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Review

Pollutant toxicology with respect to microalgae and cyanobacteria

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ABSTRACT

Microalgae and cyanobacteria are fundamental components of aquatic ecosystems. Pollution in aquatic environment is a worldwide problem. Toxicological research on microalgae and cyanobacteria can help to establish a solid foundation for aquatic ecotoxicological assessments. Algae and cyanobacteria occupy a large proportion of the biomass in aquatic environments; thus, their toxicological responses have been investigated extensively. However, the depth of toxic mechanisms and breadth of toxicological investigations need to be improved. While existing pollutants are being discharged into the environment daily, new ones are also being produced continuously. As a result, the phenomenon of water pollution has become unprecedentedly complex. In this review, we summarize the latest findings on five kinds of aquatic pollutants, namely, metals, nanomaterials, pesticides, pharmaceutical and personal care products (PPCPs), and persistent organic pollutants (POPs). Further, we present information on emerging pollutants such as graphene, microplastics, and ionic liquids. Efforts in studying the toxicological effects of pollutants on microalgae and cyanobacteria must be increased in order to better predict the potential risks posed by these materials to aquatic ecosystems as well as human health.

Introduction

Aquatic ecosystems contain numerous aquatic organisms and play critical roles in maintaining nutrient cycling and biological diversity (Leebens-Mack et al., 2019; Seymour et al., 2017). Freshwater and marine ecosystems, the two main types of aquatic environments, help regulate the global distribution of heat and provides a large amount of seafoods for humans (van Hoof et al., 2019). Meanwhile, the freshwater environment is the source of drinking water for humans. Water resource allocation in regions with increasing water demands and declines in water quality and availability is a significant societal challenge. Hence, protecting these environments from pollution, and toxicological evaluations represent an important tool in this regard (Bekturganov et al., 2016). However, owing to the open system of the aquatic environment, it captures all types of pollutants from many sources. For instance, pollutants may flow from the soil into the surface water through surface runoff (Cuevas et al., 2018). Studies have shown that high pesticide concentrations in surface waters often occur after intense rainfall (Rasmussen et al., 2015).

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Therefore, low-frequency sampling may remarkably underestimate the maximum pollutant concentrations and fluxes.

The aquatic environment consists of aquatic organisms and abiotic environments. Aquatic organisms include microorganisms, phytoplankton (algae), higher aquatic plants, invertebrates, and vertebrates, and these organisms carry out many ecological functions (Zhou et al., 2014). Among them, microalgae and cyanobacteria, as important primary producers of aquatic systems, are of fundamental importance to primary production, ecological balance, material circulation, energy flow, nitrogen fixation and pollution degradation, and their sensitivity to pollutants directly leads to overall deteriorations in many ecosystems (Ramakrishnan et al., 2010).

In-depth survey to microalgae and cyanobacteria as their interactions with different pollutants can provide insights in how pollutants change the development of organisms as well as profound impacts on the environment. This is the reason they are frequently used as model species in toxicology studies (Brayner et al., 2011; Dahms et al., 2011; Kumar et al., 2014; Qian et al., 2008). Aquatic environments contain a mix of various manmade pollutants, which can act together on all the living organisms present in the environment. These pollutants can also interact with each other (i.e., with pollutants from the same or different categories) or with abiotic factors (physical or chemical factors), thus resulting in combined stress effects (Vasquez et al., 2014). In addition, owing to the existence of a wide range of species interactions, the toxicological effects in microbial communities are more complex than those on any single species, and it is necessary to study in-depth the combined effects of pollutants and their community-based biological effects (Lu et al., 2019a; Zhang et al., 2019a). Most toxicological studies have analyzed the effects of a single pollutant on a single species. Their reductionist approach cannot fully reflect the toxicities of the pollutants present in the natural system. In addition, the toxicological effects of pollutant mixtures on microbial communities also remain unexplored, as there are too many variables involved in such an ecosystem. Owing to operational constraints (in the case of long-term studies, the microbes present in the aquatic microcosm are not accurately reflected in the natural water environment, and adding pollutants to the natural water environment would result in actual pollution), such studies are hard to perform.

Herein, we review the existing literature on aquatic environmental pollutants while focusing on the sources and fate of these pollutants in the aquatic environment as well as the toxicological effects of several major pollutant types on microalgae and cyanobacteria. We highlight the importance of considering the effects of combined toxicity and microbial community interactions on the biological effects of the pollutants based on toxicological studies.

1. Types, sources, and fates of pollutants in aquatic environments and their toxicity to microalgae and cyanobacteria

According to recent reports, the main pollutants in aquatic environments are broadly classified into the following five categories: metals, nanomaterials (NMs), pesticides, pharmaceuticals and personal care products (PPCPs), and persistent organic pollutants (POPs). The physicochemical properties, sources, fate, and toxicity to microalgae and cyanobacteria of these pollutants are different, which will be discussed in detail in the following sections. Table 1 lists the concentrations of some common pollutants in aquatic environments determined in recent years. Table 2 lists the toxic effects of several common pollutants on different microalgae and cyanobacteria, which were the most studied species, likely because they are the main producers and occupy the largest proportion of biomass in aquatic microbial communities. Fungi, heterotrophic bacteria, and zooplankton have also been investigated extensively in aquatic toxicology. However, it is worth noting that the same pollutant may have different or even opposite effects on various microorganisms, and the different responses of microbes might, sequentially, influence the metabolism of other microbes as well as the fate of the pollutants, thus, in natural aquatic system, the effects of the pollutants on a certain species are difficult to accurately predict.

1.1. *Metals*

Metals are an indispensable component of all living organisms. As key cofactors participating in many cellular processes, several metals in appropriate amounts are essential for various life processes such as cell growth and proliferation (Barnett et al., 2012; Kranzler et al., 2013). However, nonessential metals, such as Cd, Ag, and Hg, to name a few, are toxic to organisms even at low concentrations (Lemire et al., 2013). Metal pollutants in the environment can be divided into metal organic pollutants, soluble metals (i.e., metal ions), and particulate metals.

