GÖTEBORGS UNIVERSITET

EFFECTS OF MICROPLASTICS ON ORGANISMS AND IMPACTS ON THE ENVIRONMENT:

BALANCING THE KNOWN AND UNKNOWN

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Funded by the Swedish Environmental Protection Agency

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Front page picture: Giedrė Ašmonaitė

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ABBREVIATIONS

AOP	Adverse outcome pathway
CLP	Classification, Labelling and Packaging
DOM	Dissolved Organic Material
ECHA	European Chemicals Agency
EDCs	Endocrine disrupting chemicals
EFSA	European Food Safety Authority
EPA	Environmental Protection Agency
ERA	Ecological Risk Assessment
FT-IR	Fourier Transform Infrared Spectroscopy
GC/MS	Gas chromatography / Mass spectrometry
GESAMP	Group of Experts on the Scientific Aspects of Marine Environmental Protection
GHS	Globally Harmonized System
HOCs	Hydrophobic organic chemicals
IAS	Intentionally added substances
ISO	International Organization for Standardization
JRC	Joint Research Council
MPs	Microplastics
MSFD	Marine Strategic Framework Directive
NIAS	Non-intentionally added substances
NNPs	Nanoplastics
OECD	Organisation for Economic Co-operation and Development
OSPAR	Convention for the Protection of the Marine Environment in the North-East Atlantic
PBTs	Persistent, bioaccumulative and toxic substances
PEC	Predicted environmental concentration
PNEC	Predicted no-effect concentration
POM	Particulate organic matter
POPs	Persistent organic pollutants
RA	Risk assessment
UN	United Nations
UNEP	United Nations Environmental Programme
UV	Ultraviolet
vPvB	Very persistent, very bioaccumulative

Polymers

EPS	Expanded polystyrene
PA	Polyamide
PE	Polyethylene
PLA	Polylactic acid
PP	Polypropylene
PS	Polystyrene
PUR	Polyurethane
PVC	Polyvinylchloride

Chemicals

BPA	Bisphenol A
DBP	Dibutyl phthalate
DEHP	Bis(2-ethylhexyl) phthalate
DEP	Diethyl phthalate
NP	Nonylphenol
OCPs	Organochlorine pesticides
PAHs	Polyaromatic hydrocarbons
PBDEs	Polybrominated diphenyl ethers
PCBs	Polychlorinated biphenyls

SUMMARY

The task

The Swedish Environmental Protection Agency has been tasked with identifying important sources of microplastics (MPs) in the environment and suggesting actions to mitigate their emissions into the environment. This report was compiled by Gothenburg University and serves as an update to a report summarizing knowledge of the exposure and responses of wildlife to MPs that was produced by researchers at Örebro University in 2016. For this current update, scientific articles and reports were reviewed in November-December 2018. The main difference in the state of knowledge now, compared to 2016, is that the research field now has a clearer understanding of the complexity of the problem, mechanisms underlying effects of MPs, and consensus that the risks associated with microplastics are lower than we previously feared, taking into account the content of MPs is in the environment today. We provide an overview of the current knowledge in the field, focusing on conclusions where a consensus has been reached concerning the sources, fate and effects of MPs. Knowledge gaps and ongoing discussions within relevant research fields regarding the effects of MPs and associated chemicals are addressed. The major focus of this report is aquatic ecosystems (marine and freshwater), although we also review the prevalence and potential impacts of MPs in terrestrial ecosystems. However, the latter ecosystems are much less studied than the former.

Background

Production of plastic is increasing exponentially, and plastic debris or microplastics are now found in all environments, from beaches and surface water in the oceans, deep seas and sediments, arctic ice, fresh water systems, soil and terrestrial niches, to indoor environments as well as food and drinking water. Microplastics have been identified in many different organisms from the smallest planktonic animals to top predators, including large fish, birds and mammals. A search for research articles that uses the terms "microplastic and effect" shows that the number of published studies now numbers in thousands, and new studies are published daily. Early publications focused on methods for finding and identifying microplastics in different matrices, but more recently, the number of effect studies that measure the consequences of exposure have increased. As the research field has grown, the need for a common vocabulary has also grown. The term "microplastic" refers to plastic particles that are less than 5 mm in size, and the term "micro-litter" is used to include other materials such as naturally occurring polymers, rubber and cellulose.

Analysis methods and occurrence of MPs in different environments

Development of methods for testing, extracting and identifying MPs in different matrices (water, sediment, soil, biota) has continued. Spectrometric methods (Fourier transform infrared (FTIR) and Raman) are most common and have been developed to measure smaller

and irregular particles. Attenuated total reflectance FTIR and pyrolysis-gas chromatography-mass spectrometry (GC / MS) can also provide information on other chemicals present in the particles (eg, additives and environmental toxins). These methods have contributed to an increased understanding of sources of MPs and distribution over time and space. The highest levels are found in urban areas with both diffuse and point emissions. The particles are found in surface water for a limited time, and the growth of microorganisms, chemical changes of the particles (aging of the polymers, leaching of additives), and incorporation into feces contribute to increased sedimentation rate. Bottom and sediment living organisms affect the distribution of MPs in sediment.

New research has also described the existence of MPs in terrestrial systems, with a focus on sludge and agriculture. However, we know much less about the processes that affect distribution, chemical changes and degradation of MPs in these systems.

MPs in the food chain

Research concludes that intake via food is the most likely route of exposure for MP to biota and we provide an overview of the potential mechanisms that facilitate the uptake of plastic debris in aquatic nutrition chains (studies in terrestrial systems are limited). More than 690 marine species eat plastic. Growth of microorganisms and incorporation of chemical substances that affect chemosensors contribute to intake of plastic, that is, the plastic smells or tastes like "food". The color of the plastic also seems to be of importance. Intake occurs at different levels in the food chain, in animals from different taxa, and is affected by the animals' usual food, behavior and intake mechanisms (eg, filters, plankton eaters, top predators). MPs can be ingested directly or via food, i.e., via consumption of organisms that have eaten plastic.

The fate of plastics in organisms after ingestion is summarized in the report, and current information suggests that MPs have the potential to bioaccumulate and biomagnify in food chains. However, this is believed to be limited, since MPs are mainly found in intestinal contents of studied organisms. There are few studies describing the uptake of MPs from the intestine to other tissues in the body. Limited retention times in the gut and limited uptake of MPs by the epithelial tissues are important factors limiting bioaccumulation. It is likely that bioaccumulation and biomagnification are greatest with smaller MPs and nanoparticles.

Effects of exposure to MPs

We summarize the effects of MPs on biota and investigate potential consequences and mechanisms that are linked to physical interactions of particles and / or to chemical exposures. Most studies are conducted using aquatic or marine organisms, and usually include one species, one polymer, one particle size and one particle form. Effects of MPs have been studied at a number of different levels of biological organization, ranging from molecular interactions (eg, gene regulation and protein changes), tissue and organ levels (eg, inflammation and histological changes), effects in individual organisms (e.g., changes in metabolism, behavior and reproduction), to effects at the ecosystem level.

Biological mechanisms of effects

Effect studies show that MPs can cause oxidative stress, a common consequence of various types of stressors, which can be induced directly through the formation of oxygen radicals on the particles, or indirectly through exposure to radical chemicals or degradation products. This can lead to the formation of lipid peroxides and DNA damage products. MPs can affect the immune system via various cell types (neutrophils, granulocytes), or via cytokine production. Chronic exposure to MPs via food can lead to intestinal damage, cell necrosis, and changes in metabolism and fat and energy reserves.

Several different types of effects have been observed at the individual level. Exposure to MPs is believed to cause neurotoxicity and have endocrine disrupting effects, and can cause changes in an animal's behavior, eg, swimming, feeding, and / or mating. Effects on the production of gametes have been documented. Any changes in behavior (foraging, avoidance of predators, etc.) and in reproduction are expected to be important at ecosystem levels. These effects are perhaps of the greatest interest today, as these can be linked to the ecosystem level effects of MPs, but they are not well described and assumptions are often based on extrapolations.

Ecosystem effects of MPs

Other mechanisms that may mediate the effects of MPs on the ecosystem level have been investigated. MPs are a new niche in the environment, a surface where microorganisms can grow, and this can lead to potential changes in microbiomes. New studies indicate that organic hydrocarbon from MPs can stimulate growth of certain types of bacteria, and current discourse amongst researchers focuses on gene transfer between different microbes, including antibiotic resistance genes. MPs can also influence species composition and the structure of bottom-living communities in marine systems,

Toxicity, particle and chemical effects

An emerging consensus in this field of research allows us to identify the importance of a number of factors modulating the toxicity of MPs. With regard to size, smaller micro- and nanoscale particles are more biologically relevant and have a greater impact. Shape can be important, and more irregularly shaped particles can induce more effects than can rounder particles. The effects of fibers are, however, still poorly understood. Concentration, a fundamental variable in toxicological analyses, is still a problematic factor in risk assessments since the majority of effect studies use higher concentrations than those found in natural environments (two to seven orders of magnitude higher). There are, however, some limited environments (e.g., hotspots of accumulation or point sources of MPs) where risks may be imminent.

We draw some attention to discussions of potential (eco)toxicity associated with plastic additives and discuss the role of plastics as vectors or carriers of environmental pollutants. Such toxicity has been a pivotal theme in scientific discussions regarding the potential risks of MPs. MPs can act as both a sink and a source of chemicals that raise (eco)toxicological

concerns. These chemicals can be categorized into two classes by origin: chemical ingredients that originate from plastic materials and chemical substances sorbed from the environment onto MPs. The absorption of environmental contaminants and release of chemicals to biota by MPs have raised various concerns in the public, governmental, and industrial sectors. Plastic-chemicals can be toxic and can affect detoxification mechanisms, induce oxidative stress, can be androgenic or estrogenic, and can lead to behavioral changes. We are less knowledgable about how toxicity of MPs is affects by weathering processes in the environment.

A number of studies have shown that MPs can act as vectors for environmental toxins, via experiments in which animals are exposed to contaminated particles (either through in situ exposure of MPs in the environment or via artificial "spiking" with known substances). Uptake of chemicals and resulting biological effects have been described. However, these studies use high levels of MPs and/or chemicals, and have often not addressed other sources of exposure to chemicals, e.g., via natural particles or foods. A review of the most recent developments in this topic leads us to the conclusion that MPs might be of negligible importance as vectors for environmental pollutants in wildlife, based on knowledge of current MP concentrations and comparisons with uptake via contaminated food webs or water. Nanoparticles, which have a larger surface: volume, may be of greater importance.

Risk assessment

We further investigated the latest advances in ecological risk assessment of MPs and we examined remaining limitations and future needs in this topic. MPs are an all-encompassing substance that inevitably requires robust risk assessments to mitigate effects caused by exposure to MPs in the environment. Current ecological risk assessment practices apply a model that compares predicted exposures (often predicted environmental concentrations of chemical substances) with knowledge of hazard or toxicity, or more precisely, a predicted concentration where no effect is expected. A level of MPs in the environment that exceeds a "safe" level indicates risk. Risk assessments with particles (unlike soluble chemical substances) are in their infancy, and there is a mismatch between measured levels in the environment and levels used in experiments, which leads to some assumptions and extrapolations. There are also some knowledge gaps in both environmental levels of different polymers, sizes and shapes, as well as their effects in different types of organisms at different types of exposures. Preliminary calculations indicate that MP levels are currently within safe limits in most environments. Important considerations to note here are linked to large gaps in knowledge regarding levels of nanoplastic in the environment, effects of chronic exposures, and the predicted increase in plastic production and consumption in the near future.

As human exposures are inevitably linked to the presence of MPs in the aquatic environment, we also briefly explore the potential risks for human food safety and quality. The primary focus is on MP exposure occurring via the consumption of sea products (fish and shellfish). Of course, as human exposure is not limited to the consumption of seafood, we briefly examine other exposure pathways, such as drinking water, air and other foods. We critically

assess the remaining knowledge gaps in this rapidly developing interdisciplinary field of research and reflect on needs for the future. Lack of knowledge here impedes conclusions.

Risk perception

Risk assessments are primarily dealt with in science and are used as a tool to control priorities and policy decisions, and are often seen as separate from risk perception. However, risk perception is important in communication and discussions about plastics in the environment and society, and in understanding risk at a societal level. Human behavior, economic forces, social functions and the important role of plastic in our lives underly the presence of plastic in the environment. Therefore, it is important to take social aspects and risk perception into account. They play an important role in managing problems that affect the individual's behavior as well as decisions within companies and authorities.

Conclusions

In conclusion, current research greatly increases our understanding of the mechanisms of effects associated with exposure to MPs, addressing both particle and chemical effects. Over the past two years, we have gained a more nuanced and detailed picture of mechanisms of effects, exposure pathways, environmental levels and risks. Assimilating this knowledge from many levels of biological organization allows us to conclude that current levels of MPs may not be highly problematic in the environment at present. Knowledge concerning human exposure levels to microplastics and their consequences as lacking, as is knowledge concerning occurrence of nanoplastic in numerous matrices, thereby hindering conclusions in these questions. The greatest risk is likely to occur where exposure levels are highest, eg in textile workers. However, predictions of exponential increases in the use of plastic materials, together with concomitant increases in MPs in virtually every environment, lead us to end with a word of caution. Current consensus in the research field states that major threats and risks do not exist today but will likely occur in the future. MP pollution should be avoided, and mitigation efforts need to be addressed at the source.

SAMMANFATTNING

Uppdraget

Naturvårdsverket har fått uppdraget att identifiera viktiga källor för mikroplast till miljön och föreslå åtgärder som hindrar utsläppen. Kunskap om effekter av mikroplaster är viktigt för prioritering av arbetet. Denna rapport sammanställdes av Göteborgs universitet och är en uppdatering till en rapport som sammanfattade kunskap om exponering och effekter av mikroplaster (MP:er) på vilda djur som skrevs av forskare vid Örebro universitet 2016. I denna uppdatering granskades vetenskapliga artiklar och rapporter under november och december 2018. Den huvudsakliga skillnaden i kunskapsläge nu, jämfört med 2016, är att forskarvärlden nu har tydligare förståelse för komplexiteten i problemet, mekanismer som driver effekter av MP:er, och konsensus kring att riskerna förknippade med mikroplaster är lägre än vi tidigare befarat, med hänsyn till halten av MP:er i miljön idag. Vi ger en översikt över aktuell kunskap inom forskningsfältet med fokus på slutsatser där konsensus har uppnåtts om MP-källor, öde och effekter. Kunskapsbrister och pågående diskussioner inom forskningsområdena om effekter av MP:er och tillhörande kemikalier beskrivs. Huvudfokus i denna rapport är övervägande kopplat till akvatiska system (marina och sötvatten), även om vi också granskar prevalens och potentiella effekter av MP:er i terrestra ekosystem. Denna nisch är dock mycket mindre studerad jämfört med akvatiska system.

Bakgrund

Produktionen av plast ökar exponentiellt, och plastskräp eller mikroplaster hittas nu i alla miljöer, från stränder och ytvatten i haven, djuphaven och sediment, arktiska isar, söttvattenssystem, jord och mark, till inomhusmiljöer samt mat och dricksvatten. Mikroplaster har identifierats i många olika organismer från de allra minsta planktondjur till toppredatorer, inklusive stora fiskar, fåglar och däggdjur. En sökning efter forskningsartiklar som använder termerna ’microplastic and effect’ numrerar nu i tusental, och nya studier publiceras dagligen. Tidigare publikationer fokuserade på metoder för att hitta och identifiera mikroplaster i olika matriser, men på senare tid har även antal effektstudier som mäter konsekvenser av exponering ökat. Allteftersom forskningsfältet har vuxit har även behovet av en gemensam vokabulär vuxit fram. Termen ’mikroplast’ syftar på plastpartiklar som är mindre än 5 mm i storlek, och termen ’mikroskräp’ används för att inkludera andra material som tex naturligt förekommande polymerer, gummi och cellulosa.

