Review Article

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An Overview Of The Cyanobacterial Metabolites And Their Impacts On Drinking Water Quality

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1. Abstract

Climate change and intensified human activities lead to the frequent occurrence of cyanobacterial blooms worldwide. Cyanobacteria could produce diverse toxins and taste-and-odor compounds that threaten the health of human beings. Besides, algogenic organic matter introduced to the drinking water treatment plant could elevate the formation potentials of disinfection by-products (DBPs). The co-occurrence of various types of cyanobacterial metabolites with different characteristics in water resources is a growing concern, as that not only poses a threat to the water quality of drinking water source but also poses a huge challenge to the tap water production process. Here, we focused on toxins, taste-and-odor compounds, and algae-derived DBPs precursors produced by cyanobacteria, reviewed and summarized the research progresses in terms of the source of the related metabolites, analysis and detection methods, temporal and spatial changes, environmental impact factors, as well as the corresponding removal methods and technologies. This review is aimed to provide a scientific basis for the management of water sources such as lakes and reservoirs suffering cyanobacterial blooms, as well as to provide theoretical support for the safe production of drinking water treatment plants using eutrophic water bodies as sources.

Keywords:
Algogenic organic matter; Microcystins; Taste and odor compounds; Disinfection by-products; Cyanobacterial blooms

Key Contribution:
We focused on the typical cyanobacterial metabolites including toxins, taste-and-odor compounds, and algae-derived disinfection by-products precursors, reviewed and summarized the research progresses in terms of the source of the related metabolites, analysis and detection methods, temporal and spatial changes, environmental influencing factors, as well as the corresponding removal methods and technologies, aiming to provide guidance to the management of water sources and drinking water treatment plants suffering from cyanobacterial blooms.

In recent decades, eutrophication in freshwater has become increasingly serious mainly due to climate change and intensified human activities, resulting in the frequent occurrence of cyanobacterial blooms globally [1-5]. When bloom occurs in drinking water sources, high-density cyanobacteria enter water plants and seriously affect the safety of drinking water. For instance, in the Anglian Region of the United Kingdom, the density of Microcystis reached 400,000 cells/mL, resulting in the water supply being shut down for 8 weeks [6,7]. In May 2007, the outbreak of cyanobacteria in Taihu Lake caused the sudden deterioration of water quality in the Gonghu Water Works in Wuxi city, and the serious odor of tap water aroused social concern [8]. In Mid-June 2010, cyanobacteria accumulated in the eastern Chaohu Lake, leading to the peculiar smell of the water from some drinking water plants (DWPs), and the drinking water was cut off for 4 hours [9]. Besides, DWPs in the North American (Laurentian) Great Lakes basin have repeatedly faced problems in recent years with cyanobacterial toxins that cannot be effectively removed [10]. Since the harm of cyanobacteria bloom to the water environment is getting worse, it has become a hot research direction on how to control cyanobacteria in the water sources to ensure water quality safety. Currently, cyanobacteria control technologies widely used including biodegradation, chemical adsorption, and mechanical salvage, etc. The influences of cyanobacteria bloom on drinking water safety are as follows: firstly, the presence of cyanobacteria pellets. In the process of water treatment, algae and their dead residues would interfere with the coagulation process, affect the conventional operation of water plant technology, and finally reduce the water quality [11,12]. Secondly, cyanobacteria metabolites influence the water quality in water source and DWPs. Therefore, the control of cyanobacteria in drinking water sources should not only focus on the quantity of cyanobacteria but also understand the effects of cyanobacteria metabolites on water quality.

The reasons for the pollution of drinking water sources caused by cyanobacteria metabolites are mainly as follows: (1) Cyanobacteria metabolize toxins directly harm aquatic organisms and seriously affect

human health. Studies have shown that many cyanobacteria can produce toxins, such as Microcystis, Anabaena, Oscillatoria, and Amphora, etc. [13]. Cyanobacterial toxins mainly include hepatotoxins, neurotoxins, tumotoxins, and other toxins affecting the skin and gastrointestinal tract, among which microcystins (MCs) is the most common [14,15]. (2) Taste-and-odor compounds produced by cyanobacteria cause the peculiar smell in the water source and drinking water, which is the most direct manifestation of the water quality[16]. After the odor incident occurred in the Taihu Lake water source area in 2007, the odor problem in water has aroused widespread attention [17]. It has been found that dozens of substances may produce odor in eutrophic water, among which geosmin (GSM) and 2-methylisoborneol (MIB) are the substances with the highest frequency of occurrence and detection [18]. (3) Cyanobacteria and the algae source organic matter produced by their metabolism and death degradation are the precursor of disinfection by-products (DBPs). In the production of tap water with eutrophication water as raw water, chlorination disinfection greatly increases the DBPs [19-22].

In this paper, we focused on the algal toxins, taste-and-odor (T/O) compounds, and algae-derived DBPs precursors produced by cyanobacteria. The research progresses were reviewed and summarized in terms of the source of the related metabolites, analysis, and detection methods, temporal and spatial changes, environmental influencing factors, as well as the corresponding removal methods and technologies. This review aims to provide a scientific basis for the management of water sources such as lakes and reservoirs suffering from cyanobacteria, as well as to provide theoretical support for the safe production of drinking water treatment plants using eutrophic water bodies as raw water.

2. Research progress on algal toxins

2.1. Classification and properties of algal toxins

Cyanobacteria overbreed to form algal bloom and release algal toxins to water, which not only affect the sensory properties of the lake environment but also affect aquatic life, human health, and drinking water production. Studies have found that more than 40 out of 2,000 cyanobacterial species can produce toxins, and more than 50% of cyanobacteria blooms are toxic, causing harm to the animal and human health and even causing death [23-25]. In recent years, the frequency and scale of cyanobacteria producing toxins increased worldwide, and multiple toxins have been detected in one water body [26-30]. According to the mode of action, cyanobacterial toxins are divided into hepatotoxins (e.g. MCs, nodularins, and cylindrospermopsin), neurotoxins (e.g. anatoxin-a, saxitoxin and neosaxitoxin), tumourogenictoxins (e.g. MCs), and other toxins affecting the skin and gastrointestinal canal (e.g. lipopolysaccharides), etc. [14,15]. Among the toxins, MCs are the most common and widely studied cyanobacterial toxins. It is a class of cyclic polycyclic substances produced by Microcystis, Anabaena, Oscillatoria, Nostoc and Anabaenopsis, and Hapalosiphon, with large relative molecular weights of about 1000 [31]. In 1982, its molecular structure were determined firstly with MeAsp, Adda, Mdha, and Dha [32]. So far, more than 240 MC variants have been found in bloom water and isolated algal strains [33], among which MC-LR, MC-RR, and MC-YR are the most common, with the letters L, R, and Y represent leucine, arginine, and tyrosine, respectively. Meanwhile, MC-LA, MC-YM, MC-FR, MC-WR, MC-LV, MC-AA, and MC-LF were also detected in water [34-36].

In the environment, MCs are neutral or anionic and easily soluble in water, with a solubility of more than 1 g/L [37]. Due to their hydrophobicity and polar functional group, MCs are not easy to settle or be adsorbed in sediments and suspended particles. In river water containing 5 g/L MCs, only 10% and 7% of MCs were adsorbed on suspended particles and sandy sediments after three days [37]. As the stable molecular structure of the MCs with the circular structure and the interval of the double bond, they can tolerate very broad pH and temperature conditions. Under natural conditions, MCs can exist for a long time without being photolysis or biodegradation, even under an extreme temperature of 300°C. MCs have various degradation time in different water. It can be stable for up to 27 d in deionized water, for 12 d in sterilized river water, and less than 7 d in original river water [38]. When the extracted MC-LR was placed outside to receive direct sunlight and shaken regularly to allow it to enter the air for 20 days, over 50% of the MC-LR still failed to degrade [39]. When the pH values were 1 and 9 respectively, the half-period of MC-LR was 5 and 14 weeks [40]. Nevertheless, MCs still contain three regions that are easy to be oxidized, photodegraded, and biodegraded: conjugated double bonds of Adda functional group, the single and double bonds of Mdha functional groups, and the side chains of different amino acids, which can result in partial photolysis and decreased activity of MCs in the presence of long time light, strong ultraviolet light and pigment [41]. MCs are recognized as hepatotoxins and cancer-promoting agents. MCs in water mainly release after the rupture of algal cells. Long-term exposure could cause damage to human liver and endanger the human life in extreme cases [13,27,42]. Besides, MCs increase the incidence of colorectal cancer, which is prominent in developed countries. Epidemiological studies had shown that drinking water with MCs content above 50 pg/mL increased the risk of colorectal cancer by 7.9 times [24]. The toxicities of MCs isomers are different, but the most widely distributed and most virulent is MC-LR, with an LD50 value of 47 μg/kg (subcutaneous injection in mice) [35,43]. Therefore, the limit of MC-LR in drinking water was set as 1.0 g/L by the World Health Organization (WHO) [44] and China's latest sanitary standard for drinking water (GB5749-2006) [45]. Moreover, MC-RR and MC-YR were also widely detected in water bodies around the world, among which MC-RR was the type with the highest proportion in most water bodies in China [46,47], but there is no corresponding control standard for it.

2.2. Analysis methods for MCs

The determination methods of MCs mainly include cytotoxicity detection, immunoassay, and chemical analysis, etc. Cytotoxicity detection is a method by using the effect of toxins on cells to detect toxins and quantifies them by establishing the relationship between MC concentration and toxicity. This method has a large workload and high cost. Immunoassay is a technology that has been gradually developed and improved in recent

years. Chu et al. prepared anti-MC-LR polyclonal antibodies in 1989 [48]. Since then, many scholars have used the enzyme linked immunosorbent assay (ELISA) method to measure the overall content of MCs to obtain the overall status of MC pollution. MCs of 0.05–1.00 ng/rat could be detected by ELISA, and the sensitivity was 1000 times higher than that of high-performance liquid chromatography (HPLC) [6]. In recent years, ELISA has greatly developed, and the problem of false-positives has been solved to a certain extent. It has been widely used in the detection of MCs in organisms and sediments. However, recognition of multiple homologs requires spectral antibodies. Chemical analysis methods include HPLC, HPLC combined with mass spectrometry (HPLC/MS), gas chromatography/mass spectrometry (GC/MS) and capillary electrophoresis, etc. HPLC is used more frequently, followed by GC/MS and GC. HPLC is a standard MC detection method recommended by World Health Organization (WHO) and many countries such as the United States, the United Kingdom, and China. It is accurate, sensitive and reproducible, and can simultaneously analyze different MC isomers. However, HPLC monitoring techniques often require standard toxins, and so far more than 240 MCs have been found, most of which lack standard toxins and limits the further application of HPLC. HPLC/MS can solve this problem well. As long as the molecular weight of the toxin is detected, it can be qualitatively determined, even if there is no standard toxin. LC-MS can be used to quantify the toxin accurately. A high-throughput on-line concentrated liquid chromatography-tandem mass spectrometry (LC/MS/MS) workflow developed by Birbeck et al. (2019) can be used for quantitative analysis of 12 MCs and nochlotoxin in surface water and drinking water with a running time of only 8.5 min and a quantitative limit of 5–10 ng/L [49]. Wu et al. established a solid phase extraction method combined with quadrupole-high resolution time-of-flight mass spectrometry to simultaneously detect multiple MCs in water, further improving the sensitivity and accuracy of the method [50]. Besides, Zhang et al. (2018) developed an electrochemical biosensor for the detection of MC-LR with a quantitative linear range of 4–512 ng/L and a detection limit of 1.4 ng/L, 700 times lower than the guidance level recommended by the WHO [51].

Due to the algal toxins from cyanobacteria, the effective extraction and determination of algal toxin in cyanobacteria cells are also the attention focus of scientists. The intracellular algal toxin was determined in the water sample, and the GF/C filter was used for filtration. The filter membrane that retained algal cells was folded and placed in refrigerator for freezing or freeze-drying. After repeated freezing and thawing, the cells were extracted for several times with appropriate solvent and combined with vortex, microwave, ultrasonic, and other means to better break the cells and release toxins. The most common solvents were methanol, water, acetic acid, and its mixtures. Lawton (1994) obtained a good recovery rate by extracting pure methanol at room temperature for 1 h [52]. The 75% methanol can better extract multiple polar algal toxins [53]. Hydrophobic toxins can be better extracted with 5% acetic acid [54]. For different types of cyanobacteria and cyanobacterial toxins in surface waters, multiple extractions with different solvent combinations can achieve better results [55,56].

2.3. Spatiotemporal changes and influencing factors of MCs

In recent years, the occurrence of toxic cyanobacteria blooms has become more and more frequent and serious worldwide. Many studies have shown that algal toxins are prevalent in surface water and can be detected at the intake of water plants. A two-year comprehensive survey of 26 sampling sites in France showed that the maximum concentration of MC-LR was up to 1000 μg/L[57]. MCs were found in 33 drinking water sources in the United States, with 7% of the sites exceeded 1 μg/L [58]. A survey of water intake in southeastern Québec City (Canada) showed that MC-LR was the main cyanobacterial toxin with a maximum concentration of 3.5 μg/L in raw water [59]. In the water pollution incident in Wuxi, China in 2004, the maximum concentration of dissolved MCs of 35 μg/L was detected in Meiliang Bay, Taihu Lake [60]. Bui et al. (2018) detected the content of MCs in the waters of southern Vietnam as high as 11,039 μg/L [61]. In the summer of 2018, Wan et al. (2020) investigated 30 subtropical lakes in eastern China and detected MCs in 28 lakes, with the highest average concentration occurring in Chaohu Lake (26.7 μg/L) [62]. In order to reduce the potential risk of MCs to human health, the monitoring and early warning of dissolved and particle MCs, especially the concentration and generation rule of MCs in the process of natural water bloom, have aroused the researchers' interest. In the past decades, a large number of references have reported the spatial and temporal variation of MCs concentration in different water bodies. These results show that the types and concentrations of MCs in natural water bodies vary significantly in time and space, not only in different lakes but also in different sampling areas of the same lake [63-67]. The toxicity of blooms is difficult to predict under natural conditions because the toxicity produced by the same species is not necessarily the same but always changes over time [66-69]. Therefore, it is necessary to study the change rule of MCs in lake water, especially in the specific water source area, to provide the scientific basis for the management of water sources and the production of waterworks.

Microcystin levels are affected by many factors in freshwater ecosystems. It is still not fully understood. In order to understand the rule of MCs under different water quality conditions, scientists isolated cyanobacteria and studied their toxigenic ability by culture in laboratory. The influence factors including temperature [55,70], illumination [71,72], phosphorus [73,74], nitrogen [70], trace elements [75,76], pH [77], culture time [78,79] and interaction with other creatures, such as bacteria, viruses, and predators [29,70], etc. However, the conclusions by different researchers are not completely consistent. It is generally believed under the condition of different culture produced by cyanobacteria toxin type and quantity is different, and toxin production was positively correlated with nitrogen concentration and low light density [70]. The highest toxicity and intracellular toxin levels were observed at the later stage of logarithmic growth [78,80]. Perez and Chu (2020) found that a strain of Microcystis aeruginosa was able to survive at 0.25 mg/L of ZnCl2 and that extracellular MC-LR concentration increased periodically. Therefore, the toxigenic

capacity of cyanobacteria is considered to be affected by multiple factors. Only by studies in laboratory, however, it is difficult to identify the dominant factors affecting the changes of toxins in natural waters. Under natural conditions, the dynamics of toxins are often more complicated and usually affected by several environmental factors. Therefore, only in-situ investigations can better reveal the changes, and the conclusions obtained may not be consistent with the conclusions in the laboratory. In Chaohu Lake, the key factors affecting the concentration and composition of MC isomers included Microcystis biomass, water temperature, and total phosphorus concentration [81]. From May to November 2005, research on the cyanobacteria population and the content of phycotoxins in Taihu Lake showed that an important reason for the changes of cyanobacteria toxins in different months was the changes in the cyanobacteria population structure. The concentration of several MC isomers was positively correlated with water temperature and chlorophyll-a (Chl-a), but negatively correlated with dissolved oxygen (DO), conductivity, total nitrogen (TN), and nitrogen/phosphorous (N/P) ratio. There was no correlation with water depth, pH, transparency and total phosphorous (TP) [82]. Although laboratory studies showed that phosphorus was beneficial to the growth of virulent strains but not to the growth of non-virulent strains, high phosphorus did not promote the increase in the concentration of virulent strains or toxins in Taihu Lake [83]. Sampling analysis of 14 sites in Taihu Lake in May 2009 found that MCs were not correlated with TN, TP and other environmental factors, although the number of cyanobacteria was significantly positively correlated with TN and TP. However, it was positively correlated with the content of bacteria, indicating that the contents of cyanobacteria and toxins were not consistent with the influence of environmental factors [84]. Meanwhile, a survey of 30 subtropical lakes in eastern China found that sufficient nitrogen and phosphorus concentration, light and stable water column conditions were crucial to the formation dominance of MCs-producing cyanobacteria and high MCs yield [62].