Heavy metals exhibit a large concentration discrepancy in aquatic environments, which can range from nanograms per liter scale to the micrograms per liter scale (Table 1). The distribution of metal pollutants in aquatic environments is primarily dependent on the presence and types of local industry (such as mining, metallurgy, paint, battery, machinery, hair dye, and electroplating facilities) and urban activities (such as traffic emissions and urban waste) (Guo et al., 2017; Hepburn et al., 2018; Qi et al., 2018; Saleem et al., 2018). The complexation of metal pollutants with organic matter and their enrichment caused by microalgae and cyanobacteria are the two primary fates of these pollutants in aquatic environments. Heavy metals cannot be biodegraded. Adsorption by sediments and other organic matter is usually the ultimate fate for most metals (Hu et al., 2016; Mohammed et al., 2011), even if they may be desorbed under certain environmental conditions and subsequently re-enter the water bodies (Jacobson and Philippe, 2004). In addition, they accumulate in microalgae and cyanobacteria and can be transferred and aggregated at higher trophic levels through the food chain (Espejo et al., 2018; Le and Ngo, 2013). Therefore, microalgae and cyanobacteria have been employed as biological indicators for monitoring the metal pollution of waters (Karina Yew-Hoong et al., 2002; Mahmood et al., 2018; Pinto et al., 2003; Xie et al., 2015).

Metals can replace essential cofactors in enzymes and proteins, leading to abnormal metabolic activities in algae. For example, the replacement of Mg by Cu ions in photosystem II blocks the electron transport chain and inhibits photosynthesis (Thomas et al., 2016). These abnormal metabolic activities trigger the production of excessive reactive oxygen species (ROS), a category of chemically reactive chemical species containing oxygen. This, in turn, severely damages the cellular components (such as lipids, proteins, amino acids, and nucleic acids) and lipid membranes (An et al., 2013; Qian et al., 2011; Zhu et al., 2017).

In addition to metal replacement, several other mechanisms can induce ROS overproduction. For instance, some semiconductor metal oxides, such as ZnO and TiO_2, which develop electron-hole pairs, can cause oxidation-reduction reactions that lead to the ROS generation (Reddy et al., 2017). In addition, some transition metals, such as Fe^{3+} and Cu^{2+}, participate in the Haber–Weiss cycle and Fenton reactions that generate ROS (Stoiber et al., 2013). Metals with no redox capability, such as Cd^{2+}, Pb^{2+}, and Hg^{2+}, can enhance the pro-oxidant

Table 1 – Aquatic pollutants in recent years.

Chemicals	Locations	Environmental concentrations (mean value or range)	References
Metals			
Zn, Ni, Cu, Cr and Pb	Surface water of major Korean streams and rivers	0.055–0.60, 0.04–0.30, 0.01–0.055, 0.01–0.055, 0.0–0.1 mg/L	(Pandey et al., 2019)
Zn, Ni, Cu, Cr, Pb, Hg, As and Cd	Sediments of major Korean streams and rivers	1200–1400, 8–55, 45–110, 0–2, 12–22, 0–1.5, 0–2, 0–0.70 mg/kg	
Cd, Cr, Cu, Ni, Pb and Zn	Surface water of Dongting Lake, China	0.05, 1.46, 0.92, 0.95, 0.18 and 3.31 μg/L	(Fang et al., 2015)
	Deep water of Dongting Lake, China	0.07, 2.55, 1.04, 1.20, 0.30 and 4.73 μg/L	
As, Cd, Cr, Cu, Fe, Ni, Pb and Zn	Suiface water of Kralkızı Dam Reservoir in Tigris River basin, Turkey	2.39, 0.036, 22.06, 2.83, 58.63, 15.75, 2.56 and 5.02 μg/L	(Memet, 2013)
	Suiface water of Dicle Dam Reservoir in Tigris River basin, Turkey	1.61, 0.030, 18.58, 2.12, 62.07, 15.86, 1.84 and 4.12 μg/L	
	Suiface water of Batman Dam Reservoir in Tigris River basin, Turkey	0.71, 0.044, 16.50, nd, 57.66, 15.96, 1.56 and 4.09 μg/L	
Cd, Cr, Cu, Fe, Mn, Ni, Pb, Zn and As	Overlying water of Daya Bay, South China	0.021, 4.43, 1.98, 572.23, 1308.72, 0.37, 2.09, 3.71 and 3.93 μg/L	(Ni et al., 2017)
	Pore water of Daya Bay, South China	0.100, 3.13, 10.98, 148.17, 253.62, 5.37, 1.35, 21.46 and 15.55 μg/L	
Cu, Zn, Cr, Pb and Ni	Surface pore water of Northern bays in Taihu Lake, China	23.4, 42.5, 18.8, 13.5 and 12.0 μg/L	(Lei et al., 2016b)
	Overlying water of Northern bays in Taihu Lake, China	9.88, 40.9, 11.7, 10.3 and 8.86 μg/L	
Nano materials			
TiO_2, ZnO, Ag, carbon nanotube, fullerenes	Surface water in European Union	0.53, 0.09, 0.66, 0.23 and 0.11 μg/L (predicted)	(Tian et al., 2014)
	Effluents of sewage treatment plants in European Union	16, 2.3, 0.17, 4 and 1.7 μg/L (predicted)	
	Surface water in Switzerland	0.67, 0.12, 0.45, 0.35 and 0.12 μg/L (predicted)	
	Effluents of sewage treatment plants in European Union	32, 5.3, 0.32, 5.5 and 3.4 μg/L (predicted)	
SiO_2, iron oxides, Al_2O_3, CeO_2, Quantum dots	Freshwater in Northern Europe in 2020	562, 12.8, 39.6, 268 and 32.8 ng/L (predicted)	(Wang and Nowack, 2018)
	Freshwater in Southeastern Europe in 2020	2600, 44.2, 221, 1130 and 107 ng/L (predicted)	
Ag and CeO_2	Surface water of River Ijssel in Netherlands	0.3-2.5 ng/L and 1.7-5.2 ng/L (detected)	(Peters et al., 2018)
	Surface water of River Meuse in Netherlands	0.3-1.1 ng/L and 0.4-5.1 ng/L (detected)	
Pesticides			
Molinate, propachlor, trifluralin, atrazine, terbuthylazine, acetochlor, metolachlor, etc.	Surface water of the Louros River in North Western Greece	0.023, 0.081, 0.051, 0.013, 0.022, 0.026, 0.047 μg/L, etc.	(Kapsi et al., 2019)
Carbendazim, isoproturon, 2-hydroxyatrazine, atrazine, terbutryn, imidacloprid and tebuconazole, etc.	Surface water of Yangtze River Delta, China	574.91, 186.91, 277.20, 191.40, 161.54, 118.50 and 37.61 ng/L, etc.	(Peng et al., 2018)
2-Phenylphenol, acetochlor, alachlor, anthraquinone, atrazine, benfluralin, bifenthrin, biphenyl, etc.	Surface water of an agricultural area in South Georgia, USA	0.28, 0.46, 1.40, 0.29, 1.65, 0.11, 0.14, ND μg/L, etc.	(Glinski et al., 2018)
Dieldrin, endrin aldehyde, γ-Chlordane, endosulfan II, etc.	Surface water of Pampanga River in Philippines	0.28-0.30, 0.337-1.341, 0.019-0.070, 0.042-0.064 μg/L, etc.	(Navarrete et al., 2018)
Pharmaceuticals and personal care products (PPCPs)			
Acetaminophen, caffeine, diltiazem, carbamazepine, fluoxetine, sulfadiazine, sulfamethoxazole, etc.	Surface water of Baiyangdian Lake in north China	31.46, 266.24, 4.90, 72.01, 34.45, 8.79, 21.59 ng/L, etc.	(Zhang et al., 2018b)
	Pore water of Baiyangdian Lake in north China	7.22, 31.67, 0.65, 2.32, 6.86, 1.55, 4.44 ng/L, etc.	
Caffeine, metoprolol, carbamazepine, N,N-diethyl-meta-toluamide, diclofenac, bezafibrate, etc.	Surface water of seven riverside sections of the Beiyun River in Beijing, China in May	1232.5, 213.3, 44.7, 202.5, 6.4, 6.4 ng/L, etc.	(Yang et al., 2017)
	Surface water of seven riverside sections of the Beiyun River in Beijing, China in December	3020.0, 452.0, 93.1, 36.3, 810.0, 83.9 ng/L, etc	
Ciprofloxacin, acetaminophen, caffeine, benzophenone and irgasan	Effluent of a sewage treatment plant in Nagpur city, Maharashtra in summer season	31, 11, 232, 88 and 129 μg/L	(Archana et al., 2017)
	Effluent of a sewage treatment plant in Nagpur city, Maharashtra in rainy season	5, ND, 86, 23 and 24 μg/L	
	Effluent of a sewage treatment plant in Nagpur city, Maharashtra in winter season	58, ND, 148, 61 and 77 μg/L	
Persistent organic pollutants (POPs)			
Σ_{14}PCBs, HCB, BDE 100/99 and Σ_7PAHs	Surface water of Tropical Atlantic (Cape Verde Abyssal Plain)	8.5, 6, 0.3/1.6 and 83 pg/L	(Sun et al., 2016)
	Surface water of North Atlantic (Fram Strait)	0.8, 10, 0.4/0.025 and 148 pg/L	
ΣHCHs, ΣDDTs, ΣPCBs and ΣPAHs	Surface water of a branch of the Grand Canal in Hangzhou, China	1.871, 3.8, 0.52, and 104.4 ng/L	(Zhang et al., 2018a)
ΣPAHs and ΣDDTs	Marine waters of Beibu Gulf, South China Sea	3.2 and 0.015 ng/L	(Kaiser et al., 2015)

