THESE

Présentée à

L'UNIVERSITE DE LILLE

Ecole Doctorale – Sciences de la Matière, du Rayonnement et de l'Environnement
Faculté des Sciences et Technologies – Département Chimie

Pour obtenir le grade de

DOCTEUR

Discipline: Chimie Théorique, Physique, Analytique

Rédigée par

Henry MACKEOWN

EVALUATION DE LA REDUCTION DU POTENTIEL DE FORMATION DES TRIHALOMETHANES ET DES ACIDES HALOACETIQUES PAR DES PROCEDES DE TRAITEMENT DE L'EAU POTABLE : APPLICATION SUR DES EAUX IMPACTEES PAR LES IONS IODURE ET BROMURE

Soutenue publiquement le 21 novembre 2019, devant le jury:

Urs von GUNTEN	Professeur, EAWAG / ETHZurich / EPFL	Président du Jury
Jean-Luc BOUDENNE	Professeur, Université d'Aix-Marseille	Rapporteur
Hervé GALLARD	Professeur, Université de Poitiers	Rapporteur
Isabelle BAUDIN	PhD, CIRSEE, SUEZ	Examinatrice
Holly SHORNEY-DARBY	PhD, PWNT	Examinatrice
Baghdad OUDDANE	Professeur, Université de Lille	Directeur
Justine CRIQUET	Maître de Conférences, Université de Lille	Co-encadrante

THESIS

Presented at

LILLE UNIVERSITY

Doctoral School for Materials, Radiation and Environmental Sciences

Faculty of Science and Technology – Department of Chemistry

To attain the grade of

DOCTOR

Discipline: Theoretical, Physical, Analytical Chemistry

By

Henry MACKEOWN

ASSESSMENT OF TRIHALOMETHANE AND HALOACETIC ACID FORMATION POTENTIAL REDUCTION BY DRINKING WATER TREATMENT PROCESSES: APPLICATION ON IODIDE AND BROMIDE IMPACTED WATERS

Defended publicly on the 21st of November 2019, in front of the review committee:

Urs von GUNTEN	Professor, EAWAG / ETHZurich / EPFL	President of the Jury
Jean-Luc BOUDENNE	Professor, Aix-Marseille University	Examiner
Hervé GALLARD	Professor, Poitiers University	Examiner
Isabelle BAUDIN	PhD, CIRSEE, SUEZ	Co-examiner
Holly SHORNEY-DARBY	PhD, PWNT	Co-examiner
Baghdad OUDDANE	Professor, Lille University	Director
Justine CRIQUET	University Lecturer, Lille University	Co-supervisor

Assessment of trihalomethane and haloacetic acid formation potential reduction by drinking water treatment processes: Application on iodide and bromide impacted waters

Disinfectants react with natural organic matter (NOM) to produce undesired disinfection by-products (DBPs). Drinking water resources in coastal areas can be impacted by higher concentrations of iodide leading to iodinated DBP formation which are more toxic than their brominated and chlorinated analogues. Two analytical methods to analyse DBPs by headspace-trap gas chromatography were developed at Lille University: one for the trihalomethanes – with a focus on iodinated species (I-THMs) – and one for the haloacetic acids (HAAs). Those methods were implemented in a collaboration with De Watergroep to look at the removal of these DBP precursors in a full-scale drinking water treatment plant as well as in the assessment of ion exchange resins, as part of an ambitious project to modernize the plant. As well as formation potential tests on real waters, different scenarios of increasing iodide and bromide concentrations were tested. The results were also related to the characterization of the natural organic matter, with the removal of humic substances and building blocks leading to the greatest abatement of THM formation potential during coagulation-decantation and ion exchange treatment. Preformed I-THMs and HAAs were greatly removed by the granular activated carbon filters in the full scale plant, reducing the overall relative cytotoxicity. Amberlite IRA410 had the best performance amongst 4 resins on NOM removal and DBP formation potential reduction.

Key-words: DBPs, chlorination, trihalomethanes, haloacetic acids, natural organic matter, iodide, bromide, ion exchange resins.

Evaluation de la réduction du potentiel de formation des trihalométhanes et des acides haloacétiques par des procédés de traitement de l’eau potable : Application sur des eaux impactées par les ions iodure et bromure

Les désinfectants utilisés dans la production d’eau potable réagissent avec la matière organique naturelle et produisent des sous-produits de désinfection. Les ressources en eaux potables dans les régions côtières sont susceptibles d’être impactées par des concentrations plus élevées en ions iodure entraînant la formation de sous-produits iodés, plus toxiques que leurs analogues bromés et chlorés. Deux méthodes analytiques ont été développées à l’Université de Lille : une méthode pour analyser les trihalométhanes – en particulier les espèces iodées (I-THMs) – et une méthode pour les acides haloacétiques (HAAs). Ces méthodes ont été utilisées pour une étude sur une usine de traitement de l’eau ainsi que pour l’évaluation de résines échangeuses d’ions, dans le cadre d’un ambitieux projet de modernisation de l’usine. En plus de tests de potentiels de formation, différents scénarios de concentrations croissantes en ions iodure et bromure ont été testés. Les résultats ont également été reliés à la caractérisation de la matière organique naturelle, avec l’abattement des substances humiques et des building blocks entraînant la plus grande réduction en potentiel de formation des THMs, durant la coagulation-décantation et lors du traitement par résines échangeuses d’ions. Les I-THMs et HAAs déjà formés furent significativement éliminés par les filtres à charbon actif sur l’usine de traitement, abaissant la cytotoxicité relative globale. La résine Amberlite IRA410 a eu la meilleure performance parmi les 4 résines testées sur l’abattement du carbone organique dissous et la réduction en potentiel de formation des sous-produits de désinfection.

Mots-clés : sous-produits de désinfection, chloration, trihalométhanes, acides haloacétiques, matière organique naturelle, iodure, bromure, résines échangeuses d’ions.

Table of contents

Acronyms

BB: building blocks

BCIM: bromochloroiodomethane

BDCAA: bromodichloroacetic acid

BDCAN: bromodichloroacetonitrile

BDCM: bromodichloromethane

BDIM: bromodiiodomethane

BV: bed volume

CDIM: chlorodiiodomethane

CHO cells: Chinese hamster ovary cells

DBAA: dibromoacetic acid

DBAN: dibromoacetonitrile

DBCAA: dibromochloroacetic acid

DBCM: dibromochloromethane

DBP: disinfection by-product

DCAA: dichloroacetic acid

DCAN: dichloroacetonitrile

DCIM: dichloroiodomethane

DBIM: dibromoiodomethane

DL: detection limit

DOC: dissolved organic carbon

DOM: dissolved organic matter

DWTP: drinking water treatment plant

EBCT: empty bed contact time

ECD: electron capture detection

FP: formation potential

FS: full scan

GC: gas chromatography

HAA: haloacetic acid

HAA5: chloroacetic acid, bromoacetic acid, dichloroacetic acid, dibromoacetic acid, trichloroacetic acid

HAA6: HAA5 + bromochloroacetic acid

HAA9: HAA6 + bromodichloroacetic acid, dibromochloroacetic acid, tribromoacetic acid

HAcAm: haloacetamide

HAL: haloacetaldehyde

HAN: haloacetonitrile

HOC: hydrophobic organic carbon fraction

HPLC: high performance liquid chromatography

HPSEC: high performance size exclusion chromatography

HS: humic substances

HS: headspace

IAL: iodoacetaldehyde

IAN: iodoacetonitrile

IC: ion chromatography

ICM: iodinated X-ray contrast media

ICP-MS: Inductively coupled plasma mass spectrometry

IEX: ion exchange

Kow: octanol/water partition coefficient

LC-OCD: liquid chromatography – organic carbon detection

LLE: liquid-liquid extraction

LLME: liquid-liquid microextraction

LMWN: low molecular weight neutrals

LMWA: low molecular weight acids

LOD: limit of detection

LOQ: limit of quantification

MBAA: (mono)bromoacetic acid

MBAN: (mono)bromoacetonitrile

MCAA: (mono)chloroacetic acid

MCAN: (mono)chloroacetonitrile

MIAA: (mono)iodoacetic acid

mQ: Milli-Q

MS: mass spectrometry

MW: molecular weight

NOM: natural organic matter

PMS: peroxymonosulfate

PT: purge and trap

PTV inlet: programmable temperature vaporizing inlet

SH: static headspace

SIM: selected ion monitoring

SIX: suspended ion exchange

SPME: solid-phase microextraction

SUVA: specific ultraviolet light absorbance

SW: surface water

TBAA: tribromoacetic acid

TCAA: trichloroacetic acid

TCAN: trichloroacetonitrile

TCM: chloroform/ trichloromethane

TBM: bromoform/ tribromomethane

THM: trihalomethane

THM4: the 4 regulated trihalomethanes

TIM: iodoform/triiodomethane

TOBr: total organic bromine

TOCl: total organic chlorine

TOI: total organic iodine

VOCs: volatile organic compounds

General introduction

General introduction

In drinking water treatment, the addition of chemical disinfectants is widely used to serve two main purposes: inactivation of microorganisms during primary treatment and impeding their regrowth in distribution systems (National Research Council (US) Safe Drinking Water Committee 1987; Sedlak and von Gunten 2011).

Chlorine and its compounds are the most commonly used disinfectants for drinking water treatment. The regular use of chlorine in drinking water systems to kill disease causing organisms began around the beginning of the twentieth century (IARC Working Group on the Evaluation of Carcinogenic Risk to Humans 1991). However a group of disinfection by-products (DBPs) known as trihalomethanes (THMs), including chloroform, was discovered in chlorinated drinking water in 1974, first by Rook (1974), and later by Bellar et al. (1974). Since chloroform was first identified, more than 700 other DBPs have been detected in drinking water disinfection (Richardson and Ternes 2018). Only a few among many potential DBP compounds are regulated (Richardson and Ternes 2018). Also iodide and bromide play an important role in DBP speciation as iodinated and brominated DBPs are more toxic than their chlorinated analogues (Krasner 2012).

The DOC2C's project is an extensive research program of four years to investigate the possibility of improved dissolved organic carbon (DOC) removal from source waters. During drinking water treatment, the removal of dissolved organic carbon (DOC) is a key factor in the improvement of drinking water quality (Adusei-Gyamfi et al. 2019), since DOC reduces the efficiency of all treatment steps and threatens water quality due to the formation of harmful disinfection by-products (DBPs) that could also bring odour and taste issues (Sedlak and von Gunten 2011; Xie 2016; Li and Mitch 2018) or lowered biostability in the network. This research was funded by European funds through the Interreg 2 Seas Program that is focused on the North Sea and the English Channel regions, known as the 2 Seas Area. The DOC2C's consortium consists of PWNT R&D (NL), South West Water (UK), De Watergroep (BE), Lille University (FR) and Delft University of Technology (NL) (https://doc2cs.com/).

Indeed, production of safe drinking water is increasingly under pressure in the area. The main concern is the growing concentration of DOC in surface waters. This could impact project partners in coastal areas which are vulnerable to high salinity and possibly high iodide levels in source waters. And in the context of climate change, water salinity may increase in the future such as by the increase of sea water intrusion and increasing periods of droughts.

The objectives of Lille University through 2 parallel theses was to characterize NOM with emphasis on the behaviour of NOM-metal complexes in drinking water treatment processes on the one hand, and to assess the formation of DBPs in the 2 seas area on the other hand. Then, the NOM characterization in this manuscript has been performed in the laboratory by the second PhD candidate of this project Junias Adusei Gyamfi.

This PhD thesis looked at the benefits of DOC removal by the different treatments of surface waters used by the project partners on DBP formation and speciation. Conventional and innovative treatments (*e.g.* ion exchange) were studied (pilot and full scale).

Chapter I consists of a review paper on the fate of iodide in drinking water treatment and the formation of iodinated disinfection by-products. It first introduces the global iodine cycle followed by a focus on the reactivity of iodide and iodine species during drinking water oxidative treatment, the occurrence of iodinated DBPs and the parameters influencing their formation.

Chapter II describes the methodologies used for sampling and analysing drinking water samples throughout this work as well as the sampling site.

Chapter III presents the analytical development performed to set up the analysis of several classes of DBPs in the laboratory by headspace-trap gas chromatography – mass spectrometry.

Chapter IV is presented as a research article on the removal of disinfection by-product precursors in a full-scale drinking water treatment plant belonging to De Watergroep (de Blankaart). This paper focuses on bromine incorporation, iodine speciation and on the theoretical toxicity assessment as well as the impact of intermediate chlorination and activated carbon subsequent filtration on disinfection by-product formation.

Chapter V compares four ion-exchange resins in terms of disinfection by-product precursor removal. The experiments were performed on the pilot scale facilities of de Blankaart treatment works. In fact this process may be implemented on the new water treatment plant in order to improve the removal of NOM while reducing the coagulant dosage.

References:

Adusei-Gyamfi et al. 2019. J. Adusei-Gyamfi, B. Ouddane, L. Rietveld, J-P. Cornard, and J. Criquet. '**Natural Organic Matter-Cations Complexation and Its Impact on Water Treatment: A Critical Review**'. Water Research, 160 (2019), pp. 130-147. *https://doi.org/10.1016/j.watres.2019.05.064*.

Bellar et al. 1974. T. A. Bellar, J. J. Lichtenberg, and R. C. Kroner. **The Occurrence of Organohalides in Chlorinated Drinking Waters**. U.S. Environmental Protection Agency, Office of Research and Development, National Environmental Research Center (1974). *https://doi.org/10.1002/j.1551-8833.1974.tb02129.x*.

IARC Working Group on the Evaluation of Carcinogenic Risk to Humans 1991. IARC Working Group on the Evaluation of Carcinogenic Risk to Humans. *Description of the Process*. International Agency for Research on Cancer, (1991). *https://www.ncbi.nlm.nih.gov/books/NBK506936/*.

Krasner 2012. S. W. Krasner. '**Halogenated DBPs and Emerging Issues**'. In *Disinfection By-Products and Human Health*, IWA Publishing, (2012), pp. 59-71.

Li and Mitch 2018. X-F. Li and W. A. Mitch. '**Drinking Water Disinfection Byproducts (DBPs) and Human Health Effects: Multidisciplinary Challenges and Opportunities**'. Environmental Science & Technology, 52 (4) (2018), pp. 1681-1689. *https://doi.org/10.1021/acs.est.7b05440*.

National Research Council (US) Safe Drinking Water Committee 1987. National Research Council (US) Safe Drinking Water Committee. '**Disinfection Methods and Efficacy**'. In *Drinking Water and Health: Disinfectants and Disinfectant By-Products*, 7. National Academies Press (US), (1987). *https://www.ncbi.nlm.nih.gov/books/NBK217999/*.

Richardson and Ternes 2017. S. D. Richardson and T. A. Ternes. '**Water Analysis: Emerging Contaminants and Current Issues**'. Analytical Chemistry, 90 (1) (2017), pp. 398-428. *https://doi.org/10.1021/acs.analchem.7b04577*.

Rook 1974. J. J. Rook. '**Formation of Haloforms during Chlorination of Natural Waters**'. Water Treatment and Examination, 23 (1974), pp. 234-243.

Sedlak and von Gunten 2011. D. S. Sedlak and U. von Gunten. '**The Chlorine Dilemma**'. Science, 331 (6013) (2011).

Xie 2016. Y. F. Xie. **Disinfection Byproducts in Drinking Water: Formation, Analysis, and Control**. CRC Press, (2016).

I. Fate of iodide during oxidative water treatment and formation of iodine- containing disinfection by-products – A critical review

I. Fate of iodide during oxidative water treatment and formation of iodine-containing disinfection by-products – A critical review

In collaboration with Urs von Gunten[1,2,3] and Justine Criquet[4]

[1]Eawag, Swiss Federal Institute of Aquatic Science and Technology, Ueberlandstrasse 133, 8600 Duebendorf, Switzerland.
[2]Institute of Biogeochemistry and Pollutant Dynamics, ETH Zurich, 8092 Zurich, Switzerland.
[3]School of Architecture, Civil and Environmental Engineering (ENAC), École Polytechnique Fédérale de Lausanne (EPFL), 1015, Lausanne, Switzerland
[4]Univ. Lille CNRS, UMR 8516 – LASIR, Equipe Physico-Chimie de l'Environnement, F-59000, Lille, France.

1. Introduction

In drinking water treatment, the addition of chemical disinfectants is widely used to serve two main purposes: inactivation of microorganisms during primary treatment and impeding their regrowth in distribution systems (National Research Council (US) Safe Drinking Water Committee 1987; Sedlak and von Gunten 2011). Chemical oxidants are also used for iron and manganese removal (Allard et al. 2013a; Yu et al. 2015), abatement of taste and color (Bruchet et al. 2004) and micropollutants (von Gunten 2003; 2018), improving coagulation and filtration efficiency (Spellman 2013), and preventing algal growth in sedimentation basins and filters (Spellman 2013). Common chemical oxidants are chlorine, chloramine, ozone and chlorine dioxide.

Disinfectants also react with natural organic matter (NOM) to produce undesired disinfection by-products (DBPs) (Sedlak and von Gunten 2011; Xie 2016; Li and Mitch 2018). A group of DBPs known as trihalomethanes (THMs), including chloroform, was discovered in chlorinated drinking water in 1974, first by Rook (1974), and later by Bellar et al. (1974), as analytical techniques for detection became more sensitive and refined (Morris 1975). Since chloroform was first identified, more than 700 other DBPs have been detected in drinking water disinfection (Richardson and Ternes 2018). The majority of known identified DBPs in drinking water are halogenated compounds, as chloramine and especially chlorine are the most commonly used disinfectants in drinking water treatment, leading to direct chlorination or indirectly in the presence of bromide and/or iodide (Hrudey and Charrois 2012). To minimize public exposure to DBPs while maintaining adequate disinfection to control pathogens, several compounds are currently regulated by the US Environmental Protection Agency (four THMs, five haloacetic acids (HAAs), chlorite and bromate), and by the European Union (four THMs and bromate) (Hebert et al. 2010). THMs and HAAs were initially chosen for regulation as they were the first classes identified as well as generally found at the highest concentrations (Hrudey and Charrois 2012). It was assumed that applying conditions to effectively remove or prevent formation of regulated THMs and HAAs would also theoretically limit the occurrence of many other known or unknown unregulated DBPs (Krasner 2009; Hrudey and Charrois 2012; Becker et al. 2013). However, numerous studies suggest that when alternative disinfectants to chlorine are used to minimize the formation of the regulated chlorinated DBPs, this can favor the formation of potentially more toxic emerging DBPs, including iodinated compounds and nitrogenous compounds such as nitrosamines (Krasner 2009). It has been demonstrated for all classes of halogenated DBPs, that iodine-containing species (I-DBPs) are generally the most toxic (i.e., iodoacetic acid is one of the most genotoxic DBPs studied to date while diiodoacetamide is the most cytotoxic on Chinese hamster ovary (CHO) cells (Wagner and Plewa 2017)), followed by bromine-containing and chlorine-containing analogues (Krasner 2012).

Iodinated trihalomethanes (I-THMs) were the first I-DBPs studied due to early concerns about a strong medicinal taste and odor in drinking water (Hansson et al. 1987) and have received more attention subsequently to improve understanding in I-THM formation (Bichsel and von Gunten 2000a; Krasner et al. 2006; Hua and Reckhow 2007a; Jones et al. 2012a; Allard et al. 2015). Until recently, few studies dealt with other classes of I-DBPs due to the lack of commercially available analytical standards for many targeted compounds and the difficulty to find sensitive detection methods for drinking water analysis (Postigo et al. 2017), as many I-DBPs are highly polar and difficult to detect by gas chromatography – mass spectrometry (GC-MS) (Ding and Zhang 2009).

This chapter presents a comprehensive overview of the fate of iodide during oxidative drinking water treatment, the reactivity of the transient hypoiodous acid and the formation of I-DBPs. Furthermore, the main parameters influencing I-DBP formation and possible mitigation strategies are discussed as well.

