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Overview of the ecotoxicological impacts of micro and nanoplastics in aquatic environments

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Abstract

Micro- and Nanoplastics (MNPs) are emerging pollutants in aquatic environments, posing significant Eco toxicological risks to marine and freshwater ecosystems. MNPs interact with biotic and abiotic components, leading to bioaccumulation, trophic transfer, and potential toxicity in aquatic organisms. The Ecotoxicological effects of MNPs include physical damage, oxidative stress, inflammation, reproductive impairment, and disruption of metabolic functions in aquatic species. This paper provides an overview of recent findings on the sources, environmental fate, and biological effects of micro- and nanoplastics (MNPs) in aquatic ecosystems. While broad in scope, the paper offers a preliminary summary of current knowledge, highlighting the need for more research on monitoring, risk assessment, and mitigation approaches to safeguard aquatic biodiversity.

Keywords: Micro and nanoplastics; Marine pollution; Freshwater ecosystems; Bioaccumulation; Environmental risk assessment

Introduction

Globally, plastic pollution in fresh and marine water environments has become a concerning issue with plastic debris and plastic fragments contributing to water quality problems. Made for high versatility and stability from various types of synthetic or semi-synthetic organic polymers, plastics have been meeting our industrial and household needs for over 100 years with increasing annual production trend. However, poor disposal management and lack of recycling habit has resulted in their uncontrollable release into the environment (Wang *et al*. 2021). Its chemical and physical resistance has helped plastic to persist in the oceans for an indefinite time, accumulating on the sea floor or travelling over long distances (Ter Halle *et al.* 2017). Plastic pollution has been negatively impacting the aquatic life as reports have found evidences of ingestion of, or entanglement in plastic wires or nets for benthic invertebrates, birds, fish mammals, and turtles (Koelmans *et al*. 2014).

Once in the environment, plastic debris are subjected to photo, chemical and physical degradation processes which fragment them into wide range of particle sizes from macroscopic to microscopic. Microplastics (MPs) are those plastic fragments of less than 5mm in size and found in air, soil and sediments, freshwaters, seas, oceans, plants and animals (Al-Thawadi, 2020), while plastic particles less than 100nm are the nanoplastics (NPs). Both MPs and NPs are either intentionally manufactured in industries for different applications (primary) or the products (secondary) of the weathering process that larger plastic fragment undergo in the environment (Al-Thawadi, 2020).

A major portion of plastic in the ocean is made up of macro plastic debris by mass ($kg.km^{-2}$); however, MPs and NPs account for larger proportion by number ($items/km^2$) (da Costa *et al.* 2018). Global microplastic distribution is estimated to be between 93 and 236 thousand metric tonnes with about 5.25 trillion pieces of MPs at sea especially in subtropical gyres (Eriksen *et al.* 2014). It is, however, being said that the micro plastic concentration in the world

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is underestimated due to insufficiency of standardized detection and quantification methods.

On the other hand, neoplastic distribution around the world is yet to be assessed as there is no established analytical method for its detection and identification, but experiments have showing NPs' generation under laboratory conditions and the recent discovery of their presence in sea water (Ter Halle *et al.* 2017) makes them an undeniable component of plastic pollution.

With rising global plastic production, there is an emerging concern for the increasing concentrations of micro- and nanoplastics, their ecological implications as contaminants and their interactions with other contaminants in aquatic environments (Saavedra *et al.* 2019). These inert polymeric particles can be potentially ingested by a wide range of organisms causing problems such as obstruction, pseudo-satiation, loss of energy, etc., and may make their way through the food trophic levels, eventually impacting human health. Moreover, the toxic additives such as plasticisers, UV-resistance chemicals, etc. added to improve their properties may leach from the polymers, and their tendency to sorb co-contaminants such as persistent organic pollutants (POPs) and heavy metals may cause negative morphological, behavioural and reproductive changes to the organisms on exposure (da Costa *et al.* 2018), as supported by few evidences concerning their toxicity on aquatic organisms including algae, ciliates, crustaceans, fish and invertebrates (Saavedra *et al.* 2019).

While extensive studies have been done on the sources, abundance and negative impacts of plastic macroplastic in marine ecosystems, the researches on smaller sized particles are recent and still inadequate, with NPs, even being potentially the most hazardous contaminant, received the least attention of all (Koelmans *et al.* 2015). The main aim of this paper is to address the pervasive problems of plastic pollution and inform the readers about the sources, existing methods for identification and quantification, distribution, fate and transport, and ecotoxicological impacts of microplastics and nanoplastics on organisms in freshwater and marine systems by using references of the studies conducted on them.

Plastics

Considered as one of the greatest technological innovations in human history, plastics have become widespread today with its global use in industries, pharmaceutical productions, and commercial and municipal applications (Wright *et al.* 2013b; Crawford and Quinn, 2016). Since its invention in 1907 and the following mass production of plastics, a 'throw-away' culture has been created especially with the single-use plastic items. The rising rates of plastic production, lack of habits of recycling and its durability have made plastics recognized as one of the greatest challenges of environment that our species has ever faced Crawford and Quinn, 2016).

Origin of plastics

According to The International Union of Pure and Applied Chemistry (IUPAC), plastic is defined as a 'polymeric material that may contain other substances to improve performance and reduce costs'.

The exact time as to when plastic appeared in our world is quite indiscernible. But the person who succeeded in developing the first fully-synthetic polymeric compound known as Bakelite in 1907 and in commercially influencing the plastic industry was a Belgian chemist Leo Hendrick Baekeland. By the end of 1930s, more than 200,000 tonnes of Bakelite were produced and made into vast range of household products, changing the dynamics of the plastic market (Crawford and Quinn, 2016).

Types of plastic polymers and their uses

All plastics are made by the polymerisation process, i.e. the connection of individual molecules called monomers in a repeating pattern to form larger chain-like molecules (macromolecules) known as polymers. For example, the polymerisation of monomer ethylene forms the widely used plastic polyethylene polymers (shown in Figure 1 (a))which can be used to a polyethylene bag (Figure 1 (b)) (Crawford and Quinn, 2016).

There are various types of plastic polymer which can be typically either natural or synthetic. Examples of natural polymers include silk, wool, starch, and protein, while those of synthetic polymers are polyethylene(PE), polyethylene terephthalate (PET), high-density polyethylene (HDPE), polyvinyl chloride (PVC), low-density polyethylene (LDPE) , polypropylene (PP), polystyrene (PS) and polyurethane (PUR) made from raw materials such as natural gas, coal and oil and are normally classified as plastic).

Different forms of plastic exist in global markets, with polymers such as PE, PP, PVC, PS, PUR, and PET dominating the markets and are hence most commonly encountered in the environment (Al-Thawadi, 2020). PET, HDPE, PVC, LDPE, PP, PS and PUR constitute 90% of the world's total production of plastic, with PP, PE and PVC comprise 24%, 21% and 19% of total plastic production worldwide, respectively (Wright *et al.* 2013b).

Some of the types of plastic polymers, their uses and associated toxicity levels are briefly described below-

i) High-density polyethylene (HDPE) is used to make water, juice, milk, beauty products and beauty products containers. If exposed to high temperatures and sunlight, HDPE leaches synthetic estrogenic chemicals which can potentially damages endocrine system and greatly influences reproduction and health of vulnerable organisms.

ii) Polyvinyl chloride (PVC) polymers are commonly used in pipes, food wraps, jackets and toys in bath. When in contact with water, endocrine-disrupting agents (i.e. phthalates and bisphenol (A) (BPA)) are released from PVC, which are regarded highly hazardous.

iii) Polypropylene (PP), a low hazard polymer, is the most extensively produced polymer globally (Wang *et al.* 2017). It is used widely in items like medicines, carpets, automotive parts, paper currency, etc.

iv) Polystyrene (PS) is often used as a packaging material or for take-out food. The component styrene in the PS leaches out when exposed to hot liquid, is regarded 'anticipated human carcinogen' and endocrine disruptors, and may also create irritations in the respiratory system (McGoran *et al.* 2017.