Analysmetoder och förekomst av MP:er i olika miljöer

Utveckling av metoder för att provta, extrahera och identifiera MP:er i olika matriser (vatten, sediment, jord, biota) fortsätter. Spektrometriska metoder (Fourier transform infrared (FTIR) och Raman) är vanligast och har utvecklats för att kunna mäta mindre och oregelbundna partiklar. Attenuated total reflectance FTIR samt pyrolysis-gas chromatography-mass spectrometry (GC/MS) kan även ge information om andra kemikalier som finns i partiklarna (tex., tillsatser och miljögifter). Dessa metoder har bidragit till en ökad förståelse av källor till MP:er och distribution över tid och rum. De högsta halterna återfinns i urbana områden med

både diffust- samt punktutsläpp. Partiklarna återfinns i ytvatten under en begränsad tid, och påväxt av mikroorganismer, kemiska förändringar av partiklarna (åldrande av polymererna, utlakning av tillsatser), samt inkorporering i fekalier bidrar till ökad sedimenteringshastighet. Botten- och sedimentlevande organismer påverkar hur MP:er distribueras i sediment.

Ny forskning har även beskrivit förekomsten av MP:er i terrestra system, med fokus på slam och jordbruk. Vi kan dock mycket mindre om processerna som påverkar distribution, kemiska förändringar och nedbrytning av MP:er i dessa system, vilket gör att vi saknar möjlighet att dra slutsatser kring mikroplaster i denna miljö.

MP:er i näringskedjan

Forskningen visar att intag via föda är den mest troliga exponeringsvägen för MP till biota och vi ger en översikt över de potentiella mekanismerna som underlättar upptag av plastskräp i akvatiska näringskedjor (studier i terrestra system är begränsade). Fler än 690 marina arter äter plast. Påväxt av mikroorganismer och inkorporering av kemiska ämnen som påverkar kemosensorer bidrar till intag av plast, alltså plasten luktar eller smakar som ’mat’. Plastens färg verkar också vara av betydelse. Intag sker på olika nivåer i näringskedjan, i djur från olika taxa, och påverkas av djurens vanliga föda, beteende och intagsmekanismer (tex., filtrerare, planktonätare, toppredatorer). MP:er kan tas in direkt eller via föda, dvs via bytesorganismer som har ätit plast.

Plastens öde i organismer efter intag sammanfattas, och aktuell information tyder på att MP:er har potential att bioackumulera och biomagnifiera i näringskedjor. Detta antas dock vara begränsat, eftersom MP:er huvudsakligen återfinns i tarminnehåll i studerade organismer. Det finns få studier som beskriver upptag av MP:er från tarmen till andra vävnader i kroppen. Begränsade retentionstider i tarmen och begränsad upptagning av MP:er genom epitelvävnaderna är viktiga faktorer som begränsar bioackumulering Troligtvis är bioackumulering och biomagnifiering störst med mindre MP:er och nanopartiklar

Effekter av MP:er

Vi sammanfattar också effekterna av MP:er på biota och undersöker potentiella konsekvenser och mekanismer som är kopplade till fysikaliska interaktioner av partiklar och/eller till kemisk exponering. De allra flesta studier är gjorda på akvatiska eller marina organismer, och inkluderar oftast en art, en polymer, en partikelstorlek och en partikelform. Effekter av MP:er har studerats på ett antal olika nivåer av biologisk organisation, allt från molekylära interaktioner (t.ex. genreglering och proteinförändringar), vävnads- och organnivåer (t.ex. inflammation och histologiska förändringar), effekter i enskilda organismer (t.ex. förändringar i ämnesomsättning, beteende och reproduktion), till effekter på ekosystemsnivå.

Biologiska mekanismer av effekter

Effektstudier visar att MP:er kan orsaka oxidativ stress, en vanlig konsekvens av olika typer av stressorer, som kan induceras direkt via formation av syre radikaler på partiklarna, eller indirekt via radikala kemiska ämnen eller nedbrytningsprodukter. Detta kan leda till formation

av lipidperoxider och DNA-skador. MP:er kan påverka immunsystemet via olika celltyper (neutrofiler, granulocyter), eller via produktioner av cytokiner. Kronisk exponering till MP:er via mat kan leda till skador i tarmen, cellnekros, och förändringar i metabolismen och fett- och energireserver.

På individnivån har flera olika typer av effekter observerats. Exponering för MP:er tros kunna orsaka neurotoxicitet och ha hormonstörande effekter, och påverkar ett djurs beteende, tex, simning, födosökande, och/eller parning. Effekter på produktion av gameter har dokumenterats. Eventuella förändringar i beteende (födosökande, undvikande av rovdjur osv.) samt i reproduktion förutses vara viktiga på ekosystemsnivåer. Dessa effekter är kanske av största intresse idag, eftersom dessa kan kopplas till ekosystemsnivåeffekter av MP:er, men de är inte väl beskrivna och antaganden är ofta baserade på extrapoleringar.

Ekosystemseffekter av MP:er

Andra mekanismer som kan påverka effekter på ekosystemsnivån har undersökts. MP:er utgör en ny nisch i miljön, en yta där mikroorganismer kan växa, och detta kan leda till potentiella förändringar i mikrobiomer. Organisk kolväte från MP:er kan stimulera tillväxt av vissa typer av bakterier, och en ny diskussion kretsar kring genöverföring mellan olika mikrober, inklusive antibiotiska resistensgener. MP:er kan också påverka artsammansättning och strukturen i bottenlevande samhällen i marina system, vilket i sin tur kan påverka dess funktion och näringscyklar.

Toxicitet, partikel- och kemiska effekter

En växande konsensus inom forskningsområdet identifierar vikten av ett antal faktorer som modulerar toxiciteten hos MP:er. När det gäller storlek är mindre mikro- och nanopartiklar mer biologiskt relevanta och har större effekter jämfört med stora mikro- och mesopartiklar. Formen kan vara viktig och ojämnare former kan orsaka fler och större effekter jämfört med släta, runda partiklar. Effekterna av fibrer är däremot fortfarande outforskade. Koncentration eller exponeringshalt, en grundläggande variabel i toxikologiska analyser, är fortfarande en problematisk faktor i riskbedömningar eftersom majoriteten av effektstudier använder högre koncentrationer än de som finns i de naturliga miljöerna (två till sju gånger högre). Det finns dock vissa områden i miljön (t ex hotspots av ackumulering eller punktkällor för MP:er) där risker kan vara möjliga.

Vi diskuterar potentiell (ekotoxikologisk) toxicitet i samband med plastkemikalier och diskuterar plastens roll som vektorer, eller bärare, för miljöföroreningar. Detta har varit ett avgörande tema i vetenskapliga diskussioner om potentiella risker med MP:er. MP:er kan fungera som både en källa för kemikalier som ökar ekotoxikologiska faror och kan dessutom binda upp kemikalier från den omgivande miljön. Dessa kemikalier kan klassificeras i två klasser efter ursprung: kemiska ingredienser som kommer ifrån plastmaterialen och kemiska substanser som sorberas från miljön till MP:er. Idéen att MP sorberar miljöföroreningar och släpper ut dessa kemikalier i biota väckte bekymmer som uppmärksammats i offentliga, statliga, och industriella sektorer. Plastkemikalier (både additiven samt sorberade ämnen) kan vara toxiska och kan påverka detoxifierieringsmekanismer, orsaka oxidativ stress, vara

östrogena eller androgena, och leda till beteendeförändringar. Vi kan däremot mindre om hur toxiciteten påverkas av åldrandet av MP:er i miljön.

En del studier har visat att MP:er kan fungera som vektorer för miljögifter, där djur exponeras för kontaminerade partiklar (antingen genom in situ exponering av MP:er i miljön eller via artificiell 'spiking' med kända ämnen). Upptag av kemikalier och biologiska effekter har visats. Men dessa studier använder höga halter av MP:er och/eller kemikalier, och har ofta inte tittat på andra exponeringskällor via naturliga partiklar eller födoämnen. En genomgång av den senaste utvecklingen i forskning kring detta ämne leder oss till slutsatsen att MP:er kan vara av försumbar betydelse som vektorer för miljöföroreningar i vilda djur, baserat på kunskap om nuvarande MP:er-koncentrationer och jämförelser med upptag via förorenade näringskedjor eller vatten. Nanopartiklar, som har en större yta:volym ration, kan eventuellt vara av större betydelse.

Riskbedömningar

Vi undersökte vidare de senaste framstegen inom ekologisk riskbedömning av MP:er och vi granskade kvarstående begränsningar och framtida behov i detta ämne. MP:er är ett allomfattande ämne som oundvikligen kräver robusta riskbedömningar för att mildra effekter som uppstår genom exponering för MP:er i miljö. Nuvarande praxis för ekologisk riskbedömning tillämpar en modell som jämför förutspådda exponeringar (ofta förutspådda miljökoncentrationer av kemiska ämnen) med kunskap om fara eller toxicitet, eller mer exakt en förutspådd koncentration där man inte förväntar någon effekt. En MP-halt i miljön som överstiger en halt som är 'säker' indikerar risk. Riskbedömningar med partiklar (till skillnad från lösbara kemiska ämnen) är i sin linda, och det finns en missanpassning mellan uppmätta halter i miljön och halter som används i experiment, vilket leder till en del antaganden och extrapoleringar. Det finns dessutom en del kunskapsluckor i både miljöhalter av olika polymerer, storlekar och former, samt effekter av dessa i olika typer av organismer vid olika typer av exponeringar. Preliminära beräkningar tyder på att MP-halter är inom säkra gränser i de flesta miljöer. Viktiga övervägningar här är kopplade till stora kunskapsluckor när det gäller halter av nanoplaster i miljön, effekter av kroniska exponeringar, samt den förutspådda ökningen av plastproduktion och konsumtion inom en snar framtid.

Eftersom mänsklig exponering oundvikligen är kopplad till förekomsten av MP:er i miljön, undersökte vi också de potentiella riskerna för livsmedelssäkerhet och kvalitet. Primär fokus ligger på exponering för MP:er som uppstår genom konsumtion av havsmat (fisk och skaldjur). Mänsklig exponering för MP:er är inte begränsad till konsumtion av fisk och skaldjur, så vi tittar också på andra exponeringsvägar, som dricksvatten, luft och andra livsmedel. Vi bedömde kritiskt de kunskapsluckor som kvarstod i detta snabbt utvecklande tvärvetenskapliga forskningsområde och poängterar framtida forskningsbehov. Brist på kunskap här hindrar slutsatser.

Riskperception

Riskbedömningar behandlas främst inom vetenskap och används som ett verktyg för att styra prioriteringar och policybeslut, och ses ofta som separata från riskperception. Dock är

riskperception och uppfattningar viktiga i kommunikation och diskussioner kring plaster i miljön och samhället. Mänskligt beteende, ekonomiska drivkrafter, samhällsfunktioner och plastens viktiga roll i våra liv ligger bakom förekomsten av plast i miljön. Därför är det viktigt att ta hänsyn till sociala aspekter och riskperception. De spelar en viktig roll i hantering av problem som påverkar individens beteende samt beslut inom företag och myndigheter.

Slutsats

Nuvarande forskning har bidragit till en ökad förståelse för mekanismerna av effekter som uppstår vid MP-exponering, och hanterar både partikel- och kemiska effekter. Under de senaste två åren har vi fått en mer nyanserad och detaljerad bild av mekanismer av effekter, exponeringsvägar, miljöhalter och risker. En assimilering av denna kunskap från många nivåer av biologisk organisation gör att vi kan dra slutsatsen att dagens nivåer av MP:er inte utgör en stor risk i miljön i nuläget. Kunskap om mänsklig exponering och konsekvenser är bristfälliga, likaså kunskaper om förekomsten av nanoplaster. Det vetenskapliga samhället har för närvarande inte kunnat nå enighet om hälsoriskerna för människor, men störst risk förekommer troligen där exponeringshalter är högst, tex hos textilarbetare. Eftersom produktion, användning, och utsläpp av plastmaterial förväntas ökar vilket kommer att leda till högre halter av MP:er i alla miljöer, och därefter, mer betydande risker är de nyaste slutsatserna inom forskningsfälten att stora hot och risker inte förekommer idag men kan sker i framtiden. Därför vill vi uppmana tillämpning av försiktighetsprincipen. MP-föroreningar bör undvikas, och utsläpp måste åtgärdas vid källan för att minska exponeringar och ökade risker.

BACKGROUND

The production and use of plastic materials have increased steadily since the start of industrial manufacturing in the 1950s, and current global production volumes exceed 322 million tons per year[1]. Plastic materials have become indispensable for numerous applications in our everyday lives and are important in many sectors including the transportation, electronics, construction, packaging, agriculture, food safety, health and hygiene, and textile industries. Global production is dominated by six types of polymers: polypropylene (PP), polyethylene (PE), polyvinyl chloride (PVC), polyethylene terephthalate (PET) polystyrene (PS), and polyurethane (PU)[2,3]. These six materials comprise approximately 80% of total plastics production (19.3% is PP, 17.5% is low-density PE (LDPE), 12.5% is high-density PE (HDPE) and medium-density PE, 10% is PVC, 7.5% is PUR, 7.4% is PET and 6.7% is PS, including expanded PS)[4]. Packaging materials constitute a major use of produced plastics; 40% of European plastic production is used in packaging, with PE (LPPE and HDPE), PP and PET being the most commonly used polymers[5].

Plastics have become ubiquitous not only in the technosphere but also in every realm of the natural environment. This issue has been acknowledged for decades; both the general public and the scientific community have been aware of the presence of plastics in the marine environment since the 1950s and 1960s. One can trace public awareness of plastic pollution in the world's oceans to newspaper articles, such as the NY Times citing research documenting plastic objects floating on ocean surfaces[6]. In fact, we have been dumping waste in the oceans for decades. Our knowledge of this practice and the problems resulting from it is reflected in laws preventing these acts dating back to the 1970s associated with the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (The London Convention) in 1972, followed shortly thereafter by the International Convention for the Prevention of Pollution from Ships (MARPOL) in 1973.

Today, we are confronted with messages and images of plastic pollution, from macroplastics such as fishing gear and plastic bags to microbeads and fibers, via newspapers, social media, television and documentaries. Popular media communication has followed the increase in scientific publications within the field. Our knowledge concerning the sources and fate of plastics is increasing as the number of technical reports and peer-reviewed scientific studies increases. There are many overviews of plastic pollution estimating the global amounts of plastic marine litter[7], the total amount of plastic ever produced[8], the number of plastic particles according to size (from macroplastics to MPs) in the oceans[9], and the extent to which land-based sources and rivers contribute to marine debris[10,11].

Similar to the number of scientific articles addressing the occurrence and fate of MPs, the number of studies addressing their effects has increased steadily in the past few years (**Figure 1**). Although investigations of the effects of MPs are still relatively young, our understanding of the effects increases as the research community teases apart the complex questions regarding MP exposure. These questions involve numerous factors, such as polymer types, particle size and shape, chemical additives, environmental contaminants, and exposure pathways. Until recently, much of this work focused on marine and aquatic systems,

but studies of plastics in terrestrial systems are increasing, and we are gaining information about the occurrence and effects of plastics in terrestrial organisms. This gain is evident in the recent increase in publications addressing the effects of MPs (**Figure 1**).

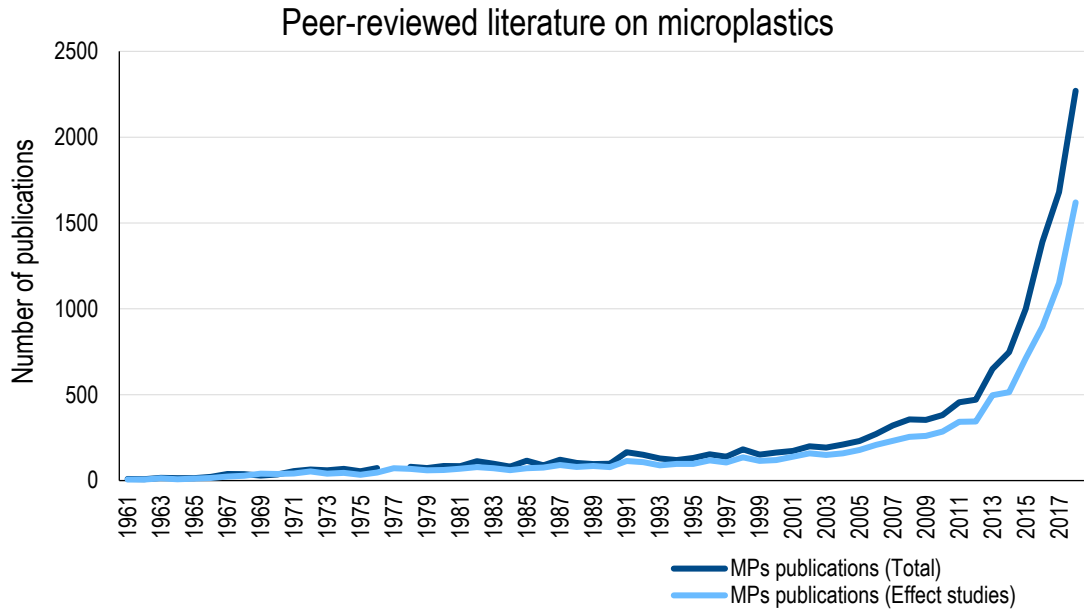

Figure 1 Cumulative number of publications in the research area of MPs. The graph demonstrates exponential growth in peer-reviewed literature since 1961 (Data source: Google Scholar 2018-12-17).