2. Aqueous iodine chemistry

Iodine is a naturally-occurring halogen, with the Earth's surface containing an estimated 6.3 x 10^{15} kg of iodine (Risher et al. 2004). Iodine exists in many chemical forms and oxidation states, ranging from -1 to +7 (Risher et al. 2004), with iodide (-1) and iodate (+5) the predominant species in an aqueous environment (Preedy et al. 2009). Iodine occurs naturally as the stable isotope (^{127}I) and the radioisotope (^{129}I), additionally 8 other radioisotopes originating from human activities exist (Risher et al. 2004).

2.1. Global cycling of iodine: transport and partitioning

Despite only representing about 1% of the total iodine (Preedy et al. 2009), the ocean is the largest compartment of accessible iodine, with 8.1 x 10^{13} kg of iodine corresponding to an average concentration of between 45 and 60 µg L^{-1} (Risher et al. 2004). By contrast, chlorine and bromine have an average seawater concentration of 19.4 g L^{-1} and 67 mg L^{-1} respectively (Millero et al. 2008). The majority of the iodine present in the Earth's crust is contained in marine and terrestrial sedimentary rocks, with iodine enrichment in sediments mostly related to marine organisms assimilating and accumulating iodine from seawater (Küpper et al. 2011). However, very little (< 0.003‰) of this iodine is actually transferable to other environmental compartments (Risher et al. 2004).

The global cycling of iodine is presented in detail by Fuge and Johnson (2015) in a review of the role of iodine environmental geochemistry on human health. The atmospheric migration of iodine is considered to be the most important part of the geochemical cycle of iodine (Johnson 2003). The following discussion of the iodine cycle will mainly focus on its transport and its role for water resources used for drinking purposes.

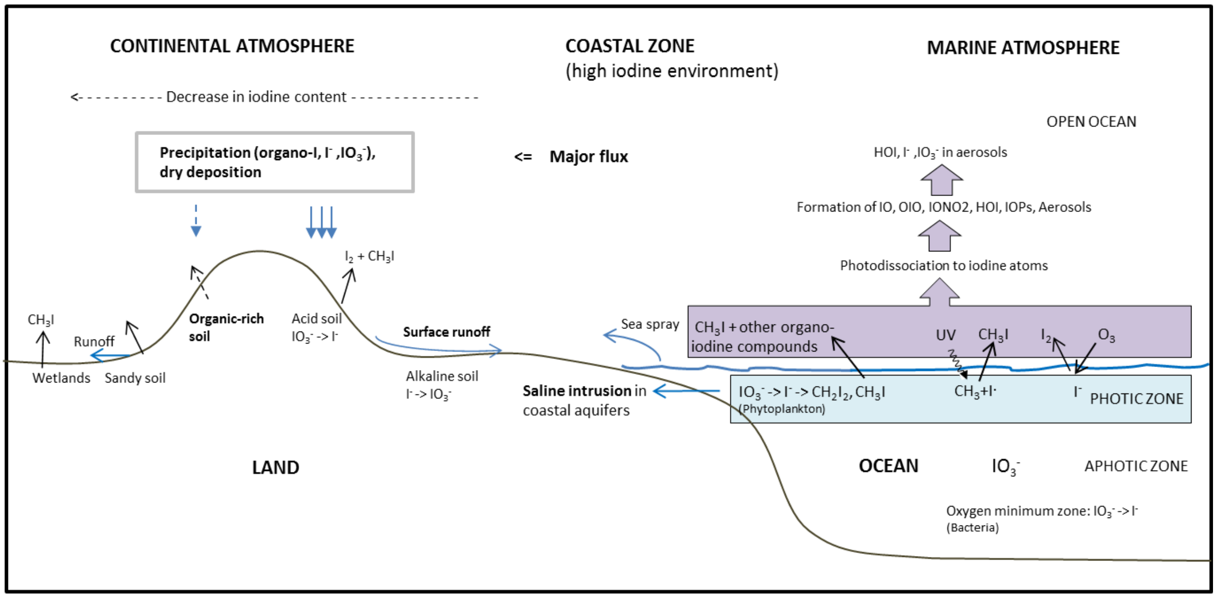

Figure I-1: Schematic representation of the iodine cycle (adapted from Fuge and Johnson (2015)), IOPs: iodine oxide particles, IO_3^-: iodate, I^-: iodide, CH_2I_2: diiodomethane, CH_3I: methyl iodide, $I^·$: iodine atom, I_2, diiodide, O_3: ozone, IO: iodine monoxide, OIO: iodine diioxide, $IONO_2$, HOI: hypoiodous acid, organo-I: organo-iodine, NOM: natural organic matter.

2.1.1. The marine environment: The largest compartment of available iodine

In the open ocean, more than 95% of the dissolved iodine is present as inorganic iodine (mainly iodide and iodate) (Preedy et al. 2009; Fuge and Johnson 2015). Iodate, is the most abundant species (Fuge and Johnson 2015), as it is the thermodynamically stable form in oxygenated, alkaline seawater (Fuge 2013). However surface and shallow shelf waters are enriched in iodide (Fuge

2013), with up to 50% of the dissolved inorganic iodine as iodide in surface seawater (Preedy et al. 2009), due to the conversion of stable iodate to metastable iodide, through the action of phytoplankton, some macroalgae, and possibly through some abiotic processes such as photolysis (Fuge and Johnson 2015). In the oxygen minimum zone, some facultative anaerobe bacteria have been proven to also reduce iodate to iodide (Farrenkopf et al. 1997). Once formed, iodide is slowly re-oxidised to iodate (Fuge 2013), after a first oxidation step of iodide to iodine, mediated by iodide-oxidising bacteria, followed by spontaneous hydrolysis to HOI, which finally disproportionates to iodate (Bichsel and von Gunten 2000b; Amachi 2008).

Unlike in the open ocean, organic iodine may present an important fraction of the dissolved iodine (40-80%) in coastal and shallow shelf waters (Preedy et al. 2009; Fuge and Johnson 2015). Volatile organo-iodine species such as CH_3I can be formed through biological activity (*e.g.* by marine algae, phytoplankton and a variety of aerobic bacteria) (Amachi 2008); CH_2I_2, CH_2ClI can be additionally formed by iodide-oxidising bacteria (Amachi 2008). Iodine is thought to be taken up by a cell wall mechanism based on the haloperoxidase mediated oxidation of I^- to HOI or I_2 which then penetrate through the plasma membrane by a facilitated diffusion mechanism (Amachi et al. 2007; Wever et al. 2018). Organoiodine compounds, mainly iodoform (CHI_3), may also be formed by HOI and I_2 – produced by ozone (see section 2.2.4.) – reacting with dissolved organic matter at the sea surface (Bichsel and von Gunten 2000a; Martino et al. 2009). The oceans provide the main source of iodine to the atmosphere through volatilization of several iodine-containing species, the two most important ones being HOI and I_2 (Legrand et al. 2018), while CH_3I is the main iodine species from terrestrial sources (Fuge and Johnson 2015). A study on Western Europe by Legrand et al. (2018) showed that increasing anthropogenic NOx emissions have led to higher surface ozone concentrations – and thus rising inorganic iodine ocean emissions. While transfer of iodine to the atmosphere through sea spray evaporation is certainly a pathway, the ratio of iodine:chlorine in the atmosphere is about 1000 times higher than in seawater. Thus, an enrichment of atmospheric iodine occurs by biological and chemical processes (Johanson 2000; Risher et al. 2004).

2.1.2. Atmospheric iodine chemistry

All the volatilized compounds photodissociate rapidly in the atmosphere (Figure I-1) – with a lifetime of a few seconds for I_2 to several days for CH_3I – to generate iodine atoms (Sherwen et al. 2016). Subsequently, iodine atoms react mainly with ozone in the lower atmosphere to generate iodine monoxide ($IO^\bullet$) (Preedy et al. 2009; Carpenter 2015). According to a model by Saiz-Lopez et al. (2014), iodine is the second most important ozone-depleting family in the global marine upper troposphere after the HO_x family and in the tropical marine boundary layer after the O_x family. Further reactions of ($IO^\bullet$) with other atmospheric species then form OIO, I_2O_2, HOI and $IONO_2$

(Carpenter 2015). Recent model simulations have shown that HOI is the dominant inorganic iodine species in the free troposphere followed by $IO^{\bullet}$ and $IONO_2$ (Legrand et al. 2018). $IO^{\bullet}$ and OIO can undergo recombination reactions leading to the formation of I_2O_2, I_2O_3 and I_2O_4, which can further react with ozone causing the formation of higher oxides such as I_2O_4 and I_2O_5 (Preedy et al. 2009; Carpenter 2015). Polymerisation of I_2O_3, I_2O_4 and I_2O_5 then lead to the production of new iodine oxide particles (IOPs) in the atmosphere. INO_2, HOI and $IONO_2$ may quickly undergo further gas-phase reactions and aerosol uptake (Preedy et al. 2009; Carpenter 2015).

The gaseous inorganic and particulate forms of iodine are transferred to the land surface through wet and dry deposition (Risher et al. 2004). The iodine content of rainwater is usually in the range 0.5 to 2.5 µg L^{-1} (Fuge and Johnson 2015). A study on Western Europe by Legrand et al. (2018) showed that increasing anthropogenic NOx emissions have promoted a change in the iodine speciation in the troposphere, with an enhanced contribution of the very water-soluble $IONO_2$ species compared to the less soluble HOI, thus leading to rising iodine wet deposition. Rainwater concentrations of iodine are higher near the coast, thus soils in the coastal zone are subjected to a greater input of iodine (Fuge and Johnson 2015). However further inland, there is apparently no significant correlation between the iodine content of the soil and the distance from the sea as this will also depend on total annual rainfall (Johnson 2003) as well as continental sources of moisture which can dominate precipitation contributions (Suess et al. 2019). Such sources have only recently been studied (Suess et al. 2019).

The deposition of iodine will also depend on parameters related to the compounds, such as particle size, concentration, and the chemical form of iodine, with alkyl iodides – such as CH_3I – having a low susceptibility to both wet and dry deposition (Risher et al. 2004). Studies on iodine speciation in rainwater have yielded contrasting results: over the sea or along the coast, iodate has been found as the dominant species in several studies (Truesdalea and Jones 1996; Hou et al. 2009), whereas further inland, organo-iodine can become the most dominant fraction, with lower fractions of iodate than iodide (Gilfedder et al. 2007; Xu et al. 2016; Suess et al. 2019).

2.1.3. Iodine in soil

In a comprehensive database of published iodine results for more than 2000 soils from all over the world, iodine concentrations varied from < 0.1 to up to 660 mg kg^{-1}, with a mean at 3.0 mg kg^{-1} (Johnson 2003; Smyth and Johnson 2011). The input of iodine to soil occurs predominantly through transfer from the atmosphere (Johanson 2000). Weathering of certain soil parent materials – recent marine sediments, or parent materials in areas recently subjected to marine inundation – may sometimes significantly enrich soils in iodine (Fuge and Johnson 2015). The input of iodine to soil also occurs through the decay of vegetation and animal parts.

Soil iodine is mainly in an organic form with variable amounts of iodate and iodide (Fuge and Johnson 2015). After transfer from the atmosphere, dissolved inorganic iodine is rapidly transformed from inorganic to organic forms – through abiotic or biotic processes (Francois 1987; Yamaguchi et al. 2006; Yamaguchi et al. 2010; Zhang et al. 2011; Seki et al. 2013). For example, MnO_2 has been found to oxidise I^- into I_2 and then to IO_3^- or iodinated organic compounds in the presence of humic substances (Gallard et al. 2009; Allard and Gallard 2013).

The ability of the soil to retain iodine, *i.e.*, the iodine fixation potential, is affected by various physico-chemical parameters, including soil type, pH, Eh, salinity, and organic matter content as well as some microbiological processes (Amachi 2008). Organic-rich soils have the highest iodine fixation potential, especially peat soils (Johnson 2003; Fuge and Johnson 2015), and humus may constitute the primary reservoir of iodine in most soils (Shetaya et al. 2012) through binding to aromatic rings (Schlegel et al. 2006). Other components such as iron and aluminium oxides and clay minerals also enhance the iodine fixation potential (Johnson 2003; Fuge and Johnson 2015), due to anion exchange processes on their positively charged adsorption sites (Preedy et al. 2009). Waterlogged soils, such as coastal wetlands, are generally depleted due to desorption from the reducing conditions and are significant sources of iodine emissions (Fuge and Johnson 2015). However, generally the volatilization of iodine from soils neither is an important reaction affecting soil iodine content nor a significant pathway to iodine within the biosphere (Söderlund et al. 2011).

2.1.4. Iodine in terrestrial waters

Most natural surface waters discussed in the literature (Table I-S1) have concentrations between 0.5 and 100 µg L^{-1} and often below 20 µg L^{-1} (Fuge 1989; Takaku et al. 1995; Longvah and Deosthale 1998; Moran et al. 2002; Richardson et al. 2008). From 807 samples taken in streams across Europe, iodide concentrations ranged from < 0.01 to 104 µg L^{-1} with a median of 0.33 µg L^{-1} (Salminen et al. 2005). Central continental areas and mountainous areas in particular tend to have lower concentrations (Day and Powell-Jackson 1972; Aquaron et al. 1993; Sharma et al. 1999), typically in the range 0.1 to 18 µg L^{-1} (Salminen et al. 2005). Compared to surface waters, groundwaters are generally more enriched in iodine. Generally iodine levels in groundwaters are below sea concentrations (Fuge 1989; Neal et al. 2007; Korobova 2010; Voutchkova et al. 2014); in a large Danish assessment (more than 2000 samples), more than 90% of results showed levels below 20 µg L^{-1} (Voutchkova et al. 2014). However, markedly higher iodine contents of drinking water were found in a few locations, with iodine levels in the range of mg L^{-1} (Tang et al. 2013; Zhang et al. 2013), higher than the sea water concentration level between 45 and 60 µg L^{-1} (Risher et al. 2004). Out of 950 samples in the southwestern Taiyuan Basin, China, Tang et al. (2013) found 72% of samples with iodine concentrations above 150 µg L^{-1}.

2.1.4.1. Natural sources

The iodine content of surface waters is influenced by several factors. Stream waters close to the coast have somewhat higher iodine concentrations than those further inland (Fuge and Johnson 2015). The geology of the river basin also plays an important role, with igneous and non-carbonate rocks releasing little iodine while sedimentary and peaty or carbonate rocks enrich streams in iodine (Fuge and Johnson 2015). Iodine is mainly bound with the organic fraction of rocks and soils. During organic matter degradation, iodine is released into the water in dissolved or particulate organic form (Moran et al. 2002). Iodine in groundwaters mainly originates from breakdown of iodine-containing organic matter, leaching from marine sediments in slow-moving aquifers, and evapo-concentration in arid areas (Fuge and Johnson 2015). The main species of iodine in groundwater is generally iodide with a small proportion of iodate (Tang et al. 2013), but in some cases organic iodine can be the main iodine fraction (Zhang et al. 2013).

2.1.4.2. Anthropogenic sources

The iodine content of surface waters is also influenced by anthropogenic activities. Several studies in the US have found that wastewaters from oil and gas extraction by conventional oil and gas wells as well as unconventional shale gas and hydraulic fracturing are characterized by high chloride, bromide and iodide concentrations (up to 56 mg L^{-1}). They may impact drinking water utilities downstream from disposal or accidental spill sites (Harkness et al. 2015). Agricultural practices are another potential source, as additional iodine may be spread on crops and agricultural soils, eventually ending in surface waters, via introduction of pesticides or herbicides containing methyl iodide or via fertilizers with trace levels of iodine (Moran et al. 2002). The use of iodine as a biocide may also contribute to iodine in surface and groundwater (Standing Committee on Biocidal Products 2013). Another source of iodide to surface and groundwaters is the discharge of persistent iodinated pollutants such as iodinated X-ray contrast media (ICM), which are a class of pharmaceuticals used in the medical imaging of soft tissues. Due to poor removal through conventional wastewater treatment – in some cases as low as 10% – ICM concentrations in natural surface waters can exceed 1 µg L^{-1}, especially for iopamidol (Schulz et al. 2008; Duirk et al. 2011; Machek 2015; Xu et al. 2017).

In some cases, iodine may be added directly to water to make it drinkable and/or to relieve endemic goitre in certain regions (Maberly et al. 1981; Vigneri et al. 1993; Smith et al. 2010) – though salt iodization for iodine supplementation to iodine-deficient populations remains the most appropriate and widespread measure to relieve endemic goitre (Eastman and Zimmermann 2000; Bürgi et al. 2001). Iodine-based disinfectants (iodine solutions or iodine resins) have been a popular option for point-of-use drinking water disinfection by campers, the military, and rural consumers in developing countries (Goodyer and Behrens 2000; Smith et al. 2010). However, use of iodination for full-scale water disinfection has been rare, due to the high cost of elemental iodine (Smith et al. 2010).

2.2. Reactivity and fate of iodide during drinking water treatment

2.2.1. Iodide removal

Due to the absence of regulatory limits on bromide and iodide – except in Australia: 0.5 mg L^{-1} for iodide (NHMRC and NRMMC 2011) – little effort has been made to develop drinking water treatment for bromide and/or iodide removal. However, based on the need to mitigate Br/I-DBP formation the potential benefits in lowering bromide and iodide concentrations have been assessed recently. Some processes, which are commonly used in drinking water treatment – while not used specifically for halide removal – may also remove bromide and iodide before a disinfection step. Some recent studies have shown that membrane techniques such as reverse osmosis, nanofiltration, ion exchange membranes, electrodialysis have excellent halide removal efficiencies – with reverse osmosis being the most effective (Watson et al. 2012). Electrolysis, capacitive deionization and membrane capacitive deionization also have good halide removal abilities but are not currently used in drinking water treatment (Watson et al. 2012).

Studies on bromide and/or iodide removal using adsorption techniques (layered double hydroxides, silver impregnated activated carbons and carbon aerogels, ion-exchange resins, aluminium coagulation) – some of which are used in drinking water treatment – showed varied results, as their efficiency was mainly limited by interference from competing anions and NOM present in drinking water sources, causing a lower halide adsorption (Sánchez-Polo et al. 2006; Watson et al. 2012). Because of the high price of bromide/iodide removal technologies, so far these treatment options have not been implemented in drinking water systems.

2.2.2. Chlorine

If iodide-containing waters are treated with chlorine, iodide (I^-) is rapidly oxidised to hypoiodous acid (HOI) or hypoiodite (OI^-) (Eqs. 1 and 2) depending on the pH (pK_a (HOI/OI^-) = 10.4±0.1) (Nagy et al. 1988; Gerritsen and Margerum 1990; Bichsel and von Gunten 1999b, 2000b).

$HOCl + I^- \rightarrow HOI + Cl^-$ (1) *(Table I-1)*

$OCl^- + I^- \rightarrow OI^- + Cl^-$ (2) *(Table I-1)*

This reaction occurs via a Cl^+ transfer and a transient formation of ICl. This intermediate product is unstable and hydrolyses to OI^- (Kumar et al. 1986). During disinfection by chlorine, HOCl (pK_a ($HOCl/OCl^-$) = 7.53, (Martell 1976)) is the dominant reactive species for the reaction with iodide ($k(HOCl + I^-) \geqslant 10^6\ k(OCl^- + I^-)$) (Deborde and von Gunten 2008). For typical concentrations of chlorine, HOCl/OCl^- transforms I^- into HOI/OI^- within < 1 ms in drinking water treatment (Bichsel and von Gunten 1999b).

The further oxidation of HOI by chlorine species occurs through a combination of second and third order reactions (Bichsel and von Gunten 1999b). There are two hypothetical pathways for HOI oxidation by HOCl: (i) The I-atom in HOI is attacked by the O-atom of HOCl, leading to iodite (IO_2^-) formation, which is quickly further oxidised by HOCl to IO_3^-; (ii) HOI is attacked by the Cl-atom of HOCl leading to the formation of an intermediate with a iodine-chlorine bond (HOI-ClOH) which then either dissociates back to the initial compounds or further reacts with a second HOCl to form $HOI(ClOH)_2$ which would quickly hydrolyse to IO_3^- (Bichsel and von Gunten 1999b). Eqs. (3) and (4) show the stoichiometries of these reactions with the corresponding rate constants in Table I-1 (Bichsel and von Gunten 1999b). In these reactions, the formation of the intermediate species is the limiting step with the corresponding second order rate constants.