The proportion of toxigenic cyanobacteria also affects the concentration of algal toxins, and the proportion of toxigenic cyanobacteria varies from lake to lake. In Lake Wannsee [85] in Germany, Oneida Lake [86] in the United States, and Mikata Lake [87] in Japan, the proportion of toxin-producing cyanobacteria was between 0% and 40%. In the Grangent reservoir in France, the proportion of toxin-producing cyanobacteria was between 6% and 93% [65], while in another lake in France, the proportion was between 30%–80% [88]. The proportion of toxigenic cyanobacteria is affected by environmental factors such as nutrients, light, and temperature [55,87,89], and studies have also shown that the proportion of toxigenic cells is inversely proportional to the density of cyanobacteria [88,89].However, only 54% of the change in algal toxin concentration can be explained by the proportion of toxin-producing cyanobacteria, which is also affected by the toxin quality per unit cell, cyanobacteria number, and other factors [90]. Carmichael studied the changes of M. aeruginosa and the toxins in the lake and found that the change of the high concentration of aqua toxin was due to the relative number of toxin-producing M. aeruginosa in the lake, rather than the toxin-producing capacity [13]. The content of toxins in Microcystis was inversely proportional to the cell density, indicating that the toxicity was reduced at the peak of bloom [91]. In addition, the rapid change of toxins in blooms of natural water may also be due to the change of algae species or the change of the same cyanobacteria toxigenic strain and non-toxigenic strain [85,92]. The influence of environmental factors on algal toxin concentration may be caused by changing the number of toxigenic genes. In natural environment, adverse growth conditions (e.g., pre - and post-bloom) lead to an increase in MCs synthesis genes [65]. Rising temperatures contribute to the proliferation of cyanobacteria and increase the secretion and release of toxins, which have been demonstrated in laboratory and field studies. Laboratory culture experiments suggested that Microcystis at 26°C–28°C showed maximum growth rate, and the investigation of many lakes around the world also proved toxic algae cells and toxins in the warm season increased [55,93,94]. There could be caused by the synergy of the temperature and other environmental factors, or due to higher temperatures increasing the algal toxin genes numbers [55,93,94], which implies that global climate change might affect the algae proliferation and release characteristics of algal toxins. Studies on the gene expression of toxins in M. aeruginensis under low nitrogen concentration (1.4 mg/L) found that nitrogen restriction would lead to the decrease of MCs synthesis gene, which would further lead to the decrease of MCs production per unit cell [95]. The environmental factors influencing MCs in different cyanobacteria bloom areas are very changeable. Studies on the influencing factors of algal toxins are conducive to the extraction of key factors in the studied areas, to take corresponding control measures to reduce the content of algal toxins in water and reduce the ecological and health risks.

2.4. Removal technologies for MCs

The removal of MCs should be focus on both intracellular and extracellular toxins. The removal of intracellular MCs is mainly through a series of means to remove cyanobacteria cells. Due to the small volume of cyanobacteria, it is difficult to remove them by physical isolation in general waterworks, and there may be potential accumulation in the process of waterworks. Such accumulation is affected by the cell volume, electric charge, motility, morphology, resistance to shear force and pressure, and other factors caused by population differences. Micro-screening is a technology used to screen cyanobacteria cells, with the efficiency of screening depending on the size of the screening hole. For instance, the screening hole at 35 μm can trap 40%–70% of phytoplankton but less than 10% of cyanobacteria (including M. aeruginosa) [96]. Additionally, many studies focused on the effects of different flocculants on cyanobacteria cells, and the results showed that the removal efficiency of flocculation and precipitation on cyanobacteria was from 62% to 98.9% [7,97,98]. However, these experimental results were observed under low concentration of cyanobacteria cells, which could not represent the removal efficiency in cyanobacterial blooms [7,97,98]. The removal efficiencies of oxidants such as ozone, chlorine, potassium permanganate and chlorine dioxide were also studied. Results showed that oxidants could lead to cell rupture and further oxidation of substances released from the cells. The release rate of toxins was decided by the extent to which

the oxidants destroyed the integrity of cell membranes [99-103]. The removal of cyanobacteria cells by oxidants is influenced by the species of cyanobacteria, the physiological state of the cells (such as growth stage), and the amount of oxidants added, while the oxidation of the released toxins depends on the type, concentration, water quality and oxidative exposure time of the toxins [99-103]. Since part of the algal toxin exists in cyanobacteria cells and can break down under certain physical pressure to release algal toxins, we should pay attention to the complete removal of cyanobacteria cells in the study of treatment technologies.

Many literatures have studied the removal of dissolved MCs by various treatment processes and established relevant experimental operating conditions. It mainly include: (1) Physical methods such as adsorption [104,105], filtration [106], air flotation [107] and ultrasonic [108], etc.; (2) Chemical methods such as enhanced coagulation [109], chemical reagent oxidation (chlorination [110], potassium permanganate [111], ozone [112] and hydrogen peroxide [177], etc.), deep oxidation (Fenton oxidation [113], photocatalytic oxidation [114], electrochemical method [115], etc.; (3) Biological method [116,117], etc. Besides, some studies considered using multiple process combinations to remove MCs. For example, Lopes et al. [118] and Zhang et al. [119] found that Fenton oxidation/activated carbon adsorption and ultraviolet (UV)/ chlorination combined could effectively remove MC-LR. However, in the actual production, the unit water purification process is difficult to reduce the algal toxin in the water to a safe level. Therefore, it is necessary to control the toxins in raw water step by step by optimizing different processes. Meanwhile, according to the actual situation, the traditional water treatment process was modified, such as using combined technologies including potassium permanganate pre-oxidation enhanced coagulation (in severe cases, powder activated carbon can be added at the same time) + air flotation + sand filtration (or granular activated carbon filtration) + chlorine disinfection [120]. The newly-built waterworks with high requirements on effluent quality can adopt ozone activated carbon deep treatment process or add microfiltration membrane treatment unit before the conventional process, which can effectively improve the removal efficiency of algal toxin in waterworks [120]. It is notable that most of the studies on removal of cyanobacterial cells and cyanobacterial toxins are confined to the laboratory and small- or medium-scale experiments,in which the effects of water quality parameters and cyanobacterial population can be quantified. But it is difficult to represent the actual operation of water plants. Most literature suggests that traditional water plant processes were inefficient in the removal of algal toxins, for example, in the monitoring of effluent samples from a DWP associated with Chaohu Lake, concentrations of MC-LR in summer, and MC-RR and MC-YR in autumn exceeded 1.0 μg/L [121]. Jurczak et al. [122] studied the actual processes of two water plants in Poland and found that although the maximum total concentration of MCs (MC-RR, MC-LR, and MC-YR) in raw water was detected at 6.7 μg/L on 13 August 2002, MCs in discharged water could be reduced below the drinking water limit (1.0 μg/L) after treatment. However, the results that studied the removal rules of cyanobacteria cells and toxins in actual water plants showed that some experimental results are inconsistent or even contradictory with the conclusions drawn in the laboratory [59]. The optimal dosage obtained by the laboratory can only be used after verification in the actual water plants [59]. Therefore, it is of great practical significance to study the removal of cyanobacteria and associated toxins in each process stage of the water plant, which can provide a scientific basis and technical advice for the water plant management.

3. Research progress on T/O compounds

In recent years, the problem of T/O in natural water bodies and drinking water has occurred frequently and attracted wide attention [123-130]. There are few reports about the harm caused by tastes and odors to human health, which is the direct and main factor used by consumers to identify the water quality [16,131-133]. Reports on tastes and odors in water can be traced back to the 1950s [134]. Monitoring and control of T/O compounds have become one of the focuses of water quality research. With the development of urban construction, industrial and agricultural production, a large number of untreated sewage and wastewater are discharged into natural water bodies, resulting in more and more reports on the peculiar smell of drinking water [134,135]. The eutrophication of water bodies has gradually become another important cause of odor problems, in which the high concentration of nutrients resulting in the excessive growth of some algae, bacteria, and microbial communities can continuously secrete and produce various secondary metabolites with the peculiar smell. This brings a series of problems, including affecting water quality, increase the cost of water treatment, cause peculiar smell in aquatic products and reduce the aesthetic value of tourist areas [130,135-137]. Therefore, the generation of T/O compounds from algal sources in eutrophic water bodies has attracted extensive attention.

3.1. Classification and source of flavor in water

Flavor in water includes odors, tastes and mouthfeel, which is a comprehensive sensation of nerve stimulation on the tongue, nose, and mouth caused by some chemicals in water [134]. Based on long-term accumulated data, scientists have drawn a Taste and Odor Wheel for odor analysis, which includes eight classes of odors, four tastes, and a mouth feel/nose feel category [138]. From the perspective of drinking water production process, T/O compounds may mainly come from three processes: (1) the olfactory substances contained in the raw water itself; (2) when the raw water is treated by the water plant, the added chemicals (e.g. chlorine and ozone) and the substances produced by the reaction with the raw water substances bring about tastes and odors; (3) when the treated water is transported to users through the water distribution system, impurities introduced in the pipe network system produce smell and taste [134,136]. Besides the first part was focused on, there have been more studies on the odor produced by disinfection and the odor problem in the pipe network. The sources of odorous substances in natural water bodies mainly include natural sources such as minerals precipitated from soil and rocks and substances produced by human activities. The latter is the main source of odorous substances in water, including the direct discharge of olfactory compounds (phenolic compounds, etc.) and decomposition of

organic matter and some microbial metabolites of Actinomycetes, fungi, and algae (such as GSM and MIB) [134]. Along with the development of industrial and agricultural production, nitrogen and phosphorus input lakes and reservoirs causing serious water eutrophication and frequently cyanobacteria blooms outbreak. As the algae metabolites are important sources of tastes and odors, attention should be paid to the changes and removal of algae-derived odorants in the production process of tap water [9,121,139].

3.2. Analysis methods for T/O compounds in water

The composition of odorants in water is very complex, and the odor threshold concentration (OTC) of odorants is extremely low, generally at the ppb level. Therefore, it is difficult to odorize substances qualitatively and quantitatively, which restricts the development of related research on the key issue. Methods for the analysis of T/O compounds in water can be divided into three categories: sensory analysis, instrumental analysis, and comprehensive analysis [140-142]. Sensory analysis means that T/O compounds enter the nasal cavity through smelling, thereby stimulating the nervous system, causing the system to generate olfactory signals, transmitting the signals to the brain, and determining the type and intensity of odors by people, which includes threshold odor number, flavor rating assessment and flavor profile analysis (FPA) [140,143]. Although the FPA method has high requirements for olfactory sensitivity and training methods for sniffers and requires a long time of training before it can be truly analyzed, it becomes one of the routine water monitoring projects in most countries and regions abroad due to the good qualitative, sensitive and classification characteristics [144]. Instrumental analysis is based on the color, luminescence, and ionization principles of the reaction products of T/O compounds, and it is analyzed by gas chromatography (GC), GC-MS, colorimetry, chemiluminescence, and other methods [141]. Among them, GC using flame ionization detector and mass spectrometry detector is commonly used. With the development of analytical techniques, especially use some new methods of pretreatment, such as steam distillation extraction method, liquid-liquid extraction method, purge and trap technique, closed-loop stripping analysis, opened-loop stripping analysis, solid-phase microextraction (SPME), a breakthrough has been made in the technique of measuring T/O compounds in water with instruments [145-148]. Among them, SPME is widely used in food, environmental safety, and other research fields because it does not need organic solvent and concentration, and is simple and easy to operate. The SPME is used to enrich trace odor compounds in water with the top space, followed by the combination with GC/MS for analysis. It greatly improves the sensitivity and becomes a common method for detection of T/O compounds [149-151].

The most commonly used method for comprehensive analysis is to analyze and test the sample water with the FPA method further to preliminarily determine the substances that cause the T/O in the water. If necessary, chemical or instrumental analysis can be used for further identification and quantitative analysis to improve work efficiency [134]. Another method is to conduct a model analysis of the main water quality indexes of a specific water body and try to establish the relationship between water quality indexes and the concentration of T/O compounds to achieve the purpose of rapid and accurate evaluation of water smell [125]. Scientists have also tried to detect T/O compounds by ELISA, which is widely used in the field of life science. It is a new detection method developed based on the principle of antigen-antibody reaction [142]. However, since the relative molecular weight of odor compounds is generally less than 300, it is not easy to obtain high-efficiency antibodies. Devi et al. (2020) proposed the application of molecular biological methods for the early detection of GSM and MIB events, providing a new idea for the monitoring and early warning of T/O compounds [152]. In addition to the determination of dissolved T/O compounds in water, the determination of these compounds in algal cells is also of great significance. Firstly, dissolved and intracellular T/O compounds were separated by filtration, and algal cells were trapped on the GF/C (Whatman) glass fiber filter membrane or 0.45 μm acetate fiber filter membrane. Then the cells were broken by freeze-thawing, grinding, or ultrasonic cell crushing apparatus, after which the T/O compounds were released into ultra-pure water for determination [126,127]. In addition, the difference subtraction method has been used to calculate the intracellular T/O compounds: the alga solution was directly determined as the total odor in content, while the dissolved (extracellular) odor in content was determined after filtration by 0.45 μm acetate fiber membrane, and the difference between the two was the intracellular T/O compounds content [153]. However, this method needs to pay attention to whether the alga cells are completely broken or not. In practical operation, the intracellular T/O compounds can freeze and thaw the cells trapped on the GF/C membrane, and then release them into ultra-pure water for instrument detection under the condition of an ice bath with ultrasonic cell crushing apparatus [121].

3.3. Sources and properties of algal-derived T/O compounds

The study on odor metabolites produced by microbial metabolism originated from GSM isolated from Actinomycetes for the first time in 1964 [154]. Subsequently, in 1969, Medsker et al. reported firstly that several Actinomyces in natural water could release MIB [155], while Rosen et al. successfully isolated MIB in 1970 [156]. Later, it was found that algae could also produce these two odorants, mainly including Phormidium, Oscillatoria, Aphanizomenon, Lyngbya, and some species of Anabaena [137]. With the deepening of the investigation, it was found that as many as 200 kinds of algae could produce T/O compounds, among which cyanobacteria could produce 25% of the known compounds [16].

As the T/O compounds produced by algae, GSM and MIB, which give off a musty smell, are the most frequent and most widely studied in the world. Both of them are terpenoids, which are difficult to be removed by oxidation process in waterworks. They are also the main substances of odor events due to their low olfactory threshold. It has been found that 132 cyanobacteria strains from 21 genera and 72 cyanobacteria strains from 13 genera could produce GSM and MIB, respectively[152]. In addition, two kinds of T/O compounds produced by cyanobacteria are also being concerned: fermentative production of hydroxyketones, and norcarotenoids such as β-cyclocitral (CYC) and β-Ionone (ION), which

are produced by the degradation of carotenoids. CYC is produced by globule cyanobacteria, especially Microcystis, which is the main odor substance produced by M. aeruginosa. In China, CYC has been paid more and more attention. In the cyanobacteria bloom outbreak in Wuxi in 2007, CYC was one of the main causes of olfactory smell [17]. Zhang et al. first detected ION in Songhua Lake, a lake in northern China, and its highest concentration exceeded the olfactory threshold [157]. In Chaohu Lake (China), CYC and ION exceeded their corresponding OTCs in 72% and 86% of the samples, respectively [121]. Therefore, we summarize the sources, spatial and temporal changes, impact factors, and removal of the above four algal derived T/O compounds in water.

3.4. Spatial and temporal variations of algal-derived T/O compounds

Climate has an important influence on the occurrence and concentration of algal-derived T/O compounds. In areas with four distinct seasons, odors change with temperature, physical and chemical properties of water, and the generation and disappearance of algae. The algal blooms in summer would inevitably cause the outbreak of T/O compounds, and the decline of cyanobacteria in autumn also increased their concentrations. Li et al. investigated the T/O compounds in Yanghe Reservoir (China) in the summer of 2007 and recorded the concentration of GSM reached the peak value (7,100 ng/L) on the day after the maximum density of Anabaena observed [67]. In addition, other high concentrations of GSM in natural water bodies were also reported in Australia (4,000 ng/L) [158] and South Africa (3,170 ng/L) [159]. Although in other seasons concentration of algal-derived T/O compounds is relatively low, but they also can not be ignored due to their extremely low smell threshold (e.g. ng/L). Many studies have investigated the spatial and temporal variation of T/O compounds in different types of lakes. Westerhoff et al. studied the characteristics of GSM and MIB in three deep-water lakes in Arizona, the United States, and found that the overall concentration of GSM was lower than MIB, but the two presented the same seasonal changes [160]. The concentrations of MIB increased from spring to late summer and vertically stratified in water with the highest concentration at 10 m underwater [160]. However, the particle and dissolved concentrations of the four T/O compounds GSM, MIB, CYC, and ION in Dianchi Lake (China) varied in different seasons, although the concentrations of the T/O compounds in Dianchi Lake peaked in summer and autumn in general [126].