As this research field is expanding and knowledge is increasing exponentially, this report intends to summarize the current consensus concerning MPs. The main aim of the report is to address our current understanding of the interactions of MPs with biota as well as the impacts of MPs from both physical and chemical perspectives. Understanding the potential mechanisms, important variables, and effects in organisms (both aquatic and terrestrial) is essential to any risk assessment of potential ecological impacts, as is information on exposure levels and types. We therefore also address current methodologies used to document the occurrence, sources and fate of MPs in different environments, although an in-depth analysis of this work is beyond the scope of the current report.

DEFINITIONS AND TERMINOLOGY

Agreement upon and appropriate use of definitions of terms are essential to avoid miscommunication and misleading interpretations of scientific findings within and across MP research disciplines and sectors (industry, governmental authorities, and the public). We therefore define the most commonly used definitions and terminology in the research area of MPs here.

Marine litter, or **marine debris**, has received much international attention. The terms 'litter' and 'debris' are often used interchangeably, for example, in the United Nations Environment Programme (UNEP) report on marine plastic debris and MPs (2016)[12]. Both terms refer to solid waste, although the former indicates more intentionally and inappropriately discarded materials, while the latter implies broken and discarded materials and may refer to, e.g., rock fragments. The terms '**microlitter'** and **'microdebris'** are also used interchangeably in reports and scientific publications. Microlitter includes other materials that may be identified in the analyses of environmental samples, such as wood or naturally occurring polymers, metals, combustion particles, rubber, and natural fibers from cellulose, cotton, and wool.

The majority of litter identified in marine environments consists of plastics. In some cases, up to 90% of collected items are plastic[13]. The word '**plastic**' refers to the capability of being shaped or molded, and the noun 'plastic' is a colloquial term that denotes a group of synthetic, organic, polymeric, high-molecular-weight materials. According to the International Organization for Standardization (ISO), plastic is a material that contains a high-molecular-weight polymer as an essential ingredient, which at some stage in its processing into finished products can be shaped by flow[14]. 'Plastics' refers to thousands of different polymers that can be further categorized into thermoplastics (such as PE, PP and PS), which can be heated and remolded, or thermosets (e.g., PUR), which undergo a chemical reaction during solidification and cannot be melted and reformed. The feedstock for synthesis of plastic materials is usually fossil fuels, including crude oils and natural gases (in fact, production of plastics currently constitutes approximately 8% of global fossil fuel usage annually, including both materials and energy use, and the vast majority of plastics are produced from fossil fuels (up to 99%))[15]. Polymer diversity is extensive. Approximately 30000 different polymer types are registered with the European Union[2] alone.

Plastics consist of not only the polymers themselves but also thousands of other chemical substances, including both intentionally added substances (IASs; i.e., monomers, solvents, processing aids, etc.) and nonintentionally added substances (NIASs; i.e., metabolites, impurities, etc.)[16]. Plastic products commonly contain additives such as fillers, plasticizers, flame retardants, colorants, stabilizers, biocides, etc[16]. Many of these compounds used to make plastics are considered hazardous[3,17].

Two terms that are often used in reference to '**environmentally friendly plastics'** are **'bioplastics'** and '**biodegradable plastics'**, although these claims remain debated and the terms themselves can be misleading. Bioplastic refers to plastic polymers that are derived from biomass instead of fossil fuels, which reduces the environmental impact of their products[18]. Examples include polylactic acid (PLA) from corn and PE derived from sugarcane.

The term ‘bioplastic’, however, is viewed as problematic, as it implies that the products are safer or more environmentally friendly, while this may not be true. The biological and ecological effects of bioplastics, while not well studied, are not likely to differ from those of petroleum-based polymers. Feedstock may not be produced in a sustainable manner, affecting the use of land, water, pesticides and food sources[19]. In addition, while the physical and chemical structures of some biopolymers may allow for degradation in composts, biotic and abiotic factors in the natural environment can hinder their decomposition[20].

The term ‘**microplastics**’ was first coined by Thompson et al. in 2004 and referred to small plastic pieces that were 5 mm or smaller in size[21]. The Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) defined MPs as plastic particles <5 mm in diameter. This definition also included a lower limit to the size range, specifically, nanosized particles as small as 1 nm[22]. However, this lower size limit has been identified as somewhat inaccurate due to the colloidal nature of nanoplastics (NNPs) in aqueous media[23] and current technological limitations in the identification of smaller plastic particles (e.g., through the use of μ-Fourier transform infrared spectroscopy (FTIR))[24]. Thus, a lower size limit of 1 μm has been suggested[24,25].

In association with this classification, the term ‘**nanoplastics**’ was proposed, complementarily defining the smallest synthetic particulate fraction based on similarities to engineered nanoparticles[25]. The term ‘nanoparticle’ has been defined as a particle that is between 1 nm and 100 nm in one dimension and usually refers to inorganic particles instead of molecules. The definition of ‘**nanoplastics**’ is currently under discussion both within the scientific community[26] and among decision makers (e.g., ECs technical working group on marine litter)[27]. The term ‘nanoplastic’ has been used to indicate particles ranging in size from 1 nm to 100 nm or 1000 nm (1 μm)[25,26,28]. These authors argue that NNPs should refer to unintentionally produced particles (i.e., those from degradation) up to 1 μm in size and that NNPs differ from ‘nanomaterials’ that are intentionally produced to have specific chemical and physical properties. This definition of ‘nanoplastic’ is in contrast to that used in numerous exposure studies, where authors have used produced NNPs (usually PS) to address their effects and toxicity[29–31], but in line with that used in the work of others producing weathered plastic nanoparticles under laboratory conditions[32] or measuring NNPs in environmental samples[33].

The term ‘microplastic’ was further refined by Cole and colleagues [34], who described ‘**primary microplastics’** and ‘**secondary microplastics’**, dividing the particles according to origin, i.e., particles that were manufactured in a microscopic size range (including pellets or beads) versus the degradation products of large debris (produced via physical, biological or chemical fragmentation)[34]. Importantly, the usage of the term ‘**degradation’** in the case of plastics usually does not indicate chemical decomposition but fragmentation to smaller particles. MPs are usually considered nondissolvable and nondegradable, although there is some evidence that MPs do in fact degrade to some degree under certain environmental conditions (usually requiring ultraviolet (UV) light). This process results in changes in hydroxyl bonds, carbonyl groups, and carbon-oxygen bonds as well as chain scission products[35,36]. A separate category of MPs (not primary nor secondary) has also be identified and includes MPs that

are generated as a result of product wear during the product's lifetime. Contrary to secondary MPs, this group of MPs is not formed during environmental weathering but is generated via mulching, abrasion of car tires, and wear of textiles and synthetic paints[37].

Although MP has become a widespread and fairly all-inclusive working definition, in recent years, its versatility has been challenged as it does not account for the complexity of these materials. The present definition of MPs does not distinguish between different polymers or chemical compositions, nor does it include information describing the size or shape of the particles. Terms used to describe shapes or forms of microparticles are important, even though the full impact of these parameters on the fate or effects of particles is not fully understood. Terms describing shape may also provide indications of sources or origins of the particles. For example, '**pellets'** or '**nurdles**' are round, oval or cylindrical in shape and 2-5 mm in size and originate from preproduction raw plastic facilities[38], whereas '**film**' refers to a thin sheet of plastic with two smooth planes[39], likely originating from packaging or bags. The arbitrary thickness of a film differs from country to country and often from material to material but in some cases is 0.25 mm[14]. '**Fragments**' are particles derived from macroplastic that have been broken down into smaller particles through weathering or mechanical forces[40]. '**Fibers**' or '**microfibers**' are commonly used terms as these particles are ubiquitous in the environment, but there is no clear definition of their size; the diameter is generally accepted to be < 20 µm, although the length may be > 5 mm[41]. Images showing common shapes of MPs are shown in **Figure 2**. These MPs originate from synthetic textiles or fishing gear. In some cases, other synthetic particles of nonplastic chemical composition (e.g., synthetic rubber, antifouling particles, and semisynthetic materials) are considered to be MPs, although they should more correctly be referred to as microlitter.

The definition of MPs is progressively evolving, but we highlight the need for consensus, which will allow easier communication and comparisons between studies. The use of differing definitions of MPs is framework dependent and is becoming tailored to suit the needs of different stakeholders (in research, reporting, policy making, and the media). For example, operational definitions used in research tend to be theoretical, all-encompassing and evolving concepts that account for increasing knowledge and complexity, e.g., heteroaggregates and associations with dissolved organic material (DOM)[42]. However, the development of a legal definition will more readily enable regulatory action(s) and should be clear and factual. Additionally, as the media landscape rapidly changes, in a time when scientists, decision makers and the general public increasingly engage in communication via multiple means, definitions used in reporting and communication should aim not only to explain fact-based issues and possible mitigation but also to prevent the dissemination of misleading or biased information[43]. Despite rapid advancements in MP research, the communication of research findings can be context dependent and may be influenced by value judgment. To reduce subjectivity and avoid misinterpretation, harmonization of definitions and terminologies is essential. This harmonization is important not only for defining MPs themselves[44] but also for defining commonly used terms (e.g., uptake, bioaccumulation, biomagnification, and aging/weathering) or phrases, such as '**environmental realism'** or '**environmental**

relevance’, ‘**ecological harm’**, **’damage’** and **‘adverse biological effects**’. These terms are discussed in more detail below.

Achieving consensus in our definitions of MPs and associated terminology across sectors may be problematic, but it is essential for acknowledging potential differences in the meaning of these terms and the context where they are used.

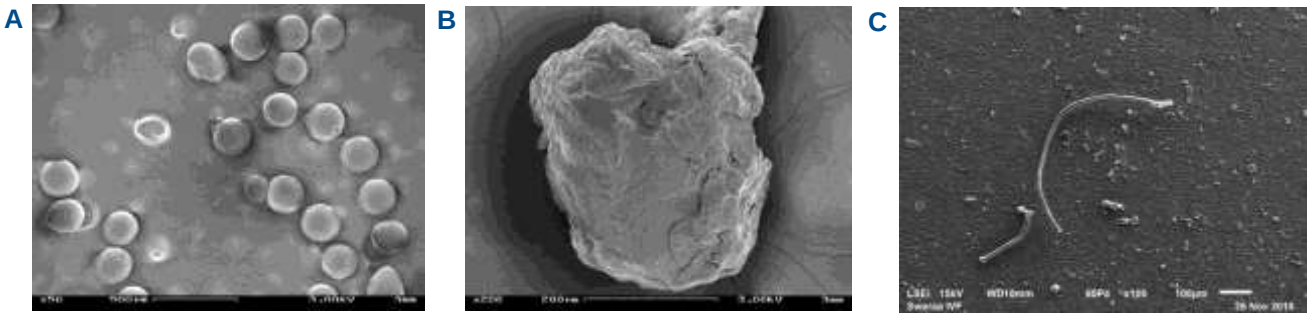

Figure 2. Scanning electron images of microplastics. A) Polyethylene beads with a smooth surface structure, B) polystyrene particle with rough surface, and C) polyethylene terephthalate fiber. Note differing size scales of images. (Photo credits: A and B: Giedrė Ašmonaitė and C: Anne-Charlotte Hanning).

A BRIEF LOOK AT ENVIRONMENTAL OCCURRENCE, SOURCES AND FATE

The presence of plastics in ocean surface waters has been evident in scientific publications for decades, dating back to the 1960s and 1970s. Preproduction pellets were identified in surface waters, birds' stomachs, and beaches around the world[45–47]. It was predicted at the time that plastic packaging materials would become more prevalent but also more problematic in waste collection, and littering and runoff from landfills were identified as pathways through which plastics entered the environment[48]. Since then, plastic litter has been identified in aquatic ecosystems globally, with coverage increasing every year. The majority (approximately 80%) of plastic in the oceans originates from land-based activities[49], while the rest stems from maritime activities, and plastics will accumulate more in major ocean gyres[50–52]. MPs exhibit similar patterns, although their occurrence can appear patchy. Understanding the sources and fate of MPs in different environments is essential for any attempt to define exposure levels, prerequisites for risk assessments, and efforts to mitigate problems.

An in-depth review of concentrations or amounts of MPs is beyond the scope of this report, and several reviews cover the issue of MP occurrence in aquatic systems[53,54]. Several sources have been studied, including release from production sites[55] and inputs from fishing and aquaculture[7,56], car tires, artificial turf[57,58], textiles[59–61] and wastewater treatment[62–64]. Many of the references in this chapter are reviews and contain summaries of numerous works.

SAMPLING AND ANALYTICAL METHODS

The methodologies utilized to sample and identify MPs in environmental matrices have been important focuses of research and development. Numerous methods for MP detection have become well established, allowing for the collection of more reliable and comparable estimates of MP abundances in various compartments of the (aquatic) environment. The alignment of sampling techniques, methodologies, and reporting is increasing, and benchmarking efforts are underway to standardize sampling, analyses and reporting to ease comparison between studies separated in time and space. Discussions about the most informative units for reporting MPs seem to indicate that the simultaneous use of several units may be useful (e.g., number of particles per area or volume and weight of particles per area or volume, with the addition of descriptors such as polymer type, size range, and shape).

Different methodologies of sample collection from surface waters have been compared, e.g., pump versus trawl[65]. While both methods are fairly common and relatively comparable, several limitations have been identified. Pump methods, which are suitable for point sources, may introduce contamination from hoses and do not sample the surface layer. In contrast, trawls can integrate over larger areas, including the sea surface, but are less accurate for determining the absolute concentration.

The mesh size used during sampling is of vital importance. A report documenting MPs in surface waters along the west coast of Sweden showed that levels of MPs were 100000-fold

higher when using a 10 µm filter than when using a 0.3 mm trawl[65]. For example, Norén and colleagues reported results from the Swedish west coast that far exceeded those previously reported from other marine waters[66] but indicated that this was likely due to different sampling methods and mesh sizes. More comparable to previous studies are the results of MPs from the west coast of Sweden (0.01-0.14 particles/m^3)[66] and the coast of Finland (0-0.74 particles/m^3)[62]. In the Baltic Sea, transects were found to average 0.27 particles/m^3 [53].

More recently, researchers have begun analyzing the occurrence of MPs in sediments and terrestrial environments, driving the development of methodologies customized for these more complex matrices[67]. Sampling techniques in such cases may involve using core samplers where depth and volume can be determined or conducting analyses per area of, for example, a beach or an agricultural plot (to a certain depth). The data reported in such cases may be the number of MPs per area or mass weight.

The treatment of samples prior to analysis has also been the subject of much research and is important as it allows the identification of MPs in complex matrices but may also affect the particles themselves. While samples from water may contain additional particulates, organic matter, and organisms, e.g., microalgae, matrices such as biota or sediment/earth samples are more challenging to analyze. Most methodologies include a digestion and/or separation step. Digestion often includes the addition of a strong acid (e.g., HCl), base (e.g., KOH or NaOH), or oxidizing agent (e.g., H_2O_2) or enzymatic degradation[68–71]. Density separation is applied at times using NaCl, NaI, or $ZnCl_2$ solutions to separate plastic particles from other particulate matter[72]. All of these techniques require some degree of cleaning as biofilms, organic and inorganic matter may produce artefacts that impede characterization of the polymers[73].