The additives such as BPA, phthalate acid esters (PAEs), perfluoroalkyl substances (PFAs), nonphenol (NP) and brominated flame retardants, known as plasticides, used in plastic products (sometimes making up to 50%) to alter or enhance their properties exacerbate the problems that come with abundance of plastic in the environment. BPA, Bisphenol S (BPS) and Bisphenol F can potentially cause obesity, asthma, and reproductive issues, and alter hormones. Their small molecular size and their not being chemically bound to plastic gets them readily leached from polymers under suitable conditions and easily get sorbed to other polymers once they are freely floating.

Plastics in the aquatic environment

This review synthesizes recent research, including key studies from 2024 and 2025, to elucidate the eco-toxicological impacts of MNPs in aquatic environments, focusing on their distribution, interactions with organisms, and implications for ecosystem health. Microplastics (MPs) have been detected across various aquatic environments, indicating their pervasive presence. For instance, in the Meghna estuary of Bangladesh, MPs were found in all surface water samples, with abundances ranging from 33.33 to 316.67 items/m^3. Fibers constituted 87% of the detected MPs, predominantly smaller than 0.5 mm in size. Similarly, studies in the Bay of Bengal have reported MPs in the gastrointestinal tracts of commercially important fish species, with varying concentrations depending on feeding habits. MNPs enter aquatic ecosystems through various pathways, including wastewater treatment plants, runoff, and atmospheric deposition. Recent studies highlight the widespread distribution of MNPs in both marine and freshwater systems. For instance, a study by Li *et al.* (2025) investigated the spatial distribution of MPs in coastal sediments, revealing concentrations ranging from 0.025 to 4.701 items/m^3 in surface water, with significant accumulation in benthic sediments (Sultana *et al.* 2024). Similarly, Wang *et al.* (2024) reported high MNP concentrations in urban aquatic systems, attributing these to industrial discharges and inadequate waste management practices (Faisal *et al.* 2025). These findings underscore the ubiquitous presence of MNPs across different aquatic compartments, from surface waters to deep-sea sediments.

NPs, due to their smaller size, exhibit distinct distribution behaviors compared to MPs. demonstrated that NPs have a higher propensity to remain suspended in the water column, increasing their bioavailability to pelagic organisms (Bappy *et al.* 2025). This size-dependent behaviour, as noted by

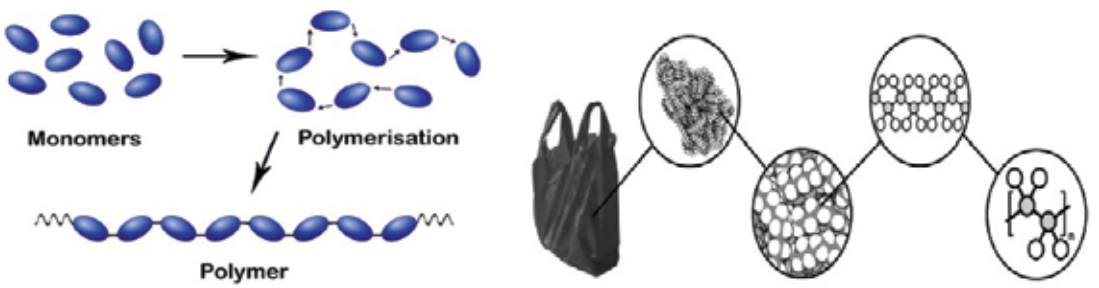

Fig. 1. (a) On the left, monomers polymerize in repeating pattern to form polymers. (b) On the right, a polyethylene bag is composed of masses of polymers which in turn are made up of monomers (Crawford and Quinn, 2016)

Zhang *et al.* (2025), influences their transport and fate, with NPs showing greater mobility and penetration into biological tissues (Hossain *et al.* 2025). These studies emphasize the need to differentiate between MPs and NPs in environmental monitoring and risk assessments due to their varying ecological impacts.

MNPs are readily ingested by aquatic organisms across trophic levels, from primary producers like phytoplankton to higher predators such as fish and marine mammals. Liu *et al.* (2025) documented significant bioaccumulation of MPs in oysters, with concentrations reaching 2.374 items/g (wet weight) in natural estuaries, highlighting their potential to enter the human food chain via seafood consumption (Paray *et al.* 2025). Similarly, Zhao *et al.* (2024) found that NPs accumulate in the tissues of commercial fish species, causing cellular alterations such as oxidative stress and histopathological damage (Hossain *et al.* 2024). Trophic transfer amplifies the ecological risks of MNPs. A study by Kim *et al.* (2025) revealed that MPs ingested by zooplankton are transferred to fish, leading to bio magnification in higher trophic levels (Parvin *et al.* 2025a) This transfer not only affects individual organisms but also disrupts food web dynamics, as MNPs can alter predator-prey interactions and reduce reproductive success. The potential for MNPs to act as vectors for adsorbed contaminants, such as heavy metals and persistent organic pollutants, further exacerbates their toxicity, as demonstrated by Yang *et al.* (2021), who found enhanced arsenic adsorption by NPs, intensifying toxic effects on submerged macrophytes (Parvin *et al.* 2025b).

The eco-toxicological effects of MNPs are multifaceted, encompassing physical, chemical, and biological impacts. Physically, MNPs can cause blockages in digestive tracts, reducing feeding efficiency and growth rates. reported that MPs induced significant mortality in mussels at high concentrations (2160 mg/L), though such effects were less pronounced at lower, environmentally relevant concentrations (Faisal *et al*., 2025). Chemically, MNPs act as carriers for pollutants, increasing their bioavailability. For example, Zhang and Goss (2020) showed that polystyrene NPs inhibit StAR expression in fish, disrupting reproductive processes via activation of HIF-1α pathways (Hossain *et al.* 2025).

Biologically, MNPs induce oxidative stress, immune suppression, and metabolic disruptions. Found that NP exposure in algae triggered reactive oxygen species (ROS) production, leading to lipid peroxidation and reduced photosynthetic efficiency (Hossain *et al.* 2024a). Similarly, observed that MPs in shrimp caused gill damage and hepatotoxicity, impairing energy metabolism. These studies collectively highlight the sublethal effects of MNPs, which may have long-term consequences for population dynamics and ecosystem stability (Hossain *et al.* 2024b). The pervasive nature of MNPs threatens aquatic ecosystem health by altering biodiversity and ecosystem services. Wang *et al.* (2024) noted that MNP accumulation in sediments disrupts benthic communities, affecting nutrient cycling and habitat quality. Furthermore, the transfer of MNPs through food webs poses risks to human health, particularly through seafood consumption (Rahman *et al.* 2024). Liu *et al.* (2025) developed an integrated risk-based framework to assess human exposure to MPs via oysters, estimating significant intake levels and potential liver damage. These findings underscore the need for comprehensive risk assessments that consider both ecological and human health endpoints. Recent advancements in MNP remediation include physical, chemical, and biological approaches. Reviewed strategies combining microbial degradation with physical pre-treatments, showing promise in reducing MNP concentrations in aquatic systems.

