12-1-2021

Quaternary Ammonium Compounds as Precursors to Disinfection Byproducts

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QUATERNARY AMMONIUM COMPOUNDS AS PRECURSORS TO DISINFECTION BYPRODUCTS

By

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2018

A thesis submitted in partial fulfillment
of the requirements for the

Master of Science in Engineering – Civil and Environmental Engineering

Department of Civil and Environmental Engineering and Construction
Howard R. Hughes College of Engineering
The Graduate College

University of Nevada, Las Vegas
December 2021

UNLV | GRADUATE COLLEGE

Thesis Approval

The Graduate College
The University of Nevada, Las Vegas

November 17, 2021

This thesis prepared by

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entitled

Quaternary Ammonium Compounds as Precursors to Disinfection Byproducts

is approved in partial fulfillment of the requirements for the degree of

Master of Science in Engineering – Civil and Environmental Engineering
Department of Civil and Environmental Engineering and Construction

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Abstract

Disinfection of drinking water is crucial for providing a safe potable water supply. Some of the most common disinfectants employed are chlorine, chloramines and ozone. While these disinfectants successfully inactivate pathogenic microorganisms, they simultaneously react with inorganic or organic precursors to form toxic byproducts, most commonly referred to as disinfection byproducts (DBPs). While there is a wide variety of organic matter and disinfection byproducts known, in this study we have looked into the potential of certain quaternary ammonium compounds (QACs) to act as precursors to form DBPs, namely trihalomethanes (THMs), haloacetonitriles (HANs) and nitrosamines.

The ubiquitous nature of QACs in municipal, hospital and industrial wastewater has motivated the need to investigate the potential of these compounds to form DBPs upon reaction with chlorine, monochloramine and ozone. Seven QACs, namely benzalkonium chloride (BZK), benzethonium chloride (BEC), tetraethylammonium chloride (TEAC), tetramethylammonium chloride (TMAC), cetylpyridinium chloride (CPC), dodecyltrimethylammonium chloride (DTMA), and cetyltrimethylammonium (CTMA), were considered in this study based on their structural differences as well as their presence in consumer products. Due to incomplete removal during wastewater treatment, these QACs end up in wastewater effluent, and, ultimately, in surface waters serving as drinking water sources.

When these seven QACs spiked at 0.1 mM were subjected to reaction with free chlorine (5 mg/L as Cl_2), trichloromethane was the dominant THM that was formed in the order of

TMAC<TEAC<BEC<CPC<DTMA<CTMA<BZK ranging from a molar yield of approximately 155 to 6700 µmol/mol. Upon reaction with monochloramine (140 mg/L as Cl_2), trichloromethane formed was in the order of TMAC<TEAC<DTMA<CTMA<BZK<BEC<CPC ranging from a molar yield of approximately 36 to 3140 µmol/mol. In the presence of bromide, a considerable shift in the THM speciation to more brominated species was observed as the concentration of bromide in the matrix increased from 0 to 50 µg/L and from 50 to 500 µg/L.

In the case of HANs, dichloroacetonitrile was the dominant species formed upon reaction between the seven QACs spiked at 0.1 mM and monochloramine (140 mg/L as Cl_2) in the order of TMAC<TEAC<CTMA<DTMA<CTMA<BZK<BEC<CPC with molar yield values ranging from 4.5 to 4000 µmol/mol. Dichloroacetonitrile formation by QACs spiked at 0.1 mM on reaction with free chlorine (5 mg/L) followed the order of TMAC<TEAC<DTMA<BEC<CTMA<BZK<CPC ranging from 8.3 to 980 µmol/mol. In the presence of bromide, a shift in speciation was observed as the concentration of bromide in the matrix increased from 0 to 50 µg/L and from 50 to 500 µg/L.

Formation of *N*-nitrosodimethylamine (NDMA) was observed only during chloramination. Of the seven QACs included in this study, benzethonium chloride and benzalkonium chloride were the only compounds that yielded detectable concentrations of NDMA. This is the first time the NDMA yield for benzethonium chloride has been reported as a result of chloramination, and the yield is approximately 3 times higher compared to the yield from the widely studied benzalkonium chloride. No other nitrosamines were detected, even though TEAC could potentially form *N*-nitrosodiethylamine.

This is the first study that shows that during the process of disinfection, BEC, CPC and BZK could form HANs, BEC and CPC could form THMs, and BEC could form NDMA. This study also investigated the potential of QACs to form brominated disinfection byproducts during disinfection treatment in the presence of bromide. Lastly, the prediction of DBP concentrations formed during the disinfection of water containing environmentally relevant concentrations of QACs was also performed. THMs and HANs formed either due to chlorination or chloramination of all of the 7 QACs (400 µg/L) was relatively low, i.e., less than 2 µg/L and 1 µg/L respectively, based on the prediction. The predicted concentration of NDMA formed due to chloramination of BEC and BZK are in the range of 22-50 ng/L. Although the DBPs formed due to the reaction of QACs with disinfectants is of lower concentration compared to regulations, it will still contribute to the total DBPs formed in a water treatment facility. Hence, care must be taken for proper removal of these QACs prior to discharge of wastewater effluent or disinfection of drinking water.

Acknowledgements

I would like to express my deepest gratitude to my advisor and mentor Dr. Erica Marti, for her continued support, guidance, and encouragement throughout this research project. I would also like to thank her for providing me with graduate research assistantship throughout the program as it not only helped me financially, but also enhanced my skills and knowledge in the environmental, water and wastewater discipline. This research would not have been possible without your continued dedication and one-on-one weekly meetings to discuss research and academic progress.

I would also like to extend my gratitude to the current UNLV graduate student George William Kajjumba for taking the time out of his busy schedule in assisting me with the experiments such as chemical extraction and chemical analysis. I am also thankful to the Graduate Student and Professional Association for providing the financial support to conduct this research.

I would like to thank all my friends, colleagues, and other graduate students, to name a few, Suraj Venkat Pochampally, Sneha Ravikumar, Maithilee Bedre, Christina Obra, Meena Ejjada, Kazi Shafayet Haider, and Amit Gajurel for your support through this research project.

Finally, I owe a part of my accomplishments to my mother-Krishnaveni, father-Krishnaswamy, sister-Indhumathi, for always having my back supporting me through this journey and my two little nephews for not failing to put a smile on my face. Thank you for your constant encouragement, support, and love.

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List of Abbreviations

BCAN	bromochloroacetonitrile
BDCAN	bromodichloroacetonitrile
BEC	benzethonium chloride
BZK	benzalkonium chloride
Br-DBP	brominated disinfection byproduct
CDC	Centers for Disease Control and Prevention
CPC	cetylpyridinium chloride
CTMA	cetyltrimethylammonium chloride
Cl-DBP	chlorinated disinfection byproduct
QACs	cationic surfactants
DBAN	dibromoacetonitrile
DBCM	dibromochloromethane
DCAN	dichloroacetonitrile
DCBM	dichlorobromomethane
DBP	disinfection byproduct
DBPFP	disinfection byproduct formation potential
DTMA	dodecyltrimethylammonium chloride
FP	formation potential
GC-MS	gas chromatography – mass spectrometry

HAA	haloacetic acid
HANs	haloacetonitriles
HANFP	haloacetonitrile formation potential
I-DBPs	iodinated disinfection byproducts
MCAN	monochloroacetonitrile
MCL	maximum contaminant level
NDBA	*N*-nitrosodibutylamine
NDEA	*N*-nitrosodiethylamine
NDPA	*N*-nitrosodiphenylamine
NDMA	*N*-nitrosodimethylamine
NMEA	*N*-nitrosomethylethylamine
NMOR	*N*-nitrosomorpholine
NOA	nitrosamine
NOM	natural organic matter
NPIP	*N*-nitrosopiperidine
NPYR	*N*-nitrosopyrrolidine
QACs	quaternary ammonium compounds
TBM	tribromomethane
TCAN	trichloroacetonitrile
TCM	trichloromethane
TEAC	tetraethylammonium chloride

TMAC	tetramethylammonium chloride
THMs	trihalomethanes
THMFP	trihalomethane formation potential
TTHM	total trihalomethanes
UNLV	University of Nevada, Las Vegas
USEPA	United States Environmental Protection Agency
UV	ultraviolet
WHO	World Health Organization
WWTP	wastewater treatment plant

Chapter 1 – Introduction, Objectives, Research Questions, and Hypotheses

Introduction

Over the recent years, contaminants such as pesticides, pharmaceuticals, surfactants, and disinfection byproducts (DBPs) have been a major concern due to their ubiquitous occurrence in the aqueous environment as well as drinking water treatment plants (Bond et al., 2011; Grassi et al., 2012; Mezzelani et al., 2018). In order to eliminate the risk of pathogenic infections, disinfection is carried out at the water treatment plants. Amongst the disinfectants, chlorine is widely employed on account of its efficiency in terms of cost as well as inactivation of the pathogens. However, besides killing bacteria and other pathogens, these disinfectants tend to react with natural organic matter (NOM) to form toxic compounds that are most commonly referred to as DBPs. Chlorine, upon reaction with NOM, tends to form compounds such as trihalomethanes (THMs) and haloacetic acids (HAAs). Four of the regulated trihalomethanes—namely trichloromethane, tribromomethane, dichlorobromomethane and chlorodibromomethane—are categorized as total trihalomethanes (TTHM). Five of the regulated HAAs are monochloroacetic acid, dichloroacetic acid, trichloroacetic acid, monobromoacetic acid and dibromoacetic acid.

In order to avoid/minimize the formation of TTHM and HAAs, alternative chemical disinfectants such as chloramines, ozone, and chlorine dioxide have come into play. While these disinfectants do not form as much TTHM or HAAs, chloramination and ozonation give rise to various other DBPs, which are unregulated, upon reaction with the relevant precursors. In fact, over 700 DBPs

were identified in drinking water systems (Richardson and Plewa, 2020). In addition to the regulated THMs and HAAs, two frequently reported DBP groups include haloacetonitriles (HANs) and nitrosamines which are unregulated at the federal level. There is a wide variety of health risks associated with the presence of DBPs in drinking water, and the toxicity varies, i.e., some largely unregulated DBP groups such as nitrosamines and HANs are relatively more toxic than some of the regulated DBP groups such as THMs and HAAs. In fact, toxicity varies within a group of DBPs based on speciation. For example, brominated and iodinated THMs and HANs are relatively more toxic than their chlorinated analogues (Echigo et al., 2004; Parvez et al., 2011; Hanigan et al., 2017). Epidemiological data over the years have shown that chronic exposure to THMs could lead to bladder cancer (Villanueva et al., 2015; Villanueva et al., 2003). Lin et al. (2016) demonstrated that zebrafish exposure to dichloroacetonitrile resulted in heart function alteration and neuronal function disturbance. Another study showed that chronic consumption of *N*-nitrosodimethylamine (NDMA), an unregulated but frequently detected nitrosamine, could possibly lead to upper aerodigestive tract cancer (Rogers et al., 1995).

Surfactants, most particularly the cationic species, are considered potential DBP precursors. In particular, they are known to react with chloramines to form NDMA (Chang et al., 2011; Ul'yanovskii et al., 2020). Quaternary ammonium-based surfactants, such as benzalkonium chlorides, are amongst the most common forms of cationic surfactants (Palmer et al., 2018).

Cationic surfactants, most of which are quaternary ammonium compounds (QACs) are found in numerous personal care products and also in fabric softeners, biocides and textile effluent (Bergé et al., 2017; Koner et al., 2011). One of the major sources of benzalkonium chlorides (BZK), a group of QACs, is hospital wastewater (Kümmerer et al. 1997; Carbajo et al., 2016). Being

biocidal in nature, biological degradation of QACs is limited (Koner et al., 2011), thereby allowing QACs to be introduced into the aquatic environment through wastewater effluent (Ishiguro et al., 2007; Zhu et al., 2018). It was also mentioned that the increase in the alkyl chain length of the quaternary ammonium compounds decreases the rate of biodegradation (Garcia et al., 2001).

With the rise in the number of DBPs discovered in aqueous systems, it is important for researchers to investigate the possible precursors as well. Knowledge and understanding of the variety of precursors are limited. There is very little information on QACs as precursors for DBPs upon ozonation, chloramination and chlorination. There is also very little evidence on the speciation of DBPs formed due to the reaction between QACs and disinfectants. All these research gaps form the basis of the objectives for this study. Moreover, given the current circumstances of COVID-19, quaternary ammonium compounds are found as an active ingredient in the composition of about 216 USEPA-recommended disinfectants that can be used to inactivate SARS-CoV-2 virus (Hora et al., 2020). This could mean a possible rise in the concentration of these quaternary ammonium compounds in water sources, thus making them potential DBP precursors of interest during the COVID-19 pandemic.

Objectives: This research aims to investigate the disinfection byproduct formation potentials of quaternary ammonium surfactants on reaction with chlorine, chloramine and ozone. We will also be looking at how speciation changes in the presence of bromide. The former part of the objective will provide us with information that fills several research gaps in the current literature related to quaternary ammonium compounds as precursors for DBPs. The latter part of the objective will give us insight into the speciation of DBPs during disinfection of water containing

bromide, which affects the toxicity of the disinfected water. The results of this research will be a valuable addition to the body of knowledge for the scientific community working on DBPs and its precursors.

Based on the objectives and identified research gaps, the research questions (RQ) and the corresponding hypotheses developed are as follows:

RQ 1: Do quaternary ammonium compounds have the potential to form THMs, HANs and NDMA upon treatment with chlorination?

Hypothesis 1 and Justification: Studies have shown that the benzalkonium chlorides, one group of cationic surfactants, have high potentials to form THMs & HAAs upon chlorination. However, their ability to form HANs has not been explored. *Since QACs have the ability to form both carbonaceous and nitrogenous DBPs* (Bond et al., 2011), *it can be expected that they may also form HANs, which are nitrogenous DBPs. QACs are not expected to form nitrosamines upon reaction with free chlorine, since secondary, tertiary and quaternary amines only form nitrosamines through reaction with chloramines* (Mitch and Sedlak, 2004; Selbes et al., 2013).

RQ 2: Do quaternary ammonium compounds have the potential to form THMs, HANs and NDMA upon treatment with chloramination?

Hypothesis 2 and Justification: Studies have shown the ability of certain quaternary ammonium compounds to form nitrosamines upon chloramination (Kemper et al., 2010a; Piazzoli et al., 2018a). However, there is a gap in literature regarding the potential of these surfactants to form THMs and HANs upon chloramination. *Although it can be expected that chloramination forms relatively fewer THMs compared to chlorination, the presence of nitrogen*

in monochloramine could accentuate the formation of HANs, which are nitrogenous DBPs. As these precursor compounds contain a dimethyl amine group (a group present in known NDMA precursors), I hypothesize that these QACs may yield NDMA upon chloramination.

RQ 3: Do quaternary ammonium compounds have the potential to form nitrosamines upon treatment with ozonation?

Hypothesis 3 and Justification: Studies have shown that ozonation of quaternary ammonium compounds reduced the surfactant concentration by 50% (Dantas et al., 2009), which means the surfactants reacted with ozone. However, that particular study did not investigate the resulting transformation products. *This means that, there is a possibility that some of the post-ozonation products could potentially be nitrosamines.* In fact, Carbajo et al. (2016) studied the degradation pathway of benzalkonium chloride, and two of the intermediates formed were a known precursor, benzyldimethylamine (Bei et al., 2016), and a probable precursor dodecyltrimethylamine, which could form *N*-nitrosamines due to the presence of the trimethylamine functional group (Sun et al., 2010; Liu et al., 2014).

RQ 4: Does the presence of bromide affect the speciation of DBPs upon chlorination, chloramination and ozonation?

Hypothesis 4 and Justification: *The presence of bromide will increase the concentration of brominated DBPs.* This is because studies show that the affinity of bromine species towards substitution, with respect to the chlorine species, is relatively higher (Bond et al., 2011). This aspect is important to confirm with experimental studies because brominated species are

relatively higher in toxicity. Thus, it is essential to understand if bromide presence is a concerning factor for all three common chemical disinfectants.

Chapter 2 – Literature Review

2.1 Disinfection in Water and Wastewater Treatment

The process of disinfection in water wastewater treatment is the inactivation of pathogenic microorganisms that could be detrimental to human and aquatic health by means of destroying the cellular structure of the microorganism, disrupting its metabolism and biosynthesis mechanisms (Hua and Reckhow, 2007). By doing so, we can ensure that the drinking water meets the regulations and standard and thereby does not cause pathogenic infections upon consumption.

Disinfection can be achieved by using certain chemicals such as chlorine, chlorine dioxide, monochloramines, and ozone, and by non-chemical methods such as ultraviolet light. The choice of the disinfectant is governed by various factors such as chemical and biological conditions of the water that is to be treated, operator preferences and most importantly cost. While these chemical disinfectants inactivate the pathogens, they tend to form toxic compounds upon reaction with certain organic matter. These toxic compounds formed are often referred to as DBPs and the organic matter which reacts with these disinfectants are called precursors.

2.2 Chlorine

Chlorine, a highly reactive and strong oxidizing agent, has been one of the most commonly used disinfectants in the water treatment process (National Academy of Sciences, 1980). Chlorine, once added to water, forms hypochlorous acid and hypochlorite ions. At a lower pH, the concentration of hypochlorous acid is dominant, and at higher pH the concentration of

hypochlorite ions is dominant. The term free chlorine refers to the combined concentrations of hypochlorous acid, hypochlorite ions and molecular chlorine in an aqueous solution. Chlorine is the most widely used disinfectant primarily due to its stability, cost and performance (Lee and Westerhoff, 2009; McGuire, 2006; Yang et al., 2012).

While chlorine is a very good disinfectant, it contributes to the formation of byproducts, some of which are toxic, upon reaction with natural organic matter such as humic and fulvic acid. The two main disinfection byproducts formed due to chlorination are TTHMs and HAAs. Four of the regulated trihalomethanes in the United States are trichloromethane, bromoform, dichlorobromomethane and chlorodibromomethane. The five regulated HAAs are monochloroacetic acid, dichloroacetic acid, trichloroacetic acid, monobromoacetic acid and dibromoacetic acid. Besides, the regulated DBPs such as THMs and HAAs, chlorination also contributes to the formation of highly toxic haloacetonitriles (Chu et al., 2017; Huang et al., 2016; Shah et al., 2011)

2.3 Chlorine Dioxide

Chlorine dioxide (ClO_2) is formed through the reaction between chlorine and sodium chlorite (Aieta and Berg, 1986). ClO_2 is an alternative to chlorine in the process of disinfection since it forms relatively low concentrations of TTHMs and HAAs when compared to the latter; however, ClO_2 gives rise to the production of regulated chlorite and chlorate ions (Gan et al., 2016). In drinking water treatment, about 70% of the chlorine dioxide is converted to chlorite and 30% to chlorate (Werdehoff and Singer, 1987). The United States Environmental Protection Agency (USEPA 1999) recommends not to use ClO_2 as a disinfectant if the oxidant demand is greater

than 1.4 mg/L. Otherwise, it will lead to excessive chlorite generation beyond its maximum contaminant level (MCL) of 1 mg/L.

2.4 Chloramines

Chloramines are formed by the reaction between chlorine and ammonia (USEPA 1999). On account of its biocidal property, there has been a progressive rise in the use of chloramines as a substitute to chlorine for secondary disinfection in order to limit the formation of halogenated disinfection byproducts such as THMs and HAAs (Krasner et al., 2013). However, studies have shown that chloramination contributes to the formation of NDMA, a carcinogenic agent (Roux et al., 2012). The NDMA formation mechanism during chloramination generally involves nucleophilic substitution between monochloramine and dimethylamine with unsymmetrical dimethylhydrazine as an intermediate compound (Mitch and Sedlak, 2001; Choi and Valentine, 2002). Research has also shown that chloramination contributes to highly toxic haloacetonitriles (Muellner et al., 2007; Yang et al., 2007).

2.5 Ozone

Ozone, due to its instability, is produced on site for the purpose of water disinfection by decomposition of diatomic oxygen (Weinberg et al., 1993). Ozone rapidly decomposes to oxygen. Since there is no long-term residual disinfectant concentration in the aqueous matrix to prevent any further microbial contamination, ozone is not used as a substitute to chlorine or chloramine as a final disinfectant in water treatment. However, it is very effective in the inactivation of certain microbes such as *Cryptosporidium* and *Giardia* which are generally resistant to chlorine disinfection (Park et al., 2016).

Ozone, too, contributes to the formation of disinfection byproducts. Some of the ozone-associated DBPs include aldehydes, ketones and di-carbonyls (Richardson et al., 2000). Studies have also shown that precursors with dimethylamine bonded to a nitrogen atom result in the formation of NDMA upon ozonation (Gunten et al., 2010; Marti et al., 2015). Additionally, it has been shown that ozonation followed by subsequent chloramination increased the formation of halonitromethanes via the oxidation of amines (Yang et al., 2012). In the presence of bromide, ozonation also contributes to bromate (Von Gunten and Hoigne, 1994), an inorganic DBP; however, this study focuses only on formation of organic DBPs.

2.6 UV

In order to avoid the formation of chlorinated disinfection byproducts, attempts have been made to provide a safer alternative disinfection technology that meets the stringent microbial contaminant standards employed in wastewater treatment systems. Ultraviolet irradiation is one such technology. The most pronounced technique to produce UV light is by passing an electric arc through a mercury vapor lamp. UV light in the electromagnetic spectrum resides between wavelengths of 40 – 400 nanometers and is classified into UV-A, UV-B and UV-C based on the wavelength (Malley and Burris, 2004). UV light is mostly germicidal at wavelengths below 265 nanometers, i.e. UV-C (Malley and Burris, 2004). The thymine bases on DNA and RNA of the microbes react with UV light and inhibit the replication of nucleic acids, thus preventing microbes from reproducing (Malley and Burris, 2004). Several studies were conducted that demonstrate the ability of UV technology to effectively inactivate a majority of *Giardia* cysts and *Cryptosporidium* oocysts in drinking water, which are typically resistant to chlorine disinfection (Linden et al., 2002; Liberti et al., 2003).

While UV disinfection minimizes the chances to form known disinfection byproducts, the technology is negatively affected by the presence of suspended particles in the water matrix (Jolis et al., 2001). The suspended particles shield the microbes from the UV radiation; thus, total disinfection of the water is prevented (Farrell et al., 2018; Jolis et al., 2001; Wu et al., 2005). Additionally, UV does not provide residual disinfection effect as well unlike chlorine. It is important to note that there is a lack of information on DBPs formed due to UV disinfection.

2.7 Hybrid Disinfection Process

The ultimate goal in the process of disinfection of wastewater is inactivating microbes while at the same time limiting the formation of disinfection byproducts. This can be achieved by using a combination of disinfection methods. One such approach involves primary disinfection using a non-chlorine disinfectant followed by secondary disinfection using a chlorine-based disinfectant. By doing so, the amount of chlorine used would be lower, thereby minimizing the formation of regulated disinfection byproducts (Chin and Berube, 2005). Fang et al. (2014) reported synergetic effects of UV/ozone co-exposure in the inactivation of *E. coli* with an increased efficiency at ozone concentrations as low as 0.05 mg/L. A similar conclusion was obtained for the inactivation of *B. subtilis* (Jung et al., 2008). Sequential UV-ozone disinfection showed a significant improvement in the inactivation of bacteriophage MS2 by 0.8 log kill at ozone and UV dose of 0.1 mg/L and 8.55 $mJ \cdot cm^{-2}$ respectively (Fang et al., 2014). Driedger et al. (2000) demonstrated a synergetic inactivation of *Cryptosporidium parvum* through sequential disinfection using ozone and free chlorine wherein ozone enabled the damage of the oocyst wall, thereby allowing enhanced permeation of chlorine through the oocyst wall. Likewise, a synergistic inactivation of *B. subtilis* spores was achieved using sequential ozone-chlorine

disinfection (Cho and Chung, 2003). A combined i.e., hybrid disinfection system is shown to reduce the formation of DBPs in certain cases (Cheema et al., 2018). However, in a study that investigated the formation of DBPs from amino acids under UV/chlorine conditions, the formation of DBPs increased (Deng et al., 2014; Gerrity et al., 2009). This is attributed to the photolytic degradation of the amino acids to primary amines which are well known precursors compared to the polyamines, thereby promoting enhanced DBP formation (Deng et al., 2014).

2.8 Regulations for DBPs

Disinfection is an important process in any drinking water treatment system. However, the toxic byproducts formed as a result of this process are a health concern. It is therefore essential to set regulatory limits for their presence in drinking water. The guidelines and/or regulatory limits for DBPs set by different agencies are listed in Table 2-1.

Table 2-1 Regulatory and guideline values of DBPs for drinking water

Regulatory agency	DBP	MCL/guideline value
U.S. EPA[a]	TTHMs	0.08
	5 HAAs	0.06
	Chlorite	1.0
	Bromate	0.01
WHO[b]	TrichloromethaneTrichloromet	0.3
	Tribromomethane	0.1
	Chloroacetic acid	0.02
	Dichloroacetic acid	0.05
	Trichloroacetic acid	0.2
	Chlorite	0.7

	Bromate	0.01
	Dichloroacetonitrile	0.02
	Dibromoacetonitrile	0.07
	N-Nitrosodimethylamine	1×10^{-4}
European Union standard	TTHMs	0.1
	Bromate	0.01

[a]USEPA (1998), [b]WHO (2011), [c]European Commission (1998)

2.9 Occurrence of DBPs

DBPs in source water, treated water and distribution systems are reported by multiple agencies and research groups across the world. In a study that analyzed the water samples from public water systems across Massachusetts, the average TTHM concentration was 40.9 µg/L with 10% of the samples exceeding 78.4 µg/L (Parvez et al., 2011). A similar study showed average TTHM concentrations of 21.6-59.9 µg/L in three water distribution systems located in France (Mouly et al., 2010). The tap waters in the Jinhua region of Zhejiang, China had average HAAs concentrations of 15.3 µg/L. The concentration of HAA5 in the waters of Taiwan was as high as 38.9 µg/L in the year 2007 (Chang et al., 2009). The surface waters in Newfoundland and Labrador, Canada had an average HAAs concentration of 129.2 µg/L whereas groundwater had an average HAAs concentration of 27.7 µg/L (Chowdhury, 2018). Several studies have observed the occurrence of HANs in natural waters. In a survey that sampled effluents from 23 different wastewater treatment plants across the US, observed HANs concentrations were 0.9-30 µg/L upon chlorination (Simpson and Hayes, 1998). The concentration of HANs in the wastewaters in Australia was as high as 36 ppb. Bond et al. (2011) stated in their review that the occurrence of these compounds in Scottish waters was quite low, on the order of 1 ppb. In general, disinfection

using chlorine tends to form THMs, HAAs and HANs; chloramine tends to form HANs and nitrosamines (NOAs), and ozone tends to form NOAs and aldehydes. Table 2-2 summarizes DBP concentrations as a result of using different disinfectants (as stated in literature).

Table 2-2 Range of reported concentration of DBPs in drinking water sources.

DBPs / Disinfectant	THMs	HAAs	Nitrosamines	HANs
Ozone	<40 µg/L[1]	28.88 µg/L[3] <35 µg/L[1]	<10-143 ng/L[6,7]	0.08-0.11 µmol/mmolC [14]
Chlorine	39-112 µg/L[2] <50-419 µg/L[12]	80-244 µg/L[2] <20-135 µg/L[12]	3.3-40 ng/L[4,5]	<20 µg/L[9,11]
Chloramine	<50-200 µg/ L[13] 0-<5 µg/ L[13]	<20-50 µg/L[12,13]	18-65 ng/L[4,5]	3-12 µg/L[7]
Chlorine dioxide	<1.5 µg/L[10]	<6 µg/L[10]	<10-70 ng/L[8]	NA

NA – Not Available

[1]Wert and Rosario-Ortiz, (2011); [2]Rodriguez et al., (2002); [3]Dojilido et al., (1999); [4]Nawrocki and Andrzejewski, (2011); [5]Charrois et al., (2007); [6]Zhao et al., (2008); [7]Bond et al., (2011); [8]Huang et al., (2016); [9]Kozari et al., (2020); [10]Yang et al., (2013a); [11]Yang et al., (2013b); [12]Goslan et al., (2009); [13]Cynthia et al., (2010); [14]De Vera et al., (2015).

2.10 TTHM

The regulated trihalomethanes, namely trichloromethane, tribromomethane, dichlorobromomethane and chlorodibromomethane (Wang et al., 2007), are generally referred to as TTHM. They are primarily formed as a result of chlorination of waters containing relevant

DBP precursors. However, chloramination also results in TTHM formation although at a relatively lower concentration compared to free chlorination.

2.10.1 Mechanism of TTHM Formation

TTHM are formed during the disinfection of drinking water using chlorine followed by sequential reaction of the disinfectant with natural organic matter. Rook (1967) showed that the meta-dihydroxylated aromatic rings in the structure of natural organic matter are the most reactive sites involved in the formation of halomethanes. The formation of trihalomethanes is time sensitive and the concentration can increase even after a several hours of halogen contact time (Trussell, 1978). As a result of this, consumers at the extreme end of the distribution system are more susceptible to higher concentrations of TTHM (Rodriguez and Sérodes, 2001; Williams et al., 1998). Further, the presence of iodide and bromide in the aqueous matrix could result in the formation of iodinated and brominated trihalomethanes (Liu et al., 2017; Nokes et al., 1999). As mentioned earlier, while free chlorine is not the only disinfectant that forms TTHM, other chlorine-based disinfectants such as monochloramine also contributes to a relatively lower concentration of TTHM (Bougeard et al., 2010; Hong et al., 2013).

2.10.2 Toxicity of TTHM

The consumption of water containing TTHM at concentrations above regulatory limits could be detrimental to health due to its proven carcinogenic effects. Studies have shown that TTHM are hepatotoxic in nature resulting in liver and renal tumors (Lilly et al., 1997; Melnick et al., 1998; Simmons and Pegram, 1998). Mutagenic effects such as low birth weight and stillbirth cases were associated with patients who were exposed to higher doses of TTHM (Toledano et al.,

2005; Wright et al., 2004). Literature shows that amongst the halogenated DBPs, the brominated and iodinated compounds are substantially more cytotoxic and genotoxic than the chlorinated counterparts (Echigo et al., 2004; Parvez et al., 2011; Richardson et al., 2008; Sharma et al., 2014). Cytotoxicity studies using mammalian cells showed that exposure to bromodichloromethane led to cell necrosis for exposure times larger than 24 hours (Lilly et al., 1997; Pagé-Larivière et al., 2016). A genotoxicity study stated that bromodichloromethane and tribromomethane exposure on mammalian cells instigated DNA damage (Geter et al., 2004).

2.10.3 Detection of TTHM

The EPA method 501.3 is used for the detection and analysis of the four regulated trihalomethanes. The method uses a gas chromatograph with mass spectrometry GC/MS in selected ion monitoring setting and has method detection limits of 0.06, 0.07, 0.05 and 0.04 µg/L for trichloromethane, bromodichloromethane, chlorodibromomethane and bromoform respectively. High purity helium or nitrogen is bubbled through the sample matrix as a result of which TTHM are purged. The purged volatile compounds are then trapped, i.e., sorbed onto a porous polymer, most commonly 2,6-diphenyl-*p*-phenylene oxide. The trapped compounds are then heated and flushed with helium into the GC/MS system.

2.11 Haloacetonitriles

Haloacetonitriles (HANs), while not present in water matrices at concentrations comparable to the regulated DBPs, are however, relatively more toxic (Ma et al., 2014). The HANs analyzed in this study are chloro-, bromo-, dichloro-, dibromo-, bromochloro-, bromodichloro-, dibromochloro-, and trichloro- acetonitrile (CAN, BAN, DCAN, DBAN, BDCAN, DBCAN, and

TCAN respectively). They are usually formed during chloramination or chlorination of waters containing NOM or effluent organic matter (Yang et al., 2012).