PCBs: polychlorinated biphenyls, HCB: hexachlorobezene, BDE: brominated diphenyl ethers, PAHs: polycyclic aromatic hydrocarbons, HCHs: hexachlorocyclohexanes, DDTs: dichlorodiphenyltrichloroethanes

Table 2 – Examples of the toxic differences of certain pollutants to different aquatic microorganisms.

Chemicals	Species	Concentration range	Toxic effects/ Detoxification strategy	References
Hg	*Hizikia fusiformis*	0.02-0.4 mg/L Hg^{2+}	Decrease of photosynthetic pigment contents, chlorophyll fluorescence rate, lipid oxidation	(Zhu et al., 2015)
	Oscillatoria tenuisa and Microcystis aeruginosa	1-250 μg/L MeHg	Growth inhibition	(Huang et al., 2012)
	Chlorella vulgaris	0.001-1 mg/L Hg^{2+} ($IC_{50} = 0.72$ mg/L)	Cell shrinkage and DNA damage.	(Hazlina et al., 2019)
Silver nanoparticles (Ag NPs)	*Scenedesmus acuminatus*	1-100 μg/L ($EC_{50} = 38.5$ μg/L)	Growth inhibition	(Pham, 2019)
	Microcystis aeruginosa	9.3-926 nmol/L	Inhibited cell yield and damaged photosynthesis, carbohydrate metabolism, antioxidant defense and protein translation	(Qian et al., 2016)
	Pseudokirchneriella subcapitata	Different sizes Ag NPs	Growth inhibition	(Ivask et al., 2014)
Glyphosate	*Fucus virsoides*	0.5-2.5 mg/L	Suppressed photosystem II	(Felline et al., 2019)
	Microcystis aeruginosa	1-20 mg/L	Growth inhibition, the presence of nanoplastics reduced the toxicity of glyphosate	(Zhang et al., 2018c)
	Prymnesium parvum	0.1-1000 μg/L	Growth stimulation at low concentrations	(Dabney and Patino, 2018)
Azoxystrobin	*Synechococcus sp.*	0.5-2.5 mg/L	No effect in short time exposure	(Lu et al, 2019a)
	Monoraphidium sp.	0.5-2.5 mg/L	Growth inhibition	(Lu et al, 2019a)
	Phaeodactylum tricornutum	0.1-20 mg/L	Growth inhibition, destruction of subcellular structures, photosystem and antioxidant system	(Du et al., 2019)
	Chlorella pyrenoidosa	0.5-10 mg/L	Growth inhibition; lesion on the cell wall and cytomembrane; disorder of genes expression related to amino acid metabolism	(Lu et al, 2018b)
	Microcystis aeruginosa	0.5-25 mg/L	No effects	(Lu et al, 2018b)
	Chlorella vulgaris	100-2000 μg/L	Excessive reactive oxygen species, photosynthesis inhibition and cell structure destruction	(Liu et al., 2015b)
Ciprofloxacin	*Chlorella pyrenoidosa*	0-150 mg/L	Stimulated algae growth at low concentration, inhibited chlorophyll a yield (40 mg/L), stimulated lipid accumulation (10 mg/L), reduced lipid accumulatio (30 mg/L), induced oxidative stress (10 mg/L) and damaged antioxidant system (60 mg/L)	(Zhang et al., 2019b)
	Raphidocelis subcapitata	3.7-19.1 μmol/L	Inhibited growth yield, induced oxidative stress and reduced activity of SOD at the concentration 19.1 μmol/L	(Aderemi et al., 2018)
	Selenastrum capricornutum	0.5-2.5 mg/L	Decreased photosynthetic rate at 0.5 mg/L, decreased maximum fluorescence, inhibited non-cyclic photophosphorylation and Ribulose-1.5-bisphosphate carboxylase activity.	(Liu et al., 2011a)

status by consuming the pool of the antioxidant glutathione (Pinto et al., 2003).

