Barboza et al. [172], Table 2).

Marine food	No of species (No of measurements)	% Fiber/ filament	Other shapes	% Not specified
Shellfish (whole)	23 (25)	87.0	4.3	8.7
Fish (stomach)	69 (90)	78.8	19.1	2.1

Analysis of large number of marine shellfish (whole organisms) and fish species (stomach content) showed the contribution of ingested microplastic fibers to total plastic to be 87% and 78.8% respectively (Table 9.1). A similar conclusion is reached for MP burden in freshwater biota where available field studies report microplastic fibers ranging from 46.6% to 100% (for references, see O'Connor et al. [164].

Most species analyzed for microplastics contained plastic particles or fibers in their stomach content or tissues. For example, field analysis revealed that 8 out of 9 tested invertebrate species from five different phyla from the North Sea and 68% of analyzed individuals of brown trout (*Salmo trutta*) from the Swedish West Coast had microplastics in them (particle sizes between 10–5000 µm). Despite the poor analytical recovery of the used method, several trends emerged from these data. Among invertebrate species, highest concentrations were found in suspension and filter feeders, such as brittle star, sponges, and mollusks. Grazers analyzed for microplastic content, such as isopods, *Patella vulgate*, and *Littorina littorea*, and also secondary consumers (*Hemigrapsus sanguineus*), showed consistently lower concentrations, and in one species (*Patella sp.*), no microplastics were found. These findings compare well with those reported in other studies. Biota subsamples analyzed with FTIR confirmed various polymer types including PS, Alkyd resin, poly (methyl methacrylate) [41]. Other studies reported variable MP contents in various fish species, such as 36.5% of specimens of pelagic and demersal fish species in the English Channel [173]. Analyzing sprat from the Belgium coast, it was found that an average of 39% of individuals had ingested microplastic (78% synthetic fibers) (Figure 9.1). Individual whole shrimps from the North Sea contained 63% ingested or internalized microplastics (96.5% synthetic fibers) (Figure 9.1). In shrimps, blue fibers were found that could be related to shrimp trawl nets These values are difficult to compare as they might be biased and use different identification and qualification methods. However, they support claims that in marine biota, fibers are an important and not rarely most common contaminant.

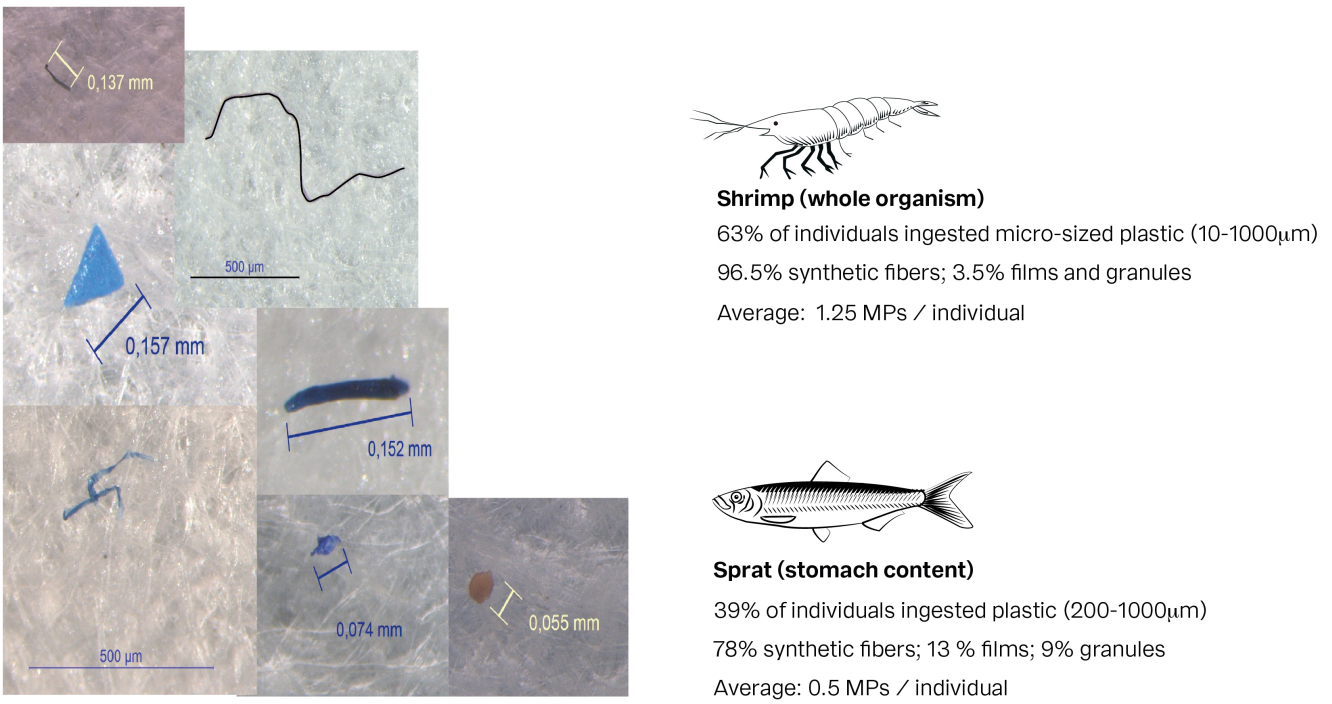

Figure 9.1.: Microplastic (MP) contamination in shrimp (*Crangon crangon*) and sprat (*Sprattus sprattus*) from the North Sea. Based on data from Devriese et al., Zoeter Vanpoucke [174, 175]. Photos ©ILVO.

Microplastics were analyzed in the freshwater duck mussel *Anodonta anatina* from a Swedish river by Berglund et al. [165]. All analyzed mussels contained both fibrous and nonfibrous microplastics. Higher abundances of microplastics were observed in larger mussels indicating a size-related uptake effect. The authors reported higher

concentrations of microplastics downstream urban areas with wastewater treatment plants and anthropogenic activities. Comparing the levels of MPs along a salinity gradient in Dutch waters, the MP concentrations from the North Sea coast to Lake IJssel was similar to the MP concentrations going inland wards. MP levels are approximately similar in freshwater mussels compared to marine mussels (Figure 9.2). In the Dutch study performed by [41], whole body concentrations reported in North Sea mussel ranged from 5–37 MPs per gram (dw) (particle sizes between 10–5000 µm), and 50% of the MPs were fibers. Based on the number of MPs per Kg (dw), the MP concentrations were approximately a factor 1000 higher than in surrounding surface water and sediment, with water, sediment and biota containing 51, 34 and 25% fibers respectively [41]. Several studies have indicated the spatial association in coastal and fresh waters and human activities [4, 5, 142, 148, 165, 176, 177]. A study on riverine macroinvertebrates (*Baetidae*, *Heptageniidae*, and *Hydropsychidae*) found that any increase in MP abundance in biota downstream of sewage treatment works, is possibly due to river flow dilution at the time of sampling [170].

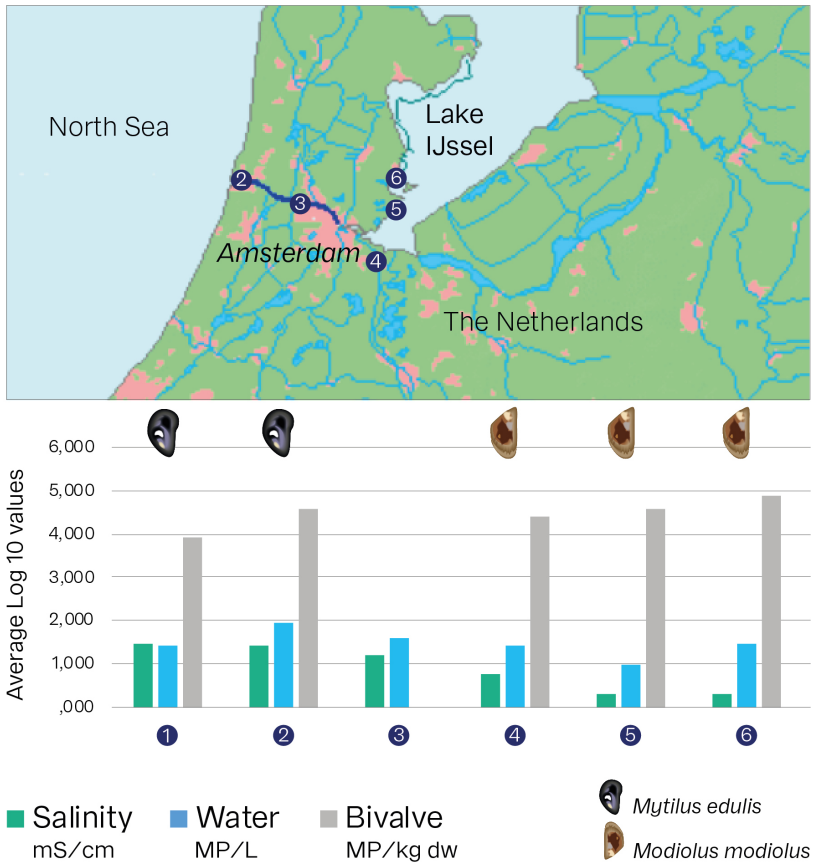

Figure 9.2.: Microplastic particle concentrations in water (MPsl^{-1}) and mussel (MPskg^{-1} dw) along a salinity gradient. Microplastics contained 25% synthetic fibers. (Based on data from Karlsson [41, 178].

The relatively high concentrations of MPs found in commercial marine species such as fish and shellfish have raised important questions about potential risks in higher trophic level species and human food safety [28, 172, 179–182]. Commercial mussels and oysters especially appear to contain high levels of microplastics, and estimates indicate that, for example, European consumers of shellfish ingest 1800 - 11,000 microplastic particles (size range 5–1000 µm) per year [183]. Therefore, oral exposure to MNPs is believed to occur through the consumption of sea products, but may also occur through the consumption of livestock fed fishmeal [23], tap water [184], commercial salts across the world, and in several terrestrial foodstuffs such as honey, beer, and sugar (reviewed by Barboza et al., Yang et al., Joon [172, 185, 186] as well as inhalation of air-born MNPs [187]. It was recently demonstrated that the risk of plastic ingestion

via mussel consumption is minimal compared to the fiber exposure during a meal via indoor dust fallout [188]. On basis of currently available data, the risk of plastic ingestion via seafood consumption is relatively small, and at these levels, it is unlikely that MP exposure will represent a significant risk to human health.

To conclude, the current data clearly show that MPs are widespread among aquatic populations and have penetrated aquatic food chains and potentially, humans. Overall, fibers are the most prevalent microplastic type found in aquatic biota.

9.7 Uptake and effects of MNPs on aquatic biota

9.7.1 Uptake

The ingestion of MNPs by aquatic biota can occur either by directly ingesting the particles from the water or from the sediment or indirectly through trophic transfer. Not readily visible plastic particles or fibers are of special concern, because they can be ingested by biota more readily than larger particles. Organisms can either preferentially feed on MPs or can accidentally ingest them. Predators, detritivores, and scavengers may indirectly ingest plastic while consuming prey or scavenging detrital matter (e.g., marine snows) containing microplastic [128, 163]. Microplastics occupy the same size fraction as some planktonic organisms and may be ingested by a wide range of organisms as outlined above. There is evidence for both size-selective [189, 190] and non-selective ingestion [191], as well as avoidance of microplastics by different species [192]. Plastic particles may also adhere on gills, external appendages or stick to the outside of the body, as on algae and deep-sea coral.

Numerous laboratory studies show that particle size is key, but particle numbers, particle shape, charge, hydrophobicity, composition, and other factors such as aggregation of particles and presence of eco-corona may play an important role in causing effects [193, 194]. Once ingested and/or internalized, plastic particles and fibers, especially the nano-size fractions, can cross cell membranes and enter the circulation where they may reach tissues and cells and induce harm, i.e., particle toxicity, including cytotoxic, oxidative stress, DNA damage, and inflammatory responses. Major sites of entry upon ingestion of MNPs include the Peyer's patches in the intestine [195]. Some studies focused on the potential mechanism of uptake of MNPs. Phagocytosis or endocytosis is possible, whereas the paracellular transport can be excluded given the size of MNPs. Based on what we know from mammalian and aquatic model studies, it can be expected that micro-sized plastic particles, owning to their size, will largely remain in the intestinal tract and excreted through faeces, while smaller microplastics (< 20 µm) and submicro-sized particles can be absorbed by the gut and retain in circularly fluids; the very small particles (e.g., < 100 nm are readily absorbed by the gut and are capable to penetrate cells [195–197]). Once within the organism, MNPs interact with serum soluble components, forming a "protein corona", which may confer a biological identity to MNPs and affects their interactions with target cells [198, 199]. There is only limited evidence that MNPs can transfer from the gut into the circulation [200–202], cells [201], organs [203], and brain [203, 204]. Unfortunately, the size-fraction of MNPs capable of being internalized within fluids and tissues of aquatic organisms largely corresponds with the unanalysed fraction of these particles in field studies, thus seriously hampering risk assessment. Generally, smallest diameter particles have prolonged retention times and higher bioavailability [205]. Smaller MNPs, including nano-sized particles, which have the capability to be internalized by organisms, will have a higher probability of ingesting and biomagnifying up the food chain [169]. It is possible that larger fibrous MPs that have a higher risk of being entangled in the gut and intestines may result in a higher probability for trophic transfer than non-fibrous particles of similar size, but this requires further research. Ultimately, this implies that each plastic particle type exhibits different gut uptake rates and pathways.

Several studies provided insights into the biokinetics of MNPs. A study by Sendra and co-workers with Mediterranean mussels (*M. galloprovincialis*) showed size dependent accumulation of PS NPs (50 nm, 100 nm and 1 µm) in tissues and provided evidence for the translocation of these particles into the hemolymph [202]. The smallest PS NPs tested were detected in the digestive gland and muscle. A fast and size-dependent internalization of particles to the hemolymph was recorded after 3 h of exposure, and this process seemed to be controlled by different endocytic

pathways (i.e., caveolae and clathrin mediated endocytosis and phagocytosis). Al-Sid-Cheikh et al. [206] provided evidence that particle size influences the biokinetics of nanoplastics in a mollusk species. In their experiment the uptake, whole body distribution, and depuration were studied in *Pecten maximus* at environmentally realistic concentrations (< 15 µg MPsl^{-1}) using pulse exposures of ^{14}C-radiolabeled nanopolystyrene (24 and 250 nm. The results suggest relatively rapid uptake of both particle sizes after 6 h, with 250 nm particles present in intestine and 24 nm particles dispersing throughout the whole body, indicating possible translocation across epithelial membranes. Model extrapolations indicated that it could take 300 days to reach equilibrium in scallop body tissues, although the MP concentrations would still be below 2.7 mg g^{-1}. A comparison with a previous work, in which scallops were exposed to nonplastic (silver) nanomaterials of similar size (20 nm, suggests that nanoparticle composition may also influence the uptake. There are some further examples in the literature on how particle surface properties may influence uptake and associated effects. Proteins secreted by *Daphnia magna* around polystyrene nanoplastics with charged and modified surface created an eco-corona which increased the NP uptake, ecotoxicity, and caused a longer gut retention time with effects on feeding rate [111]. However, the environmental relevance of these surface modifications is questionable, since such modifications are not expected to be part of the natural weathering process of MNPs. Fotopoulou et al. [207] observed significant alteration of functional groups on the surface of eroded PE pellets, whereby the altered surface acquired a negative charge due to ketone groups. Particles of weathered PP, on the other hand, were reported by these authors to remain neutral. Severe toxicity was observed in sea urchin embryos (*Paracentrotus lividus*) after incubation with amine (PS-NH_2) polystyrene, and in a lesser extent with carboxylated (PS-COOH) polystyrene. Both nanoparticle types, notably PS-COOH, aggregated partially with time and only PS-COOH accumulated inside embryo's digestive tract [208]. However, when these NPs encountered biological fluids (i.e., coelomic), they became covered by a protein-corona, which conferred identical colloidal features to PS-NH_2 and PS-COOH particles, both acquiring a monodisperse size distribution and a negative potential in the coelomic fluid [209]. It has been also evidenced that NPs translocated to mussel hemolymph after ingestion may specifically bind to plasma proteins, forming a protein corona that can affect particle uptake and toxicity in target cells [198] (see above).

