Microbial Ecosystem and Anthropogenic Impacts 1

Lalita Vithal Baragi, Dhiraj Dhondiram Narale,
Sangeeta Mahableshwar Naik, and K. M. Rajaneesh

Abstract

Oceans are the most vulnerable sites for anthropogenic waste from domestic as well as industrial origin. Usually, marine ecosystems are exposed to most anthropogenic stressors ranging from sewage disposal to nuclear waste contaminants. Most recent threats to marine ecosystems are ocean warming and ocean acidification (related to anthropogenic emission of CO_2), oil (tarball), and (micro) plastic contamination, which is proved to have a devastating impact on the marine ecosystem. Microbes are abundantly present in marine ecosystems playing essential roles in ecosystem productivity and biogeochemistry. Generally, microbial communities are the initial responders of these stressors. Altered microbial communities in response to these stressors can, in turn, have adverse impact on the marine ecosystem and later on humans. In this review, we highlight the effect of oil pollution, microplastics, and increased CO_2 on the marine microbial ecosystem. The information on the impacts of such stressors on microbial communities will be valuable to formulate appropriate remediation approaches for future use.

L. V. Baragi (✉)
Applied and Environmental Biotechnology Laboratory, Department of Biological Sciences, BITS, Pilani, KK Birla Goa Campus, Sancoale, Goa, India
e-mail: 2017proj21@goa.bits-pilani.ac.in

D. D. Narale
CSIR-National Institute of Oceanography, Dona Paula, Goa, India

S. M. Naik
National Center for Coastal Research, Government of India, Ministry of Earth Science, Chennai, India

K. M. Rajaneesh
King Fahd University of Petroleum and Minerals, Dhahran, Saudi Arabia

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1.1 Introduction

It is well known that two-thirds of planet Earth is covered by marine waters (Charette and Smith 2010). These waters have a significant role in the global biogeochemical cycles, which sustains the life in the ocean (Schlesinger 1997; Sarmiento and Gruber 2006). Even though the standing crop of marine ecosystems represents 1% of the terrestrial biomass, it contributes to approximately half of the biomass produced on Earth (Gruber et al. 2009; Hader et al. 2011). Among the marine ecosystems, one-half of global primary production occurs in the oceans (Falkowski et al. 1998; Field et al. 1998). Oceans also control the climate and weather pattern. Therefore, the ocean has a significant effect on the biosphere and much of life on Earth.

In recent years, exploitation of the ocean by human-derived activities such as fishing, tourism, oil exploration, maritime transport, and industrial activities has a substantial impact on the marine ecosystem (Nogales et al. 2011; Halpern et al. 2007, 2008). These activities affect different trophic levels of marine food web, which comprises microorganisms to animal predators. According to a recent study, the vast region of the world ocean is forecasted to have medium to high impact from these stressors (Halpern et al. 2008). Most recent stressors to these ecosystems are ocean warming and ocean acidification and oil (tarball) and microplastic contamination, which are proved to have a devastating impact on the marine ecosystem, mainly biology of the ecosystem.

Ocean warming and ocean acidification are the result of increasing concentration of atmospheric CO_2 (Caldeira and Wickett 2003; Orr et al. 2005) and are now being recognized as a major responsible factor for change in the biological system of the oceans (Lovejoy and Hannah 2005). Presently, the concentration of atmospheric CO_2 has reached 400 ppm from 280 ppm from preindustrial revolution (NOAA/ESRL; Stocker et al. 2013) with ~0.5% year^{-1} rising rate (Forster et al. 2007). Approximately one-third of the anthropogenic CO_2 generated is absorbed by the oceans and will help moderate future climate change (Sabine et al. 2004). This resulted in a decrease in pH by 0.1 unit (referred to as ocean acidification) and rise in temperature by 0.85 °C (referred to as ocean warming) (Raven et al. 2005). By the end of this century, the concentration of atmospheric CO_2 is predicted to reach 800–1000 μatm by the “business as usual” CO_2 emission scenario climate models, which will further decrease the pH (0.3–0.4 units) and increase the temperature (1–4 °C) (Stocker et al. 2013; Caldeira and Wickett 2003). Elevated CO_2 concentration will result in an increase in H^+ concentration (100–150%), which will negatively affect the marine organisms, especially calcifying organisms (Haugan and Drange 1996; Brewer 1997).