2.11.1 Mechanism and Occurrence of HANs

In order to limit and control the formation of regulated disinfection byproducts such as TTHM and HAA5, water providers tended to shift towards using alternate disinfection techniques like chloramination and ozonation (Bond et al., 2011). However, this shift in the disinfection process ultimately led to the formation of nitrogenous DBPs that are relatively more toxic than the regulated ones (Lin et al., 1986; Lipscomb et al., 2008; Yang et al., 2010). HANs are one such nitrogenous DBP group.

The two most prominently proposed pathways to the formation of HANs during chlorination or chloramination are the decarboxylation and aldehyde pathway. In the decarboxylation pathway, organic nitrogen compounds are responsible for the nitrogen in HANs, whereas, in the aldehyde pathway, chloramines as disinfectants contribute to the nitrogen in HANs (Huang et al., 2012). It was demonstrated in a study that dichloroacetonitrile (DCAN) formed during the reaction of Suwannee River NOM and monochloramine had more that 70% of the nitrogen derived from monochloramine (Huang et al., 2012; Yang et al., 2010). This confirms that DCAN formation during chloramination follows the aldehyde pathway. In this pathway, monochloramine reacts with N-organic matter to form aldehydes which further react with monochloramine to give rise to nitriles. The formation of nitriles from the reaction between aldehydes and monochloramine is confirmed by Pedersen et al. (1999). It has been shown that ozonation increases the chances for the formation of aldehydes (Richardson et al., 1999; Weinberg et al., 1993). When ozonation is subsequently followed by chloramination, the reaction between the aldehydes and chloramine

could potentially form HANs, and the type of HANs formed depends on the N-organic precursor and the corresponding aldehydes (Shah and Mitch, 2012).

2.11.2 Toxicity of HANs

Several toxicology studies reveal that HANs are highly cytotoxic, genotoxic and carcinogenic in nature (Lin et al., 1986; Lipscomb et al., 2008; Muellner et al., 2007; Yang et al., 2013). The cytotoxicity and genotoxicity of HANs are in the order of BAN > IAN≃DBAN > CAN > TCAN > DCAN and IAN > BAN ≈ DBAN > BCAN > CAN > TCAN > DCAN respectively (Lu et al., 2018; Muellner et al., 2007). In an experimental study, upon oral administration of HANs to rats, thiocyanate was found in the urine of the test subjects which inhibited the certain hepatic enzyme activity (Lin et al., 1986). Hanigan et al. (2017) demonstrated that brominated and iodinated HANs showed severe developmental effects in zebrafish. They also stated that dihalogenated- and trihalogenated-HANs are potentially more toxic than monohalogenated-HAN equivalents.

2.11.3 Detection of HANs

Sodium sulfate assisted liquid-liquid microextraction followed by programmed temperature-GC/MS can be used for the detection/analysis of HANs in a water sample (Ma et al., 2014). The addition of sodium sulfate increases the ionic strength and decreases the solubility of the extraction solvent in water (Herrera-Herrera et al., 2010). This provides enhanced phase separation between the aqueous and organic phases. In this method, the achieved range of limits of detection and recoveries were 0.4-13.2 ng/L and 79.3-103.5% respectively with relative standard deviation less than 11.9% (Ma et al., 2014). Carter et al. (2019) demonstrated a similar

process for the simultaneous analysis of 25 nitrogenous DBPs that includes 9 HANs with detection limits ranging from 0.8-1.7 μg/L.

2.12 Nitrosamines

In order to limit the formation of regulated DBPs that are generally formed as a result of chlorination, water providers have switched to alternate disinfectants such as ozone and chloramines. *N*-ntrosodimethylamine is one of the molecules that has become more prevalent in water systems due to the switch to alternate disinfectants. These compounds have a general structure of the formula R_2N-N=O where R is a representation of an alkyl group or ring. These DBPs are currently unregulated; however, their presence is of huge concern due to their carcinogenic nature (Muellner et al., 2007). These compounds are in the USEPA priority list of contaminants, where a cancer risk level of 0.7 ng/L is established (Muellner et al., 2007).

2.12.1 Mechanism and Occurrence of Nitrosamines

N-Nitrosodimethylamine (NDMA) formation in water is usually caused by the ozonation or chloramination of water matrices containing organic precursors with a quaternary amine functional group (Fiddler et al., 1977; Nawrocki and Andrzejewski, 2011). Literature shows that formation of NDMA via dimethylamine is possible through two major pathways, namely oxidation of unsymmetrical dimethylhydrazine (UDMH) and nitrosation. In the former pathway, chloramine as a disinfectant reacts with dimethylamine to form UDMH which upon subsequent oxidation forms NDMA (Mitch et al., 2003). It has been demonstrated that the oxidation of UDMH could lead to the formation of different molecules at varying pH, and the formation of NDMA via the oxidation of UDMH is favored at a neutral pH (Mitch and Sedlak, 2001). They

also showed that formation of NDMA is slow via this pathway; hence it is possible that the concentration of NDMA can increase over time in the presence of residual oxidants such as chloramine in the water distribution system (Mitch and Sedlak, 2001). In the latter mentioned pathway, i.e., nitrosation, nitrite reacts with dimethylamine with nitrosyl cation as an intermediate to form NDMA at low pH values (Mitch et al., 2003). Investigations have shown that ion exchange resins have also contributed to the formation of NDMA (Fiddler et al., 1977; Kimoto et al., 1980). This is because chlorine-/chloramine-containing water reacts with the quaternary amine functional groups in certain classes of resins, possibly following the UMDH pathway to form NDMA. The presence of nitrite further increased the concentrations of NDMA (Najm and Trussell, 2001).

Literature has shown that NDMA formation is also caused by ozonation. Marti et al. (2015) demonstrated in their experiments that there are distinct groups of NDMA precursors for chloramines and ozone. The molar conversion yield of NDMA is higher when the precursors have a dimethylamine functional group directly attached to a nitrogen atom or separated from a nitrogen atom by a leaving group (Schmidt and Brauch, 2008). Ozonation of model compounds namely 2-F-dimethylhydrazine and daminozide resulted in NDMA molar yields of 61-78% in deionized water (Marti et al., 2015). The model compounds contain the building blocks of NDMA, i.e., dimethylamine and one other nitrogen is located at the outer portion in the structure of the precursor. Also, the presence of a double bond in the structure of 2-F-dimethylhydrazine makes it more reactive towards ozone. Studies reveal that there is not much significance to hydroxyl radicals in the formation of NDMA (Oya et al., 2008; Gunten et al., 2010). In fact, advanced oxidation using ozone/H_2O_2 had little to no effect on the formation of NDMA (Marti et al., 2015).

2.12.2 Toxicity of Nitrosamines

The USEPA regards NDMA to be a probable B2 carcinogen (US Environmental Protection Agency, 2012). As a way of indicating carcinogenic risk of NDMA, the EPA has established an oral slope factor of 51 mg/kg/day (EPA IRIS 1993). NDMA intoxication in male Wistar rats showed signs of hepatic and renal DNA fragmentation (Adeleke and Adaramoye, 2017). Preneoplastic lesions were observed in the livers of White Leghorn chickens after the inoculation of NDMA and *N*-nitrosodiethylamine in the eggs (Kril et al., 2018). Despite the carcinogenicity of NDMA, it is not regulated; however, there is potential to regulate NDMA and other nitrosamines in the U.S. through the Safe Drinking Water Act. Several nitrosamines appear on the USEPA's past contaminant candidate lists (CCL), namely CCL3 and CCL4, as well as the draft CCL5.

2.12.3 Detection of NDMA

NDMA, usually found in water matrices at concentrations in ng/L, requires sensitive methods for analysis. NDMA analyzed by liquid chromatography tandem mass spectrometry (LC-MS/MS) produced recoveries over 90% with a method detection limit of 2 ng/L. Due to the low environmental concentrations, a solid phase extraction procedure is done prior to analysis using LC-MS/MS. For solid phase extraction, samples passed through 1 g activated charcoal at a rate of 5 mL/min provided 98% NDMA recovery (Plumlee et al., 2008). In this research, GC-MS method is used for analysis of nitrosamines (which includes NDMA). However, prior to analysis, the samples were extracted by liquid-liquid extraction using a modified USEPA Method 521 (Holady et al., 2012; Zeng and Mitch, 2015) due to the formation of nitrosamines in trace concentrations.

2.13 DBP Precursors

DBP precursors are those compounds, which upon reaction with disinfectants such as chlorine, monochloramine, ozone, etc., form disinfection byproducts. A variety of organic substances may act as precursors, including decaying plant matter, pharmaceutical compounds, personal care products, industrial waste, and breakdown or biodegradation products.

For instance, THM formation from pure bacterial cells such as *E.coli* during chlorination at 3 mg/L of chlorine is as high as 46.7 µg/L (Wang et al., 2013). Chloramination at 140 mg/L of Cl_2 of Sheba hospital wastewater resulted in NDMA formation potential of up to 7300 ng/L (Sack et al., 2021). Chu et al. (2017), demonstrated that microcystin-LR at a concentration of 0.5 µM upon direct reaction with 25 µM of chlorine yielded trichloromethane, chloral hydrate and DCAN at yields just over 40, 2 and 1%. Even treatment technologies have shown to promote DBP formation, however, the total cytotoxicity of the treated water is reduced (Cuthbertson et al., 2019) As such, there are numerous studies that indicate various organic compounds as precursors to DBPs.

2.14 Factors Influencing the Formation of DBPs

Several factors affect the formation as well as the speciation of DBPs. These factors include pH, temperature, residence time, metal ions, bromide ion and iodide ion.

2.14.1 Effect of pH

The formation of DBPs either increases or decreases by varying pH based on the class of DBPs. Chlorine formation potentials of THMs and HANs increased with increase in pH from 5.5 to 8.5.

In contrast, chloramine formation potentials of THMs and HANs did not show significant variations with increasing pH (Doederer et al., 2014). At higher pH, the hydrolysis of 1,1,1-trichloropropanone results in the formation of higher concentrations of THMs especially during chlorination (Doederer et al., 2014). Conversely, at lower pH, higher concentrations of THMs are observed during chloramination due to the apparent monochloramine hydrolysis to free chlorine (Zhang et al, 2010). It has been shown that the concentration of THMs increases with increase in pH whereas there is no significant change in the concentrations of HAAs with respect to pH (Ye et al., 2009). HANs increased with increase in pH from 5 to 6 and started to decrease at pH above 6 (Jia et al., 2016). At pH 7, the concentration of dimethylamine, a prominent precursor of NDMA was present at 74% and slightly lower at 67% at pH 8 and 9 (West et al., 2016). It was observed that the change in the pH did not significantly affect the concentration of NDMA formed in the presence of monochloramine and dimethylamine (West et al., 2016). It is therefore important to maintain a pH that does not promote DBP formation during the process of disinfection.

2.14.2 Effect of Temperature

The effect of temperature on the formation of DBPs depends on the volatile properties of the same. For example, Ye et al. (2009) stated that THMs continue to form only up to a certain critical temperature. As critical temperature is the temperature at which maximum formation (THMs in this case) can be obtained, an increase in the temperature above this critical temperature will cause a reduction in the levels of THMs through volatilization. This critical temperature for THMs formation is 18.97 °C (Garcia-Villanova et al., 1997). In contrast, for HAAs, heating or an increase in temperature thermodynamically favored the formation of HAAs.

This is attributed to the fact that most HAAs are not volatile and any reduction in HAA concentrations is most likely due to the thermal destruction of the compound or its precursors (Wu et al., 2001). In the case of trichloroacetonitrile and dichloroacetonitrile, the rate of thermally induced decomposition was higher than the formation (Chen et al., 2017; Jia et al., 2016). Studies have stated that an increase in the temperature of the water favors the formation of NDMA in ozonation reactions (Krasner et al., 2013; Sgroi et al., 2018). It is thus necessary to maintain the temperature of water at a value that does not promote DBP formation, and this temperature will depend on the precursors present.

2.14.3 Effect of Residence Time

While it is clear that DBPs continue to form as long as there are abundant precursors and residual disinfectants present in the water matrix, it has been reported that in an environmental matrix, residence time does not need to be proportional to the formation of DBPs such as THMs and HAAs due to the volatility and biodegradability of the compounds respectively (Ye et al., 2009). This statement is supported by evidence that THMs and HAAs at the farthest end of the distribution systems are present at low concentrations relative to the concentrations in the water treatment plant (Baribeau et al., 2005; Speight and Singer, 2005). Maximum DCAN concentrations were detected within 8 hours of chlorination and began to decrease after that. TCAN on the other hand reached a maximum concentration only after a chlorine contact time of 24 hours before it started to decrease. This decrease is attributed to the hydrolysis of the compounds to acetamides (Jia et al., 2016). During chloramination in the presence of amine precursors, NDMA was detected within 4 hours of exposure and reached a maximum

concentration of 1042 ng/L after seven days (West et al., 2016). It is thus clear that residence time i.e., water age plays a key role in the formation of DBPs along the distribution systems.

2.14.4 Effect of Metal Ions

The presence of metal ions such as Cu^{2+} can have a catalytic effect on the formation of DBPs during chlorination (Blatchley et al., 2003; Liu et al., 2013; Sharma et al., 2017). Sharma et al. (2017) states that these metal ions form complexes with the NOM which then react with chlorine to form DBPs, and the type of DBPs depend on the nature of the parent compounds. The enhancement of THM formation by certain metal ions is in the order of Cu^{2+}>Mg^{2+}>Ca^{2+} (Navalon et al., 2009). For the case of Cu^{2+}, it promotes the process of oxidative decarboxylation, thereby catalyzing the formation of HAAs (Yang et al., 2008). It has been experimentally proven that the presence of Al^{3+} in a water matrix could significantly increase the formation of DCAN by more than twice the concentration of the same in the absence of Al^{3+} (Zhang et al., 2019).

2.14.5 Effect of Bromide Ion

Brominated DBPs are generally more cytotoxic and genotoxic in nature; hence, it is important to understand the effects of bromide ions on the formation and speciation of DBPs. It has been seen that, in the case of THMs and HAAs, with increasing bromide concentration, the speciation changes from chloro- to bromochloro- and finally bromo-based compounds of the same (Chang et al., 2001). The concentration of THMs increased from 288.5 to 534.8 μg/L as bromide ion concentration increased from 3 to 68 mg/L during chlorination (Zhang et al., 2015). The incorporation of bromine in HAAs was at a higher magnitude during the disinfection of waters

containing larger proportion of hydrophilic organic matter relative to hydrophobic organic matter (Ersan et al., 2019a); consequently, the concentration of bromide as well as the DBP precursors will influence DBP formation. The reactivity of bromide to form brominated HAAs is also influenced by pH. Lower pH contributed to higher brominated HAA fractions (Hong et al., 2013). Chang et al. (2001) also observed that with increasing chlorine:bromide ratio, tribromomethane and dibromoacetic acid concentrations decrease. Similar trends were observed in the case of HANs where the concentrations of BCAN and DBAN increased with respect to increase in the concentration of bromide. This is because of the superior substitution nature of bromide ions relative to the chlorine analogues (Bond et al., 2011; Jia et al., 2016). Given the higher toxicity of brominated DBPs compared to the chlorinated analogues, it is thus necessary to make sure the concentration of bromide in the water sources is reduced before the process of disinfection.

2.14.6 Effect of Iodide Ion

Similar to the effect that the bromide has on the formation and speciation of DBPs, it is important for us to understand the effects of iodide since iodinated DBPs are more toxic than their chlorinated analogues (Muellner et al., 2007). An increase in the concentration of iodide in the water matrix increases the formation of iodinated THMs, and the speciation favors iodinated compounds as the concentration of iodide continues to increase (Liu et al., 2018). Conversely, the speciation shifted from iodinated to chlorinated DBPs after the breakpoint of chlorination due to the competitive nature of chlorine (Liu et al., 2018). However, it has been shown that for water containing high iodide concentrations, the formation of iodinated THMs continued to increase up to a dissolved organic concentration (DOC) of 4 ppm and then began to decrease at

DOC concentration of 5.4 ppm (Ersan et al., 2019b). The effect of iodide on the formation of iodinated HAAs is similar to that of the formation of iodinated THMs during chlorine disinfection (Postigo et al., 2017). Studies have demonstrated that chloramination favors the formation of iodinated THMs and HAAs over chlorination. This is because, in the presence of chlorine, hypoiodous acid is immediately oxidized to iodate whereas, during disinfection using monochloramine, hypoiodous acid reacts with NOM to potentially give rise to iodinated DBPs (Bichsel and von Gunten, 2000; Postigo et al., 2017). The information on iodinated haloacetonitriles is seldom reported due to the limited availability of analytical methods to measure them (Zhao et al., 2019). However, a study by Postigo et al. (2017) demonstrated the formation of iodochloroacetonitrile during the chlorination of waters containing iopamidol.

2.15 Surfactants

Surface active agents commonly referred to as "surfactants" are primarily classified based on the charge of their hydrophilic moiety as anionic, cationic, zwitterionic and non-ionic (Bergé et al., 2018). These surfactants occur at various proportions in detergents, personal care products, cleaning solutions, etc. (Bergé et al., 2018; Menzies et al., 2017; Olkowska et al., 2014; Sütterlin et al., 2008). Due to the ubiquitous presence of these compounds in all households, they are eventually detected in the wastewater system (Bergé et al., 2018; Menzies et al., 2017; Rebello et al., 2014). As these surfactants get discharged into water bodies, they can cause severe distress to aquatic life. It has been reported that anionic surfactants (commonly found in detergents) have the potential to disrupt liver function, as well as alter the histopathology of fish gills (Mustapha and Allah, 2020). Fortunately, most of the anionic and non-ionic surfactants are biodegraded within the wastewater treatment plant (Menzies et al., 2017). However, this is not the case for

cationic surfactants. Cationic surfactants, being biocidal in nature, are not easily biodegraded in the wastewater treatment plant. As a result, they end up in the aquatic environment through secondary effluent and point discharge (Ishiguro et al., 2007; Zhu et al., 2018).

2.16 Quaternary Ammonium Compounds

Quaternary ammonium compounds (QAC) are those molecules that contain an alkyl and/or aromatic group coupled to a positively charged nitrogen atom (Ikehata and El-Din, 2004; Zhang et al., 2015a). These QACs are predominantly found in fabric softeners, disinfectant solutions, biocides, and phase transfer agents. In addition to household applications, these surfactants are prevalent in hospitals, resulting in significant concentrations in hospital wastewaters (Lasek et al., 2019). On account of their cationic nature and large molecular size, they partition to solids and are present in large proportions in the sludge from wastewater treatment plants, although some of these compounds do escape into the aquatic environment (Boethling, 1984).

2.16.1 Toxicity

While QACs are commonly employed in personal care products and household products, they are toxic above certain concentrations. In fact, they possess comparable toxicity to the currently banned triclosan (Sreevidya et al., 2018). Sreevidya et al. (2018) showed that the exposure to BZK at a concentration of 0.1 ppm caused reduction in the size of *C. elegans* brood and higher concentrations up to 1 ppm led to increased mortality rate. *D. magna*, when exposed to BZK for 48 hours, had a half maximal effective concentration (EC_{50}) of 38.2 µg/L (Lavorgna et al., 2016). In the same study, the authors also showed that *C. dubia* expressed a LC50 of 403.7 µg/L for a BZK exposure of 24 hours. In another study, natural assemblages and mono algae expressed

EC_{50} values of 36.4 and 120 µg/L respectively for a …BZK exposure of 24 hours (Pérez et al., 2009). This is probably due to QACs increased affinity towards negatively charged algal cell walls (Zhang et al., 2015a). Studies have also shown that BZK imparts genotoxic effects to eukaryotic cells at environmentally relevant concentrations (Ferk et al., 2007; C. Zhang et al., 2015a). BZK also may have a phytotoxic effect on plants. For example, an exposure of 0.25 ppm of BZK for a period of 12 days resulted in 68% reduction in the dry weight of lettuce (Khan et al., 2018).

2.16.2 Detection

There are several methods for the analysis of QACs in water samples. Ding and Liao (2000) were able to achieve detection limits of 0.1-0.001 µg/mL for alkylbenzyldimethylammonium chlorides using RP-18 cartridges for solid phase extraction followed by GC/MS. They were able to recover 81-92% of the compound with a relative standard deviation of 5-11%. Another method involved cationic solid phase extraction followed by analysis using LC-MS/MS for benzalkonium compounds (C12-C14) in environmental samples (Voorde et al., 2012). The authors were able to achieve limits of detection ranging from 0.03-0.02 ng/L using the method with recoveries around 90%. Similar to Ding and Liao (2000), another study also uses RP-18 solid phase extraction; however, it was followed by HPLC (Kümmerer et al., 1997). They were able to attain a detection limit of 0.05 mg/L in hospital effluent samples with a precision of 1.6% and a relative standard deviation of 87-95%.

The above-mentioned analytical methods are more commonly employed to detect trace amounts at a higher sensitivity. In that regard, a more generic method developed by Patel et al. (2014)

helps detect cationic surfactants at concentrations in the range of 1-5 mg/L using UV-visible spectrophotometry at a wavelength of 517 nm.

2.16.3 Fate and Occurrence of Cationic Surfactants in Wastewater

Due to the ubiquitous presence of cationic surfactants in various household and hospital products, they are often found in the influent of many wastewater treatment plants (WWTPs) (Clara et al., 2007; Martínez-Carballo et al., 2007). The concentration of BZK was as high as 2800 ppb in the effluent of hospital wastewaters (Kreuzinger et al., 2007). The effluent from European hospitals had BZK at concentrations up to 6.03 mg/L. However, BZK concentrations in WWTP effluents are usually very low due to its adsorption affinity for particulate and organic matter such as sludge and also partly due to biotransformation of the parent compound (Zhang et al., 2015a). However, this is always not the case. Since WWTPs are not designed to remove these compounds, they eventually end up in the downstream surface waters. One surface water in Taiwan had a BZK concentration up to 65 µg/L (Ding and Liao, 2001). The concentration of BZK up to 36.6 µg/L were detected in the surface waters that are located downstream of several WWTPs in the United States (Ferrer and Furlong, 2001).

Clara et al. (2007) monitored the concentration of quaternary ammonium compounds in the influents and effluents of several WWTPs in the western region of Austria. They observed that the maximum influent and effluent concentrations of alkyl benzyl, dialkyl and trialkyl ammonium chlorides were 170 & 0.63 µg/L, 200 & 3.5 µg/L, and 27 & 1.1 µg/L, respectively.

More recently, the SARS-CoV-2 pandemic has led to an increased use of disinfectants (Hora et al., 2020). The USEPA's list of disinfectants for the inactivation of the SARS-CoV-2 virus has

216 products that contain QACs as the active ingredient (Hora et al., 2020). This suggests a potential rise in the concentration of QACs in the influent and effluents of WWTPs. The scarcity of environmental data on a variety of QACs warrants further studies.

2.16.4 QACs as Precursors for DBPs

Due to the ubiquitous nature of QACs in surface waters as we have seen in the above section, it is important for scientists and engineers to see if these compounds give rise to other toxic compounds relative to the fate of these compounds in a drinking water treatment process. In one study, ozonation of BZK resulted in the formation of benzyldimethylamine and dodecyltrimethylamine as notable intermediates in the degradation pathway (Carbajo et al., 2016). From the sections on NDMA in the literature review, we know that these intermediate compounds are well known NDMA precursors. The chloramination of 0.1 mM of BZK contributed to the formation of THMs, HAAs and NDMA at concentrations as high as 307.5 µg/L, 58.2 µg/L and 314.9 ng/L, respectively (Chang et al., 2011). In a recent study, several intermediate compounds such as chlorinated BZK, oxy- and hydroxy- BZK were identified during the chlorination of BZK, and these intermediate compounds could potentially be regarded as novel disinfection byproducts (Ul'yanovskii et al., 2020).

Chapter 3 – Materials and Experimental Methodology

3.1 Materials

A variety of chemicals were purchased to conduct this study. The list of chemicals, purity and vendor details are given in Table 3-1.

Table 3-1 List of chemicals used, their purity and the manufacturer

Chemical	**Purity/grade**	**Vendor**
TMAC	>98%	TCI
TEAC	98%	Alfa Aesar
BEC	97%	TCI
BZK	95%	Beantown chemical
CPC	98%	Alfa Aesar
DTMA	97%	Alfa Aesar
CTMA	96%	Alfa Aesar
Sodium arsenite	NA	VWR
Sodium sulfate	99%	Beantown chemical
Sodium hypochlorite	8-10%	Alfa Aesar
Sodium thiosulfate	98.3%	VWR
Sodium hydroxide	NA	VWR
Ammonium chloride	>99.5%	VWR

Dichloromethane	>99.9%	Honeywell
MTBE	HPLC/GC grade	Supelco
Methanol P&T	P&T grade	Honeywell
THM mix	>98%	Accustandard
HAN mix	>98%	Accustandard
Nitrosamine mix	>98%	Accustandard

3.1.1. Water

Deionized water was boiled and brought to room temperature (23±1 °C) prior to use. The resulting water was then buffered at neutral pH (7) with 1 mM phosphate buffer before use in all experiments.

3.1.2. Preparation of Chlorine Stock Solution

The stock solution (400-500 mg/L as Cl_2) of free chlorine was prepared by diluting 6-12% sodium hypochlorite solution in deionized water. The final concentration of the stock solution was measured using Hach method 8021 as mentioned in section 3.3.1.

3.1.3. Preparation of Chloramine Stock Solution

Fifty µL of 2 N sodium hydroxide was added per 100 mL of cold DI water in order to increase the pH during the preparation of chloramine stock solution. A calculated stoichiometric mass of ammonium chloride was weighed out and added to the cold DI water and stirred completely until

dissolution. Sodium hypochlorite was slowly added to the solution to achieve a final total chlorine concentration in the solution of 6.5-8 g/L as Cl_2. The resulting stock solution was then spiked into the sample matrix at a specific volume to achieve a final monochloramine concentration of 140 mg/L as Cl_2 and a Cl_2 to N ratio of 3.5:1 by mass in the sample.

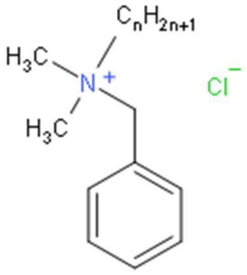
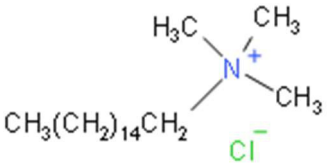
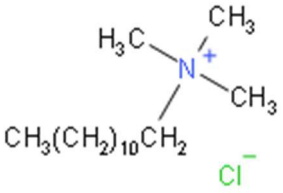
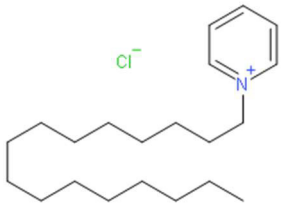
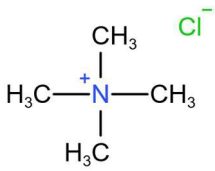
3.1.4. Preparation of Ozone Solution

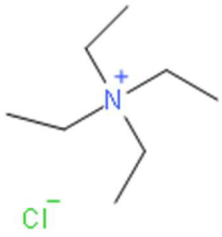
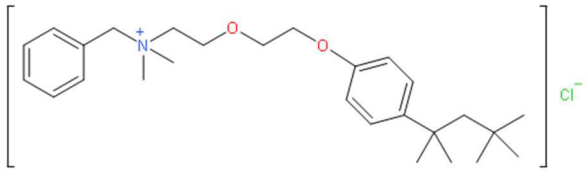
To generate ozone (O_3), a primozone GM-1 ozone generator was used. A jacketed flask connected to the ozone generator was filled with DI water and was continuously stirred using a stir plate. The jacketed flask was also connected to a chiller unit to maintain a temperature of 6 °C. Ozone from the ozone generator was allowed to mix with the cold water in the jacketed flask to produce ozonated water, which was then used for formation potential reactions. The outlet of the jacketed flask was connected to gas washing bottles containing potassium iodide (KI) as the ozone quenching agent. The KI solution was replaced with a fresh batch as it became exhausted.

3.1.5. Selection of Precursors

Seven different quaternary ammonium compounds namely, BZK (C8-C18), cetyltrimethylammonium chloride (CTMA), dodecyltrimethylammonium chloride (DTMA), cetylpyridinium chloride (CPC), tetramethylammonium chloride (TMAC), tetraethylammonium chloride (TEAC) and benzethonium chloride (BEC) were chosen based on several structural and functional characteristics. These characteristics include the presence and absence of aromatic ring, chain length, positioning of the functional groups and occurrence in commercial products. The structures of these precursors are defined in the Table (3-2) below.

Table 3-2 Structures, names, acronyms and molecular weights of quaternary ammonium compounds included as disinfection byproduct precursors in this study.

Quaternary ammonium compounds (QACs)	**Molecular Weight (g/mol)**	**Structures**
Benzalkonium chloride (BZK)	269.9-382.2	
Cetyltrimethylammonium chloride (CTMA)	320.00	
Dodecyltrimethylammonium chloride (DTMA)	263.89	
Cetylpyridinium chloride (CPC)	339.99	
Tetramethylammonium chloride (TMAC)	109.6	

Tetraethylammonium chloride (TEAC)	165.7	
Benzethonium chloride (BEC)	448.1	

3.2. Sample Preparation

The disinfection byproduct formation potential tests were conducted in 250 mL amber glass bottles in order to prevent photodegradation of the DBPs. Based on stoichiometric calculations, the precursors were spiked individually into the reaction bottles to achieve a final molar concentration of 0.1 mM. The reaction bottles were then filled with phosphate buffered DI water as stated in section 3.1.1. In order to assess the formation of brominated DBPs, bromide was spiked into the reaction bottles for some experiments. Bromide was spiked to achieve two different concentrations, i.e., 50 µg/L and 500 µg/L which are representative of low and high concentrations in the environment (VanBriesen, 2014).

3.3. DBP Formation Potential Experiments

In order to determine the yield of DBPs in natural/artificially spiked waters upon reaction with specific disinfectants, disinfection byproduct formation potential (DBPFP) tests are conducted.

In this study, the QAC precursors were individually spiked in reaction bottles as described in section 3.2 and are subjected to reaction with ozone, chloramine and chlorine separately. In order to maintain a steady pH throughout the FP tests, phosphate buffer was added to the samples prior to adding the disinfectant. The DBPFP experiments were completed in headspace-free bottles at room temperature (23±1 °C). A brief summary of the experimental matrix can be seen in Tables 3-3 and 3-4.

3.3.1. Chlorination FP Tests

In order to determine the DBPFP of QACs as a result of chlorination, hypochlorite was spiked into the reaction bottles that were prepared as described in section 3.2. The hypochlorite spike concentration into the reaction bottles was dependent on a target residual chlorine concentration of 3-5 mg/L after a period of 7 days at room temperature in the dark per Standard Method 5710 (Baird et al., 2017).

To determine the hypochlorite spike concentration, a chlorine demand test was conducted. For this test, QACs of the smallest molecular weight and the largest molecular weight at a molar concentration of 0.1 mM were allowed to react with varying doses of hypochlorite in the range of 3-11 mg/L as Cl_2. The samples were then analyzed for free chlorine after a reaction period of 7 days.

Free chlorine in samples was measured with Hach Method 8021 (Free Chlorine by DPD method) using a DR900 colorimeter. Briefly, 10 mL of the sample was transferred to a clear vial and reacted with DPD free chlorine reagent. The solution was stirred well and allowed to react for a period of 1 min. The solution was then measured using the colorimeter. Samples outside of the calibration range (0.02-2.00 mg/L as Cl_2) were diluted and re-measured.

Once the samples were analyzed for residual free chlorine, the samples were quenched using sodium arsenite. Free chlorine quenching was done to halt the reaction of free chlorine with the precursors beyond the desired 7 days reaction time.