Plastic debris are found in terrestrial, freshwater, estuarine, coastal and marine environments, and has even been found in remote places such as deep-sea sediments, submarine canyons, and Arctic sea ice (Horton *et al.* 2017). Since the commercialization of plastic products in the early 1950s, plastics production has seen a continuous rise, and this trend is likely to increase in upcoming years. The worldwide production of plastics was 1.7 million tonnes in 1950 **(**Al-Thawadi, 2020) and in 2019, it reached to 368 million tonnes (Plastic Europe, 2020). By 2050, it has been projected that further 32 million tonnes of plastic is likely to be produced (Hossain *et al*. 2020).

A major percentage of the total plastic produced annually is not recycled or reused resulting in ultimate dumping of these non-biodegradable polymeric plastics in landfills or in freshwater, estuarine and marine environments (Al-Thawadi, 2020). Additionally its extensive prevalence as a marine debris is attributed to its light weight and durability (Wright *et al*. 2013b), and also to the lack of management of fishing gears (Lusher *et al*. 2017). Between 60-80 % and up to 96.87% of all debris found in the marine environment consists of plastic materials (Lusher *et al*. 2013; Marques-Santos *et al*. 2018). It has been estimated that about 150 million tonnes of plastic have already been discarded into the oceans at a rate of 8 million tonnes per year, which means around 15 tonnes of plastic per minute (Hossain *et al*. 2020). Among all types of pollutants released by humans, plastic wastes can, therefore, be considered to be the most dominant in the environment (Marques-Santos *et al.* 2018).

The persistent nature of plastic and its impacts on the aquatic ecosystems were first identified from the recovery of several plastic pieces from the stomach of a Laysan Albatross chick carcass in 2005 (Crawford and Quinn, 2016). Plastic debris influences the ecosystem by causing problems such as entanglement and ingestion. About 100,000 marine mammal deaths were reported every year in the 1980s due to the entanglement in plastic fishing lines and nets (Moore, 2008).

Plastic degradation in the environment

Once plastics are in the environment, they undergo through various disintegration routes and thereby form macroplastics (> 25 mm), mesoplastics (5-25 mm), microplastics (< 5mm) and nanoplastics (< 0.1µm). There are two major pathways by which plastics are commonly degraded such as – a) abiotic degradation and b) biotic degradation.

a) Abiotic degradation is the mechanical disintegration of plastics, which can be caused by changes such as freezing, thawing, pressure changes, and water turbulence brought about by climatic or meteorological conditions, as well as by animal activities, which only alters the morphology of the plastics. Other abiotic types with the most intense impacts on the molecular bonds of plastic materials are the *photo*-, *thermal*, *oxidative* and *hydrolytic* degradations. Of all these, plastics in the environment are severely damaged by *photo* degradation, which is the cleavage of polymeric bonds by UV and visible light spectra. This occurs at a maximum when plastics are exposed on beach surfaces, but when present at the surface of seawater, they degrade at a much slower rate in an oxygen deficient environment (Al-Thawadi, 2020). Plastics of sizes less than 1 mm can amount to 3% by weight on highly impacted beaches (Wright *et al*. 2013a). *Thermal* degradation is rarely observed in nature, as high temperatures (375-500°C) are not reached. *Oxidative* degradation is caused by the introduction of oxygen into the polymer matrix – either photo or thermal-induced, releasing free radicals that promote further plastic degradation. Possibility of observing *hydrolytic* degradation in the environment depends on the presence of covalent bond groups such as ester and ether groups in the polymers. This degradation process alters the molecular weight and hence the strength of the plastic, making it prone to further degradation.

In marine waters, wave action and sunlight exposure are two primary causes behind plastic undergoing fragmentation, which increases the number of particles per unit area and surface area. However, fragmentation by water turbulence or wave action as in coastal areas is less likely to occur in many freshwater systems. On terrestrial lands, plastics fragments form mostly by UV radiation and temperature fluctuations (Horton *et al*. 2017). As plastic fragments, the resulting pieces end up with higher sorption capacity and higher hydrophobicity (Ma *et al*. 2016).

b) Biotic degradation is caused by the actions of organisms, including bacteria, fungi and mealworms (Horton *et al.* 2017). The high-molecular weight, hydrophobicity and cross-linked polymer chains make many polymers (e.g. polyethylene and polystyrene) extremely resistant to biodegradation. Moreover, the bio-degradation occurs only when polymers are exposed to these specific plastic-degrading organisms- such conditions are not ideally found in the environment (Horton *et al.* 2017) and requires an indefinite amount of time (Moore, 2008)

Microplastics and nanoplastics

Microplastics (MPs)

Usually the particles of sizes less than 5mm in their longest dimensions are widely accepted as MPs, particularly by organizations like the National Oceanographic and Atmospheric Administration (NOAA) of the United States of America and the Marine Strategy Framework Directive (MSFD) of the European Union. The earliest study that detected the presence of MPs in the marine environment was carried out in the early 1970s (Carpenter and Smith, 1972), but it was not until 2004 that the term ‘microplastic’ started becoming popular after findings of Thompson (2004).

Types of microplastics

There are two major types of MPs that can be observed in the environment, which are- i) primary microplastics and ii) secondary microplastics.

i) Primary microplastics are deliberately engineered to micron sizes and produced in different industries for uses in various products such in cosmetics and personal care products as microbeads, in detergents, lubricants, surface cleaning agents, pharmaceutical ingredients, etc. (Al-Thawadi, 2020). They are generally uniform in composition, colour, size, and shape (shown in Figure 2) (Syberg *et al.* 2015).

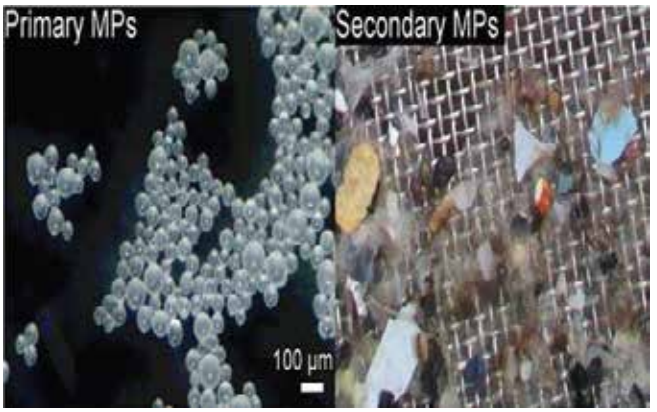

Fig. 2. Primary and secondary microplastic (2017 *Marine pollution bulletin* review of micro /nanoplastics)

ii) Secondary microplastics (shown in Figure 2) are the products of the degradation pathways that larger plastic pieces undergo to form MPs. They can also derive from the abrasion of vehicle tires, which have been blown away by wind and washed by rain into aquatic habitats (Al-Thawadi, 2020). Unlike primary MPs, they are generally much more diversified in shape, size, colour and composition (Syberg *et al.* 2015). Another source of secondary MPs can be the synthetic fibres. During washing, each garment releases 1900 fibres per garment. They travel along with primary MPs in wastewater drainage systems (Horton *et al.* 2017).

Figure 2. On the left, primary microplastics, such as the polyethylene beads (10–106 µm), are pictured. On the right, a sample collected from the Mediterranean Sea of micron-sized secondary microplastics from the degradation of larger plastic pieces is pictured (Syberg *et al.* 2015).

Nanoplastics

Nanoplastics (NPs) are synthetic or heavily modified polymeric particles with colloidal properties (Kokalj *et al.* 2021*)*. Their size range is still a matter of controversy as some authors use the size range between 1 nm to 100 nm (Lusher *et al*. 2017), whereas other authors prefer the whole nanometer range (1nm to 1000nm) as the size range (Wang *et al*. 2021).