Oil pollution is another threat to the marine environment presently. Oil spills mainly arise from either accidents or oily discharges from ships (Solberg 2012). Operational discharges from tankers cause the majority of the oil pollution cases. The effect of the spills on the marine ecosystem depends on several factors such as the quantity and quality of spilled oil, the sensitivity of the organisms exposed to the oil, location, depth, season, and meteorological and oceanic conditions (Fukuyama et al. 1998). These oil spills mainly have negative consequences on the ecology of the marine ecosystem (Fukuyama et al. 1998).

To some extent, oceans are also used as dumping sites for debris from human activities. Marine debris comprised of manufactured solid material, of which 60–80% consists of plastic (Gregory and Ryan 1997). According to a study, an estimated 1.3 plastic items can be found for every m^2 of shoreline worldwide, which is believed to be a significant threat to the marine ecosystem (Bravo et al. 2009). More than 267 species worldwide are impacted by this debris either by ingestion or entanglement (Gall and Thompson 2015). Plastics having size range between 333 μm and 5 mm are called microplastics. Smaller particles (<1 μm) also exist in marine waters but are less often detected (Arthur et al. 2009). The most commonly found micro-debris particles include polyethylene, polypropylene, and polystyrene (Andrady 2011). These are ubiquitous in marine environments. However, the harmful effects of microplastic on the marine food web are less known (Derraik 2002). Possible threats to the marine food web may include physical harm from ingestion, leaching of toxic additives, and desorption of persistent, bioaccumulative, and toxic chemicals (Nobre et al. 2015). Current literature revealed that some planktons and many classes of invertebrates and vertebrates are known to ingest and accumulate microplastics, as the size of microplastic falls in the same size range as their natural food (Wright et al. 2013).

Ocean warming, ocean acidification, oil (tarball) pollution and microplastic contamination have instant and long-lasting effects on marine organisms, from basic to complex life forms and from the cellular to the community levels. Keeping in mind the extent and the tenacity of the impact from these threats, workable and effective methods for remediation are essential. This chapter delivers a synopsis of the cause of the three key recent threats to marine environments, ecological processes, and transformation in the marine food web. Prospects for possible solutions are also discussed.

1.2 Anthropogenic Impact on Marine Organisms

1.2.1 Rising Atmospheric Carbon Dioxide: Ocean Acidification and Warming

Since the industrial revolution, due to rapid industrialization, the carbon dioxide concentration (pCO_2) has increased from 280 ppm to the present level of 400 ppm (NOAA/ESRL; Stocker et al. 2013). This has resulted in 0.1 unit reduction in pH (referred to as ocean acidification) and 0.85 °C rise in seawater temperature (referred to as ocean warming) (Raven et al. 2005). By the end of this century, the concentration of atmospheric CO_2 is predicted to reach 800–1000 μatm by the "business as usual" CO_2 emission scenario climate models, which will further decrease the pH (0.3–0.4 units) and increase the temperature (1–4 °C) (Stocker et al. 2013; Caldeira and Wickett 2003). Significant warming and acidification incidents have occurred in the earth's history, resulting in considerable changes in marine communities (Pelejero et al. 2010; Hönisch et al. 2012) and numerous mass extinctions (Pelejero et al. 2010; Clarkson et al. 2015). Though these acidification and warming incidents are not the same as present as their pace was much slower compared to the present (Pelejero et al. 2010). Thus, based on past histories, we cannot forecast the impacts of acidification and warming on marine organisms, and hence in recent years, there is increasing research in this area.