$2\,HOCl + HOI \rightarrow IO_3^- + 2\,Cl^- + 3H^+$ (3) *(Table I-1)*

$2\,OCl^- + HOI \rightarrow IO_3^- + 2\,Cl^- + H^+$ (4) *(Table I-1)*

Eq. (4) is the dominant reaction, contributing between 60% (pH 7) and 99.5% (pH 9) to the overall HOI oxidation by chlorine (Bichsel and von Gunten 1999b). This process is relatively slow, with a half-life time of 1 hour under typical drinking water treatment conditions (Table I-1).

In the presence of bromide, HOBr and OBr^- (pK_a (HOBr/OBr^-) = 8.8) formed from the reaction between chlorine and bromide are also able to oxidise I^- to HOI or OI^- (Table I-1) through an analogous reaction pathway. The formation of iodate is accelerated due to the relatively high reactivity of HOBr and OBr^- with HOI/OI^- (Table I-1) (Criquet et al. 2012).

2.2.3. Chloramines

Unlike the reaction between monochloramine and bromide, which mainly leads to the formation of a mixed haloamine NHBrCl (Trofe et al. 1980), the reaction product of monochloramine with iodide is assumed to be hypoiodous acid (Eq. (5)) (Bichsel and von Gunten 1999b). NH_2Cl transforms I^- into HOI/OI^- within < 15 minutes under typical drinking water treatment conditions (Bichsel and von Gunten 1999b).

$NH_2Cl + H^+ + I^- + H_2O \rightarrow HOI + Cl^- + NH_4^+$ (5) *(Table I-1)*

The rate constant for the reaction of iodide with dichloramine is much lower than with monochloramine: $k(NHCl_2 + H^+ + I^-) = 9.3 \times 10^5\ M^{-2}s^{-1}$ (Kumar et al. 1986). This difference is explained by the predominant reaction of NH_3Cl^+ in chloramination processes (Heeb et al. 2017) along with the difference of the pK_a of mono- (1.5) and dichloramine (-6.0) (Trogolo et al. 2017). NCl_3 can oxidise I^- with a higher rate constant than the two other chloramines (Kumar et al. 1986), leading to the formation of an intermediate NCl_3I^- (K = 6 x 10^3 M^{-1}), which further decomposes by a first-order process to $HNCl_2$ and ICl (Nagy et al. 1988).

The further oxidation of HOI to iodate by monochloramine could be neglected (Bichsel and von Gunten 1999b) with a low second-order rate constant ($k(NH_2Cl + HOI) < 2 \times 10^{-3}\ M^{-1}s^{-1}$) leading to a long lifetime of HOI and minor iodate formation (Bichsel and von Gunten 1999b) in chloramination processes. Recently, Zhu and Zhang (2016) proposed the formation of the mixed haloamine NHClI ($k = 2.0 \times 10^{-3}\ M^{-1}s^{-1}$) to explain the iodine incorporation into NOM in their model, however this species has not been experimentally confirmed. The proposed NHClI could further react with NH_2Cl leading to the formation of N_2, Cl^-, I^- and H^+ (Zhu and Zhang 2016).

Under acidic conditions, bromamine is capable of reacting with iodide to generate IBr which further reacts with iodide to form I_2 (Prütz et al. 2001). Bromide can react with monochloramine to form bromochloramine (NHBrCl) (Trofe et al. 1980) as well as HOBr, NH_2Br and $NHBr_2$ (Luh and Mariñas 2014). The mechanism for Br^- oxidation by NH_2Cl is complicated (Zhang et al. 2016c). Small amounts of HOBr may also be formed from hydrolysis of chloramines (to HOCl that can then form HOBr) (Zhai et al. 2014) or bromamines (Luh and Mariñas 2014). Bromamines are more reactive than their chloramine analogues, however the reactivity of I^- with bromamines in drinking water treatment has not been reported yet (Heeb et al. 2017).

2.2.4. Ozone

Ozone (O_3) oxidises I^- very rapidly to OI^- in equilibrium with HOI (Table I-1). The reaction occurs by direct transfer of an oxygen atom via an intermediate adduct $I\text{-}OOO^-$ (von Gunten 2003; von Sonntag and von Gunten 2012). O_3 transforms I^- to HOI/OI^- within < 1 ms during drinking water treatment (Bichsel and von Gunten 1999b). Due to the fast direct ozone reaction, the oxidation by the secondarily formed OH radicals can be neglected (Allard et al. 2013b).

$I^- + O_3 \rightarrow OI^- + O_2$ (6) $k(O_3 + I^-) = 2 \times 10^9\ M^{-1}s^{-1}$ (Garland et al. 1980)

In contrast to aqueous chlorine, both HOI and OI^- are quickly further oxidized to iodate by ozone ((Bichsel and von Gunten 1999; von Gunten 2003; von Sonntag and von Gunten 2012), Table I-1). Oxidation by ozone occurs predominantly via HOI for pH ⩽ 8. At pH > 8, the oxidation of OI^- by ozone contributes significantly to the overall oxidation process (Bichsel and von Gunten 1999, von Gunten 2003; von Sonntag and von Gunten 2012).The rate constants of these reactions (Table I-1) lead to half-life times for HOI/OI^- ranging from 0.19 s (2 mg L^{-1} O_3, pH 9) to 3.7 s (0.25 mg L^{-1} O_3, pH 6) (Bichsel and von Gunten 1999b). The overall stoichiometry of the reaction shows a consumption of 3 moles of ozone to oxidise iodide to iodate (Eq. (7)) (Bichsel and von Gunten 1999b).

$3\ O_3 + I^- = IO_3^- + 3\ O_2$ (7)

2.2.5. Chlorine dioxide

The chlorine dioxide reaction with iodide is complex and some questions still remain on the final products (Bichsel 2000). In slightly acidic to neutral solutions, chlorine dioxide (ClO_2) reacts with I^- in a multi-step reaction to form I-atoms and finally I_2 via electron-transfer (Eq. (8)) (Fabian and Gordon 1997). Two moles of chlorine dioxide per mole of iodide are necessary for this reaction (Fukutomi and Gordon 1967; Fabian and Gordon 1997).

A further redox reaction between ClO_2^- and I^- is possible (Eq. (9)) (Kern and Kim 1965), however, in slightly acidic to neutral solutions, the rate constant is several orders of magnitude lower and this reaction can be neglected under drinking water treatment conditions (Fabian and Gordon 1997).

$2\ ClO_2 + 2\ I^- \rightarrow 2\ ClO_2^- + I_2$ (8) *(Table I-1)*

$ClO_2^- + 4\ I^- + 4\ H^+ \rightarrow 2\ I_2 + Cl^- + 2\ H_2O$ (9)

While in one study iodate formation was reported, this only occurred at very low pH (1-3.5) (Lengyel et al. 1996), which is not representative of the pH of drinking waters. This iodate formation at low pH may be explained by iodate formation from chlorous acid oxidation of HOI to iodate (Eq. (10)) (chlorous acid is formed from chlorite at low pH ($pK_a = 1.72$)) (Kern and Kim 1965):

$HClO_2 + HOI \rightarrow HIO_2 + HOCl$ (10) $k(HClO_2 + HOI) = 6 \times 10^7\ M^{-1}s^{-1}$

Chlorite was also shown to oxidise iodine to iodate (Eq. (11)) in the pH range 2-5 (Grant et al. 1982):

$5\ ClO_2^- + 2\ I_2 + 2\ H_2O \rightarrow 5\ Cl^- + 4\ IO_3^- + 4\ H^+$ (11)

Chlorate can also be an important by-product (0–30%) when using chlorine dioxide (Rougé et al. 2018). The chlorate reaction with iodine is not fully understood. While iodide oxidation at neutral pH is low, chlorate has been found to oxidise triiodide to iodate in slightly acidic and neutral media (Mohammad et al. 2010).

Bromide is normally unreactive when applying ClO_2 (Hoigne and Bader 1994), despite that the presence of bromide has been shown to enhance the formation of DBPs and shift the speciation to more brominated species (Yang et al. 2013) including brominated I-THMs (Guo et al. 2014). A significant portion of ClO_2 degrades to free available chlorine (about 20-25% under drinking water conditions) which can oxidize Br^- (Rougé et al. 2018; Terhalle et al. 2018). The reactive bromine and chlorine species can further oxidize iodide/reactive iodine all the way to iodate. However, no substantial iodate formation is expected by this pathway, because the chlorine/bromine formed under these conditions are quickly consumed by natural organic matter.

2.2.6. Ferrate

In recent years, the application of ferrate(VI) has been investigated as a novel oxidant for water treatment (Lee et al. 2005; Lee and von Gunten 2010; Sharma et al. 2015; Gan et al. 2015; Jiang et al. 2016; Zhang et al. 2016b). Ferrate exists in various protonated forms in water ($H_3Fe^{VI}O_4^+$, $H_2Fe^{VI}O_4$, $HFe^{VI}O_4^-$, $Fe^{VI}O_4^{2-}$), with pK_as of 1.6, 3.5 and 7.23 (Sharma 2011). The mono-protonated form – which becomes significant between pH 3 and pH 8 – is more reactive than the di-protonated species and can undergo one-electron or two-electron transfers to oxidise inorganic compounds (Sharma 2011).

The main reaction pathway is the oxidation of iodide to HOI ($k = 2.3 \times 10^4$ $M^{-1}s^{-1}$ (Kralchevska et al. 2016)) at pH 7 and its further oxidation to iodate (Table I-1). Nonetheless, for pH > 9, the HOI disproportionation catalysed by ferrate (VI) becomes an additional transformation pathway to iodate, and is total at pH 12 (Shin et al. 2018). At high pH (9-11), ferrate(IV) decomposition produces hydrogen peroxide (Lee et al. 2014), which reduces HOI, contributing to the I^- regeneration (≤ 22% in the pH range 9-11, with a maximum at pH ≈ 9). The formation of H_2O_2 does not decrease the IO_3^- formation at neutral pH as the reduction of HOI to I^- by H_2O_2 is relatively slow compared to the oxidation of HOI to IO_3^- by Fe(VI) (Shin et al. 2018).

In another study, not only Fe(VI), but also Fe(V) and Fe(IV) were all found to oxidise I^- to I_3^- in the presence of excess I^- (Kralchevska et al. 2016). Fe(V) and Fe(IV), are postulated as intermediates in the decay of Fe(VI) as well as in the oxidation of inorganic and organic substrates (Sharma 2011; Lee et al. 2014). However, while considered more reactive than Fe(VI) (Terryn et al. 2017), Fe(V) and Fe(IV) are unstable species that self-decompose extremely fast, resulting in Fe(III) (Lee et al. 2014; Kralchevska et al. 2016). Fe(V) was only relevant at high concentrations of I^- ($[I^-]/[Fe(VI)] > 1000$), which is never the case under water treatment conditions (Kralchevska et al. 2016).

2.2.7. Permanganate and manganese dioxide

Potassium permanganate ($KMnO_4$), often used as a pre-oxidant can oxidize I^- resulting in mainly HOI at pH > 7.0 or I_2 at low pH (see part 3.1.1.) and in the precipitation of manganese dioxide (MnO_2) (Eq. (12) (Zhao et al. 2016)). MnO_4^- transforms I^- to HOI/OI^- within approximately 3 hours for typical drinking water treatment conditions *(Table I-1)*.

$2\ MnO_4^- + I^- + 2\ H^+ = IO_3^- + 2\ MnO_2 + H_2O$ (at pH 5) (12)

The subsequent formation of iodate occurs via (Eq. 13) direct oxidation of HOI to iodate by permanganate, mainly at pH < 7; (Eq. 14) permanganate-promoted disproportionation at pH > 8 (Zhao et al. 2016):

$Mn(VII)\text{-}HOI + H^+ \rightarrow IO_2^- + Mn\ (V/IV)Re$ $k = 6000$ $M^{-1}s^{-1}$ (pH < 7) (13)

$Mn(VII)\text{-}HOI + OI^- \rightarrow Mn(VII) + I^- + IO_2^-$ $k = 2 \times 10^5\ M^{-1}s^{-1}$ (pH 8-10.5) (14)

It is hypothesized that I_2 – the main product of iodide oxidation at lower pH – is also directly oxidized to iodate by permanganate at lower pH (Zhao et al. 2016).

Manganese(IV) dioxide (MnO_2) can be directly used as a mild oxidant, often for mineral spring waters when strong oxidants are prohibited, or for removal of iron, arsenic or manganese (Gallard et al. 2009; Allard et al. 2010). Manganese(IV) dioxide can oxidise iodide to iodine and iodate (Fox et al. 2009; Allard et al. 2009). In a study on the oxidation of iodide by synthetic birnessite, MnO_2 oxidizes I^- to I_2 (Eq. (15)) for pH < 7.5 (Allard et al. 2009). This reaction follows third order kinetics, second order in I^- and first order in $\delta\text{-}MnO_2$ (Eq. (15)).

$MnO_2 + 2\ I^- + 4\ H^+ \rightarrow Mn^{2+} + I_2 + 2\ H_2O$ (15) $k = 7.2 \times 10^{17}\ M^{-4.3}s^{-1}$

For a high MnO_2/I^- ratio and pH < 6, the formed iodine is rapidly further oxidized to iodate (Eqs. (16) and (17)) (Allard et al. 2009).

$5/2\ MnO_2 + ½\ I_2 + 4\ H^+ \rightarrow 5/2\ Mn^{2+} + IO_3^- + 2\ H_2O$ (16)

Overall reaction:

$3\ MnO_2 + I^- + 6\ H^+ \rightarrow 3\ Mn^{2+} + IO_3^- + 3\ H_2O$ (17)

2.2.8. Other oxides

Different types of oxides are present in distribution systems as pipe deposits because of metal pipe corrosion phenomena, including iron (Sarin et al. 2001; Hassan et al. 2006), copper (Xiao et al. 2007) and lead oxides (Kim and Herrera 2010) as well as other hydroxides and solids of various states. Lin et al. (2008) were the first to report on the reactivity of lead oxide in water using iodide as a probe compound:

$PbO_2 + 3\ I^- + 4\ H^+ \rightarrow Pb^{2+} + I_3^- + 2\ H_2O$ (18) (Lin et al. 2008)

In the presence of the same amount of oxide particles (in the range 45-150 µm), MacAskill (2010) showed that iodide oxidation by lead oxide occurs at a faster rate than iodide oxidation by manganese dioxide. Iron(III) oxide as goethite can also oxidise iodide to HOI (McAskill 2010) but to a lesser extent than lead oxide and manganese dioxide (Zhang et al. 2010b).

Cupric oxide (CuO) – one of the major corrosion products of copper pipes – can activate HOCl, thus enhancing its reactivity toward HOI and increasing the rate of formation of iodate compared to in the absence of cupric oxide (Liu et al. 2014). Iodate can be further oxidized to periodate (IO_4^-) by a CuO-activated hypohalous acid (Liu et al. 2014).

2.2.9. Peroxides

2.2.9.1. Hydrogen peroxide

The reaction between hydrogen peroxide and iodide at low pH is hypothesized to form hydroxyl and iodine atoms (Eq. 19) (Stanisavljev et al. 2011; Milenković and Stanisavljev 2012). Subsequently, the $HO^{\bullet}$ and iodine atoms combine to HOI.

$H_2O_2 + I^- + H^+ \rightarrow HO^{\bullet} + I^{\bullet} + H_2O$ (19) $k = 1.1 \times 10^{-2}\ M^{-1}s^{-1}$ (Deane and Potter 1985)

However, hydrogen peroxide reaction with hypoiodous acid leads back to iodide ions (Valent and Ševčík 1998; Shin et al. 2018). Shin et al. (2018) estimated a second order rate constant of $2.0 \times 10^8\ M^{-1}s^{-1}$ for the fast reaction between HO_2^- and HOI (Eq. 20). Shah et al. (2015) estimated this rate constant to be very fast with a proposed value close to the diffusion control $1 \times 10^{10}\ M^{-1}s^{-1}$.

$HO_2^- + HOI \rightarrow I^- + O_2 + H_2O + H^+$ (20) *(Table I-1)*

2.2.9.2. Peracetic acid

Peracetic acid (PAA) is a mixture of acetic acid and hydrogen peroxide, as a result, while PAA oxidises iodide to hypoiodous acid ((Awad et al. 2003), Eq. 21), H_2O_2 is always present in PAA solutions, thus leading back to iodide ions in a catalytic cycle (Shah et al. 2015).

$CH_3CO_3H + I^- \rightarrow HOI + CH_3CO_2^- + H_2O$ (21) *(Table I-1)*

2.2.9.3. Persulfate

The reaction between persulfate ($S_2O_8^-$) and iodide has been widely studied (Jette and King 1929; King and Jacobs 1931). The stoichiometry for the reaction between persulfate and iodide is 1:2 (Eq. (22)), leading to the formation of I_2 which then hydrolyzes quickly (House 1962). The rate-determining step is a slow initial reaction with the formation of the $IS_2O_8^{3-}$ intermediate (House 1962).

$S_2O_8^- + 2\,I^- \rightarrow 2SO_4^{2-} + I_2$ (22) $k = 1.34 \times 10^8 \exp(-12700/RT)\ M^{-1}s^{-1}$ (House, 1962)

Further reaction by persulfate to iodate is negligible (Dong et al. 2019b).

Recently, a study found that I^- could be rapidly oxidized to HOI, over the pH range 5-9, by the reactive nonradical complexes generated in a peroxydisulfate activation by carbon nanomaterials system. However further transformation of HOI to iodate was negligible (Guan et al. 2017).

2.2.9.4. Peroxymonosulfate

Peroxymonosulfate or PMS (HSO_5^-) also oxidises iodide, forming HOI (Eq. (23)) with an overall stoichiometry of 3 moles of HSO_5^- needed for the oxidation of 1 mole of I^- to 1 mole of IO_3^- (Eq. 24) (Li et al. 2017).

$HSO_5^- + I^- \rightarrow HOI + SO_4^{2-}$ (23) *(Table I-1)*

$3\ HSO_5^- + I^- \rightarrow IO_3^- + 3\ SO_4^{2-} + 3\ H^+$ (24)

2.2.10. UV

Iodide can absorb UV light to form an excited state $I^-_{aq}*$ which may relax back to the ground state or dissociate to the iodine atom and a solvated electron in close proximity (Eq. (25)). These radicals may combine once more in the cage or diffuse into the bulk (Jortner et al. 1961; Jortner et al. 1964).

$I^- + h\nu \rightleftharpoons (I^-*) \rightarrow (I^\bullet + e^-)$ (25) (Jortner et al. 1964)

In the absence of electron scavengers, the back reaction predominates (Rahn 1997). Enhancement of photoxidation of As(III) has been found in the presence of iodide through the formation of iodine atoms (Yeo and Choi 2009).

2.2.11. Radicals

2.2.11.1. Reaction with hydroxyl radicals

The reaction of hydroxyl radicals with iodide is diffusion limited (k = 1.1 x 10^{10} $M^{-1}s^{-1}$, (Table I-1)) and forms the hydroxyl iodide radical anion ($HOI^{-\bullet}$) which decomposes to $I^\bullet$ and HO^- or further reacts with iodide to form $I_2^{-\bullet}$ radical (Yamaguchi 2011). The reactivity with hydroxyl radicals in ozonation processes is however considered to be negligible (Allard et al. 2013b). The further reaction of $OH^\bullet$ with HOI is rather low (k = 7 $M^{-1}s^{-1}$) (Buxton et al. 1988).