3.3.2. Chloramination FP Tests

Chloramination experiments were conducted in order to determine the disinfection byproduct formation potential of QACs during reaction with monochloramine. Preformed chloramine (as described in section 3.1.3) was spiked into reaction bottles (prepared as described in section 3.2) to achieve a final concentration of 140 mg/L as Cl_2 and allowed to react for a period of 10 days, as used in previous FP studies (Lee et al., 2007; Mitch and Sedlak, 2004).

The total residual chlorine in samples was measured with Hach Method 8167 (Total Chlorine by DPD method) using a Hach DR900 colorimeter. Briefly, 10 mL of the sample was transferred to a clear vial and reacted with DPD total chlorine reagent. The solution was stirred well and allowed to react for a period of 3 min. The solution was then measured using the colorimeter. Samples outside of the calibration range (0.02-2.00 mg/L as Cl_2) were diluted and re-measured.

Once the samples were analyzed for residual total chlorine, the samples were quenched by adding sodium thiosulfate. The volume of sodium thiosulfate used was determined by trial and error using the sample with the highest total chlorine concentration (typically the blank samples where there is little to no chloramine demand).

Table 3-3 Experimental matrix showing disinfection byproduct formation potential tests using chlorine, chloramine and ozone.

Batch	**Concentration of QACs**	**Disinfectant dosage**
Chlorine FP	0.1 mM	Determined using chorine demand test
Chloramine FP	0.1 mM	140 mg/L as Cl_2 (2 mM Cl_2)
Ozone FP	0.1 mM	0.77 mM O_3

Table 3-4 Experimental matrix for showing disinfection byproduct formation potential tests using chlorine, chloramine and ozone in the presence of bromide at low (50 µg/L) and high (500 µg/L) dosages.

Batch	**Concentration of QACs**	**Disinfectant dosage**	**Bromide dosage***
Chlorine FP	0.1 mM	Determined using chlorine demand test	50 or 500 µg/L
Chloramine FP	0.1 mM	140 mg/L as Cl_2	50 or 500 µg/L
Ozone FP	0.1 mM	0.77 mM O_3	50 or 500 µg/L

*Typical low (50) and high (500) bromide concentrations in water sources (VanBriesen, 2014).

3.3.3. Ozonation FP tests

Ozonation experiments were conducted in order to determine the disinfection byproduct formation potential of QACs during reaction with molecular ozone. For this purpose, ozonated water (concentration of 35-40 mg/L O_3) was allowed to react with the precursors in reaction bottles (prepared as described in section 3.2) with a 7-8-fold molar excess of ozone for a period of 24 hours in the dark. The aqueous concentration of ozone in the jacketed flask was measured using the indigo blue method as described by Bader and Hoigné (1981). DBP formation results from ozonation of QACs are not included in the following chapters as no DBPs (among the ones in our analysis) were detected as a consequence of ozonation.

3.4. DBP Extraction and analytical methods

The possible DBPs formed by the end of the DBPFP tests generally occur in trace concentrations, i.e., on the order of parts per billion (PPB) and part per trillion (PPT). This concentration is too small for the analytical instruments to measure. Also, water is not a compatible solvent for most gas chromatography (GC) columns. Thus, extractions were performed to 1) isolate the compounds of interest from the water sample, 2) shift the compounds of interest to a GC-compatible solvent, and 3) concentrate the compounds before analysis by evaporating most of the solvent. All the samples from the FP tests mentioned above underwent extraction as shown in Table 3-5. Extractions were performed for HANs and nitrosamines on the same day as reactions were quenched. THM extractions were performed within 14 days per the recommended sample storage protocol.

The following subsections describe the methods of DBP extraction, and the instrument settings used for DBP analysis. For all the three DBP groups included in this study, namely THMs, HANs, and NOAs, a gas chromatography tandem mass spectrometry (GC-MS/MS) system was used for analysis. The GC-MS/MS used was a ThermoFisher Scientific TRACE 1310 coupled to a ThermoFisher Scientific TSQ 8000 Evo triple quadrupole MS/MS. Argon was used as a collision gas and helium was expended as a carrier gas. TraceFinder software (v. 4.1) was used for data acquisition and analysis.

Table 3-5 Extraction and analytical methods used for specific DBPs

DBPs	**Extraction method**	**Analytical method**	**Reference**
THMs	Purge and trap	GC-MS	1
HANs	Liquid-liquid extraction	GC-MS	2
N-Nitrosamines	Liquid-liquid extraction	GC-MS	3

[1]Zoccolillo et al., (2005); [2]Carter et al., (2019); [3]Padhye et al., (2013).

3.4.1. THM analysis

THMs in the samples were analyzed using the GC-MS/MS system. However, prior to analysis, the samples were extracted for THMs by purge and trap method using a Lumin™ Teledyne Tekmar system equipped with a AQUATek 100 automatic sampler and a VOCARB™ 3000 adsorption column. Nitrogen (99.999% purity) was used as the purging gas. A Restek Rtx®-VMS column (20 m × 0.18 mm I.D. × 1.0 µm film thickness) was used in the GC for THM

analysis and important parameters are given in Table 3-6. Argon was used a collision gas and helium was expended as a carrier gas. Based on this method of THM analysis, the MRLs for TCM, BDCM, DBCM and TBM were 0.3, 0.2, 0.2 and 0.2 μg/L, respectively.

Table 3-6 GC-MS and purge & trap parameters for THM analysis

Parameter	**Value**
Max oven temperature (°C)	240
Inlet temperature (°C)	200
Oven run time (min)	14.99
Temperature 1 (°C)/rate (°C/min)/hold time (min)	38/0/3.5
Temperature 2 (°C)/rate (°C/min)/hold time (min)	100/12/0
Temperature 3 (°C)/rate (°C/min)/hold time (min)	225/25/1.32
Injector type and mode	SSL, split with constant flow
Split flow (mL/min)	10
Carrier flow (mL/min)	0.8
Purge flow (mL/min)	5
Split ratio	20

3.4.2. HAN Extraction and Analysis

HANs in the samples were analyzed using the GC-MS/MS system. However, prior to analysis, the samples were extracted by liquid-liquid extraction as described by the USEPA Method 551.1.

Briefly, 45 mL of the sample was taken into a 60 mL clear glass vial with Teflon lined caps. The samples were spiked with 15 µL of 10,000 µg/L 1,2-dibromopropane (internal standard). Twelve g of Na_2SO_4 (sodium sulfate) was added to it in order to facilitate increased partitioning of HANs from the water matrix to the organic solvent phase. Once the Na_2SO_4 in the sample was dissolved, 3 mL of methyl tert-butyl ether (MTBE) was added to the sample and mixed thoroughly for 60 seconds. The samples were then allowed to rest for 10 min. MTBE, which formed a separate layer in the 60 mL vial, was then gently pipetted out using a Pasteur pipette and passed through a Na_2SO_4 drying column to remove any residual water. The extract was stored at -20 °C prior to analysis by GC-MS/MS.

Analysis was similar to Cuthbertson et al. (2019) and the USEPA Method 551.1. For the GC-MS/MS system, a Restek Rxi®-5ms fused silica capillary column (30 m × 0.25 mm I.D. × 0.25 µm film thickness) was used for HAN analysis. Argon was used a collision gas and helium was expended as a carrier gas. The GC settings for HAN analysis are described in the Table 3-7. Based on this method, the MRLs for MCAN, DCAN, MBAN, BCAN, DBAN, BDCAN and TCAN were 0.047, 0.273, 0.227, 0.1, 0.133, 0.227, and 0.333 µg/L, respectively.

Table 3-7 GC-MS parameters for HAN analysis

Parameter	**Value**
Max oven temperature (°C)	260
Inlet temperature (°C)	170

Oven run time (min)	18.67
Initial temperature (°C)/rate (°C/min)/hold time (min)	35/0/4
Final temperature (°C)/rate (°C/min)/hold time (min)	180/15/5
Injector type and mode	PTV, splitless with surge
Split flow (mL/min)	40
Carrier flow (mL/min)	1
Surge pressure (kPa)	300
Surge duration (min)	0.50

3.4.3. Nitrosamine Extraction and Analysis

Eight NOAs, namely NDMA, NMEA, NDEA, NDBA, NDPA, NMOR, NPIP and NPYR, were analyzed using the GC-MS/MS system. However, prior to analysis, the samples were extracted by liquid-liquid extraction using a modified USEPA Method 521 (Holady et al., 2012; Zeng and Mitch, 2015). Briefly, 15 mL of the sample was taken into a 40 mL clear glass vial with Teflon lined caps. The sample was spiked with 25 µL of 2000 µg/L internal standard (mixture of 8 deuterated nitrosamines) and 0.5 g of NaCl was added to facilitate increased partitioning of NOAs from the water matrix to the organic solvent (methylene chloride) phase. Once the Na_2SO_4 in the sample was dissolved, 2 mL of methylene chloride was added to the sample and mixed thoroughly for 60 seconds. The samples were then allowed to rest for 10 min. The denser methylene chloride layer at the bottom of the 40 mL vial was then gently pipetted out using a Pasteur pipette and passed through a Na_2SO_4 drying column to remove any residual water. The extract was then collected and concentrated to 0.5 mL by nitrogen purging using a Biotage

TurboVap LV solvent evaporation system. The concentrated extract was stored at < -5 °C prior to analysis by GC-MS/MS system.

For the GC-MS/MS system, a Restek Rtx®-200 column (30 m × 0.25 mm I.D. × 0.25 µm film thickness) was used for NOA analysis. Argon was used as a collision gas and helium was expended as a carrier gas. The GC settings for NOA analysis are described in Table 3-8. Based on this method, the MRL for NOAs are 1.2, 2.1, 6.1, 3.4, 2.6, 2.1, 1.8 and 1.3 µg/L for NDMA, NMEA, NDEA, NDBA, NDPA, NMOR, NPIP and NPYR, respectively.

Table 3-8 GC-MS parameters for NOA analysis

Parameter	**Value**
Max oven temperature (°C)	240
Inlet temperature (°C)	200
Oven run time (min)	23.93
Initial temperature (°C)/rate (°C/min)/hold time (min)	35/0/1
Temperature 1 (°°C)/rate (°C/min)/hold time (min)	120/10/0
Temperature 2 (°C)/rate (°C/min)/hold time (min)	145/5/0
Temperature 3 (°C)/rate (°C/min)/hold time (min)	230/35/7
Injector type and mode	PTV, splitless with constant flow
Split flow (mL/min)	40
Carrier flow (mL/min)	1.2

3.5. Laboratory Quality Assurance and Quality Control (QA/QC)

3.5.1. Lab Duplicates

The experiments were repeated to obtain duplicate samples for each surfactant and for each condition. Duplicates were done in order to ensure reproducibility. The average of the duplicates is reported in the results and discussion section of this study.

3.5.2. Experimental Blanks

Experimental blanks (DI water) were also included in the study as a part of quality assurance to ensure that no DBPs were detected in the background. The experimental blanks were subjected to all the steps as the real samples, including reaction with disinfectant, quenching, extraction, and analysis. In addition to an experimental blank, a DI blank was also included for extraction and analysis only to make sure background DBPs are not detected through these steps of the method.

3.5.3. Continuous Calibration Checks

A 7-point calibration curve was established prior to analysis of samples. Continuous calibration checks (CCCs) were also set up at the low, mid, and high concentration range for each of the target DBPs. This is a part of the quality control that ensures that the GC-MS/MS is performing as expected. The minimum acceptable percentage variation of the CCCs from the theoretical concentration value was 20% for mid and high CCCs and 30% for low CCCs. Extracted samples that were above or below the calibration range were diluted and re-analyzed.

3.6. Sample Storage

Samples were analyzed the same day of DBP extraction when possible. If not scheduled to analyze the same day as extraction, the extracts were stored at -20, -9 and, 4 °C for HANs, NOAs and THMs, respectively, in airtight glass vials or bottles.

Chapter 4 – Chlorination Formation Potential: Results and Discussion

4.1 Introduction

The process of disinfection by chlorination is employed in most drinking water treatment plants before distribution. The presence of organic matter in the water matrix during disinfection contributes to the formation of DBPs. There are several groups of organic matter that could be characterized in these wastewater and water sources, however, in this study, a specific group of organic compounds, quaternary ammonium compounds (QACs) are assessed for their potential to form DBPs such as THMs, HANs, and NDMA upon chlorination.

The reason why we are interested in QACs is because these compounds are present in a plethora of household products such as laundry detergents, fabric softeners, surface disinfectants/cleaners etc., as well as personal care products such as hand sanitizers, cosmetics, mouthwash, conditioners, and soap. QACs are also widely employed for surface disinfection in the hospital industry, thus contributing to a significant QAC load in hospital wastewater. In recent times, QACs are gaining a lot of attention due to their ability to inactivate the SARS-CoV-2 virus. In fact, these QACs are present in over 216 products recommended by the CDC to inactivate the SARS-CoV-2 virus on surfaces. Given the current circumstances, i.e., the COVID-19 pandemic, this could mean that QAC load in wastewaters could potentially be higher than seen in the past.

While a bulk of the QACs are removed by the WWTP, a small portion of the QACs may enter the environment via secondary effluent and eventually end up in drinking water sources. There is

research being carried out on several harmful factors imparted by QACs to the environment, such as promoting antibiotic resistance in bacteria, toxicity, etc., however, studies on their potential to form DBPs are rather scarce. On account of this research gap, the current chapter focuses on the DBP formation potential of QACs due to chlorination.

Additionally, it is also important to take into account the presence of bromide in the sample matrix since brominated DBPs are more toxic than their chlorinated analogues. Higher concentrations of bromide will push the speciation of DBPs to more brominated forms. In that regard, separate experiments were conducted to assess the potential for QACs to form DBPs during chlorination in the presence of bromide.

4.2 Methods

4.2.1 Chlorine Demand of Quaternary Ammonium Compounds

The concentration of Cl_2 required to conduct the formation potential experiments was determined based on the results of the Cl_2 demand test and the procedural goal of achieving 3-5 mg/L as Cl_2 after 7 days based on the protocol of Standard Method 5710 (Baird et al., 2017) For this test, 0.1 mM of surfactant was allowed to react with varying concentrations of Cl_2 (3-11 mg/L) in buffered DI water at pH 7. It was observed that a spike of 5 mg/L of Cl_2 was sufficient to maintain the desired 3-5 mg/L residual Cl_2 in the sample matrix for all the QACs (0.1 mM) considered in this study.

4.2.2 Formation Potential Tests

Sodium hypochlorite was individually spiked into amber reaction bottles containing 0.1 mM QACs in buffered DI water (pH 7) to achieve a final free chlorine concentration of 5 mg/L in the

sample matrix. To assess the speciation of DBPs in the presence of bromide at low and high bromide concentrations, 50 and 500 μg/L of Br^- was spiked into the sample matrix respectively. The samples were reacted for 7 days at room temperature.

4.2.3 DBP Extraction and Analysis

At the end of the 7-day reaction period, residual free chlorine was measured in the sample matrix followed by quenching of chlorine using sodium arsenite in order to terminate the chlorination reaction. DBPs were then extracted following appropriate extraction protocols as described in section 3.4 of this thesis. The extracted samples were then analyzed for the corresponding DBPs using GC-MS. The methods for GC-MS analysis for specific DBPs are described in detail in section 3.4 of this thesis.

4.3 THM Formation from QACs

The results from the formation potential tests indicated that trichloromethane (TCM) was the dominant THM formed in the presence of chlorine and no bromide. The amount of TCM formed for the seven precursors under this study due to chlorination follows the order of TMAC<TEAC<BEC<CPC<DTMA<CTMA<BZK. The micromolar yield of THMs per mole QACs at a chlorine dose of 5 mg/L is visualized in Figure 4-1. The small amount of DCBM observed for BZK might be due to bromine impurities in the BZK and sodium hypochlorite stock solutions. It is evident from the literature that chlorination yields high amounts of THMs (Chang et al., 2011; Gallard and Gunten, 2002; Heller-Grossman et al., 1993; Hong et al., 2008; Sorlini and Collivignarelli, 2005). In regard to QACs, the formation of THMs during chlorination of 1 mM BZK was also observed by Chang et al. (2011), wherein the TTHM concentration was

2713±145 µg/L at a free chlorine dose of 70 mg/L in DI water at pH 7. The higher TTHM concentration in the Chang et al. (2011) study compared to this work (80.3 µg/L) can be attributed to the higher precursor and chlorine concentrations used. The yield of THMs due to QACs is low compared to prominent THM precursor such as resorcinol which has TCM molar yield of 0.95 (Gallard and Gunten, 2002). However, this is the first time that THM formation is reported for CPC, BEC, TMAC, CTMA and DTMA.

When a spectrophotometric method (Patel et al., 2014) was employed to determine the residual concentration of the QAC after reaction with chlorine, we found out that the method worked well only for select compounds such as CPC and BZK. Additionally, this method is not compound specific, i.e., as long as there is a quaternary amine molecule in the matrix, a signal can be obtained, irrespective to the degradation of the parent compounds. Hence, future studies with specific precursor analysis (e.g., mass spectrometry) are warranted to understand DBP formation pathways of the QACs through identification of the transformation products.

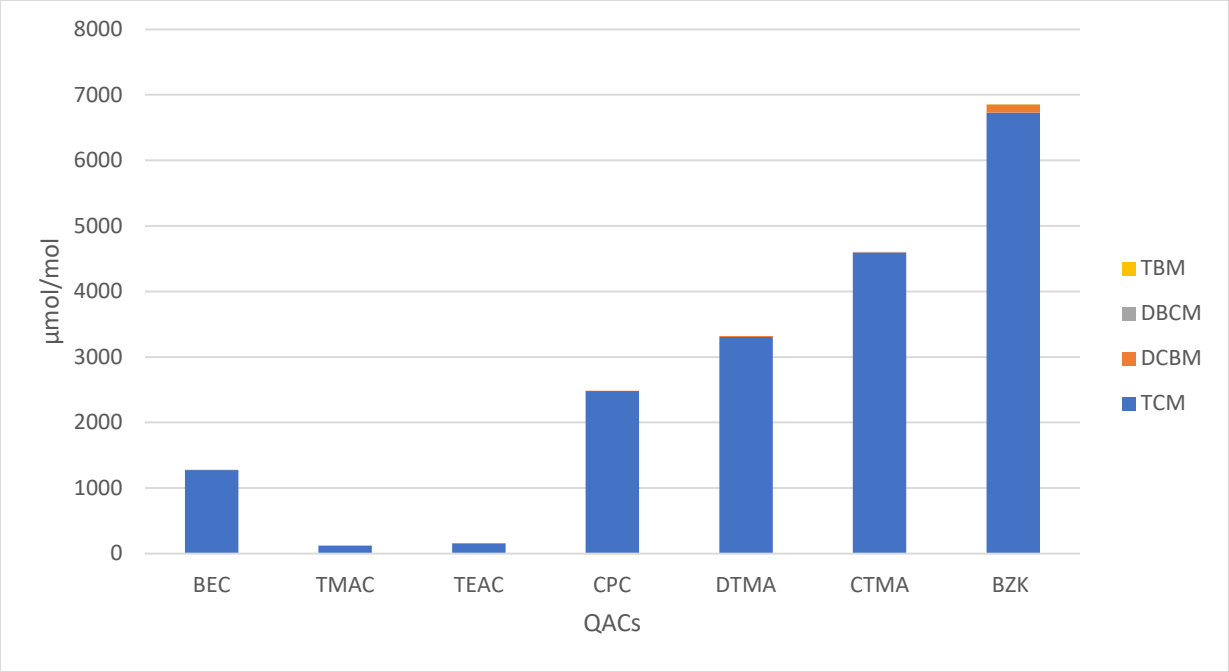


Figure 4-1 Micromolar yield of THMs (µmol THM / mol surfactant) upon reaction between 5 mg/L of chlorine and 0.1 mM QAC precursor in buffered deionized water at pH 7.

Potential THM Formation Mechanism for QACs

As seen from Table 3-2, the seven QACs included in this study are distinct in their chemical structure. From the results presented in Figure 4-1, BZK, with a benzyl moiety, formed more THMs compared to CTMA and DTMA, which do not contain a benzene ring. Additionally, the QAC with pyridine moiety formed even fewer THMs compared to the CTMA, DTMA and BZK. Although BEC contains two benzene rings, THM micromolar yield was significantly lower. In the case of compounds with short alkyl chains and no conjugated rings, i.e., TMAC and TEAC, the formation of THM was negligible.

The mechanism for QACs forming THMs is not yet known. We hypothesize the formation of BZK to THMs under chlorination based on collision-induced dissociation (CID) spectra in a study conducted by Ul'yanovskii et al. (2020). According to this study, the BZK precursor ion $[C_7H_8C_nH_{2n+1}ClN]^+$ undergoes possible rupture of the benzyl-nitrogen bond as suggested by Huang et al. (2017) and would result in the formation of a tropylium cation $[C_7H_7]^+$. This step is followed by the migration of a hydrogen from the aliphatic chain to the benzyl moiety, resulting in a subsequent loss of C_7H_8 and forming $[C_nH_{2n+1}ClN]^+$. This nitrogenous ion then loses HCl to give rise to the second product ion $[C_nH_{2n}N]^+$. Following a split of α-β C-C bonds, the long aliphatic chain leaves the second product ion, thus resulting in N, N-dimethylmethaniminium ion and a long chain alkyl molecule. A schematic representation of this pathway is presented in Figure 4-2. On conducting further analysis, no chlorinated derivatives of N, N-dimethylmethaniminium ion were found by Ul'yanovskii et al. (2020). This could mean that any chlorinated C-DBPs that were formed due to the chlorination of BZK are most likely attributed to HOCl attack on the long chain alkyl molecule which was separated from the second product ion $[C_nH_{2n}N]^+$. This possibility could also explain the THM micromolar yield from the longer chain alkyl ammonium precursors (DTMA, CTMA) as opposed to the short alkyl chain precursors (TEAC, TMAC).

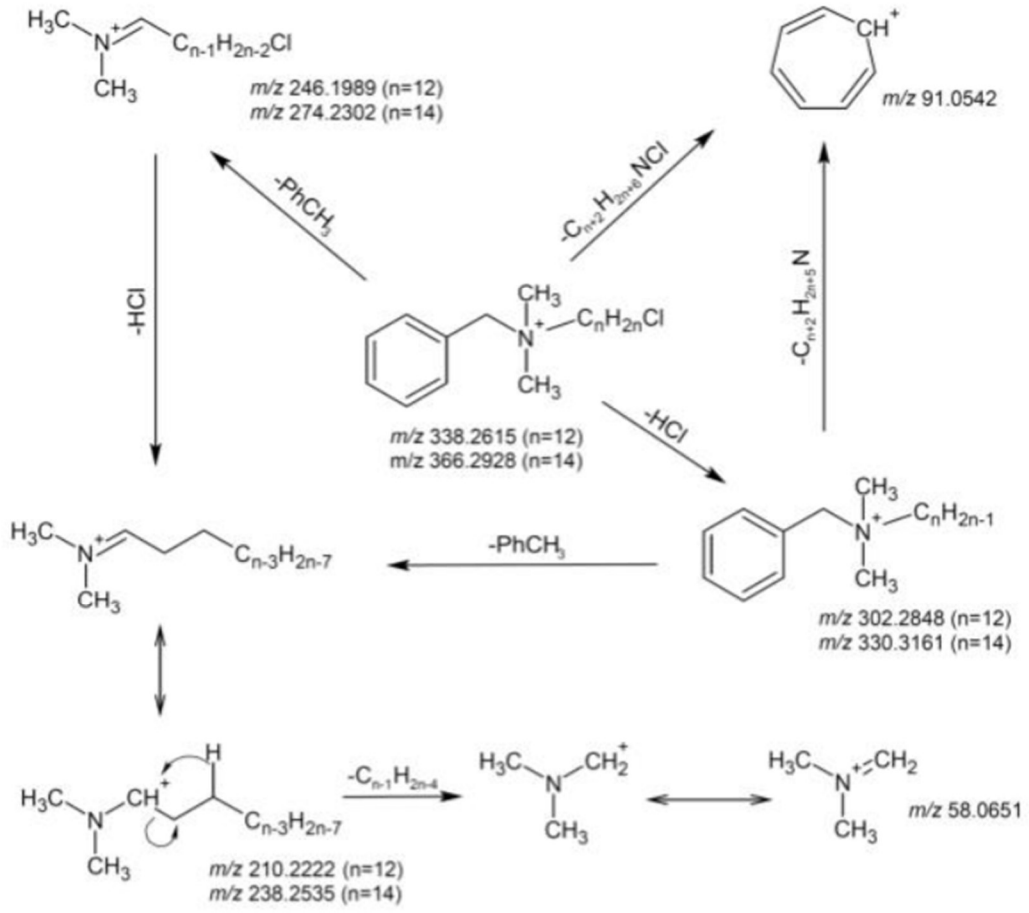


Figure 4-2 Full collision induced dissociation scheme of BZK during chlorination. Image reproduced from Ul'yanovskii et al. 2020.

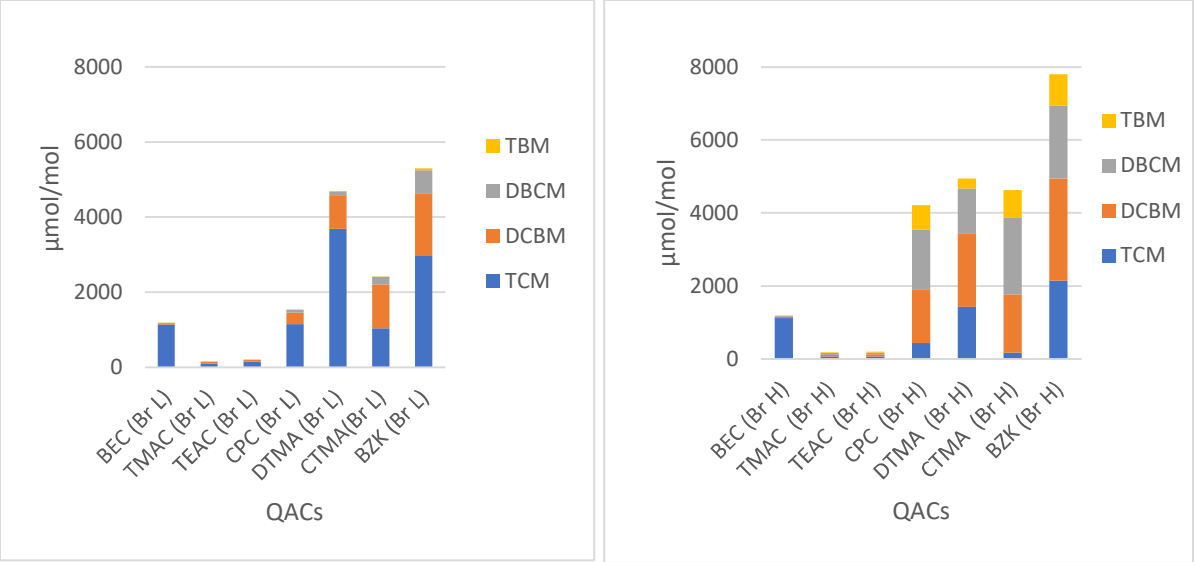
Second, quaternary amines may form TTHMs in a secondary process after demethylation. Based on a UV/chlorine oxidation study on benzalkonium chloride (Huang et al., 2017), it is possible that BZK underwent cleavage at the benzyl-nitrogen bond to form tertiary amines which further underwent demethylation to form secondary amines. The secondary and tertiary amines formed could continue to react with free chlorine to form THMs (Deborde and von Gunten, 2008; Roux

et al., 2012a; Wang et al., 2020). This formation of THMs is facilitated by the reaction of free chlorine and methyl groups from alkylamines (Roux et al., 2012b).

The short chain aliphatic amines, namely TEAC and TMAC, did not yield significant amount of THMs, as seen in Figure 4-1. Włodyka-Bergier and Bergier (2011) classified short chain aliphatic amines as hydrophilic base fractions when conducting experiments to fractionate humic acid. In that regard, it has been observed that hydrophilic base fractions were least reactive to chlorine in forming THMs (Marhaba and Van, 2000), thus giving as a possible explanation as to why TEAC and TMAC formed relatively fewer THMs.

4.4 Effect of Bromide on THM Speciation

Since brominated DBPs are relatively more toxic than their non-brominated analogues, DBPFP tests were conducted at 50 ppb and 500 ppb of bromide, thereby representing low and high levels of bromide in water sources (VanBriesen, 2014). The micromolar yield of the 4 THMs was calculated and is represented in Figure 4-2. Given the higher affinity of bromine towards substitution compared to chlorine, formation of brominated THMs is favored (Bond et al., 2011). As seen from Figure 4-2, the higher the concentration of bromide in the sample matrix, the higher the micromolar yield of brominated THMs and the lower the micromolar yield of TCM. This is because bromide upon reaction with hypochlorous acid (HOCl) forms hypobromous acid (HOBr) which has a substantially higher reactivity in terms of substitution, i.e., halogenating activity compared to HOCl (Zhou et al., 2021). This high reactivity of HOBr is due to the lower bond strength of HOBr compared to HOCl (Bond et al., 2014; Westerhoff et al., 2004).



(a) (b)

Figure 4-3 Micromolar yield of THMs (µmol THM / mol surfactant) upon reaction between 5 mg/L chlorine and 0.1 mM of QAC precursor: (a) in the presence of 50 µg/L Br- and (b) in the presence of 500 µg/L Br-.

4.5 Bromine Substitution Factor (BSF) for THMs

BSF for THMs is calculated by taking the ratio of the molar concentration of brominated THMs to the molar concentration of the total THMs formed (Zheng et al., 2020). The calculation of BSF for THM and is given by the following equation.

$$BSF_{THMs} = \frac{DCBM+(2\times DBCM)+(3\times CTBM)}{3\times (TCM+DCBM+DBCM+\quad)}$$

Based on Figure 4-4 showing the bromine substitution factor, when the concentration of Br^- increases from 0 to 50 µg/L Br^-, the increase in %BSF for the seven quaternary ammonium

compounds included in this study is in the order of BEC<DTMA<CPC<BZK<CTMA<TEAC<TMAC, and when the concentration of Br^- increases from 50 to 500 µg/L Br^-, the increase in the %BSF is in the order of BEC<DTMA<BZK<CTMA<CPC<TEAC<TMAC. Although the %BSF of TEAC is higher with increase in Br^- concentration, the yield of THMs is relatively low as seen in Figure 4-3. There are certain values in the %BSF equation that are below the MRL. As value below MRL cannot be used directly in the equation, these values are assumed to be zero prior to calculation. The assumption of zero is established based on information obtained from literature (Croghan and Egeghy, 2003), where the authors stated that in statistical analysis, values that are below reporting limits are substituted by constant values such as zero. While the best approach would be to use computational tools in order to reduce left-censored bias, a large data set would be required and in this study the data set is limited to only duplicates.

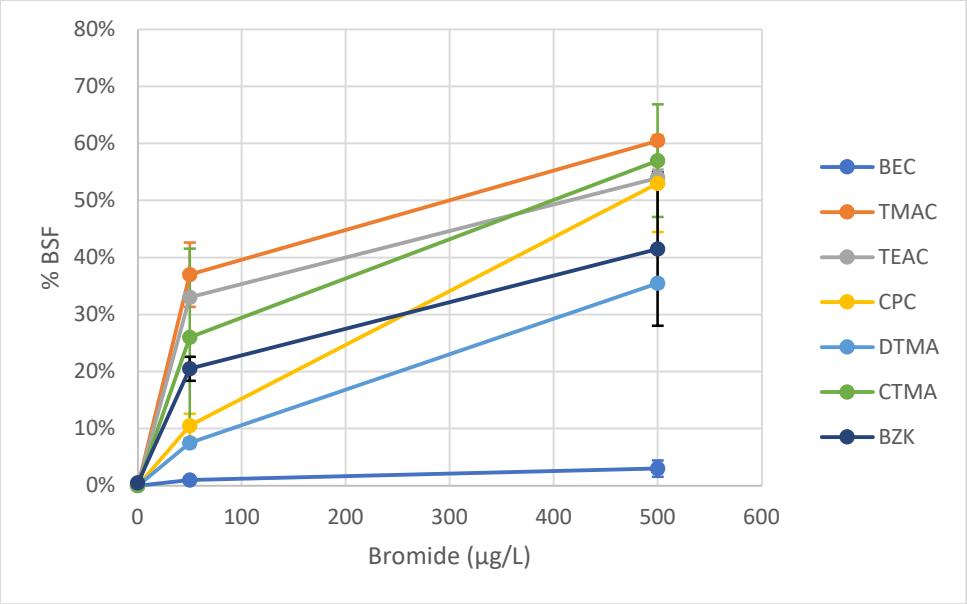


Figure 4-4 Percent bromine substitution factor for THMs upon reaction between 5 mg/L of chlorine and 0.1 mM QAC precursor.