The detection and identification of polymers are important steps in MP research and are conducted via a variety of methodologies. Visual identification is the most accessible means of handling samples, although it is more suitable for larger particles. The process is time consuming, and studies have revealed a high degree of uncertainty in these results[50,68,74,75]. Other techniques applied in the identification of MPs are utilized, although these require costly instrumentation and expertise. FTIR and µ-FTIR provide information on the chemical bonds in particles, allowing the identification of specific polymer types and, in some cases, the detection of degradation in the polymers. The application of attenuated total reflectance (ATR-FTIR) can be useful for particles with irregular surfaces. Raman spectroscopy has functions and limitations similar to those of FTIR but is also sensitive to additives and pigments. In contrast, pyrolysis-gas chromatography-mass spectrometry (GC/MS) is a destructive process that analyses thermally decomposed gases of particles and can simultaneously detect additive chemicals in certain polymers. All of these techniques require some degree of cleaning as biofilms and organic and inorganic matter may produce artifacts that impede characterization of the polymers[73].

Many sampling and analytical techniques have increased in accuracy, allowing the detection of smaller particles. The detection of particles with sizes <10 µm is still technologically difficult and costly, but this information is important. There are indications that the number of particles

greatly increases as the size of the analyzed particles decreases, and biological evidence, as discussed below, shows that particles in these lower size ranges are more hazardous to organisms.

SPATIOTEMPORAL DISTRIBUTION IN AQUATIC SYSTEMS

While the identification of point sources and pathways through which MPs enter the environment has progressed, diffuse, nonpoint sources are more challenging to address. This work is necessary as our understanding of the fate of MPs and how they spread in the environment provides us with a better understanding of the breadth of the problem as well as guidance for risk assessments and mitigation efforts. There is a discrepancy between production volumes of plastic materials, estimated entry into the marine environment, and actual sampling information[7,76], indicating gaps in knowledge about the spatiotemporal distribution of MPs in aquatic systems.

A number of **point sources** and pathways for release of MPs have been identified. For example, MPs are found in sewage effluent, usually at a concentration of 1000 particles/L, but this is dependent on the treatment processes in use. Studies show that waste treatment plants (WTP) removal efficiency for MPs can reach up to 80-99%, but particles < 10 µm are not removed. Fibers are more difficult to remove (just 20 %), due in part to their small sizes (in diameter)[77]. Additionally, particles of < 10 µm are not removed. Fibers are more difficult to remove (with a success rate of only 20%), due in part to their small sizes (in diameter)[77]. Following removal during treatment processes, a large portion of MPs entering into wastewater treatment plants will end up in sludge [62,69,78]. Biosolids can consequently become a source of MPs for terrestrial environments due to landfill.

Urban areas are important sources, and pollution gradients indicating a decreasing abundance of MPs from urban territories have been described[50,64]. Greater MP abundances have been found in areas near industrial centers[55,79]. Coastal regions generally have higher pollutant concentrations[80]. An investigation of the spread of plastic pellets from a production site showed that the MPs moved with currents and winds, decreasing in concentration with distance from the industrial site, but that the pellets were predicted to beach as prevailing winds drove them landward[55]. In fact, recent studies indicate that MPs remain in surface water for a shorter period of time than previously predicted due to water action and changes in density due to biofouling, weathering, and changes in the molecular structure of the polymers (discussed below).

The **vertical distribution** of MPs in the water column is subject to turbulent mixing (wind and wave action, shipping, and megafaunal activity) and advective transport towards neutral buoyancy. Models and empirical data suggest that size is an important factor affecting interactions between particles and turbulent forces, resulting in higher concentrations of larger, buoyant MPs in the surface layers, while smaller MPs (<100 µm and <10 µm) are found in deeper water layers[81]. Regional and seasonal differences in currents, wave action and topography will affect how particles are transported[82]. Interactions with planktonic

organisms will also affect vertical transport; MPs can become integrated into marine snow (organic-rich aggregates), dictating the vertical flux of materials in the oceans[83]. The association of MPs with marine snow leads to sedimentation of these particles, thereby modulating their fate in the water column and/or affecting their bioavailability [84]. Once MPs reach marine sediment via marine snow formation or biofouling, bioturbation activity of sediment-associated organisms will be important in determining distribution within the sediment[84].

MP research is predominantly performed in European and Atlantic waters, while data from other geographic regions, remain under-represented in the field of MPs research. This underrepresentation results in knowledge gaps in regions where macroplastic is recognized as a major problem[7] but where MP research has not reached the same level of priority. A number of studies report MPs on beaches and in organisms, e.g. birds, and pelagic waters in Southeast Asia[85], indicating levels one order of magnitude higher than in other oceans.

In addition to small- and large-scale spatial differences in MP occurrence, seasonal differences have also been observed. A high influx of freshwater, e.g., from rainfall and rivers[86] or changing rains and winds associated with monsoons[87], can flush MPs seaward or result in increased beaching.

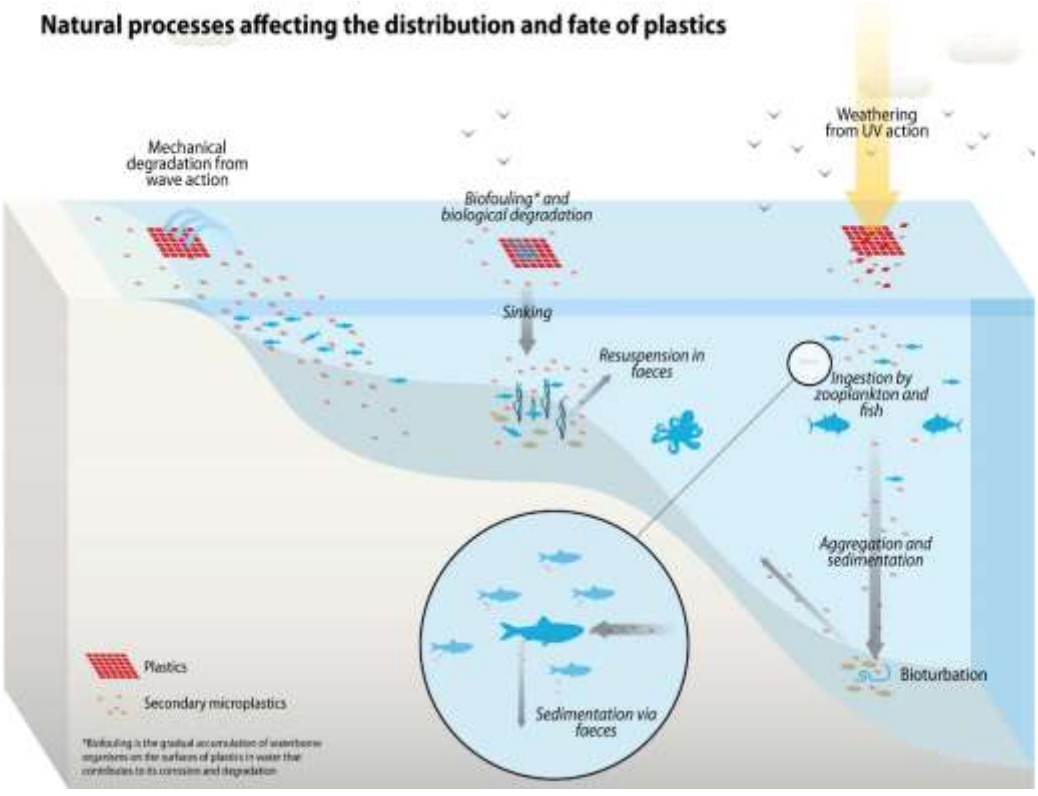

Figure 3. Environmental fate of MPs in the aquatic environment (Graphics: *Riccardo Pravettoni, Marine Litter Vital Graphics*).

WEATHERING, AGING, FRAGMENTATION, AND DEGRADATION

Once MPs have entered the environment, they undergo weathering processes, sometimes also referred to as 'aging'. '**Weathering**' and '**aging**' are used interchangeably and generally refer to changes in MPs that occur during their lifetime in the environment.

Weathering entails physical stress of MPs by various abiotic (e.g., wave activity, abrasion, temperature fluctuations, and UV radiation) and biotic (e.g., biofilm formation, microbial degradation or animal foraging and processing in gastrointestinal fluids) factors[88]. Initial degradation of plastics in the environment likely occurs via abiotic processes, driven by interactions with UV radiation and oxygen, initiating degradation of the carbon-carbon backbone. Photo-oxidation and hydrolysis reactions then proceed[89]. Plastics are found in different environments and habitats (marine, freshwater, and terrestrial environments) and are therefore subjected to different forces and processes that underpin their weathering/aging (**Figure 3**). For example, thermal and light conditions differ drastically between beach, ocean surface layers and deep-sea sediment, where UV light and higher temperature on beaches and ocean surface can induce degradation pathways that may not be relevant for MPs prevalent in the deep ocean floor. MPs in terrestrial systems are exposed to completely different abiotic conditions and (micro)organism communities than those found in marine or limnic environments, driving the aging of plastic material.

Exposure of MPs to various stressors leads to surface and structural changes (e.g., density, size, shape, crystallinity, tensile strength, and molecular weight) in the material. Weathering entails '**degradation'** that refers to a chemical change in the polymer constituents of the MPs, and '**fragmentation**', a process that is more commonly discussed as 'breakdown', refers to the physical breaking of larger particles into smaller particles. Plastic weathering leads to the formation of smaller polymeric fragments, NNPs, oligomers and/or chemical fragments derived from large polymer chains[88,90]. Fragmentation is largely induced by mechanical shearing and weathering. Experimental weathering, modeling and field studies suggest the formation of NNPs in the environment[23,32]. However, the detection of NNPs in complex environmental samples still faces many challenges[91], and it remains difficult to confirm their presence in complex natural matrices. Some preliminary data on NNP formation in natural environments exist. NNPs (PE, PS, PVC, and PET) were detected in colloidal fractions of seawater collected from the plastic accumulation zone in the western North Atlantic Ocean[33]. During plastic degradation, chemicals added to plastic can leach out, releasing not only chemical additives (plasticizers, fire retardants, heat and UV stabilizers, antioxidants, biocides, colorants, and lubricants) but also monomers and oligomers into the surrounding environment (water and sediments)[35,92–94].

After prolonged durations in the environment, MPs associate with organic and inorganic matter and form an eco-corona consisting of macromolecules (adsorbed from biological fluids), DOM, and particulate organic matter (POM))[42]. The acquired eco-corona on the MP surface provides an ideal substrate for colonization of microbial communities (microbes, phytoplankton, etc.) and in a broader sense represents a new, relatively unexploited ecological niche or "plastisphere"[95]. Association with biofilm or '**biofouling'** can lead to

microbial degradation of synthetic polymers, affecting the surface properties, volume and density of MPs and in turn modulating the leaching or partitioning of chemical contaminants[96]. While the potential for biodegradation and mineralization of anthropogenic plastics exists[97], their importance for plastic weathering is unknown.

During their lifetime in aquatic environments, MPs will not only leach chemical constituents but should also obtain a new chemical fingerprint via **association with environmental chemicals** (i.e., organic compounds and metals)[38,98–101]. Many lipophilic chemicals from the surrounding environment adhere to MPs due to their hydrophobicity. The adhering of chemicals to MPs is closely attributed to the chemicals' octanol-water coefficient[102], which indicates the chemical sorption capacity. This property of plastic materials has facilitated their use as passive samplers in environmental monitoring of chemicals[103,104], but this characteristic is now widely discussed in the context of so-called vector or carrier effects of environmental contaminants. Although the contamination of MPs by persistent organic pollutants (POPs) was reported in the 1970s[45], more extensive research on this topic was only recently initiated. In early studies, the concentration of contaminants sorbed onto plastics was shown to be orders of magnitude higher than that in the surrounding water[99]. Commonly found chemicals include POPs such as polychlorinated biphenyls (PCBs), polyaromatic hydrocarbons (PAHs), polybrominated diphenyl ethers (PBDEs) and organochlorine pesticides (OCPs)[38,105]. Most of the older, aged or weathered plastics in aquatic environments are suspected to be in chemical equilibrium with the environment, whereas the newer, more recently disposed materials may still contain higher levels of additives and are likely not equilibrated with the surrounding media[106] .

Different synthetic polymers can have different sorptive properties and mechanisms[101,107,108]. For instance, PE has been shown to have a greater capacity for the absorption of hydrophobic organic chemicals (HOCs) than other polymer types[99,109]. A general distinction between polymer types can be made by classifying polymers into two main categories: glassy and rubbery polymers[110,111]. These two classes of polymers have different sorption mechanisms and affinities to HOCs. For instance, glassy polymers (e.g., PS and PVC) are prone to '**adsorption'** (adherence of chemicals on the particle surface) as the most prominent sorption mechanism. Structurally, glassy polymers have dense structures and few void gaps, restricting the diffusion of chemicals into the polymer matrix. In contrasts, rubbery polymers (e.g., PE and PP) are more flexible and have more space in their polymer structure, facilitating molecular '**absorption'** into the bulk of the polymer matrix.

As plastic degrades, its properties change, and aged and fragmented MPs are thus expected to have different sorptive properties (surface charge, crystallinity, additive content, etc.)[88]. For example, the weathering of plastics in the environment can increase their specific surface area, leading to higher sorption capacity. Smaller particles with larger surface-to-volume ratios have been shown to enhance sorption capacity compared to larger particles[112]. In this context, NNPs are suspected to have a greater capacity to sorb environmental contaminants[113]. However, very little is known about sorption and desorption dynamics during weathering[88] and interactions with mixtures of environmental pollutants.

There is clearly a complex interplay between plastic degradation (physical and (bio)chemical) and ab/adsorption of chemical contaminants. Little is known about the interrelatedness of these interactions and their potential implications for the fate of chemicals in the environment, and the importance of this complex interplay for the MP-biota interface remains unexplored. More studies in this field are needed to predict potential pathways into food webs and associated impacts.

MICROPLASTICS IN AQUATIC FOOD WEBS

Plastic debris is known to enter food chains. Anthropogenic MPs have been described in various animals ranging from those that form the base of the trophic food chain (zooplankton)[114] to intermediate (invertebrates[115] and small fish) and top predators (large fish, seals[116], sea birds, and cetaceans[117]) (**Figure 4**). The plausibility of trophic transfer of MPs has been demonstrated in simplified food chains under laboratory conditions using model particles[30,114,118–120] and has been inferred from the detection of MPs in the tissues of animals collected in the field[117,121]. The prevalence of MP trophic transfer in natural ecosystems is not known, but coastal food webs are thought to experience higher MP exposure stress, facilitating higher intake into aquatic food webs[122], similar to areas of MP accumulation (e.g., gyres).

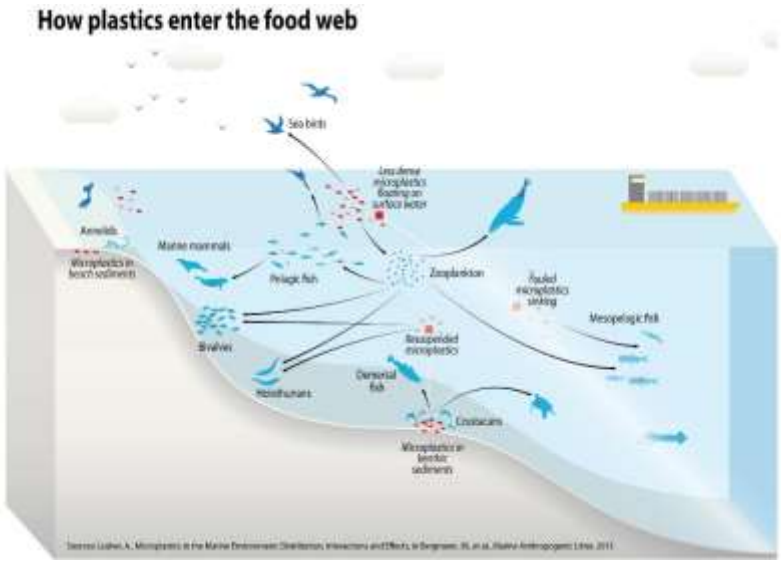

Figure 4. Presence of MPs in marine food webs[117] (Graphics: *Riccardo Pravettoni, Marine Litter Vital Graphics*).