When subjected to metal stress at nonlethal levels, algae will actively enhance their defensive system to mitigate the effects of the metal toxicity and oxidative stress (Qian et al., 2013, 2016b). Algae also enhance their production of metallothioneins and phytochelatins to prevent toxic metals from continuously replacing essential elements (Park and Jeong, 2018). Moreover, they enhance the activity of antioxidant enzymes (such as superoxide dismutase, peroxidases, and catalases, among others) and create antioxidant-active substances (glutathione, carotenoids, vitamin C, and vitamin E) to alleviate ROS damage (Jun et al., 2015). Recently, it was shown that polyphosphates (polyP) play an important role in the action of algae against the cations of toxic heavy metals. The degradation of polyP may be involved in the chelation of metals and the subsequent promotion of cell tolerance to metals (Adams et al., 2016; Samadani and Dewez, 2018).

Metals such as Hg, Cr, and Pb, and metalloids such as As can be methylated through the action of microorganisms (Juhasova and Cernansky, 2017). The toxicity of some methylated elements, such as As, can be reduced while that of others, such as Hg, is enhanced (Graham et al., 2017). Most current studies on metal methylation have tended to focus on the methylation of Hg. Mobilizing Hg into microorganisms that are capable of methylating it is one of the limiting steps in the formation of MeHg. Algal dissolved organic matter (DOM) can influence the methylation rate of Hg (Ding et al., 2019). However, Lei et al. (2019) found that DOM from *Chlorophyte* and *Euglena mutabilis* strongly inhibited Hg uptake into aerobic and anaerobic bacteria, while DOM from *Euglena gracilis* did not display this property. In short, harmful metals such as Hg have different toxic effects in different microalgae and cyanobacteria and are transformed in different ways by them (Table 2).

The various responses of microbes might sequentially influence the metabolism of other microbes as well as the fate of the metals themselves. Thus, investigating the effects of pollutants on microalgae and cyanobacteria significantly more complex, and additional research is required to elucidate the metal transformation processes, including methylation, as well as the various influencing factors that act on the ecological network of aquatic microbial communities.

1.2. Nanomaterials

Nanotechnology is evolving rapidly and affecting almost every aspect of industries and society worldwide. In 2015, the International Organization for Standardization (ISO) defined "nanoscale" as a size range of approximately 1-100 nm (ISO/TS 80004-1:2015). In the updated ISO standard, released in 2018, all NMs were divided into the following seven primary categories: carbon nanoobjects and related carbon nanoscale materials, oxides, metals, quantum dots (QDs), organic polymer NMs, bioinspired materials, and cellulose NMs (ISO/TR 12885:2018). Nanomaterial pollution in aquatic environments is related to NM production and usage, as well as to the discharge of NMs from wastewater treatment plants (Bundschuh et al., 2018; Ke et al., 2020). Piccinno et al. (2012) have estimated the production amounts of 10 different NMs (TiO_2, ZnO, FeO_x, AlO_x, SiO_2, CeO_2, Ag, QDs, carbon nanotubes (CNTs), and fullerenes) worldwide and for Europe. They found that the NM with the highest production rate was TiO_2, which had a global production amount as high as 10,000 tons. Further, the production amounts of CeO_2, FeO_x, AlO_x, ZnO, and CNTs were 100–1000 tons/year, while those of Ag, QDs, and fullerenes were less than 10 tons/year.

The NMs are endowed with unique physicochemical properties because of their nanoscale dimensions, which also increases the threat posed by their exposure to organisms. Nanomaterials are toxic to microalgae and cyanobacteria mainly through four ways, which can be summarized as follows. i) Damage owing to surface interactions: NMs get adsorbed onto the surfaces of microbes and directly interact with the cell membrane or cell wall to damage the cells. The cell wall is a shield that protects organisms from the metal toxicity arising from the chelation of heavy metals by the cell wall polysaccharides (Andrade et al., 2010). However, the cell wall does not act as a protective barrier for NMs (Sendra et al., 2018); instead, it acts as an NM interaction site and this results in toxicity (Perreault et al., 2012). ii) Internalization of NMs: NMs are readily adsorbed by cells and get assimilated into them, whereby they cause damage to organelles and proteins (Li et al., 2019). iii) Production of ROS: similar to many other pollutants, NMs can cause the excessive production of ROS, which leads to adverse effects such as lipid peroxidation. iv) Ion effect: metal and metal oxide NMs also can be dissolved in water and transformed into the corresponding metal ions. These mechanisms usually occur together in individual microalgae or cyanobacteria (Sekine et al., 2017; Xia et al., 2015).

The characteristics of NMs in aquatic environments is closely related to their toxicity. (Qian et al., 2016; Yang et al., 2012; Zhang et al., 2020). These include the following: i) Particle size: the smaller the size of NMs, the greater their toxicity to microbes will be; this may be owing to the fact that NMs with a smaller size are internalized more easily by cells (Lei et al., 2016a), and ii) Surface coatings (surfactants, polymers, and polyelectrolytes): coatings with positive and negative functional groups reduce the adsorption and aggregation behavior between NMs or with microbes, thus making the former more stable and consequently affecting their bioavailability and toxicity (Urs et al., 2018). For example, ZnO nanoparticles (ZnO-NPs) coated with 3-aminopropyltrimethoxysilane and uncoated ZnO-NPs were more potent at inhibiting growth of algal cells than ZnO-NPs coated with dodecyltrichlorosilane (Yung et al., 2017). Given the vast differences in the material types, sizes, shapes, and coatings of NMs, it is impossible to test all of the materials to the same extent. Therefore, the characterization of NMs is essential (Oberdörster, 2010). However, their characterization does not explain the dynamic changes in their properties. In fact, NMs interact with the surrounding environment, both biotic and abiotic, immediately after they are introduced into the environment, and their behavior and fate are affected by the environmental dynamics (Schirmer and Auffan, 2015). For example, water quality factors, such as the salinity and pH, can alter the agglomeration and dissolution behavior of NMs (Sendra et al., 2017). The properties of NMs may also change when they come in contact with organisms; this, in turn, may affect their bioavailability, for example, positively charged branched polyethilenimine-coated AgNPs tend to aggregate in the presence of algae, while citrate-coated AgNPs tend to dissolve (Malysheva et al., 2016).