It can be seen that the BSF for THMs formed as a consequence of reaction between BEC (0.1 mM) and 5 mg/L of free chlorine at bromide doses of 50 and 500 µg/L is rather insignificant. While there is a lack of information on the reactivity of BEC with halogens, it is possible that chlorine reacted with the aromatic moieties of BEC via electrophilic substitution to form a wide variety of intermediates. In the presence of bromide, it is possible that bromide ions would have participated in reactions with other intermediates leaving little to no bromide to form brominated trihalomethanes (Pan and Zhang, 2013). CPC, on the other hand, undergoes rapid cleavage of the heterocyclic ring upon reaction with chlorine (Zhang et al., 2017). This eventually contributes to THMs, and, in the presence of high concentrations of bromide, a sudden jump in the BSF can be observed.

4.6 HAN Formation from QACs

The results from the formation potential tests indicated that dichloroacetonitrile (DCAN) was the dominant of the 7 HANs formed in the presence of chlorine and no bromide. The amount of DCAN formed for the seven precursors under this study due to chlorination is TMAC<TEAC<DTMA<BEC<CTMA<BZK<CPC. The micromolar yield of HANs upon reaction between 5 mg/L chlorine and 0.1 mM precursor is represented in the Figure 4-5. The DCAN micromolar yield of approximately 0.001% due to chlorination of CPC is definitely lower than some precursors. For instance, the DCAN yield of the amino acid tyrosine (0.5 mM) upon reaction with chlorine for 7 days is 0.095% (Chu et al., 2015). However, this is the first time HAN formation as a consequence of reaction between BEC, TMAC, TEAC, CPC, CTMA, DTMA and BZK is reported. It should be noted that HANs undergo hydrolysis, and HAN degradation increases with higher free chlorine concentration and pH (Glezer et al., 1999; Yu and Reckhow, 2015). The formation potential test used here (Standard Method 5710) was created for THMs and may not be suitable for maximizing HAN formation. Thus, the micromolar yields for HANs might be higher in water distribution systems as compared to this study.

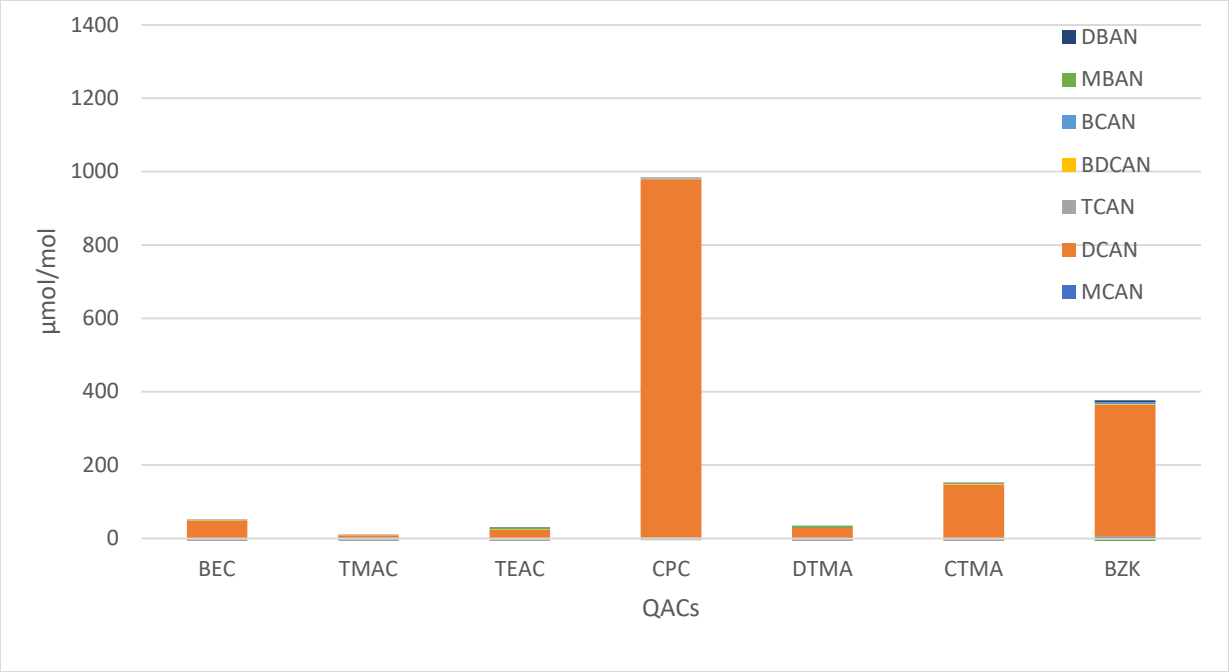


Figure 4-5 Micromolar yield of HANs (µmol HAN / mol surfactant) upon reaction between 5 mg/L chlorine and 0.1 mM QAC precursor.

4.7 Potential HAN Formation Mechanism for QACs

It can be hypothesized that, for the formation of nitrogenous DBPs during chlorination, "N" is sourced from the precursor molecule. For BZK (0.1 mM), the formation of DCAN upon reaction with 5 mg/L Cl_2 could likely be associated with two reaction pathways. As mentioned by Linge et al. (2020), DCAN is formed due to the reaction of the alkyl acetonitrile with chlorine species following the loss of the aromatic group as the aromatic group is a resonance-stabilized leaving group. Another possible formation mechanism could be the addition of chlorine to the aromatic

ring via electrophilic substitution, thereby resulting in the formation of chlorinated aromatic ring and, eventually, cleavage of the aromatic ring and subsequent formation of DCAN.

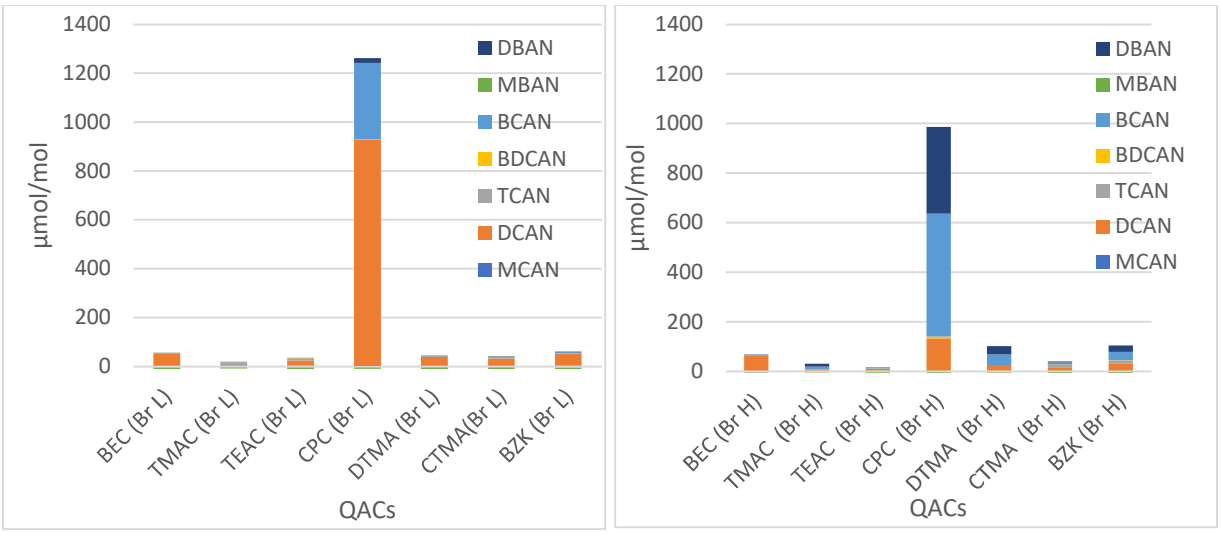
The formation of DCAN by CPC could potentially be explained by the following process: N-ethylpyridinium, a compound similar to CPC with the only difference being the alkyl chain length, was converted to N-ethylpyridone upon reaction with an oxidant (Lin Wang et al., 2014). Chakrabartty et al. (1974) also suspected the formation of 2-pyridone when sodium hypohalite reacted with pyridine. The formation of pyridones from the oxidation of pyridinium salts is also confirmed by (Fuji et al., 1975). The pyridones might undergo further halogenation using residual chlorine and could yield trichloro substituted 2-pyridone (Katritzky et al., 1984). Trichloro substituted 2-pyridone can potentially undergo hydrolysis followed by ring cleavage to form choromethanamine, chlorocarbamic acid, 1-chloro-2-iminoethan-1-ol (Zhang et al., 2017). 1-chloro-2-iminoethan-1-ol can yield HAN based on the hypothesized pathway proposed by (Zhang et al., 2017).

Pyridones can also exist in their tautomeric state which is hydroxypyridone (C. Poully et al., 2010). In the case of the QAC precursors, the tautomeric state would be as alkoxypyridones. N-alkoxypyridones would then decompose to the parent pyridine and carbonyl compound (Silwa and Tartar, 1976). The pyridine could react with sodium hypohalite to yield CO_2 and ammonia (Chakrabartty and Kretschmer, 1974). Ammonia can then react with residual chlorine to form chloramine, which in turn reacts with the alkyl or carbonyl group to yield HAN.

4.8 Effect of Bromide on HAN Speciation

Similar to THMs, brominated HANs are relatively more toxic than their non-brominated analogue. DBPFP tests were conducted at 50 and 500 µg/L Br^- of bromide, thereby representing low and high levels of bromide in water sources. The micromolar yield of the 7 HANs was calculated and is represented in Figure 4-6. Given the higher affinity of bromine towards substitution compared to chlorine, formation of brominated HANs is favored (Bond et al., 2011). As seen in Figure 4-6, the higher the concentration of bromide in the sample matrix, the higher the micromolar yield of brominated HANs and the lower the yield of DCAN. Bromide, upon reaction with HOCl, forms HOBr which has a substantially higher reactivity in terms of substitution compared to HOCl. From Figure 4-6 (a) and Figure 4-6 (b), it can be seen that there is a drop in the total HANs formed when the concentration of bromide increases from 50 to 500 µg/L.

A similar trend was observed during the chlorination of histidine and glycine where there was a drop in the yield of HANs when the concentration of bromide increased (Li et al., 2017).
It is also very likely that the HANs formed due to chlorination could hydrolyze to haloacetamides (Chen, 2011; Glezer et al., 1999). It has been seen that tri- and di-halogenated HANs are more susceptible to hydrolysis compared to the monochlorinated HANs at a pH of 7.2 (Glezer et al., 1999), similar to the pH that is maintained throughout this study. In addition, the presence of hypochlorous acid can act as a catalyst towards hydrolysis of HAN to form N-chloroamides (Glezer et al., 1999).



(a) (b)

Figure 4-6 Micromolar yield of HANs (µmol HAN / mol surfactant) upon reaction between 5 mg/L chlorine and 0.1 mM QAC precursor

4.9 Bromine Substitution Factor for HANs:

BSF for HANs is calculated by taking the ratio of the molar concentration of brominated HANs to the molar concentration of the total HANs formed (Zheng et al., 2020). The calculation of BSF for THM and is given by the following equations.

$$BSF_{HANs} = \frac{MBAN \times BDCAN \times BCAN + (2 \times DBAN)}{4 \times (MCAN + DCAN + TCAN + BDCAN + BCAN + MBAN + DBAN)}$$

Based on Figure 4-7 showing the bromine substitution factor, when the concentration of Br^- increases from 0 to 50 µg/L Br^-, there is a slight increase in %BSF for CPC and when the concentration of Br^- increases from 50 to 500 µg/L Br^-, an increase in the %BSF is observed for

some of the QACs. However, the yield of HAN for most of the QACs is below 200 µmol/mol, except for CPC, for which the HAN yield is approximately between 1000 to 1200 µmol/mol., indicating that among the 7 QACs included in this study, CPC, a compound with a heterocyclic ring formed significantly more HANs during chlorination. There are certain values in the %BSF equation that are below the MRL. As values below MRL cannot be used directly in the equation, these values are assumed to be zero prior to calculation. The assumption of zero is established based on information obtained from literature (Croghan and Egeghy, 2003), where the others stated that in statistical analysis, values that are below reporting limits are substituted by constant values such as zero. While the best approach would be to use computational tools in order to reduce left-censored bias, a large data set would be required and in this study the data set is limited to only duplicates.

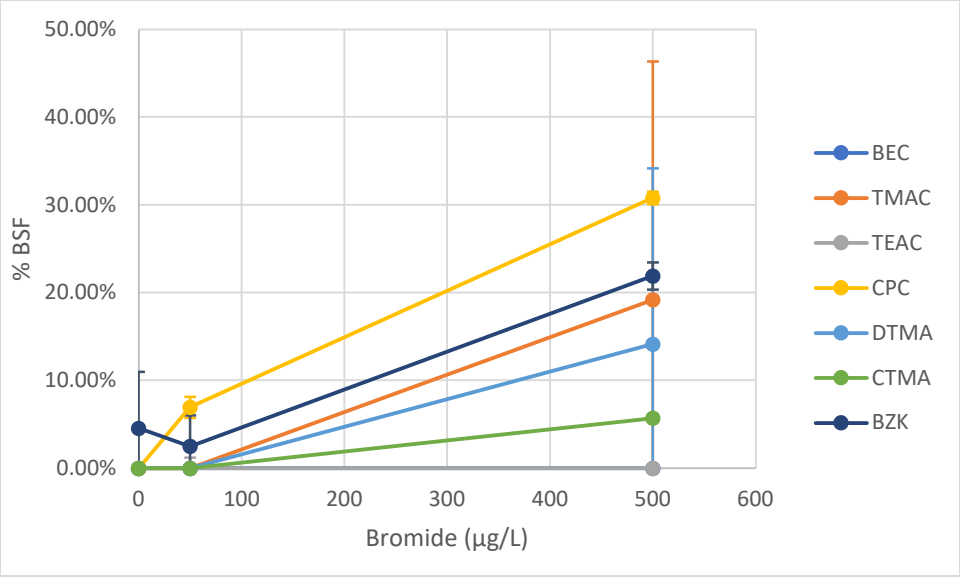


Figure 4-7 Percent bromine substitution factor for HANs upon reaction between 5 mg/L of chlorine and 0.1 mM QAC precursor.

For BEC, it can be seen that the bromine substitution in HANs is low even at a high bromide dose of 500 µg/L, similar to the BSF as seen with THMs. As discussed in section 4-5, the reaction between BEC and chlorine could have formed a variety of chlorinated and brominated intermediates, thus leaving insufficient bromine for the formation of brominated HANs. The BSF for HANs formed due to the reaction between CPC (0.1 mM) at 5 mg/L Cl_2 jumps to a higher value with higher dose of bromide just as seen with the BSF for THMs. For BZK, a some % BSF is observed at 0 µg/L bromide. However, the standard deviation is quite high making it statistically insignificant. With the exception of BEC, the %BSF increases as the concentration of bromide in the sample matrix increases for QACs considered in this study.

4.10 Nitrosamine Formation

There was no significant formation of NDMA or other nitrosamines upon reaction of the 7 QACs with chlorine. The analysis of NDMA after the chlorination formation potential indicated values below MRL for all the precursors included in this study. It is expected that formation of nitrosamines is insignificant during chlorination as the formation pathway involves direct reaction with monochloramine rather than free chlorine (Mitch and Sedlak, 2001b; Chen and Young, 2008). Even for a prominent NDMA precursor dimethylamine, NDMA formation due to chlorination is an order of magnitude lower compared to NDMA formation due to chloramination (Mitch and Sedlak, 2001b).

4.11 Conclusions

Based on the results of the chlorination formation potential experiments of the 7 QACs included in this study, it can be seen that certain precursors contribute to DBP formation. The amount of

TCM formed for the seven precursors under this study due to chlorination follows the order of TMAC<TEAC<BEC<CPC<DTMA<CTMA<BZK. The yield of THMs per mole QACs at a chlorine dose of 5 mg/L as Cl_2 ranged from approximately 160 to 6700 µmol/mol. In the presence of bromide, the speciation of THMs shifted to brominated analogues, and, as the concentration of bromide increased from 50 to 500 µg/L, the degree of bromine substitution also increased for all QACs except BEC. QACs, based on their functional groups are potential precursors to THMs, however, the yield of THMs due to QACs is low compared to prominent THM precursor such as resorcinol which has TCM molar yield of 0.95 (Gallard and Gunten, 2002).

CPC showed notable formation of HANs with DCAN yield of 980 µmol/mol followed by BZK and CTMA at 360 and 150 µmol/mol, respectively. For every other QAC, the yield of HANs was below 100 µmol/mol in the absence of bromide. In the presence of bromide, the speciation of HANs shifted to brominated analogues and, as the concentration of bromide increased from 50 to 500 µg/L, the degree of bromine substitution also increased which is attributed to the higher reactivity of $HOBr^-$. CPC is a potential precursor to HANs relative to the 7 QACs included in this study, However, in comparison to other known HAN precursors, the HAN yield from the chlorination of CPC is relatively low. For example, the DCAN yield of the amino acid tyrosine (0.5 mM) upon reaction with chlorine for 7 days is 0.095%. As expected, none of the QACs formed NDMA upon reaction with 5 mg/L free chlorine at a pH of 7, as monochloramine (electrophile) is required to attack the nitrogen of the DMA group in QACs to form any NDMA (Mitch and Sedlak, 2004).

Chapter 5 – Chloramination Formation Potential: Results and Discussion

5.1 Introduction

As studies have shown that chlorination of water samples containing organic matter contributes to higher amounts of regulated DBPs such as THMs and HAAs (Chu et al., 2017; Huang et al., 2016; Shah et al., 2011), many utilities have resorted to alternative disinfection methods. One such alternative method is chloramination as it is proven to form relatively fewer THMs and HAAs. However, chloramination of waters containing organic matter is shown to form nitrogenous DBPs such as HANs, NOAs, haloacetamides (HAMs), haloketones (HKs), etc., which are relatively more toxic than the regulated DBPs. In this study, a specific group of organic precursors—quaternary ammonium compounds (QACs)—are assessed for their potential to form DBPs such as THMs, HANs, and NDMA upon chloramination. Additionally, the speciation of DBPs is also observed when bromide is spiked into the chloramination sample matrix.

The focus on QACs is attributed to the fact that these compounds are present in a plethora of household and personal care products such as laundry detergents, fabric softeners, surface disinfectants, cosmetics, mouth wash, hand sanitizers, etc. The use of these compounds is not only limited at the household level but also widely employed in the hospital industry, thus contributing to a large QAC load in hospital wastewaters. In recent times, QACs are gaining a lot of attention due to their ability to inactivate the SARS-CoV-2 virus. In fact, these QACs are present in over 216 commercial products recommended by the CDC to inactivate the SARs-

CoV-2 virus on surfaces. Given the current circumstances, i.e., the COVID-19 pandemic, it is reasonable to expect higher QAC load in wastewater compared to QAC concentrations before the pandemic.

Eventually, QACs may end up in drinking water sources. Several studies have focused on the harmful effects of QACs. For example, QACs may promote antibiotic resistance in bacteria and are potential carcinogens (Kim et al., 2018; Harrison et al., 2020). However, studies that focus on the potential of QACs to form DBPs are rather scarce. On account of this research gap, the current chapter focuses on the DBP formation potential of QACs due to reaction with monochloramine.

Additionally, it is also important to take into account the presence of bromide in the sample matrix since bromide ion tends to form brominated DBPs. Since brominated BBPs are more toxic than their chlorinated analogues, the change in speciation due to bromide presence can substantially increase the overall toxicity of the disinfected water. In this regard, a separate set of experiments was conducted to assess the potential of QACs to form DBPs during chloramination in the presence of bromide.

5.2 Methods

5.2.1 Formation Potential Experiments

Preformed monochloramine (preparation described in section 3.1.3) was individually spiked into amber reaction bottles containing 0.1 mM QACs in buffered DI water (pH 7) to achieve a final total chlorine concentration of 140 mg/L in the sample matrix. The samples are subjected to a high concentration of monochloramine, as used in many prior studies (Beita-Sandí et al., 2020; le

Roux et al., 2011; Roux et al., 2012c), to assess the worst-case scenario for DBP formation from QACs. To assess the speciation of DBPs in the presence of bromide at low and high bromide concentrations, 50 and 500 µg/L of Br^- was spiked into the chloramination sample matrix, respectively. The samples were allowed to react for 10 days in the dark at room temperature.

5.2.2 DBP Extraction and Analysis

At the end of the 10-day reaction period, residual total chlorine was measured in the sample matrix followed by quenching of monochloramine using sodium thiosulfate in order to terminate the chloramination reaction. DBPs were then extracted from the samples following appropriate extraction protocols as described in section 3.4. The extracted samples were analyzed for the corresponding DBPs using GC-MS. The methods for GC-MS analysis for specific DBPs are detailed in section 3.4.

5.3 THM Formation

The results from the chloramination formation potential tests indicated that trichloromethane (TCM) was the dominant of the 4 THMs formed in the presence of chlorine without any bromide. The micromolar yield of TCM formed for the seven precursors under this study due to chloramination is TMAC<TEAC<DTMA<CTMA<BZK<BEC<CPC. The formation of THMs during chloramination of 1 mM BZK was also observed by Chang et al. (2011), wherein the yields of THMs were 332.2±17.5 µg/L at a monochloramine dosage of dose of 140 mg/L. The yield of THMs per mol QACs at a monochloramine dose of 140 mg/L Cl_2 is shown in Figure 5-1. The yield yield of TCM was 3100, 36, 40, 3100, 730, 990, 2700 µmol/mol for BEC, TMAC, TEAC, CPC, DTMA, CTMA and BZK, respectively.

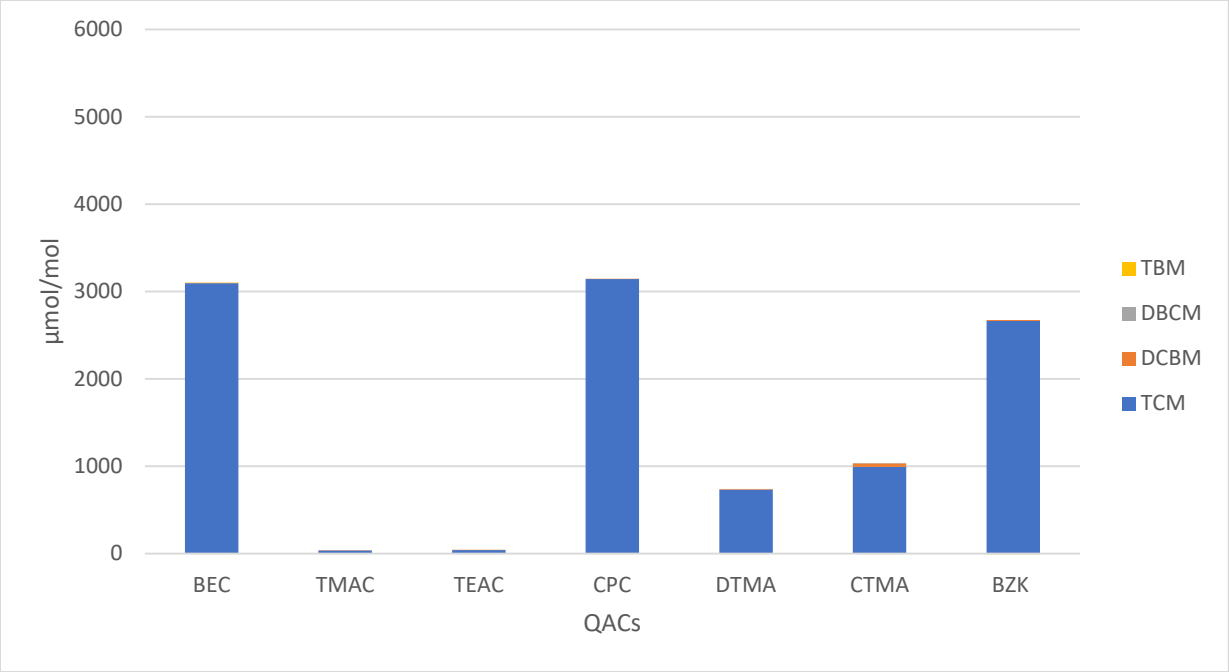


Figure 5-1 Micromolar yield (µmol THMs / mol surfactant) of THMs upon reaction between monochloramine at 140 mg/L Cl2 and 0.1 mM precursor (QAC).

5.4 THM Formation Pathway

The mechanism of formation for QACs into THMs is not yet known. It is possible that monochloramine could have been hydrolyzed to free chlorine, which then could have participated in the formation of TCM (Cimetiere et al., 2010; Nihemaiti et al., 2016); however, only a small fraction of monochloramine would hydrolyze, making this a minor reaction pathway. It is thus reasonable to assume that THM formation is likely attributed to direct oxidation of the QAC by monochloramine followed by decomposition of any oxidized intermediates formed in the process (Bi et al., 2013; Sfynia et al., 2020a). The formation of THMs as a result of reaction between BZK and monochloramine was also observed by another

group of researchers (Chang et al., 2011), but no mechanism was established. However, this is the first time THM formation due to the chloramination of BEC and CPC is reported. From Figure 5-1, it can be observed that TMAC and TEAC did not yield significant amounts of TCM, indicating that functional groups play an important role in THM formation during reaction with disinfectants. The reason for low TCM yields from TMAC and TEAC might be due to the possibility that they are unreactive to chlorine; however, information on this is unknown.

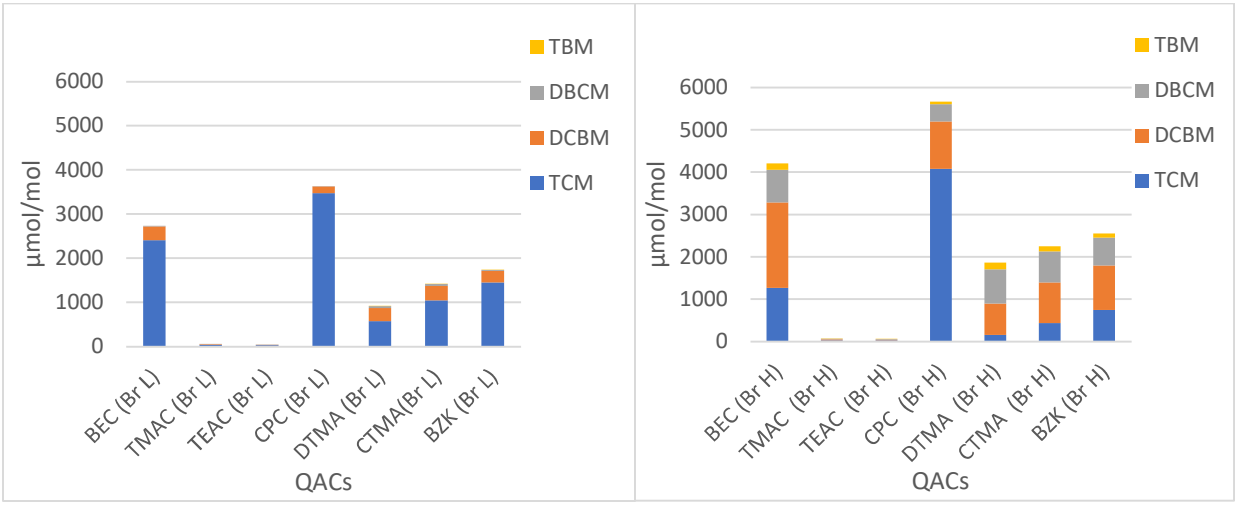
Among the precursors that contributed to TCM, the aromatic precursors, namely BEC, CPC and BZK, yielded higher amounts of TCM compared to the non-aromatic precursors, namely DTMA and CTMA. The benzyl moiety is a better leaving group compared to the alkyl moiety (Piazzoli et al., 2018a); thus, oxidation of the benzyl moiety by chloramine should dominate compared to oxidation of the alkyl chain. In any case, studies have shown that aldehydes are formed during the chloramination of benzylated amines (Cimetiere et al., 2010; Kemper et al., 2010) and chlorinated pyridinium compounds (Nihemaiti et al., 2016). The aldehydes could potentially react with free chlorine (formed as a hydrolysis product of monochloramine) to form THMs.

5.5 Effect of Bromide on THM Speciation

As mentioned before, brominated DBPs are relatively more toxic than their chlorinated counterparts. DBPFP tests using preformed monochloramine were conducted at 50 and 500 µg/Lof bromide, thereby representing low and high levels of bromide in water sources. It can be seen from Figure 5-2, the higher the concentration of bromide in the sample matrix, the higher the micromolar yield of brominated THMs and the lower the yield of trichloromethane. At low bromide concentration (50 µg/L), the formation of DCBM increased, and when the concentration

of bromide was high at 500 µg/L, the speciation of THMs shifts towards higher degree of bromination such as DBCM and TBM with a decrease in TCM concentration.

According to Figure 5-2 (a) and (b), there is an overall increase in THMs when the concentration of bromide increases from 50 to 500 µg/L. This is because bromide reacts with NH_2Cl to form NH_2Br, NHClBr and $NHBr_2$ which have more reactivity toward precursors compared to NH_2Cl (Alsulaili, 2009; R. Liu et al., 2018). Thus, at a higher bromide concentration in the sample matrix, the greater concentration of reactive bromamine species contributed to higher total THMs.



(a) (b)

Figure 5-2 Micromolar yield (µmol THMs / mol surfactant) of THMs upon reaction between monochloramine at 140 mg/L Cl_2 and 0.1 mM of QAC precursor: (a) in the presence of 50 µg/L Br^- and (b) in the presence of 500 µg/L Br^-.

5.6 Bromine Substitution Factor for THMs

BSF for THMs is calculated by taking the ratio of the molar concentration of brominated THMs to the molar concentration of the total THMs formed (Zheng et al., 2020). The calculation of BSF for THM is given by the following equations.

$$BSF_{THMs} = \frac{DCB \quad (2 \times DBCM)+(3 \times CTBM)}{3 \times (TCM+DCBM+DBCM+ \quad)}$$

Based on Figure 5-3, when the concentration of Br^- increased from 0 to 50 µg/L Br^-, the increase in %BSF for the seven quaternary ammonium compounds included in this study was in the order of TEAC<TMAC<CPC<BEC<BZK<CTMA<DTMA, and when the concentration of Br^- increased from 50 to 500 µg/L Br^-, the increase in the %BSF was in the order of CPC<BEC<BZK<TEAC <TMAC<CTMA<DTMA. Although the %BSF of TEAC and TMAC increased with higher Br^- concentration, the yield of THMs was relatively low as seen in Figure 5-3.

In the presence of bromide at 50 µg/L, the BSF ranges 2-15%. At a bromide concentration of 500 µg/L, the BSF goes as high as 50% for most of the compounds, except for CPC which is at 12%. Although bromine substitution for THMs in the case of CPC is low, from Figure 5-2 (a) and (b), it can be seen that CPC contributes to higher THM yield in comparison to the other QACs included in this study. The low BSF is likely attributed to the high monochloramine to bromide ratio (140:0.5). While brominated THMs are formed in the presence of bromide, the presence of monochloramine at a high concentration (140 mg/L Cl_2) shifts the equilibrium to favor more chlorinated species. As for the BSF of the remaining QACs in this study, it is possible that monochloramine is less reactive to these QACs even at high concentrations. In the presence of bromide, bromide reacts with monochloramine (NH_2Cl) to form NH_2Br, NHClBr and $NHBr_2$

which have more reactivity toward precursors compared to NH_2Cl (Alsulaili, 2009; R. Liu et al., 2018), hence favoring higher bromide substitution for THMs contributed by BEC, DTMA, CTMA and BZK during chloramination. There are certain values in the %BSF equation that are below the MRL. As value below MRL cannot be used directly in the equation, these values are assumed to be zero prior to calculation. The assumption of zero is established based on information obtained from literature (Croghan and Egeghy, 2003), where the others stated that in statistical analysis, values that are below reporting limits are substituted by constant values such as zero. While the best approach would be to use computational tools in order to reduce bias, a large data set would be required and in this study the data set is limited to only duplicates.