Overall, the extent to which MNP particle translocation occurs, the uptake mechanism, tissue distribution, and potential accumulation of particles are still poorly understood and represents a major obstacle in assessing their risk in aquatic organisms. The associated effects following intake of MNPs will be discussed further below.

9.7.2 Physical effects

Physical effects of MNPs

Although a few laboratory studies report little or no physical harm to aquatic biota [210–213], by far, most laboratory studies reveal adverse effects on marine and freshwater organisms at various levels of biological organization, from changes in gene expression, inflammation, and energy allocation to effects on reproduction, immune system, and central nerve system to death [23, 214, 215] (Table 9.2). In several of these cases, it is not clear if the effect is particle-related or caused by associated toxic co-contaminants or impurities. An example of a typical physical effect is perhaps the study of [18] exposing the polychaete worm, *Arenicola marina* (a deposit feeder) to unplasticised polyvinylchloride (UPVC) hold in sediments spiked at concentrations coinciding with high levels found in the environment. The worms showed reduced feeding activity, increased gut residence time of ingested particles, inflammation and significantly depleted energy stores by up to 50% compared to control. Such energy-reduced effects, in the above case mainly linked to a reduction in lipid reserves, can have an impact on growth, reproduction, and ultimately, survival. Other studies have shown similar effects on energy budget, but negative effects on the fecundity and reproductive, immune system, and behavior have also been reported. Immunological effects have been reported for micro- and nano-sized plastic particle interactions with marine invertebrates by several research groups, indicating that in marine invertebrates the immune function represents a potential target for plastic (polystyrene) nano-plastics [202, 216].

Table 9.2.: Selected examples of frequently reported effects observed in marine and freshwater biota after exposure to MNPs in laboratory studies.

Species	Observed effect	Reference
MARINE BIOTA		
Marine bacteria	Effects on ecological function, oxidative stress, grow inhibition	[217]
Microalgae	Reduced chlorophyll content	[210]
	Lower feeding rates; negative effects on growth and photosynthesis	[218]
	Reduced growth	[219]
	Inhibition of photosynthesis and promotion of ROS	[220]
Ascidians	Slowed down metamorphosis, growth inhibition	[221]
Sea urchin	Developmental defects	[208]
Copepod	Reductions in fecundity	[222]
	Increased immune response, lower feeding rates	[223]
Blue Mussel	Oxidative stress, reduced lysosomal membrane stability, cell damage, inflammation, granuloma formation	[201]
	Damaged ecophysiological functions	[224]
Mediterranean mussel	Alterations of immunological responses, lysosomal compartment, peroxisomal proliferation, antioxidant system perturbations and genotoxicity; altered gene expression	[225]
Brown mussel	Impaired larval development	[226]
Asian green mussel	Reduced filtration behavior, respiration rate and byssus production	[227]
Pacific Oyster	Immune system modulation and alteration of homeostasis	[228]
	Decrease in fertilization and embryogenesis success	[229]
Pearl oyster	Impact on assimilation efficiency and energy balance, with negative repercussions on reproduction	[230]
Lugworm	Effects on fitness and bioaccumulation, reduced feeding capacity	[210]
	Reduced feeding, weight loss and oxidative stress	[224]
	Reduced growth, inflammation	[18]
Oyster, sea urchin, copepod	Decreased fecundity; negative impacts on subsequent generations	[223, 231–233]
FRESHWATER BIOTA		
Microalgae	Growth inhibition (enhanced by enhanced growth), oxidative stress	[234]
	Growth inhibition	[219]
	Inhibited algal photosynthesis and growth; promotion of ROS indicative of oxidative stress	[220]
Polyp	Significant changes in morphology	[235]
Copepod	Effects on fecundity, survival and feeding	[236]
Nematod	Significant reduction in survival, decreased body length and reproduction impairment	[237]
Water flea	Decreased survival	[238]
	Lower feeding and reproduction at high microplastic levels	[239]
	Inhibited reproduction and induced abnormal embryonic development	[240]
Perch	Inhibition of hatching, decreased growth rates, and altered feeding preferences and innate behaviors	[241]
Crucian carp	Effects on behavioral and fat metabolism	[108]
	Brain damage, behavioral disorders	[204]
Common goby	Decrease in predatory performance	[242]

Continued on next page

Table 9.2 – continued from previous page

Species	Observed effect	Reference
Zebrafish	Increased activity of antioxidant enzymes inflammation and lipid accumulation in liver, altered metabolic profile in liver	[243]
	Intestinal damage, including cracking of villi and splitting of enterocytes	[237]
	Inhibited larvae locomotion	[244]

More toxic effects of smaller microplastics below 10 µm were reported in aquatic organisms, compared to plastic particles of larger size. The majority of lab exposure studies have used nano-sized plastic particles that appear to be more readily absorbed and accumulated in tissues of aquatic species [204, 245, 246]. Several studies indicate that microplastic fragments or irregular fluffy microplastics were readily ingested by different species [201, 247, 248], and in some cases effects of these irregular shaped microplastics were observed, while by comparison spherical microplastics did not reveal any effect [248], indicating that particle shape, in addition to their separate influence on uptake, can influence the effects of MNPs. In box 9.1 and 9.2, we have summarized laboratory studies of MNPs on phytoplankton and marine mussel respectively, which elucidate insights into possible mechanisms underlying the observed microplastic effects.

Box 9.1: Effects on phytoplankton

The risk that MNPs may affect autotrophic phytoplankton that form the basis of the aquatic food chains and represent the major global source of oxygen to the atmosphere is of particular concern. Several studies have demonstrated negative effects on marine and freshwater phytoplankton species under laboratory conditions. Negatively charged nano-PS can sorb to microalgae, inhibiting microalgal photosynthesis and promoting ROS [220], Furthermore, nano-PS (0.22 and 103 $mg l^{-1}$) exposure to microalgae affects the growth and photosynthesis in terms of reduced chlorophyll content [210]. No effects of negatively charged and uncharged PS particles (0.05, 0.5 and 6 µm) on microalgal photosynthesis were noticed. However, microalgae growth was negatively affected (up to 45%) by uncharged PS particles, but only at high concentrations (250 $mg l^{-1}$). These effects were demonstrated to increase with decreasing particle size [219]. In another study a negative effect of PVC particles on growth (up to 39.7%) was seen only at high concentration (50 $mg l^{-1}$). Toxic effects were likely to be caused by absorption and aggregation [218]. The above findings are not always consistent, for example, Besseling et al. [210] found effects on the growth and photosynthesis, while Sjollema et al. [219] and Lagarde et al. [249] observed no changes in algal growth rate but a significant reduction in photosynthesis. These differences may be due to different experimental test conditions. Lagarde et al. [249] presented the first evidence on microplastic-induced molecular toxicity in freshwater microalgae, *Chlamydomonas reinhardtii*. Exposure to high density polyethylene (HDPE) and polypropylene (PP) particles induced hetero aggregates consisting of microplastics, microalgae, and exopolysaccharides, eventually leading to vertical pathway transfer of microplastics from surface water to sediment. This was evidenced by elevated expression of genes (UGD and UGE) involved in rhamnose and xylose synthesis representing exopolysaccharide biosynthesis pathway elevated gen-expression [249].

Box 9.2: Effects on marine bivalve mollusks

The biology of filter feeding bivalves such as mussels and oysters makes them particularly suitable models for investigating the fate and effects of MNPs in organisms. Bivalves have a highly developed phagocytosis process for the cellular internalization of particles from 0.1 to 100 µm to intracellular digestion and cellular immunity [250], and they can select their food particles using various criteria, including size [18, 251]. Internalization of

MPs in bivalves can occur by means of two routes: i) MPs can be trapped by gills through microvilli activity and endocytosis processes, and; ii) they can be taken up via ciliary movement in the digestive system and subsequent translocation to the hemolymph (reviewed by Franzellitti et al. [29]. Bivalve pallial cavity is of vital importance as a portal and barrier to invade microorganism and microparticles, and the mucus layer secreted by pallial organs (gills, mantle, etc.) generally allows to effectively capture particles as small as 3–4 µm and smaller [18]. Therefore, bivalves are effective concentrators of MNPs from the surrounding environment.
The ingestion, bioaccumulation, and translocation of plastic microparticles by bivalves, with particularreference to mussels, has been demonstrated in numerous studies [252–255]. Physical blockages and biological changes have been described in *Ostrea edulis* by ingestion of MPs [256]. Polystyrene microspheres filtered by *Mytilus edulis* were found to be accumulated in gut and digestive tubules and subsequently translocated into hemolymph and hemocytes [200]. A significant reduction in the filtration rate was observed in mussels exposed to microplastic fibers (459 ± 2.25 (SE) µm), although more than 70% of fiber uptake by mussels were also found to be quickly rejected as pseudofeces, with approximately 9% ingested and $< 1\%$ excreted in feces [257]. There are evidences pointing that small MPs in the digestive gland are slower processed and eliminated than larger ones, and that a translocation of small MPs occurs from the digestive system to the gills [197]. Indeed, there is evidence indicating that bivalves are more likely to consume smaller MPs instead of larger microfibers [258]. More recently, microfibers have been observed in foot, mantle, and fused in to the byssus of mussels in experimental exposure experiments, and the adherence and fusion to these and other organs has been proposed as a novel way for mussels to uptake microplastics beyond ingestion, contributing about 50% of the microplastic uptake in mussels [259, 260].
Innate immune defence in mussels is comprised of cell-mediated and humoral mechanisms, in which hemocytes are primarily responsible for defence against parasites and pathogens. Consequently, immune functional parameters of bivalves are used as a model for investigating the major mechanisms involved, modes of actions and effects of MNPs on innate immune responses in organisms. Several *in vivo* studies have found that experimental exposure of *Mytilus galloprovincialis* to virgin MNPs may trigger upregulation of lysosome and immune-related functions, such as changes in hemolymph total cell counts, phagocytosis-related activities, extracellular lysozyme and oxyradical production and cause significant effects on the cellular (lysosomal membrane destabilization, genotoxic effects) and tissue level responses associated with infiltration of immune cells (suggestive of inflammatory processes) besides a modulation of antioxidant defenses and detoxification enzymes [201, 225, 261–263].
In vitro studies hemocytes of *M. galloprovincialis*, which have also evidenced immunomodulation (including increase in extracellular reactive oxygen species and nitric oxide production) and apoptotic processes induced by cationic polystyrene nanoparticles (PS-NH_2 NPs) [216]. The formation of a protein-corona complex around cationic polystyrene nanoparticles incubated in hemolymph serum has been described [264]. The authors found that cellular/lysosomal damage, ROS production, and p-p38MAPK levels were higher when cells were exposed to nanoplastics in hemolymph serum than in seawater, demonstrating that biological fluids can affect NP impact in marine species [264]. Lastly, it has been found that exposure to virgin micro-PS leads to an imbalance of energy reserves causing feeding alterations and reproductive disruption in oysters *Crassostrea gigas* (significant reductions in oocyte number and decreased sperm velocity) with impacts on offspring (significant reduced larval count and larval development) [231]. However, the ecological relevance of the observed sublethal effects under laboratory conditions is limited or remains unclear for reasons which are later discussed (see paragraph 9.7.5).

Physical effects of synthetic fibers

By far, most lab studies reported in the literature used regular shaped microplastics, i.e., microbeads of spheres, and not microfibers. Laboratory exposure studies using fibers are largely limited mostly to crustaceans, mostly small species (Table 9.3). Reported effects including mortality are limited to crustaceans, in particular amphipods and copepods. The majority of ingested microplastics reported in field studies were fibers, and several studies indicate that this shape of microplastics appear to have a higher potential than other forms of microplastics to enter the food chain (see above). It is unclear whether this shape is more bioavailable and may also influence the severity of resulting biological effects due to increased risk for entanglement and increased gut passage time. Differences in biological response due to the shape of microplastics, especially fibrous shapes, are however not well understood. Compared to other particle shapes, fibers may have (1) a greater potential to entangle the gut contents or externalities; (2) increased gut passage time; (3) increased chemical reactivity (and resulting effects) due to relatively large surface area to volume ratio. More research is needed to confirm whether entanglement of fiber-shaped microplastics could worsen adverse effects [13, 18].

Table 9.3.: Observed effects in marine and freshwater biota after exposure to synthetic fibers in laboratory studies.

Species	Exposure conditions	Observed effects	Reference
Marine lobster (*Nephrops norvegicus*)	PP fibers (from ropes) 3-5 mm in length and 0.2 mm in diameter; five fibers per feeding; uncertain what proportion of ingested plastic is retained in the foregut; fed and starved controls were used.	Reduced body mass and reductions in blood protein and stored lipids.	[265]
Marine crab (*Carcinus maenas*)	PP fibers (from ropes) (1-5 mm in length chronic 4 week feeding studies, crabs that ingested food containing microfibers (0.3-1.0 % plastic by weight).	Effect on food consumption and energy balance (scope for growth).	[266]
Marine copepod (*Calanus finmarchicus*)	Mixture of cultured microalgae (control), $\sim$ 50 ml^{-1} of nylon granules (10-30 µm or nylon fibers (10 × 30 µm), similar in shape and size to the microalgal prey.	Nylon fibers: altered prey selectivity ($P <$ 0.01) and nonsignificant 40% decrease in algal ingestion rates. Nylon granules and fibers: premature moulting in juvenile copepods.	[267]
Freshwater amphipod (*Hyalella Azteca*)	PE particles and PP fibers; conducting 10-d and 42-d bioassays using individual 250 ml chambers.	Acute EC50: PE particle and PP fibers 4.64×10^4 and 71.43 particles/fibers per ml respectively. Observed chronic effects: decreased growth and reproduction.	[268]
Freshwater amphipod (*Gammarus fossarum*)	PA fibers; 50-200 µm; exposure levels: 100, 540, 2680, 13,380 fibers cm^{-2} base area of glass beakers.	Reduced assimilation efficiency, reduced weigh. Found in gut after 0.5 hr of exposure; egestion in 16 h after exposure stop; No effects with PS 1.6 mm beads.	[269]
Freshwater crustacean (*Dapnia magna*)	PET textile microfibers (length range: 62-1400 µm, width 31-528 µm, thickness 1-21.5 µm).	Ingestion of very long microfibers (even 1400 µm long); increased mortality, even after 24 h of exposure in MP free medium following 48 h of MP exposure.	[13]

Blarer et al. [269] found reduced assimilation efficiency and weight after ingestion of PA fibers (50-200 µm) in freshwater amphipod (*Gammarus fossarum*) while no effects were present after ingestion of 1.6 mm PS microbeads. Jemec and Drobne [13] exposed *Daphnia magna* to a size range of PET textile microfibers (62–1400 µm) and found that while the majority of ingested fibers were around 300 µm, some very large twisted microfibers even up to 1400 µm were present inside the gut. As noted above, several studies have shown that the size, shape, and surface physicochemical characteristics of MNPs are essential determinants of their fate and effects. However, this knowledge is mainly based on studies using spherical and other non-fibrous MNPs. Cole et al. [267], exposing marine copepods to nylon fibers vs nylon granules, found that shape and chemical profile of a MP can influence its bioavailability and toxicity. Exposure to nylon fibers caused significant shifts in prey selectivity resulting in a 40% decrease in algal ingestion rates, while nylon granules negatively affected lipid accumulation.