The incidental presence of MPs in aquatic habitats (surface water, the water column or sediments) facilitates the intake of MPs by organisms. The spatial overlap between the MP distribution and the physical presence of biota is the major contributing factor for MP influx into food webs[123]. MPs enter into food webs via prey's ingestion (trophic transfer) entanglement, respiratory intake (inhalation) or adherence of MPs[64,122–126]. Nevertheless, ingestion of anthropogenic plastic debris is believed to be the most prominent pathway of MP entry into biota and trophic chains.

INGESTION AS AN ENTRY PATHWAY

Plastic **ingestion** (intake of MPs through the mouth into gastrointestinal tract via eating or drinking) is a pervasive phenomenon and is exacerbated across different trophic levels, habitats and geographic regions[127]. As a result of increasing environmental research and biomonitoring campaigns worldwide, descriptive information (both qualitative and

quantitative) about the ingestion of MPs by various aquatic organisms is steadily increasing. Detailed information about ingested MPs (e.g., plastic load, composition, size and incidence) can be found in numerous publications reporting the ingestion of MPs in the wild[128]. According to recent estimates, more than 690 marine species ingest plastic debris[129], representing more than 40% of taxa[130] (**Figure 5**).

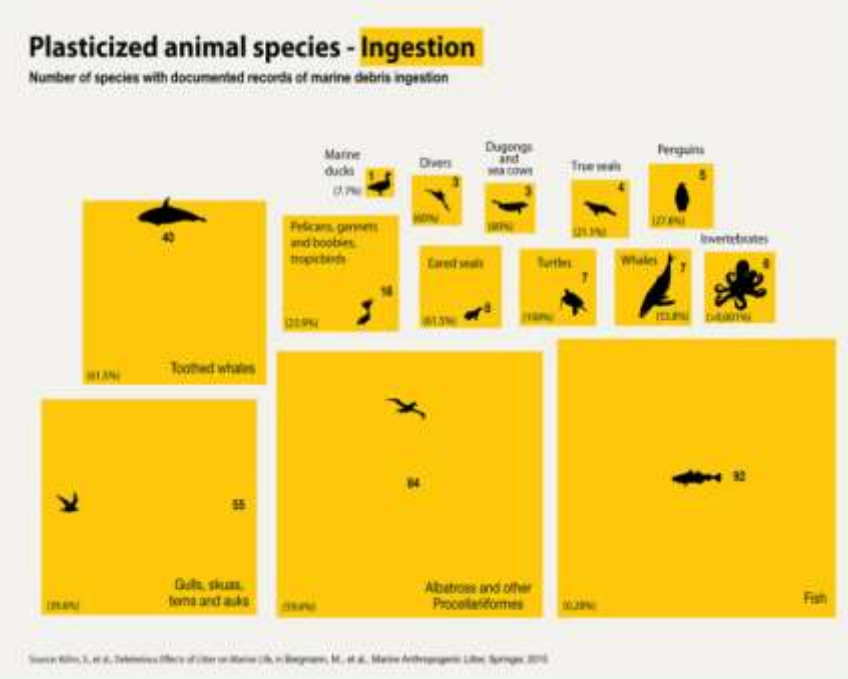

Figure 5. Graphical representation of the incidence of marine plastic debris ingestion[131] (Graphics: *Riccardo Pravettoni; Marine Litter Vital Graphics*).

Nevertheless, most of the data concerning the ingestion of MPs in the field are available only for larger MPs (> 100 µm)[128]. Information concerning the smallest-particle fractions in the natural environment unfortunately remains very limited. It is known that aquatic animals ingest a wide variety of MPs (e.g., fragments, nurdles and fibers), facilitating the entry of these MPs into food chains. The ingested MPs (their quantity, polymer type and size) can vary depending on the animal species and its ecology[132,133]. For example, MP intake is primarily driven by the overlap between the ingestible size fraction of natural prey and MPs and is limited to the size of the filtering apparatus[123,133]. Feeding mode and life history contribute to encountering and ingesting MPs[123]. Organisms in natural environments can either preferentially feed on MPs, or can accidentally ingest them, while foraging on food particles or organisms, containing these synthetic particles.

MECHANISMS FACILITATING MICROPLASTICS INGESTION IN BIOTA

Generally, organisms can ingest items directly (primary ingestion mode) or indirectly (secondary ingestion mode via prey)[134]. While plastic debris itself does not represent a good source of nutrition, plastic ingestion by aquatic organisms suggests the presence of

mechanisms facilitating the preferential intake of nondigestible and non-nutritional plastic fragments. Plastic debris collected from the field contains scraping and biting marks such as visible bite marks from large animals (e.g., sharks, turtles, and seabirds), indicating active foraging for MPs[135,136], or more subtle indications of feeding on MPs exerted by smaller animals. Small invertebrates can graze on biofilms, facilitating coingestion of plastic and associated chemicals[136]. Ingestion of MPs can also occur accidentally as predators ingest particulates, or so-called heteroaggregates**,** associated with zooplankton, bacteria, fungi, and organic or inorganic matter[137], leading to the misidentification and ingestion of plastic debris[134].

MPs may contain pigments that render the particles colorful, leading to visual resemblance with prey. Animal specimens collected from the field often contain colorful synthetic or semisynthetic particles in their guts[64]. Some experimental laboratory studies have demonstrated that animals preferentially feed on colorful items, suggesting that color may be an important factor mediating foraging interactions[138]. For example, a study using gobies found that the fish confused three different colors of MPs with their prey, artemia, and that different populations of fish foraged differently, indicating the possible role of developmental influences[139].

In addition to color, a chemosensory mechanism of MP ingestion has been proposed for a wide range of organisms of varying sizes, from microscopic invertebrates[140] to fish[141,142] and seabirds[143]. In some instances, the ingestion of MPs is facilitated by molecular attractants (e.g., algae-derived dimethyl sulfide) associated with plastics in marine environments, which then stimulate active ingestion by marine animals as a result of prey confusion and association with molecules used in chemical signaling. The presence of a biofilm on MPs has been suggested to enhance MP intake by olfactory foragers, which are attracted to chemical signaling molecules that are associated with plastic and stimulate MP foraging activity[141].

Given the wide dispersal of MPs across different aquatic compartments, there is no doubt that animals will encounter plastic debris in their natural environments and in turn ingest it, thereby facilitating its entry into aquatic food webs. This route is evident in the environment, and plastic is incorporated via trophic interactions, either as a direct food source or as a substrate/vehicle for other nutritional contents (bacteria, fungi, and algae). The importance and breadth of the underlying mechanisms are unknown and need to be investigated further.

BIOLOGICAL FATE OF INGESTED MICROPLASTICS

The fate of MPs in organisms is an important factor not only modulating the fate of MPs in aquatic environments but also determining their potential impacts on biota and biomagnification potential. This will be discussed in the following sections. Potential scenarios of the biological fate of plastic particles in an organism include particle uptake and translocation, accumulation (retention) and elimination (egestion)[144].

Particle **'retention'** (prolonged residence time in an organism) in organisms can play an important role in determining potential biological impacts[145]. Different properties of MPs (e.g., size and shape) can influence retention and its duration in biota.

Some studies suggest that smaller-sized particles can be retained for longer times in organisms[146–149], prolonging the exposure to MPs *in vivo.* Small MPs and NNPs have been shown to tightly adhere to tissues (e.g. intestinal villi in fish[150], foot and mantle of mussels[124]), cross membranes and limiting or delaying their excretion. In other studies, larger and less dense particles were shown to reside longer in organisms[151]. Furthermore, it has been advocated that indigestible synthetic particles may have short retention in gastrointestinal tracts of organisms, and can be quite quickly eliminated[152]. Indeed, a growing body of evidence suggests that ingested MPs (particles and fibers) can be readily easily egested[153] (i.e. if ingested items do not impose physical blockage). This has been experimentally shown in several crustacean species[148,154,155], echinoderms[156], amphibians[157] and in fish larvae and adults[152,153,158] . Particle '**egestion**' (defecation or elimination of particles from digestive system) may occur without exerting damage or stress on organisms. On the other hand, if particles are not excreted, but rather are retained, they may be internalized or translocate in organisms, leading to MPs accumulation in biota (within a trophic level) and further transfer in the food web[122]**.**

The '**translocation'** (movement of particles from target tissue to other tissues) of particles is primarily tied to particle size, which also determines the transport mechanism (i.e., intracellular or paracellular uptake or persorption) of particles and extent of particle distribution in tissues. Some studies have provided evidence for MP translocation in different aquatic organisms: mussels[159], crabs[160] and fish[161–163]. Studies examining the uptake and translocation of particles are highly diverse and include both laboratory (*in vitro, in vivo* and *ex vivo*) and field investigations. For instance, in experiments involving bivalves and microscopic particles, the translocation of MPs (10 µm) into the circulatory system of mussels was documented[144,159]. Particle ingestion (<80 µm HDPE) by blue mussels was followed by cellular uptake and tissue translocation to the digestive gland and gills and subsequent particle accumulation in the lysosomal system[159]. Moreover, relatively large particles (up to 250 µm) have been found to translocate in the gills, ovaries and hepatopancreas of crabs[118,164]. In fish, particles (up to 600 µm) have been found to be transported to the liver, and the occurrence of MPs (124-438 µm PE) in hepatic tissue was documented in wild fish captured in the field[161,165].

Collectively, while the uptake and translocation of larger MPs (> 100 µm) is deemed biologically possible (e.g. via specialized M-cells)[166], the uptake of such particles through intact epithelia is believed to be incidental, or is associated with methodological artefacts[64,167], thus the likelihood of such events and mechanisms remains under discussion. On the other hand, due to their small size, potentiating their bioavailability, NNPs have been experimentally shown to cross biological membranes and translocate to different organs[30,168,169].

Synthetic particles are able to cross epithelial tissues[170], enter lymphatic or systemic circulation, and be further distributed in different tissues[167]. However, the underlying

mechanisms of cellular uptake, translocation and particle retention require further investigation. The duration of particle residence in an organism may influence the fate of the particle in the trophic chain and may determine the potential for biomagnification. Furthermore, particle retention and translocation dictate the likelihood of potential biological effects, especially for the smallest particles, i.e., those reaching the nano range[167]. Thus, we need to address the importance of understanding the perseverance and toxicological implications of these particles in organisms, including humans, as a likely final consumer.

Additionally, we need to further understand the interactions and mechanisms governing the biological fate of MPs in organisms. This phenomenon could be particularly relevant, considering not only the biological fate occurring due to direct NNP exposure but also that within the context of NNP shedding during passage through alimentary tracts *in vivo.* For example, certain krill species possess a “milling stomach”, and ingested MPs fragment into smaller plastic pieces reaching the nano scale in the gut of these animals[171].

BIOMAGNIFICATION

‘Biomagnification’ (increasing concentration of particles at successively higher levels in the food chain) of MPs in food webs is thought to be a potential consequence of MP accumulation in natural environments[42,123]. While the small-sized MPs and NNP have potential to enter trophic food chains, they are expected to biomagnify in the food webs, however, very little is known about the extent of this phenomenon in the natural environment. . Smaller particles are more likely to biomagnify as they can be readily absorbed by animals’ guts and can be retained in circulatory systems or distributed to different tissues, in contrast to intermediate-sized particles, which can be easily egested from the organism. MP fragments or fibers can clog the intestinal tracts of organisms and be retained for considerably long periods of time, thus being able to trophically pass into the food web[123]. Even though, accumulation and retention of MPs in the digestive systems of organisms can facilitate the successive passage of these particles into higher trophic levels, but as these particles are not further translocated and retained in animal tissues, this may not infer to biomagnification of particles in the food web. While discussing MPs’ potential for biomagnification, it is important to acknowledge that mechanisms and pathwyas for biomagnification for solid particulates, such as MPs may be different than for e.g. fat-soluble persistent organic chemicals (where the term is inherently applied).

In the future, to better understand the potential for MP biomagnification, more information is needed about particle uptake and depuration dynamics in aquatic organisms from different trophic levels[122]. For instance, the importance of different MP properties (concentration, size, and surface properties) for defining residence times in biological compartments (i.e., the gastrointestinal tract and gills) is unknown. Integration of such information in computational modeling could allow better predictions of MP fate in organisms and trophic interactions in complex food webs[172]. Such integration is also crucial for improving our understanding of the fate, impacts and potential for bioaccumulation and biomagnification of MP-associated chemicals.

IMPACTS OF MICROPLASTICS

The impacts associated with MP exposure have been studied across different levels of biological organization: from gene to population level (**Figure 6**), thereby providing very different information regarding organisms' interactions, exposure pathways and biological consequences. However, most of the studies addressing impact of MPs have focused on organismal responses, and information on population or ecosystem levels remains fragmented. Negative impacts that are associated with direct exposure to MPs on organisms are explained briefly here, and implications are discussed further below. In general, we are gaining a more nuanced understanding about impacts (or lack thereof) and mechanisms, but herein we will address primarily adverse effects reported in association with MPs exposure. And while effects may influenced by particle properties (size, shape, polymer), chemical exposure (leachates or sorbed environmental contaminants) and exposure scenarios (exposure routes, concentrations, etc.), the importance of aspects for mediating biological effects are also addressed in this report.

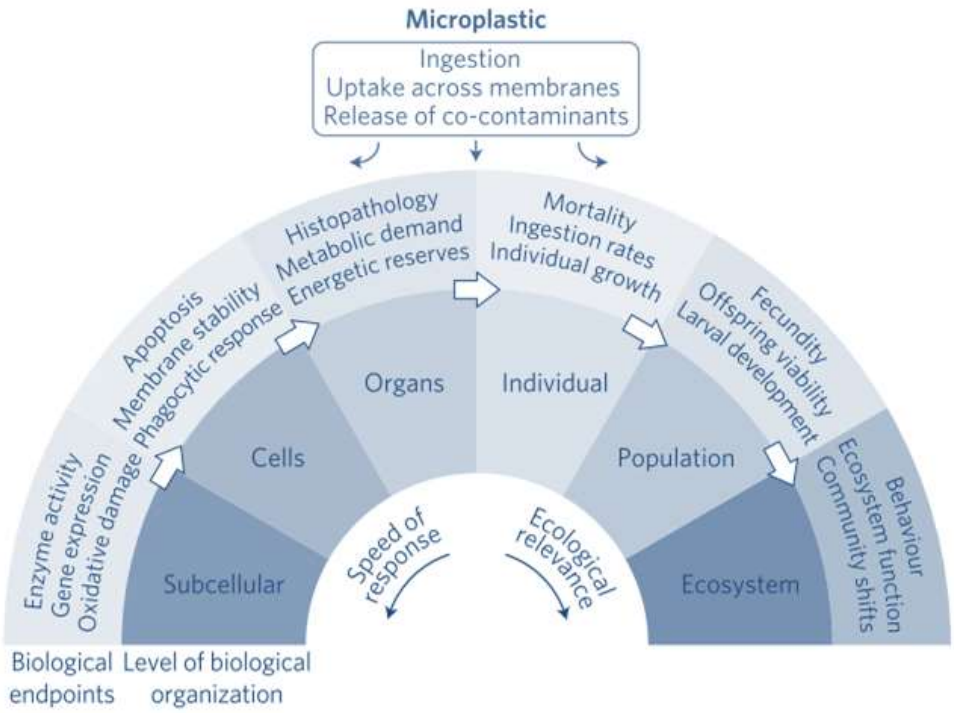

Figure 6. Schematic representation of impacts associated with MP exposure across different levels of biological organization[42] (Data source and graphics: *Galloway et al. 2017*).