2.2.11.2. Reaction with sulfate radicals

While persulfate and peroxymonosulfate (PMS) can oxidise iodide directly, these oxidants can also be activated to produce the sulfate radical ($SO_4^{-\bullet}$), which is a stronger oxidant. Wang et al. (2017) hypothesized that iodide follows the same transformation pathway as bromide when oxidised by sulfate radicals to form radical iodine species ($I^\bullet$, $I^{2-\bullet}$), which couple to each other to form free iodine (HOI, I_2). The second order rate constant of the oxidation of iodide by sulfate radicals (Eq. 26) is not yet known.

$SO_4^{-\bullet} + I^- \rightarrow SO_4^{2-} + I^\bullet$ (26)

2.2.12. Fate of other iodine sources

2.2.12.1. Iodinated X-ray contrast media (ICM)

Increasing interest in iodinated X-ray contrast media (ICM) degradation can be found in the recent literature. ICM can be degraded by direct photolysis (254 nm) leading to the release of I^-, however, not leading to the formation of HOI or IO_3^- (Allard et al. 2016). Irradiation at 254 nm was shown to be quite effective in degrading diatrizoic acid, iotalamic acid, iohexol, iopromide as well as iopamidol (quantum yield 0.034 – 0.071 mol.Einstein^{-1}) (Tian et al. 2014; Allard et al. 2016). Chlorine poorly degrades ICM, except for iopamidol (k(HOCl + iopamidol) = 1.5 x 10^{-2} - 1.66 x 10^{-3} $M^{-1}s^{-1}$ and k(OCl^- + iopamidol) = 0.45 - 0.94 $M^{-1}s^{-1}$ (Wendel et al. 2014; Tian et al. 2017a)) at slightly alkaline pH, releasing iodine mainly oxidised to iodate at the very high chlorine concentrations used in these studies (Duirk et al. 2011; Wendel et al. 2014). Monochloramines did not degrade ICM (Duirk et al. 2011; Wendel et al. 2014). Ozone has also been shown to be able to oxidise several ICM (iopamidol, iopromide, iomeprol, diatrizoate) to some extent, by hydroxide radicals (Huber et al. 2003). However most studies used conditions with very high ozone concentrations (10-16 mg L^{-1}) and degradation is neither rapid nor complete under more typical ozonation conditions, with diatrizoate and ioxitalamic acid being particularly recalcitrant (Ternes et al. 2003; Huber et al. 2003, 2005; Ning and Graham 2008; Kovalova et al. 2013; Magdeburg et al. 2014).

While no significant ICM degradation was observed under PMS oxidation alone, iopamidol was found to be degraded by PMS under catalysis of two pipe corrosion products (CuO and δ-MnO_2) with a higher degradation rate in the CuO/PMS system. After total iopamidol degradation, approximatively half of the initial total organic iodine (TOI) was present in the form of iodate (Hu et al. 2017), which is expected from the potential for an oxidation of iodide to iodate by PMS (see above, Li et al. 2017). More recently in a Co/PMS system, iopamidol and iohexol were also found to deiodinate and ultimately form iodate (Xiaoxiao Wang et al. 2019).

2.2.12.2. Iodate

Iodate is considered non-toxic because it is quickly reduced in vivo to I^- and is therefore the desired sink for iodine during water treatment. In some countries it is even added to table salt to fight against goitre (Bürgi et al. 2001). According to some recent studies, IO_3^- can be phototransformed to I^- during UV irradiation at 254 nm (Eqs. (27-29)) (Zhang et al. 2016c), following pseudo-first order kinetics, with a quantum yield of 0.0591 mol Einstein^{-1}.

$IO_3^-\ (H_2O) + h\nu \rightarrow (IO_3^{-\bullet}\ H_2O)$ (27)

$(IO_3^{-\bullet}\ H_2O) + h\nu \rightarrow OI^- + O_2 + H_2O$ (28)

$OI^- + h\nu \rightarrow I^- + ½\ O_2$ (29)

Tian et al. (2017) found that at pH 7, for an initial iodate concentration of 100 µM (17.5 mg L^{-1}), and UV doses in the range [2754 - 10728] mJ cm^{-2}, k = [1.08 - 3.79] x 10^{-4} s^{-1} which led to up to 46% iodate degradation and 30% conversion to iodide. However, the initial iodate concentration and UV dose used were very high (Tian et al. 2017b). Thus for typical drinking water conditions (40 mJ cm^{-2} (Lyon et al. 2012)), iodate phototransformation – highly correlated to UV fluence rates – will not be important.

Xia et al. (2017) recently found that iodate can also be reduced to iodide in the presence of zero valent iron (ZVI) (Eqs. 30), with an observed rate constant of 0.006 min^{-1} (for 5 g L^{-1} ZVI and 17.5 mg L^{-1} of IO_3^-).

$$IO_3^- + 3(/2)\ Fe^0 + 6\ H^+ \rightleftharpoons I^- + 3\ Fe^{2+}(/2\ Fe^{3+}) + 3\ H_2O \qquad (30)$$

2.2.13. Summary

The reactivity of oxidants towards iodide is summarized in Table I-1, this reactivity increases in the following order: $MnO_2 < MnO_4^- < ClO_2 < NH_2Cl < HFeVIO_4^- < HOCl < HOBr < O_3$. The oxidation of I^- to HOI is rapid for most oxidants (> 10^3 M^{-1}s^{-1} at pH 7), but relatively slow for permanganate. For chlorination, chloramination, ozonation and – with the right conditions – permanganate, HOI is the first oxidation product in oxidative drinking water treatment. In the case of chlorine dioxide, permanganate, ferrate, sulfate radicals, hydrogen peroxide and UV treatment, iodine atoms are formed (Ye et al. 2012; Zhang et al. 2015; Kralchevska et al. 2016).

3. Reactivity of iodine species

In oxidative water treatment, HOI formed from the oxidation of I^- can have various fates depending on the reactivity of hypoiodous acid. The possible reactions of HOI – illustrated in Figure I-2 – are its disproportionation, its oxidation to iodate or the reaction with dissolved organic matter (DOM) present in the water being treated (Bichsel 2000).

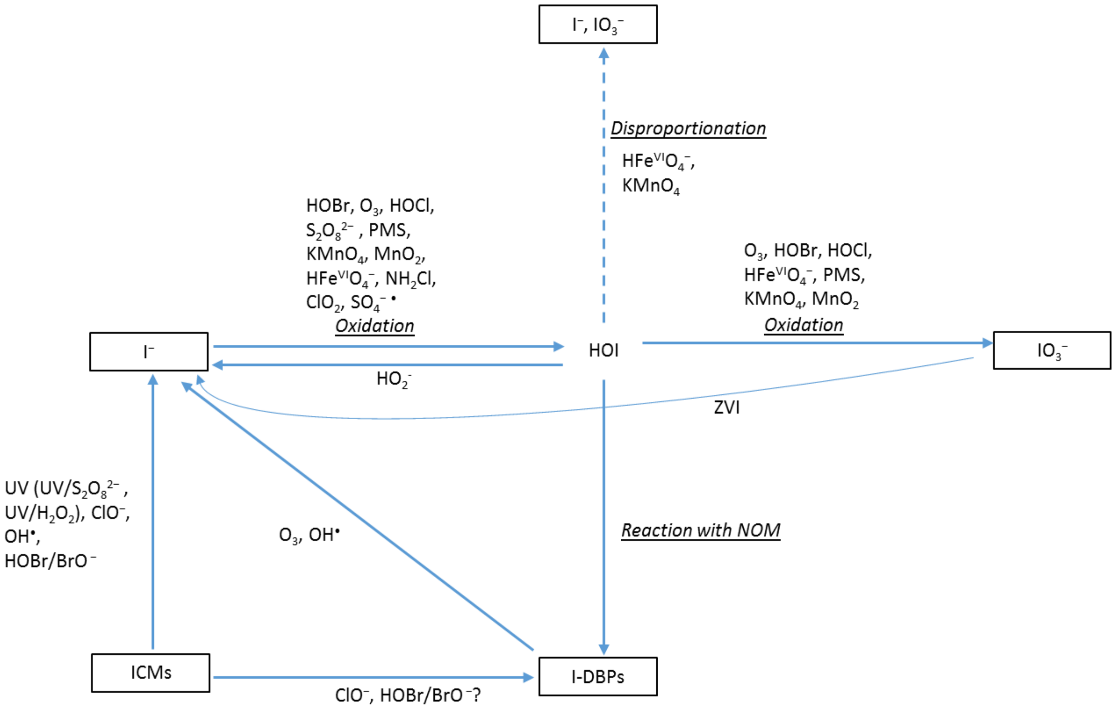

Figure I-2: Scheme of the main reactions involving HOI in drinking water.HOBr/OBr$^-$ reactivity with ICMs is supposed, from the reactivity of OCl$^-$ with ICM.

3.1. Chemistry of aqueous iodine ($HOI/OI^-/I_2/I_3^-$)

3.1.1. Aqueous iodine speciation

Once formed, HOI equilibrates to form I_2, OI^-, I_3^- and H_2OI^+ as described by equations (31-34). At low pH, I_2 is the dominant species (Figure I-3). At high pH, HOI dissociates to hypoiodite OI^- with a $pK_a = 10.4 \pm 0.1$. Therefore, under typical drinking water conditions, HOI is the major species and OI^- is less significant (Bichsel 2000).

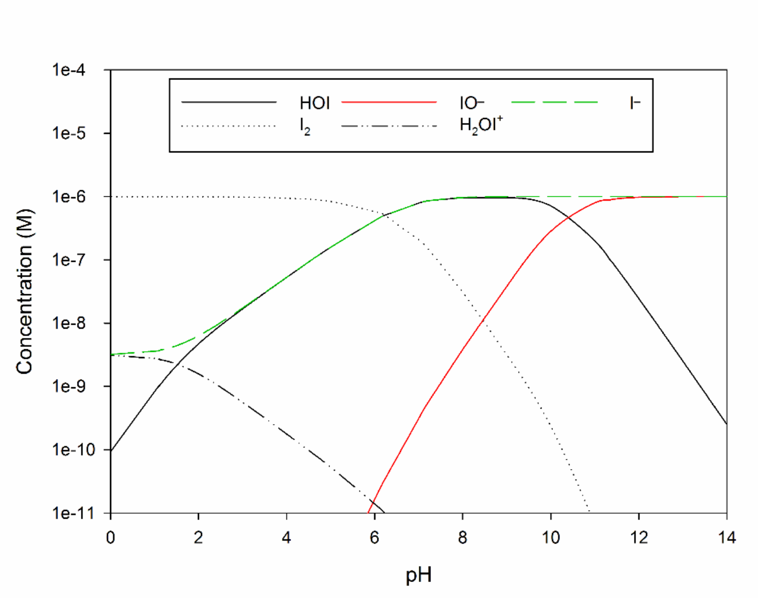

Figure I-3: pH-dependent speciation of iodine (for a total concentration of 1 µM I_2).

$I_2 + H_2O \rightleftharpoons HOI + I^- + H^+$ (31) $K = 3 \times 10^{-13} - 5.4 \times 10^{-13}\ M^2$ (Sigalla 1961)

$HOI \rightleftharpoons OI^- + H^+$ (32) $K = 4.0 \pm 0.7 \times 10^{-11}\ M$ (Bichsel and von Gunten 2000b)

$H_2OI^+ \rightleftharpoons HOI + H^+$ (33) $K = 0.03\ M$ (Belles and Gelles 1951)

I_2 can also be transformed to I_3^- and further polyiodide anions in the presence of increasing concentrations of I^- (Eq. 34 and Figure I-4). These species are both formed in the so called iodometric method used for the determination of various oxidant concentrations, then I_2 could be reduced back to iodide or transformed to I_3^-, which can be directly detected photometrically, due to its high molar absorption coefficient (Bichsel and von Gunten 1999a).

$I^- + I_2 \rightleftharpoons I_3^-$ (34) $K = 698\ M^{-1}$ (Palmer et al. 1984)

HOI can also react with chloride and bromide naturally present in drinking water sources.

$HOI + Cl^- \rightleftharpoons ICl + H_2O$ (35) $K = 22.4 \times 10^4\ M^{-1}$ (Rose and Roberts 2019)

$ICl + Cl^- \rightleftharpoons ICl_2^-$ (36) $K = 77.6\ M^{-1}$ (Rose and Roberts 2019)

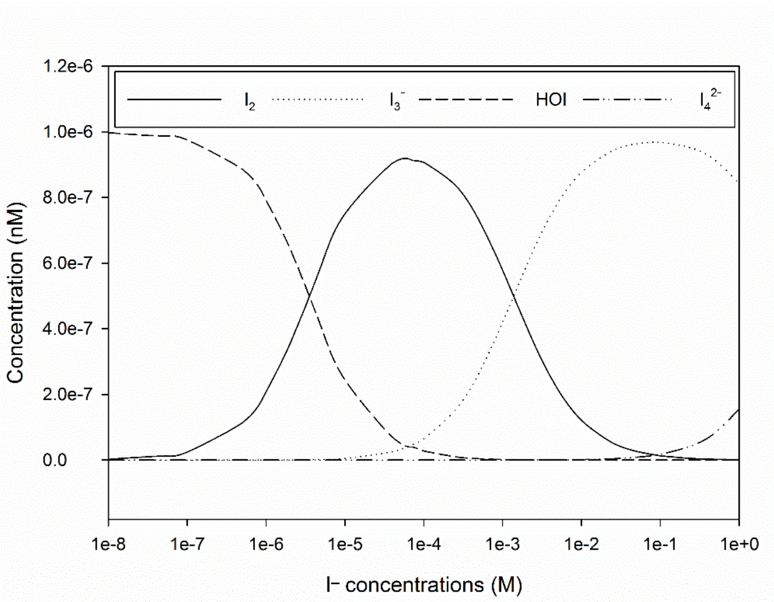

Figure I-4: Speciation of 1 µM of HOI at pH 7 for varying free I^- concentrations.

3.1.2. Disproportionation of hypoiodous acid

The disproportionation of HOI is a reaction in which HOI reacts with itself (Eqs. 37-40). The reaction may be uncatalysed or catalysed by buffers such as phosphate, borate or acetate (Bichsel and von Gunten 2000b).

$HOI + HOI \rightarrow IO_2^- + I^- + 2H^+$ (37) $k(HOI + HOI) = 0.3 \pm 0.2\ M^{-1}s^{-1}$

$HOI + OI^- \rightarrow IO_2^- + I^- + H^+$ (38) $k(HOI + OI^-) = 15 \pm 2\ M^{-1}s^{-1}$

$HOI + IO_2^- \rightarrow IO_3^- + I^- + H^+$ (39) Fast

Overall reaction:

$3HOI \rightarrow IO_3^- + 2I^- + 3H^+$ (40)

Reactions (37) and (38) are rate-determining. Buffer catalyses are extremely important in the disproportionation reactions of iodine and dominate the kinetics of HOI disproportionation (Bichsel 2000; Opoku-Agyeman 2010). The overall reaction remains very slow under typical drinking water conditions (pH 6-8, 0-5 mM carbonate, 1-10 µg L^{-1} iodine), with the initial half-life of HOI ranging from a few days to years. Therefore, IO_3^- formation by this pathway is of minor importance during drinking water treatment (Bichsel and von Gunten 2000b). Thus the fate of iodine will depend on other parameters such as the reaction of HOI with inorganic and organic compounds as well as the oxidants used.

3.2. Hypoiodous acid reactivities with chemical oxidants

The specific reactivity of iodine with various oxidants has been treated in detail in section 2.2., Table I-1 presented here shows a summary of the rate constants and reactivity of oxidants with iodine. The oxidation of HOI is usually slower than the oxidation of I^-. The lower reactivities of $HFeO_4^-$ and HSO_5^- towards iodide than towards HOI – which may be due to charge repulsion – are in contrast to the reactivities of the other oxidants (Shin et al. 2018). From the half-life of reactive iodine species such as HOI in the oxidative treatment will depend the formation of iodinated organic compounds.

Table I-1: Summary of rate constants and half-life times for oxidation of iodide and hypoiodous acid by various oxidants in typical drinking water treatment conditions (pH 7.0 and with an oxidant concentration of 10 µM)

	I^-				HOI/OI^-				
Oxidant	Second order rate constants ($M^{-1}s^{-1}$)	Main Product	Half-life time	Reference	k(HOI)	k(OI^-)	End product	Half-life time	Reference
HOCl	4.3×10^8	HOI	2.1×10^{-4} s[i]	(Nagy et al. 1988) (Gerritsen and Margerum 1990)	8.2	-	IO_3^-	1.1 h	(Bichsel and von Gunten 1999b)
OCl^-	< 30				52	-			
HOBr	5.0×10^9	HOI	1.4×10^{-5} s[ii]	(Troy and Margerum 1991)	Not significant	1.9×10^6	IO_3^-	93 s[ii]	(Criquet et al. 2012)
OBr^-	6.8×10^5				-	1.8×10^3			
HOI	-	-	-	-	0.3	15	IO_3^-	64 h	Bichsel (2000) (Opoku-Agyeman 2010)
					0.6	9.3		32 h	
NH_2Cl	2.4×10^{10} $[H^+]$ $M^{-2}s^{-1}$	HOI	29 s	(Kumar et al. 1986)	$< 2 \times 10^{-3}$	< 3	unknown	8.4 months	(Bichsel and von Gunten 1999b)
$NHCl_2$	9.3×10^5 $[H^+]$ $M^{-2}s^{-1}$	Unknown	8.6 d	(Kumar et al. 1986)	-	-	-	-	-
O_3	1.2×10^9	HOI	5.8×10^{-5} s	(Liu et al. 2001)	3.6×10^4	1.6×10^6	IO_3^-	1.9 s	(Bichsel and von Gunten 1999b)
ClO_2	1.87×10^3	$I^{\cdot}$	37 s	(Fabian and Gordon 1997)	-	-	-	-	-
$HFe^{VI}O_4^-$	4.5×10^4	HOI	2.4 s[iii]	(Shin et al. 2018)	2.5×10^5	2.5×10^5	IO_3^-	0.3 s[iii]	(Shin et al. 2018)
	2.3×10^4	$I^{\cdot} / I_2^{\cdot -}$	4.8 s[iii]	(Kralchevska et al. 2016)	1.8×10^5	-		0.4 s[iii]	(Wang et al. 2018)
$Fe^{VI}O_4^{2-}$	-	-	-		7.7×10^3	-			
$KMnO_4$	Not specified	HOI / I_2	2.8 h[iv]	(Zhao et al. 2016)	Not specified		IO_3^-, I^-	2.8 h[v]	(Zhao et al. 2016)
MnO_2	-	I_2[viii]	-	(Allard et al. 2009)	Not specified		IO_3^-	-	(Allard et al. 2009)
$S_2O_8^{2-}$	8.0×10^5	I_2	8.7×10^{-2} s	(House, 1962)	Negligible		IO_3^-, I^-	-	(Guan et al. 2017)

HSO_5^-	1.41×10^3 1.11×10^3	HOI	49 – 62 s [vi]	(Lente et al. 2009) (Li et al. 2017)	112	1.7×10^6	IO_3^- [vi]	88 s	(Li et al. 2017)
SO_5^{2-}	2.18×10^2 3.0×10^2			(Li et al. 2017) (Lente et al. 2009)	≈ 0	1.5×10^5			
H_2O_2	$4.6 \times 10^{-4} + 7 \times 10^{-3}[H^+]$ $0.011 + 0.168[H^+]$	I^- HOI	2.4 - 58 months [vii]	(Milenković and Stanisavljev 2012)	-	-	-	-	-
HO_2^-	-	-			2.0×10^8 1×10^{10}	-	I^-	0.25 - 13 s [vii]	(Shin et al. 2018) (Shah et al. 2015)
$OH^\bullet$	1.1×10^{10}	$HOI^{\bullet -}$ $I_2^{-\bullet}$	6.3×10^{-6} s	(Buxton et al. 1988)	kapp (pH9) = 7		$IO^\bullet$		(Buxton et al. 1988)
$SO_4^{-\bullet}$	unknown	HOI / I_2	-	(Wang et al. 2017)	Not specified		IO_3^-	-	(Wang et al. 2017)
Peracetic acid	422	HOI	164 s	(Awad et al. 2003)	-		-	-	-

[i] Calculated using pK_a ($HOCl/OCl^-$) = 7.53

[ii] Calculated using pK_a ($HOBr/OBr^-$) = 8.8

[iii] Calculated using pK_a ($HFe^{VI}O_4^-/Fe^{VI}O_4^{2-}$) = 7.23

[iv] Calculated using $k_{app} = 7.0\ M^{-1}s^{-1}$ (Zhao et al. 2016)

[v] Calculated using $k_{app} = 6.9\ M^{-1}s^{-1}$ (Zhao et al. 2016)

[vi] Calculated using pK_a (HSO_5^-/SO_5^{2-}) = 9.23

[vii] Calculated using pK_a (H_2O_2/HO_2^-) = 11.56

[viii] $k_{app} = 71\ M^{-2}s^{-1}$ determined from (Allard et al. 2009)

3.3. Hypoiodous acid reactivities with other inorganic compounds

Table I-2 summarizes the published rate constants for reactions of inorganic compounds with iodine. Unlike HOCl and HOBr, HOI is unreactive toward NH_4^+ (Bichsel and von Gunten 1999b). HOI/OI^- can react with Cl^- and Br^- to form the intermediates ICl and IBr, which rapidly hydrolize back ($2.4 \times 10^6\ s^{-1}$ and $8 \times 10^5\ s^{-1}$, respectively) (Vogt et al. 1999).