During the study period of May 2007 to April 2008, GSM in Xionghe Reservoir (China) appeared from May to September, with the highest concentration detected in July (about 2,700 ng/L), while MIB was detected throughout the study period with the highest concentration in April (about 400 ng/L) [161]. Ma et al. monitored the T/O compounds at 15 points along the northern coast of Gonghu Bay, Taihu Lake from January to December 2009 [162]. It was found that both dissolved and particle-bound GSM peaked in April and September, with the annual average concentrations all lower than 4 ng/L. Dissolved MIB peaked in April and September, with a maximum value of 395.6 ng/L (September), while particulate MIB peaked in September, with a maximum value of 29.8 ng/L [162]. Additionally, the maximum values of dissolved and particulate CYC were recorded in October (17.7 ng/L) and September (2,072 ng/L), respectively. No dissolved ION was detected, while the concentration of particulate ION was higher from July to October, with a maximum of 573 ng/L in August [162]. From June 2009 to May 2010, sampling and analysis of odor substances from 30 points in Taihu Lake showed that the concentration of dissolved CYC in winter was much lower than that in late spring and early summer, with the highest concentration of 49 ng/L, while the concentration of particulate CYC in winter and spring was lower than that in summer and autumn, and the maximum value was 2155 ng/L [163]. The trend of concentration changes of particulate ION and particulate CYC was the same, but the dissolved MIB was greater in spring and autumn than in summer and winter with the highest concentration of particulate MIB appeared in July [163]. Therefore, we get the following conclusions: 1) The seasonal changes of T/O compounds concentration in lakes of different regions are varied. High concentration usually occurs in summer and autumn when cyanobacteria bloom is severe, but some lakes also have peak values of T/O compounds in spring and winter; (2) The seasonal trend of different odor substances changes annually; (3) The intracellular and extracellular concentration of the same odorant substance varies seasonally. These characteristics increase the difficulty for the supervision of T/O compounds and the removal strategy of waterworks. Therefore, it is necessary to strengthen the monitoring and grasp the changing trend of the concentration of T/O compounds in the water source area, , which can better take corresponding measures to ensure the drinking water safety.

3.5. Influencing factors for T/O compounds from algal sources

The distribution of algae-derived T/O compounds in water varies in different regions and seasons. Scientists have studied the environmental and biological factors that affect the release of T/O compounds in the laboratory and natural water. In a survey of 296 rivers and lakes in Japan, Hashimoto found that musty odor was related to TN and phosphorus [164]. The 67% of rivers and lakes with odor problems with TN and TP higher than 0.6 and 0.03 mg/L, respectively, while only 8% of rivers and lakes were affected by odor with TN and TP lower than at 0.3 and 0.02 mg/L, respectively [164]. Li et al. studied the influencing factors of dissolved and particulate T/O compounds in Dianchi Lake (China, June 2002–May 2003) [126]. The results showed that there was a significant positive correlation between particulate CYC and particulate ION, and a significant positive correlation between Chl-a and the biomass of algae, cyanobacteria, and Microcystis. Particle MIB was positively correlated with the biomass of Oscillatoria and Anabaena respectively, and these two species were proved to be the sources by the experiment of species isolation [126]. However, dissolved T/O compounds had no significant correlation with the physicochemical properties of algae and water, which may be due to the fact that dissolved T/O compounds were affected by many factors, such as algal release rate, microbial biodegradation, photodegradation, particulate matter adsorption, hydrological conditions and their volatility [126]. The spatial and temporal distribution of T/O compounds were investigated in Gonghu Bay, Taihu Lake. The results firstly indicated that particulate CYC and particulate ION were significantly positively correlated with algal toxin, and believed that nitrogen was a key factor

affecting the occurrence of odor, because nitrate, TDN, and TN were significantly positively correlated with GSM [165]. Laboratory studies on two cyanobacterial strains Phormidium retzii and Microcoleus vaginatus, producing GSM and MIB, respectively, showed that odor substance production was independent of temperature and light [166].

As the occurrence of T/O compounds is affected by water quality and other factors, scientists try to establish a prediction model according to the relationship between these compounds and physicochemical and biological factors of water. Smith and Downing found that in eutrophic lakes, cyanobacteria, odorant MIB and GSM were positively correlated with Chl-a, indicating that the nutritional status, especially Chl-a, was a good indicator of odorant [167,168]. Therefore, some national regulatory authorities have established the Chl-a concentration as the standard to predict the occurrence of odor problems in lakes[169]. On a larger space-time scale, Watson et al. found that the outbreak of smell events was related to a series of physical chemistry and indicators in water [170]. Dzialowski established a more complex early warning model of GSM, which included a series of water quality parameters [171]. They suggested that Chl-a alone cannot predict the occurrence of odor events, because the concentration of GSM was also high under low nutrients and Chl-a conditions. Hence it was implied that the limit of inorganic phosphorus in the water and other environmental factors were more critical for predicting the concentration of T/O compounds. Watson et al. also found little change in nutrients and Chl-a over eight years in western Ontario, but a significant increase in GSM [170]. Moreover, some studies have found that the peak of GSM occurred in spring and even winter [172,173]. So it can be considered that even though temperature and nutritional conditions were the most important indicators affecting odors, but they were not the best indicators. Due to the outbreak of cyanobacteria in lakes and reservoirs worldwide, odor problems often occur and affect landscape ecology and drinking water safety. Therefore, strengthening the study of the spatial and temporal variation of T/O compounds and their influencing factors will be helpful to provide technical guidance for the control of cyanobacteria in drinking water source and the production of the corresponding water plant.

3.6. Removal technologies for T/O compounds

The T/O compounds in raw water would cause peculiar smell of tap water if they cannot be effectively removed after entering the water plant. A study of 59 water plants in the Great Lakes basin of United States, found that about 20% of them had odor problems every year [174]. In China, the smell problem of tap water is also very serious. The T/O problem in Taihu Lake and Chaohu Lake (China) has led to the temporary production of water plants, especially the water pollution incident in 2007, which sounded the alarm for the management of water plants [163,175]. The monitoring of T/O compounds in raw water and the effluent of 111 water plants in China showed that the raw water with mold flavor accounted for 41% [176]. Among them, the plants with lakes and reservoirs that are easy to grow algae as the raw water, MIB was positively correlated with mold flavor [176]. The raw water with a putrid taste accounting for 36% was used as the source water [176]. Moreover, the proportion of T/O compounds detected in the finished water was 45% [176]. Although most researches have focused on the removal of GSM and MIB, researches on the removal of CYC and other T/O compounds are gradually increasing in recent years. Conventional treatment technology is difficult to remove the T/O compounds effectively. Only 23% of the GSM was removed by the process of pre-chlorination-coagulation-contact filtration in Wuhan Tuanshan water plant in China, and its concentration in filtered water was 72.85 ng/L [177]. Kim et al. reported that the removal rate of GSM by conventional treatment process was 33.3% [177], while the removal rate of MIB in a water plant in Taiwan was about 40%–50% (the concentration of raw water was 30–70 ng/L), in which 16% and 40% were removed after precipitation and filtration, respectively [178]. Slow filtration (filtration rate of 4-5 m/d) was adopted to remove T/O compounds with the help of biofilm surface adhesion and microbial degradation [179]. Its effect was much higher than that of the fast filter, and the effluent MIB was below the detection limit [179]. However, due to the low filtration rate, it is not suitable to be used in large water plants [179]. In addition, it was found that aluminum-containing flocculants could not effectively remove GSM and MIB under various pH values and coagulation conditions [180]. The oxidants such as chlorine (Cl2), chlorine dioxide (ClO2), and potassium permanganate (KMnO4) also had poor removal effect, but ozone (O3) could remove 85% of MIB at the concentration of 3.8 mg/L and contact time of 6.4 min [181].

Activated carbon is the most commonly used method to remove T/O compounds in water plants, which mainly uses the physical adsorption of activated carbon and the principle of microbial degradation. Activated carbon is easy to adsorb benzene compounds in water, while the structural formula of GSM and MIB is similar to the benzene ring, which has a good adsorption effect. Therefore, many water plants adopt the temporary method of adding powdered activated carbon (PAC) to remove the smell of water [179]. Kim et al. found that it was very difficult to control GSM only through PAC treatment, and the removal efficiency could be increased to 94% by combined treatment of PAC and chlorination [182]. Zamyadi et al. conducted a study on a DWP and found that the removal degree of GSM in the water plant process was higher than that of MIB because that biofilms were attached to the surface of the GAC could improve the removal efficiency of T/O compounds [183]. Bai et al. found that after flocculation and precipitation, the concentration of deodorant could be reduced to less than 10 ng/L by using 1.0 mg/L •OH and 0.5 mg/L sand filtration, and the concentration of DBPs in the effluent also met relevant standards [184]. Due to the great difference of odor concentration in raw water, the conditions should be controlled according to the actual operation of the water plant, the method of multi-process combination should be used to improve the efficiency, with the removal of other metabolites such as toxins should be considered at the same time.

3. 6.1. Research progress focus on algogenic organic matter

Natural organic compounds (NOM) are widely found in all kinds of water bodies with complex compositions. Hydrophilic, acidic and polydispersed humus are the main constituents (50%–90%) of NOM. Humus substances

can be divided into three types: humic acid, fulvic acid, and humin, which have a similar structure but the great difference in molecular weight and functional group. Humus contains phenolic hydroxyl, carboxyl, alcohol-hydroxyl, and other functional groups, and the molecular weights are between 102–106, among which carbohydrate and its associated substances account for 50%–60%, lignin and its derivatives 10%–30%, and protein and its derivatives 1%–3% [185]. The strong hydrophilic and lower aromaticity of protein, adipose, amino acids, carbohydrates and hydrophilic acid constituted the main part of the easily biodegradable organic matter in water [186]. With the intensification of eutrophication, algae and its secretions (algogenic organic matter, AOM) have become an important part of NOM in many surface water systems [187,188]. The contribution of algae to NOM is related to different water environments [189]. Algal contribution to NOM is more significant in freshwater systems because the NOM mainly imports from runoff in the watershed [190,191]. In general, NOM is mainly affected by algae in large or high primary productivity lakes and reservoirs, while small or low ones are mainly affected by land inputs [192]. In eutrophic lakes, the composition of NOM and the proportion of its components tend to change with the seasonal variation of algal species and biomass [193]. Even in some cases, when the total amount of dissolved organic carbon (DOC) did not change significantly from season to season, each composition also showed drastic varieties. AOM is products released by algae during the growth process, mainly including proteins, neutral or charged polysaccharides, nucleic acids, fats, and other small molecular substances, of which polysaccharides account for 80%–90% [194,195]. AOM included intracellular organic matter (IOM) that existing within the cells and extracellular organic matter (EOM) releasing to extracellular. The complex composition of AOM, molecular weight (MW), DOC concentration, and content of sugars and uronic acids are related to the species and culture time of cyanobacteria [196,197]. The following is a review of the research progress of AOM based on the detection methods and characteristics.

3.6.2. Detection method for AOM

Except for substances that can be directly qualitative and quantitative, such as MCs and T/O compounds, most AOM cannot be directly qualitative and need to be detected by other means. The technical methods for determining the characteristics of AOM include elemental analysis, UV-vis spectrum, fluorescence spectroscopy, polarity analysis, affinity chromatography, high-performance molecular exclusion liquid chromatography, pyrolysis-GC/MS, and nuclear magnetic resonance (NMR) analysis, etc. Different methods can obtain information such as spectral absorption characteristics, functional group content, and so on.

4. T**he following three methods are commonly used:**

4.1. Organic carbon and organic nitrogen analysis

The concentration of organic matter in water is usually expressed by total organic carbon (TOC), including DOC and particulate organic carbon. Total organic nitrogen (TON) also includes dissolved organic nitrogen (DON) and particulate organic nitrogen (PON). The C/N ratio represents the relative content of C and N in organic matter, which can be used to identify the source of organic matter [198].

4.2. Ultraviolet spectrum analysis

The absorption of UV light varies in organic compounds with different molecular structures. The conjugated structure of aromatic compounds containing benzene rings is generally the most absorbent in the UV range of 230–280 nm, and the molecular structure and its relative content or the source of organic matter can be identified to some extent. Several commonly used indicators include characteristic absorption values at a certain wavelength, such as UV254 and UV280, which can reflect the functional groups of AOM and the degree of aroma [199]. The ratio of characteristic absorption value to TOC or DOC, such as specific ultraviolet absorbance (SUVA)254 value (=UV254×100/TOC, unit: L/mg•m), has a good correlation with the amount of aromatic functional groups measured by NMR [148]. SUVA value is a substitute indicator of the aromatic structure of organic matter, which can reflect the aromas of water to some extent. The ratio of absorption value at a specific wavelengths, such as UVA210/UVA254 (URI value), can represent the relative content of amino and aromatic structure to some extent. A larger URI value indicates more amino structure content, while a smaller URI value indicates more aromatic unsaturated structure content [200].

4.3. Fluorescence spectrum analysis

Fluorescence spectrum is a qualitative or quantitative description of organic compounds based on NOM or AOM containing a large number of structures with various functional groups. Among them, the fluorescence excitation−emission matrix (EEM) shows the fluorescence intensity at different excitation and emission wavelengths in the form of a three-dimensional projection diagram. The independent variable excitation wavelength and emission wavelength are independent of each other, and the fluorescence intensity is used to show the characteristics of the sample fluorescence spectrum [201]. It has the advantages of high sensitivity, good selectivity, no damage to the sample, less sample consumption, and fast sample measurement speed. In recent years it has been widely applied to the characterization of organic matter in water. It is used to analyze NOM or AOM fluorescence characteristics and the relationship of the chlorination DBPs generation, as well as a variety of water technology on organic matter removal process [201-203].

4.4. Characteristics of AOM

Compared with land-based organic compounds, AOM have a lower content of hydrophobic substances, aromatic hydrocarbons, phenol, etc. Organic nitrogen content is higher, which is characterized by high hydrophilicity, low aromatic hydrocarbons (low SUVA value), and high heterogeneity [200]. AOM has a high nitrogen content, mainly because of their cell walls containing N-acetyglucosamine and N-acetylmuramicacids, and other nitrogen compounds. Algal cells also contain a lot of organic nitrogen, such as protein, amino acid, and an organic amine, and can produce nitrogen secondary metabolites, such as MCs and terpenoids flavor substances such as GSM and MIB [204]. About 45% of TN fixed by

cyanobacteria can be converted to organic nitrogen [205]. Studies showed that the order of TOC/TON ratio was NOM >> EOM > IOM ≈ algal cells [20]. Therefore, it is considered that the organic matter in water with a low C:N ratio mainly comes from AOM. Besides, the values of SUVA and URI can represent the properties of NOM and AOM. The results showed that the SUVA value of IOM of M. aeruginosa was lower than that of EOM, and both of them were far lower than NOM. It indicated that the content of aromatic structure in the three was IOM < EOM < NOM [20]. Humic acid has the most aromatic structure, with a URI value of 1.59; followed by fulvic acid with a URI value of 1.88; and the least aromatic structure is bovine serum albumin, with a URI value of 13.50 [200].

The nature of AOM is related to the growth stage of algae with the content of proteins, sugars, and lipids varying in different growth stages of algae, especially in the process of EOM release [206]. Extracellular organics are released in two forms, one is the equilibrium diffusion of intracellular and extracellular, the other is the irreversible release after the damage or death of algal cells. In the first form, intermediate metabolites with low molecular weight, such as glucose and amino acids, are released through the cell membrane from high concentration to low concentration to achieve a certain balance of intracellular and extracellular concentration. They are released as extracellular organic substances and occur mainly at the early stage of growth [186]. In this process, extracellular organics are mainly intermediates with low molecular weight, such as alcohol and amino acids, 90% of which are polysaccharides and a small amount of proteins, amino acids, and trace nitrogen-containing organics [195]. Experiment showed that the algae concentration dilution was caused by the increase of EOM by diffusion and release [186]. Due to the algal densities in natural water bodies are generally below that of laboratory cultures, the release of EOM dynamics are following the balance concentration diffusion characteristics in nature [186]. The second release mode mainly produces substances with higher molecular weight from the surface of senescent cells, which are produced in the later stage of cell growth. Compared with natural water, more EOM is released from decaying cells under laboratory culture conditions due to high cell density [186]. Proteins in EOM increased with culture time. When cells decay, IOM precipitates out in large quantities from cells, and the EOM reached maximum [21]. The properties of AOM are affected by algae species. Studies have shown that different algae species produce various DOC yields per unit biomass. Diatoms and cyanobacteria produce DOC > 20 mg/L, while green algae produce DOC ranging from 10 to 12 mg/L under experimental conditions [207]. According to the DOC generation rate per unit optical density, the lowest rate of Chlorophyta was 2.1 mg/L/(cm•h), while that of cyanobacteria and diatoms was 3.8 and 21 mg/L/(cm•h), respectively. In terms of the DOC yield per unit of Chl-a, the highest yield of cyanobacteria was 9.0 g C/g Chl-a/h, and that of Chlorophyta and diatom were 3.6 and 1.1 g C/g Chl-a /h, respectively. The proportion of proteins and lipids was also different. The protein content of cyanobacteria was 41%–69%, which was higher than that of diatoms (12%–50%) [208]. Diatom lipids were 5%–43% higher than those of cyanobacteria (2%–30%) [208]. In general, the concentration of EOM increased continuously with the growth of culture time, and Microcystis released more than Anabaena, reaching 2.26 mg/L before the release of IOM in the decay phase [21].