MOLLECULAR-CELLULAR LEVEL

Oxidative stress is a prominent mechanism of biological response to stressors and is commonly associated with MP exposure[146,147,173]. Oxidative stress can be induced by direct particulate exposure (i.e., formation of reactive oxygen species on the particle surface), indirectly via chemical exposures, or associated with the general stress response. Additionally, inflammation occurs in association with MP exposure, suggesting that MPs have the capacity to interfere with immune system components and induce immunological responses (e.g., granulocytoma formation, lysosomal membrane destabilization, neutrophil trap release, and cytokine regulation)[159,174–176]. Non-specific innate immune responses may occur as a result of direct particle interaction or/and intrusion (e.g. internalization of MPs or NNPs[167]). Immunological responses as well as other large-scale molecular responses have been studied using gene expression analyses and proteomics. For example, using proteomics analysis, alterations in the hemolymph proteome affecting genes involved in inflammation and structural development in mussels were defined[177]. Other reported molecular effects resulting from MP exposure include lipid peroxidation and DNA strand breaks[180], necrosis[167], perturbation of membranes and activation of detoxification pathways.

TISSUE-ORGAN LEVEL

Several studies have focused on the impact of MP exposure in the gastrointestinal tract, which is the main site of MP exposure, while the potential effects in other primary organs of exposure, e.g., the gills and epidermis, remain largely understudied.

The ingestion-related impacts of (micro)plastics are related to clogging and mechanical damage in the gastrointestinal tract[181]. At the tissue level, MPs induce inflammation[174], increase mucus production[182], or lead to goblet cell hyperplasia[152]. Long-term exposure to MPs results in structural deterioration of the fish intestine[183] but may not exert any measurable effects on intestinal function[152]. Apart from the alimentary tract, effects in hepatic tissue have been reported and include disturbances in hepatic lipid and energy metabolism[162], glycogen depletion, fatty vacuolation and single-cell necrosis[184].

ORGANISM-POPULATION LEVEL

Recent work indicates that MPs can exert effects on behavior in organism from various trophic levels indiscriminately[132].The exposure to MPs have been shown to cause false satiation[181], reduced energy reserves[186], growth impediments[173], and lead to overall reduced fitness and survival[187] in aquatic animals. Some studies also indicate potential of MPs to induce neurotoxicity[180] and swimming behavior alterations [188], disruption of foraging and feeding behavior[31,189–191].

On that note, some organism level responses have shown to translate to population-level effects. For example, impaired gametogenesis and gamete quality , potentially could leading to reduced fecundity and diminished reproduction success and reduced offspring viability in oysters[190]. Also, reduced ingestion, causing to energy depletion and reduced fecundity and

survival in marine copepods[31]. These findings collectively indicate the potential of biological implications, cascading across populations.

POTENTIAL IMPACTS ON COMMUNITIES AND ECOSYSTEMS

To date, most of the research examining potential consequences of MPs for biota has focused on examining biological consequences at the organismal or suborganismal level (e.g., the molecular, cellular, and tissue levels). Although the presence of ecological impacts is predicted based on theory, the evidence for ecological harm caused by marine debris is currently limited and remains inconclusive[192] (**Figure 7**). However, implications at the population or ecosystem level are expected.

Due to the widespread occurrence and persistence of these physical entities (i.e., MPs) in the environment, they are hypothesized to physically alter biogeochemical cycling, change the dynamics of aquatic food webs, and have an impact on large-scale ecosystem processes[8]. For example, the accumulation of buoyant particulates in surface waters likely reduces light penetration, whereas sedimentation of plastic debris on the seafloor could impede the gas exchange between surface water and the interstitial water of sediments, potentially causing hypoxia[193]. These consequences are more likely to be attributed to larger plastic debris that, due to its bulky size, has a greater potential to influence and cause physical damage in natural environments. While MPs have a smaller mass per item, their other properties (i.e., a high surface-to-volume ratio, small size, and greater affinity for biological systems) have ecological consequences. Foreseen ecological consequences in pelagic habitats could have negative impacts on phyto- and zooplankton communities and subsequent consequences for carbon cycling[42]. Furthermore, biofouling of MPs can influence fluxes of organic matter in aquatic ecosystems[194]. Additionally, an association of MPs with alien species could suggest that floating MPs can serve as vectors for invasive species and pathogens[195,196]. These newly formed associations could have negative consequences for biological diversity[95].

Furthermore, plastics and MPs create a new niche that supports microbial growth and has the potential to affect carbon cycling in aquatic ecosystems. Experimental evidence demonstrates that leaching of organic carbon from MPs can stimulate the activity of heterotrophic microbes[197]. MPs can also expedite gene exchange in aquatic environments[198]. A recent review discusses the impacts of MPs on microbial communities, where horizontal gene transfer in the biofilm can affect the metabolic diversity of the microbes, potentially influencing cycling of organic carbon in the aquatic ecosystem. There are indications the proximity of MPs to humans and human microbiomes (e.g. in sewage effluent) could potentiate evolution of pathogenic species, or increase antibiotic resistance reservoirs in the environment.

In sediments, it is believed that presence of MPs could affect the nutrient, oxygen and trace element permeability/transport into sediments, increasing nourishment and food availability for interstitial organisms, which subsequently could alter redox conditions in sediment layer.[199]

There are some experimental studies suggesting that incorporation of MPs in beach soils can alter thermal diffusivity in sediments, and it was predicted that this may have an impact on temperature-dependent organisms and their eggs (i.e. interstitial meiofauna, molluscs, crustaceans and turtles), further affecting associated ecological communities, inhabiting these regions[199]. Exposure to MPs could cause population shifts or affect animals' behavior (e.g., burrowing and prey-predator interactions), which could lead to amendments of ecological functions in ecosystems[42]. For example, in benthic communities, MPs affected the fitness and behavior of lungworms by inducing indirect changes in primary productivity and nutrient cycling in their habitats[200]. MP exposure affects the structure of benthic assemblages[201], influencing the functioning and structure of sedimentary habitats[202].

Due to the inherent complexity and interconnectedness of aquatic food webs, the assessment of potential ecological impacts strongly relies on predictions. The evaluation and prediction for ecosystem-level effects is in its infancy and is based on preliminary findings from existing experimental studies, modelling studies, and theoretical assumptions[42]. In order to better predict potential impacts of MPs on higher levels of biological complexity, the refinement of data concerning concentrations and fate of MPs in the environment and impacts in the organism from numerous trophic levels in the field, is needed.

KNOWLEDGE INTEGRATION

Numerous stand-alone studies have provided valuable information about the potential mechanisms and toxicological effects associated with MP exposure, but collective knowledge integration is urgently needed not only to better understand linkages and relationships between observed biological responses but also to assimilate research findings and stimulate constructive research[190,244]. Existing conceptual toxicological frameworks, such as adverse outcome pathways (AOPs), have been proposed to accommodate knowledge integration and harmonization of MP research with the intention of facilitating knowledge transfer from research to policy making[42]. Although only recently introduced, this approach is starting to be used in MPs research[146]. For example, a tentative AOP was constructed to conceptually illustrate pathways that connect MP ingestion to potential adverse outcomes across different levels of biological complexity and incorporate various subcellular, cellular and individual responses (**Figure 7**)[190,244].

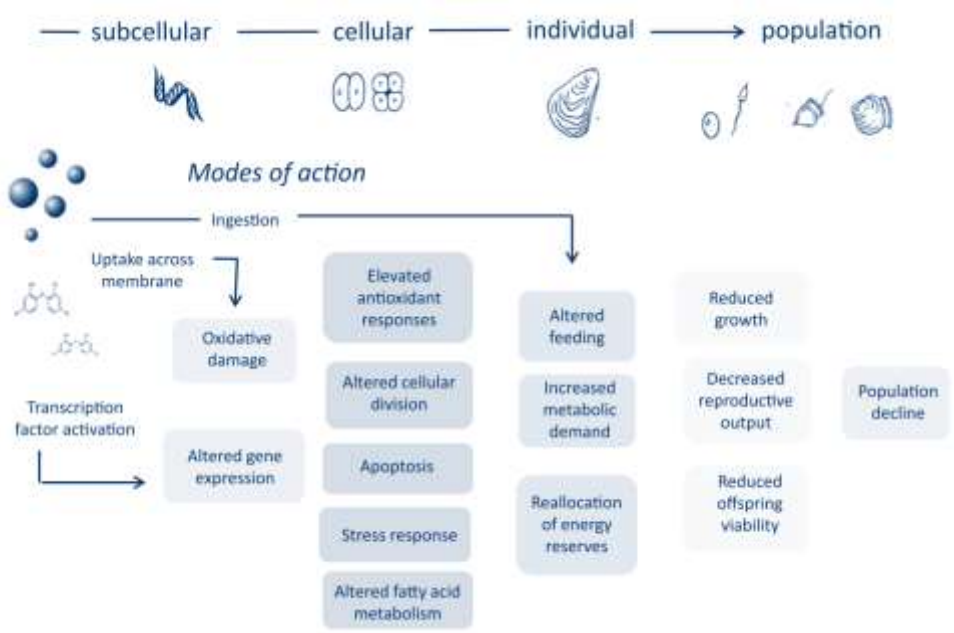

Figure 7. Adverse outcome pathway tentatively constructed for aquatic species exposed to MPs[244] (Data source and graphics: *Galloway et al., 2016*).

Inherently, such a framework provides a generalized sequence of key biological events that are not tied to specific chemicals (i.e., polymers and additives) or particle properties (shape, size, surface charge, etc.). Thus, while integrating knowledge about the potential biological impacts and harm of MPs on/to organisms, it is important to consider the conditions under which experimental studies are conducted (exposure doses and shape and size of particles) and how they could relate to the diversity of observed adverse biological effects and MP exposures in natural environments. For example, the majority of effect-based investigations have addressed the potential toxicity of spherical commercially available MP products without considering the shape or surface topology of the particles. (Eco)toxicological knowledge of environmentally relevant MPs (i.e., those that undergo aging/weathering or environmentally realistic chemical exposures) is currently very limited. There is an evident mismatch between

the types of MPs commonly used in laboratory experiments and those that are detected in, extracted from, and identified in complex biological matrices.

On the basis of existing experimental data, a higher prevalence of biological effects occurs at lower levels of biological organization, e.g., cellular as opposed to organismal. Additionally, the strongest interactions and negative consequences occur at the macromolecular and cellular levels at the lowest particle size ranges (**Figure 8**). The term “microplastics” has been used in studies of the potential effects of MPs across vast size gradients ranging from several nanometers up to 5 mm. Generally, plastics and MPs are referred to as biochemically inert materials that are not capable of directly interacting with biological receptors or molecules[101]. However, owing to their small size, submicron-sized particles and NNPs are bioavailable, can cross biological membranes, and can directly interact with cellular components[245], in contrast to larger plastic debris. Therefore, larger MPs are not expected to pose great risks to aquatic organisms, and stronger (eco)toxicological effects are expected of NNPs and submicron-sized MPs[64,120,146]. Lessons learned from elucidating the (eco)toxicological aspects of nanotechnologies and nanomaterials have been valuable for predicting the potential impacts of NNPs[25,246,247]. Additionally, detrimental effects of large macro- and mesoplastics are highly prevalent in natural environments and pose significant threats to large marine fauna, e.g., by leading to entanglement[12] (GESAMP 2015, 2016). While these consequences are induced by plastics beyond the MP size fraction, large plastic debris remains a major cause of severe and irreversible impacts on aquatic organisms, which are often exempted from MP discussions.

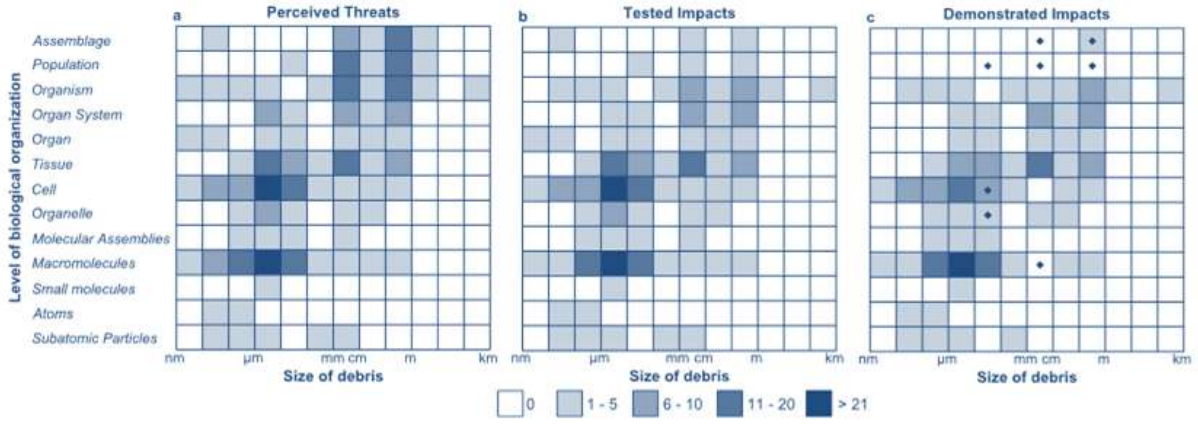

Figure 8. Perceived, tested and demonstrated effects of plastic debris across different levels of biological organization[192] (Data source and graphics: *Rochman et al., 2016*).

Different effect studies are performed under relatively diverse experimental conditions (experimental designs, exposure conditions, target organisms, endpoints etc.), providing variable findings, regarding potential effects, which sometimes can be contradictory. There have been many discussions regarding the exposure levels used in (eco)toxicological studies, which are not considered to be environmentally relevant. For example, the majority of (eco)toxicological studies have focused on the effects of MPs at concentrations 10^2-10^7-fold greater than currently reported environmental concentrations[74,248,249]. The consideration of exposure levels (and the bioavailability) of MPs is important as they determine the potential for toxicological effects. In recent years, an increase in the number of studies reporting an absence of adverse negative effects associated exposure to MPs [152,185,224] (and is growing), providing less alarming findings and balancing ongoing discussions on the danger/harm to biota by MPs. For example, a meta-study summarizing the impacts of MP exposure on aquatic invertebrates and fish reported the absence of or neutral biological effects on consumption, growth, reproduction, and survival[187].

MPs are recognized as particulate stressors in marine and freshwater ecosystems. Thus, the relative importance of MPs in effects on wildlife has been compared to that of naturally occurring particulates (e.g., sand and clay). While this line of research is emerging and more studies will be conducted, MPs are expected to have the same toxicity as naturally occurring particles across multiple levels of biological organization[250]. There is a call for the inclusion of ‘natural’ particles in controlled laboratory studies to either create a more realistic exposure scenario or help differentiate particle effects of MPs from chemical effects.

Conclusive generalizations with regard to the magnitude of MP effects on biota are still difficult to make. There is an emerging consensus that the adversity of MP-mediated effects is associated with the size of the MPs. The chemical exposures of MPs, resulting from chemical additives or via coexposures with sorbed environmental pollutants, need to be considered and will be addressed in the following section of this report.

CHEMICAL EXPOSURE ASSOCIATED WITH (MICRO)PLASTICS

The chemical toxicity of (micro)plastics is associated with the release of constituting monomers (or oligomers) and various chemical additives.

Due to their high molecular mass, polymers themselves are considered inert and nonreactive materials incapable of crossing biological membranes[203]. Nevertheless, a number of plastic-associated chemicals (i.e., unpolymerized residual monomers, NIASs, transformation and degradation products and chemical additives) are able to migrate from polymeric materials and induce toxicological effects. All plastic products contain some degree of chemicals, which can potentially leach into the environment during manufacture, use and disposal[16].