Table I-2: Summary of rate constants for hypoiodous acid reaction with various inorganic compounds (not in Table I-1).

Inorganic compounds	Rate constants	Main product	References
Cl^-	$2.9 \times 10^{10}\ M^{-2}s^{-1}$	ICl	(Vogt et al. 1999)
Br^-	$3.3 \times 10^{12}\ M^{-2}s^{-1}$	IBr	(Vogt et al. 1999)
$HClO_2$	$6.0 \times 10^7\ M^{-1}s^{-1}$	HIO_2	(Kern and Kim 1965)
$S_2O_3^{2-}$	$3.2 \pm (0.2) \times 10^{10}\ M^{-1}s^{-1}$	I^-	(Varga et al. 2010)
$S_2O_3^{2-}$	$9.4 \pm (0.6) \times 10^9\ M^{-1}s^{-1}$	$S_2O_3I^-$	(Varga et al. 2010)
$N_2H_5^+$	$4 \times 10^9\ M^{-1}s^{-1}$	I^-	(Deane and Potter 1985)
H_2O_2	$1.1 \times 10^2\ M^{-1}s^{-1}$	I^-	(Deane and Potter 1985)
I^-	$5.0 \times 10^9\ M^{-1}s^{-1}$	I_2OH^-	(Deane and Potter 1985)

Overall, the information on HOI reactivity with inorganic compounds is scarce especially compared to HOCl or HOBr (Heeb et al. 2014). In particular, no specific rate constants are available for the reactions with metals and metalloids of interest such as soluble iron (Fe(II)), manganese (Mn(II)) and arsenic (As(III)), nor with other anionic inorganic compounds such as sulfite, cyanide, nitrite and thiocyanate. However, according to certain authors, HOI reaction with sulfite proceeds rapidly, forming hydrogen sulfate and releasing iodide (Morgan 1954). Knowing the rate constants of I_2 with sulfite (3.1×10^9 $M^{-1}s^{-1}$) and hydrogen sulfite (1.7×10^9 $M^{-1}s^{-1}$) (Yiin and Margerum 1990), and the rate constants of HOCl (7.6×10^8 $M^{-1}s^{-1}$) (Fogelman et al. 1989), and HOBr (5.0×10^9 $M^{-1}s^{-1}$) (Troy and Margerum 1991) with sulfite, the rate constant of HOI with sulfite and hydrogen sulfite can be assumed to be $> 10^9$ $M^{-1}s^{-1}$ (Pechtl et al. 2007; Xu 2014).

3.4. Hypoiodous acid reactivity with organic compounds

The two main types of moieties which have been investigated for HOI reactivity in the literature are phenols (Table I-3) and α-methyl carbonyl compounds (Table I-4). These classes of compounds are known constituents of NOM and have often been used as model moieties to represent substructures within NOM when studying the reactivity of halogenated oxidants (mainly with chlorine and bromine) (Bichsel 2000; Deborde and von Gunten 2008; Heeb et al. 2014; Criquet et al. 2015). Humic substances which contain carboxyl or phenol groups are known precursors of polar I-DBPs such as iodinated acids or iodinated phenols (Ding and Zhang 2009). Warner et al. (2000) studied the interaction kinetics of $I_{2(aq)}$ with substituted phenols and humic substances and found that the rates of iodination of humic substances fell within the range measured for substituted phenols. Recently, several other types of compounds have been studied as well such as carboxylic acids and flavones (Zhao et al. 2017).

Table I-3: Second order rate constants for the reactions of phenolic compounds with HOI.

	Rate constants ($M^{-1}s^{-1}$)				
Compound	**kH$_2$OI$^+$/PhOH**	**kHOI/PhOH**	**kHOI/PhO$^-$**	***kapp***	**Reference**
p-methoxyphenol	(5 ± 1) x 10^4	< 100	(7 ± 2) x 10^8	4.5 x 10^5 *(pH 7)*	(Bichsel and von Gunten 2000a)
p-cresol	(4.0 ± 1.5) x 10^4	(3 ± 1) x 10^2	(7 ± 3) x 10^8	3.8 x 10^5 *(pH 7)*	
p-chlorophenol	(3 ± 1) x 10^3	< 5	(1.6 ± 0.5) x 10^5	6.0 x 10^2 *(pH 7)*	
	-	-	-	(1.9 ± 0.7) x 10^3 *(pH 7)*	(Vikesland et al. 2013)
p-iodophenol	(4 ± 1) x 10^3	(20 ± 8)	(1.5 ± 0.8) x 10^5	9.6 x 10^2 *(pH 7)*	(Bichsel and von Gunten 2000a)
	-	-	-	(9.5 ± 0.6) x 10^2 *(pH 7)*	(Vikesland et al. 2013)
p-cyanophenol	(2 ± 1) x 10^2	(1.5 ± 0.5)	(4 ± 2) x 10^2	5.0 x 10^1 *(pH 7)*	(Bichsel and von Gunten 2000a)
phenol	(5 ± 2) x 10^4	(1.0 ± 0.3) (2.0 x 10^2	(2 ± 1) x 10^6	2.1 x 10^3 *(pH 7)* 2.0 x 10^4 *(pH 8)*	
	-	-	-	(1.5 ± 0.2) x 10^3 *(pH 7)*	(Vikesland et al. 2013)
	-	-	-	(2.1 ±0.2) x 10^3 *(pH 8)*	(Zhao et al. 2017)
	-	-	-	(1.9 ± 0.1) x 10^3 *(pH 7)*	(Li et al. 2018a)
phloroglucinol	-	-	-	(3.1 ±0.5) x 10^7 *(pH 8)*	(Zhao et al. 2017)
resorcinol	-	-	-	(4.2 ±0.3) x 10^6 *(pH 8)*	
4-hydroxybenzoic acid	-	-	-	(7.8 ± 0.1) × 10^3 *(pH 8)*	
4-nitrophenol	-	-	-	(2.6 ± 0.4) × 10 *(pH 8)*	
flavone	-	-	-	(2.5 ± 0.2) × 10^3 *(pH 8)*	
methylparaben	-	-	-	(1.02 ± 0.03) x 10^3 *(pH 7)*	(Li et al. 2018a)
2-bromophenol	-	-	-	(4.8 ± 0.4) x 10^3 *(pH 7)*	
3-bromophenol	-	-	-	(1.54 ± 0.09) x 10^4 *(pH 7)*	
4-bromophenol	-	-	-	(4.0 ± 0.3) x 10^3 *(pH 7)*	
p-hydroxybenzoic acid	-	-	-	(1.6 ± 0.2) x 10^3 *(pH 7)*	
bisphenol A	-	-	-	(8.7 ± 0.5) x 10^4 *(pH 7)*	(Li et al. 2018b)
bisphenol AF	-	-	-	(8.3 ± 0.5) x 10^3 *(pH 7)*	
bisphenol AS	-	-	-	(6.4 ± 0.3) x 10^2 *(pH 7)*	

3.4.1. Phenolic compounds

Free iodine reacts with phenolic compounds via (i) electrophilic aromatic substitution orienting to the ortho and para positions of the hydroxyl group (Wang et al. 2017), or via (ii) electron transfer, leading to the formation of benzoquinones (Zhao et al. 2017). The reactivity of HOI with phenolic moieties varies significantly depending on the nature and location of the substituents on the aromatic ring (Zhao et al. 2017). The rate for the reactions of iodine with phenolic compounds is pH-dependent (Bichsel and von Gunten 2000a): at low pH, the formation of H_2OI^+ has been proposed by protonation of HOI (pK_a =1.4 ± 0.3) (Bell and Gelles 1951; Burger and Liebhafsky 1973; Paquette and Ford 1985), thus its reaction (Eq. 36) would dominate at low pH. However I_2 that is formed at much higher concentration at low pH (Figure I-3) has the same pH dependence. At increasing pH (3.5-5), the reaction involving HOI and the phenol dominates (Eq. 37) and above pH 5, the reaction between HOI and the phenolate form dominates (Eq. 38).

I_2 or H_2OI^+ + (substituted)phenol → products (36) (Bichsel and von Gunten 2000a)

HOI + (substituted)phenol → products (37) (Bichsel and von Gunten 2000a)

HOI + (substituted)phenolate → products (38) (Bichsel and von Gunten 2000a)

A comparison of the species-specific rate constants clearly shows that: $k_{HOI/PhO-} > k_{H2OI+/PhOH} > k_{HOI/PhOH}$
The higher reactivity of H_2OI^+ compared to HOI towards phenol can be explained by the higher electrophilicity of H_2OI^+ compared to HOI, whereas the different reactivities of HOI towards phenol and phenolate can be explained by the increased nucleophilicity of the phenolate compared to the phenol (Bichsel 2000).

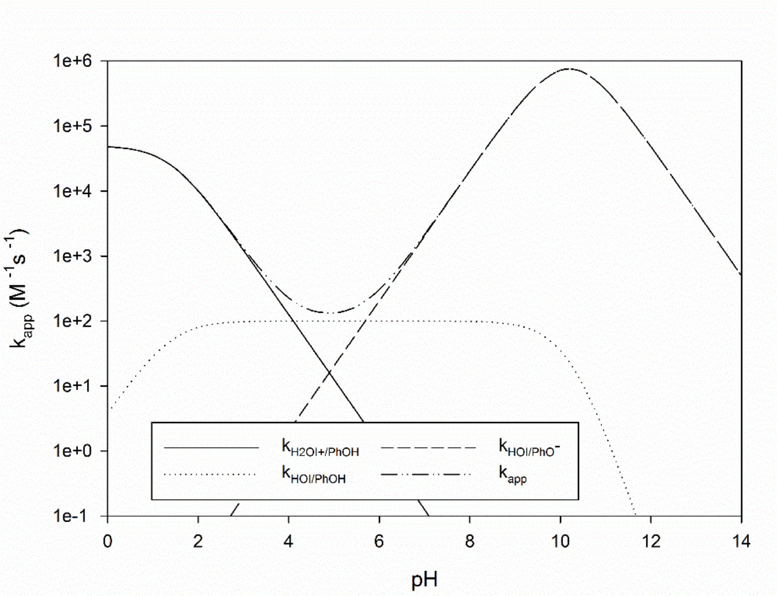

Figure I-5: Modelled apparent second-order rate constants for the reactions of iodine with phenol as a function of pH, considering reactions 36-38 (using data from (Bichsel and von Gunten 2000a)).

According to Figure I-5, the main contribution to phenol transformation above pH 6 is the reaction between HOI and phenolate. The highest second-order rate is observed at pH 10.2, average of the *pKa*s of the two couples involved. Below this pH, the reaction between HOI and the non-dissociated phenol is only significant until pH 4, then the reaction between H_2OI^+ (or I_2) is the main pathway in acidic conditions. However, the impact of buffers should also be considered. Iodination of phenol is a buffer catalysed process (Berliner 1951). Zhao et al. (2017) showed a significant increase of the apparent second-order rate constant of HOI reaction with phenol in the presence of increasing concentrations of phosphate and borate buffers. In addition, a shift of the pH with the maximum apparent second-order rate constant is observed.

3.4.2. Ketones and carboxylic acids

Similarly to chlorine and bromine, 3 HOI molecules react successively with enolized carbonyl compounds to form a triiodomethyl group. At neutral pH, the enolization step is rate-determining and the overall reaction depends only on the carbonyl concentration and the pH (Table I-4). The triiodocarbonyl compound then hydrolyzes with the formation of CHI_3 and the corresponding carboxylic acid (RCOOH) (Bichsel and von Gunten 2000a). Bichsel and von Gunten (2000a) found acetaldehyde to be the most reactive amongst 3 α-methyl carbonyl compounds, and pinacolone the least reactive.

Table I-4: Second order rate constants for the reactions of α-methyl carbonyl compounds with HOI.

α-methyl carbonyl compounds	kH_2O (s^{-1})	kHO^- ($M^{-1}s^{-1}$)	kapp (s^{-1})	References
acetaldehyde	9×10^{-8}	9.0×10^{-1}	4.5×10^{-7} (pH 7)	(Bichsel and von Gunten 2000a)
pinacolone	4×10^{-9}	3.7×10^{-2}	1.9×10^{-8} (pH 7)	
acetone	1.3×10^{-8}	2.5×10^{-1}	1.1×10^{-7} (pH 7)	
3-oxopentanedioic acid	-	-	$(4.0 \pm 0.3) \times 10^{3}$ (pH 8)	(Zhao et al. 2017)
Citric acid	-	-	<1 (pH 8)	

Considering carboxylic acids, their reactivity is low, the rate constant of iodine with citric acid has been found to be < 1 $M^{-1}s^{-1}$ (pH 8) (Zhao et al. 2017). Acetic acid reacts with iodine to form iodoacetic acid (K= 0.24 ± (0.05 M^{-1})) (Urbansky et al. 1997).

3.4.3. Other compounds

Besides phenols and carbonyl compounds, other organic model compounds (allyl alcohol, glucosamine, glycine, toluene, and oxalate) were tested for their reactivity toward HOI. Rate constants were below 1 $M^{-1}s^{-1}$ or even below 0.1 $M^{-1}s^{-1}$ (Bichsel and von Gunten 2000a). Second-order rate constants for the reaction of DOM with HOI have been determined to be in the range

0.1-0.4 $M^{-1}s^{-1}$ on a C-atom base for the DOM in two natural surface waters with DOCs of 1.3 and 3.5 mgC L^{-1} respectively (Bichsel and von Gunten 2000a).

Other compounds are known to react with hypoiodous acid. For example uric acid ($C_5H_4N_4O_3$) which reacts with HOI (1.00×10^6 $M^{-1}s^{-1}$) to form mainly alloxan and urea and iodide (Pejić et al. 2012). Aldoses are also known to be oxidated by HOI to aldonic acids with release of iodide (Ingles and Israel 1948). Oxalate ($HC_2O_4^-$) reacts with HOI releasing carbon dioxide and iodide (Griffith and McKeown 1932).

Overall, the information on HOI reactivity with organic compounds is scarce especially compared to HOCl or HOBr. Nonetheless, HOI second-order rate constants with phenols are lower than those for HOBr (Heeb et al. 2014) but higher than those for HOCl (Deborde and von Gunten 2008).

4. Analytical methods for iodinated disinfection by-product classes

I-DBPs are formed from the reactions of HOI with certain DOM moieties (Figure I-2). First identified in drinking waters in 1976 (Shackelford and Keith 1976), I-THMs were the first I-DBPs studied due to early concerns about medicinal taste and odours in drinking water (Hansson et al. 1987) with the lowest odour threshold at 0.03 µg L^{-1} for iodoform (Cancho et al. 2001). Later on, it was discovered that I-DBPs are more toxic than their Cl^-/Br^- analogues (Wagner and Plewa 2017).

The different I-DBP classes can generally be grouped into two categories: aliphatic and aromatic compounds. To date, the reported aliphatic I-DBPs mainly comprise iodinated trihalomethanes (THMs), iodinated aliphatic acids, and iodinated haloacetamides. The reported aromatic I-DBPs are mainly classified into four groups based on their structures: iodinated phenols, iodinated hydroxybenzaldehydes, iodinated hydroxybenzoic acids, and iodinated nitrophenols (Hu et al. 2018b). DBPs vary in molecular weight, volatility and polarity, therefore, there is no single analytical method capable of extracting and identifying them all (Weinberg 2009). Over the last 25 years, most efforts for new DBP discovery and quantification have used gas chromatography (GC) – mass spectrometry (MS) with electron ionization (EI), largely because of the availability of large mass spectral library databases. This makes the identification of unknown compounds much easier, along with easy-to-spot chromatographic peaks and the absence of matrix effects (Shi et al. 2012; Richardson and Postigo 2016; Richardson and Kimura 2016). Thus a majority of the first I-DBPs species (first I-THMs and more recently I-HAAs) reported in drinking water were quantified by GC-MS, or by gas chromatography - electron capture detection (GC-ECD) (Plewa et al. 2004; Krasner et al. 2006; Hua and Reckhow 2007b; Plewa et al. 2008). The corresponding methods are briefly described below for the different substances.

4.1. Iodinated trihalomethanes

THMs are halogen-substituted single-carbon volatile organic carbons classified as DBPs with the general formula CHX_3, where X represents a halogen such as chlorine, bromine, or iodine (Pérez Pavón et al. 2008). Generally the term THM is associated to the chlorinated and brominated analogues (chloroform ($CHCl_3$), bromodichloromethane ($CHBrCl_2$), dibromochloromethane ($CHBr_2Cl$), and bromoform ($CHBr_3$)) (WHO 2005), which are the most commonly detected DBPs and are regulated worldwide (*e.g.* US EPA: EPA 1998; EU: CELEX-EUR 1998)). The 6 additional I-THMs are: iodoform (CHI_3), chlorodiiodomethane ($CHClI_2$), bromodiiodomethane ($CHBrI_2$), dichloroiodomethane ($CHCl_2I$), dibromoiodomethane ($CHBr_2I$) and bromochloroiodomethane ($CHBrClI$). I-THMs have been the most studied I-DBPs in order to understand their formation (Bichsel 2000; Krasner et al. 2006; Hua and Reckhow 2007a; Jones et al. 2012b; Criquet et al. 2012; Zhang et al. 2016a, b). Amongst I-THMs, iodoform was considered to be the most cytotoxic, while chlorodiiodomethane was considered the only genotoxic I-THM (Richardson et al. 2008).

Low I-THM concentrations require an initial step of extraction and concentration of the analytes before analysis by GC. Headspace, purge and trap, closed-loop stripping analysis and liquid-liquid extraction (LLE) were evaluated as analytical methods for determining I-THMs. It was initially found that the LLE technique, the main technique used for determining I-THMs (Cancho et al. 2000; Weinberg et al. 2002; Ye et al. 2012), was the most sensitive for I-THM quantification, nevertheless, this method requires extensive sample preparation, uses a lot of solvent and is time-consuming. Alternative extraction methods for GC analysis such as solid-phase extraction or microextraction techniques and headspace techniques have rapidly improved in sensitivity for I-THM detection in drinking water and other aqueous solutions (Chinn et al. 2007; Luo et al. 2013) (Cancho and al. 1999; Richardson et al. 2008; Allard et al. 2012; Montesinos and Gallego 2014; Cardador and al. 2015, 2016; Cardador and Gallego 2016; Zhang et al. 2016a).