4.5. AOM is one of the precursors of DBPs

The presence of AOM affects the unit operating efficiency in the water treatment process by increasing the amount of coagulant and reducing the coagulation efficiency [209], and it is difficult to be filtered and removed [210]. Additionally, carbon and nitrogen in AOM may require more chlorine for pre-oxidation or disinfection to reduce the effectiveness of chlorine and generate DBPs [211,212], and even lead to the growth of bacteria in the pipe network [213]. Rook [214] and Bellar and Kroner [215] detected chloroform and other trihalomethanes (THMs) firstly in chlorine disinfected tap water, which was also the earliest DBPs detected. Nowadays, data from drinking water monitoring and laboratory test show that disinfection with chlorine or other disinfectants such as chloramine, ozone, and chlorine dioxide can produce more than 600 types of DBPs [216,217]. The sources of DBPs precursors and the influencing factors for DBPs generation are summarized below.

4.5.1. Source of the DBPs precursors

The precursor of DBPs generally comes from humus in water, and its components mainly include hydrophilic acids, sugars, carboxylic acids, and amino acids [218,219]. In recent years, algal cells and their metabolites have also become an important source of DBPs with the appearance of algal bloom [19,21,22,220]. Studies on the formation of DBPs of humus in water showed that humic acid with higher molecular weight had more active sites than fulvic acid with the THM yields of 58.6 and 42.6 μmol/mg C, respectively [221]. Hong et al. conducted a DBPs generation experiment using bovine serum albumin (BSA), starch, and fish oil as simulated substances of proteins, carbohydrates and lipids respectively [22]. The results showed that the amount of chloroform generated by BSA and fish oil was 9 times higher than that of starch. However, BSA was the most susceptible to HAA production, which was 9 times higher than that of fish oil and starch [22]. Proteins such as BSA, pepsin, chymosin, and cytochrome could form THMs in large quantities (20–51 μg/mg C), but in smaller quantities than humic acid (78 μg/mg C THMs) [222]. Carbonaceous DBPs (C-DBPs) can be generated by algae and AOM as precursors, and THMs yield was 0.23–3.20 μmol/mg C [223]. Nitrogenous substances in AOM are important precursors of DBPs, which can generate C-DBPs and nitrogenous DBPs (N-DBPs), and affect the types of DBPs generated.

Studies have shown that in the chlorinated experiment of amino acids, the order of chloroform production was tryptophan, tyrosine, histidine, aspartic acid, threonine, lysine, alanine, and serine [224]. The chlorination of amino acids produced unstable intermediate dichloroacetonitrile (DCAN) and continue to react to form THMs and HAAs [225]. Because proteins in AOM are composed of different amino acids, they have different DBPs generation potentials. As a precursor, DON derived from algae can also generate N-DBPs (such as halogenated acetonitrile, halogenated acetamide, halogenated nitromethane, and dimethyl nitrosamine, etc.)

[217,226]. Cyanobacteria have also been shown to be precursors for haloacetonitriles (HANs) and trichloronitromethane production [227]. Nitrosamines, especially dimethylnitrosamines, could also be generated during chlorination or chlorination of AOM, while dimethylnitrosamines were more generated during chlorination because chloramines provide more pathways for its generation [228]. Due to the high nitrogen content in AOM, N-DBPs produced by nitrochlorinated substances are more toxic than C-DBPs [227,229,230], which may be more harmful than C-DBPs although the reported concentration of N-DBPs was relatively low [231]. Other small amounts of organic matters such as deoxyribonucleic acid, ribonucleic acid, organic acids, and Chl-a in algae are also progenies for HAAs [232]. The difference in HAAs generated by cyanobacteria and Chlorophyta may be due to Chl-a and b in Chlorophyta but only Chl-a in cyanobacteria [233]. DBPs generated in the chlorination experiment of algal cells may be generated by the reaction of chlorine with the intact cell wall or with the dissolved IOM, so it is difficult to identify whether the cell wall is the precursor of DBPs [21]. In eutrophic lakes, both exogenous and endogenous organic matter are the precursors of DBPs, and the nature of the organic matter in the water source water changes due to the source. Exogenous organic matter mainly includes humus washed by rainwater or brought by rivers in the basin, with complex composition, slow conversion rate, and long existence time [234]. The endogenous sources are mainly algal biomass, extracellular metabolites, and products released or transformed by cell death, which are more unstable in nature, related to biological cycles, and generally only account for a small part of the TOC pool [235,236]. However, as the degree of eutrophication intensifies, the proportion of endogenous organic matter in eutrophication lakes increases, and the DBPs generated when it is used as a reference water source increase, which should be taken seriously.

4.5.2. Factors influencing the generation of algal-sourced DBPs

Algal cells, IOM, and EOM are all precursors of C-DBPs and N-DBPs, and the DBPs yield is affected by the properties of the precursors by indirect influences including source, algae species, growth stage, and chlorination conditions [21,186,237,238].

5. The effects of precursors properties on DBPs production

5.1. The comparison of AOM and NOM

Reference showed that the trichloromethane (TCM) and trichloroacetic acid (TCAA) yields generated by AOM and fulvic acid were similar, but they were significantly lower than that of humic acid [239]. The average value of dichloroacetic acid (DCAA) produced by algae cells was 13.69 μg/L, which was significantly higher than that of humic acid (1.38 μg/L) and fulvic acid (1.20 μg/L). At the same time, the ratio of TCAA/DCAA generated by the chlorination of algal cells was 0.95, lower than humic acid (6.38) and fulvic acid (6.15). This may be due to the fact that humic acid and fulvic acid contain higher aromatic hydrocarbons and fewer nitrogen-containing substances than AOM, which were more prone to TCAA generation [239]. Compared with NOM, algal cells chlorinated with more organic nitrogen produced more N-DBPs, similar DCAA but lower C-DBPs (such as THM, TCAA, and haloketones (HKs)) [227]. DBPs generated by algae and NOM as a precursor are different due to their different properties, with NOM containing more complex fulvic acid and humic acid formed by biological degradation, but AOM mainly containing more proteins, amino acids and polysaccharides [206]. The organic nitrogen content in the algal cells was 230 mg N/mg C (organic carbon), but it was only 28 mg N/mg C in NOM. Accordingly, the carbon-nitrogen ratio in AOM was 4.3–4.8, and that in NOM was 35.2 [198]. The organic nitrogen in AOM reacts quickly with the active chlorine to generate organochloramines, some of which would decompose and generate N-DBPs [198]. At the same time, chlorine consumption hinders the reaction between chlorine and organic carbon leading to low C-DBPS production [198].

5.2. Effects of algal species

Cyanobacteria, diatoms and Chlorophyte were commonly used microalgae to investigate the role of biochemical substances of AOM in the DBPs yield and composition. The DOC produced by green algae culture was more likely to produce THMs and HAAs than that produced by cyanobacteria and diatoms. The chloroform produced by chlorination of green algae, diatoms and cyanobacteria were 0.40, 0.24, and 0.25 μmol/mg C, respectively [186]. In other studies, the yields of chloroform from chlorination of extracellular organics of Chlorophyta, diatoms, and cyanobacteria were 0.18–0.34, 0.42, and 0.50 μmol/mg C, respectively [239]. In chlorinating cells, green algae had higher HAAs than cyanobacteria, and DCAA and TCAA were much higher than MCAA. Additionally, the TCAA/DCAA values were 0.86 and 0.98, respectively, suggesting higher DCAA precursor content in algal cells [233]. Another study showed that diatoms produced higher chloroform and lower HAA than cyanobacteria and Chlorophyta, and organics, especially lipids, in diatoms were the main precursors of THM [22]. This might be attributed to that diatoms usually grow in nutrient-poor water and lipids usually accumulate under nutrient-limited conditions.

Controlling diatoms and lipids in water sources and reducing humic acid usually limit THM production. On the other hand, under the same nutrient conditions, low light and temperature, such as at high latitude conditions, accumulate proteins and lipids to maintain photosynthesis and prevent frostbite of cells, whereas high light conditions produce relatively low proteins but high sugars. Therefore, it can be inferred that algae at low temperatures (cold regions at high latitudes) were more likely to produce chloroform than algae in warm regions [22]. In addition, DBPs produced by AOM in different Cyanophyta species were also different. In the presence of bromine, THMs yields of Microcystis cells and extracellular organics were 5.76 and 3.47 μmol/mmol C, respectively, while HAAs yields were 9.73 and 4.61 μmol/mmol C, respectively [21]. However, the THMs yield of Anabaena was similar to that of Microcystis, but the HAAs generated by Anabaena were lower than that of Microcystis, which may be due to its lower hydrophobicity and HAA precursor compared with Microcystis [21].

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5.3. The influence of algal growth stage

Studies showed that most DBPs increased with the algal growth stage [232]. Hoehn et al. studied THM generation of two green algae and two cyanobacteria species at a 24 h chlorination time, indicating that the highest yield was achieved in the late-exponential growth stage [240]. Another study has shown that when Microcystis cells and extracellular organisms were chlorinated at different stages, the production of total THMs (TTHMs) and total HAAs (THAAs) changed with the growth stage of Microcystis [21]. The production of TTHMs and THAAs was relatively stable during the lag growth phase, increased slightly at the beginning of the logarithmic phase, fluctuated during the logarithmic phase, and then steadily increased during the stationary phase. The maximum value of TTHMs and THAAs in chlorinated cells occurred during the logarithmic phase (1.41 μmol/L) and late-stationary phase (3.06 μmol/L) [21]. During the decay period, the production of THAAs by chlorinated cells and extracellular organics decreased, while TTHMs produced by extracellular organics increased [21]. The changing trend of DBPs generated by Anabaena chloride cells and EOM was similar, indicating that IOM released by cell death in the late growth stage was more favorable to THM generation than HAA. Compared with the total production, it indicated that the unit carbon yield of DBPs did not change much with the growth stage when Microcystis and AOM were chlorinated. The unit carbon yield of Microcystis was at the late lag stage, while that of Anabaena was higher at the late growth stage [21]. The type of DBPs also changes with the growth stage. Monohalogenated HAA was the main product of cell and extracellular organics during lag and logarithmic growth stage of Anabaena [21]. At the stationary phase, higher halogenated products were produced, especially in the chlorination reaction of cells [21]. For Microcystis, monohalogenated HAA only appeared during the transition from lag to logarithmic phase. The ratio of trihalogenated HAA to dihalogenated HAA was 1.2 μmol/ μmol and 0.66 μmol/ μmol for cell and EOM, respectively [21]. In the decay phase, the proportion of trihalide HAA in extracellular organic samples increased rapidly, indicating that IOM in dying cells was the precursor of major trihalide HAAs [17]. The reason for the high HAA ratio at the beginning of the algal growth stage was still not clear, but polysaccharides as the main metabolite may be the reason for the low HAA production [21].

In addition, the different growth stages also led to the conversion of C-DBPs and N-DBPs. With the increasing growth time, the protein and organic nitrogen of M. aeruginosa decreased [206]. Chlorination reaction and nitrogen functional groups, such as inorganic nitrogen -NH2 react faster may hinder the reaction of chlorinated organic carbon [198]. So along with the growth stage, organic nitrogen content in the algal cells decreased, more residual chlorine can react with organic carbon to generate C-DPBs. High residual chlorine in algal cells at the late growth stage led to more 1,1-dichloropropanone (1,1-DCP) conversion to 1,1,1-tri-chloropropanone (1,1,1-TCP) [227]. With the increasing growth stage, the increase of N-DBPS generation might due to the precursor of N-DBP transformed from organic nitrogen to organic carbon.

5.4. The influence of algae cell number and its comparison with EOM

Relative study showed that the amount of chloroform and THM produced by algae cells with Chl-a concentration of 10 and 20 μg/L did not change significantly, but only affected HAAs production. HAAs yield was 17.18–24.96 and 26.51–37.24 μg/L when cyanobacteria Chl-a was 10 and 20 μg/L, respectively [233]. While the yield of HAAs was 17.86–27.10 and 27.17–47.75 μg/L when the Chl-a of green algae was 10 and 20 μg/L, respectively [233]. The increase of Chl-a may be the precursor of HAAs, and the difference between the two algae may be because green algae contain Chl-a and b, while cyanobacteria only contain Chl-a [233]. In contrast to DBPs produced by algae cells and extracellular organisms, the interaction between algae cells and extracellular organisms was also investigated. The algae cells of M. aeruginosa and Anabaena produce more THM and HAA than EOM. The amount of THMs and HAAs produced per unit carbon of Microcystis cells was 2–3 times highter than that of EOM, which was also observed in other algae species [21]. It indicates that physical removal of algae cells is more effective in controlling the production of DBPs than pre-ozonation and pre-chlorination methods that may cause cell rupture and release intracellular substances. When algae cells and extracellular organisms simultaneously exist, they would produce antagonism. The reason may be as follows: 1) Cell debris can adsorb THMs and HAAs. THMs are relatively hydrophobic and more easily adsorbed than hydrophilic HAAs, so the antagonism of THMs is more serious. 2) In the chlorination process, the intermediate products of THMs or HAAs consume the final product, or the intermediate product reacts with cells or EOM to produce products other than THMs and HAAs. 3) The degree of cell damage during the chlorination reaction is related to the cell morphology and the ratio of chloride to the release of intracellular substances, which may have an antagonistic effect.

6. The effects of chlorination conditions on DBPs production

The production and types of DBPs are influenced by chlorination amount, contact time, reaction temperature, pH, and other ions, which would ultimately affect DBPs contents in tap water.

6.1. The influence of disinfectant dosage

The generation rate of DBPs is affected by the amount of chlorine added and the remaining chlorine concentration. When the amount of chlorine added increases, the yield of THMs increases, but the response of DBPs types is different [221]. Chlorination experiments on organics of M. aeruginosa (chlorination 3 d) showed that with the increase of chlorine dosage, most DBPs (except for DCAN and 1,1-DCP) increased with the increase of chlorine concentration, however, DCAN first increased and then decreased, while 1,1-DCP always decreased [227]. This might be because the final generation concentration of unstable DBPs depended on the generation rate and degradation rate, so 1,1,1-TCP, chloral hydrate (CH), trichloroacetonitrile (TCAN), and trichloronitromethane (TCNM) increased all the time after 3 d [227]. The degradation rate of DCAN by oxidation or hydrolysis was faster than the generation rate even at a very high chlorine dosage (10.2 mg/L), because the concentration first

increases and then decreases [227]. The reason that 1,1-DCP decreased with the increase of chlorine dose was that 1,1-DCP could be oxidized to 1,1,1-TCP, and the presence of chlorine could accelerate the hydrolysis rate [241].

6.2. The influence of reaction time

Chlorination time has different effects on different DBPs production. THMs and HAAs increased with longer reaction time [242], but haloacetonitrile and haloacetonitrile decreased due to hydrolysis or reaction with residual chlorine [225,241,243]. When chlorinating algal cells, longer chlorination time may lead to cell damage and intracellular DOC release, resulting in more DBPs. DBPs types also varied with different reaction times: 60% chloroform was generated in a 5 min reaction, but 57%–71% HAAs were generated in a 30 min reaction [241]. The changing trend of N-DBPs over time can be explained by the oxidation and hydrolysis of chlorine [241]. DCAN and TCAN are hydrolyzed when pH was 7, and the hydrolysis rate increased with chlorine [241].

6.3. The influence of temperature

Generally, it is believed that as the reaction temperature rises, the reaction of chlorine and organic matter speeds up, and finally DBPs generation increases. However, the decomposition of some DBPs, such as dihalogenated acetonitrile and halogenated ketone, also speeds up accordingly [221]. After chlorinated algae cells at 10°C and 25°C for 3 d, it was found that most of the products increased with the increase of temperature, but 1,1-DCP and 1,1,1-TCP decreased with the increase of temperature, possibly because the degradation efficiency was higher than the generation efficiency at high temperature [227]. Besides, cyanobacteria tend to grow well at high temperatures, in which case they may produce more organic matter and thus higher DBPs [227].

6.4. The influence of pH

Different DBPs have different responses to pH changes during chlorination reactions. Studies have shown that high a pH value may increased the yield of THMs, but it would decrease the yield of HAAs, dichloroacetonitrile (DCAN), and 1,1,1-TCP [242,244]. This may be due to hydrolysis and oxidative degradation, and the instability of DBPs under alkaline pH and chlorine [243]. With the increase of pH from 4 to 9, it was found that the amount of residual chlorine decreased rapidly, TCM increased, but the production of 1,1-DCP and 1,1,1-TCP decreased [241]. This was because pH affected DBPs production by influencing the chlorination process. At a high pH value, the consumption of chlorine was large, which leaded to more DBP formation [241]. At the same time, pH value also affects the stability of unstable DBPs, for example, DCAN, TCAN, 1,1-DCP, 1,1,1-TCP, and CH can undergo catalytic hydrolysis, and the hydrolysis rate of unstable DBPs increased with the increase of pH [241].

6.5. The influence of ammonia

Under the condition of adding 0, 2.5, and 8.0 mg/L (calculated as N) of ammonia, the chlorinated algae cells for 3 d showed that the addition of ammonia reduced the production of most DBPs, but TCNM was not affected, and 1,1-DCP increased [227]. When 2.5 and 8.0 mg/L (calculated as N) of ammonia were added, only monochloramine remained at the end of the reaction, and the conversion of residual chlorine to monochloramine reduced the content of most DBPs, e.g. trichlorinated DBPs such as TCAA, CH, TCAN and 1,1,1-TCP were almost reduced to the detection limit [227,245]. TCNM was different from 1,1-DCP, and the generation of TCNM was only affected by the presence of ammonia. The yield of 1,1-DCP was higher in chloramination than in chlorination, because 1,1-DCP could be oxidized to 1,1,1-TCP by chlorine, but it was stable in the presence of monochloramine [227,241].