Some NMs promote algae growth at low concentrations rather than reduce it (some carbon NMs promote algae growth even at high concentrations). They also promote other characteristics of algae. For instance, they enhance the processes of photosynthesis and lipid synthesis (He et al., 2017). However, these so-called "low concentrations" were found to be significantly higher than the prescribed limits for various other pollutants and far greater than their concentrations in the environment. Therefore, it remains to be seen whether engineered NMs pose a significant threat to the aquatic environment. Unlike other pollutants, some NMs are inherently inert and unlikely to induce biological effects; however, their dose makes them a poison (Ottoboni, 2011). Krug (2018) suggested that more than 60% of the reported nanotoxicological results require a critical reevaluation and cannot be considered in risk assessment standards, given the high doses of the materials in question as well as our inability to control NMs and the current lack of essential physicochemical characterization data.

Different aquatic organisms may exhibit very different sensitivities to NMs (see Table 2). For example, the silver nanoparticles (AgNPs), which are among the most widely used commercial nanoproducts, were found to be significantly more toxic to the growth, photosynthesis, antioxidant systems, and carbohydrate metabolism of the cyanobacteria *Microcystis aeruginosa* than to those of the green alga *Chlorella vulgaris* grown in laboratory batch cultures (Chen et al., 2016). *C. vulgaris* could efficiently detoxify the ROS induced by the AgNPs by producing more antioxidant enzymes, while *M. aeruginosa* failed to detoxify in a similar manner when exposed to AgNPs at the same concentration (Qian et al., 2016). AgNPs also had negative impacts on zooplankton grazing and the photosynthesis of phytoplankton at concentrations greater than 500 μg/L (Baptista et al., 2015). However, trace amounts of AgNPs (10 μg/L) could activate the growth of eukaryotes (green algae and zooplankton) in microcosm experiments (Lu et al., 2020a).

1.3. Pesticides

Depending on the target organisms, pesticides can be categorized as herbicides, fungicides, insecticides, and plant growth regulators (Delorenzo et al., 2001; Qu et al., 2020). Worldwide pesticide use has increased dramatically over the past few decades; a trend is expected to continue in the coming years. Liu et al. (2015c) reviewed worldwide pesticide usage from 1990 to 2010. A total of 82 countries consumed an average of 342,000 tons of insecticides each year. The average annual consumption of herbicides in 75 countries was 566,000 tons, while 77 countries consumed an average of 353,000 tons of bactericides and fungicides per year. The concentration of pesticides in aquatic environments typically ranges from the order of nanograms per liter to micrograms per liter (Table 1). Given their higher usage, herbicides may cause more pollution in aquatic environments than do other types of pesticides (Glinski et al., 2018; Peng et al., 2018), primarily due to excessive use in agriculture (Kuster et al., 2009). Moreover, approximately 25% of commercial pesticides are chiral (Garrison, 2006), and their effects on microorganism growth and development are consistently enantioselective (Asad et al., 2017). Thus, evaluating the toxicity of chiral pesticides becomes more difficult.

The removal of pesticides from the aquatic environment is mainly dependent on their biodegradation by microorganisms (Liu et al., 2018b). Bacterial strains belonging to the genera *Pseudomonas, Bacillus, Xanthomonas*, and *Rhodococcus*, as well as some fungi exhibit a strong ability to degrade various pesticides (Alexandrino et al., 2020; Lin et al., 2020; Priya et al., 2020). Some of the genes corresponding to pesticide-degrading enzymes, such as *phnGHIJKLM, puhA/puhB*, and *atzABCD/trzDN*, act on broad-spectrum herbicides such as glyphosate, diuron, and simazine (Mauffrey et al., 2017). Moreover, pesticide removal may be affected by seasonal changes in the hydrological and hydrochemical conditions, for example, Maillard et al. (2011) found that, in wetlands, pesticides like diuron and glyphosate could be removed more efficiently in spring than in summer. Finally, sediments in aquatic systems can increase the persistence of pesticides (Laabs et al., 2007).

Pesticides are bioactive chemicals that have been formulated to kill target organisms. Given that insecticides, herbicides, and fungicides have different sites of action, their specific effects on macroinvertebrates, algae, bacteria, and fungi also vary. Therefore, pesticides belonging to different categories may inhibit a certain microorganism but have significantly different microecological effects in the aquatic environment. i) Insecticides are designed to kill insects. Thus, zooplankton with metabolic characteristics similar to those of insects would also be strongly inhibited by them. The growth of microalgae and cyanobacteria might be out of control due to the inhibition of their predator (zooplankton) under insecticides stress. For instance, insecticide mixtures reduce the diversity and community structure of aquatic microbial communities dominated by macroinvertebrates such as mosquito larvae, while herbicide mixtures do not (Muturi et al., 2017b). ii) The most important mechanism of action of herbicides is to inhibit photosynthesis through electron-transfer disruption by blocking the Hill reaction in Photosystem II. Other mechanisms of action include inhibiting carotenoids, lipids, microtubules, cellulose, or folate synthase, as well as the enzymes involved in the amino acid synthesis pathway. Herbicide pollution inevitably inhibits aquatic producers like microalgae and cyanobacteria and changes their metabolic characteristics (Qian et al., 2008; Qian et al., 2014). iii) Fungicides, which serve as general biocides, have many different mechanisms of action, such as inhibiting the biosynthesis of intracellular membrane components and preventing oxidative phosphorylation (Rodrigues et al., 2013). Non-target microbe like *Chlorella pyrenoidosa* could also inhibited by fungicide and subsequently led to uncontrolled growth of cyanobacteria (Lu et al., 2019a).

In addition to being toxic to target organisms, pesticides and their subsequent intermediates may also have unintended effects on nontarget microorganisms including microalgae and cyanobacteria (Table 2). For example, azoxystrobin (AZ), a broad-spectrum fungicide that prevents ATP generation, was found to inhibit the growth of *Chlorella pyrenoidosa* by inducing oxidative damage and perturbing amino acid metabolism, ascorbate synthesis, fatty acid metabolism, and RNA translation (Lu et al., 2018b). Further, a dose as low as 2 μg/L of the triazole fungicide tebuconazole inhibits substrate-induced respiration and photosynthesis in natural biofilms (Artigas et al., 2014).