4.2. Iodinated haloacetic acids

HAAs are halogen-substituted carboxylic acids for which one or more hydrogen atoms attached to carbon atoms have been replaced by a halogen (chlorine, bromine and/or iodine). There are nine different species of brominated or chlorinated HAAs, however, only 5 are regulated by the US EPA (chloroacetic acid $ClCH_2CO_2H$, dichloroacetic acid Cl_2CHCO_2H, trichloroacetic acid Cl_3CCO_2H, bromoacetic acid $BrCH_2CO_2H$ and dibromoacetic acid Br_2CHCO_2H) (EPA 2016) and none in the EU, although all 9 have been cited in a proposal for a revised drinking water directive (The European

Commission 2018). Compared to I-THMs, I-HAAs require an additional methylation step for GC-MS analysis because of their low volatility (Cardador et al. 2008). I-HAAs are even more challenging to analyse than their brominated or chlorinated analogues, as (i) I-HAAs are less stable and (ii) I-HAAs can be present at much lower concentrations in drinking water (Table I-5).

4.3. Other iodinated disinfection by-products

While GC-MS allowed the discovery of the first I-DBPs, the use of GC-MS is limited or of no use when the target DBPs have high molecular masses, are non-volatile or very polar (Shi et al. 2012). Ding and Zhang (2009) developed a precursor ion scan method (UPLC ESI-tqMS) enabling a proposal for the structures of 17 polar I-DBPs (mainly iodoacids). Further studies have used this method to detect and identify polar I-DBPs in simulated drinking water, tap water and chlorinated saline wastewater effluents, with some detection limits reaching less than 1 ng L^{-1} (Yang and Zhang 2013; Gong and Zhang 2015; Pan et al. 2016a, b). Recently electrospray ionization ultrahigh resolution Fourier transform ion cyclotron resonance mass spectrometry interfaced with electrospray ionization (ESI FT-ICR MS) allowed the identification of 206 previously unknown polar I-DBPs in simulated chlorinated/chloraminated drinking water (Wang et al. 2016). Another method using solid-phase extraction-HPLC-MS/MS was developed for 6 polar I-DBPs including iodinated haloacetic acids, phenols and aldehydes.

4.3.1. Iodinated haloacetamides

Haloacetamides (HAcAms) are a class of halogenated nitrogen-containing disinfection by-products (N-DBPs) (Richardson 2008). Several chlorinated and brominated HAcAms were quantified for the first time in drinking water treatment plants during a US DBP occurrence study (Weinberg et al. 2002; Krasner et al. 2006). HAcAm analysis is particularly complex due to their low volatility and mono- and di-halogenated HAcAms having significantly different polarities to their trihalogenated analogues. Thus, a method combining solid-phase extraction enrichment with liquid chromatography triple quadrupole mass spectrometry with atmospheric pressure chemical ionization (SPE-HPLC/tqMS) using selective ion monitoring in the positive mode has been recently used for the analysis of HAcAms, including 4 I-HAcAms (Chu W. et al. 2012) which are classified as both N-DBPs and I-DBPs, with LOQs in the 20 to 50 ng L^{-1} range (Chu et al. 2012; Chu et al. 2016a; Liu et al. 2017). Following this recent discovery, HAcAm formation studies have remained mainly focused on chlorinated and brominated species (Chu et al. 2013; Bond et al. 2015; Chen et al. 2015; Chu et al. 2016b; Sfynia et al. 2017; Zeng et al. 2016). Thus, less is known about their iodinated analogues (Fang et al. 2018), although dihalogenated species are expected to have higher formation compared to tri- or mono-halogenated species, since this is the case for brominated and chlorinated species (Chu et al. 2012; 2013).

4.3.2. Iodinated haloacetaldehydes

Haloacetaldehydes (HALs) constitute the third largest group by weight of identified DBPs. Jeong et al. (2015) were the first to include iodoacetaldehyde as a target analyte using a new GC-MS method with O-(2,3,4,5,6-pentafluorobenzyl)hydroxylamine derivatization. Following studies which also targeted this compound used the same method (Postigo et al. 2017; Postigo al. 2018a).

4.3.3. Iodinated haloacetonitriles

They are an important class of N-DBPs that are nearly ubiquitous in drinking waters (Yu and Reckhow 2015). The few I-HANs that have been studied so far in drinking water have been measured using LLE-GC-MS (Postigo et al. 2018b; Kimura et al. 2019) or LLE-GC-ECD (Ding et al. 2013).

4.3.4. Iodinated acyclic alkanes

A nontargeted screening method involving GC×GC–quadrupole MS (qMS) combined with OECD QSAR Toolbox Ver. 3.2 was able to tentatively identify 1-Iodo-2-methylundecane and 1-Iodo-2,3-epoxypropane in lab scale chlorination and chloramination experiments of raw water.

4.4. Monitoring of total iodine, total organic iodine and iodate

4.4.1. Total iodine

The inductively coupled plasma – mass spectrophotometer (ICP-MS) has superior sensitivity for iodine in comparison to other detection techniques (see Tables I-S3, I-S4, I-S5 in the Supporting Information). For that reason, many studies have utilized ICP-MS in various natural waters to study total iodine.

Total I concentrations have also been determined after the decomposition of dissolved organic matter with chromatographic systems designed to detect inorganic iodine (Zhang et al. 2010a).

4.4.2. Iodate and inorganic iodine

Several methods for the determination of iodide and iodate by ion chromatography have been reported in the literature (Table I-S3). The main advantage of ion chromatography is the possibility of determining both iodide and iodate simultaneously among other ions. While detection can be either amperometric, conductimetric, electrochemical, or spectrophotometric, detection limits are often quite high for iodide, often around 10 µg L^{-1}, and even higher for iodate (Table I-S3). A postcolumn reaction has been found to improve those limits and the selectivity throught the formation of Br_3^- or

I_3^- and the detection by a UV/vis detector (Weinberg and Yamada 1998; Salhi and Von Gunten 1999).

Other chromatographic techniques include HPLC, often after iodate reduction to iodide (Wang et al. 2015), or GC-MS, where iodate is reduced to iodide and iodide derivatized to other organo-iodine forms before analysis (Shin et al. 1996; Mishra et al. 2000; Zhang et al. 2010a). However despite lower detection limits for GC-MS methods, these are time-consuming due to extensive sample pretreatment, and suffer from possible interferences (Zhang et al. 2010a).

Finally, HPLC or IC methods coupled to ICP-MS are usually more sensitive, with detection limits in the ng L^{-1} range, able to analyse iodide and iodate simultaneously, and do not require sample pretreatment for fresh water samples (Table I-S4).

4.4.3. Total organic iodine

Methods for total organic iodine (TOI) measurement have usually involved four major steps: (i) after adsorption of the organic iodine in an acidified water sample onto activated carbon, (ii) pyrolysis of the adsorbed organic iodine to form hydrogen iodide, (iii) absorption of the hydrogen iodide into a solution, (iiii) and finally off-line iodide separation/detection using either ion chromatography (Hua and Reckhow 2005; Kristiana et al. 2009), offline ultra-performance liquid chromatography/ electrospray ionization – mass spectrometry (Ding and Zhang 2009; Pan and Zhang 2013; Gong and Zhang 2013) and more recently offline ICP-MS (Sayess and Reckhow 2017). However these steps are time-consuming and the detection limits usually high, with 0.95 µg L^{-1} as I the lowest detection limit reported by Sayess and Reckhow (2017). Usually TOI can be measured by measuring total iodine and total inorganic iodine (iodide + iodate) separately and substracting inorganic from total organic iodine (Gilfedder et al. 2007, 2010, 2009, 2010).

5. Factors influencing the formation of I-DBPs

5.1. Role of oxidant type, dose and kinetics

5.1.1. Role of oxidant type

Iodate is shown as a major sink for iodide in ozonation, ferrate and PMS processes (Table I-2). Overall, for chlorine, chlorine dioxide, permanganate, and chloramine the formation of I-DBPs can be of concern when treating iodide-containing waters due to the slow oxidation kinetics of HOI. However, during chlorination of bromide-containing waters, the oxidation of HOI to IO_3^- is also significantly enhanced by bromine (Criquet et al. 2012).

Then, the yield of iodinated DBPs for the main oxidants in typical drinking water treatment conditions should generally follow the order summarized in Table I-4.

Table I-4: Relative yield of I-DBPs for the main oxidants in drinking water treatment conditions

High I-DBP yield	Medium I-DBP yield	Low I-DBP yield
Chloramines Chlorine dioxide Persulfate	Permanganate Chlorine	Peroxymonosulfate Ozone Ferrate

This general pattern has been observed experimentally comparing 4 different processes on the same water matrix (Hua et al. 2006).

Surprisingly, in the case of Parsons et al. (2009), the plants using chloramination and those using chlorination in the study did not show significantly different I-THM formation levels and the highest value for the sum of the only two I-THMs measured ($CHCl_2I$ and CHBrClI) was found for a plant using chlorine (3.7 µg L^{-1}). This could be explained by the relatively long period of free chlorine contact time (approx. 30 minutes), before ammonia addition in the case of the plants using chloramination (see part 5.1.3). No information is available either on ammonia content in source waters for the plants using chlorination. Only the plant with a pre-ozonation step showed much lower I-THM production – which is likely due to enhanced iodate formation.

5.1.2. Role of oxidant dose and residence time

Several studies have reported that increased residence time lead to higher I-THM concentrations in chlorinated or chloraminated distribution networks (Serrano et al. 2015; Ioannou et al. 2016). Lower free chlorine residual concentrations will also lead to higher I-THM concentrations (Ioannou et al. 2016). Overall higher doses of chlorine reduce the formation of I-DBPs (Hua et al. 2006).

Increasing contact times will also lead to further transformation of I-DBPs. According to Gong and Zhang (2015), phenols can quickly form 4-iodophenol, and then 4-iodophenol might further transform to 2,4,6-triiodophenol and 2,6-diiodo-4-nitrophenol in the presence of monochloramine (Gong and Zhang 2015).

5.1.3. Timing of ammonia and chlorine addition: (pre)-chlorination vs chloramination

Due to the inability of chloramines to further oxidize HOI to iodate, the highest occurrence of I-DBPs is generally found for this process due to the long lifetime of HOI and the minor iodate formation (Bichsel and von Gunten 1999b) (cf. sections 2.2.3 and 4.). Hence, naturally occuring ammonia, as well as organic amines, in raw water could have an impact on I-DBP formation during chlorination processes (Hansson et al. 1987; Karpel Vel Leitner et al. 1998; Richardson et al. 2008; Allard et al. 2015; Tugulea et al. 2015, 2018). Also, the formation of I-DBPs via chloramination has been shown to be dependent on the timing of ammonia and chlorine additions during the disinfection process. For example, more than 5 µg L^{-1} of CHI_3 were detected in water samples when adding chlorine after ammonia compared to less than 1 µg L^{-1} of CHI_3 when adding chlorine before ammonia (Hansson et al. 1987). When chlorine is added ahead of ammonia it allows a period of contact time with free chlorine, where some iodide is oxidized to iodate, thus reducing the concentration in HOI and the subsequent formation of I-DBPs. This coupled with high levels of naturally-occurring ammonium in some drinking water sources has led to similar levels of I-THMs in drinking water treated by either chloramination or chlorination in several occurrence studies (Goslan et al. 2009; Tugulea et al. 2015). Other reactions with chlorine may also come into play during pre-chlorination: in the case of I-HAcAms for example, interactions with various reactive halogen species may transform HAcAms precursors to non-HAcAms precursors, therefore further limiting I-HAcAms formation during subsequent ammonia addition (Fang et al. 2018).

Therefore, if chloramines are used as a disinfectant it is recommended to increase free chlorine contact times before ammonia addition to reduce iodo-DBP formation as has been reported in several studies (Richardson et al. 2008; Jones et al. 2011; Criquet et al. 2012; Allard et al. 2015). However, the formation of I-DBPs, and especially their speciation, in the pre-chlorination/post-chloramination process depends strongly on the concentration of bromide (Allard et al. 2015). Indeed, while pre-chlorination minimized iodoform formation, short pre-chlorination times form more I-THMs as compared to preformed NH_2Cl due to a large increase in the formation of brominated I-THM species (Jones et al. 2011; Criquet et al. 2012; Allard et al. 2015; Zhang et al. 2016a).

5.2. Total iodine/iodide and bromide concentrations

5.2.1. Iodine concentrations

Generally higher iodide concentrations are associated with higher I-DBP formation by oxidants in terms of concentration and the number of species formed, as shown through several occurrence studies (Richardson et al. 2008) and laboratory studies (Bichsel 2000; Kristiana et al. 2009; Jones et al. 2012a; Zhang et al. 2015). Dillon et al. (2015) state that for chlorination at typical chlorine doses of around 1-3 mg L^{-1} in the UK, 20% of the iodide would form I-DBPs in the water below pH 8. Moreover, the same trend of iodine and bromine incorporation has been observed, ranging from 10 to 40% of halogen incorporated into NOM (Criquet et al. 2015; Zhao et al. 2017) which is usually higher than chlorine substitution. In fact, the bromine substitution has been found to be approximately one order of magnitude higher than chlorine substitution (Westerhoff et al. 2004; Hua et al. 2006). Also, a iodine incorporation from 7.7 up to 18.5% has been found by Hua et al. (2006) depending on the iodide level. The highest iodide concentration showed the highest incorporation due to a lower yield of iodate formation for the same chlorination dose. In this study, the iodine substitution was however much lower than the one of bromine for the same initial bromide and iodide concentrations due to the formation of iodate. In some cases, even low levels of iodide (≤ 1 µg L^{-1}) can yield measurable levels of I-DBPs (Richardson et al. 2008). Generally higher iodide concentrations are also associated with a higher number of species of I-THMs (Zhang et al. 2015). A change in the nature of the dominant species can also occur shifting from less iodinated THMs, such as $CHCl_2I$, to highly iodinated THMs such as CHI_3 with increasing iodide concentration (Bichsel 2000; Zhang et al. 2015). In the case of I-HAAs, higher I^- concentrations lead to an increase in diiodoacetic acid and bromoiodoacetic acid concentrations in chlorinated and chloraminated water samples (Postigo et al. 2017). However, no consistant correlation between total I-DBPs formed and source water iodide concentration appears. Richardson et al. (2008), who only measured $CHCl_2I$ and CHBrClI out of the 6 I-THMs in 23 source waters, found no clear correlation with iodide concentrations despite most of the plants surveyed using the same chloramination process. These results could be explained by the differences in period of free contact time (see part 5.1.3), which varied immensely (< 1 to > 45 min). The high iodide and bromide concentrations could have lead to the formation of substantial amounts of more iodinated and brominated I-THMs, thus perhaps measuring the other I-THMs as well would have led to a better trend. Another explanation is the formation of I-THMs from other sources of iodine (for example, ICM such as iopamidol were later found in some of these source waters with low natural iodide levels). Indeed, Richardson et al. (2008) found that out of the 23 studied source waters, 4 had high (µg L^{-1}) concentrations of iodo-acids and I-THMs despite having low or non-detectable levels of naturally occurring iodide present. The assumption of the presence of ICM has been later observed in different source waters (Duirk et al. 2011; Tugulea et al. 2018). Total iodine measurements (and iodate) in raw waters therefore is also important to follow to also take into account other iodinated precursors.

5.2.2. Bromide concentrations and bromide to iodide ratio

Increasing the bromide concentrations in waters significantly enhances the rate of iodate formation possibly reducing the yield of other I-DBPs (Criquet et al. 2012). It has been shown that HOBr accelerates the oxidation of HOI to iodate during chlorination in a bromide-catalysed process. The rate constants of bromine towards iodine ($k(HOBr/OI^-) = 1.9 \times 10^6\ M^{-1}s^{-1}$ and $k(OBr^-/OI^-) = 1.8 \times 10^3\ M^{-1}s^{-1}$) are much higher than the corresponding rate constants for HOCl ($k(HOCl/OI^-) = 8.2\ M^{-1}s^{-1}$ and $k(OCl^-/HOI) = 52\ M^{-1}s^{-1}$, Table I-1). The effect of the bromide/iodide ratio on the formation and speciation of I-THMs by chlorine followed by ammonia addition showed that $CHCl_2I$ is the main I-THM in the absence of bromide. At a bromide/iodide ratio of 10, mixed Cl-/Br-/I-THMs without any clear dominating species were formed. When the bromide/iodide ratio was increased to 50, Br-/I-THMs clearly became the major species with $CHBr_2I$ being the dominating species (Allard et al. 2015). This change in I-THM speciation is due to the formation of HOBr which is a better substituting agent than chlorine (Westerhoff et al. 2004; Acero et al. 2005). This may lead to higher concentrations of brominated precursors available for mixed I-THM formation (Allard et al. 2015). In addition, increasing bromide concentrations in chloraminated solutions also lead to a significant decrease in the I-THM formation and changes in species distribution due to the formation of bromine (Jones et al. 2012a; Zhang et al. 2016a; Liu et al. 2017).

5.3. Natural organic matter

Natural organic matter (NOM) is the major precursor for the generation of organic DBPs during disinfection. NOM reacts with hypohalous acids (HOX) by electrophilic aromatic substitution and electrophilic addition leading to the incorporation of the halogen atom into the organic matrix or by electron transfer leading to the reduction of the oxidant and the release of the halide ions (Criquet et al. 2015). The specific ultraviolet absorbance at 254 nm (SUVA) can be used as an indicator of aromatic content of NOM – with a high SUVA value generally indicating a high aromatic content (Reckhow et al. 1990), as well as more hydrophobic high molecular weight compounds. In contrast low SUVA values often indicate hydrophilic, low molar mass and low charge density compounds (Hofman-Caris and Hofman 2017). SUVA has been widely used as an indicator of the DBP formation potential of NOM (Hua and Reckhow 2007a; Jones et al. 2012a; Hua et al. 2015; Liu et al. 2017). Liu et al. (2017) showed from the chloramination of model compounds that low SUVA substances favored the formation of I-DBPs compared to high SUVA substances. The same behaviour has been observed with natural waters or NOM extracts in several studies (Hua and Reckhow 2007a; Jones et al. 2012a; Criquet et al. 2015; Liu et al. 2017) based on I-THMs and or other I-DBPs but the opposite was found in other studies (Wang et al. 2014b; Zhang et al. 2016a). However, when the TOI was measured, the high SUVA NOM shows the highest amount of unknown I-DBPs formed (Hua and Reckhow 2007b) or lower iodate formation (Criquet et al. 2012). Consequently, the aromatic fraction seems to be more reactive with iodine to form TOI. Conversely, low SUVA NOM with aliphatic moieties, already containing smaller molecular weight (MW) precursors, could lead to the formation of measurable DBPs such as I-THMs. This behaviour has been hypothesized for brominated DBPs (Hua and Reckhow 2007b). However, these conclusions on I-DBPs are weakened by the fact that iodine incorporation into NOM depends on other factors (*e.g.* bromide concentration, oxidant exposure, iodate formation) that are not always well or easy to set in the different studies with natural waters. One should also be aware that the formation potential test (DBP-FP) is not appropriate in the case of I-DBPs due to the high conversion of iodide into iodate in the presence of such high concentration of chlorine. Hua et al. (2015) showed in a study on different DBPs that

SUVA is a good indicator for the formation of unknown DBPs but the relationships are usually weaker for specific DBPs. In case of iodide, a competition with chlorine occurs either for the formation of iodate or the chlorination of DOM. Indeed at increasing SUVA and/or for high molecular weight fractions, a higher chlorine consumption is expected by direct reaction with the NOM, producing more chlorinated reactive sites and leaving less chlorine available for oxidation of HOI to iodate (Allard et al. 2015; Zhang et al. 2016a). Considering the oxidation state of NOM, its partial pre-oxidation could change the production of DBP, as shown in the case of UV/H_2O_2 pretreatment before chloramination for which the production of I-THM increased, due to the partial oxidation of the DOM, to low aromatic and small MW DOM fractions (Zhang et al. 2018).