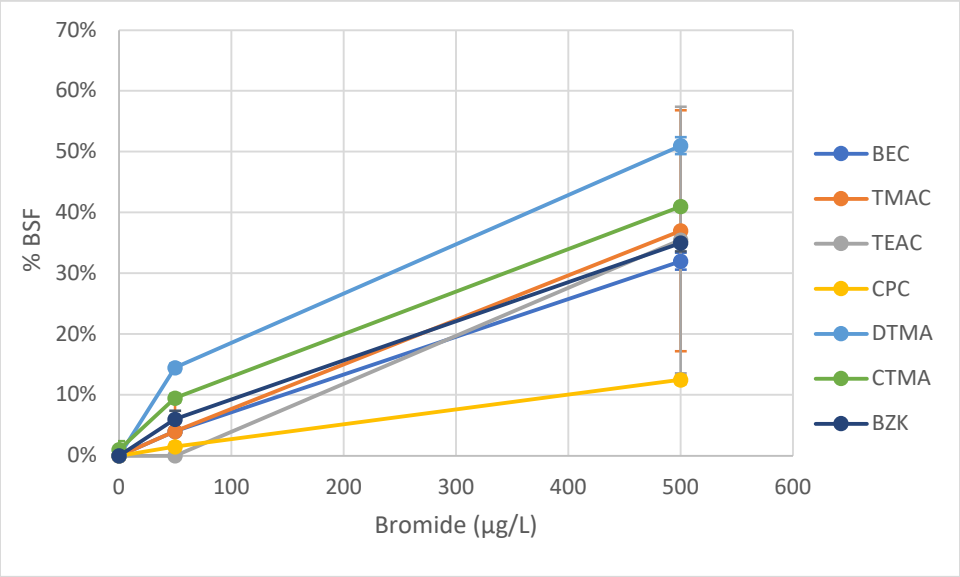


Figure 5-3 Percent bromine substitution factor for THMs upon reaction between monochloramine at 140 mg/L Cl2 and 0.1 mM QAC precursor.

5.7 HAN Formation

The results from the formation potential tests indicated that DCAN was the dominant of the 7 HANs that were formed in the presence of chlorine with no bromide (Figure 5-4). The amount of DCAN formed for the seven precursors under this study due to chloramination is TMAC<TEAC<CTMA<DTMA<CTMA<BZK<BEC<CPC. The molar formation of DCAN upon disinfection varies depending on the disinfection dose, compound structure, and functional groups (Le Roux et al., 2011; Yang et al., 2012b). Chloramination of phenol at pH 7 yielded a molar ratio of 34,600 µmol/mol while under similar conditions 1,2,4-trihydroxybenzene yielded 450 µmol/mol (Shao et al., 2020). In our experiments, the highest DCAN micromolar yield was obtained for CPC at 4300 µmol/mol.

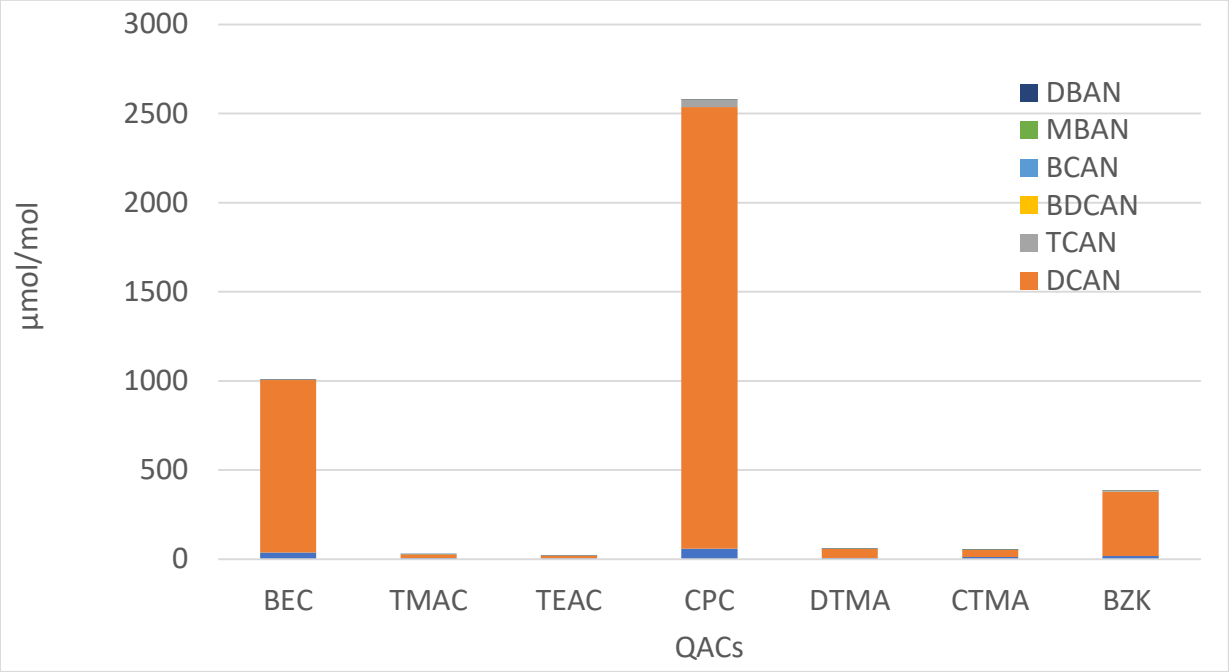


Figure 5-4 Micromolar yield (µmol HAN / mol surfactant) of HANs upon reaction between monochloramine at 140 mg/L Cl_2 and 0.1 mM QAC precursor

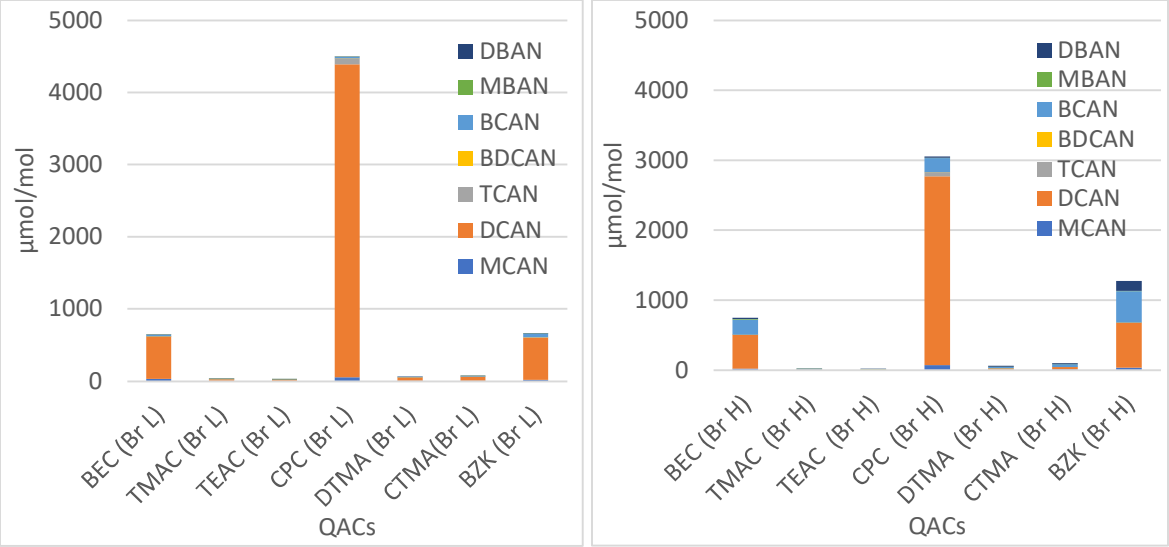
5.8 HAN Formation Pathway

The nitrogen in HAN could potentially be sourced from the nitrogen in the precursor molecule or from the chloramine molecule. It is likely that the alkyl pyridinium compound would form its oxidation product, pyridone. Upon reaction with excess preformed monochloramine, trichloro substituted pyridone could be formed. Eventually, cleavage of the heterocyclic ring occurs upon continued reaction with monochloramine resulting in the formation of haloacetaldehydes and haloacetamides (Nihemaiti et al., 2016). Nucleophillic addition of monochloramine on the carbonyl carbon of the haloacetaldehyde molecule takes place resulting in the formation of carbinolamines. Carbinolamines lose a water molecule to halo-imines followed by decomposition to form haloacetonitriles (Kimura, 2013).

BEC, CPC and BZK were the only compounds of the 7 precursors included in this study that formed significant amounts of HANs upon reaction between 0.1 mM precursor and 140 mg/L monochloramine. Based on the structures of these compounds (Table 3-2), it can thus be assumed that aromatic precursors react well with monochloramine to form HANs compared to the non-aromatic precursors considered in this study (L. Linge et al., 2020). Similar observations were recorded where DCAN formation was higher for aromatic amines than the non-aromatic analogues. The authors also concluded that the popular aldehyde formation pathway for DCAN formation from amino acids is not followed for DCAN formed from amino acids with an aromatic ring system. The most probable HAN formation pathway would involve the cleavage of the aromatic structure followed by electrophilic substitution of chlorine and the addition of monochloramine to the ring system (Linge et al., 2020). Interestingly, it is highly possible that nitrogen in DCAN is sourced from mostly from monochloramine (Linge et al., 2020).

5.9 Effect of Bromide on HAN Speciation

Similar to THMs, the brominated species of HANs are more toxic than the chlorinated analogues. DBPFP tests were conducted at 50 and 500 ppb of bromide, thereby representing low and high levels of bromide in water sources. The micromolar yield of the 7 HANs was calculated and is represented in Figure 5-5. At a bromide concentration of 50 µg/L Br^-, the speciation had not shifted much compared to samples with no $bromide^-$; DCAN was the greatest fraction. However, at a high bromide concentration of 500 µg/L Br^-, the speciation shifts slightly resulting in the formation of BCAN and DBAN. Overall, even in the presence of bromide, DCAN is the dominant HAN formed during chloramination.



(a) (b)

Figure 5-5 Micromolar yield (µmol HAN / mol surfactant) of HANs upon reaction between monochloramine at 140 mg/L Cl_2 and 0.1 mM of QAC precursor: (a) in the presence of 50 µg/L Br- and (b) in the presence of 500 µg/L Br-.

5.10 Bromine Substitution Factor for HANs

BSF for HANs is calculated by taking the ratio of the molar concentration of brominated HANs to the molar concentration of the total HANs formed (Zheng et al., 2020). The calculation of BSF for HANs is given by the following equations.

$$BSF_{HANs} = \frac{MBAN + BDCAN + BCAN + (2 \times DBAN)}{4 \times (MCAN + DCAN + TCAN + BDCAN + BCAN + MBAN + DBAN)}$$

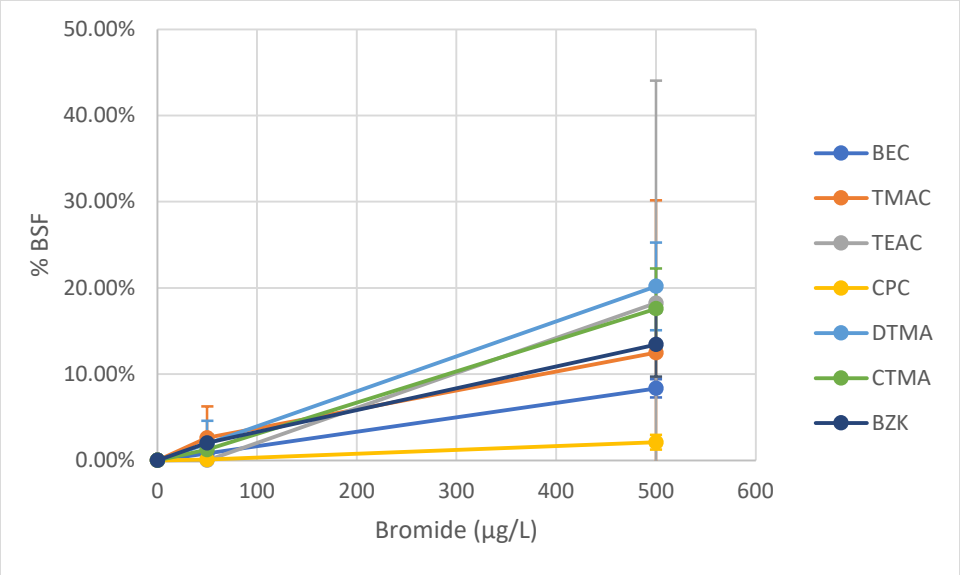


Figure 5-6 Percent bromine substitution factor for HANs upon reaction between monochloramine at 140 mg/L Cl_2 and 0.1 mM QAC precursor.

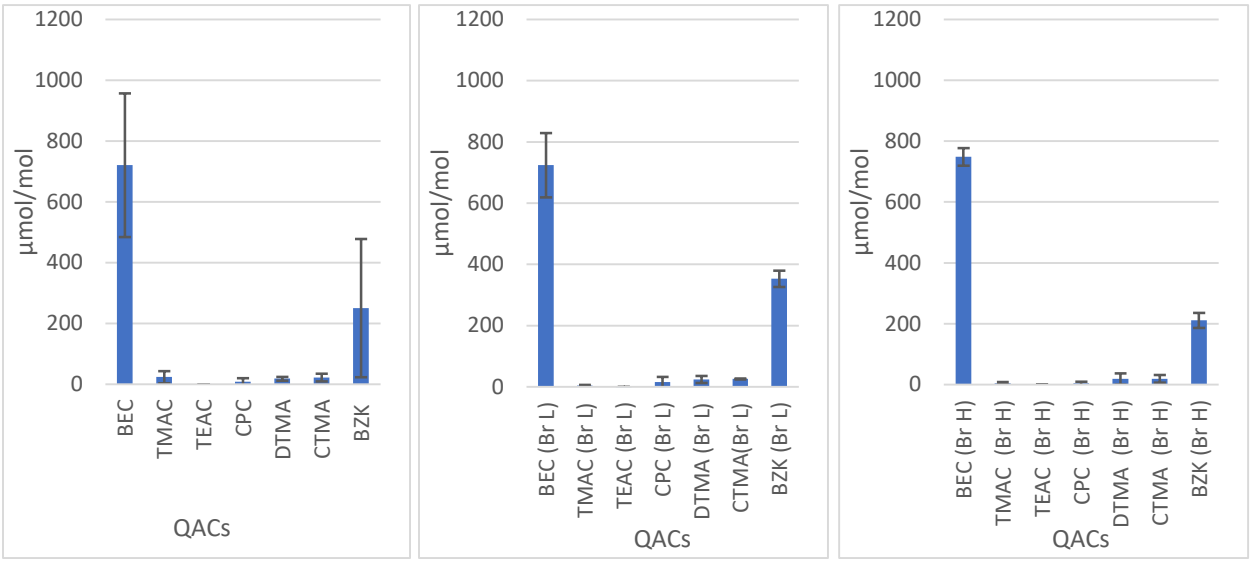
Based on Figure 5-6, when the concentration of Br^- increases from 0 to 50 µg/L Br^-, the increase in %BSF for the seven quaternary ammonium compounds included in this study is in the order of TEAC<CPC<BEC<CTMA<DTMA<BZK<TMAC, and when the concentration of Br^- increases from 50 to 500 µg/L Br^-, the increase in the %BSF is in the order of CPC<BEC<TMAC<BZK<TEAC<CTMA<DTMA. Although the %BSF of TEAC and TMAC is high with increase in Br^- concentration, the yield of HANs is relatively low as seen in Figure 5-5. It is very likely that bromine could have been consumed for substitution in the intermediate compounds that would have formed due to the reaction between BEC, CPC and BZK and monochloramine, similar to that suggested by Chang et al. (2011), thus explaining the low BSF for those precursors. For precursors such as TMAC, TEAC, DTMA and CTMA, although BSF values are higher than that of CPC, BEC, and BZK, the former group of precursors did not yield

significant amounts of HANs. There are certain values in the %BSF equation that are below the MRL. As value below MRL cannot be used directly in the equation, these values are assumed to be zero prior to calculation. The assumption of zero is established based on information obtained from literature (Croghan and Egeghy, 2003), where the others stated that in statistical analysis, values that are below reporting limits are substituted by constant values such as zero. While the best approach would be to use computational tools in order to reduce bias, a large data set would be required and in this study the data set is limited to only duplicates.

5.11 NDMA Formation

The results of the chloramine FP tests clearly show that among the 7 QACs included in this study, BEC and BZK are the only precursors that formed NDMA as seen in Figure 5-7. The formation of NDMA for TMAC, TEAC, CPC, DTMA, and CTMA were all below the MRLs. In addition, other nitrosamines were not detected above their reporting limits. The average micromolar yield of NDMA from BEC and BZK due to chloramination at 140 mg/L Cl_2 in the absence of bromide were 720 and 250 μmol/mol respectively. The micromolar yield of NDMA by BEC is approximately 3 times that of BZK, which is a well-known precursor for NDMA (Chang and Wang, 2011; Kemper et al., 2010; Piazzoli et al., 2018). In a study by Piazzoli et al. (2018), the authors reported NDMA formation below 0.09% molar yield upon reaction of BZK with a 10-fold molar excess of monochloramine at a pH of 7, and this is consistent with the findings of this research. However, when the pH was increased to 8, the NDMA molar yield increased to 0.56±0.03% (Piazzoli et al., 2018). This indicates that the molar yield of NDMA is pH dependent. This pH dependency could be attributed to the increased stability of monochloramine at higher pH, thereby enabling higher monochloramine exposure. Upon

comparing total nitrosamine yield and NDMA yield, it was observed that NDMA accounted only for a minor portion of the total nitrosamines (Piazzoli et al., 2018). Further analysis indicated that N-nitroso-N-methyldodecylamine, a novel nitrosamine, contributed to a major fraction of nitrosamines (Piazzoli et al., 2018). In that regard, although our analysis did not include this novel nitrosamine, it cannot be neglected that BZK is a nitrosamine precursor, and QACs may form nitrosamines that were not analyzed in this study.



(a) (b) (c)

Figure 5-7 Micromolar yield (µmol NDMA / mol surfactant) of NDMA upon reaction between monochloramine at 140 mg/L Cl_2 and 0.1 mM of QAC precursor: (a) absence of Br^- (b) in the presence of 50 µg/L Br^- and (c) in the presence of 500 µg/L Br

The formation of NDMA upon chloramination of BZK could potentially be due to the electron donating nature of quaternary ammonium and benzyl radical functional groups (Chang et al., 2011; Kemper et al., 2010). The cleavage of the nitrogen and benzyl bond as a result of nucleophilic substitution reaction by monochloramine likely results in the formation of dimethylalkylamine and a tropylium cation. Dimethylalkylamine, upon continued reaction with preformed monochloramine, could yield NDMA (Chang et al., 2011; Tezel and Pavlostathis, 2009). From Figure 5-7 it can also be seen that the presence of bromide at 50 or 500 μg/L has no influence on the formation of NDMA since the micromolar yield is similar.

This is the first time the formation of NDMA is reported due to reaction between BEC and monochloramine. Due to extremely scarce information on the potential intermediates of BEC upon the reaction with monochlormaine, it is difficult to predict a formation pathway at this point of time. However, given the fact that benzyl group is a better leaving group compared to an alkyl group, the benzyl group imparts a stronger electron donating effect on the DMA moiety; therefore, higher NDMA formation rates would be expected during chloramination (Piazzoli et al., 2018; Shen and Andrews, 2011). The presence of the benzyl group could explain why BEC and BZK were the only ones that formed NDMA upon reaction with monochloramine out of the 7 QACs included in this study as they were the only compounds that contain benzyl groups. This could also potentially explain why BEC, which has 2 benzyl groups, formed NDMA at a relatively higher micromolar yield compared to BZK which only has 1 benzyl group. Nevertheless, further studies are warranted to investigate potential NDMA formation pathways due to reaction between BEC and monochloramine by possibly conducting a full scan GC-MS analysis and identifying potential intermediates.

It was also reported by Piazzoli et al. (2018) that the reaction between 200-fold molar excess of monochloramine with 0.1 mM TMAC and CTMA did not yield significant NDMA, which is consistent with the findings of our studies. Given that the structures of TEAC and DTMA are similar to TMAC and CTMA respectively, it is reasonable to assume that there was no significant NDMA formation because TMAC, TEAC, CTMA, and DTMA are less susceptible to reactions with monochloramine compared to benzylated quaternary amines. Also, it has been stated by Mitch and Sedlak (2004) that QACs contribute to relatively lower NDMA yields because of the absence of a free lone pair on the DMA-nitrogen in QAC molecules, thus making it difficult for monochloramine (electrophile) to react.

5.12 Conclusions

Based on the results of the chloramination formation potential experiments of the 7 QACs included in this study, it can be seen that certain precursors contribute to DBP formation. Trichloromethane formed was in the order of TMAC<TEAC<DTMA<CTMA<BZK<BEC<CPC with the yield ranging from approximately 36 to 3140 µmol/mol. In the presence of bromide, the speciation of THMs shifted to brominated analogues, and, as the concentration of bromide increased from 50 to 500 µg/L, the degree of bromine substitution also increased for all QACs,

Dichloroacetonitrile was the dominant species formed upon reaction between the seven quaternary ammonium compounds (0.1 mM) and monochloramine (140 mg/L as Cl_2) in the order of TMAC<TEAC<CTMA<DTMA<CTMA<BZK<BEC<CPC with yields ranging from 4.5 to 4000 µmol/mol. In the presence of bromide, the speciation of HANs shifted slightly to brominated analogues and, as the concentration of bromide increased from 50 to 500 µg/L, the

degree of bromine substitution also increased. However, DCAN remained as the primary HAN formed.

Notable formation of NDMA was observed during the reaction between QACs (0.1 mM) and monochloramine (140 mg/L Cl_2), especially with benzylated precursors, namely BEC and BZK. Given that the benzyl group is a relatively better leaving group comapred to alkyl groups and also that the electron donating nature of the benzyl group increases the elecrtron density on the N atom, a neucleophillic attack on the chloramine is favoredClick or tap here to enter text., thus explaining why the benzylated QACs formed NDMA in comparison to the remaining 5 QACs. It was also interesting to note that BEC formed NDMA with a yield 3 times that formed by BZK. This is likely due to the presence of two aromatic rings in BEC as compared to one ring in BZK.

Chapter 6 - Prediction of DBP Concentrations in the Environment by QAC Precursors

Based on the DBP micromolar yield values obtained from this study, predictive calculations were done to assess concentrations of DBPs that could be formed by chlorination of QACs at environmentally relevant concentrations. In order to achieve this, an assumption was made that the concentrations of all QACs in this study are the same. This assumption was made based on the lack of information regarding environmental concentrations of QACs in water sources. Among the 7 QACs included in this study, BZK was the only QAC for which environmental concentrations were reported. The concentration of BZK in Taiwanese rivers with industrial effluents were as high as 65 and 100 µg/L respectively (Ding and Liao, 2000). BZK concentration in the surface water downstream from five wastewater treatment plants (WWTP) in the US was as high as 36.6 µg/L. Pati and Arnold (2020) reported a BZK concentration of approximately 8 µg/L upon analyzing the wastewater effluents from several WWTPs in Minnesota, US. However, all these values reported are before the COVID-19 pandemic. Considering the functionality and application of QACs, it can be expected that the concentrations of QACs in the wastewater effluents can be higher today as a consequence of the COVID-19 pandemic (Hora et al., 2020). In that regard, a QAC value of 400 µg/L is assumed to be present in the environment and the corresponding concentration of DBPs at a pH of 7 is summarized from Table 6-1 through Table 6-5. 400 µg/L represents one order of magnitude increase in BZK concentrations compared to measured concentrations in surface water downstream from WWTPs in the US. The prediction is made based on the DBP micromolar yield results obtained from the experiments in the research.

Table 6-1 Predicted concentration of THMs during chlorination (5 mg/L Cl_2) of 400 ug/L of QACs.

CBD – Cannot Be Determined (Because one or more experimental values are below MRL)

Sample	**Bromide Conc. (µg/L)**	**TCM (µg/L)**	**DCBM (µg/L)**	**DBCM (µg/L)**	**TBM (µg/L)**	**Total THM Env. Conc. (µg/L)**
BEC	0	0.14	CBD	CBD	CBD	0.14
TMAC	0	0.051	CBD	CBD	CBD	0.051
TEAC	0	0.044	CBD	CBD	CBD	0.044
CPC	0	0.34	CBD	CBD	CBD	0.34
DTMA	0	0.61	CBD	CBD	CBD	0.61
CTMA	0	0.69	CBD	CBD	CBD	0.69
BZK	0	1.2	0.030	CBD	CBD	1.2
BEC	50	0.12	0.0042	CBD	CBD	0.12
TMAC	50	0.043	0.024	0.0091	CBD	0.076
TEAC	50	0.043	0.015	CBD	CBD	0.058
CPC	50	0.16	0.058	0.018	0.003	0.24
DTMA	50	0.67	0.22	0.034	CBD	0.92
CTMA	50	0.16	0.24	0.051	CBD	0.45
BZK	50	0.52	0.41	0.19	0.020	1.10
BEC	500	0.12	0.0051	0.0055	CBD	0.13
TMAC	500	0.028	0.027	0.037	0.023	0.12
TEAC	500	0.02	0.024	0.023	0.011	0.078
CPC	500	0.061	0.28	0.39	0.20	0.93
DTMA	500	0.25	0.51	0.38	0.10	1.2
CTMA	500	0.027	0.33	0.55	0.25	1.1
BZK	500	0.38	0.68	0.62	0.32	2.0

Table 6-2 Predicted concentrations of HANs during chlorination (5 mg/L Cl_2) of 400 ug/L of QACs.

CBD – Cannot Be Determined (Because one or more experimental values are below MRL)

Sample	**Bromide Conc. (µg/L)**	**MCAN (µg/L)**	**DCAN (µg/L)**	**BCAN (µg/L)**	**DBAN (µg/L)**	**Total HAN Env. Conc. (µg/L)**
BEC	0	CBD	0.0084	CBD	CBD	0.0084
TMAC	0	CBD	CBD	CBD	CBD	CBD
TEAC	0	CBD	0.011	CBD	CBD	0.011
CPC	0	CBD	0.22	CBD	CBD	0.22
DTMA	0	CBD	0.0082	CBD	CBD	0.0082
CTMA	0	CBD	0.035	CBD	CBD	0.035
BZK	0	CBD	0.10	CBD	CBD	0.10
BEC	50	CBD	0.0093	CBD	CBD	0.0093
TMAC	50	CBD	CBD	CBD	CBD	CBD
TEAC	50	CBD	0.011	CBD	CBD	0.011
CPC	50	CBD	0.21	0.057	0.0047	0.27
DTMA	50	CBD	0.011	CBD	CBD	0.011
CTMA	50	CBD	0.0076	CBD	CBD	0.0076
BZK	50	CBD	0.015	CBD	CBD	0.015
BEC	500	CBD	0.011	CBD	CBD	0.011
TMAC	500	CBD	CBD	0.0061	0.0091	0.015
TEAC	500	CBD	CBD	CBD	CBD	CBD
CPC	500	CBD	0.03	0.09	0.082	0.20
DTMA	500	CBD	0.007	0.0098	0.01	0.027
CTMA	500	CBD	0.0037	0.0014	CBD	0.0051
BZK	500	CBD	0.0086	0.0076	0.0077	0.024

Table 6-3 Predicted concentration of HANs during chloramination (140 mg/L Cl_2) of 400 ug/L of QACs.

CBD – Cannot Be Determined (because one or more experimental values are below MRL)

Sample	Bromide Conc. (µg/L)	MCAN (µg/L)	DCAN (µg/L)	BCAN (µg/L)	DBAN (µg/L)	Total HAN Env. Conc. (µg/L)
BEC	0	0.0013	0.17	CBD	CBD	0.17
TMAC	0	CBD	0.014	CBD	CBD	0.014
TEAC	0	CBD	0.0074	CBD	CBD	0.0074
CPC	0	0.0026	0.55	CBD	CBD	0.55
DTMA	0	CBD	0.015	CBD	CBD	0.015
CTMA	0	CBD	0.0098	CBD	CBD	0.0098
BZK	0	0.0021	0.10	CBD	CBD	0.10
BEC	50	0.0021	0.10	0.0029	CBD	0.10
TMAC	50	CBD	0.015	CBD	CBD	0.015
TEAC	50	CBD	0.0078	CBD	CBD	0.0078
CPC	50	0.0048	0.97	0.0034	CBD	0.97
DTMA	50	CBD	0.011	0.0015	CBD	0.013
CTMA	50	CBD	0.017	0.0018	CBD	0.018
BZK	50	0.0018	0.17	0.012	CBD	0.18
BEC	500	0.0016	0.082	0.030	0.0029	0.11
TMAC	500	CBD	CBD	CBD	CBD	CBD
TEAC	500	CBD	CBD	0.0025	CBD	0.0025
CPC	500	0.0065	0.61	0.036	0.0040	0.65
DTMA	500	CBD	0.0056	0.0057	0.0036	0.015
CTMA	500	0.0011	0.0084	0.0071	0.0040	0.019
BZK	500	0.0039	0.18	0.10	0.041	0.33

Table 6-4 Predicted concentration of NDMA during chloramination (140 mg/L Cl_2) of 400 ug/L of QACs.

CBD – Cannot Be Determined (because one or more experimental values are below MRL)

Sample	**Bromide concentration (µg/L)**	**NDMA (ng/L)**
BEC	0	48
TMAC	0	CBD
TEAC	0	CBD
CPC	0	CBD
DTMA	0	CBD
CTMA	0	CBD
BZK	0	27
BEC	50	48
TMAC	50	CBD
TEAC	50	CBD
CPC	50	CBD
DTMA	50	CBD
CTMA	50	CBD
BZK	50	38
BEC	500	50
TMAC	500	CBD
TEAC	500	CBD
CPC	500	CBD
DTMA	500	CBD
CTMA	500	CBD
BZK	500	23

Based on the predicted concentration of DBPs, it can be seen that the concentration of THMs formed either due to chlorination or chloramination of all of the 7 QACs was relatively low, i.e., less than 2 μg/L in comparison to the drinking water MCL for THMs at 80 μg/L. The same applies for HANs, where the concentration of HANs formed either by chlorination or chloramination of QACs is relatively low at values <1 μg/L compared to the World Health Organization's drinking water guidelines at 20 μg/L for dichloroacetonitrile.

In the case of predicted NDMA concentration, the values were in the range of 22–50 ng/L for BEC and BZK as they were the only compounds that yielded NDMA of the 7 QACs included in this study. Although the predicted NDMA concentration is less than the WHO guideline of 100 ng/L, the values are well above the 1×10^{-6} cancer risk level of 0.7 ng/L for drinking water (EPA IRIS, 1993). Thus, an increase in QAC concentration could lead to a significant NDMA concentrations for water treatment plants practicing chloramination. However, it is also to be noted that the DBP concentration values obtained from this study are due to formation potential conditions, i.e., to achieve maximum DBP yield. Thus, the concentrations overestimate DBP formation and are not indicative of typical disinfection systems.

It should be noted that, even though the predicted concentration of DBPs formed due to reaction between QACs and disinfectants is low compared to the drinking water standards and guidelines, QACs could still contribute to the overall DBPs that are formed from drinking water sources. QACs are one among a plethora of precursors that form DBPs during disinfection of water; thus, it is still important to take note of the low DBP concentrations formed by ubiquitous QACs.