However, the aquatic ecotoxicity of some pesticides remains a matter of debate. For example, a few studies claim that the most commonly used herbicide (i.e., glyphosate) is environmentally friendly (Williams et al., 2000), even when this is used to control noxious aquatic weeds (Gardner and Grue, 1996; Mansor, 1996). However, several recent reports have strongly refuted this claim (Pizarro et al., 2016; Reno et al., 2016; Vera et al., 2012). Microalgae may be affected because of their physiological similarity to plants, while nontargeted aquatic organisms are probably adversely affected because of the imbalances in the aquatic ecosystem (Cedergreen and Streibig, 2005; Muturi et al., 2017a; Smedbol et al., 2018). Ren et al. (2017) found that in a phosphorus-deficient environment, if glyphosate is the only source of phosphorus, the competition between algae may be affected, providing *M. aeruginosa* a strong competitive advantage in freshwater. Moreover, Lu et al. (2020b) found that, in the aquatic microbial community, *Synechococcus* can mitigate the toxicity of glyphosate to other species by using glyphosate as a P source. These studies suggest that glyphosate can destroy the ecological equilibrium that exists in aquatic microbial communities, which will result in some of the species such cyanobacteria having a distinct growth advantage.

To sum up, pesticides can cause direct or indirect ecotoxicity to microalgae and cyanobacteria, and it is recommended that evaluations of the effects of any given pesticide on microalgae or cyanobacteria should not be limited to its toxicity to a single species; instead, evaluations should focus on its ability to induce changes through ecological relations.

1.4. *Pharmaceutical and personal care products*

The term PPCPs refers to nine categories of pharmaceuticals: hormones, antibiotics, lipid regulators, nonsteroidal anti-inflammatory drugs, beta-blockers, antidepressants, anticonvulsants, antineoplastic agents, and diagnostic contrast agents (Doerr-MacEwen and Haight, 2006). PPCPs also include four classes of personal care products: perfumes, preservatives, disinfectants, and sunscreens (Wang and Wang, 2016). The concentration of PPCPs in aquatic environments is usually on the order of nanograms per liter to micrograms per liter (Table 1), and their distribution in aquatic environments is affected by the source and hydrological conditions (Yang et al., 2017). Untreated sewage from hospitals, pharmaceutical plants, and aquafarms, and sewage treatment plant (STP) effluents are the main sources of PPCPs (Dai et al., 2016; Lu et al., 2018a; Rodriguez-Mozaz et al., 2015; Xu et al., 2014; Zhang et al., 2018b). Most PPCPs are difficult to remove through wastewater treatment processes. Archana et al. (2017) reported on the seasonal changes and environmental quality control data for five PPCPs (i.e. acetaminophen, ciprofloxacin, caffeine, irgasan, and benzophenone) in the influent and effluent of a STP. The total concentration of the PPCPs was 12–373 μg/L in the influent and 11–233 μg/L in the effluent, and thus, inefficient removal was detected. In addition, pharmaceuticals may be biomagnified through aquatic food webs, where they can cause widespread ecological damage. The adsorption of PPCPs by sediments is one of the most important pathways for their removal from aquatic environments (Tang et al., 2019). Photochemical and biological degradation are also important removal pathways (Baena-Nogueras et al., 2017).

Antibiotics are an important class of PPCPs. In some developing countries with large populations, such as China, antibiotics are being used in large quantities. In 2013, a total of 162,000 tons of antibiotics were used in China, which is approximately nine-fold and 150-fold the amount used in the United States and the United Kingdom, respectively. Studies have shown that cyanobacteria are sensitive to antibiotics (Lu et al., 2019b; Qian et al., 2012) and that the toxicological effects of antibiotics on eukaryotes such as green algae are less of a concern. Liu et al. (2011b) found that three antibiotics, namely, erythromycin, ciprofloxacin, and sulfamethoxazole, affect the photosynthesis of the green alga *Selenastrum capricornutum*, and changes in its primary photochemistry and electron transport, photophosphorylation, and carbon assimilation characteristics were observed. Chloramphenicol and roxithromycin can affect green algae by inhibiting fatty acid synthesis, inducing protein and DNA aggregation, aiding the accumulation of lipid peroxidation products, and accelerating the loosening of beta-sheet structural proteins and the conversion of B-DNA into Z-DNA (Xiong et al., 2019). Antibiotics in aquatic environments directly affect the growth of bacteria and destroy the diversity of aquatic microbial communities (Davies, 2010), as a reslut, microalgae and cyanobacteria would finally be influence due to the community disturbance. Ciprofloxacin (CIP) can inhibit most cyanobacteria even at trace concentrations. However, interestingly, Lu et al. (2019b) showed that 7 μg/L CIP in a freshwater microcosm was tolerable to cyanobacteria and unexpectedly inhibited eukaryota growth; this was in direct contrast to the antibacterial classification of CIP and the enhanced tolerance of cyanobacteria to CIP is likely to be associated with microbial mutualisms.

In general, PPCPs pollution (take antibiotics as an example) not only directly influences the growth of microalgae and cyanobacteria in the aquatic environment, but also results in antibiotic-resistant bacteria, which might change the microecological balance of aquatic environment.

1.5. *Persistent organic pollutants*

In May 1995, the Governing Council of the United Nations Environment Programme drafted a list of POPs. Initially, the list only contained 12 chemicals: aldrin, chlordane, dieldrin, endrin, heptachlor, hexachlorobenzene, mirex, toxaphene, polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane, dioxins, and polychlorinated dibenzofurans. Since 2001, the list already includes additional compounds, such as polycyclic aromatic hydrocarbons (PAHs) and brominated flame retardants. Some organophosphorus pesticides (OPs) are also classified as POPs based on their characteristics. The concentration of most POPs in aquatic environments usually ranges from the order of picograms per liter to nanograms per liter (Table 1). The book *Silent Spring* by Rachel Carson brought these pollutants to the attention of the world. Chemicals that are POPs are highly toxic to organisms and difficult to degrade. They can migrate long distances and spread across the world. Wang et al. (2018) studied the transcriptomic response of the dinoflagellate genus *Prorocentrum* to PCBs by using microarrays and found that while PCBs may not cause chloroplast and oxidative damage, they result in cell cycle arrest and apoptosis. Experiments by Halm-Lemeille et al. (2014) have shown that the toxicity of PCBs depends on the target species and that they have a nonspecific mode of action on green algae (e.g., anesthesia). However, mixtures of PAHs as well as those of OPs were found to reduce the expression of the photosynthetic genes *rbc*L (RuBisCO large subunit) and *psb*A (PSII D1 protein) in oceanic *Prochlorococcus* populations (Fernandez-Pinos et al., 2017).