The limited use of plastic microfibers and nanofibers in experimental work has provided a very limited view on their fate and potential effects. In addition to this, there are several limitations related with the experimental design of published laboratory findings that hamper translation of the findings to the field situation in nature, which will be described in 9.7.6.

9.7.3 Chemical-mediated effects of MNPs

With the sorption of chemicals on MNPs, the transfer of harmful substances such as persistent organic pollutants (POPs), can lead to organic bioaccumulation of such contaminants within the upper part of the food chain [210]. Nevertheless, laboratory studies and model estimates indicate that the relative importance of MNPs as a carrier of hydrophobic organic chemicals is low compared to other media (for review, see Koelmans et al. [90, 270]). However, the bioaccumulation can vary depending on polymer nature, chemical type, and fish species traits [88]. Microfibers exhibit a high surface to volume ratio compared to non-fibrous microplastic shapes and thus may exhibit enhanced pollutant-sorption performance and bioavailability of toxic compounds. Thus, the combined effects of microplastics, in particular microfibers, and other pollutants, including persistent organic pollutants (POPs) and heavy metals, through aquatic food chains requires further attention. The real risk for aquatic and human consumers still needs to be elucidated [172, 181].

Mechanisms potentially contributing to chemical impacts of microplastic particles and microfibers on aquatic biota include leaching of plastic additives and transfer of sorbed compounds such as POPs and metals. The chemical effect on organisms mediated via the ingestion of MNPs depends on the amount and nature of both the polymeric materials and sorbed/desorpted hydrophobic pollutants. Uptake and negative effects of additives and sorbed contaminants attributed to leaching from plastics has been documented in several cases [224, 225, 232, 271–275]. Several of these studies showed that microplastics and sorbed chemicals had a greater effect than chemicals alone; other studies did not reveal such effects [276, 277]. This can be largely contributed to the different experimental test conditions used in these studies. The combined effect of microplastics and sorbed co-contaminants in aquatic organisms is still not well understood. Some studies found additive effects of microplastics and sorbed contaminants, others did not find such combined effects. For example, Rainieri et al. [275] exposed zebrafish to feed spiked with 2% microplastics (LD-PE 125–250 µm of diameter) or a sorbed mixture of PCBs, BFRs, PFCs and methylmercury and a combination of both. After 3 weeks of exposure, microplastics alone did reveal no relevant health effects on zebrafish, but the combined effect of microplastics and sorbed contaminants caused significant organ toxicity in a greater manner than the contaminants alone.

The above studies suggest that ingestion of microplastics can move additives and sorbed pollutants into the tissues of organisms at concentrations that are sufficient to cause health effects or ecophysical functions (e.g., [83, 224, 278]. However, in most of these cases, it will be difficult to separate physical effects of the polymeric material and chemical effects caused by desorption from the particles.

Several studies indicate that so-called virgin microplastics are toxic to the larval stage of aquatic organisms through the leaching of chemicals, with subsequent consequences for the evaluation of microplastic laboratory experiments.

Martínez-Gómez et al. [233] evaluated the effects of virgin, weathered, and leachate of PS 6 µm and HDPE fluff particles < 80 µm in the sea urchin, *Paracentrotus lividus*. During the 48-h exposure time, fertilization and larval development were significantly impaired. The authors concluded that sea urchin embryotoxicity was attributed to the chemical leachate of the exposed plastic particles. Major identified compounds likely responsible for the observed effects were aliphatic hydrocarbons (C14, C16, C18, C20, and C22) from HDPE and styrene monomers from PS particles [279]. Embryotoxicity and impairment of larval development was also reported after exposure of brown mussel to PP microplastic leachate [226]. This makes the point that previous studies using virgin microplastics may have difficulties of separating between physical and chemical effects, and that weathered microplastics should provide more realistic exposure scenarios. Cleaning the MP particles with organic solvent may be insufficient to stop continuous leakage of chemical co-contaminants or residual monomers, such as styrene, which is known to be toxic to aquatic organisms.

Despite this, there is some evidence for effects of MPs on organisms at environmentally relevant concentrations. Rochman et al. [83] demonstrated that the ingestion of plastic microdebris (PE particles and associated chemicals) by Japanese medaka (*Oryzias latipes*) at environmentally relevant concentrations in sea water is capable of altering endocrine system function. However, a critical review of published studies and empirical evidence by Koelmans et al. [270] concluded that the ingestion of contaminated microplastics was not likely to increase the overall exposure to and risk of hydrophobic organic chemicals in marine organisms, highlighting current uncertainty in impacts. In a study by our group, ingestion of PCB-spiked microplastics by Norway lobster (*Nephros novergicus*) showed limited PCB bioaccumulation for polyethylene and negligible PCB bioaccumulation for polystyrene. No PCB depuration from lobster tissue to microplastics was observed [93]. A similar conclusion was reached independently using a modelling approach. Transfer of sorbed organic contaminants from microplastics was modelled to marine biota using the model OMEGA (Optimal Modelling for Ecotoxicological Applications) with different conditions of temperature, pH and gut surfactants. There was a negligible input from plastic compared to intake from food and water under both relevant and worst-case scenarios [280]. However, given the discrepancies in research results and the many uncertainties in risk assessment of microplastics and sorbed chemicals, more research is needed in this area.

9.7.4 Microbial effects of MNPs

An increasing number of studies provide ample evidence that potential pathogenic bacteria (e.g., *Vibrio spp.*, *Aeromonas salmonicida*) bind particularly strong to plastic debris [91, 100, 101, 281]. In several studies, for example, the bacterial richness including *Vibrio spp.* was higher on MPs than in the surrounding seawater, [102, 282, 283]. Other studies were unable to confirm an enrichment of *Vibrio spp.* with plastic [282–284]. Gene sequencing analysis has shown that the microbial assemblages associated with marine MPs are very diverse and different between geographical zones, the greater species richness found at lower latitudes [30, 97, 285]. Research on riverine microplastics showed that bacteria assembling differently on riverine microplastics have lower taxa richness, diversity, and evenness on plastic than on non-plastic substrates [286]. Especially, plastic polluted surface waters which directly receive urban storm water or raw sewage effluents may pose direct plastic associated microbial risks, including the spread of potential pathogens and antimicrobial resistance (AMR). It has been shown that microplastic particles and fibers passing through sewage treatment plants may become enriched with pathogens [100] and subsequently may then be dispersed to the marine environment [99]. However, so far there is no evidence that microplastic-associated pathogens, AMR, and biotoxins caused health effects in aquatic organisms. Extensive multidisciplinary research is needed to identify disease risks, ecological impacts and potential implications for human health [99]. A key question concerns whether and how long microbial pathogens may persist on aquatic MPs and whether they pose a potential risk to aquatic ecosystems.

9.7.5 Potential ecological effects of MNPs

Although hypothesized, the ubiquitous, persistent, and anticipated increase of microplastic pollution could in the long-term reach concentrations that physically alter ecologically significant processes. Several studies have proposed potential ecological impacts of microplastics that could provide avenues for future research. For example, the potential of MP pollution to alter ecosystem microbial function [287], whether MP pollution could affect the functioning and structure of sedimentary environments and the structure of benthic assemblages [288, 289], and the potential impact of microplastics on marine pelagic primary producers [290]. The assessment of potential ecological impacts strongly relies on predictions from experimental studies and modelling studies [291] and more thorough research is needed before refined and more reliable assessments can be made.

9.7.6 Field evidence and ecological relevance of laboratory studies

Biological effects of microplastics in the field have not been demonstrated, mainly due to the fact that the environmental concentrations detected are below laboratory effect thresholds and these effects are non-specific and difficult to separate from the effect of associated chemicals [27, 292] and similar effects caused by other multiple natural and anthropogenic stressors acting on aquatic animals and ecosystems. The many uncertainties and knowledge gaps as highlighted throughout this chapter seriously hamper an adequate MNP risk assessment.

In the absence of field evidence for MNPs, one must rely on the available laboratory exposure studies, which for several reasons are difficult to extrapolate meaningfully to effects likely to occur in the field. Most of the studies looking at the effects of microplastics on model organisms are fraught with one or more of the following limitations: (1) conducted at unrealistically high MP concentrations, using only one type of particular polymeric particle, while aquatic organisms in nature are exposed to much lower concentrations and also much larger diversity of plastic particles; (2) lack information on dose-response curves; (3) used particles smaller than those reported from the field; (4) used virgin particles not taking the physical-chemical weathering, eco-corona/microbial biofilm interactions into their effect assessment; (3) lack of reference natural (non-plastic) particles as control treatment; (5) laboratory studies are typically short term (and high concentration), whilst in reality exposure is of a chronic nature and at low concentration (high dose acute exposure versus low dose chronic exposure). Clearly, these experimental limitations hinder translation of the observed effects to the actual field situation. Moreover, not unimportantly, to date, only a handful of experiments with fibrous MNPs, the environmentally most prevalent plastic particle shape, have been conducted. It is evident that the experimental results derived of lab-based MNP studies have only limited ecological relevance and are difficult to translate to the real nature [293].

Adverse effects in biota have rarely been shown experimentally at environmentally relevant concentrations of MNPs. Still, despite their limitations, the majority of reported laboratory studies on MNP health effects are at levels that are sub-organismal or affect individual organisms. That said, it is clear that most of the observed toxicity endpoints, such as survival, growth, and reproduction are of high ecological relevance and can potentially impact ecosystem function [246]. Provisional risk assessments show that current MP levels likely do not pose a widespread ecological risk, except in some coastal locations [28]. However, as said previously, a full characterisation of the risk of MNPs in the aquatic environment is not yet feasible, due to large data gaps. Laboratory and field-derived data to inform more comprehensive ecological risks assessments of MNPs are needed and expected to rapidly grow, as the levels of MNP are expected to increase in the future and the subject will continue to attract significant scientific attention.

9.8 Key conclusions

- Micro and nanoplastics are highly complex mixtures of polymer materials, chemical contaminants, and biological agents. They exhibit variable properties (particle composition, size, density, charge, toxicity, aggregation

state, age, associated epibiota), representing a diverse and complex class of environmental contaminants.
- Microplastics, often in the form of microfibers, are common in both marine and freshwater systems from around the world. Current field data generally document only the larger plastic particles and fibers, seriously limiting field assessments. PP, PE, polyester, PET, HDPE, LDPE, PVC, and cellophane are most common polymer types; rayon, polyacrylonitrile, nylon only occasionally reported.
- Synthetic fibers are the most prevalent type of aquatic microplastics, and their occurrence appears to be related to effluents, fishing activity, and atmospheric deposition.
- Heterogeneous sampling methodologies and units are often applied resulting in poor quality data, hampering comparison with other studies. Microfibers are particularly difficult to monitor as they are pervasive throughout environment and laboratory.
- MNP debris is an emerging multi-stressor in the aquatic environment due to the physical hazards of the polymeric material, and the chemical and microbial hazards associated with it.
- There is ample laboratory evidence that exposure to MNPs can impact aquatic species across many taxa at various levels of biological organization, albeit at high concentrations. But there is only limited evidence from nature with many gaps in our understanding. Smaller particles are more harmful than large particles and micro-fibers are more harmful than non-fibrous particles.
- The fate and effects of synthetic and natural microfibers are understudied and poorly understood.

9.9 Key knowledge gaps and research priorities

Our overview of the current state of knowledge on aquatic MNPs displays important research gaps which need to be filled in order to investigate the possible risks posed by MNPs. Major knowledge gaps are summed up below.
- For all aquatic compartments, there is a pressing need for improved characterization and quantification of MNPs and standardized methods for sampling and analysis. Good quality data on microfibers deserves special attention.
- A particular challenge is the development of analytical methods for small microplastics up to several micrometers and down to the nanoscale in aquatic matrices.
- There is a clear need for chronic experiments with environmentally relevant concentrations, particle sizes, shapes and compositions relative to natural particles. Special attention deserves microfiber fate and effects.
- Given the ubiquity of microplastics in our aquatic environments, there is a need to assess population and ecosystem level effects of increasing MNP pollution.
- Another challenge is to better understand how MNPs are cycled through aquatic ecosystems, where they go and how they degrade.
- Future research should elucidate the role of MNPs as a transport vector of pathogens, antibiotic resistance and biotoxins and the potential for dispersing diseases in aquatic systems and humans.

References

[1] C. Wilcox, E. Van Sebille, and B. D. Hardesty. “Threat of plastic pollution to seabirds is global, pervasive, and increasing”. In: *Proceedings of the national academy of sciences* 112.38 (2015), pages 11899–11904 (cited on page 99).

[2] Statista. *Global production of plastics since 1950*. 2017. URL: `https://www.statista.com/statistics/282732/global-production-of-plastics-since-1950/` (cited on page 99).

[3] J. Boucher and D. Friot. *Primary microplastics in the oceans: a global evaluation of sources*. IUCN Gland, Switzerland, 2017 (cited on pages 99–101).

[4] D. Eerkes-Medrano, R. C. Thompson, and D. C. Aldridge. “Microplastics in freshwater systems: a review of the emerging threats, identification of knowledge gaps and prioritisation of research needs”. In: *Water research* 75 (2015), pages 63–82 (cited on pages 99–101, 109).