Chapter 7 – Conclusions and Recommendations

Quaternary ammonium compounds are present in a plethora of household cleaning products such as surfactants, laundry detergents, and fabric conditioners, as well as consumer care products such as mouth wash, cosmetics, hand sanitizers, etc. Besides wide use at the household level, QACs are extensively employed in hospitals as disinfectants. In fact, among the products recommended by the CDC to inactivate COVID-19 virus on surfaces, about 216 of these products contain QACs. Eventually, large quantities of these compounds end up in the wastewater stream and a portion of the load escapes into the environment. Pati and Arnold, (2020) found QACs in wastewater effluents at concentrations as high as 8.3 µg/L from the wastewater effluents in Minnesota, USA. BZK was also detected in surface water downstream of wastewater treatment plants in the US at concentrations as high as 36.3 µg/L (C. Zhang et al., 2015b). While these compounds have environmental implications of their own such as promoting antibiotic resistance, disinfection of wastewater (municipal, hospital and industrial) and drinking water using chlorine or chloramine, or combinations of both can give rise to disinfection byproducts such as THMs, HANs and NDMA. It is thus important to understand the potential of these QACs to form disinfection byproducts. Ding et al. (2018) used the caption "Big database" for the formation of DBPs from pharmaceuticals and personal care products. Information gathered from this study could contribute to the "Big database" as QACs are found in many personal care products. However, it is also important to note that in the case of chloramination formation potential, the dose of monochloramine is very high compared to standard disinfection practices by water and wastewater utilities; thus, the DBPs are an overestimation. In the case of chlorination formation potential tests, the chlorine dose is more

consistent with actual disinfection practices; however, DBP yields could also be underestimated as some distribution systems have booster stations that would increase chlorine residual and continue DBP formation.

On basis of the results obtained through this study, the following conclusions are made:

- Quaternary ammonium compounds do have the potential to form THMs upon treatment with chlorine and monochloramine. At a precursor dose of 0.1 mM and a free chlorine dose of 5 mg/L as Cl_2, the yield of TCM was in the order of TMAC<TEAC<BEC<CPC<DTMA<CTMA<BZK for the seven precursors included in this study. At the same precursor dose and a monochloramine dose of 140 mg/L as Cl_2, the yield of TCM was found out to be in the order of TMAC<TEAC<DTMA<CTMA<BZK<BEC<CPC.
- Quaternary ammonium compounds do have the potential to form HANs upon treatment with chlorine and monochloramine. DCAN is the dominant HAN formed upon reaction of 0.1 mM of the precursors with free chlorine (5 mg/L as Cl_2). The amount of DCAN formed for the seven precursors under this study due to chlorination is TMAC<TEAC<DTMA<BEC<CTMA<BZK<CPC. DCAN is again the dominant of the HANs formed upon reaction of 0.1 mM of precursor with monochloramine (140 mg/L as Cl_2). The amount of DCAN formed for the seven precursors under this study due to chloramination is TMAC<TEAC<CTMA<DTMA<CTMA<BZK<BEC<CPC.
- Quaternary ammonium compounds do have the potential to form NDMA upon treatment with monochloramine. It can be seen that, upon reaction of 0.1 mM of the seven precursors with of monochloramine (140 mg/L Cl_2), only BEC and BZK formed relatively significant

amounts of NDMA. The yield of NDMA due to rest of the precursors was below the limit of detection. The average micromolar yield of NDMA from BEC and BZK due to chloramination at 140 mg/L as Cl_2 in the absence of bromide were 721.2 and 250.80 µmol/mol, respectively, thus making the micromolar yield of NDMA by BEC approximately 3 times that of BZK.

- The presence of bromide at low and high concentrations affects the speciation of THMs and HANs upon treatment with chlorine and monochloramine. As the concentration of bromide in the aqueous matrix increases from 0 to 50 µg/L, the speciation of the THMs shifts from chlorinated methanes to mono- and di-bromo substituted methanes. As the concentration of bromide increases from 50 to 500 µg/L, the tri-bromo substituted halomethane (bromoform) is formed.
- Quaternary ammonium compounds upon reaction with aqueous ozone do not yield *N*-nitrosamines or the concentration of any *N*-nitrosamines formed are below the limit of detection. This could be attributed to the degradation of QACs by hydroxyl radicals formed by ozone or selective oxidation of double bonds leading to intermediates that are not part of the aldehyde formation pathway. Additionally, ozone selectively reacts with particular precursors to form NDMA. Thus the structural characteristic of a compound is important for determining if the compound forms DBPs due to ozonation.

Considering the research gaps for the potential of QACs to form disinfection byproducts, the following recommendations for future work are addressed below:

- As there is currently no research done to determine the speciation of HANs and THMs due reaction of QACs with chlorine and chloramine in the presence of iodide, a study on this

matter can be conducted. This would be important for the DBP field because iodinated species of DBPs are relatively more toxic than the chlorinated or brominated analogues.

- As this is the first time the potential of cetylpyridinium chloride to form HANs upon chlorination and chloramination is reported, a full scan GC-MS analysis of the final extract is warranted to determine potential intermediates formed. With the information on the potential intermediates formed post chlorination, a more refined HAN formation pathway can be proposed (L. Linge et al., 2020). In fact, experiments can be conducted to determine the source of nitrogen in HANs i.e., whether nitrogen is sourced from the oxidant (monochloramine) or the precursor (QACs).
- As this is the first time the potential of benzethonium chloride to form NDMA upon chloramination is reported, a full scan LC-MS analysis of the final extract is warranted to determine potential intermediates formed. This information would pave the way to propose probable NDMA formation pathways as a result of the reaction between BEC and monochloramine.
- As formation of DBPs are affected by pH, DBP formation potential experiments can be conducted by varying the pH and measuring DBP yields.
- Due to the likelihood of HAN hydrolysis to haloacetamides, an analysis of the potential of QACs to form haloacetamides upon reaction with chlorine and monochloramine is warranted. Additionally, a full scan GC-MS spectra could help determine the corresponding haloacetamide reaction pathway through the identification of possible intermediate compounds.
- Development of methods for the analysis of individual QACs and analysis of the residual concentration of each precursor is needed to determine the consumption of the same during

the process of disinfection. Doing so will also help us in determining the degradation kinetics of individual QACs during disinfection.

- Once methods for analyzing QACs are developed, environmental samples can be collected and measured for individual QACs.

Appendix

Note – Entries in red are micromolar yield values where the analyte concentration is below the method reporting limit (MRL). NA = Not available; *the value of one duplicate is below the MRL.

Micromolar yield of TCM due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

TCM						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	970	1600	1285	450	35%
0	TMAC	<25	<25	<25	NA	NA
0	TEAC	<25	<25	<25	NA	NA
0	CPC	2300	2600	2450	210	8.7%
0	DTMA	3500	3200	3350	210	6.3%
0	CTMA	4600	4600	4600	0	0%
0	BZK	5200	8200	6700	2100	32%
50	BEC	1100	1100	1100	0	0%
50	TMAC	<25	<25	<25	NA	NA
50	TEAC	<25	<25	<25	NA	NA
50	CPC	1300	960	1130	240	21%

50	DTMA	2400	5000	3700	1800	50%
50	*CTMA	<25	2100	*2100	NA	NA
50	BZK	970	4900	2900	2800	95%
500	BEC	850	1400	1100	390	35%
500	TMAC	<25	<25	<25	NA	NA
500	TEAC	<25	<25	<25	NA	NA
500	*CPC	<25	880	*880	NA	NA
500	DTMA	1600	1200	1400	280	20%
500	*CTMA	<25	360	*360	NA	NA
500	BZK	810	3500	2200	1900	88%

Micromolar yield of DCBM due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

DCBM						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<12	<12	<12	NA	NA
0	TMAC	<12	<12	<12	NA	NA
0	TEAC	<12	<12	<12	NA	NA
0	CPC	<12	<12	<12	NA	NA
0	DTMA	<12	<12	<12	NA	NA

0	CTMA	<12	<12	<12	NA	NA
0	BZK	5.1	240	120	170	136%
50	BEC	32	26	29	4.2	15%
50	TMAC	26	54	40	20	49%
50	TEAC	27	49	38	16	41%
50	CPC	370	230	300	99	33%
50	DTMA	550	1200	880	460	53%
50	CTMA	1400	990	1200	290	24%
50	BZK	460	2900	1700	1700	103%
500	BEC	31	39	35	5.7	16%
500	TMAC	20	71	46	36	79%
500	TEAC	40	81	61	29	48%
500	CPC	1600	1300	1500	210	15%
500	DTMA	2300	1800	2050	350	17%
500	CTMA	2100	1100	1600	710	44%
500	BZK	2500	3100	2800	420	15%

Micromolar yield of DBCM due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

DBCM						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<9.6	<9.6	<9.6	NA	NA
0	TMAC	<9.6	<9.6	<9.6	NA	NA
0	TEAC	<9.6	<9.6	<9.6	NA	NA
0	CPC	<9.6	<9.6	<9.6	NA	NA
0	DTMA	<9.6	<9.6	<9.6	NA	NA
0	CTMA	<9.6	<9.6	<9.6	NA	NA
0	BZK	<9.6	<9.6	<9.6	NA	NA
50	BEC	<9.6	<9.6	<9.6	NA	NA
50	*TMAC	<9.6	15	*15	NA	NA
50	TEAC	<9.6	<9.6	<9.6	NA	NA
50	CPC	58	90	74	23	31%
50	DTMA	65	150	110	60	56%
50	CTMA	160	230	200	49	25%
50	BZK	330	900	620	400	66%
500	BEC	36	23	30	9.2	31%
500	TMAC	23	75	49	37	75%
500	TEAC	31	59	45	20	44%

500	CPC	2000	1200	1600	570	35%
500	DTMA	1300	1100	1200	140	12%
500	CTMA	1500	2700	2100	850	40%
500	BZK	2600	1400	2000	850	42%

Micromolar yield of TBM due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

TBM						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<7.9	<7.9	<7.9	NA	NA
0	TMAC	<7.9	<7.9	<7.9	NA	NA
0	TEAC	<7.9	<7.9	<7.9	NA	NA
0	CPC	<7.9	<7.9	<7.9	NA	NA
0	DTMA	<7.9	<7.9	<7.9	NA	NA
0	CTMA	<7.9	<7.9	<7.9	NA	NA
0	BZK	<7.9	<7.9	<7.9	NA	NA
50	BEC	<7.9	<7.9	<7.9	NA	NA
50	TMAC	<7.9	<7.9	<7.9	NA	NA
50	TEAC	<7.9	<7.9	<7.9	NA	NA
50	CPC	<7.9	<7.9	<7.9	NA	NA

50	DTMA	<7.9	<7.9	<7.9	NA	NA
50	CTMA	<7.9	<7.9	<7.9	NA	NA
50	BZK	18	90	54	51	94%
500	BEC	<7.9	<7.9	<7.9	NA	NA
500	TMAC	10	39	25	21	84%
500	TEAC	10	26	18	11	63%
500	CPC	660	690	680	21	3.1%
500	DTMA	300	240	270	42	16%
500	CTMA	170	1400	790	870	110%
500	BZK	860	860	860	0	0%

Micromolar yield of MCAN due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

MCAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<6.2	<6.2	<6.2	NA	NA
0	TMAC	<6.2	<6.2	<6.2	NA	NA
0	TEAC	<6.2	<6.2	<6.2	NA	NA
0	CPC	<6.2	<6.2	<6.2	NA	NA
0	DTMA	<6.2	<6.2	<6.2	NA	NA

0	CTMA	<6.2	<6.2	<6.2	NA	NA
0	BZK	<6.2	<6.2	<6.2	NA	NA
50	BEC	<6.2	<6.2	<6.2	NA	NA
50	TMAC	<6.2	<6.2	<6.2	NA	NA
50	TEAC	<6.2	<6.2	<6.2	NA	NA
50	CPC	<6.2	<6.2	<6.2	NA	NA
50	DTMA	<6.2	<6.2	<6.2	NA	NA
50	CTMA	<6.2	<6.2	<6.2	NA	NA
50	BZK	<6.2	<6.2	<6.2	NA	NA
500	BEC	<6.2	<6.2	<6.2	NA	NA
500	TMAC	<6.2	<6.2	<6.2	NA	NA
500	TEAC	<6.2	<6.2	<6.2	NA	NA
500	CPC	<6.2	<6.2	<6.2	NA	NA
500	DTMA	<6.2	<6.2	<6.2	NA	NA
500	CTMA	<6.2	<6.2	<6.2	NA	NA
500	BZK	<6.2	<6.2	<6.2	NA	NA

Micromolar yield of DCAN due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

DCAN

Br^- Conc. (µg/L)	Sample	Set 1 µmol/mol	Set 2 µmol/mol	Average µmol/mol	Std. Dev	CV (%)
0	BEC	24	74	49	35	72%
0	TMAC	<14	<14	<14	NA	NA
0	*TEAC	37	<14	*37	NA	NA
0	CPC	930	1000	970	49	5.1%
0	*DTMA	55	<14	*55	NA	NA
0	CTMA	84	210	150	89	61%
0	BZK	660	57	360	430	120%
50	BEC	46	63	55	12	22%
50	TMAC	<14	<14	<14	NA	NA
50	*TEAC	43	<14	*43	NA	NA
50	CPC	900	940	920	28	3.1%
50	DTMA	20	58	39	27	69%
50	*CTMA	56	<14	*56	NA	NA
50	BZK	70	34	52	25	49%
500	BEC	38	89	64	36	57%
500	TMAC	<14	<14	<14	NA	NA
500	TEAC	<14	<14	<14	NA	NA
500	CPC	200	64	130	96	73%
500	*DTMA	42	<14	*42	NA	NA

500	*CTMA	26	<14	*26	NA	NA
500	BZK	47	14	31	23	77%

Micromolar yield of TCAN due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

TCAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<23	<23	<23	NA	NA
0	TMAC	<23	<23	<23	NA	NA
0	TEAC	<23	<23	<23	NA	NA
0	CPC	<23	<23	<23	NA	NA
0	DTMA	<23	<23	<23	NA	NA
0	CTMA	<23	<23	<23	NA	NA
0	BZK	<23	<23	<23	NA	NA
50	BEC	<23	<23	<23	NA	NA
50	TMAC	<23	<23	<23	NA	NA
50	TEAC	<23	<23	<23	NA	NA

50	CPC	<23	<23	<23	NA	NA
50	DTMA	<23	<23	<23	NA	NA
50	CTMA	<23	<23	<23	NA	NA
50	BZK	<23	<23	<23	NA	NA
500	BEC	<23	<23	<23	NA	NA
500	TMAC	<23	<23	<23	NA	NA
500	TEAC	<23	<23	<23	NA	NA
500	CPC	<23	<23	<23	NA	NA
500	DTMA	<23	<23	<23	NA	NA
500	CTMA	<23	<23	<23	NA	NA
500	*BZK	<23	25	*25	NA	NA

Micromolar yield of BDCAN due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

BDCAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<7	<7	<7	NA	NA
0	TMAC	<7	<7	<7	NA	NA
0	TEAC	<7	<7	<7	NA	NA
0	CPC	<7	<7	<7	NA	NA

0	DTMA	<7	<7	<7	NA	NA
0	CTMA	<7	<7	<7	NA	NA
0	BZK	<7	<7	<7	NA	NA
50	BEC	<7	<7	<7	NA	NA
50	TMAC	<7	<7	<7	NA	NA
50	TEAC	<7	<7	<7	NA	NA
50	CPC	<7	<7	<7	NA	NA
50	DTMA	<7	<7	<7	NA	NA
50	CTMA	<7	<7	<7	NA	NA
50	BZK	<7	<7	<7	NA	NA
500	BEC	<7	<7	<7	NA	NA
500	TMAC	<7	<7	<7	NA	NA
500	TEAC	<7	<7	<7	NA	NA
500	CPC	<7	<7	<7	NA	NA
500	DTMA	<7	<7	<7	NA	NA
500	CTMA	<7	<7	<7	NA	NA
500	BZK	<7	<7	<7	NA	NA

Micromolar yield of BCAN due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

BCAN

Br^- Conc. (µg/L)	Sample	Set 1 µmol/mol	Set 2 µmol/mol	Average µmol/mol	Std. Dev	CV (%)
0	BEC	<6.5	<6.5	<6.5	NA	NA
0	TMAC	<6.5	<6.5	<6.5	NA	NA
0	TEAC	<6.5	<6.5	<6.5	NA	NA
0	CPC	<6.5	<6.5	<6.5	NA	NA
0	DTMA	<6.5	<6.5	<6.5	NA	NA
0	CTMA	<6.5	<6.5	<6.5	NA	NA
0	*BZK	<6.5	8	*8	NA	NA
50	BEC	<6.5	<6.5	<6.5	NA	NA
50	TMAC	<6.5	<6.5	<6.5	NA	NA
50	TEAC	<6.5	<6.5	<6.5	NA	NA
50	CPC	260	370	315	78	25%
50	DTMA	<6.5	<6.5	<6.5	NA	NA
50	CTMA	<6.5	<6.5	<6.5	NA	NA
50	*BZK	<6.5	8.6	*8.6	NA	NA
500	BEC	<6.5	<6.5	<6.5	NA	NA
500	*TMAC	<6.5	18	*18	NA	NA
500	TEAC	<6.5	<6.5	<6.5	NA	NA
500	CPC	750	240	495	360	73%
500	*DTMA	80	<6.5	*80	NA	NA

500	CTMA	7.2	7.4	7.3	0.14	1.9%
500	BZK	49	17	33	23	69%

Micromolar yield of MBAN due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

MBAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<19	<19	<19	NA	NA
0	TMAC	<19	<19	<19	NA	NA
0	TEAC	<19	<19	<19	NA	NA
0	CPC	<19	<19	<19	NA	NA
0	DTMA	<19	<19	<19	NA	NA
0	CTMA	<19	<19	<19	NA	NA
0	BZK	<19	<19	<19	NA	NA
50	BEC	<19	<19	<19	NA	NA
50	TMAC	<19	<19	<19	NA	NA
50	TEAC	<19	<19	<19	NA	NA
50	CPC	<19	<19	<19	NA	NA

50	DTMA	<19	<19	<19	NA	NA
50	CTMA	<19	<19	<19	NA	NA
50	BZK	<19	<19	<19	NA	NA
500	BEC	<19	<19	<19	NA	NA
500	TMAC	<19	<19	<19	NA	NA
500	TEAC	<19	<19	<19	NA	NA
500	CPC	<19	<19	<19	NA	NA
500	DTMA	<19	<19	<19	NA	NA
500	CTMA	<19	<19	<19	NA	NA
500	BZK	<19	<19	<19	NA	NA

Micromolar yield of DBAN due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

DBAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<6.7	<6.7	<6.7	NA	NA
0	TMAC	<6.7	<6.7	<6.7	NA	NA
0	TEAC	<6.7	<6.7	<6.7	NA	NA
0	CPC	<6.7	<6.7	<6.7	NA	NA
0	DTMA	<6.7	<6.7	<6.7	NA	NA

0	CTMA	<6.7	<6.7	<6.7	NA	NA
0	*BZK	<6.7	9.6	*9.6	NA	NA
50	BEC	<6.7	<6.7	<6.7	NA	NA
50	TMAC	<6.7	<6.7	<6.7	NA	NA
50	TEAC	<6.7	<6.7	<6.7	NA	NA
50	CPC	16	24	20	5.7	28%
50	DTMA	<6.7	<6.7	<6.7	NA	NA
50	CTMA	<6.7	<6.7	<6.7	NA	NA
50	BZK	<6.7	<6.7	<6.7	NA	NA
500	BEC	<6.7	<6.7	<6.7	NA	NA
500	*TMAC	<6.7	21	*21	NA	NA
500	TEAC	<6.7	<6.7	<6.7	NA	NA
500	CPC	510	190	350	230	65%
500	*DTMA	67	<6.7	*67	NA	NA
500	CTMA	<6.7	<6.7	<6.7	NA	NA
500	BZK	36	16	26	14	54%

Micromolar yield of TCM due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

TCM						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	3400	2800	3100	420	14%
0	TMAC	40	31	35.5	6.4	18%
0	TEAC	51	29	40	16	39%
0	CPC	3200	3000	3100	140	5%
0	DTMA	790	670	730	85	12%
0	CTMA	1400	560	980	590	61%
0	BZK	2000	3400	2700	990	37%
50	BEC	3100	1700	2400	990	41%
50	TMAC	27	48	37.5	15	40%
50	*TEAC	<25	38	*38	NA	NA
50	CPC	3600	3400	3500	140	4%
50	DTMA	700	440	570	180	32%
50	CTMA	1300	830	1065	330	31%
50	BZK	1800	1100	1450	490	34%
500	BEC	1500	1100	1300	280	22%
500	*TMAC	<25	46	*46	NA	NA
500	*TEAC	<25	46	*46	NA	NA

500	CPC	3800	4400	4100	420	10%
500	DTMA	180	130	155	35	23%
500	CTMA	470	400	435	49	11%
500	BZK	760	730	745	21	3%

Micromolar yield of DCBM due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

DCBM						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<12	<12	<12	NA	NA
0	TMAC	<12	<12	<12	NA	NA
0	TEAC	<12	<12	<12	NA	NA
0	CPC	<12	<12	<12	NA	NA
0	DTMA	<12	<12	<12	NA	NA
0	*CTMA	85	<12	*85	NA	NA
0	BZK	<12	<12	<12	NA	NA
50	BEC	400	220	310	130	41%

50	*TMAC	<12	15	*15	NA	NA
50	TEAC	<12	<12	<12	NA	NA
50	CPC	180	120	150	42	28%
50	DTMA	340	250	295	64	22%
50	CTMA	420	230	325	130	41%
50	BZK	300	230	265	49	19%
500	BEC	2300	1700	2000	420	21%
500	TMAC	13	25	19	8.5	45%
500	*TEAC	<12	19	*19	NA	NA
500	CPC	1100	1100	1100	0	0%
500	DTMA	840	650	745	130	18%
500	CTMA	1000	870	935	92	10%
500	BZK	1100	970	1035	92	9%

Micromolar yield of DBCM due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

DBCM						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<9.6	<9.6	<9.6	NA	NA
0	TMAC	<9.6	<9.6	<9.6	NA	NA

0	TEAC	<9.6	<9.6	<9.6	NA	NA
0	CPC	<9.6	<9.6	<9.6	NA	NA
0	DTMA	<9.6	<9.6	<9.6	NA	NA
0	CTMA	<9.6	<9.6	<9.6	NA	NA
0	BZK	<9.6	<9.6	<9.6	NA	NA
50	*BEC	13	<9.6	*13	NA	NA
50	TMAC	<9.6	<9.6	<9.6	NA	NA
50	TEAC	<9.6	<9.6	<9.6	NA	NA
50	CPC	<9.6	<9.6	<9.6	NA	NA
50	DTMA	54	37	45.5	12	26%
50	CTMA	56	26	41	21	52%
50	BZK	26	23	24.5	2.1	9%
500	BEC	840	690	765	110	14%
500	TMAC	16	18	17	1.4	8%
500	TEAC	14	14	14	0	0%
500	CPC	400	420	410	14	3%
500	DTMA	870	750	810	85	10%
500	CTMA	820	650	735	120	16%
500	BZK	730	590	660	99	15%

Micromolar yield of TCM due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

TBM						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<7.9	<7.9	<7.9	NA	NA
0	TMAC	<7.9	<7.9	<7.9	NA	NA
0	TEAC	<7.9	<7.9	<7.9	NA	NA
0	CPC	<7.9	<7.9	<7.9	NA	NA
0	DTMA	<7.9	<7.9	<7.9	NA	NA
0	CTMA	<7.9	<7.9	<7.9	NA	NA
0	BZK	<7.9	<7.9	<7.9	NA	NA
50	BEC	<7.9	<7.9	<7.9	NA	NA
50	TMAC	<7.9	<7.9	<7.9	NA	NA
50	TEAC	<7.9	<7.9	<7.9	NA	NA
50	CPC	<7.9	<7.9	<7.9	NA	NA
50	DTMA	<7.9	<7.9	<7.9	NA	NA
50	CTMA	<7.9	<7.9	<7.9	NA	NA
50	BZK	<7.9	<7.9	<7.9	NA	NA
500	BEC	160	150	155	7.1	5%
500	TMAC	<7.9	<7.9	<7.9	NA	NA
500	*TEAC	9	<7.9	*9	NA	NA

500	CPC	49	70	59.5	15	25%
500	DTMA	160	150	155	7.1	5%
500	CTMA	120	110	115	7.1	6%
500	BZK	110	91	100.5	13	13%

Micromolar yield of MCAN due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

MCAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	*BEC	38	<6.2	*38	NA	NA
0	*TMAC	7.8	<6.2	*7.8	NA	NA
0	TEAC	<6.2	<6.2	<6.2	NA	NA
0	*CPC	59	<6.2	*59	NA	NA
0	*DTMA	6.9	<6.2	*6.9	NA	NA
0	*CTMA	12	<6.2	*12	NA	NA
0	BZK	26	11	18.5	11	57%
50	BEC	33	28	30.5	3.5	12%

50	*TMAC	9.1	<6.2	*9.1	NA	NA
50	*TEAC	6.5	<6.2	*6.5	NA	NA
50	CPC	61	46.	53.5	11	20%
50	*DTMA	8.2	<6.2	*8.2	NA	NA
50	*CTMA	10	<6.2	*10	NA	NA
50	*BZK	27	<6.2	*27	NA	NA
500	BEC	41	6.2	23.6	25	104%
500	*TMAC	6.5	<6.2	65	NA	NA
500	TEAC	<6.2	<6.2	<6.2	NA	NA
500	CPC	36	110	73	52	72%
500	*DTMA	9.1	<6.2	*9.1	NA	NA
500	*CTMA	17	<6.2	*17	NA	NA
500	BZK	39	30	34.5	6.4	18.%

Micromolar yield of DCAN due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

DCAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	690	1300	995	430	43%
0	*TMAC	29	<14	*29	NA	NA

0	*TEAC	19	<14	*19	NA	NA
0	CPC	4300	640	2470	2600	105%
0	DTMA	63	43	53	14	27%
0	CTMA	48	34	41	9.9	24%
0	BZK	620	100	360	370	102%
50	BEC	620	560	590	42	7.2%
50	*TMAC	<14	28	*28	NA	NA
50	*TEAC	<14	23	*23	NA	NA
50	CPC	4700	3900	4300	570	13%
50	DTMA	44	51	47.5	4.9	10%
50	CTMA	53	62	57.5	6.4	11%
50	BZK	650	520	585	92	16%
500	BEC	470	490	480	14	3.0%
500	*TMAC	<14	<14	<14	NA	NA
500	*TEAC	<14	<14	<14	NA	NA
500	CPC	2000	3400	2700	990	37%
500	DTMA	22	17	19.5	3.5	18%
500	CTMA	32	38	35	4.2	12.12%
500	BZK	500	790	645	210	31.79%

Micromolar yield of TCAN due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

TCAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<23	<23	<23	NA	NA
0	TMAC	<23	<23	<23	NA	NA
0	TEAC	<23	<23	<23	NA	NA
0	*CPC	43	<23	*43	NA	NA
0	DTMA	<23	<23	<23	NA	NA
0	CTMA	<23	<23	<23	NA	NA
0	BZK	<23	<23	<23	NA	NA
50	BEC	<23	<23	<23	NA	NA
50	TMAC	<23	<23	<23	NA	NA
50	TEAC	<23	<23	<23	NA	NA
50	CPC	80	83	82	2.1	2.6%
50	DTMA	<23	<23	<23	NA	NA
50	CTMA	<23	<23	<23	NA	NA
50	BZK	<23	<23	<23	NA	NA
500	BEC	<23	<23	<23	NA	NA
500	TMAC	<23	<23	<23	NA	NA
500	TEAC	<23	<23	<23	NA	NA

500	CPC	34	83	59	35	59%
500	DTMA	<23	<23	<23	NA	NA
500	CTMA	<23	<23	<23	NA	NA
500	BZK	<23	<23	<23	NA	NA

Micromolar yield of BDCAN due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

BDCAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<7	<7	<7	NA	NA
0	TMAC	<7	<7	<7	NA	NA
0	TEAC	<7	<7	<7	NA	NA
0	CPC	<7	<7	<7	NA	NA
0	DTMA	<7	<7	<7	NA	NA
0	CTMA	<7	<7	<7	NA	NA
0	BZK	<7	<7	<7	NA	NA
50	BEC	<7	<7	<7	NA	NA
50	TMAC	<7	<7	<7	NA	NA
50	TEAC	<7	<7	<7	NA	NA
50	CPC	<7	<7	<7	NA	NA

50	DTMA	<7	<7	<7	NA	NA
50	CTMA	<7	<7	<7	NA	NA
50	BZK	<7	<7	<7	NA	NA
500	BEC	<7	<7	<7	NA	NA
500	TMAC	<7	<7	<7	NA	NA
500	TEAC	<7	<7	<7	NA	NA
500	CPC	<7	<7	<7	NA	NA
500	DTMA	<7	<7	<7	NA	NA
500	CTMA	<7	<7	<7	NA	NA
500	BZK	<7	<7	<7	NA	NA

Micromolar yield of BCAN due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

BCAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<6.5	<6.5	<6.5	NA	NA
0	TMAC	<6.5	<6.5	<6.5	NA	NA
0	TEAC	<6.5	<6.5	<6.5	NA	NA
0	CPC	<6.5	<6.5	<6.5	NA	NA
0	DTMA	<6.5	<6.5	<6.5	NA	NA

0	CTMA	<6.5	<6.5	<6.5	NA	NA
0	BZK	<6.5	<6.5	<6.5	NA	NA
50	BEC	20	22	21	1.4	6.7%
50	*TMAC	<6.5	7.3	*7.3	NA	NA
50	TEAC	<6.5	<6.5	<6.5	NA	NA
50	CPC	17	20	18.5	2.1	11%
50	*DTMA	<6.5	9.2	*9.2	NA	NA
50	CTMA	7.0	8.0	7.5	0.71	9.4%
50	BZK	51	50	51	0.71	1.4%
500	BEC	200	230	215	21	9.9%
500	*TMAC	<6.5	7.3	*7.3	NA	NA
500	*TEAC	<6.5	8.6	*8.6	NA	NA
500	CPC	210	190	200	14	7.1%
500	DTMA	21	28	24.5	4.9	20%
500	CTMA	28	46	37	13	34.%
500	BZK	250	640	445	280	62%

Micromolar yield of MBAN due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

MBAN

Br^- Conc. (µg/L)	Sample	Set 1 µmol/mol	Set 2 µmol/mol	Average µmol/mol	Std. Dev	CV (%)
0	BEC	<19	<19	<19	NA	NA
0	TMAC	<19	<19	<19	NA	NA
0	TEAC	<19	<19	<19	NA	NA
0	CPC	<19	<19	<19	NA	NA
0	DTMA	<19	<19	<19	NA	NA
0	CTMA	<19	<19	<19	NA	NA
0	BZK	<19	<19	<19	NA	NA
50	BEC	<19	<19	<19	NA	NA
50	TMAC	<19	<19	<19	NA	NA
50	TEAC	<19	<19	<19	NA	NA
50	CPC	<19	<19	<19	NA	NA
50	DTMA	<19	<19	<19	NA	NA
50	CTMA	<19	<19	<19	NA	NA
50	BZK	<19	<19	<19	NA	NA
500	BEC	<19	<19	<19	NA	NA
500	TMAC	<19	<19	<19	NA	NA
500	TEAC	<19	<19	<19	NA	NA
500	CPC	<19	<19	<19	NA	NA
500	DTMA	<19	<19	<19	NA	NA

500	CTMA	<19	<19	<19	NA	NA
500	BZK	<19	<19	<19	NA	NA

Micromolar yield of DBAN due to chloramination (140 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

DBAN						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<6.7	<6.7	<6.7	NA	NA
0	TMAC	<6.7	<6.7	<6.7	NA	NA
0	TEAC	<6.7	<6.7	<6.7	NA	NA
0	CPC	<6.7	<6.7	<6.7	NA	NA
0	DTMA	<6.7	<6.7	<6.7	NA	NA
0	CTMA	<6.7	<6.7	<6.7	NA	NA
0	BZK	<6.7	<6.7	<6.7	NA	NA
50	BEC	<6.7	<6.7	<6.7	NA	NA
50	TMAC	<6.7	<6.7	<6.7	NA	NA
50	TEAC	<6.7	<6.7	<6.7	NA	NA
50	CPC	<6.7	<6.7	<6.7	NA	NA
50	DTMA	<6.7	<6.7	<6.7	NA	NA
50	CTMA	<6.7	<6.7	<6.7	NA	NA

50	BZK	<6.7	<6.7	<6.7	NA	NA
500	BEC	11	22	17	7.8	47%
500	TMAC	<6.7	<6.7	<6.7	NA	NA
500	*TEAC	<6.7	7.3	NA	NA	NA
500	CPC	16	18	17	1.4	8.3%
500	DTMA	10	14	12	2.8	24%
500	CTMA	11	21	16	7.1	44%
500	BZK	58	220	139	110	82%

Micromolar yield of NDMA due to chlorination (5 mg/L Cl_2) of 0.1 mM QAC with and without bromide.