In marine organisms, ocean acidification affects their metabolism, acid-base balance, and calcification (Chan et al. 2012; Pörtner et al. 2004; Pörtner 2008; Ries et al. 2009; Nilsson et al. 2012; Andersson and Gledhill 2013; Lane et al. 2013; Mostofa 2016). The marine organisms show species-specific (Hendriks et al. 2010; Kroeker et al. 2010, 2013; Harvey et al. 2013) and habitat-specific response (Andersson et al. 2008; Clark et al. 2009) to ocean acidification. Bacterial community structure and diversity is also known to change in acidified conditions (Allgaier et al. 2008; Kerfahi et al. 2014; Witt et al. 2011; Lidbury et al. 2012). Phytoplankton may either benefit from rising pCO_2 or be affected by the related reduction in pH depending on species (Gao and Campbell 2014; Torstensson et al. 2015). In phytoplankton, acidification influences their growth, energy allocation, photosynthesis, calcification, carbon acquisition, cellular fluxes, particulate carbon production, elemental composition, and biochemical composition (Rost et al. 2006; Rickaby et al. 2010; Sett et al. 2014; Bautista-Chamizo et al. 2016; Jin and Gao 2016; Kottmeier et al. 2016). Phytoplankton shows species-specific response to acidification such as positive, neutral, and negative (Gao et al. 2012; Johnson et al. 2013; Baragi and Anil 2016; Jin and Gao 2016). Such variation in response might be due to variation in carbon-concentrating mechanism (CCM), which uses high metabolic energy to transport and convert HCO_3^- into CO_2 around RuBisCO in the cell under CO_2-limited condition (Raven et al. 2008; Reinfelder 2011). Under the acidified condition, the elevated CO_2 downregulates CCM, thus reducing the energy demand of the cell (Burkhardt et al. 2001; Beardall and Raven 2004; Spijkerman 2008; Holtz et al. 2015; Wu et al. 2015).

A meta-analysis reported the effect of moderate acidified condition (936 μatm) on all animal taxa (corals, echinoderms, mollusks, crustaceans, fishes) with more significant impact on those having weak acid-base regulation abilities and calcified structures (corals, echinoderms, and mollusks); however, crustaceans are relatively unaffected (Wittmann and Pörtner 2013). Compared to non-calcifying organisms, calcifying organisms are highly susceptible to acidification (Hendriks et al. 2010; Kroeker et al. 2010; Byrne and Przeslawski 2013); however, few calcifying organisms are robust to acidification either due to their adaptive capacity to naturally acidic habitats (Talmage and Gobler 2011) or due to the higher buffering ability of local waters (Range et al. 2012). Recently, it has been observed that ocean acidification affects some non-calcifying organisms (Wage et al. 2016; Borges et al. 2018). Invertebrates are capable of tolerating acidification by investing more energy in compensatory mechanisms like maintenance of acid-base homeostasis rather than basic mechanisms like growth and reproduction (Pörtner 2008; Melzner et al. 2009; Kroeker et al. 2013; Xu et al. 2016).

On the other hand, warming significantly affects growth and metabolism of marine organisms (Eppley 1972; Brown et al. 2004), consequently changing their abundance and distribution (Thomas et al. 2004; Harley et al. 2006). Organisms show species-specific response to temperature and rely on the organism's thermal tolerance window and its capability to adapt to varying temperature. Beyond these limits, the rising temperature can impose physiological stress and a decline in biochemical and metabolic processes such as development and growth in the organisms (Pörtner and Farrell 2008; Poloczanska et al. 2014). Under warming situations, organisms might display poleward migrations as organisms generally adapt to warming by shifting to regions with optimal temperature (Nguyen et al. 2012; Kamya et al. 2014; Poloczanska et al. 2014). Moreover, warming is also predicted to have severe negative impacts on tropical species than temperate as the former species have already comparatively small thermal windows and are usually living near their maximum thermal limit (Stillman and Paganini 2015). Phytoplankton shows enhanced photosynthesis, growth, calcification, and reduction in size in response to warming (De Bodt et al. 2010; Müller et al. 2014; Sett et al. 2014). In natural plankton community, warming increased phytoplankton abundance (Lewandowska et al. 2014) and changed the distribution of phytoplankton groups (Thomas et al. 2012). Warming is the significant reason for the speedy reduction (~1% yearly) of phytoplankton biomass globally (Boyce et al. 2010), with diatoms being the significantly affected group (Toseland et al. 2013). Further, warming is forecasted to cause a reduction in tropical phytoplankton diversity and a poleward shift in the species' thermal niches (Thomas et al. 2012).