Types of nanoplastics

Like microplastics, nanoplastics can be either manufactured in nano-scale (primary), or unintentionally produced from larger plastic debris (secondary) (Kokalj *et al.* 2021). Primary and secondary NPs are briefly described below-

a) Primary nanoplastics are bottomed-up synthesized or top-down milled for uses in coatings, medical diagnostics drug delivery, magnetics, optoelectronics and electronic devices (shown in Figure. 3) (Al-Thawadi, 2020).

b) Secondary nanoplastics are unintentionally formed from the weathering degradation (nanofragmentation) of larger plastic objects (shown in Figure 3), and also from c) microplastics inside personal care products or from food and beverage packaging (Kokalj *et al.* 2021). Weathering produces NPs of different sizes as demonstrated by Lambert and Wagner (2016) and Mattsson *et al.* (2021). Secondary NPs with higher surface areas are more hazardous than spherically synthesized primary NPs as they have stronger adsorption capability of contaminants, which may become bioavailable to organisms (Baudrimont *et al.* 2020).

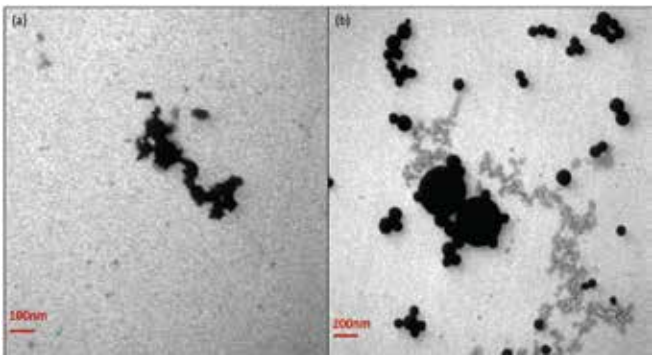

Fig. 3. Electronic microscopy images of (a) polyethylene NPs degraded by UV from aged-microplastics sampled in North Atlantic Ocean (b) a mixture of standard polystyrene latex particles of different sizes (primary nanoplastics) (Gigault *et al.* 2018).

Sources of Micro- and Nanoplastic Contamination in Aquatic Environments

Aquatic environments mainly receive primary micro- and nanoplastics from diffuse sources. One of their fundamental diffuse (indirect) sources is wastewater from households and industries. Even though some Waste Water Treatment Plants (WWTPs) are capable of removing 99.9% primary MPs from domestic or industrial drainage systems, still a small percentage that may bypass filtration systems represent a huge number of MPs which typically get discharged in effluents to surface water bodies (Horton *et al.* 2017). Additionally, many countries do not have such efficient sewage systems and even discharge untreated wastewater directly into water courses. Many studies have found that microfibers are the most abundant of all microplastic forms, with primary microbeads from beauty products as another major contributor to microplastic pollution in freshwater and marine environments (Horton *et*

al. 2017). Sludge from WWTPs also contains substantial amounts of plastic particles. The uses of urban and industrial waste water (treated or untreated) and sludge applications on agricultural lands are another two of the major indirect routes that MPs and NPs are released in the environment. Moreover, the injection of effluents from WWTP and industries into aquifers as one of the many techniques for managed aquifer recharge (MAR) may potentially contaminate fresh groundwater aquifers. Studying the fate of MPs and NPs in WWTPs is therefore imperative to understand their behaviour and transport means within different treatment stages. It is also crucial to analyse the proportions of plastics that are leaving through the treated effluents against those retained in the sludge, and also determine the areas along the treatment trains where MPs and NPs may be building-up. Urbanisation of the area near the water bodies is also a crucial factor determining the presence and abundance of particles, and can result in large variation in a relatively small area by introducing substantial particle concentrations to the environment (Horton *et al.* 2017).

Other common indirect routes of contamination include accidental release, improper disposal methods and undiscriminating discards especially near areas where many industries operate. They inadvertently release micro- and nanoplastics during manufacture, transport and use, becoming one of the significant sources of aquatic MP and NP contamination. Runoff from urban and rural areas depending on their land-use, runoff from agricultural lands through drainage ditches or storm water drains from roads containing worn-tire particles, fragments of road-marking paintings etc. are also major sources of macro-, micro- and nanoplastics in riverine systems (Thompson, 2015). Wind action can transport macro- and microplastics to freshwater systems as studies found evidences of substantial amounts of microplastic fibres in the atmosphere. Construction materials and household dust can also be carried by wind (Horton *et al.* 2017). The sources of microplastic contamination in aquatic bodies are graphically illustrated in Figure 4.

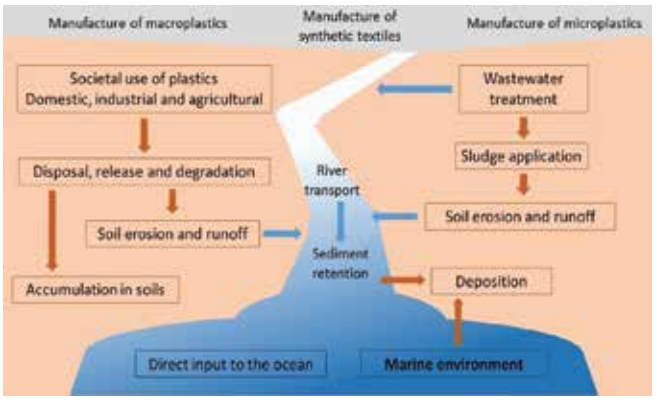

Fig. 4. A graphical representation of the sources of MPs in aquatic environments (Horton *et al.* 2017)

Identification and Quantification of Micro- and Nanoplastics

Assessment of risks and hazards posed by the MPs are understood from quantifying MPs released in the aquatic systems and determining their fate and transport (Horton *et al.* 2017). While the analysis of concentration of macro- and microplastics has been widely done using conventional sampling methods (plankton nets), the assessment of nanoplastic presence, types and abundance in the oceans is still controversial as there has been insufficiency of established sampling and of polymer-type identification techniques (Marques-Santos *et al.* 2018; Koelmans *et al.* 2015).

Sampling and Pre-Treatments

Sampling methods and their associated pre separation, separation and analysis methods are summarized in Figure 5. Sampling method depends on the kind of samples: biological, water or sediment. For biological samples, dissection is employed mainly for larger organisms such as fish and sharks to separate gastrointestinal tract to visually identify microplastics (Nguyen *et al.* 2019). In case of water and sediment samples, mid-water column and benthic nets, neuston nets, manta trawls plankton nets and sieves/filter of different ranges of pore sizes are used to collect plastic particles particularly of larger sizes.

Following sampling, biological tissues or organs are commonly digested in acids or bases to assess the presence of MPs or NPs (Nguyen *et al.* 2019). Separation of MPs from minerals is typically done using density floatation

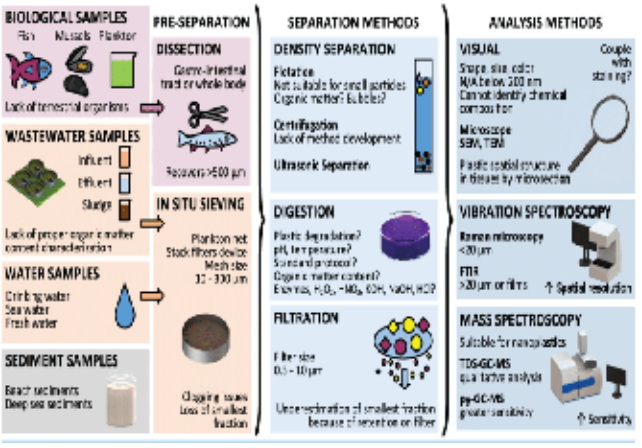

Fig. 5. A summarized representation of separation and analysis of micro- and nanoplastics in the environment (Nguyen *et al.* 2019)

techniques. Microplastic coatings (i.e. biogenic materials or biofilms) and microplastic embedded in organic-rich matrices requires pre-treatments such as using Fenton's reagent (H_2O_2 + Fe catalyst) or enzyme digestion to separate and quantify MPs (< 1mm in size).