NDMA						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	<160	<160	<160	NA	NA
0	TMAC	<160	<160	<160	NA	NA
0	TEAC	<160	<160	<160	NA	NA
0	CPC	<160	<160	<160	NA	NA
0	DTMA	<160	<160	<160	NA	NA
0	CTMA	<160	<160	<160	NA	NA
0	BZK	<160	<160	<160	NA	NA

50	BEC	<160	<160	<160	NA	NA
50	TMAC	<160	<160	<160	NA	NA
50	TEAC	<160	<160	<160	NA	NA
50	CPC	<160	<160	<160	NA	NA
50	DTMA	<160	<160	<160	NA	NA
50	CTMA	<160	<160	<160	NA	NA
50	BZK	<160	<160	<160	NA	NA
500	BEC	<160	<160	<160	NA	NA
500	TMAC	<160	<160	<160	NA	NA
500	TEAC	<160	<160	<160	NA	NA
500	CPC	<160	<160	<160	NA	NA
500	DTMA	<160	<160	<160	NA	NA
500	CTMA	<160	<160	<160	NA	NA
500	BZK	<160	<160	<160	NA	NA

Micromolar yield of NDMA due to chloramination (140 mg/L Cl2) of 0.1 mM QAC with and without bromide.

NDMA						
Br^- Conc. (µg/L)	**Sample**	**Set 1 µmol/mol**	**Set 2 µmol/mol**	**Average µmol/mol**	**Std. Dev**	**CV (%)**
0	BEC	890	550	720	240	33%

0	TMAC	<160	<160	<160	NA	NA
0	TEAC	<160	<160	<160	NA	NA
0	CPC	<160	<160	<160	NA	NA
0	DTMA	<160	<160	<160	NA	NA
0	CTMA	<160	<160	<160	NA	NA
0	*BZK	410	<160	*410	NA	NA
50	BEC	800	650	725	110	15%
50	TMAC	<160	<160	<160	NA	NA
50	TEAC	<160	<160	<160	NA	NA
50	CPC	<160	<160	<160	NA	NA
50	DTMA	<160	<160	<160	NA	NA
50	CTMA	<160	<160	<160	NA	NA
50	BZK	370	330	350	28	8%
500	BEC	730	770	750	28	4%
500	TMAC	<160	<160	<160	NA	NA
500	TEAC	<160	<160	<160	NA	NA
500	CPC	<160	<160	<160	NA	NA
500	DTMA	<160	<160	<160	NA	NA
500	CTMA	<160	<160	<160	NA	NA
500	BZK	230	190	210	28	13%

References

Adeleke, G.E., Adaramoye, O.A., 2017. Modulatory role of betulinic acid in N - nitrosodimethylamine-induced hepatorenal toxicity in male Wistar rats. Human and Experimental Toxicology 36, 734–743. https://doi.org/10.1177/0960327116661399

Aieta, E.M., Berg, J.D., 1986. A review of chlorine dioxide in drinking water treatment. Journal / American Water Works Association 78, 62–72. https://doi.org/10.1002/j.1551-8833.1986.tb05766.x

Alsulaili, A.D., 2009. Impact of bromide, NOM, and prechlorination on haloamine formation, speciation, and decay during chloramination.

Bader, H., Hoigné, J., 1981. Determination of ozone in water by the indigo method, Water Res. 15, 449-456. doi: https://doi.org/10.1016/0043-1354(81)90054-3.

Baird, R. B., Eaton, A. D., Rice, E. W., & Bridgewater, L., 2017. Standard methods for the examination of water and wastewater (Vol. 23). Washington, DC: American Public Health Association.

Baribeau, H., Krasner, S.W., Chinn, R., Singer, P.C., 2005. Impact of biomass on the stability of HAAs and THMs in a simulated distribution system. Journal - American Water Works Association 97, 69–81. https://doi.org/10.1002/J.1551-8833.2005.TB10826.X

Beita-Sandí, W., Erdem, C.U., Karanfil, T., 2020. Effect of bromide on NDMA formation during chloramination of model precursor compounds and natural waters. Water Research 170, 115323. https://doi.org/10.1016/J.WATRES.2019.115323

Bergé, A., Wiest, L., Baudot, R., Giroud, B., Vulliet, E., 2018. Occurrence of multi-class surfactants in urban wastewater: contribution of a healthcare facility to the pollution transported into the sewerage system. Environmental Science and Pollution Research 25, 9219–9229. https://doi.org/10.1007/s11356-017-0470-8

Bi, X., Xu, B., Lin, Y.-L., Hu, C.-Y., Ye, T., Qin, C., 2013. Monochloramination of Oxytetracycline: Kinetics, Mechanisms, Pathways, and Disinfection By-Products Formation. CLEAN – Soil, Air, Water 41, 969–975. https://doi.org/10.1002/CLEN.201200344

Bichsel, Y., von Gunten, U., 2000. Formation of iodo-trihalomethanes during disinfection and oxidation of iodide-containing waters. Environmental Science and Technology 34, 2784–2791. https://doi.org/10.1021/es9914590

Blatchley, E.R., Margetas, D., Duggirala, R., 2003. Copper catalysis in chloroform formation during water chlorination. Water Research 37, 4385–4394. https://doi.org/10.1016/S0043-1354(03)00404-4

Boethling, R.S., 1984. Environmental fate and toxicity in wastewater treatment of quaternary ammonium surfactants. Water Research 18, 1061–1076. https://doi.org/10.1016/0043-1354(84)90220-3

Bond, T., Huang, J., Graham, N.J.D., Templeton, M.R., 2014. Examining the interrelationship between DOC, bromide and chlorine dose on DBP formation in drinking water — A case study. Science of The Total Environment 470–471, 469–479. https://doi.org/10.1016/J.SCITOTENV.2013.09.106

Bond, T., Huang, J., Templeton, M.R., Graham, N., 2011. Occurrence and control of nitrogenous disinfection by-products in drinking water - A review. Water Research 45, 4341–4354. https://doi.org/10.1016/j.watres.2011.05.034

Bougeard, C.M.M., Goslan, E.H., Jefferson, B., Parsons, S.A., 2010. Comparison of the disinfection by-product formation potential of treated waters exposed to chlorine and monochloramine. Water Research 44, 729–740. https://doi.org/10.1016/j.wateres.2009.10.008

Carbajo, J.B., Petre, A.L., Rosal, R., Berná, A., Letón, P., García-Calvo, E., Perdigón-Melón, J.A., 2016. Ozonation as pre-treatment of activated sludge process of a wastewater containing benzalkonium chloride and NiO nanoparticles. Chemical Engineering Journal 283, 740–749. https://doi.org/10.1016/j.cej.2015.08.001

Carter, R.A.A., Liew, D.S., West, N., Heitz, A., Joll, C.A., 2019. Simultaneous analysis of haloacetonitriles, haloacetamides and halonitromethanes in chlorinated waters by gas chromatography-mass spectrometry. Chemosphere 220, 314–323. https://doi.org/10.1016/j.chemosphere.2018.12.069

Chakrabartty, S. K., & Kretschmer, H. O., 1974. Sodium hypochlorite as a selective oxidant for organic compounds. Journal of the Chemical Society, Perkin Transactions 1, 222-228.

Chang, E.E., Lin, Y.P., Chiang, P.C., 2001. Effects of bromide on the formation of THMs and HAAs. Chemosphere 43, 1029–1034. https://doi.org/10.1016/S0045-6535(00)00210-1

Chang, H., Chen, C., Wang, G., 2011. Identification of potential nitrogenous organic precursors for C-, N-DBPs and characterization of their DBPs formation. Water Research 45, 3753–3764. https://doi.org/10.1016/J.WATRES.2011.04.027

Chang, H.H., Wang, G.S., 2011. Correlations between surrogate nitrogenous organic precursors and C-, N-DBP formation. Water Science and Technology 64, 2395–2403. https://doi.org/10.2166/WST.2011.823

Charrois, J. W., Boyd, J. M., Froese, K. L., & Hrudey, S. E., 2007. Occurrence of N-nitrosamines in Alberta public drinking-water distribution systems. Journal of Environmental Engineering and Science, 6(1), 103-114.

Cheema, W.A., Andersen, H.R., Kaarsholm, K.M.S., 2018. Improved DBP elimination from swimming pool water by continuous combined UV and ozone treatment. Water Research 147, 214–222. https://doi.org/10.1016/J.WATRES.2018.09.030

Chen, B., 2011. Hydrolytic Stabilities of Halogenated Disinfection Byproducts: Review and Rate Constant Quantitative Structure–Property Relationship Analysis. https://home.liebertpub.com/ees 28, 385–394. https://doi.org/10.1089/EES.2010.0196

Chen, W., Liu, Z., Tao, H., Xu, H., Gu, Y., Chen, Z., Yu, J., 2017. Factors affecting the formation of nitrogenous disinfection by-products during chlorination of aspartic acid in

drinking water. Science of the Total Environment 575, 519–524. https://doi.org/10.1016/j.scitotenv.2016.09.010

Chen, W.-H., Young, T.M., 2008. NDMA formation during chlorination and chloramination of aqueous diuron Solutions. Environmental Science and Technology 42, 1072–1077. https://doi.org/10.1021/ES072044E

Cho, M.I.N., Chung, H., Yoon, J., 2003. Quantitative Evaluation of the Synergistic Sequential Inactivation of *Bacillus subtilis* spores with ozone followed by chlorine 37, 2134–2138.

Choi, J., Valentine, R.L., 2002. Formation of N-nitrosodimethylamine (NDMA) from reaction of monochloramine: A new disinfection by-product. Water Research 36, 817–824. https://doi.org/10.1016/S0043-1354(01)00303-7

Chowdhury, S., 2018. Occurrences and changes of disinfection by-products in small water supply systems. Environmental Monitoring and Assessment 190. https://doi.org/10.1007/s10661-017-6410-8

Chu, W., Li, D., Gao, N., Yin, D., Zhang, Y., & Zhu, Y., 2015. Comparison of free amino acids and short oligopeptides for the formation of trihalomethanes and haloacetonitriles during chlorination: Effect of peptide bond and pre-oxidation. Chemical Engineering Journal, 281, 623-631.

Chu, W., Yao, D., Deng, Y., Sui, M., Gao, N., 2017. Production of trihalomethanes, haloacetaldehydes and haloacetonitriles during chlorination of microcystin-LR and

impacts of pre-oxidation on their formation. Journal of Hazardous Materials 327, 153–160. https://doi.org/10.1016/J.JHAZMAT.2016.12.058

Cimetiere, N., Dossier-Berne, F., de Laat, J., 2010. Effect of some parameters on the formation of chloroform during chloramination of aqueous solutions of resorcinol. Water Research 44, 4497–4504. https://doi.org/10.1016/J.WATRES.2010.06.010

Clara, M., Scharf, S., Scheffknecht, C., Gans, O., 2007. Occurrence of selected surfactants in untreated and treated sewage. Water Research 41, 4339–4348. https://doi.org/10.1016/j.watres.2007.06.027

C. Poully, J., P. Schermann, J., N. Nieuwjaer, F. Lecomte, G. Grégoire, C. Desfrançois, A. Garcia, G., L. Nahon, D. Nandi, L. Poisson, M. Hochlaf, 2010. Photoionization of 2-pyridone and 2-hydroxypyridine. Physical Chemistry Chemical Physics 12, 3566–3572. https://doi.org/10.1039/B923630A

Croghan, C and P P. Egeghy. Methods of dealing with values below the limit of detection using sas. Presented at Southeastern SAS User Group, St. Petersburg, FL, September 22-24, 2003.

Cuthbertson, A.A., Kimura, S.Y., Liberatore, H.K., Summers, R.S., Knappe, D.R., Stanford, B.D., Maness, J.C., Mulhern, R.E., Selbes, M. and Richardson, S.D., 2019. Does granular activated carbon with chlorination produce safer drinking water? From disinfection byproducts and total organic halogen to calculated toxicity. Environmental science & technology, 53(10), 5987-5999.

Deborde, M., von Gunten, U., 2008. Reactions of chlorine with inorganic and organic compounds during water treatment—Kinetics and mechanisms: A critical review. Water Research 42, 13–51. https://doi.org/10.1016/J.WATRES.2007.07.025

Deng, L., Huang, C.H., Wang, Y.L., 2014. Effects of Combined UV and Chlorine Treatment on the Formation of Trichloronitromethane from Amine Precursors. Environmental Science and Technology 48, 2697–2705. https://doi.org/10.1021/ES404116N

De Vera, G. A., Stalter, D., Gernjak, W., Weinberg, H. S., Keller, J., & Farré, M. J., 2015. Towards reducing DBP formation potential of drinking water by favouring direct ozone over hydroxyl radical reactions during ozonation. Water Research, 87, 49-58.

Ding, S., Chu, W., Bond, T., Wang, Q., Gao, N., Xu, B., Du, E., 2018. Formation and estimated toxicity of trihalomethanes, haloacetonitriles, and haloacetamides from the chlor(am)ination of acetaminophen. Journal of Hazardous Materials 341, 112–119. https://doi.org/10.1016/J.JHAZMAT.2017.07.049

Ding, W.H., Liao, Y.H., 2000. Determination of Alkylbenzyldimethylammonium Chlorides in River Water and Sewage Effluent by Solid-Phase Extraction and Gas Chromatography/Mass Spectrometry. Analytical Chemistry 73, 36–40. https://doi.org/10.1021/AC000655I

Doederer, K., Gernjak, W., Weinberg, H.S., Farré, M.J., 2014. Factors affecting the formation of disinfection by-products during chlorination and chloramination of secondary effluent for the production of high quality recycled water. Water Research 48, 218–228. https://doi.org/10.1016/j.watres.2013.09.034

Dojlido, J., Zbieć, E., & Świetlik, R., 1999. Formation of the haloacetic acids during ozonation and chlorination of water in Warsaw waterworks (Poland). Water research, 33(14), 3111-3118.

Driedger, A.M., Rennecker, J.L., Mariñas, B.J., 2000. Sequential inactivation of Cryptosporidium parvum oocysts with ozone and free chlorine. Water Research 34, 3591–3597. https://doi.org/10.1016/S0043-1354(00)00097-X

Echigo, S., Itoh, S., Natsui, T., Araki, T., Ando, R., 2004. Contribution of brominated organic disinfection by-products to the mutagenicity of drinking water. Water Science and Technology 50, 321–328. https://doi.org/10.2166/wst.2004.0344

EPA Integrated Risk Information System (IRIS). 1993. "N-Nitrosodimethylamine; CASRN 62-75-9." www.epa.gov/iris/subst/0045.htm.

Ersan, M.S., Liu, C., Amy, G., Karanfil, T., 2019a. The interplay between natural organic matter and bromide on bromine substitution. Science of the Total Environment 646, 1172–1181. https://doi.org/10.1016/j.scitotenv.2018.07.384

Ersan, M.S., Liu, C., Amy, G., Plewa, M.J., Wagner, E.D., Karanfil, T., 2019b. Chloramination of iodide-containing waters: Formation of iodinated disinfection byproducts and toxicity correlation with total organic halides of treated waters. Science of the Total Environment 697, 134142. https://doi.org/10.1016/j.scitotenv.2019.134142

European Commission, 1998. Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption. Official journal of the European communities, 41, 32-54.

Fang, J., Liu, H., Shang, C., Zeng, M., Ni, M., Liu, W., 2014. E. coli and bacteriophage MS2 disinfection by UV, ozone and the combined UV and ozone processes. Frontiers of Environmental Science and Engineering 8, 547–552. https://doi.org/10.1007/s11783-013-0620-2

Farrell, C., Hassard, F., Jefferson, B., Leziart, T., Nocker, A., Jarvis, P., 2018. Turbidity composition and the relationship with microbial attachment and UV inactivation efficacy. Science of the Total Environment 624, 638–647. https://doi.org/10.1016/j.scitotenv.2017.12.173

Ferk, F., Mišík, M., Hoelzl, C., Uhl, M., Fuerhacker, M., Grillitsch, B., Parzefall, W., Nersesyan, A., Mičieta, K., Grummt, T., Ehrlich, V., Knasmüller, S., 2007. Benzalkonium chloride (BAC) and dimethyldioctadecyl-ammonium bromide (DDAB), two common quaternary ammonium compounds, cause genotoxic effects in mammalian and plant cells at environmentally relevant concentrations. Mutagenesis 22, 363–370. https://doi.org/10.1093/mutage/gem027

Fiddler, W., Pensabene, J.W., Doerr, R.C., Dooley, C.J., 1977. The presence of dimethyl- and diethyl-nitrosamines in deionized water. Food and Cosmetics Toxicology 15, 441–443. https://doi.org/10.1016/S0015-6264(77)80010-2

Gan, W., Huang, H., Yang, X., Peng, Z., Chen, G., 2016. Emerging investigators series: Disinfection by-products in mixed chlorine dioxide and chlorine water treatment. Environmental Science: Water Research and Technology 2, 838–847. https://doi.org/10.1039/c6ew00061d

Garcia-Villanova, R.J., Garcia, C., Alfonso Gomez, J., Paz Garcia, M., Ardanuy, R., 1997. Formation, evolution and modeling of trihalomethanes in the drinking water of a town: I. At the municipal treatment utilities. Water Research 31, 1299–1308. https://doi.org/10.1016/S0043-1354(96)00335-1

Gerrity, D., Mayer, B., Ryu, H., Crittenden, J., & Abbaszadegan, M., 2009. A comparison of pilot-scale photocatalysis and enhanced coagulation for disinfection byproduct mitigation. Water research, 43(6), 1597-1610.

Geter, D.R., Chang, L.W., Hanley, N.M., Ross, M.K., Pegram, R.A., DeAngelo, A.B., 2004. Analysis of in vivo and in vitro DNA strand breaks from trihalomethane exposure. Journal of Carcinogenesis 3, 1–8. https://doi.org/10.1186/1477-3163-3-2

Glezer, V., Harris, B., Tal, N., Iosefzon, B., Lev, O., 1999. Hydrolysis of haloacetonitriles: Linear free energy relationship, kinetics and products. Water Research 33, 1938–1948. https://doi.org/10.1016/S0043-1354(98)00361-3

Goslan, E. H., Krasner, S. W., Bower, M., Rocks, S. A., Holmes, P., Levy, L. S., & Parsons, S. A., 2009. A comparison of disinfection by-products found in chlorinated and chloraminated drinking waters in Scotland. Water research, 43(18), 4698-4706.

G. Pati, S., A. Arnold, W., 2020. Comprehensive screening of quaternary ammonium surfactants and ionic liquids in wastewater effluents and lake sediments. Environmental Science: Processes & Impacts 22, 430–441. https://doi.org/10.1039/C9EM00554D

Gunten, U. von, Salhi, E., Schmidt, C.K., Arnold, W.A., 2010. Kinetics and mechanisms of N -nitrosodimethylamine formation upon ozonation of N, N -dimethylsulfamide-containing waters: Bromide catalysis. Environmental Science and Technology 44, 5762–5768. https://doi.org/10.1021/es1011862

Hanigan, D., Truong, L., Simonich, M., Tanguay, R., Westerhoff, P., 2017. Zebrafish embryo toxicity of 15 chlorinated, brominated, and iodinated disinfection by-products. Journal of Environmental Sciences 58, 302–310. https://doi.org/10.1016/j.jes.2017.05.008

Harrison, K. R., Kappell, A. D., & McNamara, P. J. (2020). Benzalkonium chloride alters phenotypic and genotypic antibiotic resistance profiles in a source water used for drinking water treatment. Environmental Pollution, 257, 113472.

Herrera-Herrera, A. V., Asensio-Ramos, M., Hernández-Borges, J., Rodríguez-Delgado, M.Á., 2010. Dispersive liquid-liquid microextraction for determination of organic analytes. TrAC - Trends in Analytical Chemistry 29, 728–751. https://doi.org/10.1016/j.trac.2010.03.016

Gallard. H, Gunten, U. V., 2002. Chlorination of Phenols: Kinetics and Formation of Chloroform. Environmental Science and Technology 36, 884–890. https://doi.org/10.1021/ES010076A

Holady, J.C., Trenholm, R.A., Snyder, S.A., 2012. Use of automated solid-phase extraction and GC-MS/MS to evaluate nitrosamines in water matrices. https://doi.org/10.2/JQUERY.MIN.JS

Hong, H., Xiong, Y., Ruan, M., Liao, F., Lin, H., Liang, Y., 2013. Factors affecting THMs, HAAs and HNMs formation of Jin Lan Reservoir water exposed to chlorine and monochloramine. Science of The Total Environment 444, 196–204. https://doi.org/10.1016/J.SCITOTENV.2012.11.086

Hora, P.I., Pati, S.G., McNamara, P.J., Arnold, W.A., 2020. Increased Use of Quaternary Ammonium Compounds during the SARS-CoV-2 Pandemic and Beyond: Consideration of Environmental Implications. Environmental Science & Technology Letters 7, 622–631. https://doi.org/10.1021/ACS.ESTLETT.0C00437

Hua, G., Reckhow, D.A., 2007. Comparison of disinfection byproduct formation from chlorine and alternative disinfectants. Water Research 41, 1667–1678. https://doi.org/10.1016/j.watres.2007.01.032

Huang, H., Wu, Q.Y., Hu, H.Y., Mitch, W.A., 2012. Dichloroacetonitrile and dichloroacetamide can form independently during chlorination and chloramination of drinking waters, model organic matters, and wastewater effluents. Environmental Science and Technology 46, 10624–10631. https://doi.org/10.1021/es3025808

Huang, H., Wu, Q.Y., Tang, X., Jiang, R., Hu, H.Y., 2016. Formation of haloacetonitriles and haloacetamides and their precursors during chlorination of secondary effluents. Chemosphere 144, 297–303. https://doi.org/10.1016/J.CHEMOSPHERE.2015.08.082

Ikehata, K., El-Din, M.G., 2004. Degradation of recalcitrant surfactants in wastewater by ozonation and advanced oxidation processes: A review. Ozone: Science and Engineering 26, 327–343. https://doi.org/10.1080/01919510490482160

Jia, A., Wu, C., Duan, Y., 2016. Precursors and factors affecting formation of haloacetonitriles and chloropicrin during chlor(am)ination of nitrogenous organic compounds in drinking water. Journal of Hazardous Materials 308, 411–418. https://doi.org/10.1016/j.jhazmat.2016.01.037

Jolis, D., Lam, C., Pitt, P., 2001. Particle Effects on Ultraviolet Disinfection of Coliform Bacteria in Recycled Water. Water Environment Research 73, 233–236. https://doi.org/10.2175/106143001x139218

Jung, Y.J., Oh, B.S., Kang, J.W., 2008. Synergistic effect of sequential or combined use of ozone and UV radiation for the disinfection of Bacillus subtilis spores. Water Research 42, 1613–1621. https://doi.org/10.1016/j.watres.2007.10.008

Karl, G. L., Gwy-Am, S., Gaetan, F., William, C., and Sobsey, M.D., 2002. UV Disinfection of Giardia lamblia Cysts in Water. Environmental Science and Technology 36, 2519–2522. https://doi.org/10.1021/ES0113403

Katritzky, A.R., Khan, G.R., Leahy, D.E., de Rosa, M., 1984. 44(a), 44; Chem. Abstr. 1968, 69, 27253w. (5) (a) De Rosa, M. J. Chem. Soc., Chem. Commun 49, 4142.

Kemper, J.M., Walse, S.S., Mitch, W.A., 2010. Quaternary Amines as Nitrosamine Precursors: A Role for Consumer Products? Environmental Science and Technology 44, 1224–1231. https://doi.org/10.1021/ES902840H

Khan, A.H., Libby, M., Winnick, D., Palmer, J., Sumarah, M., Ray, M.B., Macfie, S.M., 2018. Uptake and phytotoxic effect of benzalkonium chlorides in Lepidium sativum and Lactuca sativa. Journal of Environmental Management 206, 490–497. https://doi.org/10.1016/j.jenvman.2017.10.077

Kim, M., Weigand, M. R., Oh, S., Hatt, J. K., Krishnan, R., Tezel, U., ... & Konstantinidis, K. T., 2018. Widely used benzalkonium chloride disinfectants can promote antibiotic resistance. Applied and environmental microbiology, 84(17), e01201-18.

Kimoto, W.I., Dooley, C.J., Carré, J., Fiddler, W., 1980. Role of strong ion exchange resins in nitrosamine formation in water. Water Research 14, 869–876. https://doi.org/10.1016/0043-1354(80)90267-5

Kimura, S.Y., Dissertation, H., 2013. Formation of (halo)acetamides and (halo)acetonitriles from the reaction of monochloramine and (halo)acetaldehydes in water.

Krasner, S.W., Mitch, W.A., McCurry, D.L., Hanigan, D., Westerhoff, P., 2013. Formation, precursors, control, and occurrence of nitrosamines in drinking water: A review. Water Research 47, 4433–4450. https://doi.org/10.1016/J.WATRES.2013.04.050

Kreuzinger, N., Fuerhacker, M., Scharf, S., Uhl, M., Gans, O., Grillitsch, B., 2007. Methodological approach towards the environmental significance of uncharacterized

substances - quaternary ammonium compounds as an example. Desalination 215, 209–222. https://doi.org/10.1016/j.desal.2006.10.036

Kril, A., Georgieva, A., Nikolov, B., Pepovich, R., Hristov, K., Stoimenov, G., Nikolova, E., Petrova, R., Ananiev, J., Manov, V., 2018. In ovo hepatocarcinogenicity of N-nitrosodimethylamine and N-nitrosodimethylamine in White Leghorn chickens. Journal of the Hellenic Veterinary Medical Society 69, 1117–1124. https://doi.org/10.12681/JHVMS.18884

Kümmerer, K., Eitel, A., Braun, U., Hubner, P., Daschner, F., Mascart, G., Milandri, M., Reinthaler, F., Verhoef, J., 1997. Analysis of benzalkonium chloride in the effluent from European hospitals by solid-phase extraction and high-performance liquid chromatography with post-column ion-pairing and fluorescence detection. Journal of Chromatography A 774, 281–286. https://doi.org/10.1016/S0021-9673(97)00242-2

Lasek, F., Karpel Vel Leitner, N., Rauwel, G., Blanchier, L., Castel, O., Ayraud-Thevenot, S., Deborde, M., 2019. Discharge of biocidal products from healthcare activities into a sewage system—a case study at a French university hospital. Environmental Science and Pollution Research 26, 4938–4951. https://doi.org/10.1007/s11356-018-3882-1

Lavorgna, M., Russo, C., D'Abrosca, B., Parrella, A., Isidori, M., 2016. Toxicity and genotoxicity of the quaternary ammonium compound benzalkonium chloride (BAC) using Daphnia magna and Ceriodaphnia dubia as model systems. Environmental Pollution 210, 34–39. https://doi.org/10.1016/j.envpol.2015.11.042

Lee, C., Schmidt, C., Yoon, J., von Gunten, U., 2007. Oxidation of N-nitrosodimethylamine (NDMA) precursors with ozone and chlorine dioxide: Kinetics and effect on NDMA formation potential. Environmental Science and Technology 41, 2056–2063. https://doi.org/10.1021/ES062484Q/SUPPL_FILE/ES062484QSI20070208_074829.PDF

Lee, W., Westerhoff, P., 2009. Formation of organic chloramines during water disinfection – chlorination versus chloramination. Water Research 43, 2233–2239. https://doi.org/10.1016/J.WATRES.2009.02.009

Li, C., Gao, N., Chu, W., Bond, T., Wei, X., 2017. Comparison of THMs and HANs formation potential from the chlorination of free and combined histidine and glycine. Chemical Engineering Journal 307, 487–495. https://doi.org/10.1016/J.CEJ.2016.08.110

Liberti, L., Notarnicola, M., Petruzzelli, D., 2003. Advanced treatment for municipal wastewater reuse in agriculture. UV disinfection: Parasite removal and by-product formation. Desalination 152, 315–324. https://doi.org/10.1016/S0011-9164(02)01079-2

Lilly, P.D., Ross, T.M., Pegram, R.A., 1997. Trihalomethane comparative toxicity: Acute renal and hepatic toxicity of chloroform and bromodichloromethane following aqueous gavage. Fundamental and Applied Toxicology 40, 101–110. https://doi.org/10.1006/faat.1997.2372

Lin, E.L.C., Daniel, F.B., Herren-Freund, S.L., Pereira, M.A., 1986. Haloacetonitriles: Metabolism, Genotoxicity, and Tumor-Initiating Activity. Environmental Health Perspectives 69, 67–71.

Lin Wang, Lingliang Long, Liping Zhou, Yanjun Wu, Chi Zhang, Zhixiang Han, Junli Wang, Zulin Da, 2014. A ratiometric fluorescent probe for highly selective and sensitive detection of hypochlorite based on the oxidation of N -alkylpyridinium. RSC Advances 4, 59535–59540. https://doi.org/10.1039/C4RA10633D

Lipscomb, J.C., El-Demerdash, E., Ahmed, A.E., 2008. <Emphasis Type="Bold">Haloacetonitriles: Metabolism and Toxicity</Emphasis>. Reviews of Environmental Contamination and Toxicology 198, 169–200. https://doi.org/10.1007/978-0-387-09647-6_5

Liu, R., Tian, C., Hu, C., Qi, Z., Liu, H., Qu, J., 2018. Effects of bromide on the formation and transformation of disinfection by-products during chlorination and chloramination. Science of The Total Environment 625, 252–261. https://doi.org/10.1016/J.SCITOTENV.2017.12.253

Liu, S., Li, Z., Dong, H., Goodman, B.A., Qiang, Z., 2017. Formation of iodo-trihalomethanes, iodo-acetic acids, and iodo-acetamides during chloramination of iodide-containing waters: Factors influencing formation and reaction pathways. Journal of Hazardous Materials 321, 28–36. https://doi.org/10.1016/j.jhazmat.2016.08.071

Liu, S.G., Zhu, Z.L., Tan, X.C., Feng, X.H., Huang, Z.Y., Qiu, Y.L., Zhao, J.F., 2013. The influence of Cu(II) on the formation and distribution of disinfection by-products during the chlorination of drinking water. Water, Air, and Soil Pollution 224, 1–13. https://doi.org/10.1007/S11270-013-1493-5/TABLES/3

Liu, Z., Lin, Y.L., Xu, B., Hu, C.Y., Wang, A.Q., Gao, Z.C., Xia, S.J., Gao, N.Y., 2018. Formation of iodinated trihalomethanes during breakpoint chlorination of iodide-containing water. Journal of Hazardous Materials 353, 505–513. https://doi.org/10.1016/j.jhazmat.2018.04.009

L. Linge, K., Ina Kristiana, Deborah Liew, Adam Holman, A. Joll, C., 2020. Halogenated semivolatile acetonitriles as chloramination disinfection by-products in water treatment: a new formation pathway from activated aromatic compounds. Environmental Science: Processes & Impacts 22, 653–662. https://doi.org/10.1039/C9EM00603F

Lu, G., Qin, D., Wang, Y., Liu, J., Chen, W., 2018. Single and combined effects of selected haloacetonitriles in a human-derived hepatoma line. Ecotoxicology and Environmental Safety 163, 417–426. https://doi.org/10.1016/j.ecoenv.2018.07.104

Ma, H., Li, Y., Zhang, H., Shah, S.M., Chen, J., 2014. Salt-assisted dispersive liquid-liquid microextraction coupled with programmed temperature vaporization gas chromatography-mass spectrometry for the determination of haloacetonitriles in drinking water. Journal of Chromatography A 1358, 14–19. https://doi.org/10.1016/j.chroma.2014.06.021

Malley, J.P., Burris, B., 2004. Ultraviolet Disinfection. World Water Congress.