Certain metals can bind with functional groups in NOM, affecting NOM reactivity with HOI and thus I-DBP formation. A recent study demonstrated that the complexation of Cu^{2+} with humic acid lead to a decrease in I-THM formation during chlorination. During chloramination low Cu^{2+} leads to some reduction in I-THM formation, but higher metal concentrations lead to some increase (Liu et al. 2019).

5.4. pH

As seen previously, iodine speciation as well as chlorine and chloramine speciation depend on pH. During chloramination, several studies have found that the formation of I-THMs increased with increasing pH in the pH range 6-8.5 for source waters spiked with either iodide or iopamidol (Hua and Reckhow 2008; Duirk et al. 2011). This could be explained by the hydrolysis of monochloramine to chlorine at low pH leading to more iodate formation (Jafvert and Valentine 1992). However, Jones et al. (2012) found the opposite trend explained by the possible oxidation of the NOM by monochloramine hydrolysis and the formation of additional oxidants and substituting agents (HOCl, HOBr, HOI) at lower pH, producing compounds more reactive in forming I-THMs. Similarly, a decrease in the concentration of bromine-containing I-THMs with increasing pH has been observed for all studied MW fractions (Zhang et al. 2016a), and either iodide or iopamidol as iodine source (Wang et al. 2014a).

In contrast, for chlorination, it has been shown that I-THM formation increases for low MW fractions of NOM with increasing pH (Zhang et al. 2016a). However, the lesser amount of HOCl available at higher pH equals to less breakdown of high MW fractions to THM precursors. While the TOI remained stable at higher pH, the fraction corresponding to I-THMs increased (and the other fraction decreased) (Hua and Reckhow 2008). The formation of iodate by chlorination (or bromination) has an optimal pH at the average of chlorine (or bromine) and iodine pK_a values (Criquet et al. 2015). This optimal formation can also vary with the bromide to iodide ratio, chlorine dose and concomitant competition with NOM. In order to assess the different parameters in the formation of iodate vs I-DBPs, kinetic modelling has been performed (see below).

Finally, pH also affects the stability of I-DBPs: similarly to TOCl or TOBr, TOI degradation increases with both pH and temperature, notably due to the dehalogenation of I-DBPs (Rahman 2015; Abusallout et al. 2017). Acidic conditions have been shown to also increase the decomposition of certain I-DBPs such as iodophenols (Hu et al. 2018b).

5.5 Modeling

To assess the simultaneous effect of different parameters on I-DBP formation, kinetic modeling has been performed. In this kinetic model (software Copasi, (Hoops et al. 2006)), 26 reactions (Table I-S8) were used to take into account the formation of different reactive species (chlorine, bromine and iodine) with their speciation with pH, and their reactivity towards phenol as a model compound for organic matter. The reaction rates of iodine with phenol determined by Bichsel and von Gunten (2000a) have been used despite the latter discrepancy. These values allow to take into account the specific rate constants and the evolution of reactivity with pH. A similar behavior can be assumed for NOM with the acid-base speciation of phenolic moieties. Substituted phenol structures are associated with fast reacting NOM moieities controlling the rapid initial HOI consumption phase (Westerhoff et al. 2004). The slower phase for the reactions between HOI and other NOM moieities has not been taken into account in this model. A concentration of 20 µg L^{-1} of iodide has been considered and increasing concentrations of bromide up to 400 µg L^{-1} corresponding to a ratio of 20/1.

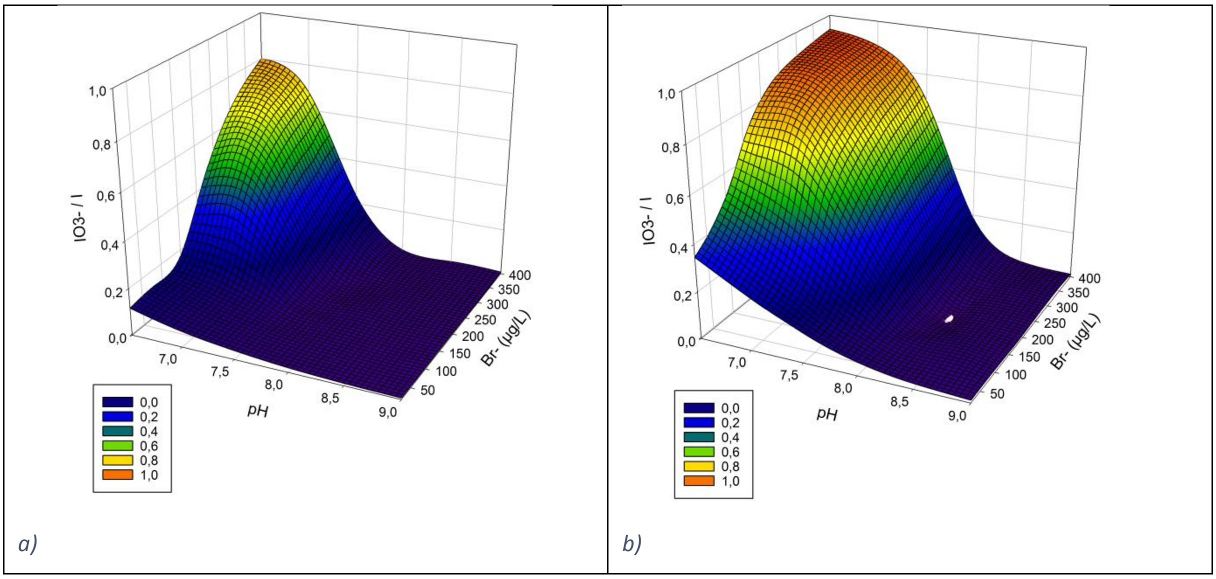

Figure I-6: Modelisation of the effect of pH and bromide concentration on iodide speciation after chlorination of waters with phenol. The results of the model are illustrated in fractions of the initial iodide as iodate. The model was run for a reaction time of 4000 s, with initial concentrations of I^- = 20 µg L^{-1}, PhOH = 10 µM (equivalent to 0.72 mgC L^{-1}, so the number of fast reaction sites roughly matches the concentration of phenolic compounds in natural water), a) HOCl = 15 µM, b) HOCl = 50 µM.

Figure I-6 shows the formation of iodate as a function of the bromide concentration and pH. After the chosen reaction time (4000 s), 90% of iodide is converted either into iodate or incorporated in the organic matter for pH > 7.0. A percentage of 20% of effective incorporation has been considered to mimic the behaviour of chlorine, bromine and iodine reactive species with NOM. 80% of the reaction of iodine reactive species with the organic moietie results in the release of iodide and the oxidation of the organic substance.

According to Figure I-6, at low bromide levels, most of the HOI formed during chlorination is incorporated into the organic moieties as the reactions between HOI and phenolate is orders of magnitude higher than the oxidation of HOI to iodate by chlorine whatever the pH of the solution. At higher bromide levels, the competition with HOBr leads to the formation of brominated phenolic compounds instead. The organic structure becomes limiting, thus the remaining HOI is either rapidly oxidised to iodate by the remaining HOBr, or when HOBr is depleted, slowly oxidised to iodate by HOCl. With an increasing chlorine dose compared to organic compounds, the iodate formation is favored. For example at pH 7, the formation of iodate increases from 6 to 23% of the initial iodide concentration by increasing the chlorine dose from 15 to 50 µM in the absence of bromide, and from 68 to 90% with 400 µg L^{-1} of Br^-. From this modeling, considering the fast reaction step between NOM and the oxidants, it is shown that iodate formation only occurs when the highly reactive sites of NOM are limiting. For pH < 7, HOI has a higher stability due to the depletion of bromine and phenolic structures and the slower oxidation by chlorine.

6. Occurrence of iodinated disinfection by-product classes in drinking water

With the exception of one Spanish survey (Cancho et al. 2000), two large US surveys (Weinberg et al. 2002; Richardson et al. 2008) and one Scottish survey (Goslan et al. 2009), most of the major studies which analysed I-DBPs in drinking water samples were published after 2012. Most of the data published was carried out in the US and Canada, or – since 2012 – in China, and mainly focus on drinking water treatment plant effluents, although some samples were also taken from some distribution networks and consumer taps. Initially the focus was on I-THMs, but more polar I-DBPs, I-HAAs in particular, have been measured as well.

Table I-5: Occurrence of I-DBPs in drinking waters. Publications in order of appearance.

Samples	Water source, country	Disinfectant(s)	I-DBPs analysed/ number of sampling dates (n)	DBPs detected (usually in plant effluent unless specified), concentration ranges and comments	Reference
1 DWTP	Mundaring lake, Australia	Chloramines	CHI_3 n = NS	NH_3/Cl_2: $CHI_3 > 5$ µg L^{-1} Cl_2/NH_3: $CHI_3 < 1$ µg L^{-1}	(Hansson 1987)
1 DWTP	Llobregat River, Spain	Chlorine	6 I-THMs n=21	Σ I-THMs < 1 µg L^{-1} after pre-chlorination and sand filtration, CHBrClI dominated All I-THMs < LODs in DS	(Cancho et al. 2000)
12 DWTPs	SW (10) and GW (2), US	Various, including: chlorine dioxide (3), ozone (3), chlorine (10), chloramines (10)	6 I-THMs n = 4-5	Σ I-THMs: < LOD - 19.0 µg L^{-1} Σ I-THMs in the DS : < LOD - 24.7 µg L^{-1} $CHCl_2I$ dominated	(Weinberg et al. 2002)
1 tap water sample	City of Cincinnati, US	NS	3 iodophenols n = 1	Only 2,4,6-triiodophenol detected (1.40 ng L^{-1})	(Wuilloud et al. 2003)
23 DWTPs	Rivers and GW, US and Canada	Chloramines (21) and chlorine (2)	$CHCl_2I$, CHBrClI, 5 iodoacids n = 2	Σ I-THMs: < 0.09 - 12.3 µg L^{-1} Σ iodoacids: < LOD - 2.04 µg L^{-1} ICH_2CO_2H median: < 0.1 µg L^{-1}	(Richardson et al. 2008)
7 DWTPs	River and reservoir, Scotland	Chlorine (3) and chloramines (4)	$CHCl_2I$, CHBrClI, n = 3 seasons	Σ I-THMs: < LOD - 3.7 µg L^{-1} (max as $CHCl_2I$, with chlorination)	(Goslan et al. 2009)
1 advanced water recycling plant	Secondary treated wastewater, Perth, Australia	Chlorine	6 I-THMs n = 1	$CHBr_2I$ (2 ng L^{-1}) $CHCl_2I$ (8 ng L^{-1})	Allard 2012
7 DWTPs	China	Chlorine (4) and chloramines (3)	4 I-HAcAms n = 3	Only $ClICHCONH_2$ and $BrICHCONH_2$ detected Σ: < LOD - 0.36 µg L^{-1}	(Chu et al. 2012)
70 DWTPs	Rivers, lakes, reservoirs and a few GWs, China	Chlorine (60), chlorine dioxide (3) chloramines (7)	$CHCl_2I$, CHBrClI, $CHBrI_2$, CHI_3, ICH_2CN n = 1	Σ I-THMs: <LOD - 5.58 µg L^{-1} (median < 0.1 µg L^{-1}) $CHCl_2I$ most abundant ICH_2CN: (0,2 - 4,2 µg L^{-1})	(Ding et al. 2013)
1 DWTP	Inland river, Southern China	Chloramines	$CHCl_2I$ n = 2	1.42 µg L^{-1}	(Wei et al. 2013c)
13 DWTPs	Yangtze river and Huangpu river, China	Mainly pre-chlorination and chloramines	CHI_3, ICH_2CO_2H n = 2 seasons January (salt water intrusion period) and July	CHI_3: 0.01 - 1.25 µg L^{-1} ICH_2CO_2H: 0.03 - 1.66 µg L^{-1}	(Wei et al. 2013a)
4 DWTPs	Yangtze River (2), Huangpu river (2), Shanghai, China	NS		ICH_2CO_2H: 0.03 - 2.18 µg L^{-1} CHI_3: < LOD - 0.86 µg L^{-1}	(Wei et al. 2013b)
2 DWTPs	China	Chlorine	4 I-HAAs n = 1	Σ I-HAAs: approx. 2 µg L^{-1} in one plant No I-HAAs detected in the	(Luo et al. 2013)

				other plant (< LOD = 0.5 µg L^{-1})	
2 DWT plants	Qiangwei River, Lianyungang, China	Ozone (1), chlorine (2)	6 I-THMs n = 1	Σ I-THMs: 1.7 and 4.07 µg L^{-1} $CHCl_2I$ most abundant	(Luo et al. 2014)
7 DWT plants	US	Chloramines (5), chlorine (2)	IAL n = 1	0.62 to 4.6 µg L^{-1} in 3/5 chloraminated waters	(Jeong et al. 2015)
1 tap water sample	Boulder Creek, US	Chlorination	$CHCl_2I$, CHBrClI, $CHBr_2I$ n = 10	$CHCl_2I$: 0.05 µg L^{-1} (CHBrClI, $CHBr_2I$) < MDL of 0.02 µg L^{-1}	(Barber et al. 2015)
1 desalination plant	Red Sea coast, Saudi Arabia	Chlorine	6 I-THMs n = 1	0.92 ($CHBr_2I$) 0.58 ($CHBrI_2$)	(Le Roux et al. 2015)
16 DWTPs (10 selected with sodium level > 200 mg L^{-1} and ammonium in the source water)	GW (12), river water (3), lake water (1), Canada	Chlorine	6 I-THMs n = 2 seasons	Σ I-THM: < LOD - 26.82 µg L^{-1} in treated waters. $CHCl_2I$ most frequent, CHI_3 less frequent but had the highest concentrations measured	(Tugulea et al. 2015)
1 DWTP	Guadalmellato Reservoir, Spain	Pre-oxidation (ClO_2), chloramines	$CHCl_2I$, ICH_2CO_2H n = 16 at each sampling location (4 seasons)	In the DS: $CHCl_2I$: 0.07 - 0.33 µg L^{-1} ICH_2CO_2H: 0.10 - 0.70 µg L^{-1} (not detected prior to chloramination)	(Serrano et al. 2015)
9 tap waters	Yangtze River, China	Chlorine	11 polar I-DBPs: 6 iodoacids, 2 IALs, 3 iodophenols n = 2	Σ I-DBPs: 0.45 - 64.85 ng L^{-1} $ClICHCO_2H$ and 3-iodo-4-hydroxy-5-methylbenzoicacid most abundant (ICH_2CO_2H not detected < 0.5 ng L^{-1})	(Pan et al. 2016a)
9 DWTPs	7 GW and 2 SW sources, England and Wales	Chlorine + 2 Sites using pre-ozonation and intermediate ozonation	5 I-THMs (no CHI_3) n = 1	Σ I-THMs: 7.3 - 12.1 µg L^{-1} $CHCl_2I$ dominated	(Goslan 2016)
5 full advanced treatment trains for potable reuse	Secondary or tertiary wastewater effluents, US	Chloramines	6 I-THMs n = 1 or 2	$CHCl_2I$, CHBrClI and $CHBr_2I$ detected in some treatment trains after chloramination followed by microfiltration. None detected in the produced waters (LODs = 0.17-0.21 µg L^{-1})	(Zeng et al. 2016)
Tap samples from 37 individual households (from 2 DS)	SW (dams), Cyprus	Chlorine	$CHCl_2I$, CHBrClI, n = 1	Sum: 0.039 - 1.79 µg L^{-1} $CHCl_2I$ dominant	(Ioannou et al. 2016)
1 DWTP	Creek water near St. Louis, MO, USA	Chlorine	4 I-HAAs	Σ I-HAAs = 1.7 µg L^{-1} ICH_2CO_2H dominant	(Xue et al. 2016)
1 tap water	GW, USA			Σ I-HAAs = 1.7 µg L^{-1} (only ICH_2CO_2H)	

1 conventional DWTP	Pudong, China	Pre-chlorination, chloramines	5 I-THMs (n = 1)	Σ I-THM: 918 (raw water) and 2848 ng L^{-1} (finished water)	(Xu et al. 2017)
1 advanced DWTP	Yangshupu, China	Pre-chlorination, ozone, chloramines		Σ I-THM: 225 (raw water) and 356 ng L^{-1} finished water)	
65 DWTPs	Lake, river, GW, Canada	Post-disinfection agent mainly chlorine or chloramine	6 I-THMs	Σ I-THM: 0.02 - 21.66 μg L^{-1} Median < 1 μg L^{-1} Highest concentrations for waters with naturally occurring ammonium especially after chloramination Overall $CHCl_2I$ most frequent, but CHI_3 had the highest concentrations measured	(Tugulea et al. 2018)
70 water purification plants	Mainly from 4 major rivers, Korea	Pre-/ postchlorination or ozonation/ postchlorination	6 I-THMS (207 samples)	Σ I-THM: 0.04 - 1.59 μg L^{-1} $CHCl_2I$ most frequently detected. Higher concentrations in coastal areas and in pre-/postchlorination facilities	(Woo et al. 2018)
2 DWTPs + 1 reverse osmosis-desalinated water + 3 mixtures between the final waters	Llobregat River, Ter River, and seawater, Spain	Chlorine	6 I-THMs 4 I-HAAs ICH_2CN (high LOQs) (n = 1)	Σ I-THM: 0.18 - 0.31 μg L^{-1}, $CHCl_2I$ dominant Only ICH_2CO_2H found in 1 DWTP and its mixtures, always < 2.5 μg L^{-1} (LOQ)	(Postigo et al. 2018b)
Water from a DWTP just prior to disinfection (Lab study)	NS	Chlorine	6 I-THMs, 4 I-HAAs, IAL	Σ I-THM = 1.3 μg L^{-1} (only $CHClI_2$ and $CHCl_2I$) Σ I-HAAs = 0.4 μg L^{-1} (only ICH_2CO_2H and $ClICHCO_2H$)	(Postigo et al. 2018a)
9 tap waters	Nanjing, China	NS	6 polar I-DBPs (3 iodoacids, 2 iodophenols and 1 iodoaldehyde) (n = 1)	Σ I-DBPs: 1.23 - 5.62 ng L^{-1} ICH_2CO_2H dominant	(Hu et al. 2018a)
22 Samples from a DWTP and consumer taps	NUST University, Islamabad, Pakistan	Chlorine	CHI_3, CH_2ClI	CHI_3: <LOD–0.433 μg L^{-1} CH_2ClI: <LOD–21.5 μg L^{-1} and > 5 μg L^{-1} on 8 sites.	(Khan et al. 2018)
3 DWTPs	Lake Michigan (2), GW (1), US	Chlorine (2), pre-formed chloramine (1)	6 I-THMs, ICH_2CN, $ClICHCONH_2$, $BrICHCONH_2$, $I_2CHCONH_2$	Only $CHCl_2I$, $CHBrClI$, $CHBr_2I$ detected, $CHCl_2I$ dominant Σ I-THM: 0.05 - 0.49 μg L^{-1}	(Kimura et al. 2019)

DS: distribution system; DWTP: drinking water treatment plant; GW: Groundwater; IAL: iodoacetaldehyde; LOD: limit of detection; LOQ: limit of quantification; NS: not specified; SW: surface water.

6.1. Iodinated trihalomethanes

Iodinated trihalomethanes (I-THMs) have been found as DBPs in drinking water in many countries. Furthermore, the total I-THM concentration has even been found to exceed the total regulated THM concentration under specific treatment conditions (Richardson et al. 2008; Tugulea et al. 2015; 2018). However, the ratio I-THM/THM4 was 2% on a median basis for the survey of all 12 plants in the US (Krasner et al. 2006). Indeed, most of the plants had some form of pre-oxidation (mainly by chlorine) before chloramination, which would reduce I-THM formation and increase THM4 formation.