Additives are commonly used in plastics and include solvents, surfactants, plasticizers, stabilizers, biocides, flame retardants, accelerators, and colorants. Groh and colleagues recently published an overview of available data concerning chemicals that can be associated with plastic packaging materials, reporting evidence of 906 substances that are likely associated with plastics; they classified these substances according to data from classification, labeling and packaging (CLP) regulations implementing the United Nations' Globally Harmonized System (GHS)[16]. The authors identified 7 persistent, bioaccumulative, and toxic (PBT) or very persistent, very bioaccumulative (vPvB) compounds and 15 endocrine-disrupting chemicals (EDCs). Thirty-four of the 906 chemicals were also recognized as EDCs or potential EDCs in a recent report by the UNEP. Some additives used in plastics are known to be carcinogenic, mutagenic, or toxic to reproduction and may have detrimental acute effects or long-term health impacts[3]. Such classification has been a long-standing focus in human toxicology and epidemiological studies and has largely focused on certain chemical compounds (e.g., bisphenol A (BPA) and DEHP)[204]. Only recently have these compounds raised ecotoxicological concerns. High levels of phthalates (DEP, DBP and DEHP or its metabolites) were detected in tissues of marine animals (e.g., stranded marine turtles[141], whales[205], and sharks[206]), demonstrating that plastic-derived chemicals are widespread in aquatic environments and can be detected in aquatic organisms. Field studies have reported the presence of additives in organisms in association with ingested MPs[207,208], suggesting that plastic additives are found in biota and that the animals may be exposed to chemical derivatives via the ingestion of MPs[205]. With the current limited evidence (which is also based on correlative assumptions), it remains difficult to confirm that additives are released as a result of plastic ingestion and not via other exposure pathways. Some modeling studies suggest that the migration of certain chemicals (e.g., nonylphenol (NP) and BPA) into biota (lungworms and fish) upon ingestion is minimal[209]. In contrast, it has been experimentally shown that additives can disassociate from plastic and migrate into organisms[210]. Additive-rich plastic materials have the potential to release chemical substances into organisms due to a high fugacity gradient[211] and physiological conditions that enhance chemical desorption[212]. However, the ingestion-mediated effects of plastic additives remain largely unknown.

The toxicity of plastic-associated chemicals has been investigated by testing chemical mixtures or leachates obtained from different plastic materials. Recent studies suggest large

differences in toxicological potency between additive-free versus additive-rich materials[64,213]. These findings suggests that major toxicological concerns are associated with the additives found in plastics[64]. Plastic materials composed of certain polymers (e.g., PVC) may contain up to 60% additives (by mass), significantly increasing the hazard potential of such plastic materials[3]. Laboratory studies testing the toxicity of leachates from commercial plastic products (e.g., single-use packaging materials) have shown leakage of chemicals and increased mortality of target organisms[214]. The toxicity of plastic leachates from recyclable plastics was shown to reduce larval survival and impair the settlement of sessile barnacle species[215]. Additionally, a recent study demonstrated that plastic leachates can affect prey-predator interactions by affecting the prey's behavioral vigilance and antipredatory behavior[216]. At the molecular level, plastic additives are known to possess estrogenic or androgenic activities *in vitro*[217], leading to oxidative stress. Evidently, biochemically active components of leachates derived from new or recycled materials have great potential to induce toxicological effects on biota at multiple levels of biological organization. Additionally, aged/weathered materials can release unreacted monomers or oligomers and chemical additives[35,94] and induce toxicological effects[218]. The intrinsic chemical properties (including IASs and NIASs), age, degradation state, origin and fate in the environment of the polymer can determine its potential to release chemical substances[88]. However, hazards of the chemical mixture(s) liberated from MPs by weathering are often unidentified, and the potential toxicity of weathered materials remains underinvestigated[88].

Overall, chemical exposures associated with plastic-derived chemicals are important when discussing the hazard potential of (micro)plastics. Due to the inherent diversity of plastic materials, potential chemical drivers of toxicity remain largely unknown (e.g., due to analytical challenges and costs of identifying constituents of complex mixtures in nontarget chemical screening). Currently, chemical toxicity testing of plastic-associated chemicals is limited to explorative studies testing leachates (or extracts) of specific plastic products under acute exposure conditions.

MICROPLASTICS AS VECTORS FOR ENVIRONMENTAL CONTAMINANTS

Another aspect associated with chemical exposures of (micro)plastics arises from the reported ability of MPs to sorb contaminants from the surrounding environment[99].

The notion that desorption of MP-bound chemicals in organisms could facilitate an alternative pathway of entry for these chemicals into food chains and increase bioaccumulation and/or adverse effects on organisms[207,219] has raised various concerns, which has in turn stimulated vivid discussions in the scientific community and fostered research on this topic. The hypothesis/assumption that MPs act as **vectors for HOCs or metals** into biota has been an important theme in discussions regarding the potential toxicological effects of MPs. However, investigations of vector effects have mostly focused the role of MPs in delivering persistent chemical contaminants sorbed from the environment, and little attention has been paid to plastic additives[220].

Vector effects have been studied by using measurements from the field (e.g., chemical concentrations in different matrices, correlated with MPs), experimental studies and modeling studies. This research formed the basis of our understanding of the prevalence of this phenomenon and the adversity of MP-mediated vector effects on biota. Vector effects have been studied in a broad range of animal groups (invertebrates[221–223], fish[152,183,184,224,225], and seabirds[226]) and include predictions of MP-mediated effects on trophic food chains[172]. A few studies included desorption experiments in artificial gut fluids[64,227]. It has been experimentally shown that the chemical transfer of MP-bound chemicals is biologically plausible, leading to desorption of chemicals *in vivo* with subsequent biological impacts[183,184]. For example, *in vivo* desorption is expected to be attenuated by the presence of gut surfactants, potentially leading to increased uptake of chemicals by an organism.

As outlined in a critical review by Koelmans et al. (2016)[211], investigations of this vector hypothesis have used different methodologies and experimental designs and have addressed different aspects of the issue. Thus, we lack a clear understanding of mechanisms and potentials as well as speculations about and misinterpretations of the issue. A growing body of experimental evidence and models (using environmental concentrations and chemical properties of specific substances) suggests that chemical transfer (bioaccumulation) and biological effects associated with MPs vector effects is limited and are expected to be greater via other pathways (prey contamination, uptake via water, etc.)[211,223,225].

As previously discussed, we have been aware of the presence of MPs, specifically fibers, in the terrestrial environment for decades, and fibers have even been suggested as a trace marker for the spread of sewage sludge in terrestrial environments[228,229]. MPs are ubiquitous and abundant in terrestrial environments. However, the global distribution of MPs in different types of terrestrial habitats remains inadequately assessed[230], and information about their abundance and composition is fragmented. One of the reasons for this lack of information is that the detection of MPs in complex matrices (e.g., soil and sludge) is procedurally and analytically challenging. Separating, identifying and quantifying is considerably more difficult in terrestrial samples than in aquatic systems (see section above “Sampling and analytical methods”).

One of the largest sources of terrestrial MPs is sludge applied as a fertilizer to agricultural fields[63,228]. The deposition of sewage sludge containing MPs (beads, fibers, and NNPs) in agricultural soils can reach 63000–430000 tons (Europe) and 44000–300000 tons (North America)[231] annually. The widespread agricultural practice of plasticulture, i.e., the use of plastic films or mulch to reduce reliance on rain, nutrient application and pesticides by covering and protecting seedbeds, is applied across large spatial scales[120,232] and contributes to MPs in agricultural lands[233]. Furthermore, degradation and fragmentation of macroplastic debris on land facilitates the entry of MPs into the upper layers of soil. For example, wind dispersal of anthropogenic macroplastic debris in remote arid environments contributes to the accumulation of this debris in desert environments[234]. Other pathways for the entry of MPs into terrestrial environments include runoff from roads or urban areas and municipal water. With increasing influx of MPs into terrestrial ecosystems, soils are becoming long-term storage reservoirs of MPs and NNPs[235].

Additionally, the inclusion of MPs in soil is hypothesized to affect the properties of the soil, change geochemical and biophysical environments, and influence the structural and functional diversity of microbial soil communities[236]. These effects include impacts on soil microbiota and the functional diversity of soil enzymes[237]. MPs have been recognized as anthropogenic components of soil organic carbon cycles[238,239]. Additionally, the incorporation of MPs into terrestrial habitats (by biota) could pose a risk of chemical leaching into soil and groundwater[240]. MPs may affect the bioavailability and transport of organic MP-associated chemicals and/or NNPs[235]. These aspects need to be further investigated, with a particular focus on understanding the physicochemical processes governing plastic degradation, retention times and fate in soil and the interplay between MPs and associated chemicals.

Studies examining the interactions and effects of MPs on biota in terrestrial environments are relatively scarce and have been limited to a few key organisms. The feeding mode of animals influences their exposure[230], and MPs are recognized as food by animals or incorporated into their habitats. Ingestion has been documented in soil-dwelling organisms[240], and MPs can be incorporated into burrows of earthworms and vertically transported through different soil layers[240]. Furthermore, micro-, meso- and macroplastics are ingested by camels, terrestrial

birds[241] and livestock (i.e., cows/cattle)[242]. Regarding potential impacts on biota, studies have focused on the role of MPs as carriers of contaminants (agrochemicals) in soil organisms[243], and although no strong evidence has been found, these questions warrant more detailed research in the future. We have yet to elucidate the impacts of plastic pollution in terrestrial ecosystems as well as the extent to which MPs accumulate in terrestrial food webs, and studies should involve a broader range of organisms (plants, microorganisms, and invertebrates, such as insects)[120].

RISK ASSESSMENT

The identification of risks can differ between disciplines. As Backhaus and Wagner (2018) noted, a marine biologist may consider MPs to be a risk based on their ubiquitous presence in marine environments without any consideration of hazard or harm[251]. Definitions of harm can also be controversial, and the Joint Research Council (JRC) has therefore clarified the use of this term with regard to the prevention of "harm", as indicated in the Marine Strategic Framework Directive (MSFD). The JRC identify three types of harm: ecological (mortality or sublethal effects on plants and animals), social (reduction in aesthetic value or public safety) and economic (costs associated with tourism, vessels, fishing, and clean-up)[27]. Here, we will focus on ecological harm and risk.

Current practices for environmental risk assessment (ERA) in Europe apply a model that compares predicted exposures (often predicted environmental concentrations, PECs) with knowledge of the hazard, or more precisely, a predicted no-effect concentration (PNEC). A PEC value that exceeds the PNEC value indicates risk. The basic premise of an ERA is to inform policymakers of a potential problem, i.e., exceeding an established threshold, for example, the concentration of a specific chemical, which is predicted to result in negative effects. These effect concentrations are based on experimental data using exposures and endpoint measurements from standardized tests, often with a single model organism. This type of risk assessment can be applied to specific regions (e.g., specific bodies of water), groups of organisms (e.g., birds or periphyton communities), and specific processes (e.g., wastewater treatment plants).

ERA frameworks commonly used to assess the risks of soluble chemical contaminants have been adapted to insolvable synthetic polymer particles across different size and chemistry gradients. There are limitations and difficulties in applying this type of risk assessment to MPs, in part since standardized ERA models are used for single chemical compounds and are calculated as a function of chemical concentration (in weight or molarity). MPs, on the other hand, are polymeric particles potentially containing hundreds or thousands of chemical substances. The potential risks associated with exposure may be impacted by additional variables including size, shape, and biofouling. In addition, the scientific community is struggling to agree on which units MPs should be assigned to in exposure scenarios or reports of environmental concentrations: number, bulk weight, surface area, etc.

Risk assessment of MPs is in its infancy and is implemented by adapting established procedures (for soluble chemicals) for application in questions concerning MPs. Currently, there is an evident mismatch between exposures in the environment and reported effects from laboratory studies. While the abundance of MPs is well documented in various compartments (surface waters, the water column and sediments) of aquatic ecosystems, MP interactions with and impacts on biota at these levels of contamination are not evident. The first attempts to conduct ERAs of MPs have only recently been made[252,253]. Preliminary findings suggest that average (reported) global MP concentrations are still within safe limits, with exceptions in certain geographical regions with higher levels of MP pollution pressure (i.e., coastal regions, narrow straits, and urban areas), which have higher anticipated impacts.

Based on the current state of the art of the field, concentrations of MPs are inferred to be safe, according to EU legislation. Notably, current ERA does not incorporate the smallest MPs fractions (and NNPs), which are also considered having greatest impact on biota, thus this assessment inherently remains limited to the largest sized MPs.

An important consideration here is the expected future growth in the global production of plastics. The occurrence of MPs and exposure of biota to MPs are also expected to increase; thus, frequent updates involving ERAs will be needed. Moreover, there is a need to address existing uncertainties (lack of data, accuracy of data, technological implications, particle sizes, plastic chemistry, etc.) in such a framework, allowing the potential risks associated with MPs to be adequately assessed. As there are great uncertainties associated with the nanofraction, future risk analysis should include and consider particles in this size fraction. The existing knowledge base regarding the potential impacts of MPs has increased rapidly in recent years. However, despite recent advancements, the data available for hazard assessment of MPs are not sufficient for robust risk assessment. In the following section, we will examine prevalent knowledge gaps remaining in the contexts of MP exposure and hazard assessment, which are the components of ERA.

KNOWLEDGE GAPS AND PREREQUISITES IN EXPOSURE AND HAZARD ASSESSMENTS

Interdisciplinary knowledge obtained from various scientific disciplines (ecotoxicology, marine ecology, limnology, environmental chemistry, hydrology, oceanography, etc.) has facilitated growth and development in the rapidly expanding research field of MPs. In recent years, scientific progress has substantially improved our knowledge about the occurrence, environmental fate and potential impacts of MPs, but it has also revealed technological limitations, highlighted remaining knowledge gaps and uncertainties in regard to (micro)plastic pollution, and addressed needs for the future.

EXPOSURE ASSESSMENT

A widely addressed implication and challenge in MP research is the lack of international standardization of research methodologies suitable for monitoring (sampling and identification) of environmental MPs. Complementarily, quality criteria for conducting research and reporting data are needed to enhance transparency and facilitate better knowledge integration within and across disciplines[254].

Additionally, methodologies for the detection and quantification of submicron MPs and NNPs in aquatic environments and organisms are needed to determine exposure levels to these small particles[255].

Increased knowledge is also needed on the mechanisms and kinetics involved in plastic degradation and better elucidation of the role that degradation/weathering plays in chemical ab/adsorption in natural environments, in addition to the potential consequences of these processes for MP-mediated vector effects on biota, are needed[256].

HAZARD ASSESSMENT

Despite the recent increase in research examining interactions of MPs with biota and their potential impacts, many unanswered questions remain. Because the definition of MPs includes particles with different chemical compositions and morphologies, detailed reporting and categorization of hazard data are needed. Such processes could include better material characterization for ecotoxicity testing and tailoring hazard assessment towards more environmentally relevant materials and testing conditions[257,258].

The majority of hazard/effect laboratory studies have examined potential adverse effects of MPs at exposure levels orders of magnitude greater than "environmentally relevant" concentrations and/or under relatively "artificial" exposure conditions (type, size and chemical attributes of MPs). Thus, a transition from investigating the acute effects of high exposure levels of MPs to examining long-term or chronic exposure to low levels of MPs is strongly advised to increase the environmental realism of hazard studies. Additionally, current hazard testing is conspicuously skewed towards investigations using commercial MP products (e.g., NNPs and MPs that are uniformly spherical or contain surfactants and suspension-stabilizing agents) with little resemblance and relevance to plastic particles detected in the environment. Therefore, a shift in hazard testing towards environmentally relevant materials is needed and should go beyond single-polymer-type and single-shaped particles.

The diversity of chemicals used in the plastics industry is substantial and can vary greatly between manufacturers; thus, studies that examine the potential toxicity of plastic materials (MPs and plastic leachates) should incorporate chemical analysis to identify potential chemical drivers (plastic degradation products, additives, and monomers)[220].

The majority of studies have primarily focused on marine biota; the impacts on freshwater species are considerably less explored, and the impacts in terrestrial systems are largely unknown. Benthic habitats have been described as a prominent sink for MPs[76,259]. However, information about the impacts on sediment-dwelling organisms in different aquatic habitats is not readily available and is therefore identified as a necessary future research direction. Furthermore, to date, very little attention has been paid to exploring impacts on terrestrial organisms or communities.

Information for impact assessment at higher tiers of biological organization (i.e. population, community) is needed to assess the weight of evidence for MP (ecological) effects on biota and predict their downstream impacts on ecosystem functions.

HUMAN HEALTH RISK

There is evidence for potential health risks associated with (micro)plastic exposure in humans, and MPs have been found in drinking water and food[166,260,261]. The greatest concerns regarding human health are related to potential microparticulate contamination in drinking water, seafood and freshwater products, which were discussed in previous sections of this report. While there are other pathways of human exposure (airborne contamination, contamination of land-based food commodities, medical devices, etc.)[166], they will not be discussed in detail here. An integrative approach of assessing and quantifying the potential for human exposures as well as hazards is needed to estimate potential risks. This section briefly explores human exposures and potential risks in relation to some suspected exposure pathways.