In the future climatic scenario, both acidification and warming are known to co-occur. Some studies reported positive synergistic effect of these stressors on some species of phytoplankton wherein it caused an increase in growth and repair rate of UV-damaged PSII machinery of microalgae (Connell and Russell 2010; Fiorini et al. 2011; Li et al. 2012). However, other studies reported the insignificant synergistic effect of these stressors on cyanobacteria (Fu et al. 2007; Hutchins et al. 2007) and coccolithophores (De Bodt et al. 2010). Some microalgal species showed temperature-dependent response to acidification, wherein the optimum temperature for the growth, carbon fixation, and calcification increased under elevated pCO_2 in contrast to ambient pCO_2 concentration (Sett et al. 2014). Our understanding of the effect of acidification and warming is limited only to single species of microalgae. However, few studies have been carried out to know the response of the natural microalgal community to these stressors (Kim et al. 2006; Calbet et al. 2014; Sommer et al. 2015). Moreover, very limited snapshots are available for prolonged adaptation of microalgae to acidification (Lohbeck et al. 2012; Jin et al. 2013; Low-Décarie et al. 2013; Jin and Gao 2016). Marine invertebrates show species-specific response to the synergistic effect of acidification and warming. Acidification is known to shorten the thermal window of some species, thus making them highly susceptible to warming (Schalkhausser et al. 2013). However, in other species, there was no such effect observed (Zittier et al. 2015).

In invertebrates, compared to adults, the larval and initial life phases of invertebrates are severely susceptible to acidification and warming (Dupont et al. 2010a; Kroeker et al. 2013). Thus, it is crucial to emphasize on the effects of altering environmental conditions on larval phases as they signify a bottleneck for populations that try to tolerate the altering environmental conditions (Havenhand et al. 2008; Dupont et al. 2010b). Any influence on larval development and growth will show a high impact on population. The larval sensitivity to these stressors differs with individuals and taxa because of the variation in the maternal nutritional and energetic investment and history (Byrne et al. 2009; Przeslawski and Webb 2009; Donelson et al. 2012). Some studies have observed significant "carryover" effects from one life stage to another (transgenerational effect) (Kurihara 2008; Putnam and Gates 2015; Manno et al. 2016; Borges et al. 2018). For example, Parker et al. (2015) reported that larvae and juveniles of barnacle were able to survive better under acidified conditions due to positive carryover effects from an adult. Thus, it is essential to investigate the impact of these stressors on different life stages through parental and transgenerational effects.

Acidification and warming may indirectly drive ecological change through biotic interactions, which are the crucial "pressure point" (Gaylord et al. 2015). Studies also discovered that most remarkable impacts of acidification and warming would arise through changed species interactions (Rossoll et al. 2012; De Kluijver et al. 2013; Poore et al. 2013; Kroeker et al. 2014). Recently, it is observed that food supply alleviates the negative effects of these stressors on marine invertebrates (Melzner et al. 2011; Thomsen et al. 2013; Asnaghi et al. 2014; Pansch et al. 2014; Uthicke et al. 2015, 2016; Ramajo et al. 2016).

1.2.2 Oil (Petroleum/Tarball) Pollution

Petroleum is one of the common contaminants in the aquatic environment. As a consequence of rising global demand for energy, there is increased crude oil exploration and transportation in the marine environment, thus making them vulnerable to crude oil pollution (National Research Council 2003). Oil spill as a result of accidents or discharge of ballast waters is a common occurrence nowadays. On the contrary to the common belief, even the small oil spills and their repetitive nature can have an immediate adverse biological impact on marine biota, thereby affecting the marine ecosystem functioning (Brussaard et al. 2016). In addition to the tanker-derived oil pollution, land-derived inputs due to urbanization and industrialization coupled with domestic petroleum production also contribute to coastal oil pollution (Zakaria et al. 2000). The compounds such as isoprenoid alkanes, steranes, hopanes, and polycyclic aromatic hydrocarbons (PAH) have been proposed as the molecular compounds or biomarker to identify the sources of oil pollution (Zakaria et al. 2000).