[5] J. Li, H. Liu, and J. P. Chen. “Microplastics in freshwater systems: A review on occurrence, environmental effects, and methods for microplastics detection”. In: *Water research* 137 (2018), pages 362–374 (cited on pages 99, 109).
[6] H. A. Leslie, S. H. Brandsma, M. J. M. Van Velzen, and A. D. Vethaak. “Microplastics en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota”. In: *Environment international* 101 (2017), pages 133–142 (cited on pages 99, 101, 106).
[7] C. J. Foley, Z. S. Feiner, T. D. Malinich, and T. O. Höök. “A meta-analysis of the effects of exposure to microplastics on fish and aquatic invertebrates”. In: *Science of the total environment* 631 (2018), pages 550–559 (cited on page 99).
[8] M. Wagner, S. Lambert, and M. W. Lambert. *Freshwater microplastics*. Springer International Publishing Cham, Switzerland, 2018 (cited on page 99).
[9] W. Luo, L. Su, N. J. Craig, F. Du, C. Wu, and H. Shi. “Comparison of microplastic pollution in different water bodies from urban creeks to coastal waters”. In: *Environmental pollution* 246 (2019), pages 174–182 (cited on page 99).
[10] J. R. Jambeck, R. Geyer, C. Wilcox, T. R. Siegler, M. Perryman, A. Andrady, R. Narayan, and K. L. Law. “Plastic waste inputs from land into the ocean”. In: *Science* 347.6223 (2015), pages 768–771 (cited on page 99).
[11] P. Villarrubia-Gómez, S. E. Cornell, and J. Fabres. “Marine plastic pollution as a planetary boundary threat–The drifting piece in the sustainability puzzle”. In: *Marine policy* 96 (2018), pages 213–220 (cited on page 99).
[12] R. C. Thompson, Y. Olsen, R. P. Mitchell, A. Davis, S. J. Rowland, A. W. John, D. McGonigle, and A. E. Russell. “Lost at sea: where is all the plastic?” In: *Science* 304.5672 (2004), pages 838–838 (cited on page 99).
[13] A. Jemec and D. Drobne. “From nanoparticles to microplastics: retrospective and prospective views”. In: *Arhiv za higijenu rada i toksikologiju* 67 (2016), page 31 (cited on pages 99, 115, 116).
[14] P. Kershaw, A. Turra, F. Galgani, et al. “Guidelines for the Monitoring and Assessment of Plastic Litter in the Ocean-GESAMP Reports and Studies No. 99”. In: *GESAMP Reports and Studies* (2019) (cited on pages 99, 104, 105).
[15] N. B. Hartmann, T. Hüffer, R. C. Thompson, M. Hassellöv, A. Verschoor, A. E. Daugaard, S. Rist, T. Karlsson, N. Brennholt, M. Cole, M. P. Herrling, M. C. Hess, N. P. Ivleva, A. L. Lusher, and M. Wagner. “Are we speaking the same language? Recommendations for a definition and categorization framework for plastic debris”. In: *Environmental science & technology* 53.3 (2019), pages 1039–1047 (cited on page 99).
[16] K. Liu, T. Wu, X. Wang, Z. Song, C. Zong, N. Wei, and D. Li. “Consistent transport of terrestrial microplastics to the ocean through atmosphere”. In: *Environmental science & technology* 53.18 (2019), pages 10612–10619 (cited on pages 99–101).
[17] M. Cole. “A novel method for preparing microplastic fibers”. In: *Scientific reports* 6 (2016), page 34519 (cited on page 99).
[18] S. L. Wright, R. C. Thompson, and T. S. Galloway. “The physical impacts of microplastics on marine organisms: a review”. In: *Environmental pollution* 178 (2013), pages 483–492 (cited on pages 100, 104, 111–115).
[19] M. Cole, P. Lindeque, C. Halsband, and T. S. Galloway. “Microplastics as contaminants in the marine environment: a review”. In: *Marine pollution bulletin* 62.12 (2011), pages 2588–2597 (cited on page 100).
[20] J. A. I. do Sul and M. F. Costa. “The present and future of microplastic pollution in the marine environment”. In: *Environmental pollution* 185 (2014), pages 352–364 (cited on page 100).
[21] M. Wagner, C. Scherer, D. Alvarez-Muñoz, N. Brennholt, X. Bourrain, S. Buchinger, E. Fries, C. Grosbois, J. Klasmeier, T. Marti, et al. “Microplastics in freshwater ecosystems: what we know and what we need to know”. In: *Environmental sciences Europe* 26.1 (2014), pages 1–9 (cited on page 100).
[22] P. Kershaw. *Sources, fate and effects of microplastics in the marine environment: a global assessment*. Technical report. International Maritime Organization, 2015 (cited on page 100).
[23] P. J. Kershaw and C. M. Rochman. “Sources, fate and effects of microplastics in the marine environment: part 2 of a global assessment”. In: *Reports and studies-IMO/FAO/Unesco-IOC/WMO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) eng no. 93* (2015) (cited on pages 100, 102, 103, 106, 107, 109, 111).
[24] K. Duis and A. Coors. “Microplastics in the aquatic and terrestrial environment: sources (with a specific focus on personal care products), fate and effects”. In: *Environmental sciences Europe* 28.1 (2016), page 2 (cited on pages 100, 101, 106).
[25] H. Auta, C. Emenike, and S. Fauziah. “Distribution and importance of microplastics in the marine environment: a review of the sources, fate, effects, and potential solutions”. In: *Environment international* 102 (2017), pages 165–176 (cited on page 100).

[26] A. A. Horton, A. Walton, D. J. Spurgeon, E. Lahive, and C. Svendsen. "Microplastics in freshwater and terrestrial environments: evaluating the current understanding to identify the knowledge gaps and future research priorities". In: *Science of the total environment* 586 (2017), pages 127–141 (cited on pages 100, 101, 106).

[27] E. E. Burns and A. B. Boxall. "Microplastics in the aquatic environment: Evidence for or against adverse impacts and major knowledge gaps". In: *Environmental toxicology and chemistry* 37.11 (2018), pages 2776–2796 (cited on pages 100, 106, 118).

[28] B. Koelmans, S. Pahl, T. Backhaus, F. Bessa, G. van Calster, N. Contzen, R. Cronin, T. Galloway, A. Hart, L. Henderson, et al. *A scientific perspective on microplastics in nature and society*. SAPEA, 2019 (cited on pages 100, 109, 118).

[29] S. Franzellitti, L. Canesi, M. Auguste, R. H. Wathsala, and E. Fabbri. "Microplastic exposure and effects in aquatic organisms: A physiological perspective". In: *Environmental toxicology and pharmacology* (2019) (cited on pages 100, 114).

[30] J.-Q. Jiang. "Occurrence of microplastics and its pollution in the environment: A review". In: *Sustainable production and consumption* 13 (2018), pages 16–23 (cited on pages 100, 117).

[31] J. U. Skåre, J. Alexander, M. Haave, I. Jakubowicz, H. K. Knutsen, A. Lusher, M. Ogonowski, K. E. Rakkestad, I. Skaar, L. E. Sverdrup, et al. "Microplastics; occurrence, levels and implications for environment and human health related to food. Scientific opinion of the Scientific Steering Committee of the Norwegian Scientific Committee for Food and Environment". In: *VKM Report* (2019) (cited on pages 100, 105).

[32] S. Estahbanati and N. L. Fahrenfeld. "Influence of wastewater treatment plant discharges on microplastic concentrations in surface water". In: *Chemosphere* 162 (2016), pages 277–284 (cited on page 100).

[33] A. Sanchez-Vidal, R. C. Thompson, M. Canals, and W. P. de Haan. "The imprint of microfibres in southern European deep seas". In: *PloS one* 13.11 (2018) (cited on pages 100, 107).

[34] L. C. Lebreton, J. Van Der Zwet, J.-W. Damsteeg, B. Slat, A. Andrady, and J. Reisser. "River plastic emissions to the world's oceans". In: *Nature communications* 8 (2017), page 15611 (cited on page 100).

[35] G. Gatidou, O. S. Arvaniti, and A. S. Stasinakis. "Review on the occurrence and fate of microplastics in Sewage Treatment Plants". In: *Journal of hazardous materials* 367 (2019), pages 504–512 (cited on pages 100, 101).

[36] C. M. Rochman. "Microplastics research—from sink to source". In: *Science* 360.6384 (2018), pages 28–29 (cited on pages 100, 106).

[37] F. S. Cesa, A. Turra, and J. Baruque-Ramos. "Synthetic fibers as microplastics in the marine environment: a review from textile perspective with a focus on domestic washings". In: *Science of the total environment* 598 (2017), pages 1116–1129 (cited on page 100).

[38] P. J. Kole, A. J. Löhr, F. G. Van Belleghem, and A. M. Ragas. "Wear and tear of tyres: a stealthy source of microplastics in the environment". In: *International journal of environmental research and public health* 14.10 (2017), page 1265 (cited on page 100).

[39] F. Sommer, V. Dietze, A. Baum, J. Sauer, S. Gilge, C. Maschowski, and R. Gieré. "Tire abrasion as a major source of microplastics in the environment". In: *Aerosol and air quality research* 18.8 (2018), pages 2014–2028 (cited on page 100).

[40] R. Dris, J. Gasperi, C. Mirande, C. Mandin, M. Guerrouache, V. Langlois, and B. Tassin. "A first overview of textile fibers, including microplastics, in indoor and outdoor environments". In: *Environmental pollution* 221 (2017), pages 453–458 (cited on page 100).

[41] T. M. Karlsson, A. D. Vethaak, B. C. Almroth, F. Ariese, M. van Velzen, M. Hassellöv, and H. A. Leslie. "Screening for microplastics in sediment, water, marine invertebrates and fish: method development and microplastic accumulation". In: *Marine pollution bulletin* 122.1-2 (2017), pages 403–408 (cited on pages 100, 102, 105–109).

[42] K. Laitala, I. G. Klepp, and B. Henry. "Does use matter? Comparison of environmental impacts of clothing based on fiber type". In: *Sustainability* 10.7 (2018), page 2524 (cited on page 101).

[43] B. Henry, K. Laitala, and I. G. Klepp. "Microfibres from apparel and home textiles: prospects for including microplastics in environmental sustainability assessment". In: *Science of the total environment* 652 (2019), pages 483–494 (cited on pages 101, 102).

[44] I. E. Napper and R. C. Thompson. "Release of synthetic microplastic plastic fibres from domestic washing machines: Effects of fabric type and washing conditions". In: *Marine pollution bulletin* 112.1-2 (2016), pages 39–45 (cited on page 101).

[45] Eunomia Research & Consulting Ltd. Bristol. *Plastics in the Marine Environment.* 2016. URL: `http://www.eunomia.co.uk/reports-tools/plastics-in-the-marine-environment/` (cited on page 101).

[46] M. A. Browne, P. Crump, S. J. Niven, E. Teuten, A. Tonkin, T. Galloway, and R. Thompson. “Accumulation of microplastic on shorelines woldwide: sources and sinks”. In: *Environmental science & technology* 45.21 (2011), pages 9175–9179 (cited on page 101).

[47] J. C. Prata. “Microplastics in wastewater: State of the knowledge on sources, fate and solutions”. In: *Marine pollution bulletin* 129.1 (2018), pages 262–265 (cited on page 101).

[48] UNESCO. “The United Nations world water development report 2017: wastewater: the untapped resource; facts and figures.” In: (2017). URL: `https://unesdoc.unesco.org/ark:/48223/pf0000247153` (cited on page 101).

[49] A. Verschoor, L. De Poorter, R. Dröge, J. Kuenen, and E. de Valk. *Emission of microplastics and potential mitigation measures: Abrasive cleaning agents, paints and tyre wear*. National Institute for Public Health and the Environment: Bilthoven, The Netherlands. 2016 (cited on page 101).

[50] M. Bläsing and W. Amelung. “Plastics in soil: Analytical methods and possible sources”. In: *Science of the total environment* 612 (2018), pages 422–435 (cited on page 101).

[51] S. Ziajahromi, P. A. Neale, L. Rintoul, and F. D. Leusch. “Wastewater treatment plants as a pathway for microplastics: development of a new approach to sample wastewater-based microplastics”. In: *Water research* 112 (2017), pages 93–99 (cited on page 101).

[52] I. A. Kane and M. A. Clare. “Dispersion, accumulation and the ultimate fate of microplastics in deep-marine environments: A review and future directions”. In: *Frontiers in earth science* 7 (2019), page 80 (cited on page 101).

[53] L. Simon-Sánchez, M. Grelaud, J. Garcia-Orellana, and P. Ziveri. “River Deltas as hotspots of microplastic accumulation: The case study of the Ebro River (NW Mediterranean)”. In: *Science of the total environment* 687 (2019), pages 1186–1196 (cited on page 101).

[54] H. K. Imhof, N. P. Ivleva, J. Schmid, R. Niessner, and C. Laforsch. “Contamination of beach sediments of a subalpine lake with microplastic particles”. In: *Current biology* 23.19 (2013), R867–R868 (cited on page 101).

[55] L. C. Woodall, A. Sanchez-Vidal, M. Canals, G. L. Paterson, R. Coppock, V. Sleight, A. Calafat, A. D. Rogers, B. E. Narayanaswamy, and R. C. Thompson. “The deep sea is a major sink for microplastic debris”. In: *Royal society open science* 1.4 (2014), page 140317 (cited on pages 101, 107).

[56] K. Zhang, J. Su, X. Xiong, X. Wu, C. Wu, and J. Liu. “Microplastic pollution of lakeshore sediments from remote lakes in Tibet plateau, China”. In: *Environmental pollution* 219 (2016), pages 450–455 (cited on page 101).

[57] M. Bergmann, V. Wirzberger, T. Krumpen, C. Lorenz, S. Primpke, M. B. Tekman, and G. Gerdts. “High quantities of microplastic in Arctic deep-sea sediments from the HAUSGARTEN observatory”. In: *Environmental science & technology* 51.19 (2017), pages 11000–11010 (cited on pages 101, 107).

[58] I. Peeken, S. Primpke, B. Beyer, J. Gütermann, C. Katlein, T. Krumpen, M. Bergmann, L. Hehemann, and G. Gerdts. “Arctic sea ice is an important temporal sink and means of transport for microplastic”. In: *Nature communications* 9.1 (2018), pages 1–12 (cited on page 101).

[59] R. Ambrosini, R. S. Azzoni, F. Pittino, G. Diolaiuti, A. Franzetti, and M. Parolini. “First evidence of microplastic contamination in the supraglacial debris of an alpine glacier”. In: *Environmental pollution* 253 (2019), pages 297–301 (cited on page 101).

[60] M. Haave, C. Lorenz, S. Primpke, and G. Gerdts. “Different stories told by small and large microplastics in sediment-first report of microplastic concentrations in an urban recipient in Norway”. In: *Marine pollution bulletin* 141 (2019), pages 501–513 (cited on pages 101, 106).

[61] B. D. Hardesty, J. Harari, A. Isobe, L. Lebreton, N. Maximenko, J. Potemra, E. van Sebille, A. D. Vethaak, and C. Wilcox. “Using numerical model simulations to improve the understanding of micro-plastic distribution and pathways in the marine environment”. In: *Frontiers in marine science* 4 (2017), page 30 (cited on page 101).

[62] T. S. Galloway, M. Cole, and C. Lewis. “Interactions of microplastic debris throughout the marine ecosystem”. In: *Nature ecology & evolution* 1.5 (2017), pages 1–8 (cited on pages 101, 103).

[63] C. M. Rochman, C. Brookson, J. Bikker, N. Djuric, A. Earn, K. Bucci, S. Athey, A. Huntington, H. McIlwraith, K. Munno, et al. “Rethinking microplastics as a diverse contaminant suite”. In: *Environmental toxicology and chemistry* 38.4 (2019), pages 703–711 (cited on page 102).

[64] S. Mintenig, P. S. Bäuerlein, A. Koelmans, S. Dekker, and A. Van Wezel. “Closing the gap between small and smaller: towards a framework to analyse nano-and microplastics in aqueous environmental samples”. In: *Environmental Science: nano* 5.7 (2018), pages 1640–1649 (cited on page 102).

[65] R. Dris, H. Imhof, W. Sanchez, J. Gasperi, F. Galgani, B. Tassin, and C. Laforsch. “Beyond the ocean: contamination of freshwater ecosystems with (micro-) plastic particles”. In: *Environmental chemistry* 12.5 (2015), pages 539–550 (cited on pages 102, 106).