Marhaba, T.F., Van, D., 2000. The variation of mass and disinfection by-product formation potential of dissolved organic matter fractions along a conventional surface water treatment plant. Journal of Hazardous Materials 74, 133–147. https://doi.org/10.1016/S0304-3894(99)00190-9

Marti, E.J., Pisarenko, A.N., Peller, J.R., Dickenson, E.R.V., 2015. N-nitrosodimethylamine (NDMA) formation from the ozonation of model compounds. Water Research 72, 262–270. https://doi.org/10.1016/j.watres.2014.08.047

Martínez-Carballo, E., Sitka, A., González-Barreiro, C., Kreuzinger, N., Fürhacker, M., Scharf, S., Gans, O., 2007. Determination of selected quaternary ammonium compounds by liquid chromatography with mass spectrometry. Part I. Application to surface, waste and indirect discharge water samples in Austria. Environmental Pollution 145, 489–496. https://doi.org/10.1016/j.envpol.2006.04.033

McGuire, M.J., 2006. Eight revolutions in the history of US drinking water disinfection. Journal / American Water Works Association 98, 123–149. https://doi.org/10.1002/j.1551-8833.2006.tb07612.x

Melnick, R. L., Kohn, M. C., Dunnick, J. K., & Leininger, J. R., 1998. Regenerative Hyperplasia Is Not Required for Liver Tumor Induction in Female B6C3F1Mice Exposed to Trihalomethanes. Toxicology and applied pharmacology, 148(1), 137-147.

Menzies, J.Z., McDonough, K., McAvoy, D., Federle, T.W., 2017. Biodegradation of nonionic and anionic surfactants in domestic wastewater under simulated sewer conditions. Biodegradation 28, 1–14. https://doi.org/10.1007/s10532-016-9773-6

Mitch, W.A., Sedlak, D.L., 2004. Characterization and Fate of N-Nitrosodimethylamine Precursors in Municipal Wastewater Treatment Plants. Environmental Science and Technology 38, 1445–1454. https://doi.org/10.1021/ES035025N

Mitch, W.A., Sedlak, D.L., 2001. Formation of N-Nitrosodimethylamine (NDMA) from Dimethylamine during Chlorination. Environmental Science and Technology 36, 588–595. https://doi.org/10.1021/ES010684Q

Mitch, W.A., Sharp, J.O., Trussell, R.R., Valentine, R.L., Alvarez-Cohen, L., Sedlak, D.L., 2003. N-nitrosodimethylamine (NDMA) as a drinking water contaminant: A review. Environmental Engineering Science 20, 389–404. https://doi.org/10.1089/109287503768335896

Mizgireuv, I. V., Majorova, I.G., Gorodinskaya, V.M., Khudoley, V. v., Revskoy, S.Y., 2004. Carcinogenic Effect of N-Nitrosodimethylamine on Diploid and Triploid Zebrafish (Danio rerio). Toxicologic Pathology 32, 514–518. https://doi.org/10.1080/01926230490496311

Mouly, D., Joulin, E., Rosin, C., Beaudeau, P., Zeghnoun, A., Olszewski-Ortar, A., Munoz, J.F., Welté, B., Joyeux, M., Seux, R., Montiel, A., Rodriguez, M.J., 2010. Variations in trihalomethane levels in three French water distribution systems and the development of a predictive model. Water Research 44, 5168–5179. https://doi.org/10.1016/j.watres.2010.06.028

Muellner, M.G., Wagner, E.D., Mccalla, K., Richardson, S.D., Woo, Y.T., Plewa, M.J., 2007. Haloacetonitriles vs. regulated haloacetic acids: Are nitrogen-containing DBFs more toxic? Environmental Science and Technology 41, 645–651. https://doi.org/10.1021/es0617441

Mustapha, D. S., & Bawa-Allah, K. A., 2020. Differential toxicities of anionic and nonionic surfactants in fish. Environmental Science and Pollution Research, 27(14), 16754-16762.

Najm, I., Trussell, R.R., 2001. NDMA Formation in Water and Wastewater. Journal - American Water Works Association 93, 92–99. https://doi.org/10.1002/J.1551-8833.2001.TB09129.X

Navalon, S., Alvaro, M., Garcia, H., 2009. Ca2+ and Mg2+ present in hard waters enhance trihalomethane formation. Journal of Hazardous Materials 169, 901–906. https://doi.org/10.1016/J.JHAZMAT.2009.04.031

Nawrocki, J., Andrzejewski, P., 2011. Nitrosamines and water. Journal of Hazardous Materials 189, 1–18. https://doi.org/10.1016/j.jhazmat.2011.02.005

Nihemaiti, M., Roux, J. le, Hoppe-Jones, C., Reckhow, D.A., Croué, J.-P., 2016. Formation of Haloacetonitriles, Haloacetamides, and Nitrogenous Heterocyclic Byproducts by Chloramination of Phenolic Compounds. Environmental Science and Technology 51, 655–663. https://doi.org/10.1021/ACS.EST.6B04819

Nokes, C.J., Fenton, E., Randall, C.J., 1999. Modelling the formation of brominated trihalomethanes in chlorinated drinking waters. Water Research 33, 3557–3568. https://doi.org/10.1016/S0043-1354(99)00081-0

Olkowska, E., Polkowska, Ż., Ruman, M., Namieśnik, J., 2014. Similar concentration of surfactants in rural and urban areas. Environmental Chemistry Letters 13, 97–104. https://doi.org/10.1007/s10311-014-0485-z

Oya, M., Kosaka, K., Asami, M., & Kunikane, S. (2008). Formation of N-nitrosodimethylamine (NDMA) by ozonation of dyes and related compounds. Chemosphere, 73(11), 1724-1730.

Pagé-Larivière, F., Tremblay, A., Campagna, C., Rodriguez, M.J., Sirard, M.A., 2016. Low concentrations of bromodichloromethane induce a toxicogenomic response in porcine embryos in vitro. Reproductive Toxicology 66, 44–55. https://doi.org/10.1016/j.reprotox.2016.09.010

Pan, Y., Zhang, X., 2013. Four Groups of New Aromatic Halogenated Disinfection Byproducts: Effect of Bromide Concentration on Their Formation and Speciation in Chlorinated Drinking Water. Environmental Science and Technology 47, 1265–1273. https://doi.org/10.1021/ES303729N

Park, K.Y., Choi, S.Y., Lee, S.H., Kweon, J.H., Song, J.H., 2016. Comparison of formation of disinfection by-products by chlorination and ozonation of wastewater effluents and their toxicity to Daphnia magna. Environmental Pollution 215, 314–321. https://doi.org/10.1016/j.envpol.2016.04.001

Parvez, S., Rivera-Núñez, Z., Meyer, A., Wright, J.M., 2011. Temporal variability in trihalomethane and haloacetic acid concentrations in Massachusetts public drinking water systems. Environmental Research 111, 499–509. https://doi.org/10.1016/j.envres.2010.12.008

Patel, P., Varshney, P., Patel, D., Meshram, D., 2014. Method development and validation of benzalkonium chloride in marketed formulation by UV-visible spectrophotometry using silver nitrate and eosin solution. World Journal of Pharmacy and Pharmaceutical Sciences

Pati, S.G., Arnold, W.A., 2020. Comprehensive screening of quaternary ammonium surfactants and ionic liquids in wastewater effluents and lake sediments. Environmental Science: Processes & Impacts 22, 430–441. https://doi.org/10.1039/C9EM00554D

Pedersen, E.J., Urbansky, E.T., Mariñas, B.J., Margerum, D.W., 1999. Formation of cyanogen chloride from the reaction of monochloramine with formaldehyde. Environmental Science and Technology 33, 4239–4249. https://doi.org/10.1021/es990153q

Pérez, P., Fernández, E., Beiras, R., 2009. Toxicity of benzalkonium chloride on monoalgal cultures and natural assemblages of marine phytoplankton. Water, Air, and Soil Pollution 201, 319–330. https://doi.org/10.1007/s11270-008-9947-x

Piazzoli, A., Breider, F., Aquillon, C.G., Antonelli, M., von Gunten, U., 2018a. Specific and total N-nitrosamines formation potentials of nitrogenous micropollutants during chloramination. Water Research 135, 311–321. https://doi.org/10.1016/J.WATRES.2018.02.019

Plumlee, M.H., López-Mesas, M., Heidlberger, A., Ishida, K.P., Reinhard, M., 2008. N-nitrosodimethylamine (NDMA) removal by reverse osmosis and UV treatment and analysis via LC-MS/MS. Water Research 42, 347–355. https://doi.org/10.1016/j.watres.2007.07.022

Postigo, C., Richardson, S.D., Barceló, D., 2017. Formation of iodo-trihalomethanes, iodo-haloacetic acids, and haloacetaldehydes during chlorination and chloramination of iodine containing waters in laboratory controlled reactions. Journal of Environmental Sciences 58, 127–134. https://doi.org/10.1016/j.jes.2017.04.009

Rebello, S., Asok, A.K., Mundayoor, S., Jisha, M.S., 2014. Surfactants: Toxicity, remediation and green surfactants. Environmental Chemistry Letters 12, 275–287. https://doi.org/10.1007/s10311-014-0466-2

Richardson, S.D., Fasano, F., Ellington, J.J., Crumley, F.G., Buettner, K.M., Evans, J.J., Blount, B.C., Silva, L.K., Waite, T.J., Luther, G.W., Mckague, A.B., Miltner, R.J., Wagner, E.D., Plewa, M.J., 2008. Occurrence and mammalian cell toxicity of iodinated disinfection byproducts in drinking water. Environmental Science and Technology 42, 8330–8338. https://doi.org/10.1021/es801169k

Richardson, S.D., Plewa, M.J., 2020. To regulate or not to regulate? What to do with more toxic disinfection by-products? Journal of Environmental Chemical Engineering 8, 103939. https://doi.org/10.1016/J.JECE.2020.103939

Richardson, S.D., Thruston, A.D., Caughran, T. v., Chen, P.H., Collette, T.W., Floyd, T.L., Schenck, K.M., Lykins, B.W., Sun, G.R., Majetich, G., 1999. Identification of new ozone disinfection byproducts in drinking water. Environmental Science and Technology 33, 3368–3377. https://doi.org/10.1021/es981218c

Richardson, S.D., Thruston, A.D., Caughran, T. v., Chen, P.H., Collette, T.W., Schenck, K.M., Lykins, B.W., Rav-Acha, C., Glezer, V., 2000. Identification of new drinking water

disinfection by-products from ozone, chlorine dioxide, chloramine, and chlorine. Water, Air, and Soil Pollution 123, 95–102. https://doi.org/10.1007/978-94-011-4369-1_9

Rodriguez, M.J., Sérodes, J.B., 2001. Spatial and temporal evolution of trihalomethanes in three water distribution systems. Water Research 35, 1572–1586. https://doi.org/10.1016/S0043-1354(00)00403-6

Rodriguez, M. J., Vinette, Y., Sérodes, J. B., & Bouchard, C., 2003. Trihalomethanes in drinking water of greater Québec region (Canada): occurrence, variations and modelling. Environmental monitoring and assessment, 89(1), 69-93.

Rook, J.J., 1967. Guide to the Chemicals Used in Crop Protec-tion. Bull. Environ. Contam. Toxicol 2, 1571.

Roux, J. L, Gallard, H., Croué, J.-P., 2012a. Formation of NDMA and Halogenated DBPs by Chloramination of Tertiary Amines: The Influence of Bromide Ion. Environmental Science and Technology 46, 1581–1589. https://doi.org/10.1021/ES203785S

Roux, J. L., Gallard, H., Croué, J.P., 2011. Chloramination of nitrogenous contaminants (pharmaceuticals and pesticides): NDMA and halogenated DBPs formation. Water Research 45, 3164–3174. https://doi.org/10.1016/J.WATRES.2011.03.035

Sack, S., Avisar, D., Kaplan, A., & Lester, Y., 2021. Detection of N-nitrosodimethylamine (NDMA) and its formation potential in hospital wastewater. *Environmental Science and Pollution Research*, *28*(11), 14199-14206.

Schmidt, C.K., Brauch, H.J., 2008. N,N-dimethylsulfamide as precursor for N-nitrosodimethylamine (NDMA) formation upon ozonation and its fate during drinking water treatment. Environmental Science and Technology 42, 6340–6346. https://doi.org/10.1021/es7030467

Selbes, M., Kim, D., Ates, N., Karanfil, T., 2013. The roles of tertiary amine structure, background organic matter and chloramine species on NDMA formation. Water Research 47, 945–953. https://doi.org/10.1016/J.WATRES.2012.11.014

Sfynia, C., Bond, T., Kanda, R., Templeton, M.R., 2020a. The formation of disinfection by-products from the chlorination and chloramination of amides. Chemosphere 248, 125940. https://doi.org/10.1016/J.CHEMOSPHERE.2020.125940

Sfynia, C., Bond, T., Kanda, R., Templeton, M.R., 2020b. The formation of disinfection by-products from the chlorination and chloramination of amides. Chemosphere 248, 125940. https://doi.org/10.1016/J.chemosphere.2020.125940

Sgroi, M., Vagliasindi, F.G.A., Snyder, S.A., Roccaro, P., 2018. N-Nitrosodimethylamine (NDMA) and its precursors in water and wastewater: A review on formation and removal. Chemosphere 191, 685–703. https://doi.org/10.1016/j.chemosphere.2017.10.089

Shah, A.D., Dotson, A.D., Linden, K.G., Mitch, W.A., 2011. Impact of UV disinfection combined with chlorination/chloramination on the formation of halonitromethanes and haloacetonitriles in drinking water. Environmental Science and Technology 45, 3657–3664. https://doi.org/10.1021/ES104240V/

Shah, A.D., Mitch, W.A., 2012. Halonitroalkanes, halonitriles, haloamides, and N-nitrosamines: A critical review of nitrogenous disinfection byproduct formation pathways. Environmental Science and Technology 46, 119–131. https://doi.org/10.1021/es203312s

Shao, K.L., Ye, Z.X., Huang, H., Yang, X., 2020. ClO2 pre-oxidation impacts the formation and nitrogen origins of dichloroacetonitrile and dichloroacetamide during subsequent chloramination. Water Research 186. https://doi.org/10.1016/j.watres.2020.116313

Sharma, V.K., Yang, X., Cizmas, L., McDonald, T.J., Luque, R., Sayes, C.M., Yuan, B., Dionysiou, D.D., 2017. Impact of metal ions, metal oxides, and nanoparticles on the formation of disinfection byproducts during chlorination. Chemical Engineering Journal 317, 777–792. https://doi.org/10.1016/J.CEJ.2017.02.071

Sharma, V.K., Zboril, R., McDonald, T.J., 2014. Formation and toxicity of brominated disinfection byproducts during chlorination and chloramination of water: A review. Journal of Environmental Science and Health - Part B Pesticides, Food Contaminants, and Agricultural Wastes 49, 212–228. https://doi.org/10.1080/03601234.2014.858576

Shen, R., Andrews, S.A., 2011. Demonstration of 20 pharmaceuticals and personal care products (PPCPs) as nitrosamine precursors during chloramine disinfection. Water Research 45, 944–952. https://doi.org/10.1016/J.WATRES.2010.09.036

Sliwa, H., & Tartar, A. 1976. Isolation and alkaline decomposition of the intermediated pyridinium salts occurring in the pyridine N-oxide oxidation of. alpha.-halo esters or acids. The Journal of Organic Chemistry, 41(1), 160-163.

Simmons, T. E. K. J. E., & Pegram, R. A., 1998. NOAEL and LOAEL determinations of acute hepatotoxicity for chloroform and bromodichloromethane delivered in an aqueous vehicle to F344 rats. Journal of toxicology and environmental health Part A, 55(1), 65-75.

Simpson, K.L., Hayes, K.P., 1998. Drinking water disinfection by-products: An Australian perspective. Water Research 32, 1522–1528. https://doi.org/10.1016/S0043-1354(97)00341-2

Sorlini, S., & Collivignarelli, C. (2005). Trihalomethane formation during chemical oxidation with chlorine, chlorine dioxide and ozone of ten Italian natural waters. Desalination, 176(1-3), 103-111.

Speight, V.L., Singer, P.C., 2005. Association between residual chlorine loss and HAA reduction in distribution systems. Journal - American Water Works Association 97, 82–91. https://doi.org/10.1002/J.1551-8833.2005.TB10827.X

Sreevidya, V.S., Lenz, K.A., Svoboda, K.R., Ma, H., 2018. Benzalkonium chloride, benzethonium chloride, and chloroxylenol - Three replacement antimicrobials are more toxic than triclosan and triclocarban in two model organisms. Environmental Pollution 235, 814–824. https://doi.org/10.1016/j.envpol.2017.12.108

Sütterlin, H., Alexy, R., Coker, A., Kümmerer, K., 2008. Mixtures of quaternary ammonium compounds and anionic organic compounds in the aquatic environment: Elimination and biodegradability in the closed bottle test monitored by LC-MS/MS. Chemosphere 72, 479–484. https://doi.org/10.1016/j.chemosphere.2008.03.008

Tezel, U., Pavlostathis, S.G., 2009. Transformation of Benzalkonium Chloride under Nitrate Reducing Conditions. Environmental Science and Technology 43, 1342–1348. https://doi.org/10.1021/ES802177F

Toledano, M.B., Nieuwenhuijsen, M.J., Best, N., Whitaker, H., Hambly, P., de Hoogh, C., Fawell, J., Jarup, L., Elliott, P., 2005. Relation of trihalomethane concentrations in public water supplies to stillbirth and birth weight in three water regions in England. Environmental Health Perspectives 113, 225–232. https://doi.org/10.1289/ehp.7111

Trussell, R.R., 1978. Formation of Trihalomethanes. Journal / American Water Works Association 70, 604–612. https://doi.org/10.1002/j.1551-8833.1978.tb04255.x

Ul'yanovskii, N. v., Kosyakov, D.S., Varsegov, I.S., Popov, M.S., Lebedev, A.T., 2020. Identification of novel disinfection byproducts in pool water: Chlorination of the algaecide benzalkonium chloride. Chemosphere 239, 124801. https://doi.org/10.1016/J.chemosphere.2019.124801

VanBriesen, J. M., 2014. Potential drinking water effects of bromide discharges from coal-fired electric power plants. EPA NPDES Comments, 1-38.

Van de Voorde, A., Lorgeoux, C., Gromaire, M.C., Chebbo, G., 2012. Analysis of quaternary ammonium compounds in urban stormwater samples. Environmental Pollution 164, 150–157. https://doi.org/10.1016/J.envpol.2012.01.037

Villanueva, C.M., Cordier, S., Font-Ribera, L., Salas, L.A., Levallois, P., 2015. Overview of Disinfection By-products and Associated Health Effects. Curr. Environ. Heal. reports 2, 107–115. https://doi.org/10.1007/s40572-014-0032-x

Villanueva, C.M., Fernández, F., Malats, N., Grimalt, J.O., Kogevinas, M., 2003. Meta-analysis of studies on individual consumption of chlorinated drinking water and bladder cancer. J. Epidemiol. Community Health 57, 166–173. https://doi.org/10.1136/jech.57.3.166

Von Gunten, U., & Hoigne, J., 1994. Bromate formation during ozonization of bromide-containing waters: interaction of ozone and hydroxyl radical reactions. Environmental science & technology, 28(7), 1234-1242.

Wang, G.S., Deng, Y.C., Lin, T.F., 2007. Cancer risk assessment from trihalomethanes in drinking water. Science of The Total Environment 387, 86–95. https://doi.org/10.1016/J.SCITOTENV.2007.07.029

Wang, J., Liu, X., Ng, T.W., Xiao, J., Chow, A.T., Wong, P.K., 2013. Disinfection byproduct formation from chlorination of pure bacterial cells and pipeline biofilms, Water Research 47, 2701-2709. doi: 10.1016/j.watres.2013.02.038.

Wang, Y., Dong, H., Wu, Z., Qiang, Z., 2020. Organic Amines Enhance the Formation of Iodinated Trihalomethanes during Chlorination of Iodide-Containing Waters. Environmental Science & Technology 54, 4651–4657. https://doi.org/10.1021/ACS.EST.9B07234

Weinberg, H.S., Glaze, W.H., Krasner, S.W., Sclimenti, M.J., 1993. Formation and removal of aldehydes in plants that use ozonation. Journal / American Water Works Association 85, 72–85. https://doi.org/10.1002/j.1551-8833.1993.tb05988.x

Werdehoff, K.S., Singer, P.C., 1987. Chlorine Dioxide Effects on Thmfp, Toxfp, and the Formation of Inorganic By-Products. Journal / American Water Works Association 79, 107–113. https://doi.org/10.1002/j.1551-8833.1987.tb02908.x

West, D.M., Wu, Q., Donovan, A., Shi, H., Ma, Y., Jiang, H., Wang, J., 2016. N-nitrosamine formation by monochloramine, free chlorine, and peracetic acid disinfection with presence of amine precursors in drinking water system. Chemosphere 153, 521–527. https://doi.org/10.1016/j.chemosphere.2016.03.035

Westerhoff, P., Chao, P., Mash, H., 2004. Reactivity of natural organic matter with aqueous chlorine and bromine. Water Research 38, 1502–1513. https://doi.org/10.1016/J.WATRES.2003.12.014

WHO, 2011. Guidelines for drinking-water quality. World Health Organization, 216, 303-4.

Williams, D.T., Benoit, F.M., Lebel, G.L., 1998. Trends in levels of disinfection by-products. Environmetrics 9, 555–563. https://doi.org/10.1002/(SICI)1099-095X(199809/10)9:5<555::AID-ENV323>3.0.CO;2-W

Włodyka-Bergier, A., Bergier, T., 2011. The influence of organic matter quality on the potential of volatile organic matter. Archives of Environmental Protection 37, 25-35.

Wright, J.M., Schwartz, J., Dockery, D.W., 2004. The effect of disinfection by-products and mutagenic activity on birth weight and gestational duration. Environmental Health Perspectives 112, 920–925. https://doi.org/10.1289/ehp.6779

Wu, W.W., Benjamin, M.M., Korshin, G. v., 2001. Effects of thermal treatment on halogenated disinfection by-products in drinking water. Water Research 35, 3545–3550. https://doi.org/10.1016/S0043-1354(01)00080-X

Wu, Y., Clevenger, T., Deng, B., 2005. Impacts of goethite particles on UV disinfection of drinking water. Applied and Environmental Microbiology 71, 4140–4143. https://doi.org/10.1128/AEM.71.7.4140-4143.2005

Yang, X., Fan, C., Shang, C., Zhao, Q., 2010. Nitrogenous disinfection byproducts formation and nitrogen origin exploration during chloramination of nitrogenous organic compounds. Water Research 44, 2691–2702. https://doi.org/10.1016/j.watres.2010.01.029

Yang, X., Guo, W., Lee, W., 2013. Formation of disinfection byproducts upon chlorine dioxide preoxidation followed by chlorination or chloramination of natural organic matter. Chemosphere 91, 1477–1485. https://doi.org/10.1016/j.chemosphere.2012.12.014

Yang, X., Shang, C., Lee, W., Westerhoff, P., Fan, C., 2008. Correlations between organic matter properties and DBP formation during chloramination. Water Research 42, 2329–2339. https://doi.org/10.1016/j.watres.2007.12.021

Yang, X., Shang, C., Shen, Q., Chen, B., Westerhoff, P., Peng, J., Guo, W., 2012. Nitrogen origins and the role of ozonation in the formation of haloacetonitriles and

halonitromethanes in chlorine water treatment. Environmental Science and Technology 46, 12832–12838. https://doi.org/10.1021/es302993u

Yang, X., Shang, C., Westerhoff, P., 2007. Factors affecting formation of haloacetonitriles, haloketones, chloropicrin and cyanogen halides during chloramination. Water Research 41, 1193–1200. https://doi.org/10.1016/j.watres.2006.12.004

Yang, X., Shen, Q., Guo, W., Peng, J., Liang, Y., 2012b. Precursors and nitrogen origins of trichloronitromethane and dichloroacetonitrile during chlorination/chloramination. Chemosphere 88, 25–32. https://doi.org/10.1016/j.chemosphere.2012.02.035

Ye, B., Wang, W., Yang, L., Wei, J., 2009. Factors influencing disinfection by-products formation in drinking water of six cities in China. Journal of Hazardous Materials 171, 147–152. https://doi.org/10.1016/j.jhazmat.2009.05.117

Yu, Y., & Reckhow, D. A. (2015). Kinetic analysis of haloacetonitrile stability in drinking waters. Environmental science & technology, 49(18), 11028-11036.

Zeng, T., Mitch, W.A., 2015. Contribution of N-Nitrosamines and Their Precursors to Domestic Sewage by Greywaters and Blackwaters. Environmental Science and Technology 49, 13158–13167. https://doi.org/10.1021/ACS.EST.5B04254/SUPPL_FILE/ES5B04254_SI_001.PDF

Zhang, B., Xian, Q., Gong, T., Li, Y., Li, A., Feng, J., 2017. DBPs formation and genotoxicity during chlorination of pyrimidines and purines bases. Chemical Engineering Journal 307, 884–890. https://doi.org/10.1016/J.CEJ.2016.09.018

Zhang, C., Cui, F., Zeng, G. ming, Jiang, M., Yang, Z. zhu, Yu, Z. gang, Zhu, M. ying, Shen, L. qing, 2015a. Quaternary ammonium compounds (QACs): A review on occurrence, fate and toxicity in the environment. Science of The Total Environment 518–519, 352–362. https://doi.org/10.1016/J.SCITOTENV.2015.03.007

Zhang, H., Dong, H., Adams, C., Qiang, Z., Luan, G., Wang, L., 2015. Formation and speciation of disinfection byproducts during chlor(am)ination of aquarium seawater. Journal of Environmental Sciences (China) 33, 116–124. https://doi.org/10.1016/j.jes.2014.11.016

Zhang, M., Ma, H., Wang, H., Du, T., Liu, M., Wang, Y., Zhang, T., Li, Y., 2019. Effects of ion species on the disinfection byproduct formation in artificial and real water. Chemosphere 217, 706–714. https://doi.org/10.1016/J.CHEMOSPHERE.2018.11.066

Zhao, C., Dong, Yan, Feng, Y., Li, Y., Dong, Yong, 2019. Thermal desorption for remediation of contaminated soil: A review. Chemosphere 221, 841–855. https://doi.org/10.1016/j.chemosphere.2019.01.079

Zhao, Y. Y., Boyd, J. M., Woodbeck, M., Andrews, R. C., Qin, F., Hrudey, S. E., & Li, X. F. (2008). Formation of N-nitrosamines from eleven disinfection treatments of seven different surface waters. Environmental Science & Technology, 42(13), 4857-4862.

Zheng, L., Sun, H., Wu, C., Wang, Y., Zhang, Y., Ma, G., Lin, H., Chen, J., Hong, H., 2020. Precursors for brominated haloacetic acids during chlorination and a new useful indicator for bromine substitution factor. Science of The Total Environment 698, 134250. https://doi.org/10.1016/J.SCITOTENV.2019.134250

Zhou, K., Ye, S., Yu, Q., Chen, J., Yong, P., Ma, X., Li, Q., Dietrich, A.M., 2021. Derivates variation of phenylalanine as a model disinfection by-product precursor during long term chlorination and chloramination. Science of The Total Environment 771, 144885. https://doi.org/10.1016/J.SCITOTENV.2020.144885

Curriculum Vitae

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EDUCATION

University of Nevada Las Vegas 2021
Master of Science in Civil and Environmental Engineering (GPA: 3.67)

Anna University, India 2018
Bachelor of Engineering in Civil Engineering (GPA: 7.94/10)

SKILLS AND CERTIFICATIONS

Skills: ArcGIS Pro, AutoCAD Civil 3D, Revu Bluebeam, Microsoft Office, experimental research, technical report & proposal writing.
Certifications: 40 Hour HAZWOPER training

RESEARCH/WORK EXPERIENCE

- **Engineering Intern, S&B Christ Consulting, Las Vegas, NV** 2021
  - Performed water sampling for volatile compounds from activated carbon groundwater treatment system.
  - Assisted in the design of site plans using AutoCAD Civil 3D as well as quantities estimation.
  - Conducted field surveying using Trimble R10 GNSS.
- **Graduate Research Assistant, Environmental Engineering Laboratory, UNLV** 2019

1. Adsorption of TCE and PCE from water using biochar; 2. Disinfection byproduct formation potential of cationic surfactants

  - Developed experimental plan and methodology.
  - Assisted in the development of a WRF project proposal for approaches to minimize Br^- and I^- DBPs in distribution systems.
  - Conducted batch experiments for adsorption equilibrium and kinetics.
  - Prepared working standards and chemical stock solutions
  - Prepared standard operating procedures and managed work tasks and schedules for undergraduate research assistants.
  - Conducted disinfection byproducts formation potential tests and extractions
  - Operated purge & trap and GC-ECD/MS for analysis of trace organic compounds in water.
- **Research Intern, Environmental Engineering Laboratory, IIT, India** 2018

Remediation of fuel oil contaminated soil by microwave heating using spent graphite as susceptor.

- Operated GC-MS, FTIR, TOC-S analysis and physical characterization of the samples.
- Performed solvent extraction of total petroleum hydrocarbon from soil using gravimetric analysis.
- Set up and engineered bench scale testing
- Co-authored the manuscript of the findings for publication

➢ **Undergraduate Research Intern (International exchange program), UNLV** 2017

Evaluation of Biological Degradation of Hexavalent Chromium and perchlorate

- Prepared ground water Influent mix for pilot scale biological column feed
- Samples effluent and measured various chemicals following acceptable methods
- Set up columns and plumbing for Fluidized Bed Reactor

INTERNATIONAL PUBLICATIONS AND CONFERENCES

- Comparison of Biochars Obtained from Different Feedstocks for the Adsorption of TCE and PCE in Water. **WM Symposia**, March 2021
- Adsorption of trichloroethylene and tetrachloroethylene by biochar derived from three different feedstocks – A comparative study. **ACS Fall 2020 Virtual Meeting and Expo**.
- Upgrading City of Henderson Kurt R. Segler Water Reclamation Facility for Biological Nitrogen Removal, **WEF student design competition**, October 2020
- Microwave remediation of hydrocarbon contaminated soil using spent graphite – an approach for waste as a resource. **Journal of Environmental Management**, 230(2019)151-158.
- Study on decontamination of aged spent wash using activated carbon from *Limonia acidissima* shell, **Desalination and Water Treatment**, 82(2017)369-378.

AWARDS & SCHOLARSHIPS

- NWRA graduate scholarship - $1000 2021
- UNLV Grad Rebel Slam semi-finalist - $100 cash prize. 2020
- GPSA award of $1000 from UNLV for thesis research. 2020
- Achievers Award (2017-2018)-Department of Civil Engineering, BIT. 2018
- Best Student Project National Award – Dr. Kalam Educational Trust. 2017