Of all the marine life forms, planktonic organisms are prone to oil spill contamination. Planktons are on the mercy of currents and hence cannot avoid the crude oil areas, compelling them into the polluted waters and causing unplanned encounters with the polluted regions. The crude oil effect on marine species has gathered attention from ages; however, mostly higher organisms have been in the limelight compared to the marine microbes. Phytoplankton plays a vital role in biogeochemistry of the marine ecosystem, and hence any alterations in the ecosystem can have a profound effect on the food web dynamics. PAHs, a significant fraction of crude oil, are primarily responsible for crude oil toxicity in phytoplankton (Ozhan and Bargu 2014). They even accumulate in the sediment, posing a severe threat to the benthic community (e.g., Ozhan and Bargu 2014). Oil pollution can result in acute and chronic effects of phytoplankton productivity and community composition which can alter the whole planktonic ecosystem. Therefore, any alterations due to contaminants can result in ecosystem alterations (Othman et al. 2018), while some work has suggested a positive impact of crude oil on water chemistry which has enhanced the phytoplankton biomass. Most studies have shown that it has negatively altered the growth of phytoplankton. This impact is also determined by the concentration of crude oil exposed to the phytoplankton (Huang et al. 2011). Several factors influence the harmfulness of crude oil to phytoplankton, and it is not clear. Echeveste et al. (2011) found that the phytoplankton cellular size was key factor in determining the susceptibility to PAHs, with the pico-sized group showing the synergistic relationship between PAHs and UV radiation. Temperature is also another critical factor influencing the toxicity of crude oil in phytoplankton as demonstrated by Huang et al. (2011). They showed that *Skeletonema costatum* was highly tolerant to water-accommodated fraction (WAF) during winter but in summer even low concentration of WAF restricted their growth. They attributed this to the increased metabolic rate due to increased temperature resulting in more excellent absorption on toxicants.

Zooplankton is also the key player in marine food web dynamics, biogeochemical processes, and fish population dynamics (Banse 1995; Castonguay et al. 2008; Alcaraz et al. 2010). The impact of crude oil on zooplankton depends on several factors such as type of species, life stages, size, oil concentration, chemical dispersant, exposure time, temperature, salinity, UV radiation, etc. (Almeda et al. 2013a, b). The impact of oil on the zooplankton can also be reduced or counteracted by the presence of another organism which is important in the fate or degradation of crude oil in the marine environment (Almeda et al. 2013b). The effect of hydrocarbons on marine includes changes in feeding behavior, growth, and reproduction (Almeda et al. 2013a, b).

1.2.3 Plastic Pollution

In the ocean, plastic debris is ubiquitous and abundantly reported in natural habits. Plastic waste in the oceans was firstly reported in the 1970s (Carpenter et al. 1972; Carpenter and Smith 1972; Colton and Knapp 1974); afterward, they had got little public attention. Over a period, the ocean contains over 150 million metric tons of plastic (MacArthur et al. 2016). About 8 million metric tons of plastics waste enters marine environment, annually (Jambeck et al. 2015; Science Daily, 12 February 2015). By 2050, there will be excess plastic (based on weight) in the oceans compared to fish (MacArthur et al. 2016). Depending upon polymers used, plastic material can persevere up to several years (2–450 years) (Kibria 2018). Thus, plastic pollution is persistent in the oceans and has adverse ecological effects. Accumulation of marine plastic litter over a period has openly threatened marine biota.

In nature, plastic litter gets fragmented by UV radiation, hydrolysis, oxidation physical abrasion, and/or biodegradation into micro- or nanoscopic particles. Based on the size, they are referred to as nano- (<1 μm) and microplastics (1–5 mm) (Germanov et al. 2018). However, in most scientific articles, particles smaller than 5 mm are referred to as microplastics (Arthur et al. 2009; Andrady 2011; Hidalgo-Ruz et al. 2012). Depending on the source of origin, these fragmented plastics are called as secondary microplastics (Andrady 2011; Mrowiec 2017; Germanov et al. 2018). The primary microplastics arise from makeup products, dyes, fabrics, and waste from plastic industry (Mrowiec 2017; Germanov et al. 2018). A particular concern over microplastic debris over large size plastic litter is that they remained under-investigated due to their non-visibility to the naked eye. Incidences on the entanglement or ingestion of plastic material by marine species are extensively documented worldwide (Laist 1997; Clapham et al. 1999; Mascarenhas et al. 2004). Overall, the accumulation of large plastics remains stable or disappears, whereas that of microplastic rises (Eriksen et al. 2014). Scientific investigations reported the devastating effect of microplastics at higher trophic organisms. However, the possible impact of microplastics is under-evaluated on marine microorganisms, which are the foundation of marine food web. Considering this fact, the present section is restricted toward only the influencing mechanisms of microplastics on diverse marine microorganisms.