[66] A. L. Andrady. "The plastic in microplastics: A review". In: *Marine pollution bulletin* 119.1 (2017), pages 12–22 (cited on pages 102, 103).
[67] V. Hidalgo-Ruz, L. Gutow, R. C. Thompson, and M. Thiel. "Microplastics in the marine environment: a review of the methods used for identification and quantification". In: *Environmental science & technology* 46.6 (2012), pages 3060–3075 (cited on pages 102, 104, 106).
[68] C. Munari, V. Infantini, M. Scoponi, E. Rastelli, C. Corinaldesi, and M. Mistri. "Microplastics in the sediments of Terra Nova Bay (Ross Sea, Antarctica)". In: *Marine pollution bulletin* 122.1-2 (2017), pages 161–165 (cited on page 102).
[69] V. Ambrogi, C. Carfagna, P. Cerruti, and V. Marturano. "Additives in polymers". In: *Modification of Polymer Properties*. Elsevier, 2017, pages 87–108 (cited on page 102).
[70] G. Pritchard. *Plastics additives: an AZ reference*. Volume 1. Springer Science & Business Media, 2012 (cited on page 102).
[71] D. Lithner, Å. Larsson, and G. Dave. "Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition". In: *Science of the total environment* 409.18 (2011), pages 3309–3324 (cited on page 102).
[72] K. J. Groh, T. Backhaus, B. Carney-Almroth, B. Geueke, P. A. Inostroza, A. Lennquist, H. A. Leslie, M. Maffini, D. Slunge, L. Trasande, et al. "Overview of known plastic packaging-associated chemicals and their hazards". In: *Science of the total environment* 651 (2019), pages 3253–3268 (cited on page 102).
[73] I. Langelaan, I. Nuyoens, and M. Jansen. "Microplastics in de Nederlandse zoete wateren". In: *H20, januari* (2015) (cited on page 102).
[74] J.-H. Kwon, S. Chang, S. H. Hong, and W. J. Shim. "Microplastics as a vector of hydrophobic contaminants: Importance of hydrophobic additives". In: *Integrated environmental assessment and management* 13.3 (2017), pages 494–499 (cited on page 102).
[75] H. L. De Frond, E. van Sebille, J. M. Parnis, M. L. Diamond, N. Mallos, T. Kingsbury, and C. M. Rochman. "Estimating the mass of chemicals associated with ocean plastic pollution to inform mitigation efforts". In: *Integrated environmental assessment and management* 15.4 (2019), pages 596–606 (cited on page 102).
[76] E. L. Teuten, J. M. Saquing, D. R. Knappe, M. A. Barlaz, S. Jonsson, A. Björn, S. J. Rowland, R. C. Thompson, T. S. Galloway, R. Yamashita, et al. "Transport and release of chemicals from plastics to the environment and to wildlife". In: *Philosophical transactions of the royal society B: biological sciences* 364.1526 (2009), pages 2027–2045 (cited on page 102).
[77] A. Bakir, S. J. Rowland, and R. C. Thompson. "Competitive sorption of persistent organic pollutants onto microplastics in the marine environment". In: *Marine pollution bulletin* 64.12 (2012), pages 2782–2789 (cited on page 102).
[78] H. Lee, W. J. Shim, and J.-H. Kwon. "Sorption capacity of plastic debris for hydrophobic organic chemicals". In: *Science of the total environment* 470 (2014), pages 1545–1552 (cited on page 102).
[79] C. Wu, K. Zhang, X. Huang, and J. Liu. "Sorption of pharmaceuticals and personal care products to polyethylene debris". In: *Environmental Science and pollution research* 23.9 (2016), pages 8819–8826 (cited on page 102).
[80] L. M. Ziccardi, A. Edgington, K. Hentz, K. J. Kulacki, and S. K. Driscoll. "Microplastics as vectors for bioaccumulation of hydrophobic organic chemicals in the marine environment: A state-of-the-science review". In: *Environmental toxicology and chemistry* 35.7 (2016), pages 1667–1676 (cited on page 102).
[81] N. B. Hartmann, S. Rist, J. Bodin, L. H. Jensen, S. N. Schmidt, P. Mayer, A. Meibom, and A. Baun. "Microplastics as vectors for environmental contaminants: exploring sorption, desorption, and transfer to biota". In: *Integrated environmental assessment and management* 13.3 (2017), pages 488–493 (cited on page 102).
[82] H. Zhang, Q. Zhou, Z. Xie, Y. Zhou, C. Tu, C. Fu, W. Mi, R. Ebinghaus, P. Christie, and Y. Luo. "Occurrences of organophosphorus esters and phthalates in the microplastics from the coastal beaches in north China". In: *Science of the total environment* 616 (2018), pages 1505–1512 (cited on page 102).
[83] C. M. Rochman, T. Kurobe, I. Flores, and S. J. Teh. "Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment". In: *Science of the total environment* 493 (2014), pages 656–661 (cited on pages 102, 116, 117).
[84] A. Turner and L. A. Holmes. "Adsorption of trace metals by microplastic pellets in fresh water". In: *Environmental chemistry* 12.5 (2015), pages 600–610 (cited on page 102).
[85] F. Wang, C. S. Wong, D. Chen, X. Lu, F. Wang, and E. Y. Zeng. "Interaction of toxic chemicals with microplastics: a critical review". In: *Water research* 139 (2018), pages 208–219 (cited on page 102).
[86] M. Llorca, G. Schirinzi, M. Martínez, D. Barceló, and M. Farré. "Adsorption of perfluoroalkyl substances on microplastics under environmental conditions". In: *Environmental pollution* 235 (2018), pages 680–691 (cited on page 102).

[87] R. M. Razanajatovo, J. Ding, S. Zhang, H. Jiang, and H. Zou. “Sorption and desorption of selected pharmaceuticals by polyethylene microplastics”. In: *Marine pollution bulletin* 136 (2018), pages 516–523 (cited on page 102).
[88] A. A. Koelmans, E. Besseling, A. Wegner, and E. M. Foekema. “Plastic as a carrier of POPs to aquatic organisms: a model analysis”. In: *Environmental science & technology* 47.14 (2013), pages 7812–7820 (cited on pages 102, 103, 116).
[89] M. Davranche, C. Veclin, A.-C. Pierson-Wickmann, H. El Hadri, B. Grassl, L. Rowenczyk, A. Dia, A. Ter Halle, F. Blancho, S. Reynaud, et al. “Are nanoplastics able to bind significant amount of metals? The lead example”. In: *Environmental pollution* 249 (2019), pages 940–948 (cited on page 102).
[90] A. A. Koelmans. “Modeling the role of microplastics in bioaccumulation of organic chemicals to marine aquatic organisms. A critical review”. In: *Marine anthropogenic litter*. Springer, Cham, 2015, pages 309–324 (cited on pages 103, 116).
[91] E. R. Zettler, T. J. Mincer, and L. A. Amaral-Zettler. “Life in the “plastisphere”: microbial communities on plastic marine debris”. In: *Environmental science & technology* 47.13 (2013), pages 7137–7146 (cited on pages 103, 117).
[92] J. Reisser, J. Shaw, G. Hallegraeff, M. Proietti, D. K. Barnes, M. Thums, C. Wilcox, B. D. Hardesty, and C. Pattiaratchi. “Millimeter-sized marine plastics: a new pelagic habitat for microorganisms and invertebrates”. In: *PloS one* 9.6 (2014) (cited on page 103).
[93] C. De Tender, L. I. Devriese, A. Haegeman, S. Maes, J. Vangeyte, A. Cattrijsse, P. Dawyndt, and T. Ruttink. “Temporal dynamics of bacterial and fungal colonization on plastic debris in the North Sea”. In: *Environmental science & technology* 51.13 (2017), pages 7350–7360 (cited on pages 103, 117).
[94] C. D. Rummel, A. Jahnke, E. Gorokhova, D. Kühnel, and M. Schmitt-Jansen. “Impacts of biofilm formation on the fate and potential effects of microplastic in the aquatic environment”. In: *Environmental science & technology letters* 4.7 (2017), pages 258–267 (cited on page 103).
[95] C. A. De Tender, L. I. Devriese, A. Haegeman, S. Maes, T. Ruttink, and P. Dawyndt. “Bacterial community profiling of plastic litter in the Belgian part of the North Sea”. In: *Environmental science & technology* 49.16 (2015), pages 9629–9638 (cited on page 103).
[96] G. Oberdörster, A. Maynard, K. Donaldson, V. Castranova, J. Fitzpatrick, K. Ausman, J. Carter, B. Karn, W. Kreyling, D. Lai, et al. “Principles for characterizing the potential human health effects from exposure to nanomaterials: elements of a screening strategy”. In: *Particle and fibre toxicology* 2.1 (2005), page 8 (cited on page 103).
[97] D. Debroas, A. Mone, and A. Ter Halle. “Plastics in the North Atlantic garbage patch: a boat-microbe for hitchhikers and plastic degraders”. In: *Science of the total environment* 599 (2017), pages 1222–1232 (cited on pages 103, 117).
[98] A. K. Urbanek, W. Rymowicz, and A. M. Mirończuk. “Degradation of plastics and plastic-degrading bacteria in cold marine habitats”. In: *Applied microbiology and biotechnology* 102.18 (2018), pages 7669–7678 (cited on page 103).
[99] S. Oberbeckmann, M. G. Löder, and M. Labrenz. “Marine microplastic-associated biofilms–a review”. In: *Environmental chemistry* 12.5 (2015), pages 551–562 (cited on pages 103, 117).
[100] I. V. Kirstein, S. Kirmizi, A. Wichels, A. Garin-Fernandez, R. Erler, L. Martin, G. Gerdts, et al. “Dangerous hitchhikers? Evidence for potentially pathogenic Vibrio spp. on microplastic particles”. In: *Marine environmental research* 120 (2016), pages 1–8 (cited on pages 103, 117).
[101] M. K. Viršek, M. N. Lovšin, Š. Koren, A. Kržan, and M. Peterlin. “Microplastics as a vector for the transport of the bacterial fish pathogen species Aeromonas salmonicida”. In: *Marine pollution bulletin* 125.1-2 (2017), pages 301–309 (cited on pages 103, 117).
[102] L. Frère, L. Maignien, M. Chalopin, A. Huvet, E. Rinnert, H. Morrison, S. Kerninon, A.-L. Cassone, C. Lambert, J. Reveillaud, et al. “Microplastic bacterial communities in the Bay of Brest: Influence of polymer type and size”. In: *Environmental pollution* 242 (2018), pages 614–625 (cited on pages 103, 117).
[103] M. M. Silva, G. C. Maldonado, R. O. Castro, J. de Sá Felizardo, R. P. Cardoso, R. M. dos Anjos, and F. V. de Araújo. “Dispersal of potentially pathogenic bacteria by plastic debris in Guanabara Bay, RJ, Brazil”. In: *Marine pollution bulletin* 141 (2019), pages 561–568 (cited on page 103).
[104] J. P. Harrison, T. J. Hoellein, M. Sapp, A. S. Tagg, Y. Ju-Nam, and J. J. Ojeda. “Microplastic-associated biofilms: a comparison of freshwater and marine environments”. In: *Freshwater microplastics*. Springer, Cham, 2018, pages 181–201 (cited on page 103).
[105] M. Imran, K. R. Das, and M. M. Naik. “Co-selection of multi-antibiotic resistance in bacterial pathogens in metal and microplastic contaminated environments: An emerging health threat”. In: *Chemosphere* 215 (2019), pages 846–857 (cited on page 103).

[106] M. E. Larsson, O. F. Laczka, I. M. Suthers, P. A. Ajani, and M. A. Doblin. "Hitchhiking in the East Australian Current: rafting as a dispersal mechanism for harmful epibenthic dinoflagellates". In: *Marine ecology progress series* 596 (2018), pages 49–60 (cited on page 103).
[107] M. Kooi, E. H. v. Nes, M. Scheffer, and A. A. Koelmans. "Ups and downs in the ocean: effects of biofouling on vertical transport of microplastics". In: *Environmental science & technology* 51.14 (2017), pages 7963–7971 (cited on page 103).
[108] T. Cedervall, L.-A. Hansson, M. Lard, B. Frohm, and S. Linse. "Food chain transport of nanoparticles affects behaviour and fat metabolism in fish". In: *PloS one* 7.2 (2012) (cited on pages 103, 112).
[109] M. Lundqvist, J. Stigler, T. Cedervall, T. Berggård, M. B. Flanagan, I. Lynch, G. Elia, and K. Dawson. "The evolution of the protein corona around nanoparticles: a test study". In: *ACS nano* 5.9 (2011), pages 7503–7509 (cited on page 103).
[110] D. Docter, U. Distler, W. Storck, J. Kuharev, D. Wünsch, A. Hahlbrock, S. K. Knauer, S. Tenzer, and R. H. Stauber. "Quantitative profiling of the protein coronas that form around nanoparticles". In: *Nature protocols* 9.9 (2014), page 2030 (cited on page 103).
[111] F. Nasser and I. Lynch. "Secreted protein eco-corona mediates uptake and impacts of polystyrene nanoparticles on Daphnia magna". In: *Journal of proteomics* 137 (2016), pages 45–51 (cited on pages 103, 111).
[112] L. Canesi and I. Corsi. "Effects of nanomaterials on marine invertebrates". In: *Science of the total environment* 565 (2016), pages 933–940 (cited on page 103).
[113] A. L. Andrady. *Plastics and the Environment*. John Wiley & Sons, 2003 (cited on page 103).
[114] A. Ter Halle, L. Ladirat, X. Gendre, D. Goudounèche, C. Pusineri, C. Routaboul, C. Tenailleau, B. Duployer, and E. Perez. "Understanding the fragmentation pattern of marine plastic debris". In: *Environmental science & technology* 50.11 (2016), pages 5668–5675 (cited on page 103).
[115] B. Gewert, M. M. Plassmann, and M. MacLeod. "Pathways for degradation of plastic polymers floating in the marine environment". In: *Environmental science: processes & impacts* 17.9 (2015), pages 1513–1521 (cited on page 103).
[116] D. Hodgson, A. Bréchon, and R. Thompson. "Ingestion and fragmentation of plastic carrier bags by the amphipod Orchestia gammarellus: effects of plastic type and fouling load". In: *Marine pollution bulletin* 127 (2018), pages 154–159 (cited on page 103).
[117] D. K. Barnes, F. Galgani, R. C. Thompson, and M. Barlaz. "Accumulation and fragmentation of plastic debris in global environments". In: *Philosophical transactions of the royal society B: biological sciences* 364.1526 (2009), pages 1985–1998 (cited on page 103).
[118] A. L. Dawson, S. Kawaguchi, C. K. King, K. A. Townsend, R. King, W. M. Huston, and S. M. B. Nash. "Turning microplastics into nanoplastics through digestive fragmentation by Antarctic krill". In: *Nature communications* 9.1 (2018), pages 1–8 (cited on page 103).
[119] S. Summers, T. Henry, and T. Gutierrez. "Agglomeration of nano-and microplastic particles in seawater by autochthonous and de novo-produced sources of exopolymeric substances". In: *Marine pollution bulletin* 130 (2018), pages 258–267 (cited on page 104).
[120] E. Besseling, J. T. Quik, M. Sun, and A. A. Koelmans. "Fate of nano-and microplastic in freshwater systems: A modeling study". In: *Environmental pollution* 220 (2017), pages 540–548 (cited on page 104).
[121] I. Isachenko, L. Khatmullina, I. Chubarenko, and N. Stepanova. "Settling velocity of marine microplastic particles: laboratory tests". In: *EGU General Assembly Conference Abstracts*. Volume 18. 2016 (cited on page 104).
[122] A. Bagaev, A. Mizyuk, L. Khatmullina, I. Isachenko, and I. Chubarenko. "Anthropogenic fibres in the Baltic Sea water column: Field data, laboratory and numerical testing of their motion". In: *Science of the total environment* 599 (2017), pages 560–571 (cited on page 104).
[123] S. Ye and A. L. Andrady. "Fouling of floating plastic debris under Biscayne Bay exposure conditions". In: *Marine pollution bulletin* 22.12 (1991), pages 608–613 (cited on page 104).
[124] D. Lobelle and M. Cunliffe. "Early microbial biofilm formation on marine plastic debris". In: *Marine pollution bulletin* 62.1 (2011), pages 197–200 (cited on page 104).
[125] I. Jalón-Rojas, X. H. Wang, and E. Fredj. "A 3D numerical model to Track Marine Plastic Debris (TrackMPD): Sensitivity of microplastic trajectories and fates to particle dynamical properties and physical processes". In: *Marine pollution bulletin* 141 (2019), pages 256–272 (cited on page 104).
[126] F. M. Fazey and P. G. Ryan. "Biofouling on buoyant marine plastics: An experimental study into the effect of size on surface longevity". In: *Environmental pollution* 210 (2016), pages 354–360 (cited on page 104).
[127] M. Cole, P. K. Lindeque, E. Fileman, J. Clark, C. Lewis, C. Halsband, and T. S. Galloway. "Microplastics alter the properties and sinking rates of zooplankton faecal pellets". In: *Environmental science & technology* 50.6 (2016), pages 3239–3246 (cited on page 104).