POPs are known for the property of persistent, therefore, their toxicological properties may be quite different. The toxic mechanisms of POPs to microalgae and cyanobacteria vary widely, which may include but is not limited to oxidative damage, cell cycle arrest and apoptosis and photosynthetic inhibition. Aquatic toxicology studies on POPs remain limited. Further researches on toxicology and tolerance thresholds needs to be widely carried out.

1.6. *Emerging pollutants*

Owing to the advancements in new industries, a large number of new materials and chemicals have been developed to support the strategic concept of efficient and green sustainable development. However, thorough chemical and toxicological research on the environmental behavior of these new materials and chemicals is required to identify new types of environmental pollutants. For example, graphene, microplastics, and ionic liquids have been identified as emerging environmental pollutants, which can easily enter aquatic systems through various ways and cause unpredictable ecotoxicological risks (Banchi et al., 2019; Luo et al., 2019; Mao et al., 2018; Thuy Pham et al., 2010). Other emerging pollutants like artificial sweeteners, salicylic acid and fullerenes also got a lot of attentions.

Graphene is a two-dimensional monolayer carbon atom sheet with a hexagonal structure (Novoselov et al., 2012). It has attracted widespread attention in the industry because of its electronic and thermal conductivity and gas impermeability characteristics (Gogotsi, 2015). The annual output of graphene has increased from approximately 120 tons in 2015 to 1000 tons in 2019, and investments are expected to reach approximately 400 million US dollars by 2020 (Ghaffarzadeh, 2016). This exponential increase in production may lead to the massive release of graphene-based materials (GBM) in aquatic habitats due to loss of graphene-rich products and poor waste disposal (Banchi et al., 2019). Studies have shown that graphene has toxic effects on freshwater algae (Nogueira et al., 2015; Martin de et al., 2018). Generally, because the cell wall thickness of freshwater algae is approximately 20 nm, graphene has a strong internalization ability and bioavailability. The GBM exposure will lead to excessive production of ROS and cause membrane damage of *Chlamydomonas reinhardtii* and *Raphidocelis subcapitata* (Nogueira et al., 2015; Martin de et al., 2018). Hence, additional research is required to understand how graphene toxicity is influence by differences in the morphological structure of algal species.

Global plastics production reached 359 million tons in 2018, an increase of 9 million tons from 2017, which has brought about ecological pollution problems that cannot be ignored (The China Plastics Industry Editorial Office). Microplastics, an emerging pollutant, are commonly detected in freshwater ecosystems (Lambert and Wagner, 2018). Studies have shown that microplastics have different toxicological effects on freshwater algae according to their particle size and the properties of surface functional groups (Liu et al., 2016; Zhang et al., 2018a). Microplastics usually adsorb on the surface of algae (Bhattacharya et al., 2010; Zhang et al., 2018a), thereby forming a light shield that affects photosynthesis, or the microplastics can cause the algal cell wall to crack after adsorption (Zhang et al., 2016). Meanwhile, microplastics of a few nanometers can enter cells and destroy organelles, leading to algal cell death (Besseling et al., 2014; Zhang et al., 2018a). In addition, because of the different particle sizes and surface charges of microplastics, the strength of the agglomeration effect in freshwater is different, and the bioavailability also changes (Turner et al., 2010). Therefore, the characteristics of microplastics must be considered when investigating their ecological toxicity in freshwater environments.

Ionic liquids (ILs) are a new type of organic compound being used as a substitute in the green chemical industry (Thuy Pham et al., 2010). Thorough aquatic ecotoxicological assessments should be conducted owing to the solubility of ILs in water and the large number of literature reports on their toxicity to aquatic organisms (Anthony et al., 2001). In particular, studies on the toxicological risks of ILs revealed that freshwater algae, as primary producers, have assumed important aquatic ecological roles (Fan et al., 2019a; Jin et al., 2019). ILs are usually divided into cations and anions, and it was found that cationic ILs presented a greater toxicity to *Scenedesmus obliquus*; additionally, the toxicity of anions was found to be related to ionic properties; for example, bromide is a more powerful toxicant than chloride (Fan et al., 2019b). After exposure to ILs (cations), *C. pyrenoidosa* showed obvious photosynthesis inhibition and oxidative stress responses, algae cell deformations, cell wall and cell membrane damage; meanwhile, this toxicity of ILs to *C. pyrenoidosa* was positively correlated with the exposure time and concentration (Jin et al., 2019). To sum up, cationic ILs are usually more toxic to microalgae than anions ILs, which mainly cause cell membrane damage, oxidative stress and photosynthesis inhibition in algal cells.

While existing pollutants are being discharged into the environment daily, new ones are also being produced continuously. The above pollutants are considered as emerging water pollutants for their proven or potential adverse effects on microalgae, cyanobacteria, aquatic ecology and human health. Novel investigative and analytical approaches for future monitoring studies and toxic mechanism researches are proposed.

2. Interactions between pollutants and abiotic factors in the aquatic environment

2.1. Combined toxic effects of different pollutants

The combined toxic effects of pollutants in the aquatic environment can be highly noticeable. Pollutants never appear alone in water bodies (Table 1). At present, the combined toxicity of metals has received wide concerns (Gao et al., 2020; Ramakrishnan et al., 2010). The synergistic effect of Cu and Cd was an increase in the ROS content, disruption of chlorophyll synthesis, and inhibition of *C. vulgaris* growth (Qian et al., 2009; Wei et al., 2014). This may be attributable to the fact that Cd can replace Cu in various cytoplasmic and membrane proteins, thereby increasing the amount of free Cu ions involved in the generation of oxidative stress by the Fenton reaction. Interestingly, metallic Cu was also found to reduce the bioaccumulation of Cd in *Ulva compressa* (Kovacik et al., 2018). This difference could be related to both the metal concentration and the species involved (Babu et al., 2014; Franklin et al., 2002). In addition, the toxic effects of different combinations of metals are also crucial. An evaluation of the combined toxicity of binary heavy metals to *Microcystis aeruginosa* found that when the mixed concentration is low, such as with Cu^{2+} + Cd^{2+}, Cu^{2+} + Cr^{3+}, Zn^{2+} + Cr^{3+}, there is a synergistic effect, and when the mixed concentration is higher, there is an antagonistic effect; however, the binary whole of Zn^{2+} + Cd^{2+}, Pb^{2+} + Cr^{3+}, Pb^{2+} + Cd^{2+}, Pb^{2+} + Zn^{2+}, and Cr^{3+} + Cd^{2+} always showed antioxidative effects on *Microcystis aeruginosa* (Gao et al., 2020).