Plastics are incredibly resistant to biodegradation because of their high molecular weight and hydrophobicity. However, some microbial species have the ability to biodegrade the plastic material (Sivan et al. 2006; Shah et al. 2008; Mor and Sivan 2008; Harshvardhan and Jha 2013). The biodegradation process is generally initiated by surface assimilation of organic molecules on the microplastic, which further supports bacterial colonization (biofilm formation) (Gilan et al. 2004; Mor and Sivan 2008; Balasubramanian et al. 2010). Scanty information is available on the exact mechanisms involved in the biodegradation. Possibly biodegradation could be due to the interplay of various oxidative mechanisms caused by the microorganisms alone or in combination with the atmospheric oxygen, and the mechanisms would be complicated (Glass and Swift 1989). However, studies have proved that biodegradation process may reduce the weight of plastic molecules (Harshvardhan and Jha 2013) and surface hydrophobicity (Gilan et al. 2004).

The microplastic-associated microbial community is remarkably different from those present in the adjacent water (Ogonowski et al. 2018; Parrish and Fahrenfeld 2019). In water, the microbial population and metabolic processes depend on the quality of dissolved organic matter (Ruiz-González et al. 2015; Pernthaler 2017).

The microplastic-derived dissolved organic carbon attracts suitable microbial population, which could differ from the natural one. This difference in the utilization of carbon substances could partially explain the profile difference between the microplastic microbes and microbes present in adjacent water (Arias-Andres et al. 2018). Microplastics are known to release carbon thus, can affect the other free-living bacterial population by adding to dissolved organic carbon pool (Arias-Andres et al. 2018) and particulate organic matter bioavailability (Zhang et al. 2016). Considering the additional volume of anthropogenic dissolved organic carbon escaping from the tons of plastic per year (up to 23,600 metric tons from 4.8 to 12.7 million tons of plastics in 2010) entering marine environment (Romera-Castillo et al. 2018) could influence the natural carbon cycle in the ocean.

The genetic diversity within the microplastic biofilms increases gene transfer, which can affect the metabolic diversity of different microorganisms (Fazey and Ryan 2016; Rummel et al. 2017; Arias-Andres et al. 2018). This could further modulate various biochemical functions (Flemming et al. 2016) and bioadsorption capabilities of different toxic chemical and metal molecules on microorganisms. It is, therefore, the microbial biofilm formation on the consistently accumulating microplastics in the environment that can not only influence the fate of microplastic itself (Rummel et al. 2017) but also their impacts on the working of whole microbiomes (Arias-Andres et al. 2018). Ecologically also, plastic debris has an essential role in the spread of invasive, harmful microorganisms and algae species (Maso et al. 2003; Bryant et al. 2016; NOAA Marine Debris Program 2017) and thus changes in microbial biogeography. Biofilms established on microplastic attract other fouling organisms (biofouling), which do enhance microplastic sinking (Kaiser et al. 2017). The developed biofilm increases the sinking velocity of negatively buoyant microplastics, whereas macrofouling causes positively buoyant microplastics to sink (Kaiser et al. 2017). Further sinking in marine aggregates could further impact other planktonic and benthic feeding organisms (Long et al. 2015).