Quantification and characterization

Quantifying and characterizing can be done visually for microplastic particles of sizes greater than 500 µm. Visual identification is inexpensive and simple, but it produces incorrect results for MPs prone to embrittlement, fragmentation or bleaching, or having biota crusts on them (Lusher *et al.* 2017), and it also misidentifies natural particles like aluminium silicate, quartz or calcium carbonate as microplastics. Several studies have supported this method to be unreliable with significant over- and under estimation with more than 70% identification errors. More reliable instruments- mid-infrared (FT-MIR) spectroscopy, near-infrared (NIR), Conventional Raman Spectroscopy, Coherent Anti-Stokes Raman Scattering (CARS), pyrolysis gas chromatography mass spectrometry (Py-GC-MS) and thermal extraction desorption gas chromatography mass spectrometry (TED-GC-MS) – can be used instead. Among them FT-MIR and Raman spectroscopy are commonly used in microplastic analysis.

In NPs' detection, techniques such as UV-VIS spectrometry, electron microscopy, field flow fractionation (FFF) or dynamic light scattering (DLS) commonly employed for nanomaterials may help under controlled laboratory experiments (Koelmans *et al*. 2015), and commercially produced fluorescently labelled particles are mostly used which helps in detection or tracing by e.g. flow cytometry, fluorometry, fluorescence microscopy and confocal microscopy, thus overcoming the typical analytical difficulties associated with NPs (Kokalj *et al.* 2021).

Table I. Global distribution of microplastics in freshwater and marine environments

Water body type	Study location	Study findings	Study
River	Rhine river from Basel, Switzerland to Rotterdam, The Netherlands	52,364 to 3,931,062 particles/km^2	Mani *et al.* 2015
Aquifers (Groundwater)	Karst aquifers in Illinois, USA	Samples from springs and wells from two karst aquifers contained 6.4 particles/L with a maximum concentration of 15.2 particles/L	Panno *et al.* 2019
Lake (Sediments)	Lake Ontario	Bottom sediment contained 241 microplastic pieces per kg dry weight	Corcoran *et al.* 2015
Lakes and Rivers	Wuhan, China	1660 ± 639 n/m3 to 8925 ±1591 n/m^3	Wang *et al.* 2017
Ocean	North Atlantic Ocean	Deep-sea water samples contained 70.8 particles/m^3	Courtene-Jones *et al*. 2017
Sea	Northwestern Mediterranean Sea	Surface water - 112 * 10^3 (particles / km^3)	Schmidt *et al.* 2018
Polar water	Arctic	surface water 0.34 (± 0.31) (particles/m^3)	Lusher *et al.* 2015
Ocean	North Atlantic subtropical gyre	10 000 - 250 000 pieces/km	Ter Halle *et al.* 2017

Global distribution of Micro- and nanoplastics in freshwater and marine environments

MPs are ubiquitous in the environment and are considered to be the most abundant form of solid waste on Earth (Al-Thawadi, 2020). Distribution of MPs is a complicated matter as it is affected by several factors including physical, chemical and biological factors (Sun *et al*. 2018). The perpetual rise in the usage of plastics is causing the amount of MPs to continually increase along with the potential damages to the aquatic organisms (Hossain *et al.* 2021). The presence of MPs has been found in surface waters, beaches, deep sea sediments, water columns, coastal waters, estuaries, rivers, and even in aquifers with gyres, industrial and heavily populated coastal areas (Sun *et al*. 2018) as MP hotspots (Wright *et al*. 2013b). Additionally, one of the most impacted regions in the world by microplastic abundance has been the 'Mediterranean Sea' (Marques-Santos *et al*. 2018). While numerous studies on the distribution and abundance of MPs in marine water bodies have been done, there have been relatively fewer studies on the freshwater aquatic systems.

In case of NPs, it is difficult to get a clear picture of their distribution in aquatic environments due to lack of adequately established analytical methods (Baudrimont *et al.* 2020). However, after the discovery of the presence of NPs in sea water samples from the North Atlantic Gyre (Ter Halle *et al.* 2017), there is a fear that nanoplastic concentration will rise with the increasing plastic debris degradation (Baudrimont *et al.* 2020), and hence its ecological consequences must also be considered. Findings from several studies on the distribution of MPs in aquatic environments have been provided in Table I.

Transport and fate of Micro- and nanoplastics in aquatic environments

The overall transport and subsequent fate of MPs are governed by various factors such as number of local sources, water surface area, river water velocity and ocean currents, water body depth, particle characteristics such as density, colour, shape and size, sediment transport, weather conditions like wind, rainfall pattern and flooding, and topographical and hydrological characteristics of the environment. The mobile marine organisms such as mammals and fish can play part in the dispersal of MPs over long distances through ingestion and following egestion of consumed microplastics (Horton *et al.* 2017). The rotation of the strong Ekman ocean currents can get MPs trapped and accumulated in higher concentrations in central areas of ocean gyres and convergent zones happening globally in oceans (Thompson, 2015).

Nanoplastics' surface properties and different environmental conditions influence their fate and transport in water (Oriekhova and Stoll, 2018). They also frequently collide with water molecules and existing ionic species which may prevent it from settling down the water column as often seen with macro- and microplastics. Consequently, they randomly move throughout the water solution resulting in a phenomenon known as Brownian motion. Like all colloidal substances, nanoplastic particles have also the potential to be associated with dissolved organic matter and inorganic (trace metal, metal oxides, etc.) colloids and hence form aggregates (hetero-aggregation) which can both be stable and unstable in the presence of physical (UV light, temperature etc.) and chemical (ionic strength, pH etc.) conditions. The shape, size and concentrations of the aggregates influence the dispersion properties of nanoplastic (Gigault *et al.* 2018)

Factors determining the fate, bioavailability and toxicity of micro- and nanoplastics

Sizes, shapes, surface charge, colours, functional groups and compositions of polymers (density) of plastic particles are important in evaluating their toxicity and interactions with their co-contaminants, as these affect the sorption capacity, bioavailability and uptake in an organism (Bhagat *et al*. 2020). Many studies have been found to focus on size, shape, colour, and polymer density of MPs as factors determining their fate, while in case of NPs, much attention has been drawn upon their surface functional groups. The morphological characteristics of MPs and NPs influencing their availability, toxicity and uptake are briefly described below:

a) Size determines the extent of its impacts on the range of organisms in the aquatic environments and hence is a vital aspect to consider when studying the particles .The smaller size of MPs means they are more available to organisms at the lower trophic levels than those with larger dimensions (Lusher *et al*. 2017), as evident in Sun *et al*. (2018)'s study where zooplankton retained about 72% of <200μm MPs and 96% of <500μm MPs. Cellular damages are also more likely to occur by smaller sized particles (Bhagat *et al*. 2020). Small dimensions of microplastics also correspond to high surface area to volume ratio which dictates the leaching and uptake abilities of chemicals (Lee *et al*. 2019). Majority of lower-trophic organisms differentiates between particles to a limited extent and hence ingest anything of proper size. Organisms at higher trophic level may intake microplastics when mistaking them for prey or during normal feeding activity (Wright *et al*. 2013b). Besides particle size, the physiological (particle to mouth ratio) and behavioural

characters of the aquatic organisms also dictate the ingestion possibility of the particle by vertebrates and invertebrates (Horton *et al.* 2017).