The combined aquatic toxicity of nano-type pollutants and other pollutants also has received many attentions in recent years. Because of the huge specific surface area, electrostatic capacity, and recombination ability of nano-pollutants, they can adsorb and bind to a variety of other aquatic pollutants, which leads to more complex toxic effects on aquatic algae (Fan et al., 2010; Liu et al., 2018a; Whiteley et al., 2013). For instance, it was reported that the presence of nanoscale Al_2O_3 increased the bioabsorption of Pb in the marine alga *Isochrysis galbana* (Hu et al., 2018); the graphene oxide NPs and zinc oxide showed a superimposed toxic effect on *Scenedesmus obliquus* (Wu et al., 2019); the combined toxicity between nano-copper oxide (nCuO) and the nano-zinc oxide (nZnO) to freshwater algae *Scenedesmus obliquus* was greater than that of the nZnO (Ye et al., 2017). In addition to the synergistic effects, the combination of nanoscale ZnO and cetyltrimethylammonium chloride exhibits an antagonistic effect on *C. vulgaris*; this is because of the inhibitory effect of cetyltrimethylammonium chloride on the dissolution of the nanoscale ZnO and the accumulation of Zn in algae (Liu et al., 2018a).

Combined toxic effects also have been observed in the cases of mixtures of many other categories of pollutants, such as pesticides, nanoplastics and PPCPs (Affek et al., 2018; Carusso et al., 2018; Liu et al., 2013; Tien and Chen, 2012; Zhang et al., 2018a). Petersen et al. (2014) found that the combined effects of the multi-pollutant mixture (PPCPs, biocides, PAHs and alkylphenols) were clearly additive, and indicated that these pollutants will collectively cause environmental risks. Antibiotics can reduce the toxicity of herbicides to *Microcystis aeruginosa* by stimulating photosynthesis and reducing oxidative stress, thus leading to increased microcystin production (Yu et al., 2019). The combination of different ratios of macrolide antibiotics and the azole fungicide ketoconazole produced a strong synergistic effect and enhanced the growth inhibition effect on the green alga *Pseudokirchneriella subcapitata* (Yamagishi et al., 2017). The nanoplastics, which often carry different functional groups because of their various industrial uses, also show different adsorption capacities to other pollutants. For example, polystyrene cationic amino-modified nanoplastics ($PS\text{-}NH_2$) can adsorb glyphosate on their surface, which enhances the stability of the dispersion system, causing a greater number of $PS\text{-}NH_2$ to be adsorbed on the surfaces of cyanobacteria. This may result in the further enrichment of $PS\text{-}NH_2$ in the food chain (Zhang et al., 2018c). Meanwhile, the carboxylated polystyrene nanoparticles (PS-COOH NPs) did not absorb the Cu ions and no difference in the growth inhibition of *Raphidocelis subcapitata* were observed after exposure to PS-COOH NPs and Cu (Bellingeri et al., 2019).

In combined toxicity studies, the concentration addition (CA) and independent action (IA) models, which assume a similar action or a dissimilar action of mixture components, respectively, are generally used to predict the combined toxic effects of pollutants (Faust et al., 2003; Petersen et al., 2014).However, these experiments are generally restricted to the assessment and prediction of the combined toxicity. Future studies should focus on the interactions between the pollutants as well as the underlying mechanisms of the molecular toxicity caused by the pollutants in combination.

2.2. Combined effects of abiotic factors and pollutants

Abiotic environments may alter the biological effects of pollutants and the growth state of the microalgae and cyanobacteria (Zhang et al., 2021). Van Regenmortel and De Schamphelaere (2018) reported that different water chemical conditions may lead to changes in the combined action of metals on algae. In the case of low cation competition (i.e., in the case of soft water), the antagonism between metals is stronger than that during high cation competition (i.e., in the case of hard water). Organic materials such as humic acid can reduce the bioavailability of metals by adsorption, as revealed by Bai et al. (2019) who showed that humic acids with larger molecular weights have stronger inhibitory effects on the bioavailability of Pb to the algae *C. pyrenoidosa*. In addition, inorganic nutrients can also affect the toxicity of metals. Inorganic nitrogen and phosphorus compounds have been found to reduce the toxicity of Cr to the freshwater alga *C. vulgaris* (Liu et al., 2015a; Qian et al., 2013). However, an increase in the phosphorus concentration increased the inhibitory effect of Cu on the growth of a freshwater amphipod, *Hyalella azteca* (Li et al., 2012). Further, temperature is an important factor. Van de Perre et al. (2018) found that that Zn toxicity had a smaller effect at higher temperatures. Chalifour et al. (2014) reported that temperature can affect the toxic influence of herbicides (norflurazon and fluridone) on photosynthesis as well as the pigment and fatty acid composition of the freshwater alga *Chlamydomonas reinhardtii*. Rico et al. (2018) found that temperature along with genetic variations and species competition affect the sensitivity of algae to the antibiotic enrofloxacin. Therefore, in toxicology studies, the effects of abiotic environmental factors, such as water chemical conditions (including N, P contents, pH value etc.), organic materials present, and temperature, can greatly affect the toxicity of pollutants and should not be ignored.

3. Conclusions

Human behavior affects natural processes of ecological systems via physical and chemical forces and interferes with biodiversity. Changes in the dynamics of microalgae and cyanobacteria in ecosystems can be useful indicators of the pollution impact and extent. However, information on the ecotoxicology of various aquatic pollutants on microalgae and cyanobacteria is still fragmentary.

The characteristics of pollutants determine their toxicological mechanisms in microalgae and cyanobacteria, which can be generally divided into physical damage and cytotoxicity. Physical damage is mainly caused by cell damage and shielding effects, as it occurs with the emerging pollutant microplastics, whereas the cytotoxicity of highly toxic pollutants, such as organic pesticides, mainly destroys small molecules and causes cellular metabolic damage. Although some studies have integrated the toxicity data of pollutants in a model to predict the toxicity of specific pollutants, results did not consider the interactions among the entire ecological community and the combined toxic effects of multiple pollutants.

Therefore, further study on the toxic effects of pollutants on aquatic microorganisms is required to continually identify pollutants and understand the combined ecological risks of multiple pollutants. This equates to shifting from a reductionist perspective to a holistic perspective for understanding pollutants and freshwater ecosystem health, allowing us to better predict the risks of these substances to ecosystem health.

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