[128] S. Zhao, J. E. Ward, M. Danley, and T. J. Mincer. “Field-based evidence for microplastic in marine aggregates and mussels: implications for trophic transfer”. In: *Environmental science & technology* 52.19 (2018), pages 11038–11048 (cited on pages 104, 110).
[129] M. G. Löder and G. Gerdts. “Methodology used for the detection and identification of microplastics—A critical appraisal”. In: *Marine anthropogenic litter*. Springer, 2015, pages 201–227 (cited on page 104).
[130] E. Fries, J. H. Dekiff, J. Willmeyer, M.-T. Nuelle, M. Ebert, and D. Remy. “Identification of polymer types and additives in marine microplastic particles using pyrolysis-GC/MS and scanning electron microscopy”. In: *Environmental science: processes & impacts* 15.10 (2013), pages 1949–1956 (cited on page 104).
[131] C. G. Avio, S. Gorbi, and F. Regoli. “Experimental development of a new protocol for extraction and characterization of microplastics in fish tissues: first observations in commercial species from Adriatic Sea”. In: *Marine environmental research* 111 (2015), pages 18–26 (cited on page 104).
[132] W. J. Shim, S. H. Hong, and S. E. Eo. “Identification methods in microplastic analysis: a review”. In: *Analytical methods* 9.9 (2017), pages 1384–1391 (cited on page 104).
[133] T. Rocha-Santos and A. C. Duarte. “A critical overview of the analytical approaches to the occurrence, the fate and the behavior of microplastics in the environment”. In: *TrAC Trends in analytical chemistry* 65 (2015), pages 47–53 (cited on page 104).
[134] E. Hermsen, S. M. Mintenig, E. Besseling, and A. A. Koelmans. “Quality criteria for the analysis of microplastic in biota samples: a critical review”. In: *Environmental science & technology* 52.18 (2018), pages 10230–10240 (cited on pages 104, 105).
[135] C. G. Avio, S. Gorbi, and F. Regoli. “Plastics and microplastics in the oceans: from emerging pollutants to emerged threat”. In: *Marine environmental research* 128 (2017), pages 2–11 (cited on page 104).
[136] L. Zada, H. A. Leslie, A. D. Vethaak, G. H. Tinnevelt, J. J. Jansen, J. F. de Boer, and F. Ariese. “Fast microplastics identification with stimulated Raman scattering microscopy”. In: *Journal of Raman spectroscopy* 49.7 (2018), pages 1136–1144 (cited on pages 104, 106).
[137] E. C. Ebere, V. E. Ngozi, et al. “Microplastics, an emerging concern: A review of analytical techniques for detecting and quantifying microplatics”. In: *Analytical methods in environmental chemistry journal* 2.2 (2019), pages 13–30 (cited on page 104).
[138] I. R. Comnea-Stancu, K. Wieland, G. Ramer, A. Schwaighofer, and B. Lendl. “On the identification of rayon/viscose as a major fraction of microplastics in the marine environment: discrimination between natural and manmade cellulosic fibers using Fourier transform infrared spectroscopy”. In: *Applied spectroscopy* 71.5 (2017), pages 939–950 (cited on page 105).
[139] Y. Yu, D. Zhou, Z. Li, and C. Zhu. “Advancement and challenges of microplastic pollution in the aquatic environment: a review”. In: *Water, air, & soil pollution* 229.5 (2018), page 140 (cited on page 105).
[140] C. Wesch, K. Bredimus, M. Paulus, and R. Klein. “Towards the suitable monitoring of ingestion of microplastics by marine biota: A review”. In: *Environmental pollution* 218 (2016), pages 1200–1208 (cited on pages 105, 107).
[141] F. Remy, F. Collard, B. Gilbert, P. Compère, G. Eppe, and G. Lepoint. “When microplastic is not plastic: the ingestion of artificial cellulose fibers by macrofauna living in seagrass macrophytodetritus”. In: *Environmental science & technology* 49.18 (2015), pages 11158–11166 (cited on page 105).
[142] T. Maes, R. Jessop, N. Wellner, K. Haupt, and A. G. Mayes. “A rapid-screening approach to detect and quantify microplastics based on fluorescent tagging with Nile Red”. In: *Scientific reports* 7 (2017), page 44501 (cited on pages 105, 106, 109).
[143] G. Erni-Cassola, M. I. Gibson, R. C. Thompson, and J. A. Christie-Oleza. “Lost, but found with Nile Red: a novel method for detecting and quantifying small microplastics (1 mm to 20 μm) in environmental samples”. In: *Environmental science & technology* 51.23 (2017), pages 13641–13648 (cited on page 105).
[144] L. C. Woodall, C. Gwinnett, M. Packer, R. C. Thompson, L. F. Robinson, and G. L. Paterson. “Using a forensic science approach to minimize environmental contamination and to identify microfibres in marine sediments”. In: *Marine pollution bulletin* 95.1 (2015), pages 40–46 (cited on page 105).
[145] A. A. Koelmans, E. Besseling, and W. J. Shim. “Nanoplastics in the aquatic environment. Critical review”. In: *Marine anthropogenic litter*. Springer, Cham, 2015, pages 325–340 (cited on page 105).
[146] K. Mattsson, S. Jocic, I. Doverbratt, and L.-A. Hansson. “Nanoplastics in the aquatic environment”. In: *Microplastic contamination in aquatic environments*. Elsevier, 2018, pages 379–399 (cited on page 105).
[147] J. Gigault, B. Pedrono, B. Maxit, and A. Ter Halle. “Marine plastic litter: the unanalyzed nano-fraction”. In: *Environmental science: nano* 3.2 (2016), pages 346–350 (cited on page 105).

[148] S. Rezania, J. Park, M. F. M. Din, S. M. Taib, A. Talaiekhozani, K. K. Yadav, and H. Kamyab. "Microplastics pollution in different aquatic environments and biota: A review of recent studies". In: *Marine pollution bulletin* 133 (2018), pages 191–208 (cited on pages 106, 109).

[149] S. Klein, E. Worch, and T. P. Knepper. "Occurrence and spatial distribution of microplastics in river shore sediments of the Rhine-Main area in Germany". In: *Environmental science & technology* 49.10 (2015), pages 6070–6076 (cited on page 106).

[150] J. Wang, J. Peng, Z. Tan, Y. Gao, Z. Zhan, Q. Chen, and L. Cai. "Microplastics in the surface sediments from the Beijiang River littoral zone: composition, abundance, surface textures and interaction with heavy metals". In: *Chemosphere* 171 (2017), pages 248–258 (cited on page 106).

[151] R. R. Hurley, A. L. Lusher, M. Olsen, and L. Nizzetto. "Validation of a method for extracting microplastics from complex, organic-rich, environmental matrices". In: *Environmental science & technology* 52.13 (2018), pages 7409–7417 (cited on page 106).

[152] C. Schmidt, T. Krauth, and S. Wagner. "Export of plastic debris by rivers into the sea". In: *Environmental science & technology* 51.21 (2017), pages 12246–12253 (cited on page 106).

[153] S. Eo, S. H. Hong, Y. K. Song, J. Lee, J. Lee, and W. J. Shim. "Abundance, composition, and distribution of microplastics larger than 20 μm in sand beaches of South Korea". In: *Environmental pollution* 238 (2018), pages 894–902 (cited on page 106).

[154] J. A. Brandon, A. Freibott, and L. M. Sala. "Patterns of suspended and salp-ingested microplastic debris in the North Pacific investigated with epifluorescence microscopy". In: *Limnology and oceanography letters* 5.1 (2020), pages 46–53 (cited on page 106).

[155] J. Gago, O. Carretero, A. Filgueiras, and L. Viñas. "Synthetic microfibers in the marine environment: A review on their occurrence in seawater and sediments". In: *Marine pollution bulletin* 127 (2018), pages 365–376 (cited on page 106).

[156] L. Van Cauwenberghe, A. Vanreusel, J. Mees, and C. R. Janssen. "Microplastic pollution in deep-sea sediments". In: *Environmental pollution* 182 (2013), pages 495–499 (cited on page 107).

[157] V. Fischer, N. O. Elsner, N. Brenke, E. Schwabe, and A. Brandt. "Plastic pollution of the Kuril–Kamchatka Trench area (NW pacific)". In: *Deep Sea Research Part II: Topical Studies in Oceanography* 111 (2015), pages 399–405 (cited on page 107).

[158] M. Taylor, C. Gwinnett, L. Robinson, and L. Woodall. "Plastic microfibre ingestion by deep-sea organisms". In: *Scientific reports* 6.1 (2016), pages 1–9 (cited on page 107).

[159] S. E. Nelms, T. S. Galloway, B. J. Godley, D. S. Jarvis, and P. K. Lindeque. "Investigating microplastic trophic transfer in marine top predators". In: *Environmental pollution* 238 (2018), pages 999–1007 (cited on page 107).

[160] E. M. Duncan, A. C. Broderick, W. J. Fuller, T. S. Galloway, M. H. Godfrey, M. Hamann, C. J. Limpus, P. K. Lindeque, A. G. Mayes, L. C. Omeyer, et al. "Microplastic ingestion ubiquitous in marine turtles". In: *Global change biology* 25.2 (2019), pages 744–752 (cited on page 107).

[161] T. Romeo, B. Pietro, C. Pedà, P. Consoli, F. Andaloro, and M. C. Fossi. "First evidence of presence of plastic debris in stomach of large pelagic fish in the Mediterranean Sea". In: *Marine pollution bulletin* 95.1 (2015), pages 358–361 (cited on page 107).

[162] F. Bessa, N. Ratcliffe, V. Otero, P. Sobral, J. C. Marques, C. M. Waluda, P. N. Trathan, and J. C. Xavier. "Microplastics in gentoo penguins from the Antarctic region". In: *Scientific reports* 9.1 (2019), pages 1–7 (cited on page 107).

[163] A. Lusher, N. Welden, P. Sobral, and M. Cole. "Sampling, isolating and identifying microplastics ingested by fish and invertebrates". In: *Analytical methods* 9.9 (2017), pages 1346–1360 (cited on pages 107, 110).

[164] J. D. O'Connor, A. M. Mahon, A. F. Ramsperger, B. Trotter, P. E. Redondo-Hasselerharm, A. A. Koelmans, H. T. Lally, and S. Murphy. "Microplastics in freshwater biota: a critical review of isolation, characterization, and assessment methods". In: *Global challenges* (2019), page 1800118 (cited on pages 107, 108).

[165] E. Berglund, V. Fogelberg, P. Nilsson, and J. Hollander. "Microplastics in a freshwater mussel (Anodonta anatina) in Northern Europe". In: *Science of the total environment* 697 (2019), page 134192 (cited on pages 107–109).

[166] L. Hu, M. Chernick, D. E. Hinton, and H. Shi. "Microplastics in small waterbodies and tadpoles from Yangtze River Delta, China". In: *Environmental science & technology* 52.15 (2018), pages 8885–8893 (cited on page 107).

[167] W. Sanchez, C. Bender, and J.-M. Porcher. "Wild gudgeons (Gobio gobio) from French rivers are contaminated by microplastics: preliminary study and first evidence". In: *Environmental research* 128 (2014), pages 98–100 (cited on page 107).

[168] C. B. Brookson, S. R. De Solla, K. J. Fernie, M. Cepeda, and C. M. Rochman. “Microplastics in the diet of nestling double-crested cormorants (Phalacrocorax auritus), an obligate piscivore in a freshwater ecosystem”. In: *Canadian Journal of fisheries and aquatic sciences* 76.11 (2019), pages 2156–2163 (cited on page 107).
[169] O. Setälä, M. Lehtiniemi, R. Coppock, and M. Cole. “Microplastics in marine food webs”. In: *Microplastic contamination in aquatic environments*. Elsevier, 2018, pages 339–363 (cited on pages 107, 110).
[170] F. M. Windsor, R. M. Tilley, C. R. Tyler, and S. J. Ormerod. “Microplastic ingestion by riverine macroinvertebrates”. In: *Science of the total environment* 646 (2019), pages 68–74 (cited on pages 107, 109).
[171] C. Wesch, A.-K. Barthel, U. Braun, R. Klein, and M. Paulus. “No microplastics in benthic eelpout (Zoarces viviparus): An urgent need for spectroscopic analyses in microplastic detection”. In: *Environmental research* 148 (2016), pages 36–38 (cited on page 107).
[172] L. G. A. Barboza, A. D. Vethaak, B. R. Lavorante, A.-K. Lundebye, and L. Guilhermino. “Marine microplastic debris: An emerging issue for food security, food safety and human health”. In: *Marine pollution bulletin* 133 (2018), pages 336–348 (cited on pages 107, 109, 116).
[173] A. Lusher, M. Mchugh, and R. Thompson. “Occurrence of microplastics in the gastrointestinal tract of pelagic and demersal fish from the English Channel”. In: *Marine pollution bulletin* 67.1-2 (2013), pages 94–99 (cited on page 108).
[174] L. I. Devriese, M. D. van der Meulen, T. Maes, K. Bekaert, I. Paul-Pont, L. Frère, J. Robbens, and A. D. Vethaak. “Microplastic contamination in brown shrimp (Crangon crangon, Linnaeus 1758) from coastal waters of the Southern North Sea and Channel area”. In: *Marine pollution bulletin* 98.1-2 (2015), pages 179–187 (cited on page 108).
[175] M. Zoeter Vanpoucke. “Impact of microplastic uptake: contamination in sprat and microplastic-mediated uptake of PAHs by European shore crab. Institute of Agricultural and Fisheries Research (ILVO) Ghent, EMBC+”. PhD thesis. MSc thesis, 2015 (cited on page 108).
[176] A. Naji, Z. Esmaili, S. A. Mason, and A. D. Vethaak. “The occurrence of microplastic contamination in littoral sediments of the Persian Gulf, Iran”. In: *Environmental Science and pollution research* 24.25 (2017), pages 20459–20468 (cited on page 109).
[177] A. Naji, M. Nuri, and A. D. Vethaak. “Microplastics contamination in molluscs from the northern part of the Persian Gulf”. In: *Environmental pollution* 235 (2018), pages 113–120 (cited on page 109).
[178] T. Karlsson. “Can microliter in sediment and biota be quantified? Method development and analysis of microliter in field collected biota and sediment”. PhD thesis. Master thesis, University of Gothenburg and VU University of Amsterdam-IVM, 2015 (cited on page 109).
[179] A. Vethaak and H. Leslie. “Plastic debris is a human health issue”. In: *Environmental science & technology* 50.13 (2016), pages 6825–6826 (cited on page 109).
[180] A. Petersen. “Statement on the presence of microplastics and nanoplastics in food, with particular focus on seafood”. In: (2016) (cited on page 109).
[181] M. Carbery, W. O’Connor, and T. Palanisami. “Trophic transfer of microplastics and mixed contaminants in the marine food web and implications for human health”. In: *Environment international* 115 (2018), pages 400–409 (cited on pages 109, 116).
[182] W. Wang, H. Gao, S. Jin, R. Li, and G. Na. “The ecotoxicological effects of microplastics on aquatic food web, from primary producer to human: A review”. In: *Ecotoxicology and environmental safety* 173 (2019), pages 110–117 (cited on page 109).
[183] L. Van Cauwenberghe and C. R. Janssen. “Microplastics in bivalves cultured for human consumption”. In: *Environmental pollution* 193 (2014), pages 65–70 (cited on page 109).
[184] WHO (World Health Organization). *Microplastics in drinking-water*. ISBN: 9789241516198. Geneva, Switzerland: World Health Organization, 2019, 101 p. (Cited on page 109).
[185] D. Yang, H. Shi, L. Li, J. Li, K. Jabeen, and P. Kolandhasamy. “Microplastic pollution in table salts from China”. In: *Environmental science & technology* 49.22 (2015), pages 13622–13627 (cited on page 109).
[186] M. Joon. “Trophic transfer of microplastics in zooplanktons towards its speculations on human health: A review”. In: *Journal of biomedical and therapeutic sciences* 6.1 (2019), pages 8–14 (cited on page 109).
[187] S. L. Wright and F. J. Kelly. “Plastic and human health: a micro issue?” In: *Environmental science & technology* 51.12 (2017), pages 6634–6647 (cited on page 109).
[188] A. I. Catarino, V. Macchia, W. G. Sanderson, R. C. Thompson, and T. B. Henry. “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”. In: *Environmental pollution* 237 (2018), pages 675–684 (cited on page 110).