6.5.1. Removal methods for algae-derived precursors of DBPs

In recent years, some studies have discussed the removal methods of algae-derived precursors of DBPs. Ma et al. studied the chitosan-aluminum chloride (CTSAC) composite coagulation process to reduce AOM-related carbon and nitrogen DBPs formation [202]. Theie results showed that CTSAC significantly improved the removal rate of dissolved organic matter, humic acid, polysaccharide, and protein, compared to aluminium chloride (AC) and chitosan (CTS). Moreover, Fourier transforms infrared spectroscopy analysis proved the interaction of CTS and AC in the composite coagulant CTSAC improved the removal performance, thereby reducing the carbon and nitrogen of DBPs generated [202]. Using M. aeruginosa, Anabaena aequalis, and Oscillatoria tenuis, Bernat-Quesada et al. (2020) found that pre-ozonation with cyanobacteria suspension did not reduce the production of all the DBPs, with increasing THMs and trichloronitromethane, and only reducing the formation of haloacetonitriles, a nitrogen-containing disinfection byproduct [246]. Therefore, pre-ozonation for algae-containing water was not an appropriate method to reduce DBPs formation during chlorination, but rather cyanobacteria should be removed from raw water before chlorination or ozonation [246]. Besides, some scholars proposed the pretreatment of algae-containing water source by ultraviolet disinfection. The results showed that the ultraviolet radiation with low UV absorption rate could effectively reduce the formation of trihalomethanes and dichloacetic acid from EOM and IOM in the subsequent chlorination process. It was believed that the ultraviolet pretreatment process was a potential technology for the treatment of algal-rich water [247,248]. However, the above studies were carried out in laboratory. In the actual production, many factors such as technology, economy, and operation management need to be taken into consideration in the selection of removal technologies for the algae-derived precursors of DBPs.

7. Conclusions

Cyanobacterial blooms pose a threat to the water quality in drinking water sources and plants. The formation of cyanobacterial blooms and their associated metabolites were affected by many factors with spatial and temporal variations differed in water. Further in-depth research is needed from the perspective of the molecular mechanism of metabolite production, which will help managers targeted control cyanobacterial blooms and the release of its metabolites. The establishment of an

early warning system of cyanobacteria in water source is an important prerequisite to ensure drinking water safety. Previous research mainly focused on determining cyanobacteria warning values according to algal toxins. Besides, the impact of cyanobacteria on the safety of drinking water includes odorants and other algae-derived organic substances that form DBPs after chlorination. The development of an early warning value system for cyanobacteria based on the combination of algal toxins, odorous substances, and DBPs precursors as reference indicators can more comprehensively guide the control of cyanobacteria in water sources. The occurrence of cyanobacteria and their metabolites has seasonal changes. The establishment of early warning values for cyanobacteria at different growth stages is of great significance to the operability of actual work. Using online monitoring technologies to determine the density of cyanobacteria can improve the precision of the established cyanobacteria warning value system based on the acquisition of a large amount of data, and can detect in time that the cyanobacteria density exceeding the limit in the water sources so that corresponding control measures can be taken in time. Moreover, the combined toxicological effects of multiple algal origin organisms in the water source may affect the data of cyanobacteria warning values, which needs further study.

Algal metabolites in eutrophication water were difficult to be removed effectively in the conventional water production processes, thus affecting drinking water quality. Metabolites produced by cyanobacteria could be followed by algal cells into water plants. It released in the process of drinking water production due to the destruction of algae cells and leading to the pollution load of water. Besides using the concentration of odorants, algae toxins, and organic pollutants in the water body to characterize the water quality of the water source, it is necessary to figure out the content of metabolites in cyanobacteria cells and integrate the removal efficiency of metabolites in water plants in different seasons to establish a cyanobacterial density safety early warning system, to guarantee the safety of drinking water production. In addition, the development of effective and economically feasible removal technologies and processes for different metabolites is still the top priority, and attention should also be paid to their application in actual water plants.

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References

1. Almanza V, Parra O, Bicudo CEDM, Baeza C, Beltran J, Figueroa R, Urrutia R. Occurrence of toxic blooms of Microcystis aeruginosa in a central Chilean (36 degrees Lat. S) urban lake. Revista Chilena De Historia Natural 2016, 89, doi:10.1186/s40693-016-0057-7.
2. Barros MUG, Wilson AE, Leitao JIR, Pereira SP, Buley RP, Fernandez-Figueroa EG, Capelo-Neto J. Environmental factors associated with toxic cyanobacterial blooms across 20 drinking water reservoirs in a semi-arid region of Brazil. Harmful Algae. 2019; 86: 128-137, doi:10.1016/j.hal.2019.05.006.
3. Carmichael WW, Boyer GL. Health impacts from cyanobacteria harmful algae blooms: Implications for the North American Great Lakes. Harmful Algae. 2016; 54: 194-212, doi:10.1016/j.hal.2016.02.002.
4. Clark JM, Schaeffer BA, Darling JA, Urquhart EA, Johnston JM, Ignatius AR,et al. Satellite monitoring of cyanobacterial harmful algal bloom frequency in recreational waters and drinking water sources. Ecological Indicators 2017; 80: 84-95, doi:10.1016/j.ecolind.2017.04.046.
5. Huisman J, CoddGA, Paerl HW, Ibelings BW, Verspagen JMH, Visser PM. Cyanobacterial blooms. Nature Reviews Microbiology. 2018; 16: 471-483, doi:10.1038/s41579-018-0040-1.
6. Greene L, Hayes C. The impact of eutrophication on water treatment and supplies in the Anglian Water Authority. Journal of the Institution of Water Engineers and Scientists 1981; 35.
7. Henderson R, Parsons SA, Jefferson B. The impact of algal properties and pre-oxidation on solid–liquid separation of algae. Water Research. 2008; 42: 1827-1845. doi:10.1016/j.watres.2007.11.039.
8. Qin B, Zhu G, Gao G, Zhang Y, Li W, Paerl H, Carmichael WA. drinking water crisis in Lake Taihu, China: linkage to climatic variability and lake management. Environmental Management 2010; 45: 105-112, doi:10.1007/s00267-009-9393-6.
9. Shang L. Research on cyanobacterial pollution and establishment of cyanobacterial threshold system in drinking water resources. Graduate University of Chinese Academy of Sciences, Beijing, 2015.
10. 10. 10. Miller TR, Beversdorf LJ, Weirich CA, Bartlett SL. Cyanobacterial Toxins of the Laurentian Great Lakes, Their Toxicological Effects, and Numerical Limits in Drinking Water. Marine Drugs 2017; 15. doi:10.3390/md15060160.
11. Ghernaout B, Ghernaout D, Saiba A. Algae and cyanotoxins removal by coagulation/flocculation: A review. Desalination and Water Treatment 2010; 20: 133-143, doi:10.5004/dwt.2010.1202.
12. Henderson RK, Parsons SA, Jefferson B. The impact of differing cell and algogenic organic matter (AOM) characteristics on the coagulation and flotation of algae. Water Research. 2010; 44: 3617-3624, doi:10.1016/j.watres.2010.04.016.
13. Carmichael WW. Cyanobacteria secondary metabolites–the cyanotoxins. Journal of Applied Bacteriology. 1992; 72: 445-459, doi:10.1111/j.1365-2672.1992.tb01858.x.
14. Codd GA. Cyanobacterial toxins: Occurrence, properties and biological significance. Water Science and Technology. 1995; 32: 149-156, doi:10.1016/0273-1223(95)00692-3.
15. Percival SL, Williams DW. Cyanobacteria. In Microbiology of Waterborne Diseases (Second Edition), Percival SL, Yates MV, Williams DW, Chalmers RM, Gray NF, Eds. Academic Press: London, 2014; https://doi.org/10.1016/B978-0-12-415846-7.00005-6pp. 79-88.
16. Watson SB. Cyanobacterial and eukaryotic algal odour compounds: signals or by-products? A review of their biological activity.

Phycologia. 2003; 42: 332-350, doi:10.2216/i0031-8884-42-4-332.1.

17. Yu J, Li Z, Cao N, Yang M, Ding J, Miao T, Zhang J. Analyses on cause for odor and potential problems in water source during odor episode event in Wuxi. Acta Scientiae Circumstantiae. 2007; 27: 1771-1777, doi:10.13671/j.hjkxxb.2007.11.025.
18. Watson SB, Ridal J. Periphyton: a primary source of widespread and severe taste and odour. Water Science and Technology. 2004; 49: 33-39, doi:10.2166/wst.2004.0527.
19. Chu W, Gao N, Deng Y, Krasner SW. Precursors of dichloroacetamide, an emerging nitrogenous DBP formed during chlorination or chloramination. Environmental Science & Technology. 2010: 44; 3908-3912, doi:10.1021/es100397x.
20. Fang J, Yang X, Ma J, Shang C, Zhao Q. Characterization of algal organic matter and formation of DBPs from chloramination. Water Research. 2010; 44: 5897-5906. doi:10.1016/j.watres.2010.07.009.
21. Huang J, Graham N, Templeton M, Zhang Y, Collins C, Nieuwenhuijsen M. A comparison of the role of two blue–green algae in THM and HAA formation. Water research 2009; 43: 3009-3018. doi:10.1016/j.watres.2009.04.029.
22. Hong HC, Mazumder A, Wong MH, Liang Y. Yield of trihalomethanes and haloacetic acids upon chlorinating algal cells, and its prediction via algal cellular biochemical composition. Water research. 2008; 42: 4941-4948, doi: 10.1016/j.watres.2008.09.019.
23. Skulberg OM, Codd GA, Carmichael WW. Toxic blue-green algal blooms in Europe: A growing problem. Ambio: journal of human environment. 1984; 13: 245-247.
24. Hernández JM, López-Rodas V, Costas E. Microcystins from tap water could be a risk factor for liver and colorectal cancer: A risk intensified by global change. Medical hypotheses. 2009; 72: 539-540. doi:10.1016/j.mehy.2008.11.041.
25. Dao TS, Do-Hong LC, Wiegand C. Chronic effects of cyanobacterial toxins on Daphnia magna and their offspring. Toxicon. 2010; 55: 1244-1254, doi:10.1016/j.toxicon.2010.01.014.
26. 26. 26. Codd GA. Cyanobacterial toxins, the perception of water quality, and the prioritisation of eutrophication control. Ecological Engineering. 2000: 16: 51-60, doi:10.1016/S0925-8574(00)00089-6.
27. Falconer IR. Toxic cyanobacterial bloom problems in Australian waters: risks and impacts on human health. Phycologia. 2001; 40: 228-233. doi:10.2216/i0031-8884-40-3-228.1.
28. Ernst B, Hoeger SJ, O'Brien E, Dietrich DR. Abundance and toxicity of Planktothrix rubescens in the pre-alpine Lake Ammersee, Germany. Harmful Algae. 2009; 8: 329-342, doi:10.1016/j.hal.2008.07.006.
29. Paerl H, Otten T. Harmful cyanobacterial blooms: Causes, consequences, and controls. Microbial Ecology. 2013; 65: 995-1010, doi:10.1007/s00248-012-0159-y.
30. Pitois F, Fastner J, Pagotto C, Dechesne M. Multi-toxin occurrences in ten french water resource reservoirs. Toxins. 2018; 10: 283. doi:10.3390/toxins10070283.
31. Agrawal A, Gopal K. Toxic cyanobacteria in water and their public health consequences. In Biomonitoring of Water and Waste Water, Springer: 2013; 135-147.
32. PBotes D, CViljoen C, HeléneKruger, LWessels P, HWilliams D. Configuration assignments of the amino acid residues and the presence of N-methyl dehydroalanine intoxins from the blue-green alga Microcystis aeruginosa. Toxicon. 1982; 20: 1037-1042.
33. Tamele IJ, Vasconcelos V. Microcystin incidence in the drinking water of mozambique: challenges for public health protection. Toxins 2020; 12. doi:10.3390/toxins12060368.
34. Amé MV, Galanti LN, Menone ML, Gerpe MS, Moreno VJ, Wunderlin DA. Microcystin–LR, –RR, –YR and –LA in water samples and fishes from a shallow lake in Argentina. Harmful Algae 2010; 9: 66-73, doi:http://dx.doi.org/10.1016/j.hal.2009.08.001.
35. Botes DP, Wessels PL, Kruger H, Runnegar MT, Santikarn S, Smith RJ, Et al. Structural studies on cyanoginosins-LR,-YR,-YA, and-YM, peptide toxins from Microcystis aeruginosa. Journal of the Chemical Society, Perkin Transactions. 1985; 1: 2747-2748.
36. Lehman P, Boyer G, Satchwell M, Waller S. The influence of environmental conditions on the seasonal variation of Microcystis cell density and microcystins concentration in San Francisco Estuary. Hydrobiologia 2008; 600: 187-204, doi:https://doi.org/10.1007/s10750-007-9231-x.
37. Rivasseau C, Martins S, Hennion MC. Determination of some physicochemical parameters of microcystins (cyanobacterial toxins) and trace level analysis in environmental samples using liquid chromatography. Journal of chromatography A 1998; 799: 155-169, doi:https://doi.org/10.1016/S0021-9673(97)01095-9.
38. Rapala J, Lahti K, Sivonen K, Niemelä S. Biodegradability and adsorption on lake sediments of cyanobacterial hepatotoxins and anatoxin-a. Letters in applied microbiology 1994; 19: 423-428, doi:https://doi.org/10.1111/j.1472-765X.1994.tb00972.x.
39. Deng Y. Study on photocatalytic degradation of microcystins in lake water by TiO2 nanotubes. Graduate University of Chinese Academy of Sciences, Beijing, 2008.
40. Harada K, Tsuji K, Watanabe MF, Kondo F. Stability of microcystins from cyanobacteria-III.* Effect of pH and temperature. Phycologia 1996; 35: 83-88, doi:https://doi.org/10.2216/i0031-8884-35-6S-83.1.
41. Westrick JA, Szlag DC, Southwell BJ, Sinclair J. A review of cyanobacteria and cyanotoxins removal/inactivation in drinking water treatment. Analytical and Bioanalytical Chemistry 2010; 397: 1705-1714, doi:10.1007/s00216-010-3709-5.
42. Azevedo SMFO, Carmichael WW, Jochimsen EM, Rinehart KL, Lau S, ShawGR, Eaglesham GK. Human intoxication by microcystins during renal dialysis treatment in Caruaru-Brazil. Toxicology 2002; 181–182: 441-446, doi:http://dx.doi.org/10.1016/S0300-483X(02)00491-2.
43. Lam AKY, Fedorak PM, Prepas EE. Biotransformation of the cyanobacterial hepatotoxin microcystin-LR, as determined by HPLC and protein phosphatase bioassay. Environmental science & technology 1995; 29: 242-246, doi:https://doi.org/10.1021/es00001a030.