b) Shapes of MPs are generally categorized as fragments, fibres, beads, foams, and pellets (Lusher *et al*. 2017), each likely having different adverse impacts on the aquatic organisms (Wright *et al*. 2013b) and also on their egestion with microspheres more easily released than irregular one (Santana *et al*. 2017). In many studies, fibres in aquatic organisms seemed to be the dominant among all microplastic shapes (Sun *et al.* 2018). 68.3%, 16.1%, and 11.5% of the microplastics in gastrointestinal tract of fishes sampled in Lusher *et al.* (2013)'s study were composed of fibres, fragments, and beads, respectively.

c) Microplastic colour like size also determines the extent of uptake by aquatic organisms. Predators like pelagic invertebrates and some commercially important fish which ingest their prey based on colour can accidently eat microplastic due to colour resemblance to their prey items (Wright *et al.* 2013b).

d) Polymer density determines the positions of MPs in water column, their buoyancy and their subsequent differences in interactions with the aquatic biota. Microplastic polymers like PVC sink in the water column because of their higher density than that of sea water, whereas low-density polymers like PE are likely to stay afloat at the water surface (Lusher *et al*. 2017). However, there are processes like bio-fouling, colonization of organisms onto the plastic surface, bio-film formation, degradation and fragmentation of MPs, and the leaching of chemicals added during manufacture which can alter their inherent density and consequently their location in the water (Lusher *et al*. 2017). Biofilm development on plastic surface or hetero-aggregation with suspended solids, algae and detritus, may cause particles to sink to the sediments (sedimentation) (Koelmans *et al*. 2015) making them available to benthic suspension and deposit feeders and detritivores. However, this biofilm can also be removed by foraging organisms (de-fouling), which makes MPs lighter to rise back to the water surface where these might encounter filter feeders, planktivores and suspension feeders residing at the top layers of water column (Wright *et al.* 2013b). MPs may remain suspended in the water column due to turbulence and water flow (McGoran *et al.* 2017).

e) Surface functionalization - Surface properties such as charge and functional groups of NPs determine their behaviour, and ecotoxicological consequences causing potential severe damages in single cells, embryos or whole organisms (Marques-Santos *et al.* 2018). Coating development on the particles' surfaces by natural organic matter, such as humic substances, proteins, extracellular polymeric substances, etc., affect their stability and toxicity to organisms (Saavedra *et al.* 2019). A study conducted by Saavedra *et al.* (2019) found that the positive amidine ($PS(-CNH_2NH_2^+)$) nanoplastics have stronger negative impacts on D. magna, T. platyrus and B.calyciflorus in freshwater than negative carboxyl (PS(-COO-) nanoplastics due to electrostatic attraction, as microorganisms are, by default, negatively charged. Despite the importance of surface functionalization in determining the impacts of MPs and NPs, it has not received much attention for comprehensive study.

Ingestion and interaction routes with aquatic fauna

Numerous studies on aquatic species, particularly from marine water, have reported ingestion of MPs in a wide range of species with different feeding techniques including amphibods, lugworms, mussels, fishes etc., their accumulation in lower trophic level organisms and also their trophic transfer between species especially bivalves and crustaceans (Kokalj *et al.* 2021). Besides the above factors dictating bioavailability of MPs and NPs, species initial susceptibility to these particles also determines their likelihood to be harmed by their interactions with plastics. Different species have different feeding strategies, so are their interactions with MPs and NPs, among which selective feeding for particle ingestion is widely exhibited (Wright *et al.* 2013b).

Deposit and detritus feeders

Benthic inhabitants (i.e. detritivores and deposit feeders) are exposed to MPs that has sunk and deposited in the sediments. Deposit feeder A.marina ingest MPs selectively based on size, whereas scavengers feeding on debris exhibits non-selective feeding strategy ingesting MPs along with the sediment (in table II) (Wright *et al.* 2013b).

Suspension feeders, planktivores and filter-feeders

Several laboratory studies have reported that suspension feeding marine ciliates such as sea urchin, sea star and sea cucumber, and filter feeders such as echinoderm larvae (table II) capture and engulf MPs of appropriate sizes. However, whether the MPs are egested or accumulated in the gut has not been experimentally determined (Wright *et al.* 2013b).

Marine zooplankton, particularly of the herbivorous members, has been found to eat low-density MPs floating on the sea surface, and benthic suspension feeders like bivalves are exposed to sinking microplastics. Prior to ingestion of particles, bivalves capture facilitated by cilia, retain, sort them according to size, shape and density and discard unwanted particles. However, the sorting is done irrespective of the particle quality, and hence microplastic particles are not rejected and get ingested. Besides entering the food chain via ingestion, smaller plastic particles have the capacity to electrostatically adsorb to the lowest trophic level organisms such as freshwater and marine algal cells (in table II), which depend on factors like algal morphology and motility (Wright *et al.* 2013b).

Fish ingestion of plastic particles has also been reported possibly during their normal feeding activity. Lusher *et al.* (2013)'s study found such phenomenon by substantial numbers of 10 fish species examined from the English Channel, and 92.4 % of the plastics was MPs of sizes smaller than 5mm. Several studies have observed that MPs are retained in zooplankton community with an average of 12.24±25.70 pieces/m (Sun *et al.* 2018).

Trophic cascades of micro- and nanoplastics in aquatic environments

MPs and NPs may enter food chain (shown in Figure 6) starting with microalgae at the base of the chain, which in turn, are ingested by zooplankton (for example, copepod, brine-shrimp, and daphnia), bi-valves, marine ciliates (Wright *et al.* 2013b), fish and other organisms. Some of the particles accumulate in their bodies over longer than expected duration (Kokalj *et al.* 2021) or adhere to surfaces or external appendages, and a portion of them are probably released from bodies in faecal pellets (Santana *et al.* 2017) mostly without any damage (Ma *et al.* 2016). However, expecting particles cascade from one trophic level to another as predators eat prey shortly after MPs intake would depict an environmentally inaccurate exposure scenario as particle distribution are influenced by many biotic and abiotic forces, and exposures with preys are variable with time (Santana *et al.* 2017).

Several studies have been conducted demonstrating the uptake of MPs from water or sediment, but without much focus on the trophic interactions with the contaminated

Table II. Marine and freshwater organisms' vulnerability to microplastic and nanoplastic ingestion and their interaction routes (Wright *et al.* 2013b)

Species	Interaction
Marine algae e.g. *Scenedesmus*	On encounter with nanoplastics, it adsorbs them especially when positively charged
Grazing microzooplankton e.g the marine ciliate *Strombidium sulcatum*	Selective feeding activity according to size indicates the ability of uptake microplastics of appropriate size.
Benthic deposit feeders e.g. the polychaete *Arenicola marina* and the holothurian *Holothuria floridana*	*A.marina* exhibits size-based, deposit-feeding activities in high-density microplastic rich sediment at the sea floor. This suggests their capability to eat MPs of proper size; *H.Floridana* shows selective feeding of particles preferably for fibrous shaped microplastics.
Benthic scavengers e.g. the crustacean *Nephrops norvegicus*	Passive ingestion of fibrous microplastics through food it forages or sediment evident from the gut analysis. Sediment in marine water was observed to be enriched with fibres.
Mesozooplankton e.g. echinoderm larvae, calanoid copepods , chaetognaths	Size-based selective feeding activity.
Benthic suspension feeders e.g. the bivalve *Mytilus edulis*	Ingestion of sinking microplastics of low numbers

food. Then there have been many experiments which supported trophic transference by finding presence of micro-sized plastics in the gut cavities of the consumers. These findings did not provide any evidence of these particles persistence in their tissues, an important aspect in assessing the potential impacts of transference along the food web (Santana *et al.* 2017). Some studies have found MPs in the tissues of predators after feeding with highly contaminated preys, which increases the risks associated with microplastics, but using high MP concentrations is not representing realistically accurate situation. Santana *et al.* (2017)'s experiment maintained standards by addressing the inconsistencies raised with the experiments carried out for plastic bio-transference. It showed microplastic transfer from Perna perna mussels to predators like crab and pufferfish confirming the trophic cascading, but found no MPs remaining in their tissues proving that they have been egested. However, the transfer of microplastic between trophic levels is a concerning matter in itself.