[189] J. van den Hoff, C. Eriksson, H. Burton, and M. Schultz. "Size-Selective Feeding by Mesopelagic Fish Can Impact Ocean Surface Abundance of Small Plastic Particles". In: *Proceedings of the International Conference on Microplastic Pollution in the Mediterranean Sea*. Springer. 2018, pages 151–157 (cited on page 110).

[190] C. Gebhardt and S. Forster. "Size-selective feeding of Arenicola marina promotes long-term burial of microplastic particles in marine sediments". In: *Environmental pollution* 242 (2018), pages 1777–1786 (cited on page 110).

[191] H. S. H. Chan, C. Dingle, and C. Not. "Evidence for non-selective ingestion of microplastic in demersal fish". In: *Marine pollution bulletin* 149 (2019), page 110523 (cited on page 110).

[192] M. G. Ryan, L. Watkins, and M. T. Walter. "Hudson River juvenile Blueback herring avoid ingesting microplastics". In: *Marine pollution bulletin* 146 (2019), pages 935–939 (cited on page 110).

[193] N. Hussain, V. Jaitley, and A. T. Florence. "Recent advances in the understanding of uptake of". In: *Advanced drug delivery reviews* 50 (2001), pages 107–142 (cited on page 110).

[194] D. A. Norris and P. J. Sinko. "Effect of size, surface charge, and hydrophobicity on the translocation of polystyrene microspheres through gastrointestinal mucin". In: *Journal of applied polymer science* 63.11 (1997), pages 1481–1492. ISSN: 1097-4628 (cited on page 110).

[195] T. S. Galloway. "Micro-and nano-plastics and human health". In: *Marine anthropogenic litter*. Springer, Cham, 2015, pages 343–366 (cited on page 110).

[196] A. L. Lusher, C. O'Donnell, R. Officer, and I. O'Connor. "Microplastic interactions with North Atlantic mesopelagic fish". In: *ICES Journal of marine science* 73.4 (2016), pages 1214–1225 (cited on page 110).

[197] B. Fernández and M. Albentosa. "Insights into the uptake, elimination and accumulation of microplastics in mussel". In: *Environmental pollution* 249 (2019), pages 321–329 (cited on pages 110, 114).

[198] L. Canesi, T. Balbi, R. Fabbri, A. Salis, G. Damonte, M. Volland, and J. Blasco. "Biomolecular coronas in invertebrate species: Implications in the environmental impact of nanoparticles". In: *NanoImpact* 8 (2017), pages 89–98 (cited on pages 110, 111).

[199] L. Marques-Santos, G. Grassi, E. Bergami, C. Faleri, T. Balbi, A. Salis, G. Damonte, L. Canesi, and I. Corsi. "Cationic polystyrene nanoparticle and the sea urchin immune system: biocorona formation, cell toxicity, and multixenobiotic resistance phenotype". In: *Nanotoxicology* 12.8 (2018), pages 847–867 (cited on page 110).

[200] M. A. Browne, A. Dissanayake, T. S. Galloway, D. M. Lowe, and R. C. Thompson. "Ingested microscopic plastic translocates to the circulatory system of the mussel, Mytilus edulis (L.)" In: *Environmental science & technology* 42.13 (2008), pages 5026–5031 (cited on pages 110, 114).

[201] N. Von Moos, P. Burkhardt-Holm, and A. Köhler. "Uptake and effects of microplastics on cells and tissue of the blue mussel Mytilus edulis L. after an experimental exposure". In: *Environmental science & technology* 46.20 (2012), pages 11327–11335 (cited on pages 110, 112–114).

[202] M. Sendra, A. Saco, M. P. Yeste, A. Romero, B. Novoa, and A. Figueras. "Nanoplastics: From tissue accumulation to cell translocation into Mytilus galloprovincialis hemocytes. resilience of immune cells exposed to nanoplastics and nanoplastics plus Vibrio splendidus combination". In: *Journal of hazardous materials* (2019), page 121788 (cited on pages 110, 111).

[203] J. Ding, S. Zhang, R. M. Razanajatovo, H. Zou, and W. Zhu. "Accumulation, tissue distribution, and biochemical effects of polystyrene microplastics in the freshwater fish red tilapia (Oreochromis niloticus)". In: *Environmental pollution* 238 (2018), pages 1–9 (cited on page 110).

[204] K. Mattsson, E. V. Johnson, A. Malmendal, S. Linse, L.-A. Hansson, and T. Cedervall. "Brain damage and behavioural disorders in fish induced by plastic nanoparticles delivered through the food chain". In: *Scientific reports* 7.1 (2017), pages 1–7 (cited on pages 110, 112, 113).

[205] M. D. Prokić, T. B. Radovanović, J. P. Gavrić, and C. Faggio. "Ecotoxicological effects of microplastics: Examination of biomarkers, current state and future perspectives". In: *TrAC Trends in analytical chemistry* 111 (2019), pages 37–46 (cited on page 110).

[206] M. Al-Sid-Cheikh, S. J. Rowland, K. Stevenson, C. Rouleau, T. B. Henry, and R. C. Thompson. "Uptake, whole-body distribution, and depuration of nanoplastics by the scallop Pecten maximus at environmentally realistic concentrations". In: *Environmental science & technology* 52.24 (2018), pages 14480–14486 (cited on page 111).

[207] K. N. Fotopoulou and H. K. Karapanagioti. "Surface properties of beached plastic pellets". In: *Marine environmental research* 81 (2012), pages 70–77 (cited on page 111).

[208] C. Della Torre, E. Bergami, A. Salvati, C. Faleri, P. Cirino, K. Dawson, and I. Corsi. "Accumulation and embryotoxicity of polystyrene nanoparticles at early stage of development of sea urchin embryos Paracentrotus lividus". In: *Environmental science & technology* 48.20 (2014), pages 12302–12311 (cited on pages 111, 112).

[209] G. Grassi, C. Landi, C. Della Torre, E. Bergami, L. Bini, and I. Corsi. “Proteomic profile of the hard corona of charged polystyrene nanoparticles exposed to sea urchin Paracentrotus lividus coelomic fluid highlights potential drivers of toxicity”. In: *Environmental science: nano* 6.10 (2019), pages 2937–2947 (cited on page 111).
[210] E. Besseling, A. Wegner, E. M. Foekema, M. J. Van Den Heuvel-Greve, and A. A. Koelmans. “Effects of microplastic on fitness and PCB bioaccumulation by the lugworm Arenicola marina (L.)” In: *Environmental science & technology* 47.1 (2013), pages 593–600 (cited on pages 111–113, 116).
[211] J. Hämer, L. Gutow, A. Köhler, and R. Saborowski. “Fate of microplastics in the marine isopod Idotea emarginata”. In: *Environmental science & technology* 48.22 (2014), pages 13451–13458 (cited on page 111).
[212] K. L. Kaposi, B. Mos, B. P. Kelaher, and S. A. Dworjanyn. “Ingestion of microplastic has limited impact on a marine larva”. In: *Environmental science & technology* 48.3 (2014), pages 1638–1645 (cited on page 111).
[213] A. A. Koelmans, E. Besseling, and E. M. Foekema. “Leaching of plastic additives to marine organisms”. In: *Environmental pollution* 187 (2014), pages 49–54 (cited on page 111).
[214] C. M. Rochman, M. A. Browne, A. J. Underwood, J. A. Van Franeker, R. C. Thompson, and L. A. Amaral-Zettler. “The ecological impacts of marine debris: unraveling the demonstrated evidence from what is perceived”. In: *Ecology* 97.2 (2016), pages 302–312 (cited on page 111).
[215] C. Barría, I. Brandts, L. Tort, M. Oliveira, and M. Teles. “Effect of nanoplastics on fish health and performance: A review”. In: *Marine pollution bulletin* (2019), page 110791 (cited on page 111).
[216] L. Canesi, C. Ciacci, E. Bergami, M. Monopoli, K. Dawson, S. Papa, B. Canonico, and I. Corsi. “Evidence for immunomodulation and apoptotic processes induced by cationic polystyrene nanoparticles in the hemocytes of the marine bivalve Mytilus”. In: *Marine environmental research* 111 (2015), pages 34–40 (cited on pages 111, 114).
[217] X. Sun, B. Chen, Q. Li, N. Liu, B. Xia, L. Zhu, and K. Qu. “Toxicities of polystyrene nano-and microplastics toward marine bacterium Halomonas alkaliphila”. In: *Science of the total environment* 642 (2018), pages 1378–1385 (cited on page 112).
[218] C. Zhang, X. Chen, J. Wang, and L. Tan. “Toxic effects of microplastic on marine microalgae Skeletonema costatum: interactions between microplastic and algae”. In: *Environmental pollution* 220 (2017), pages 1282–1288 (cited on pages 112, 113).
[219] S. B. Sjollema, P. Redondo-Hasselerharm, H. A. Leslie, M. H. Kraak, and A. D. Vethaak. “Do plastic particles affect microalgal photosynthesis and growth?” In: *Aquatic toxicology* 170 (2016), pages 259–261 (cited on pages 112, 113).
[220] P. Bhattacharya, S. Lin, J. P. Turner, and P. C. Ke. “Physical adsorption of charged plastic nanoparticles affects algal photosynthesis”. In: *The journal of physical chemistry C* 114.39 (2010), pages 16556–16561 (cited on pages 112, 113).
[221] S. Messinetti, S. Mercurio, G. Scarì, A. Pennati, and R. Pennati. “Ingested microscopic plastics translocate from the gut cavity of juveniles of the ascidian Ciona intestinalis”. In: *The European zoological journal* 86.1 (2019), pages 189–195 (cited on page 112).
[222] K.-W. Lee, W. J. Shim, O. Y. Kwon, and J.-H. Kang. “Size-dependent effects of micro polystyrene particles in the marine copepod Tigriopus japonicus”. In: *Environmental science & technology* 47.19 (2013), pages 11278–11283 (cited on page 112).
[223] M. Cole, P. Lindeque, E. Fileman, C. Halsband, R. Goodhead, J. Moger, and T. S. Galloway. “Microplastic ingestion by zooplankton”. In: *Environmental science & technology* 47.12 (2013), pages 6646–6655 (cited on page 112).
[224] M. A. Browne, S. J. Niven, T. S. Galloway, S. J. Rowland, and R. C. Thompson. “Microplastic moves pollutants and additives to worms, reducing functions linked to health and biodiversity”. In: *Current biology* 23.23 (2013), pages 2388–2392 (cited on pages 112, 116).
[225] C. G. Avio, S. Gorbi, M. Milan, M. Benedetti, D. Fattorini, G. d’Errico, M. Pauletto, L. Bargelloni, and F. Regoli. “Pollutants bioavailability and toxicological risk from microplastics to marine mussels”. In: *Environmental pollution* 198 (2015), pages 211–222 (cited on pages 112, 114, 116).
[226] P. P. G. e Silva, C. R. Nobre, P. Resaffe, C. D. S. Pereira, and F. Gusmão. “Leachate from microplastics impairs larval development in brown mussels”. In: *Water research* 106 (2016), pages 364–370 (cited on pages 112, 117).
[227] S. E. Rist, K. Assidqi, N. P. Zamani, D. Appel, M. Perschke, M. Huhn, and M. Lenz. “Suspended micro-sized PVC particles impair the performance and decrease survival in the Asian green mussel Perna viridis”. In: *Marine pollution bulletin* 111.1-2 (2016), pages 213–220 (cited on page 112).
[228] C. Détrée and C. Gallardo-Escárate. “Single and repetitive microplastics exposures induce immune system modulation and homeostasis alteration in the edible mussel Mytilus galloprovincialis”. In: *Fish & shellfish immunology* 83 (2018), pages 52–60 (cited on page 112).

[229] K. Tallec, A. Huvet, C. Di Poi, C. González-Fernández, C. Lambert, B. Petton, N. Le Goïc, M. Berchel, P. Soudant, and I. Paul-Pont. "Nanoplastics impaired oyster free living stages, gametes and embryos". In: *Environmental pollution* 242 (2018), pages 1226–1235 (cited on page 112).

[230] T. Gardon, C. Reisser, C. Soyez, V. Quillien, and G. Le Moullac. "Microplastics affect energy balance and gametogenesis in the pearl oyster Pinctada margaritifera". In: *Environmental science & technology* 52.9 (2018), pages 5277–5286 (cited on page 112).

[231] R. Sussarellu, M. Suquet, Y. Thomas, C. Lambert, C. Fabioux, M. E. J. Pernet, N. Le Goïc, V. Quillien, C. Mingant, Y. Epelboin, et al. "Oyster reproduction is affected by exposure to polystyrene microplastics". In: *Proceedings of the national academy of sciences* 113.9 (2016), pages 2430–2435 (cited on pages 112, 114).

[232] C. Nobre, M. Santana, A. Maluf, F. Cortez, A. Cesar, C. Pereira, and A. Turra. "Assessment of microplastic toxicity to embryonic development of the sea urchin Lytechinus variegatus (Echinodermata: Echinoidea)". In: *Marine pollution bulletin* 92.1-2 (2015), pages 99–104 (cited on pages 112, 116).

[233] C. Martínez-Gómez, V. M. León, S. Calles, M. Gomáriz-Olcina, and A. D. Vethaak. "The adverse effects of virgin microplastics on the fertilization and larval development of sea urchins". In: *Marine environmental research* 130 (2017), pages 69–76 (cited on pages 112, 117).

[234] Y. Mao, H. Ai, Y. Chen, Z. Zhang, P. Zeng, L. Kang, W. Li, W. Gu, Q. He, and H. Li. "Phytoplankton response to polystyrene microplastics: perspective from an entire growth period". In: *Chemosphere* 208 (2018), pages 59–68 (cited on page 112).

[235] F. Murphy and B. Quinn. "The effects of microplastic on freshwater Hydra attenuata feeding, morphology & reproduction". In: *Environmental pollution* 234 (2018), pages 487–494 (cited on page 112).

[236] M. Cole, P. Lindeque, E. Fileman, C. Halsband, and T. S. Galloway. "The impact of polystyrene microplastics on feeding, function and fecundity in the marine copepod Calanus helgolandicus". In: *Environmental science & technology* 49.2 (2015), pages 1130–1137 (cited on page 112).

[237] L. Lei, S. Wu, S. Lu, M. Liu, Y. Song, Z. Fu, H. Shi, K. M. Raley-Susman, and D. He. "Microplastic particles cause intestinal damage and other adverse effects in zebrafish Danio rerio and nematode Caenorhabditis elegans". In: *Science of the total environment* 619 (2018), pages 1–8 (cited on pages 112, 113).

[238] K. Mattsson, K. Adolfsson, M. T. Ekvall, M. T. Borgström, S. Linse, L.-A. Hansson, T. Cedervall, and C. N. Prinz. "Translocation of 40 nm diameter nanowires through the intestinal epithelium of Daphnia magna." In: *Nanotoxicology* 10 (2016), pages 1160–1167 (cited on page 112).

[239] M. Ogonowski, C. Schür, Å. Jarsén, and E. Gorokhova. "The effects of natural and anthropogenic microparticles on individual fitness in Daphnia magna". In: *PloS one* 11.5 (2016) (cited on page 112).

[240] R. Cui, S. W. Kim, and Y.-J. An. "Polystyrene nanoplastics inhibit reproduction and induce abnormal embryonic development in the freshwater crustacean Daphnia galeata". In: *Scientific reports* 7.1 (2017), pages 1–10 (cited on page 112).

[241] O. M. Lönnstedt and P. Eklöv. "Environmentally relevant concentrations of microplastic particles influence larval fish ecology". In: *Science* 352.6290 (2016), pages 1213–1216 (cited on page 112).