44. WHO. Guidelines for drinking water quality, incorporating first addendum. 2006.
45. Health MO. Standards for drinking water quality. In National Standard of the People's Republic of China, China Standard Press: Beijing, 2006; Vol. GB 5749-2006.
46. Chen W, Song L, Gan N, Li L. Sorption, degradation and mobility of microcystins in Chinese agriculture soils: risk assessment for groundwater protection. Environmental pollution 2006; 144: 752-758, doi:https://doi.org/10.1016/j.envpol.2006.02.023.
47. Song L, Chen W, Peng L, Wan N, Gan N, Zhang X. Distribution and bioaccumulation of microcystins in water columns: A systematic investigation into the environmental fate and the risks associated with microcystins in Meiliang Bay, Lake Taihu. Water Research 2007; 41: 2853-2864, doi:10.1016/j.watres.2007.02.013.
48. Chu FS, Huang X, Wei RD, Carmichael WW. Production and characterization of antibodies against microcystins. Applied and environmental microbiology 1989; 55: 1928-1933, doi:10.1128/AEM.55.8.1928-1933.1989.
49. Birbeck JA, Westrick JA, O'Neill GM, Spies B, Szlag DC. Comparative analysis of microcystin prevalence in Michigan Lakes by online concentration LC/MS/MS and ELISA. Toxins 2019; 11: doi:10.3390/toxins11010013.
50. Wu J, Ni Q, Ren Y, Xu R, Liang N, Huang J, Dai Z. Determination of 8 microcystinsin water by high performance liquid chromatography-quadrupole- igh resolution time of flight mass spectrometry. The Administration and Technique of Environmental Monitoring 2020; https://doi.org/10.19501/j.cnki.1006-2009.20201111.003 1-4, doi:https://doi.org/10.19501/j.cnki.1006-2009.20201111.003
51. Zhang K, Ma H, Yan P, Tong W, Huang X, Chen DDY. Electrochemical detection of microcystin-LR based on its deleterious effect on DNA. Talanta 2018, 185, 405-410, doi:10.1016/j.talanta.2018.03.051.
52. Lawton LA, Edwards C, Codd GA. Extraction and high-performance liquid chromatographic method for the determination of microcystins in raw and treated waters. Analyst 1994; 119: 1525-1530, doi:https://doi.org/10.1039/AN9941901525.
53. Spoof L, Vesterkvist P, LindholmT, Meriluoto J. Screening for cyanobacterial hepatotoxins, microcystins and nodularin in environmental water samples by reversed-phase liquid chromatography–electrospray ionisation mass spectrometry. Journal of Chromatography A 2003; 1020: 105-119, doi:https://doi.org/10.1016/S0021-9673(03)00428-X.
54. Harada KI, Nakano T, Fujii K, Shirai M. Comprehensive analysis system using liquid chromatography–mass spectrometry for the biosynthetic study of peptides produced by cyanobacteria. Journal of Chromatography A 2004; 1033: 107-113, doi:10.1016/j.chroma.2004.01.006
55. Davis TW, Berry DL, Boyer GL, Gobler CJ. The effects of temperature and nutrients on the growth and dynamics of toxic and non-toxic strains of Microcystis during cyanobacteria blooms. Harmful Algae 2009; 8: 715-725, doi:http://dx.doi.org/10.1016/j.hal.2009.02.004.
56. Vasas G, Szydlowska D, Gáspár A, Welker M, Trojanowicz M, Borbély G. Determination of microcystins in environmental samples using capillary electrophoresis. Journal of Biochemical and Biophysical Methods 2006; 66: 87-97, doi:http://dx.doi.org/10.1016/j.jbbm.2005.12.002.
57. (AFSSET) A.F.d.S.S.d.A.A.a.A.F.d.S.S.d.l.E.e.d.T. Avis de relatif à l'évaluation des risques liés à la présence de cyanobactéries dans les cours d'eau destinés à la baignade et/ou à d'autres usages; France, 2006.
58. Haddix PL, Hughley CJ, LeChevallier MW. Occurrence of microcystins in 33 US water supplies. Journal (American Water Works Association) 2007; https://doi.org/10.1002/j.1551-8833.2007.tb08033.x, 118-125, doi:https://doi.org/10.1002/j.1551-8833.2007.tb08033.x.
59. Zamyadi A, MacLeod SL, Fan Y, McQuaid N, Dorner S, Sauve S, Prevost M. Toxic cyanobacterial breakthrough and accumulation in a drinking water plant: a monitoring and treatment challenge. Water Research 2012, 46, 1511-1523, doi:10.1016/j.watres.2011.11.012.
60. 60. Liu Y, Chen W, Li D, Huang Z, Shen Y, LiuY. Cyanobacteria-/cyanotoxin-contaminations and eutrophication status before Wuxi drinking water crisis in Lake Taihu, China. Journal of Environmental Sciences 2011; 23: 575-581, doi:10.1016/S1001-0742(10)60450-0.
61. Bui T, Dao TS, Faassen E, Lurling M. Cyanobacterial blooms and microcystins in Southern Vietnam. Toxins 2018, 10, doi:10.3390/toxins10110471.
62. Bui T, Dao T S, Faassen E, Lurling M, Cyanobacterial blooms and microcystins in Southern Vietnam. Toxins 2018, 10, doi:10.3390/toxins10110471.
63. Wan X, Steinman A D, Gu, Y, Zhu G, Shu, X, Xue Q, Zou W, Xie L. Occurrence and risk assessment of microcystin and its relationship with environmental factors in lakes of the eastern plain ecoregion, China. Environmental Science and Pollution Research 2020, 10.1007/s11356-020-10384-0, doi:10.1007/s11356-020-10384-0.
64. Sivonen K, Niemelä S I, Niemi R M, Lepistö L, Luoma T H, Räsänen L A. Toxic cyanobacteria (blue-green algae) in Finnish fresh and coastal waters. Hydrobiologia 1990, 190, 267-275, doi:10.1007/BF00008195.
65. Aboal M, Pui M Á, Intracellular and dissolved microcystin in reservoirs of the river Segura basin, Murcia, SE Spain. Toxicon 2005, 45, 509-518, doi:http://dx.doi.org/10.1016/j.toxicon.2004.12.012.
66. Briand E, Escoffier N, Straub C,Sabart M, Quiblier C, Humbert J F, Humbert J F. Spatiotemporal changes in the genetic diversity of a bloom-forming Microcystis aeruginosa (cyanobacteria) population. The ISME Journal 2009, 3, 419-429, doi:doi: 10.1038/ismej.2008.121.
67. 67. Zhang L, Liu J, Zhang D, Luo L, Liao Q, Yuan L,Wu N. Seasonal and spatial variations of microcystins in Poyang Lake, the largest freshwater lake in China. Environmental Science and Pollution Research 2018, 25, 6300-6307, doi:10.1007/s11356-017-0967-1.
68. Li Z, Yu J, Yang M, Zhang J, Burch M D, Han W. Cyanobacterial population and harmful metabolites dynamics during a bloom in Yanghe Reservoir, North China. Harmful Algae 2010, 9, 481-488,

doi:10.1016/j.hal.2010.03.003.

69. Yang H, Xie P, Xu J, Zheng L, Deng D, Zhou Q, Wu S. Seasonal variation of microcystin concentration in lake Chaohu, a shallow subtropical lake in the people's republic of China. Bulletin of Environmental Contamination and Toxicology 2006, 77, 367-374, doi:10.1007/s00128-006-1075-y.
70. Chen W, Song L, Peng L, Wan N, Zhang X, Gan, N. Reduction in microcystin concentrations in large and shallow lakes: Water and sediment-interface contributions. Water Research 2008, 42, 763-773, doi:10.1016/j.watres.2007.08.007.
71. Sivonen K. Effects of light, temperature, nitrate, orthophosphate, and bacteria on growth of and hepatotoxin production by Oscillatoria agardhii strains. Applied and Environmental Microbiology 1990, 56, 2658-2666.
72. Utkilen H, Gjølme N. Toxin production by Microcystis aeruginosa as a function of light in continuous cultures and its ecological significance. Applied and Environmental microbiology 1992, 58, 1321-1325, doi:doi:10.1128/AEM.58.4.1321-1325.1992.
73. Wiedner C, Visser P M, Fastner J, Metcalf J S, Codd G A, Mur LR. Effects of light on the microcystin content of Microcystis strain PCC 7806. Applied and Environmental Microbiology 2003, 69, 1475-1481, doi:10.1128/aem.69.3.1475-1481.2003.
74. Watanabe M F, Oishi S. Effects of environmental factors on toxicity of a cyanobacterium (Microcystis aeruginosa) under culture conditions. Applied and Environmental microbiology 1985, 49, 1342-1344, doi:https://doi.org/10.1128/AEM.49.5.1342-1344.1985.
75. Lehtimaki J, Sivonen K, Luukkainen R, Niemela S. The effects of incubation time, temperature, light, salinity, and phosphorus on growth and hepatotoxin production by Nodularia strains. Archiv für Hydrobiologie 1994, 130, 269-282.
76. Lukač M, Aegerter R. Influence of trace metals on growth and toxin production of Microcystis aeruginosa. Toxicon 1993, 31, 293-305, doi:http://dx.doi.org/10.1016/0041-0101(93)90147-B.
77. Utkilen H, Gjølme N. Iron-stimulated toxin production in Microcystis aeruginosa. Applied and Environmental Microbiology 1995, 61, 797-800, doi:https://doi.org/10.1128/AEM.61.2.797-800.1995.
78. van der Westhuizen AJ, Eloff J N. Effect of culture age and pH of culture medium on the growth and toxicity of the blue-green alga Microcystis aeruginosa. Zeitschrift für Pflanzenphysiologie 1983, 110, 157-163, doi:http://dx.doi.org/10.1016/S0044-328X(83)80162-7.
79. Watanabe M, Harada K I, Matsuura K, Watanabe M, Suzuki M. Heptapeptide toxin production during the batch culture of two Microcystis species (Cyanobacteria). Journal of Applied Phycology 1989, 1, 161-165, doi:10.1007/BF00003879.
80. Heyan WU, Jin S U, Wei S.H.I. Study on growth and toxin production of microcystis aeraginosa strain under different conditions. Journal of Environment and Health 2006, 23, 304-307.
81. Codd G.A, Bell S G. Brooks, W.P. Cyanobacterial toxins in water. Water Science & Technology 1989, 21, 1-13.
82. Yu L, Zhu G, Kong F, Li S, Shi X, Zhang M, Yang, Z, Xu H, Zhu M. Spatiotemporal characteristics of microcystin variants composition and associations with environmental parameters in Lake Chaohu,China. Journal of Lake Sciences 2019, 31, 700-713, doi:10.18307/2019.0309.
83. Chen W, Peng L, Wan N, Song L. Mechanism study on the frequent variations of cell-bound microcystins in cyanobacterial blooms in Lake Taihu: Implications for water quality monitoring and assessments. Chemosphere 2009, 77, 1585-1593, doi:http://dx.doi.org/10.1016/j.chemosphere.2009.09.037.
84. Vézie C, Rapala J, Vaitomaa, J, Seitsonen, J, Sivonen K. Effect of nitrogen and phosphorus on growth of toxic and nontoxic Microcystis strains and on intracellular microcystin concentrations. Microbial Ecology 2002, 43, 443-454, doi:10.1007/s00248-001-0041-9.
85. Wilhelm SW, Farnsley S E, LeCleir G R, Layton A C. Satchwell M F, DeBruyn J M, Boyer G L, Zhu G,Paerl H W. The relationships between nutrients, cyanobacterial toxins and the microbial community in Taihu (Lake Tai), China. Harmful Algae 2011, 10, 207-215, doi:https://doi.org/10.1016/j.hal.2010.10.001.
86. Kurmayer R, Christiansen G, Chorus I. The abundance of microcystin-producing genotypes correlates positively with colony size in Microcystis sp. and determines its microcystin net production in Lake Wannsee. Applied and Environmental Microbiology 2003, 69, 787-795, doi:10.1128/AEM.69.2.787-795.2003.
87. Ha J H, Hidaka T, Tsuno H. Quantification of toxic microcystis and evaluation of its dominance ratio in blooms using real-time PCR. Environmental Science & Technology 2009, 43, 812-818, doi:10.1021/es801265f.
88. Wang J, Xie P, Guo N. Effects of nonylphenol on the growth and microcystin production of Microcystis strains. Environmental Research 2007, 103, 70-78, doi:10.1016/j.envres.2006.05.013.
89. Briand E, Gugger M, François J C, Bernard C, Humbert J F, Quiblier C. Temporal variations in the dynamics of potentially microcystin-producing strains in a bloom-forming Planktothrix agardhii (cyanobacterium) population. Applied and environmental microbiology 2008, 74, 3839-3848, doi:doi: 10.1128/AEM.02343-07.
90. Briand E, Yéprémian C, Humbert J F, Quiblier C. Competition between microcystin- and non-microcystin-producing Planktothrix agardhii (cyanobacteria) strains under different environmental conditions. Environmental Microbiology 2008, 10, 3337-3348, doi:10.1111/j.1462-2920.2008.01730.x.
91. Sabart M, Pobel D , Briand E, Combourieu B, Salencon J, Humbert J F, Latour D. Spatiotemporal variations in microcystin concentrations and in the proportions of microcystin-producing cells in several Microcystis aeruginosa populations. Applied and Environmental Microbiology 2010, 76, 4750-4759, doi:10.1128/AEM.02531-09.
92. Welker M, Von Döhren H, Täuscher H, Steinberg C E, Erhard M. Toxic Microcystis in shallow lake Müggelsee (Germany)–dynamics, distribution, diversity. Archiv für Hydrobiologie 2003, 157, 227-248, doi:DOI: 10.1127/0003-9136/2003/0157-0227.
93. Via-Ordorika L, Fastner J, Kurmayer R, Hisbergue M, Dittmann

E, Komarek J,Erhard M, Chorus I. Distribution of microcystin-producing and non-microcystin-producing Microcystis sp. in european freshwater bodies: detection of microcystins and microcystin genes in individual colonies. Systematic and Applied Microbiology 2004, 27, 592-602, doi:10.1078/0723202041748163.

94. Rinta-Kanto J M, Konopko E A, DeBruyn J M, Bourbonniere R A, Boyer G L, Wilhelm, S.W. Lake Erie Microcystis: relationship between microcystin production, dynamics of genotypes and environmental parameters in a large lake. Harmful Algae 2009, 8, 665-673, doi:10.1016/j.hal.2008.12.004.
95. Westhuizen A J v d, Eloff, J N, Effect of temperature and light on the toxicity and growth of the blue-green alga Microcystis aeruginosa (UV 006). Planta 1985, 163, 55-59, doi:10.1007/BF00395897.
96. Harke M J, Gobler C J. Global transcriptional responses of the toxic cyanobacterium, Microcystis aeruginosa, to nitrogen stress, phosphorus stress, and growth on organic matter. Plos One 2013, 8, doi:10.1371/journal.pone.0069834.
97. Chorus I, Bartram J. Toxic cyanobacteria in water: A guide to their public health consequences, monitoring and management; Spon Press: 1999.
98. Jiang J Q, Graham N J, Harward C. Comparison of polyferric sulphate with other coagulants for the removal of algae and algae-derived organic matter. Water Science & Technology 1993, 27, 221-230, doi:https://doi.org/10.2166/wst.1993.0280.
99. Drikas M, Chow C W, House J, Burch M D. Using coagulation, flocculation, and settling to remote toxic cyanobacteria. Journal-American Water Works Association 2001, 93, 100-111, doi:https://doi.org/10.1002/j.1551-8833.2001.tb09130.x.
100. Fan J, Hobson P, Ho L, Daly R, Brookes J. The effects of various control and water treatment processes on the membrane integrity and toxin fate of cyanobacteria. Journal of Hazardous Materials 2014, 264, 313-322, doi:10.1016/j.jhazmat.2013.10.059.
101. Zamyadi A, Dorner S, Sauvé S, Ellis D, Bolduc A, Bastien C, Prévost M. Species-dependence of cyanobacteria removal efficiency by different drinking water treatment processes. Water Research 2013, 47, 2689-2700, doi:http://dx.doi.org/10.1016/j.watres.2013.02.040.
102. un F, Pei H Y, Hu W R, Ma C X. The lysis of Microcystis aeruginosa in AlCl3 coagulation and sedimentation processes. Chemical Engineering Journal 2012, 193–194, 196-202, doi:10.1016/j.cej.2012.04.043.
103. Ma M, Liu R, Liu H, Qu J. Chlorination of Microcystis aeruginosa suspension: Cell lysis, toxin release and degradation. Journal of Hazardous Materials 2012, 217–218, 279-285, doi:10.1016/j.jhazmat.2012.03.030.
104. Ho L, Dreyfus J, Boyer J, Lowe T, Bustamante H, Duker P, Meli T, Newcombe G. Fate of cyanobacteria and their metabolites during water treatment sludge management processes. Science of The Total Environment 2012, 424, 232-238, doi:10.1016/j.scitotenv.2012.02.025.
105. Villars K, Huang Y, Lenhart J J. Removal of the cyanotoxin microcystin-lr from drinking water using granular activated carbon. Environmental Engineering Science 2020, 37, 585-595, doi:10.1089/ees.2020.0017.
106. Ampiaw R E, Yaqub M, Lee W. Adsorption of microcystin onto activated carbon: A review. Membrane and Water Treatment 2019, 10, 405-415, doi:10.12989/mwt.2019.10.6.405.
107. Dixon M B, Falconet C, Ho L, Chow, CWK, O'Neill B K, Newcombe G. Removal of cyanobacterial metabolites by nanofiltration from two treated waters. Journal of Hazardous Materials 2011, 188, 288-295, doi:doi: 10.1016/j.jhazmat.2011.01.111.
108. Wu Y, Wang, Q, Wang Y, Shi D, An Y. Optimization of parameters in coagulation / flotation process for algae removal. China Water & Wastewater 2008, 24, 95-99.
109. Rajasekhar P, Fan L, Thong N, Roddick F A. A review of the use of sonication to control cyanobacterial blooms. Water Research 2012, 46, 4319-4329, doi:10.1016/j.watres.2012.05.054.
110. Ma C, Hu W Pei, H Xu H, Pei R. Enhancing integrated removal of Microcystis aeruginosa and adsorption of microcystins using chitosan-aluminum chloride combined coagulants: Effect of chemical dosing orders and coagulation mechanisms. Colloids and Surfaces a-Physicochemical and Engineering Aspects 2016, 490, 258-267, doi:10.1016/j.colsurfa.2015.11.056.
111. Zamyadi A. Ho L, Newcombe G, Daly R I. Burch M. Baker P. Prévost M. Release and oxidation of cell-bound saxitoxins during chlorination of Anabaena circinalis cells. Environmental Science & Technology 2010, 44, 9055-9061, doi:https://doi.org/10.1021/es102130b.
112. Laszakovits J R, MacKay A A. Removal of cyanotoxins by potassium permanganate: Incorporating competition from natural water constituents. Water Research 2019, 155, 86-95, doi:10.1016/j.watres.2019.02.018.
113. Loganathan K. Ozone-based advanced oxidation processes for the removal of harmful algal bloom (HAB) toxins: a review. Desalination and Water Treatment 2017, 59, 65-71, doi:10.5004/dwt.2016.0346.
114. Micheletto J, Torres M d A, de Paula V d C S, Cerutti V E, Pagioro T A, Cass Q B, Martins LRR, de Liz M V, de Freitas A M. The solar photo-Fenton process at neutral pH applied to microcystin-LR degradation: Fe2+, H(2)O(2)and reaction matrix effects. Photochemical & Photobiological Sciences 2020, 19, 1078-1087, doi:10.1039/d0pp00050g.
115. 115. Liu S, Hu X, Jiang W, Ma L, Cai M, Xu H, Wu M, Ma F. Degradation of Microcystins from Microcystis aeruginosa by 185-nm UV Irradiation. Water Air and Soil Pollution 2016, 227, doi:10.1007/s11270-016-2805-3.
116. 116. 116. Liang W. Inactivation of algae and degradation of microcystins by eleetrochemieal oxidation and pulsed maguetic field. Beijing Forestry University, Beijing, 2005.
117. 117. 117. Harada K i. Imanishi S, Kato H, Mizuno M, Ito E, Tsuji K. Isolation of Adda from microcystin-LR by microbial degradation. Toxicon 2004, 44, 107-109, doi:https://doi.org/10.1016/j.toxicon.2004.04.003.
118. Wang S F, Guo C R, Xu X Y, Zhu L. Review on biological treatment

of cyanobacterial toxin in natural waters. Ying yong sheng tai xue bao = The journal of applied ecology 2016, 27, 1683-1692, doi:10.13287/j.1001-9332.201605.026.