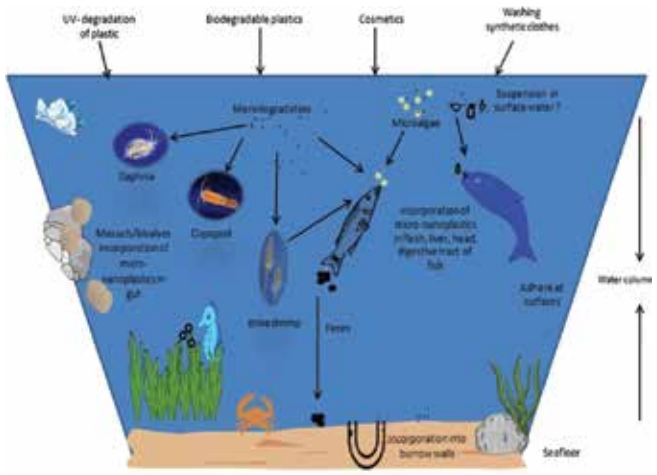

Fig. 6. The fate of micro- and nanoplastics in an aquatic environment (Al-Thawadi, 2020)

Ecotoxicological impacts of micro- and nanoplastics on aquatic organisms

MPs and NPs as environmental pollutants have been gaining interest among scientists and researchers in this plastic age (Bhagat *et al.* 2020; Horton *et al.* 2017). Between these two, NPs are considered to be the most hazardous pollutant found in marine litter (Al-Thawadi, 2020), yet have been least studied (Koelmans *et al.* 2015). To understand the ecotoxicological impacts of MPs and NPs, it is important to know the meaning of ecotoxicology, which can be defined as 'the study and effect of toxic agents in ecosystems' (Bradl *et al.* 2005). As per definition, this seminar paper will address MPs and NPs alone as toxicants, and also their interactions with other toxic contaminants.

Microplastics and nanoplastics as environmental toxicants and their effects

Globally, there have been extensive researches conducted on the impacts of macroplastic ingestion on vertebrates, which have reported internal or external abrasions, ulcers and blockages of digestive tract leading to false satiation, poor physical health and starvation. These in turn caused drowning, impaired feeding activity, reduced avoidance from predators, diminished reproduction and ultimate demise. These same consequences may be faced by smaller organisms (e.g. zooplankton and zoobenthos) which ingest MPs (Wright *et al*. 2013b). Digestive system and feeding appendage obstructions, lacerations from sharp objects, inhibition of enzyme production, oxidative stress, reduced feeding inclination (Wright *et al.* 2013a) (table III), dilution of nutrients, diminished growth rate, reduced energy reserves, reproductive failure, low levels of steroid hormones and absorption of toxic pollutants are some of the potential impacts on the marine invertebrates (Wright *et al.* 2013b; Barboza *et al*. 2018). Understanding these impacts requires knowledge about the residence times of the plastic present in the gut (McGoran *et al*. 2017), for longer residence time means energy-intensive digestion (Wright *et al*. 2013a). However, McGoran *et al*. (2017)'s study didn't find any such abrasions or blockage in digestive tracts of fishes examined. No physical damage (Ma *et al*. 2016) and no significant influences on motility and survival (Horton *et al.* 2017) from MPs ingestion were found in Daphnia magna as well.

NPs have more potential to be hazardous as they are likely to have increased interactions with biota including internalisation due to endocytosis or phagocytosis, increased surface reactivity due to higher surface area as well as different kinetics for release of potentially toxic chemical additives (Kokalj *et al*. 2021). NPs may penetrate (Lee *et al*. 2019), or get adsorbed by small organisms (Ma *et al*. 2016), which may cause immobilisation.

Aquatic vegetation

Aquatic macrophytes in freshwater systems are home to a wide variety of periphyton, zooplankton, invertebrates, fish and frogs. They aid in keeping the water clear by weakening wave actions and by diminishing resuspension, thus enhancing the conditions for plant growth. Additionally, nutrient accumulation and removal through uptake and increased denitrification are also attributed to the macrophytes (van Weert *et al*. 2019).

The impacts of MPs and NPs on aquatic vegetation have been inadequately researched (van Weert *et al*. 2019) with much emphasis given on the phytoplankton (Kalčíková *et al*. 2017). Only two studies on a floating plant (duckweed) (Kalčíková *et al.* 2017) and sediment rooted macrophytes (van Weert *et al.* 2019) are known as of the time this seminar paper was prepared (the results shown in table III). Considering the roles played by macrophytes in the functioning of fresh water ecosystems, it is imperative to put more interest in studying the effects of micro- and nanoplastics on the aquatic macrophytes.

Interactions of toxic contaminants with micro- and nanoplastics and their combined impacts

Naturally, MPs and NPs interact with contaminants with varied toxic potential found in the aquatic environment, which is considered as a possible additional exposure route of harmful chemicals to aquatic organism. Several experimental studies have been conducted focusing on the encounters of MPs and NPs with environmental persistent, toxic and bioaccumulative chemicals such as heavy metals, polycyclic aromatic hydrocarbons (PAHs), styrenes, polychlorinated biphenyls, pesticides, pharmaceuticals, organohalogens, nanoparticles and other emerging contaminants (Bhagat *et al.* 2020; Lusher *et al.* 2017). Plastic particles may impair their degradation and also of their metabolites, therefore increasing potentials of bioaccumulation and toxicity of these chemicals in the environment (Ma *et al*. 2016). Toxic additives added during plastic production can potentially migrate from the widely present plastics to organisms (Bhagat *et al*. 2020) and get released in gut producing undesirable effects (McGoran *et al.* 2017). Additionally, these chemicals may be released to the environment following degradation and disintegration of the plastics (Bhagat *et al*. 2020). In ecosystems especially estuaries which are influenced by heavy industrialized, urban and agricultural areas, MPs are likely to accumulate environmental pollutants (McGoran *et al*. 2017). NPs with higher surface area have more potential to retain organic toxic chemicals or heavy metals more than MPs, which poses a real hazard if NPs are capable of permeating membranes, crossing cell walls, translocation and residing in epithelial tissues for longer times, posing an 'unforseen risks' for the organisms in contact with NPs (Koelmans *et al*. 2015). Nanoscale additives such as engineered carbon nanotubes used to improve the polymer durability better may also be released during nanofragmentation process contributing to the overall risk associated with NPs (Koelmans *et al*. 2015). The comprehensive reports, prepared by Bhagat *et al.* (2020) and Alimi *et al*. (2018) show experimental studies done on the interactions of MPs and NPs with environmental contaminants.