PRESENCE OF MICROPLASTICS IN DRINKING WATER

Concern regarding the presence of MPs in the water supply chain was recently raised as MP particles were detected in tap water[262]. This discovery has fostered a search for MPs in the water supply chain, i.e., in sewage effluent, tap water, groundwater and commercial water[145]. Recent investigations of raw and treated drinking water (residuals from sewage treatment) suggest that MPs are not completely removed, especially particles in the low size range (< 10 µm). The smallest particles are among the most abundant, accounting for 95% of particles[263]. An analysis of MPs in drinking tap water in the Nordic region (Denmark and Norway) did not find significant levels of MPs[264]. In Denmark, 15.6 particles/50 L was found, and 3% of these particles were identified as MPs (e.g., PET, PP, and PS), and the majority of the detected particles were made of cellulose-like material. In Norway, low levels of MPs were detected in drinking water (> 100 µm)[265]. A study in the Netherlands suggested low levels of MPs (>20 µm) in groundwater[63]. The presence of MPs in the water purification and supply chain was explained by the wear of plastic equipment used in water processing. Furthermore, contamination of bottled water by synthetic polymers is an emerging concern regarding the potential implications of MP pollution on human health. Up to 93% of commercial bottled water is contaminated by MPs, with an average of 10 particles per liter (> 100 µm) and most of them are fibers[266]. Evidence for airborne contamination is limited in published studies, but new evidence is being generated that plastic packaging can release MPs into mineral water[267].

Our current understanding of the potential occurrence of MPs in drinking water is based on a few pioneering studies that raised awareness of the potential prevalence of MPs as emerging contaminants in water. More detailed studies are needed and would be beneficial for establishing MP levels for human exposure. Therefore, attention should be paid not only to refining analytical methodologies for MP detection (capturing and identifying the smallest fractions, i.e., NNPs, which currently are not analyzed) but also to increasing confidence in current research findings about the levels of larger MPs in drinking water from both raw water treatment facilities and commercial sources.

POTENTIAL IMPACTS OF ANTHROPOGENIC DEBRIS ON SEAFOOD SAFETY

Since MPs have been detected in the aquatic food chain, popular narratives have advocated for addressing potential risks to human health, especially as the ingestion of MPs in commercial marine organisms is a prevalent phenomenon[128]. Anthropogenic MPs has been found in more than 25% of tested aquatic animals on the market, including fish and bivalves[268]. MPs have been found in mussels and oysters[269], crayfish[270], fish[161,271,272], and seaweed[64] suitable for human consumption (**Figure 9**). MPs have been detected in animals primarily harvested from the wild[128], but there are early indications that MPs can be present in aquaculture or mariculture facilities[56,268,273,274].

These findings not only exemplify the pervasiveness of MP pollution but also highlight the need to increase our understanding of the pathways though which MPs are consumed by humans. This issue has also interested governments, the public and nongovernmental organizations worldwide[56,268], highlighting it as an emerging issue. However, the direct presence of MPs in animals does not necessarily imply direct human exposure, as the digestive systems containing particles are often removed before consumption (for example, in fish)[56]. However, in smaller fish and invertebrates, the removal of MPs is not possible, inherently leading to MP exposure[56,275]. Nevertheless, the exposure to MPs via ingestion of mussels and shellfish is expected to be minimal, especially when contextualized and compared to airborne MPs from dust fallout during a meal[276].

In addition to direct ingestion of MPs by humans, concerns regarding the safety of consuming products has increased as MPs have been implicated in the accumulation of environmental toxicants and transfer to biota[128] or the food chain[277]. These assumptions have not been confirmed as relevant in the context of food safety, and current knowledge about MP-mediated chemical transfer is limited. Some evidence suggests that MPs can transfer pollutants to animals (under restricted laboratory conditions), but the chemical transfer of PBTs and plastic additives from MPs into biota is not predicted to exceed 0.1%[56]. Granby et al. (2017) found that MPs in feed given to seabass and salmon influenced the uptake of brominated flame retardants but that these effects did not influence the toxicokinetics of the compounds in nature, where uptake via natural food and water are expected to be much more significant[278]. According to some theoretical calculations, exposure to PCBs via the consumption of mussels is as low as 0.0001% of the daily intake established by the European Food Safety Authority (EFSA), and exposures to POPs mediated by MP desorption are negligible[279]. Similarly, chemical exposure to a common plastic additive (BPA) via consumption of seafood containing contaminated MPs is likely to be negligible compared to other exposure pathways[260]. Taken together, these findings suggest that the potential risks associated with chemical release from MPs into organisms consumed by humans are low. However, plastic additives (NP, BPA, and phthalates) have been detected in fresh seafood commodities, such as mussels, fish, and prawns[220,280]. Such findings indicate that organisms are exposed to plastic-derived chemicals and that such exposures may be associated with MPs but more likely are the result of exposure via water and prey.

On the other hand, it has also been proposed that MP-related chemical exposures can indirectly affect the safety of commercial fish products by enhancing the formation of hazardous substances in fish fillets or affect the oxidation stability of the fillets[224]. Additionally, the action of MPs as pathogen carriers (e.g., polyamide MPs in mussel farms) deserves further research[281]. While potential indirect impacts on the quality of edible products are likely, more studies are needed.

With increasing concerns about the impacts of plastic pollution, questions regarding the potential impacts of MPs on seafood safety and suitability for human consumption will increase. Uncertainties, common to MPs (e.g., documentation, identification and quantification of MPs at the lowest size ranges; presence of NNPs; and chemical exposures) prevail in the context of seafood contamination by MPs. Currently, there is too little information to confidently assess the risk for human consumption. According to an EFSA report (2016), toxicity and toxicokinetic data on MPs and NNPs in food (not limited to sea products) are lacking and are insufficient to support risk assessment[282]. Analytical methods for the detection and quantification of plastic particles (MPs, but especially NNPs) in food are needed for the quantification of exposures[279] as well as an increased understanding of the potential effects and toxicokinetics in the human gastrointestinal tract.

Furthermore, the identification of sources of MPs in different world regions and the establishment of management strategies[268] are called for. Expanding knowledge about the presence of MPs in seafood products on the market beyond the most commonly used fish and shellfish species will aid in risk assessment and provide guidance for mitigation efforts. Moreover, regarding the potential impacts of (micro)plastics on food safety, the leakage of chemicals during food processing and packaging needs to be investigated[16,224].

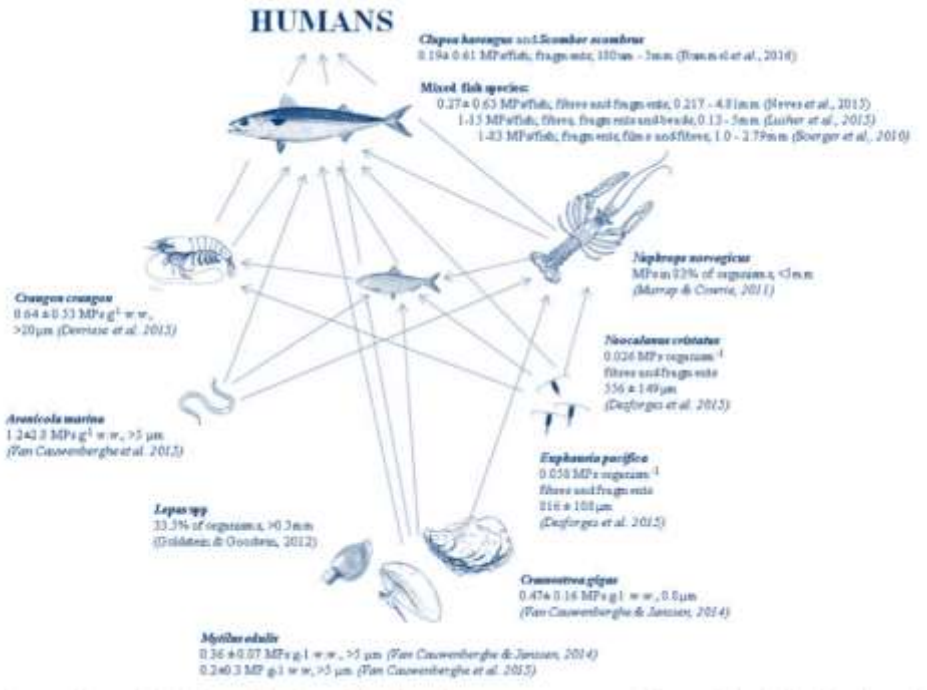

Figure 9. Illustration depicting the occurrence of MPs in the marine food web (Data source and graphics: *Carbery et al., 2018)*.

RISKS FROM A SOCIOECOLOGICAL PERSPECTIVE AND RISK PERCEPTION

As we have described above, work by the scientific community is steadily increasing knowledge concerning the effects of MPs on individuals of species inhabiting numerous niches as well as on ecosystems. However, our understanding of the occurrence, fate and consequences of plastics in the environment is still limited, as is our ability to perform proper assessments of the risks posed by these materials. This lack of understanding has promoted a call for the treatment of marine plastic as a planetary boundary threat[283], as marine plastics have been identified as having the potential to cross precautionary boundaries meant to identify a safe operating space for humanity or to avoid shifts in Earth system functioning[284]. MPs are an irreversible contaminant displaying global ubiquity and should therefore be treated using the precautionary principle.

Societal awareness of environmental problems associated with plastics is widespread, and the increasing media attention is likely due to the pervasiveness, visibility and relatability of the problem. Media coverage of plastic- or MP-related issues is correlated with the increasing number of scientific publications. A recent study in Germany found that media articles focusing on MPs shifted the framing of the threat from a distant and indistinct marine problem to a discussion focused on the occurrence of MPs in local waters or food and drink[64]. Further, media analysis highlights an existing discrepancy between public risk perception and scientific findings described in the scientific literature[285].

While frameworks for calculating risks are well established in the scientific community, perception of risk is more complicated. The risk narrative is based on hypotheses and not established facts, granting fear, anger, etc., and leading to misinterpretation of scientific findings based on superficial knowledge. The risks of MPs may be exaggerated by the media as journalists dwell on uncertainties and knowledge gaps, introducing biases for more scandalous headlines. One example is the recent articles about the findings of MPs in human feces (e.g., a report from SVT) based on a non-peer-reviewed report from conference proceedings, where methodologies and sample sizes were not available for examination. This lack of peer review renders this type of communication virtually useless, as it does not account for methodological contamination or provide a frame of reference for the findings. Indeed, one of greatest challenges for scientists communicating research results lies in contextualizing findings without trivializing the complex plastic pollution problem[260]. As a result, human exposure to and effects of MPs are currently understudied.

Nevertheless, the framing of risk perception as something opposed to risk assessment assumes that beliefs among the general public are shaped by bias, prior beliefs and knowledge (or lack thereof) and previous personal experiences[286]. The public opinions expressed should, however, not be seen purely as an opportunity to engage in scientific outreach and communication as a means to diminish the information gap. The differences between risk assessment and perception may instead represent the framing of a risk or problem that differs from purely scientific questions and takes into account societal inequalities, historical events and governmental policy practices[287].

The media can influence the spread of knowledge, which in some cases can lead to the establishment of new social norms and pro-environmentalism, including support for new policy action. For example, recognizing the impact of plastic waste on megafauna in marine environments, numerous countries and regions around the world have introduced market-based strategies and policies (bans, taxes or fees) related to single-use plastic carrier bags or microbeads in facewash and cosmetics[288]. However, as the authors note, few studies evaluate the effectiveness of these activities in mitigating problems, and strategies for implementation and monitoring are important.

Additionally, the relationships between pro-environmental behaviors and policy support are not clear and are in some cases negative[289]. Thus, while norms can positively affect policy, individuals who actively exhibit low-cost, pro-environmental behaviors (such as avoiding the use of plastic bags) do not always demonstrate positive spillover and support for pro-environmental policies. The opposite is also considered in the literature; implementations of new policies can induce new social norms that are pro-environmental, although this relationship is not linear. This relationship is complicated by factors including the strengths of existing norms (health, safety, and antilittering), levels of policy jurisdiction (local, regional, national, and international), the influence of plastics industries in different regions, materials interests and the manufacturing economy[290]. The popularity of nudge theory has led to attempts to apply these interventions to behavioral economics and decision making regarding plastics and the use of noncoercive strategies to induce change, both in norms and behavior[291]. Rivers et al. (2017) showed that while nudging had a limited impact on the use of reusable bags, other factors including social groups and economic status confounded the results[292].

Importantly, as argued by Wagner, individual actions, like recycling or choosing not to use plastic carrier bags, may not be effective in combating environmental decline[293]. Many of the problems we face today are beyond the scope of the individual, and will require massive restructuring on a societal level and international collaboration. This is realized in the push for industries to change their practices, become active in technological advancements, and in producing plastics that will enable for the incorporation into material flows in a circular economy[15].

SUMMARIZING REMARKS

Microplastics are recognized as an emerging wide-spread contaminants in natural ecosystems. MPs are detected everywhere, however the global abundance and distribution of MPs is currently is known to larger-sized particles. Our current knowledge is rather limited about exposures to relatively small MPs, which are currently exempted from environmental monitoring, due to lacking technological and analytical tools. Quantitatively resolving spatial and temporal distribution of smaller micro-size fractions or NNPs in complex environmental matrices, is immanently needed to increase our understanding about prevalence and distribution of these particles and subsequent estimation of exposure levels to biota.

Ever-growing number of studies have demonstrated potential of MPs to inflict (eco)toxicological effects on biota, and depict impacts caused either from direct (particle) exposure, or exposure to plastic-associated chemicals (i.e. additives or chemicals absorbed onto MPs). The majority of negative effects that are reported in association with MPs exposure detect effects on molecular-cellular and organismal level. Many of studies that grant our understanding about potential consequences of MPs are derived from short-term acute exposures with unrealistically high exposures of MPs or performed in artificial test conditions, and provide us essential information on mechanisms and pathways in which MPs interact with biota. However, these findings entail precautionary principal and forecast “worse-case” scenarios, which may not prevail in the natural environment and may not translate to impacts on ecological communities or ecosystems.

Laboratory studies demonstrate sub-micron MPs and NNPs uptake and accumulation in a wide range of organisms [24,105,231] and prompt towards the high likelihood for toxicological effects on biota, compared to larger plastic particles, which have limited uptake and translocation. Potential of such particles to cause harm and adversity in natural environment are unknown and remain one of the main (eco)toxicological concerns and require future investigations. Apart from small size, hazard potential (micro)plastics is also associated with chemical content (additives, monomers), which are biochemically active and can commence potential toxicity (i.e. oxidative stress, endocrine disruption) after leaching from synthetic polymer material. The toxicity mediated via ab/adsorbed environmental contaminants onto MPs (so-called vector effects) is expected to be of minor importance for subsequent toxicological effects and bioaccumulation of these chemicals in organisms.

Up-to this day, the global ambient abundance and distribution of MPs is considered to be low, predicting minimal exposures, thus impacts on biota. Current risk predictions are at its infancy, based on fragmented information, hampered by uncertainties, thus do not comprehensively assess ecological risks, associated with MPs exposures. Refinement of information about environmental MPs exposures, as well as associated impacts (across multiple levels of biological complexity) is vital for improving risk assessment of MPs, allowing us to adequately grasp the full-extent of potential consequences and impacts of (micro)plastic pollution.

It has been demonstrated that MPs interact with various organisms from different trophic levels, facilitating their entrance to (aquatic) food webs, also reaching humans, top consumers. This raise concerns about the potential risks not only to organisms directly ingesting them, but also for potential of these particles accumulate and biomagnify in the food chains. While, interactions, pathways and potential implications of MPs entering aquatic food chains are being investigated, involvement of various stakeholders (i.e. research, governmental, private and public sectors) is observed, not only raising the awareness and interest in this issue, but also setting monitoring and mitigation measures.

AKNOWLEDGEMENTS

The authors are thankful to Dr. Agathe Bour for reviewing this report and providing feedback. We would also like to acknowledge Marine Litter Vital Graphics (GRID Arendal, Author: Riccardo Pravettoni) and Anne-Charlotte Hanning at RISE for the possibility to use illustrations for this report.

Source: https://www.grida.no/resources/6933 (2018-12-18).

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