Plastic can concentrate contaminants such as persistent organic pollutants and heavy metals and increase their concentration up to 10^6 order (Mato et al. 2001). Moreover, plastic can act as vectors of contaminants which are already present in the waters, thus increasing their harmfulness to marine species (Bejgarn et al. 2015). Enormous literature is available on the effect of nanoparticles together with contaminants on higher-trophic-level organisms. However, the datasets on the toxicological effect of nano- or microplastic particles on common marine microbes (bacterial, phytoplankton) are limited. These studies are mostly based on synthetic microplastic particles under laboratory conditions. Hydrophobic contaminants such as persistent organic pollutants (POPs) get readily accumulated on the surface of plastics (Avio et al. 2017; Bhattacharya et al. 2010). The degraded or aged microplastics have uneven outer layers, which further promote POP adsorption (Cole et al. 2011). However, contradictorily, recent laboratory studies revealed microplastic surfaces modulate the toxicity of pollutants and made them less available to the microorganisms (Garrido et al. 2019; Yi et al. 2019). A recent study observed that polyethylene microplastics alone do not have any inhibitory effect on

Isochrysis galbana, whereas the negative effect was observed upon exposure to chlorpyrifos (CPF) in lethal concentration (2–3 mg L^{-1}) (Garrido et al. 2019). However, upon following incubation, chlorpyrifos gets adsorbed on the microplastic surface and not much available for microalgae. Similarly, Yi et al. (2019) observed adsorption of triphenyltin chloride (TPTCl) on polystyrene surface control bioavailability and toxicity of TPTCl to green algae.

Under laboratory conditions, microplastic toxicity exhibits conflicting results mainly due to confounding factors of microplastic size and dosages (Long et al. 2017). In *Chlorella pyrenoidosa*, upon exposure to polystyrene microplastics, dose-dependent adverse effect was observed at initial growth phases. Initially, in *C. pyrenoidosa*, reduced photosynthetic activity, indistinct pyrenoids, and impaired cell membranes were detected. Later, cellular wall thickening together with homo- and hetero-aggregation triggers cell growth and algal photosynthesis (Mao et al. 2018). Likewise, a study by Sjollema et al. (2016) observed no effect of polystyrene microplastics on photosynthetic activity but had effect on the growth rate of marine flagellate *Dunaliella tertiolecta*, only at high exposure level (250 mg L^{-1}) with smaller particle size (0.05 mm). However, in marine diatom *Phaeodactylum tricornutum*, the polystyrene nanoparticles (50 and 100 nm) possibly have effects at different physiological and cellular levels (Sendra et al. 2019). At higher concentrations (50 mg L^{-1}), smallest (50 nm) nanoparticles significantly damage the photosynthetic apparatus; damage DNA, depolarization of mitochondria, and cell membrane; and later inhibit chlorophyll content and population growth. An adverse effect of micro-PVC (~1 mm) was observed on the growth and chlorophyll fluorescence of *Skeletonema costatum* at high level (50 mg L^{-1}); this might have resulted from the obstruction of alveoli and impairment of cell surface (Zhang et al. 2017). Compared to microplastics, nanoplastics might more actively interact with microalgal membrane by covering or obstructing the pores or gas exchange (Bhattacharya et al. 2010). The physiological adaptive strategies of test species also greatly influence the effect of microplastics. Seoane et al. (2019) observed that the marine diatom *Chaetoceros neogracil* modulates the oil body level to overcome the stress produced upon polystyrene microbead exposure.

In conclusion, the marine microbial population could have a diverse impact on increasing anthropogenic plastic pollution. Overall, the "plastisphere" could control the microgeography and microbial diversity in the oceans.

1.3 Concluding Remarks

Human activities have caused marine ecosystems and, in turn, microbial communities to suffer a lot. Anthropogenic stressors like ocean warming and ocean acidification (related to anthropogenic emission of CO_2), oil (tarball) pollution, and (micro) plastic contamination have been proved to have a devastating impact on the marine ecosystem, which can have severe consequences on socioeconomic levels. Available pieces of literature have revealed significant alterations in marine species community dynamics. Based on the available literature, we are yet not confident enough to predict the functioning of the marine ecosystem in the future, mainly if the stressor is either still present or increases. Therefore, to know the effect of these stressors, we need to focus on multidisciplinary approaches/holistic frameworks linking modeling, observations, and experiments including new technologies. Further, there is a need to promote awareness within stakeholders and governments. Thus, with this approach, we may able to take necessary steps to slow or minimize the impacts of these stressors on the marine microbial ecosystem and in turn on human populations.

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