Heavy metals

Heavy metals from industrial waste and fuel combustion, and in antifouling paints, and have contributed to metal pollution in aquatic environments, especially prevailing within marinas and harbours (Brennecke *et al*. 2016). As they are known to affect the cellular systems of the contaminated organisms (Alexandre, 2017), the heavy metals association with MPs and NPs is a matter of concern. Microplastic adsorption of heavy metals facilitated by direct adsorption of cations onto charged sites or neutral regions of plastic surfaces have been supported by various studies such as Brennecke *et al.* (2016)'s, Ashton *et al.* (2010')s and Ahechti *et al*. (2020)'s. Concentrations of Cu and Zn 800 times more near the particles than in the seawater were found (Brennecke *et al.* 2016), with the adsorption behaviour depending on the type of metal, exposure time, salinity and pH (Ahechti *et al.* 2020). High salinity and pH means high adsorption of particles (Ahechti *et al.* 2020). NP affinity for heavy metals depends on their pH and redox potential and can be a load of heavy metals by forming precipitation and complexation with them, thereby increasing their potential to cause harm (Singh *et al.* 2019). There have been more studies done on the co-transport of MPs and heavy metals, and their effects on organisms than on the interactions between NPs and metals. In table III, some of the studies and their findings have been shown.

Persistent organic pollutants (POPs)

Due to wide usage, long-range transport, and persistence of the organic pollutants (POPs) such as polychlorinated biphenyls (PCBs) and organochlorine pesticides (e.g. DDT), they are found abundantly in aquatic systems (Ogata *et al.* 2009). POPs such as PCBS are toxic congeners, which have high affinity for Ah receptor and hence display dioxin-like toxicity (Velzeboer *et al.* 2014). Plastic pellets or debris are found to sorb these hydrophobic pollutants from the adjacent sea water (Mato *et al*. 2001; Rios *et al*. 2007) with concentration factors of up to 10^6 (Ogata *et al*. 2009). MPs and NPs both have potentials to influence the transport, uptakes and toxicity of the associated persistent organic pollutants. For example- in Ma *et al*. (2016)'s study on Daphnia magna, the uptake of nanoplastics caused enhanced bioaccumulation of phenanthrene-derived residues in daphni's body. Phenanthrene is a polycyclic hydrocarbon, which possess serious carcinogenic and mutagenic toxicity to organism (Ma *et al.* 2016). Impacts of some of the studies conducted on the MPs and NPs' sorption of POPs and their potentiation in mediat-

Table III. Ecotoxicological impacts of micro- and nanoplastics on aquatic organisms

Organisms	Toxic agents	Study Findings	Study
European Sea Bass *Dicenthrarchus labrax*	MP accumulation of mercury	Oxidative stress in gills and liver	Barboza *et al*. 2018
Shrimps *D.sargus* juveniles	Virgin MPs, and Cu and Zn	Oxidative stress in muscle by MPs alone, and in muscles, gills, and livers by MPs contaminated with Cu and Zn	Alexandre, 2017
Freshwater algae *Chlamydomonas reinhardtii* and *Ochromonas danica.*	Polystyrene nanoplastics(PsNPs) and silver nanoparticles (AgNPs)	PsNPs alone and as well as with the uptake of AgPs inhibited the growth of both algal species	Huang *et al*. 2019
Zebra Fish Embryo	Differently sized PS nanoplastic alone and in combination with Au ion	Accumulation of smallest PS in the whole embryo especially in yolk lipids; subcellular damage especially mitochondria. PS +AU combination have more severe effects on hatching rate, development and survival than PS alone.	Lee *et al*. 2019
	500nm and 20 nm PS alone and in combination with phenanthrene	Increased phenanthrene level in embryos from co-exposure of 20nm; delayed hatching and increased EROD activity.	Zhang and Goss, 2020
Japanese medaka	MP uptake of persistent and bioaccumulative toxic PAHs, PCBs, and PBDE's	Induced liver stress; 74% of fish exhibited. Glycogen depletion	Rochman *et al.* 2013
Lugworm *Arenicola marina*	Polyvinly chloride	Suppressed feeding activity and reduction in energy reserves; Decreased bioturbation in sediments.	Wright *et al.* 2013a
	Uptake of MP with nonylphenol and phenanthrene pollutants and Triclosan and PBDE-47 additives	Transfer of Triclosan into tissues induced greater than 55% mortality; Nonylphenol reduced ability to remove pathogenic bacteria; Microplastic alone caused oxidative stress.	Browne *et al.* 2013
Algae (*C. pyrenoidosa***)**	Nanoplastics	Reduced growth and photosynthetic activity, and induction of oxidative stress	Yang *et al*. 2021
Daphnia Magna	50 nm Nanoplastics	Induced immobilization and physical damage due to adhesion to thoracopods, essential for swimming. Possibly caused by toxic impurities from NPs (e.g. styrene)	Ma *et al*. 2016
Freshwater green algae *(Scenedesmus subspicatus)* Marine diatom *(Thalassiosira weissiflogii)* Freshwater bivalve *Corbicula fluminea*	Nano-sized polyethylene particles	Inhibition of growth for both algae especially at high concentrations ; No effect on filtering ability but increased pseudo-faeces as defence mechanism	Baudrimont *et al.* 2020
Macrophytes *Myriophyllum spicatum and Elodea sp*	Polystyrene micro- and nanoplastics	Overall reductions in root and shoot dry weights and root –shoot ratios but particularly at an unrealistic concentrations of MPs and NPs.	van Weert *et al.* 2019
Duckweed Lemna minor	Polyethylene microbeads	Reduction in root growth	Kalčíková *et al*. 2017

ing their toxicity are given in table III. Nanoplastics with their higher surface area have more affinity for PCBs than the microplastics. If MPs are rapidly released from the body, they cannot have adequate exposure bodies of aquatic organisms, whereas PCBs transfer from NPs to biota lipid is highly likely (Velzeboer *et al*. 2014).

Conclusion

This paper finds most of the experimental studies demonstrating negative impacts of MPs and NPs on the subject organisms, with some even finding mortality, cellular damages and presence of NPs in yolk lipids. However, most of these studies have been carried out at unrealistically high concentrations of MPs and NPs, under controlled laboratory conditions, deliberately contaminating the MPs and NPs with co-pollutants and across a limited range of aquatic organisms, with only two studies conducted on aquatic macrophytes as to my knowledge. Moreover, there has been far less information on the freshwater organism as much as there is for the marine fauna. Therefore, in overall, existing methods for detection, sampling and identification of MPs and NPs must be more refined, and more experiments needs to be undertaken of a wide range of organisms (both marine and freshwater) in the existing environmental conditions with realistic concentrations of MPs and NPs. To create more awareness of the MPs and NPs in public domains, greater understandings of their fate and behaviour mechanisms across time and space and of their potentials of toxicity alone and also with other environmental contaminants on exposure to aquatic organisms are vital. The micro- and nanoplastic pollution and its apparent consequence on the aesthetics, environmental repercussions and economic outcomes are not limited to just individual countries (da Costa *et al.* 2018), in fact, it has become a global concern that needs utmost attention in the arena of plastic pollution. The research lacks comprehensive data on the impacts of MPs and NPs in freshwater ecosystems and their interactions with diverse organisms under realistic environmental conditions (da Costa *et al.* 2018). Future research should prioritize refining detection and sampling methodologies, conducting field studies with environmentally relevant concentrations, and exploring the long-term ecological and human health consequences of MPs and NPs in both marine and freshwater systems.

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