119. Lopes W S, Buriti J S, Cebalos B S O, Sousa J T, Leite V D, Vieira F F. Removal of microcystin-LRfrom drinking water using a system involving oxidation and adsorption. Water Air and Soil Pollution 2017, 228, doi:10.1007/s11270-017-3504-4.
120. Zhang X, He J, Lei, Y, Qiu Z, Cheng S ,Yang X. Combining solar irradiation with chlorination enhances the photochemical decomposition of microcystin-LR. Water Research 2019, 159, 324-332, doi:10.1016/j.watres.2019.05.030.
121. li l, Gao N, Yin D, Zhang K. Treatment processes in waterworks for removal of algae and algal toxins from raw water. China Water & Wastewater 2008, 6.
122. Shang L, Feng M, Xu X, Liu F, Ke F, Li W. Co-occurrence of microcystins and taste-and-odor compounds in drinking water source and their removal in a full-scale drinking water treatment plant. Toxins 2018, 10, doi:10.3390/toxins10010026.
123. Jurczak T, Tarczynska M, Izydorczyk K, Mankiewicz J, Zalewski M, Meriluoto J. Elimination of microcystins by water treatment processes - examples from Sulejow Reservoir, Poland. Water Research 2005, 39, 2394-2406, doi:DOI 10.1016/j.watres.2005.04.031.
124. Izaguirre G, Hwang C J, Krasner S W, McGuire M J. Geosmin and 2-methylisoborneol from cyanobacteria in three water supply systems. Applied and Environmental Microbiology 1982, 43, 708-714, doi:10.1128/AEM.43.3.708-714.1982.
125. Durrer M, Zimmermann U, Jüttner F. Dissolved and particle-bound geosmin in a mesotrophic lake (lake Zürich): spatial and seasonal distribution and the effect of grazers. Water Research. 1999; 33: 3628-3636, doi:10.1016/s0043-1354(99)00069-x.
126. Davies J.M, Roxborough M and Mazumder A. Origins and implications of drinking water odours in lakes and reservoirs of British Columbia, Canada. Water Research. 2004; 38:1900-1910, doi: 10.1016/j.watres.2004.01.008.
127. Li L, Wan N, Gan N, Xia B and Song L. Annual dynamics and origins of the odorous compounds in the pilot experimental area of Lake Dianchi, China. Water science and technology. 2007; 55: 43-50. doi:https://doi.org/10.2166/wst.2007.160.
128. Peter A, Köster O, Schildknecht A and von Gunten U. Occurrence of dissolved and particle-bound taste and odor compounds in Swiss lake waters. Water Research. 2009; 43: 2191-2200, doi:http://dx.doi.org/10.1016/j.watres.2009.02.016.
129. Journey C.A, Arrington J.M, Beaulieu K.M Graham J.L and Bradley P.M. Limnological conditions and occurrence of taste-and-odor compounds in lake William C. Bowen and Municipal Reservoir# 1, Spartanburg County, South Carolina, 2006-2009; U. S. Geological Survey: 2011.
130. Jo J-O, Kim S.D, Lee, H.-J and Mok, Y.S. Decomposition of taste-and-odor compounds produced by cyanobacteria algae using atmospheric pressure plasma created inside a porous hydrophobic ceramic tube. Chemical Engineering Journal. 2014; 247:291-301, doi:10.1016/j.cej.2014.03.018.
131. Pham T.L, Bui M.H, Driscoll M, Shimizu K and Motoo, U. First report of geosmin and 2-methylisoborneol (2-MIB) in Dolichospermum and Oscillatoria from Vietnam. Limnology 10.1007/s10201-020-00630-2, doi:10.1007/s10201-020-00630-2.
132. Wei L, Sun G, Jiang W and Li J. A contamination accident of tap water Preventive Medicine Tribune. 1999; 5: 366, doi:10.16406/j.pmt.issn.1672-9153.1999.04.053.
133. Jardine C, Gibson N and Hrudey S. Detection of odour and health risk perception of drinking water. Water Science and Technology. 1999; 40: 91-98, doi:10.1016/S0273-1223(99)00543-0.
134. Gagné F, Ridal J, Blaise C and Brownlee B. Toxicological effects of geosmin and 2-methylisoborneol on rainbow trout hepatocytes. Bulletin of Environmental Contamination and Toxicology. 1999; 63: 174-180, doi:10.1007/s001289900963.
135. Li Y, Zhang X and Chen C. Review on the tastes and odors compounds in drinking water of China. Environmental Science. 2009; 30: 583-588, doi:10.13227/j.hjkx.2009.02.041.
136. Wu T, Rui M Zhu, H, Li K and Chu L. Analysis of cyanobacteria odorous compounds and study on influence factor of odor levels in Lake Chaohu. Environmental Science & Technology 2015; 38: 147-151, doi:10.3969/j.issn.1003-6504.2015.03.028.
137. Proulx F, Rodriguez M.J, Serodes, J.B and Bouchard C. Spatio-temporal variability of tastes and odors of drinking water within a distribution system. Journal of Environmental Management. 2012; 105: 12-20, doi:10.1016/j.jenvman.2012.03.006.
138. Smith J.L, Boyer G.L and Zimba P.V. A review of cyanobacterial odorous and bioactive metabolites: Impacts and management alternatives in aquaculture. Aquaculture. 2008; 280: 5-20, doi:10.1016/j.aquaculture.2008.05.007.
139. Schweitze L and Khiari D. Olfactory and chemical analysis of taste and odor episodes in drinking water supplies. Reviews in Environmental Science and Biotechnology. 2004; 3: 3-13, doi:10.1023/B:RESB.0000040012.94870.48.
140. Kehoe M.J, Chun K.P and Baulch, H.M. Who Smells? Forecasting Taste and Odor in a Drinking Water Reservoir. Environmental Science & Technology. 2015; 49: 10984-10992, doi:10.1021/acs.est.5b00979.
141. Xu X. Study on the detection and prewarning system of odor in the water source of Tailake. Nanjing University of Science & Techinology Najing, 2010.
142. Song, L, Li L, Chen W and Gan N. Research progress on the off-flavours and secondary metabolites of algae in the aquatic environment. Acta Hydrobiologica Sinica. 2004; 28: 434-439.
143. Chung S.Y, Johnsen P.B and Klesius P.H. Development of an ELISA using polyclonal antibodies specific for 2-methylisoborneol. Journal of Agricultural and Food Chemistry 1990; 38: 410-415, doi:10.1021/jf00092a015.
144. Krasner S.W, McGuire M.J and Ferguson V.B. Tastes and odors: The flavor profile method. Journal American Water WorksAssociation.1985;77:34-39, doi:https://doi.

org/10.1002/j.1551-8833.1985.tb05504.x.

145. Rashash D.M.C,Dietrich A.M and Hoehn R.C. FPA of selected odorous compounds. Journal American Water Works Association.1997;89:131-141. doi:DOI: 10.1002/j.1551-8833.1997.tb08213.x.
146. Saito K, Okamura K and Kataoka H. Determination of musty odorants, 2-methylisoborneol and geosmin, in environmental water by headspace solid-phase microextraction and gas chromatography–mass spectrometry. Journal of Chromatography A. 2008; 1186:434-437, doi:10.1016/j.chroma.2007.12.078.
147. Watson S.B, Brownlee B, Satchwill T and Hargesheimer E.E. Quantitative analysis of trace levels of geosmin and MIB in source and drinking water using headspace SPME. Water Research. 2000; 34: 2818-2828, doi:https://doi.org/10.1016/S0043-1354(00)00027-0.
148. Deng X, Liang G, Chen J, Qi M and Xie P. Simultaneous determination of eight common odors in natural water body using automatic purge and trap coupled to gas chromatography with mass spectrometry.Journal of Chromatography A 2011;1218:3791-3798, doi:10.1016/j.chroma.2011.04.041.
149. Chen W, Westerhoff P, Leenheer J.A. and Booksh, K. Fluorescence excitation−emission matrix regional integration to quantify spectra for dissolved organic matter. Environmental science & technology. 2003; 37:5701-5710, doi:10.1021/es034354c.
150. Shin H.S and Ahn H.S. Simple, rapid, and sensitive determination of odorous compounds in water by GC–MS. Chromatographia. 2004; 59:107-113, doi:https://doi.org/10.1365/s10337-003-0125-4.
151. Malleret L, Dugay J, Bruchet, A and Hennion M.-C. Simultaneous determination of “earthy-musty” odorous haloanisoles and their corresponding halophenols in water samples using solid-phase microextraction coupled to gas chromatography with electron-capture detection. Journal of Chromatography A. 2003; 999:135-144. doi:10.1016/s0021-9673(03)00526-0.
152. Taehyo I and se-uk c. Lee In-jeong; Lee Gyeong-rak; Park Jeong-ja. Determination of geosmin and 2-MIB in Nakdong River using headspace solid phase microextraction and GC-MS. Analytical Science & Technology. 2013; 26: 326-332, doi:10.5806/ast.2013.26.5.326.
153. Devi A, Chiu Y-T, Hsueh H.-T and Lin T.-F. Quantitative PCR based detection system for cyanobacterial geosmin/2-methylisoborneol (2-MIB) events in drinking water sources: Current status and challenges. Water research.2020;188:116478-116478, doi:10.1016/j.watres.2020.116478.
154. Yu J, Che K,Su M, Yang M and Liu D. Influence of nutrient sources on Anabaena spiroides growth and odorous compounds production characteristics. Envirenmental Science. 2011; 32:2254-2259, doi:10.13227/j.hjkx.2011.08.006.
155. Gerber N and Lechevalier H. Geosmin, an earthy-smelling substance isolated from actinomycetes. Applied Microbiology.1965; 13:935-938.
156. Medsker L.L, Jenkins D, Thomas J.F and Koch C. Odorous compounds in natural waters. 2-exo-Hydroxy-2-methylbornane, the major odorous compound produced by several actinomycetes. Environmental Science & Technology.1969;3:476-477, doi:10.1021/es60028a008.
157. Rosen A, Mashni C and Safferman, R. Recent developments in the chemistry of odour in water: the cause of earthy/musty odour. WATER TREAT EXAM. 1970; 19:106-119, doi:10.1016/j.chroma.2015.09.008.
158. Zhang R, Qi F, Liu C, Zhang Y, Wang Y, Song Z, Kumirska, J and Sun D. Cyanobacteria derived taste and odor characteristics in various lakes in China: Songhua Lake, Chaohu Lake and Taihu Lake. Ecotoxicology and Environmental Safety.2019;181:499-507, doi:10.1016/j.ecoenv.2019.06.046.
159. Jones G.J and Korth W. In situ production of volatile odour compounds by river and reservoir phytoplankton populations in Australia. Water Science and Technology.1995; 31: 145-151, doi:https://doi.org/10.1016/0273-1223(95)00469-4.
160. Wnorowski A and Scott W. Incidence of off-flavors in South African surface waters. Water Science & Technology.1992; 25:225-232, doi:https://doi.org/10.2166/wst.1992.0056.
161. Westerhoff P, Rodriguez-Hernandez M, Baker L and Sommerfeld M. Seasonal occurrence and degradation of 2-methylisoborneol in water supply reservoirs. Water Research 2005; 39: 4899-4912.
162. Zhang T, Li L, Zuo Y, Zhou Q and Song L. Biological origins and annual variations of earthy-musty off-flavours in the Xiong he Reservoir in China. Aqua 2010, 59, 243, doi:https://doi.org/10.2166/aqua.2010.019.
163. Ma Z, Xie P, Chen J, Niu Y, Tao M and Qi M. Microcystis blooms influencing volatile organic compounds concentrations in Lake Taihu. Fresenius Environmental Bulletin 2013; 22: 95-102, doi:10.1080/10643389.2012.671750.
164. Qi M, Chen J, Sun X, Deng X, Niu Y and Xie P. Development of models for predicting the predominant taste and odor compounds in Taihu Lake, China. Plos One 2012, 7, doi:10.1371/journal.pone.0051976.
165. 167. 165. Hashimoto T. Phytoplankton and nutrient in musty odour occurring reservoirs in Japan. Proceedings of 36th Zenkoku suidokenkyn Happyolai 1995; 493- 495.
166. 168. 166. Chen J, Xie P,Ma Z, Niu Y, Tao M, Deng X and Wang, Q. A systematic study on spatial and seasonal patterns of eight taste and odor compounds with relation to various biotic and abiotic parameters in Gonghu Bay of Lake Taihu, China. Science of the Total Environment. 2010; 409:314-325. doi:10.1016/j.scitotenv.2010.10.010.
167. 169. 167. Alghanmi H.A, Alkam F.a.M and Al-Taee M.M. Effect of light and temperature on new cyanobacteria producers for geosmin and 2-methylisoborneol. Journal of Applied Phycology 2018; 30:319-328, doi:10.1007/s10811-017-1233-0.
168. Smith V.H, Sieber-Denlinger J, Jr F.d, Campbell S, Pan S, Randtke S.J and et al. Managing taste and odor problems in a eutrophic drinking water reservoir. Lake and Reservoir Management. 2002;

18, 319-323, doi:10.1080/07438140209353938.