[242] L. C. de Sá, L. G. Luís, and L. Guilhermino. "Effects of microplastics on juveniles of the common goby (Pomatoschistus microps): confusion with prey, reduction of the predatory performance and efficiency, and possible influence of developmental conditions". In: *Environmental pollution* 196 (2015), pages 359–362 (cited on page 112).

[243] Y. Lu, Y. Zhang, Y. Deng, W. Jiang, Y. Zhao, J. Geng, L. Ding, and H. Ren. "Uptake and accumulation of polystyrene microplastics in zebrafish (Danio rerio) and toxic effects in liver". In: *Environmental science & technology* 50.7 (2016), pages 4054–4060 (cited on page 113).

[244] C. Chen, A. D. Townsend, S. A. Sell, and R. S. Martin. "Microchip-based 3D-cell culture using polymer nanofibers generated by solution blow spinning". In: *Analytical methods* 9.22 (2017), pages 3274–3283 (cited on page 113).

[245] S. Bhargava, S. S. Chen Lee, L. S. Min Ying, M. L. Neo, S. Lay-Ming Teo, and S. Valiyaveettil. "Fate of nanoplastics in marine larvae: a case study using barnacles, Amphibalanus amphitrite". In: *ACS Sustainable chemistry & engineering* 6.5 (2018), pages 6932–6940 (cited on page 113).

[246] R. H. Waring, R. M. Harris, and S. C. Mitchell. "Plastic contamination of the food chain: A threat to human health?" In: *Maturitas* 115 (2018), pages 64–68 (cited on pages 113, 118).

[247] R. J. Vroom, A. A. Koelmans, E. Besseling, and C. Halsband. "Aging of microplastics promotes their ingestion by marine zooplankton". In: *Environmental pollution* 231 (2017), pages 987–996 (cited on page 113).

[248] J. S. Choi, Y.-J. Jung, N.-H. Hong, S. H. Hong, and J.-W. Park. "Toxicological effects of irregularly shaped and spherical microplastics in a marine teleost, the sheepshead minnow (Cyprinodon variegatus)". In: *Marine pollution bulletin* 129.1 (2018), pages 231–240 (cited on page 113).
[249] F. Lagarde, O. Olivier, M. Zanella, P. Daniel, S. Hiard, and A. Caruso. "Microplastic interactions with freshwater microalgae: hetero-aggregation and changes in plastic density appear strongly dependent on polymer type". In: *Environmental pollution* 215 (2016), pages 331–339 (cited on page 113).
[250] M. Moore. "Do nanoparticles present ecotoxicological risks for the health of the aquatic environment?" In: *Environment international* 32.8 (2006), pages 967–976 (cited on page 113).
[251] E. P. Espinosa and B. Allam. "Reverse genetics demonstrate the role of mucosal C-type lectins in food particle selection in the oyster Crassostrea virginica". In: *Journal of experimental biology* 221.6 (2018), jeb174094 (cited on page 113).
[252] A. Mathalon and P. Hill. "Microplastic fibers in the intertidal ecosystem surrounding Halifax Harbor, Nova Scotia". In: *Marine pollution bulletin* 81.1 (2014), pages 69–79 (cited on page 114).
[253] L. Van Cauwenberghe, M. Claessens, M. B. Vandegehuchte, and C. R. Janssen. "Microplastics are taken up by mussels (Mytilus edulis) and lugworms (Arenicola marina) living in natural habitats". In: *Environmental pollution* 199 (2015), pages 10–17 (cited on page 114).
[254] J. Li, X. Qu, L. Su, W. Zhang, D. Yang, P. Kolandhasamy, D. Li, and H. Shi. "Microplastics in mussels along the coastal waters of China". In: *Environmental pollution* 214 (2016), pages 177–184 (cited on page 114).
[255] J. Li, C. Green, A. Reynolds, H. Shi, and J. M. Rotchell. "Microplastics in mussels sampled from coastal waters and supermarkets in the United Kingdom". In: *Environmental pollution* 241 (2018), pages 35–44 (cited on page 114).
[256] D. S. Green. "Effects of microplastics on European flat oysters, Ostrea edulis and their associated benthic communities". In: *Environmental pollution* 216 (2016), pages 95–103 (cited on page 114).
[257] M. N. Woods, M. E. Stack, D. M. Fields, S. D. Shaw, and P. A. Matrai. "Microplastic fiber uptake, ingestion, and egestion rates in the blue mussel (Mytilus edulis)". In: *Marine pollution bulletin* 137 (2018), pages 638–645 (cited on page 114).
[258] X. Qu, L. Su, H. Li, M. Liang, and H. Shi. "Assessing the relationship between the abundance and properties of microplastics in water and in mussels". In: *Science of the total environment* 621 (2018), pages 679–686 (cited on page 114).
[259] P. Kolandhasamy, L. Su, J. Li, X. Qu, K. Jabeen, and H. Shi. "Adherence of microplastics to soft tissue of mussels: a novel way to uptake microplastics beyond ingestion". In: *Science of the total environment* 610 (2018), pages 635–640 (cited on page 114).
[260] Q. Li, C. Sun, Y. Wang, H. Cai, L. Li, J. Li, and H. Shi. "Fusion of microplastics into the mussel byssus". In: *Environmental pollution* 252 (2019), pages 420–426 (cited on page 114).
[261] I. Paul-Pont, C. Lacroix, C. G. Fernández, H. Hégaret, C. Lambert, N. Le Goïc, L. Frère, A.-L. Cassone, R. Sussarellu, C. Fabioux, et al. "Exposure of marine mussels Mytilus spp. to polystyrene microplastics: toxicity and influence on fluoranthene bioaccumulation". In: *Environmental pollution* 216 (2016), pages 724–737 (cited on page 114).
[262] I. Brandts, M. Teles, A. Gonçalves, A. Barreto, L. Franco-Martinez, A. Tvarijonaviciute, M. Martins, A. Soares, L. Tort, and M. Oliveira. "Effects of nanoplastics on Mytilus galloprovincialis after individual and combined exposure with carbamazepine". In: *Science of the total environment* 643 (2018), pages 775–784 (cited on page 114).
[263] L. Pittura, C. G. Avio, M. E. Giuliani, G. d'Errico, S. H. Keiter, B. Cormier, S. Gorbi, and F. Regoli. "Microplastics as vehicles of environmental PAHs to marine organisms: combined chemical and physical hazards to the Mediterranean mussels, Mytilus galloprovincialis". In: *Frontiers in marine science* 5 (2018), page 103 (cited on page 114).
[264] L. Canesi, C. Ciacci, R. Fabbri, T. Balbi, A. Salis, G. Damonte, K. Cortese, V. Caratto, M. P. Monopoli, K. Dawson, et al. "Interactions of cationic polystyrene nanoparticles with marine bivalve hemocytes in a physiological environment: role of soluble hemolymph proteins". In: *Environmental research* 150 (2016), pages 73–81 (cited on page 114).
[265] N. A. Welden and P. R. Cowie. "Long-term microplastic retention causes reduced body condition in the langoustine, Nephrops norvegicus". In: *Environmental pollution* 218 (2016), pages 895–900 (cited on page 115).
[266] A. J. Watts, M. A. Urbina, S. Corr, C. Lewis, and T. S. Galloway. "Ingestion of plastic microfibers by the crab Carcinus maenas and its effect on food consumption and energy balance". In: *Environmental science & technology* 49.24 (2015), pages 14597–14604 (cited on page 115).
[267] M. Cole, R. Coppock, P. K. Lindeque, D. Altin, S. Reed, D. W. Pond, L. Sørensen, T. S. Galloway, and A. M. Booth. "Effects of nylon microplastic on feeding, lipid accumulation, and moulting in a coldwater copepod". In: *Environmental science & technology* 53.12 (2019), pages 7075–7082 (cited on pages 115, 116).

[268] S. Y. Au, T. F. Bruce, W. C. Bridges, and S. J. Klaine. "Responses of Hyalella azteca to acute and chronic microplastic exposures". In: *Environmental toxicology and chemistry* 34.11 (2015), pages 2564–2572 (cited on page 115).

[269] P. Blarer and P. Burkhardt-Holm. "Microplastics affect assimilation efficiency in the freshwater amphipod Gammarus fossarum". In: *Environmental science and pollution research* 23.23 (2016), pages 23522–23532 (cited on pages 115, 116).

[270] A. A. Koelmans, A. Bakir, G. A. Burton, and C. R. Janssen. "Microplastic as a vector for chemicals in the aquatic environment: critical review and model-supported reinterpretation of empirical studies". In: *Environmental science & technology* 50.7 (2016), pages 3315–3326 (cited on pages 116, 117).

[271] K. Tanaka, H. Takada, R. Yamashita, K. Mizukawa, M.-a. Fukuwaka, and Y. Watanuki. "Accumulation of plastic-derived chemicals in tissues of seabirds ingesting marine plastics". In: *Marine pollution bulletin* 69.1-2 (2013), pages 219–222 (cited on page 116).

[272] A. Bakir, S. J. Rowland, and R. C. Thompson. "Enhanced desorption of persistent organic pollutants from microplastics under simulated physiological conditions". In: *Environmental pollution* 185 (2014), pages 16–23 (cited on page 116).

[273] E. M. Chua, J. Shimeta, D. Nugegoda, P. D. Morrison, and B. O. Clarke. "Assimilation of polybrominated diphenyl ethers from microplastics by the marine amphipod, Allorchestes compressa". In: *Environmental science & technology* 48.14 (2014), pages 8127–8134 (cited on page 116).

[274] S. O'Donovan, N. C. Mestre, S. Abel, T. G. Fonseca, C. C. Carteny, B. Cormier, S. H. Keiter, and M. J. Bebianno. "Ecotoxicological effects of chemical contaminants adsorbed to microplastics in the clam Scrobicularia plana". In: *Frontiers in marine science* 5 (2018), page 143 (cited on page 116).

[275] S. Rainieri, N. Conlledo, B. K. Larsen, K. Granby, and A. Barranco. "Combined effects of microplastics and chemical contaminants on the organ toxicity of zebrafish (Danio rerio)". In: *Environmental research* 162 (2018), pages 135–143 (cited on page 116).

[276] S. Garrido, M. Linares, J. A. Campillo, and M. Albentosa. "Effect of microplastics on the toxicity of chlorpyrifos to the microalgae Isochrysis galbana, clone t-ISO". In: *Ecotoxicology and environmental safety* 173 (2019), pages 103–109 (cited on page 116).

[277] J. R. Rivera-Hernández, B. Fernández, J. Santos-Echeandia, S. Garrido, M. Morante, P. Santos, and M. Albentosa. "Biodynamics of mercury in mussel tissues as a function of exposure pathway: natural vs microplastic routes". In: *Science of the total environment* 674 (2019), pages 412–423 (cited on page 116).

[278] C. Peda, L. Caccamo, M. C. Fossi, F. Gai, F. Andaloro, L. Genovese, A. Perdichizzi, T. Romeo, and G. Maricchiolo. "Intestinal alterations in European sea bass Dicentrarchus labrax (Linnaeus, 1758) exposed to microplastics: preliminary results". In: *Environmental pollution* 212 (2016), pages 251–256 (cited on page 116).

[279] M. Björnsdotter. *Leaching of Residual Monomers, Oligomers and Additives from Polyethylene, Polypropylene, Polyvinyl Chloride, High-density Polyethylene and Polystyrene Virgin Plastics*. 2015 (cited on page 117).

[280] A. Bakir, I. A. O'Connor, S. J. Rowland, A. J. Hendriks, and R. C. Thompson. "Relative importance of microplastics as a pathway for the transfer of hydrophobic organic chemicals to marine life". In: *Environmental pollution* 219 (2016), pages 56–65 (cited on page 117).

[281] M. Van der Meulen, L. De Vriese, J. Lee, T. Maes, J. Van Dalfsen, A. Huvet, P. Soudant, J. Robbens, and A. Vethaak. *Socio-economic impact of microplastics in the 2 Seas, Channel and France Manche Region. An initial risk assessment*. MICRO Interreg project Iva. 2014, p.10 (cited on page 117).

[282] J. A. Bryant, T. M. Clemente, D. A. Viviani, A. A. Fong, K. A. Thomas, P. Kemp, D. M. Karl, A. E. White, and E. F. DeLong. "Diversity and activity of communities inhabiting plastic debris in the North Pacific Gyre". In: *MSystems* 1.3 (2016), e00024–16 (cited on page 117).

[283] C. Dussud, C. Hudec, M. George, P. Fabre, P. Higgs, S. Bruzaud, A.-M. Delort, B. Eyheraguibel, A.-L. Meistertzheim, J. Jacquin, et al. "Colonization of non-biodegradable and biodegradable plastics by marine microorganisms". In: *Frontiers in microbiology* 9 (2018), page 1571 (cited on page 117).

[284] K. Kesy, S. Oberbeckmann, B. Kreikemeyer, and M. Labrenz. "Spatial environmental heterogeneity determines young biofilm assemblages on microplastics in Baltic Sea mesocosms". In: *Frontiers in microbiology* 10 (2019), page 1665 (cited on page 117).

[285] L. A. Amaral-Zettler, E. R. Zettler, B. Slikas, G. D. Boyd, D. W. Melvin, C. E. Morrall, G. Proskurowski, and T. J. Mincer. "The biogeography of the Plastisphere: implications for policy". In: *Frontiers in ecology and the environment* 13.10 (2015), pages 541–546 (cited on page 117).

[286] A. R. McCormick, T. J. Hoellein, M. G. London, J. Hittie, J. W. Scott, and J. J. Kelly. “Microplastic in surface waters of urban rivers: concentration, sources, and associated bacterial assemblages”. In: *Ecosphere* 7.11 (2016), e01556 (cited on page 117).

[287] M. Arias-Andres, U. Klümper, K. Rojas-Jimenez, and H.-P. Grossart. “Microplastic pollution increases gene exchange in aquatic ecosystems”. In: *Environmental pollution* 237 (2018), pages 253–261 (cited on page 118).

[288] D. S. Green, B. Boots, J. Sigwart, S. Jiang, and C. Rocha. “Effects of conventional and biodegradable microplastics on a marine ecosystem engineer (Arenicola marina) and sediment nutrient cycling”. In: *Environmental pollution* 208 (2016), pages 426–434 (cited on page 118).

[289] D. S. Green, B. Boots, N. E. O’Connor, and R. Thompson. “Microplastics affect the ecological functioning of an important biogenic habitat”. In: *Environmental science & technology* 51.1 (2017), pages 68–77 (cited on page 118).

[290] T. A. Troost, T. Desclaux, H. A. Leslie, M. D. van Der Meulen, and A. D. Vethaak. “Do microplastics affect marine ecosystem productivity?” In: *Marine pollution bulletin* 135 (2018), pages 17–29 (cited on page 118).

[291] T. S. Galloway and C. N. Lewis. “Marine microplastics spell big problems for future generations”. In: *Proceedings of the national academy of sciences* 113.9 (2016), pages 2331–2333 (cited on page 118).

[292] A. A. Koelmans, E. Besseling, E. Foekema, M. Kooi, S. Mintenig, B. C. Ossendorp, P. E. Redondo-Hasselerharm, A. Verschoor, A. P. Van Wezel, and M. Scheffer. “Risks of plastic debris: unravelling fact, opinion, perception, and belief”. In: *Environmental science & technology* 51.20 (2017), pages 11513–11519 (cited on page 118).

[293] N. N. Phuong, A. Zalouk-Vergnoux, L. Poirier, A. Kamari, A. Châtel, C. Mouneyrac, and F. Lagarde. “Is there any consistency between the microplastics found in the field and those used in laboratory experiments?” In: *Environmental pollution* 211 (2016), pages 111–123 (cited on page 118).