169. Downing J.A, Watson S.B and McCauley E. Predicting cyanobacteria dominance in lakes. Canadian Journal of Fisheries and Aquatic Sciences 2001; 58: 1905-1908, doi:10.1139/cjfas-58-10-1905.
170. Board O.W.R. Justification for chlorophyll-a criteria to protect the public and private water supply beneficial use of sensitive water supplies. Availabe online: (accessed on)
171. Watson S.B, Charlton M, Rao Y.R, Howell, T, Ridal J, Brownlee B, and et al. Off flavours in large waterbodies: physics, chemistry and biology in synchrony. Water Science and Technology. 2007; 55, doi:10.2166/wst.2007.155.
172. Dzialowski A.R, Smith V.H, Huggins D.G, deNoyelles F, Lim N.-C, Baker D.S and Beury J.H. Development of predictive models for geosmin-related taste and odor in Kansas, USA, drinking water reservoirs. Water Research 2009; 43: 2829-2840, doi:10.1016/j.watres.2009.04.001.
173. Yagi M. Musty odour problems in Lake Biwa 1982-1987. Water Science & Technology 1988; 20:133-142, doi:https://doi.org/10.2166/wst.1988.0234.
174. Xu Y, Li W, Wu W and Zhang Y. Study on aquaticd off-flavors in eutrophic Donghu Lake. Acta Ecologica Sinica. 1999; 19: 212-216.
175. Watson S.B, Ridal J and Boyer G.L. Taste and odour and cyanobacterial toxins: impairment, prediction, and management in the Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences. 2008; 65:1779-1796, doi:10.1139/f08-084.
176. Ma Z, Niu Y, Xie P, Chen J, Tao M and Deng X. Off-flavor compounds from decaying cyanobacterial blooms of Lake Taihu. Journal of Environmental Sciences. 2013; 25: 495-501, doi:10.1016/s1001-0742(12)60101-6.
177. Sun D, Yu J, Yang M, An W, Zhao Y, Lu N, Yuan S and Zhang D. Occurrence of odor problems in drinking water of major cities across China. Frontiers of Environmental Science & Engineering. 2014; 8:411-416, doi:10.1007/s11783-013-0577-1.
178. Kim Y, Lee Y, Gee C.S and Choi E. Treatment of taste and odor causing substances in drinking water. Water Science and Technology. 1997; 35:29-36, doi:http://dx.doi.org/10.1016/S0273-1223(97)00148-0.
179. Lin T.F, Wong J.Y and Kao H.P. Correlation of musty odor and 2-MIB in two drinking water treatment plants in South Taiwan. Science of the total environment. 2002; 289: 225-235, doi:10.1016/s0048-9697(01)01049-x.
180. Ji R, Lv, X and Li X. Removal of taste and odor compounds from eutrophic raw water. Water & Wastewater Engineering 2004; 30: 8-13, doi:10.13789/j.cnki.wwe1964.2004.10.003.
181. Bruce D, Westerhoff P and Brawley-chesworth, A. Removal of 2-methylisoborneol and geosmin in surface water treatment plants in Arizona. Journal of Water Supply: Research and Technology – AQUA. 2002; 51: 183-197, doi:DOI: 10.2166/aqua.2002.0016.
182. Jung S, Baek K and Yu M. Treatment of taste and odor material by oxidation and adsorption. Water science and technology: a journal of the International Association on Water Pollution Research. 2004; 49: 289, doi:https://doi.org/10.2166/wst.2004.0588.
183. Kim T.K. Removal of geosmin with inter-chlorination using powdered activated carbon in full-scale WTP.Desalination and Water Treatment.2015;532850-2855, doi:10.1080/19443994.2014.931532.
184. Zamyadi A, Henderson R, Stuetz R, Hofmann R, Ho L and Newcombe G. Fate of geosmin and 2-methylisoborneol in full-scale water treatment plants. Water Research. 2015; 83: 171-183, doi:10.1016/j.watres.2015.06.038.
185. Bai M, Huang X, Zhong Z, Cao M and Gao M. Comparison of center dot OH and NaClO on geosmin degradation in the process of algae colonies inactivation at a drinking water treatment plant. Chemical Engineering Journal. 2020; 393, doi:10.1016/j.cej.2019.123243.
186. Shang D. Study on the disinfection by-products in drinking water and the method of determination. Nanjing University of Science and Technology, Nanjing 2004.
187. Nguyen M.-L, Paul Westerhoff, P.E, Baker L, Hu Q, Esparza-Soto M and Sommerfeld, M. Characteristics and reactivity of algae-produced dissolved organic carbon. Journal of Environmental Engineering.2005;131:1574-1582, doi:DOI: 10.1061/(ASCE)0733-9372(2005)131:11(1574).
188. Lin JH, Kao WC, Tsai KP, Chen CY. A novel algal toxicity testing technique for assessing the toxicity of both metallic and organic toxicants. Water Research 2005; 39: 1869-1877, doi:http://dx.doi.org/10.1016/j.watres.2005.02.015.
189. Nalewajko C, Lean DRS. Growth and excretion in planktonic algae and bacteria. Journal of Phycology 1972; 8: 361-366, doi:10.1111/j.1529-8817.1972.tb04049.x.
190. Rositano J, Newcombe G, Nicholson B, Sztajnbok P. Ozonation of nom and algal toxins in four treated waters. Water Research 2001, 35, 23-32, doi:http://dx.doi.org/10.1016/S0043-1354(00)00252-9.
191. Buffle J. Complexation reactions in aquatic systems: An analytical approach; Wiley & Sons Ltd./Ellis Horwood Ltd.: Chichester 1988; https://doi.org/10.1002/aheh.19890170220.
192. Hall T, Hart J, Croll B, Gregory R. Laboratory-scale investigations of algal toxin removal by water treatment. Water and Environment Journal 2000; 14: 143-149, doi:10.1111/j.1747-6593.2000.tb00241.x.
193. Widrig DL, Gray KA, McAuliffe KS. Removal of algal-derived organic material by preozonation and coagulation: Monitoring changes in organic quality by pyrolysis-GC-MS. Water Research 1996; 30: 2621-2632, doi:10.1016/S0043-1354(96)00162-5.
194. Bruchet A Rousseau C, Mallevialle J. Pyrolysis-GC-MS for investigating high-molecular-weight THM precursors and other refractory organics. Journal-American Water Works Association 1990; 82: 66-74, doi:https://doi.org/10.1002/j.1551-8833.1990.tb07022.x.
195. Fogg G. The ecological significance of extracellular products of phytoplankton photosynthesis. Botanica Marina 1983; 26: 3-14, doi:https://doi.org/10.1515/botm.1983.26.1.3.
196. Myklestad SM. Release of extracellular products by phytoplankton with special emphasis on polysaccharides. Science of The Total Environment 1995; 165: 155-164, doi:http://dx.doi.org/10.1016/0048-9697(95)04549-G.

197. Bernhardt H, Hoyer O, Schell H, Lüsse B. Reaction mechanisms involved in the influence of algogenic organic matter on flocculation. Zeitschrift für Wasser-und Abwasser-Forschung 1985; 18: 18-30.
198. Hoyer O, Lusse B, Bernhardt H. Isolation and characterization of extracellular organic matter (EOM) from algae. Zeitschrift für Wasser-und Abwasser-Forschung 1985; 18: 76-90.
199. Morris JC. Kinetics of reactions between aqueous chlorine and nitrogen compounds; John Wiley & Sons: New York, NY, 1967.
200. Chin YP, Aiken G, O'Loughlin E. Molecular weight, polydispersity, and spectroscopic properties of aquatic humic substances. Environmental Science & Technology 1994; 28: 1853-1858, doi:10.1021/es00060a015.
201. Her N, Amy G, Park HR, Song M. Characterizing algogenic organic matter (AOM) and evaluating associated NF membrane fouling. Water Research 2004; 38: 1427-1438, doi:10.1016/j.watres.2003.12.008.
202. Zhang Y. Biological-optical properties of Lake Taihu and its ecological significance. Graduate University of Chinese Academy of Sciences, Beijing, 2005.
203. Ma C, Pei H, Hu W, Wang Y, Xu H, Jin Y. The enhanced reduction of C- and N-DBP formation in treatment of source water containing Microcystis aeruginosa using a novel CTSAC composite coagulant. Science of the Total Environment 2017; 579: 1170-1178, doi:10.1016/j.scitotenv.2016.11.099.
204. McKnight DM, Harnish R, Wershaw RL, Baron JS, Schiff S. Chemical characteristics of particulate, colloidal, and dissolved organic material in Loch Vale Watershed, Rocky Mountain National Park. Biogeochemistry 1997; 36: 99-124, doi:10.1023/A:1005783812730.
205. Wetzel RG, Likens GE. Limnological analysis; Springer: 2000.
206. Westerhoff P, Mash H. Dissolved organic nitrogen in drinking water supplies: A review. Aqua 2002; 51: 415-448, doi:DOI: 10.2166/aqua.2002.0038.
207. Brown M, Garland C, Jeffrey S, Jameson I Leroi J. The gross and amino acid compositions of batch and semi-continuous cultures of Isochrysis sp.(clone T. ISO), Pavlova lutheri and Nannochloropsis oculata. Journal of Applied Phycology 1993; 5: 285-296.
208. Tomlinson A, Drikas M, Brookes JD. The role of phytoplankton as pre-cursors for disinfection by-product formation upon chlorination. Water Research 2016; 102: 229-240, doi:10.1016/j.watres.2016.06.024.
209. Hong HC, Wong MH, Liang Y. Amino acids as precursors of trihalomethane and haloacetic acid formation during chlorination. Archives of Environmental Contamination and Toxicology 2009; 56: 638-645, doi:10.1007/s00244-008-9216-4.
210. Paralkar A, Edzwald JK. Effect of ozone on EOM and coagulation. Journal-American Water Works Association 1996; 88: 143-154, doi:https://doi.org/10.1002/j.1551-8833.1996.tb06540.x.
211. Ma J, Liu W. Effectiveness and mechanism of potassium ferrate (VI) preoxidation for algae removal by coagulation. Water Research 2002; 36: 871-878, doi:https://doi.org/10.1016/S0043-1354(01)00282-2.
212. Croue J, Debroux J, Amy G, Aiken G, Leenheer J. Natural organic matter: structural characteristics and reactive properties. Formation and control of disinfection by-products in drinking water 1999; 65-93.
213. Croue J, Violleau D, Labouyrie-Rouillier L, Leenheer J, Aiken G. DBP formation potentials of hydrophobic and hydrophilic NOM fractions: A comparison between a low and a high-humic water. In Proceedings of ABSTRACTS OF PAPERS OF THE AMERICAN CHEMICAL SOCIETY; pp. U722-U722.
214. Escobar IC, Randall AA, Taylor JS. Bacterial growth in distribution systems: effect of assimilable organic carbon and biodegradable dissolved organic carbon. Environmental science & technology 2001; 35: 3442-3447, doi:https://doi.org/10.1021/es0106669.
215. Rook JJ. Formation of haloforms during chlorination of natural waters. Water Treat Examination 1974; 23: 234-243.
216. Bellar T, Lichtenberg J Kroner R. Occurrence of organohalides in chlorinated drinking waters. J Am Water Works Assoc 1974; 66: 703-706, doi:https://doi.org/10.1002/j.1551-8833.1974.tb02129.x.
217. Krasner SW, Weinberg HS, Richardson SD, Pastor SJ, Chinn R, Sclimenti, MJ, et al. Occurrence of a new generation of disinfection byproducts. Environmental Science & Technology 2006; 40: 7175-7185, doi:10.1021/es060353j.
218. Bond T, Huang J, Templeton MR, Graham N. Occurrence and control of nitrogenous disinfection by-products in drinking water – A review. Water research 2011; 45: 4341-4354, doi:10.1016/j.watres.2011.05.034.
219. Trehy ML, Yost RA, Miles CJ. Chlorination byproducts of amino acids in natural waters. Environmental science & technology 1986; 20: 1117-1122, doi:https://doi.org/10.1021/es00153a006.
220. 222. 220. Yang X, Shang C, Lee W, Westerhoff P, Fan C. Correlations between organic matter properties and DBP formation during chloramination. Water research 2008; 42: 2329-2339, doi:https://doi.org/10.1016/j.watres.2007.12.021.
221. 223. 221. Yang X, Fan C, Shang C, Zhao Q. Nitrogenous disinfection byproducts formation and nitrogen origin exploration during chloramination of nitrogenous organic compounds. Water Research 2010; 44: 2691-2702, doi:http://dx.doi.org/10.1016/j.watres.2010.01.029.
222. 224. 222. Gao N, Chu W, Yan M, Xu B. Formation and control of disinfection by-products in drinking water; China Construction Industry Press: Beijing, 2011.
223. 225. 223. Scully FE, Howell GD, Kravitz R, Jewell JT, Hahn V, Speed M. Proteins in natural waters and their relation to the formation of chlorinated organics during water disinfection. Environmental Science & Technology 1988; 22: 537-542, doi:https://doi.org/10.1021/es00170a009.
224. 226. 224. Wardlaw V, Perry R, Graham N. Role of algae as trihalomethane precursors: A review. Journal of Water Supply Research and Tecnology-Aqua 1991; 40: 335-345.
225. Hureiki L, Croue J, Legube B. Chlorination studies of free and combined amino acids. Water Research 1994; 28: 2521-2531, doi:https://doi.org/10.1016/0043-1354(94)90070-1.
226. Reckhow DA, Platt TL, MacNeill A, McClellan JN. Formation and

degradation of DCAN in drinking waters. Journal of Water Supply Research and Technology, AQUA 2001; 50: 1-13.

227. Bond T, Templeton MR, Graham N. Precursors of nitrogenous disinfection by-products in drinking water – A critical review and analysis. Journal of Hazardous Materials 2012; 235-236: 1-16, doi:10.1016/j.jhazmat.2012.07.017.
228. Fang J, Ma J, Yang X, Shang C. Formation of carbonaceous and nitrogenous disinfection by-products from the chlorination of Microcystis aeruginosa. Water research 2010; 44: 1934-1940, doi:https://doi.org/10.1016/j.watres.2009.11.046.
229. Xie P, Ma J, Fang J, Guan Y, Yue S, Li X, Chen L. Comparison of permanganate preoxidation and preozonation on algae containing water: Cell integrity, characteristics, and chlorinated disinfection byproduct formation. Environmental science & technology 2013; 47: 14051-14061, doi:https://doi.org/10.1021/es4027024.
230. Oliver BG. Dihaloacetonitriles in drinking-water - Algae and fulvic-acid as precursors. Environmental Science & Technology 1983; 17: 80-83, doi:https://doi.org/10.1021/es00108a003.
231. Muellner MG, Wagner ED, McCalla K, Richardson SD, Woo YT, Plewa MJ. Haloacetonitriles vs. regulated haloacetic acids: are nitrogen-containing DBPs more toxic? Environmental science & technology 2007; 41: 645-651, doi:https://doi.org/10.1021/es0617441.
232. Richardson SD, Plewa MJ, Wagner ED, Schoeny R, DeMarini DM. Occurrence, genotoxicity, and carcinogenicity of regulated and emerging disinfection by-products in drinking water: A review and roadmap for research. Mutation Research/Reviews in Mutation Research 2007; 636: 178-242, doi:10.1016/j.mrrev.2007.09.001.
233. 235. 233. Geider RJ, Roche JL. Redfield revisited: variability of C: N: P in marine microalgae and its biochemical basis. European Journal of Phycology 2002; 37: 1-17, doi:doi:10.1017/S0967026201003456.
234. 236. 234. El-Aty AMA, Ibrahim MBM, El-Dib MA, Radwan EK. Influence of chlorine on algae as precursors for trihalomethane and haloacetic acid production. World Applied Sciences Journal 2009; 6: 1215-1220, doi:http://www.idosi.org/wasj/wasj6(9)/8.pdf.
235. 237. 235. Pomes ML, Larive CK, Thurman EM, Green WR, Orem WH, Rostad CE, et al. Sources and haloacetic acid/trihalomethane formation potentials of aquatic humic substances in the Wakarusa River and Clinton Lake near Lawrence, Kansas. Environmental science & technology 2000; 34: 4278-4286, doi:https://doi.org/10.1021/es991376j.
236. 238. 236. Hong HC, Wong MH, Mazumder A, Liang Y. Trophic state, natural organic matter content, and disinfection by-product formation potential of six drinking water reservoirs in the Pearl River Delta, China. Journal of Hydrology 2008; 359: 164-173, doi:https://doi.org/10.1016/j.jhydrol.2008.06.024.
237. Suksomjit M, Nagao S, Ichimi K, Yamada T, Tada K. Variation of dissolved organic matter and fluorescence characteristics before, during and after phytoplankton bloom. Journal of oceanography 2009; 65: 835-846, doi:10.1007/s10872-009-0069-x.
238. Huang W, Chu H, Dong B. Characteristics of algogenic organic matter generated under different nutrient conditions and subsequent impact on microfiltration membrane fouling. Desalination 2012; 293: 104-111, doi:10.1016/j.desal.2012.03.001.
239. Wert EC, Rosario-Ortiz FL. Intracellular organic matter from cyanobacteria as a precursor for carbonaceous and nitrogenous disinfection byproducts. Environmental Science & Technology 2013; 47: 12617-12617, doi:10.1021/es404500p.
240. Plummer JD, Edzwald JK. Effect of ozone on algae as precursors for trihalomethane and haloacetic acid production. Environmental Science & Technology 2001; 35: 3661-3668, doi:https://doi.org/10.1021/es0106570.
241. Hoehn RC, Barnes DB, Thompson BC, Randall CW, Grizzard TJ, Shaffer PT. Algae as sources of trihalomethane precursors. Journal (American Water Works Association) 1980; 344-350, doi:https://doi.org/10.1002/j.1551-8833.1980.tb04528.x.
242. Yang X, Shang C, Westerhoff P. Factors affecting formation of haloacetonitriles, haloketones, chloropicrin and cyanogen halides during chloramination. Water Research 2007; 41: 1193-1200, doi:10.1016/j.watres.2006.12.004.
243. Pourmoghaddas H, Stevens AA. Relationship between trihalomethanes and haloacetic acids with total organic halogen during chlorination. Water Research 1995; 29: 2059-2062, doi:10.1016/0043-1354(95)00026-H.
244. Xie Y. Disinfection byproducts in drinking water: Formation, analysis, and control; CRC Press: Florida, 2003.
245. Heller-Grossman L, Idin A, Limoni-Relis B, Rebhun M. Formation of cyanogen bromide and other volatile DBPs in the disinfection of bromide-rich lake water. Environmental science & technology 1999, 33, 932-937, doi:https://doi.org/10.1021/es980147e.
246. Zhang X, Echigo S, Minear RA, Plewa MJ. Characterization and comparison of disinfection by-products of four major disinfectants. In Proceedings of ACS Symposium Series; pp. 299-315.
247. Bernat-Quesada F, Alvaro M, Garcia H, Navalon S. Impact of chlorination and pre-ozonation on disinfection by-products formation from aqueous suspensions of cyanobacteria: Microcystis aeruginosa, Anabaena aequalis and Oscillatoria tenuis. Water Research 2020, 183, doi:10.1016/j.watres.2020.116070.
248. Chen Y, Bai F, Li Z, Xie P, Wang Z, Feng X, Liu Z, Huang LZ. UV-assisted chlorination of algae-laden water: Cell lysis and disinfection byproducts formation. Chemical Engineering Journal 2020; 383, doi:10.1016/j.cej.2019.123165.
249. Chen S, Deng J, Li L, Gao N. Evaluation of disinfection by-product formation during chlor(am)ination from algal organic matter after UV irradiation. Environmental Science and Pollution Research 2018; 25: 5994-6002, doi:10.1007/s11356-017-0918