While many laboratory studies on I-THM formation found CHI_3 as the major species, these were often not conducted at typical Br^-/I^- mass ratios. CHI_3 is the least occurring I-THM in most real water samples (Table I-5). However, CHI_3 has been recently reported as the main I-THM in several distributed waters ((Wei et al. 2013a; Tugulea et al. 2015, 2018), Table I-5) usually accompanied by high total I-THM concentrations. The most frequently quantified I-THMs are usually $CHCl_2I$ followed by CHBrClI (Table I-5) with the highest measured concentrations for $CHCl_2I$ and CHBrClI being 15 and 6 µg L^{-1} respectively (in distribution systems), in a treatment plant in the US which added chlorine and ammonia simultaneously to form chloramines (Weinberg et al. 2002). At this plant, I-THMs were nearly as present as regulated THMs (81% mass ratio). Furthermore, plants using source waters with naturally occurring ammonia had higher average and median I-THM values, regardless of the use of chlorine or chloramines as secondary disinfectant (Tugulea et al. 2015).

The largest surveys focusing mainly on chlorination were conducted in China (Ding et al. 2013), Korea (Woo et al. 2018), the UK (Goslan 2016), and Canada (Tugulea 2018). Amongst the data presented by these authors in Table I-5, the sum of 5 I-THMs measured by Goslan (2016) may appear excessively high, as they found values in the 7.3-12.1 µg L^{-1} range for 9 plants using chlorination. While the ammonia content of source waters was not measured, and some chloramination may have taken place, 2 of the plants used both pre- and intermediate-ozonation, thus one would not expect such a high formation of I-THMs after final chlorination. Additionally, not only did the final I-THM concentrations systematically exceed the initial iodide concentrations of source waters, but in some cases they even exceeded the total iodine concentrations of source waters. The authors hypothesized that possible coelution may have lead to overestimation of certain I-THM peaks.

Parsons et al. (2009) and Tugulea et al. (2018) found a tendency for more I-THMs during summer than during winter, and this was linked to rising iodide levels during summer. Weinberg et al. (2002), also showed some plant-specific seasonal trend with more I-THMs formed in warmer waters.

But the lack of data on iodide levels in source waters makes it impossible to distinguish possible seasonal effects due to variations in temperature and due to variations in iodide levels.

It is worth mentioning that, while I-THM levels experienced mostly unsignificant variations along the distribution network in the study by Tugulea et al. (2018), the highest I-THM sum measured (21.66 µg L^{-1}), mid-point through the distribution, was drastically higher than for the chloraminated treated sample (4.25 µg L^{-1}), due to some unoxidized inorganic iodide still remaining after treatment (the iodide concentration in the source water was extremely high, at 131 µg L^{-1}). Therefore for processes using chloramination as final disinfection, monitoring not only the iodide level in source water, but the remaining iodide concentration in treated water as well could be important to evaluate the risk of further I-DBP formation in the network.

Overall it remains difficult to compare occurrence and speciation of I-THMs between sites (Table I-5), because many of the early studies did not measure all 6 I-THMs. Additionally, many of the parameters which greatly influence their formation were not systematically measured (bromide, iodide, total iodine and ammonia content of raw water and/or free chlorine contact time/ order of ammonia and chlorine addition and/or free chlorine residual concentration…).

Other iodinated halomethanes have also been detected (although not necessarily quantified) in water: chloroiodomethane was first reported in drinking water samples in the UK in the mid-70's, but has not been often monitored since then (Rumsby et al. 2009). Chloroiodomethane has been detected both in chlorinated waters (Khan et al. 2018) and in lab chloraminated source waters (Postigo et al. 2016). Bromoiodomethane and diiodomethane were also identified in chloraminated water in 1995 (Symons et al. 1998). In lab-scale chlorination and chloramination reactions performed on NOM, diiodomethane and iodomethane were also qualitatively identified (Cojocariu et al. 2016). One should notice that diiodomethane and iodomethane have also been identified as thermal degradation products of iodoform during solid-phase micro-extraction (SPME) (Frazey et al. 1998). Several other iodinated dihalomethanes (including chloroiodomethane, bromoidomethane and diiodomethane) have been identified as degradation products of I-THMs in spiked river water (Cancho et al. 2000). This may perhaps explain the extremely high value for chloroiodomethane found to exceed 20 µg L^{-1} at one sampling point in a chlorinated distribution network of Islamabad (Khan et al. 2018), as they used HS-SPME. No study has yet looked at possible thermal degradation of chlorodiiodomethane or chloroiodoacetic acid to chloroiodomethane during HS and gas chromatography. Nonetheless, Chinn et al. (2007) have demonstrated loss of iodinated THMs at higher injection port temperatures.

6.2. Iodinated haloacetic acids

Four iodinated haloacetic acids (I-HAAs) were found in different studies: (mono)iodoacetic acid (ICH_2CO_2H), chloroiodoacetic acid ($ClICHCO_2H$), bromoiodoacetic acid ($BrICHCO_2H$), and diiodoacetic acid (I_2CHCO_2H) in chlorinated and chloraminated drinking waters (Plewa et al. 2004; Krasner et al. 2006; Richardson et al. 2008) at low ng L^{-1} to low µg L^{-1} levels with maximum concentrations of 1.7 µg L^{-1} for iodoacetic acid and approximately 0.5 µg L^{-1} for chloro- and bromoiodoacetic acid. Diiodoacetic acid has so far only been quantified below 1 ng L^{-1} in drinking waters. Triiodoacetic acid has been reported in several bench scale formation studies. However, trihalo-HAAs generally decompose very fast in drinking water under ambient temperature (Zhang and Minear 2002), and it is unlikely to find them in drinking water. The most frequently monitored I-HAA, until now, has been ICH_2CO_2H (Table I-5).

Other iodoacids (Z)-3-bromo-3-iodopropenoic acid, (E)-3-bromo-3-iodopropenoic acid and (E)-2-iodo-3-methylbutenedioic acid were also identified in chloraminated drinking water in a U.S. nationwide DBPs occurrence study (Weinberg et al. 2002; Krasner et al. 2006). Richardson et al. (2008) found concentrations < 1 µg L^{-1} amongst 23 drinking water treatment plants (Table I-5) for these compounds. Another iodoacid, 3,5-diiodosalicylic acid was found at concentrations in the 0.17-1.48 ng L^{-1} range in 9 tap waters in China (Hu et al. 2018a).

Iodoacids were found in finished drinking waters which also had relatively high levels of I-THMs (Weinberg et al. 2002; Wei et al. 2013a; Wei et al. 2013b).

6.3. Other iodinated disinfection by-products

6.3.1. Iodinated haloacetaldehydes and other iodinated aldehydes

Iodobutanal was the first iodinated haloaldehyde (I-HAL) identified – but not quantified – in US surveys (Weinberg et al. 2002; Krasner et al. 2006). Iodoacetaldehyde was then quantified in 4 waters from US drinking water plants at 0.62 to 4.5 µg L^{-1}, even with iodide below the detection limit (5 µg L^{-1}) in the source waters, but only for chloraminated waters (Jeong et al. 2015). While iodoacetaldehyde has not been further investigated in drinking water since then, disinfection experiments directly on river water have shown its formation is possible after chlorination as well (Postigo et al. 2017).

6.3.2. Iodophenols

Various types of iodophenols have been quantified in tap water including iodinated hydroxybenzaldehydes, iodinated hydroxybenzoic acids, and iodinated nitrophenols (2-iodophenol; 4-iodophenol; 2,4,6-triiodophenol; 3-iodo-4-hydroxy-5-methylbenzoic acid; 2,6-diiodo-4-nitrophenol;

2,4,6-triiodophenol, 2,4-diiodo-6-nitrophenol, and 3,5-diiodo-4-hydroxybenzaldehyde) (Wuilloud et al. 2003; Pan et al. 2016a; Hu et al. 2018a). Individual concentrations were below 1 ng L^{-1} in 18 Chinese tap waters (Pan et al. 2016a; Hu et al. 2018a), except for the two nitrogenous phenolic I-DBPs in 2 samples with high amounts of dissolved organic nitrogen, with a maximum of 24 ng L^{-1} for 2,4-diiodo-6-nitrophenol (Pan et al. 2016a).

6.3.3. Iodinated haloacetonitriles

Only one survey has actually monitored iodinated haloacetonitriles (I-HANs) in drinking waters: Ding et al. (2013) measured iodoacetonitrile (IAN), which was found (0.2-4.2 µg L^{-1}) in only 4 out of 60 DWTPs using chlorine in China (Ding et al. 2013). For the two highest IAN values, iodoacetonitrile was the main HAN measured (6 other chlorinated and brominated HANs were measured as well), while no I-THMs were measured. Surprisingly, iodoacetonitrile was not detected in the chloraminated samples, although the number (7) of treatment plants sampled was smaller.

6.3.4. Iodinated haloacetamides

3 out of 7 samples in China had chloroiodoacetamide (<0.04 - 8.5 µg L^{-1}) and 3 out of 7 samples had bromoiodoacetamide (<0.05 - 6.1 µg L^{-1}) (Chu et al. 2012).

6.3.5. Iodinated trihalomethanols

They have yet to be reported in drinking waters, but some have been proposed (not confirmed) in chlorinated saline primary effluent such as chlorobromoiodomethanol and chlorodiiodomethanol (Gong and Zhang 2015), and in simulated ClO_2-treated drinking water such as iododichloromethanol, chlorobromoiodomethanol and chlorodiiodomethanol (Han et al. 2017).

6.3.6. Iodinated dipeptides

4 Iodinated dipeptides (3-I-/3,5-di-I-Tyr-Ala and 3-I-/3,5-di-I-Tyr-Gly) have been identified in tap water samples (Huang et al. 2018).

7. Risk assessment and mitigation

7. 1. Toxicity of iodinated disinfection by-products

Although toxicity evidence from in vivo assays and molecular epidemiology studies for I-DBPs is still lacking (Cortés and Marcos 2018), different cytotoxicity and genotoxicity assays have been performed on various I-DBPs in the literature (Tables I-S6 and I-S7). Iodoacetic acid is the I-DBP which has been subjected to the most number of different assays, allowing a comparison of hamster

(CHO-AS52 and CHO-K1), mouse (NIH3T3), and human (HepG2, Caco-2, and CCD 841 CoN) cell lines. However most of the other I-DBPs have only been subjected to quantitative cytotoxicity assays and genotoxicity SCGE or comet assays using a Chinese hamster ovary (CHO) cell platform. Assessing individual cytotoxicity and genotoxicity of I-DBPs may underestimate possible interactions among them and with the organic matter present in the water. Additionally, many I-DBPs remain unindentified. However due to the lack of quantitative comparative toxicity data on most of the known I-DBPs, these toxicity data aid in prioritizing DBPs and their related compounds for future in vivo toxicological studies and risk assessments.

A recent review compiled for 103 DBPs, their lowest cytotoxic concentration, their LC_{50} value (lethal concentration for 50% of a population), their lowest concentration that induced a genotoxic response and the 50% TDNA (intensity of the DNA that migrated away from the nucleus) or midpoint of Tail moment (integrated value of migrated DNA density multiplied by the migration distance) with CHO cells (Wagner and Plewa 2017), and demonstrated that I-DBPs are more cytotoxic and genotoxic than their brominated and chlorinated analogues. A more detailed discussion on the toxicity of each class of I-DBPs and a comparison to their brominated and chlorinated analogues can be found in a recent article by Dong et al. (2019a), in which the authors used mammalian cell cytotoxicity and genotoxicity data to provide a rank ordering of the relative toxicities of I-DBPs.

7. 2. Prioritization of I-DBPs

To prioritize DBP classes, a new metric called "TIC-Tox" (Plewa et al. 2017), compares measured DBP concentrations – or semi-quantitative TIC peak areas – weighted by metrics of toxic potency (cytotoxicity or genotoxicity index value) to determine chemical drivers of overall toxicity in water (Jeong et al. 2012; Allard et al. 2015; Li and Mitch 2018; Chuang et al. 2019; Dong et al. 2019 a). From this methodology, iodoacetaldehyde, iodoacetonitrile as well as iodoacetic acid could be important DBPs due to their high toxicities and because they have already been found at the µg L^{-1} level. In the case of iodophenols and iodoacetamides, both classes of compounds have reportedly high toxicity – especially the three iodoacetamides (iodoacetamide, bromoiodoacetamide, chloroiodoacetamide), more cytotoxic I-DBPs than the I-HAAs with the exception of iodoacetic acid (Sayess et al. 2017) – but have yet to be quantified at concentrations above dozens of ng L^{-1} in treated waters. There are some I-DBPs with no toxicity data as of yet, and more occurrence data is needed on I-DBPs in real drinking water samples so as to better prioritise I-DBPs. Such prioritization can provide useful information to develop control strategies for the minimization of human health risk resulting from the exposure of the consumer to I-DBPs, including by the establishment of new threshold values.

An indexing method based on 3 criteria (concentration level, reported frequency and toxicity) was recently developed by Mian et al. (2018) to prioritize unregulated DBPs in three categories (i.e., critical, medium, and low priorities). The I-DBP group was categorized as of low priority compared to other DBP classes. However only iodoacetic acid and iodoform were considered. An earlier paper (Hebert et al. 2010) gave scores to DBPs based on 4 criteria (toxicity, occurrence, US-EPA and/or IARC classification, and regulation). This procedure allowed the identification and ranking of three different groups of emerging DBPs: Dichloroidomethane and iodoacetonitrile were ranked in Group I, which consists of the most critical emerging DBPs with regard to their potential health effects (moderate occurrence but the highest toxicity). Other I-THMs as well as I-acids and I-HAcAms were categorized in Group II (moderate to elevated occurrence and relevant toxicity).

7. 3. Mitigation strategies

Precursor removal prior to disinfection is an attractive strategy for DBP minimisation, in that it non-specifically minimises all DBPs, both regulated and emerging (Watson et al. 2012). As NOM is the major precursor for the generation of DBPs, NOM removal prior to disinfection is extremely important. When focusing on I-DBPs, there may be however a need for effective bromide and iodide removal in the context of more and more complicated and stringent regulations as well as the increase in the use of salinity impacted waters for drinking water purposes. However, current drinking water treatment schemes are challenged to effectively remove halides (bromide and iodide) from drinking water sources before final disinfection (Watson et al. 2012).

Considering iodine, the transformation into iodate, a safe end-product could be a good alternative. A pre-ozonation step before post-chlorination or especially post-chloramination to completely oxidize iodide to iodate is indeed an efficient process to avoid the formation of I-DBPs (Allard et al. 2013). Also iodide oxidation by ozone is sufficiently fast to avoid excessive formation of bromate, a regulated DBP linked to ozonation (Allard et al. 2013). A higher ozone dose was required for waters with high SUVA254 values (Gruchlik et al. 2015). As already stated in part 5.1.3., adequate prechloration before chloramination can also form more iodate and reduce the subsequent formation of I-DBPs.

One should pay attention however on the oxidant type used, for example hydrogen peroxide exhibited no appreciable effect on I-THM formation during chloramination (Jones 2009), and led to I-HAcAm formation during chlorination (Chu et al. 2014). This oxidant is therefore not suitable as a pre-oxidant for I-DBP minimisation. Further research should be done on the use of potassium permanganate as a pre-oxidant. Indeed, while potassium permanganate as a pre-oxidant before chloramination was shown to increase I-THM formation (Jones 2009), the pH was not controlled during this experiment. At more acidic or basic pH, iodide is rapidly oxidized ($pH < 6$) or disproportionated (pH > 8) to iodate (see 2.2.7.), therefore one would expect permanganate to reduce I-DBP formation during subsequent chloramination. Chlorine dioxide as a pre-oxidant has been shown to reduce I-THM formation for high bromide and iodide concentrations (800 µg L^{-1} / 80 µg L^{-1}), but to increase I-THM formation for lower bromide and iodide levels (200 µg L^{-1} / 20 µg L^{-1}) (Jones 2009). Ferrate pre-oxidation is also very effective in minimizing I-DBPs formation before chloramination and even before chlorination, by oxidising iodide to iodate (Zhang et al. 2016b).

8. Conclusion

The increasing demand on water resources will inevitably require increasing use of water sources, containing higher concentrations of iodide/iodine especially in coastal areas. Reactive iodine species can be formed during oxidative water treatment of iodide-containing waters with various oxidants and react with dissolved inorganic and organic compounds. HOI is generally the major reactive species for the reaction with organic matter. Iodine reactive species are similar to chlorine and bromine in terms of speciation, mechanism of reaction or NOM incorporation. The fewer available rate constants do not allow a solid conclusion in terms of reactivity, which however generally appears to be intermediate between those of chlorine and bromine. The biggest difference consists of the further possible oxidation of HOI into iodate (by chlorine), considered as a safe end-product for iodine. The formation of iodinated disinfection by-products is then favored when the oxidant is not able to effectively oxidize HOI into iodate. The incorporation of I^- is highly dependent on the oxidant type, I^-, Br^- and NOM concentrations, NOM type, solution pH as well as on the ammonium concentration. Thus, disinfection practices intending to reduce THM4 levels may ultimately increase the formation of I-DBPs in the treated waters, especially in the case of chloramination. Ever-improving detection techniques allow for increasingly good detection of I-DBPs in the ng L^{-1} range or even lower. Occurrence studies mainly look at I-THMs as well as several iodoacids. However in recent years, more studies have started looking at nitrogenated I-DBPs as well as other polar I-DBPs. One of the main difficulties faced is being able to quantify many I-DBPs from different classes with different properties and at low detection limits.

I-DBPs are known to have generally higher toxicity than their chlorinated and brominated analogues. Iodoacetic acid and iodoacetamides exibit particularly strong cytotoxicity. In terms of mitigation, processes which favor the formation of iodate contribute to the reduction of I-DBP formation, in particular controlled ozonation allows to form iodate without formation of bromate and a pre-chlorination contact time before chloramination has been proved to be efficient in the control of I-DBP formation.

9. Bibliography

Abusallout et al. 2017. I. Abusallout, S. Rahman, and G. Hua. '**Effect of Temperature and pH on Dehalogenation of Total Organic Chlorine, Bromine and Iodine in Drinking Water**'. Chemosphere, 187 (2017), pp. 11-18. *https://doi.org/10.1016/j.chemosphere.2017.07.149*.

Acero et al. 2005. J. L. Acero, P. Piriou, and U. von Gunten. '**Kinetics and Mechanisms of Formation of Bromophenols during Drinking Water Chlorination: Assessment of Taste and Odor Development**'. Water Research, 39 (13) (2005), pp. 2979-2993. *https://doi.org/10.1016/j.watres.2005.04.055*.

Allard et al. 2016. S. Allard, J. Criquet, A. Prunier, C. Falantin, A. Le Person, J. Yat-Man Tang, and J-P. Croué. '**Photodecomposition of Iodinated Contrast Media and Subsequent Formation of Toxic Iodinated Moieties during Final Disinfection with Chlorinated Oxidants**'. Water Research, 103 (2016), pp. 453-461. *https://doi.org/10.1016/j.watres.2016.07.050*.

Allard et al. 2013a. S. Allard, L. Fouche, J. Dick, A. Heitz, and U. von Gunten. '**Oxidation of Manganese(II) during Chlorination: Role of Bromide**'. Environmental Science & Technology, 47 (15) (2013), pp. 8716-8723. *https://doi.org/10.1021/es401304r*.

Allard et al. 2010. S. Allard, H. Gallard, C. Fontaine, and J-P. Croué. '**Formation of Methyl Iodide on a Natural Manganese Oxide**'. Water Research, 44 (15) (2010), pp. 4623-4629. *https://doi.org/10.1016/j.watres.2010.06.008*.

Allard et al. 2013b. S. Allard, C. E. Nottle, A. Chan, C. Joll, and U. von Gunten. '**Ozonation of Iodide-Containing Waters: Selective Oxidation of Iodide to Iodate with Simultaneous Minimization of Bromate and I-THMs**'. Water Research, 47 (6) (2013), pp. 1953-1960. *https://doi.org/10.1016/j.watres.2012.12.002*.

Allard et al. 2009. S. Allard, U. von Gunten, E. Salhi, R. Nicolau, and H. Gallard. '**Oxidation of Iodide and Iodine on Birnessite (δ-MnO$_2$) in the pH Range 4-8**'. Water Research, 43 (14) (2009), pp. 3417-3426. *https://doi.org/10.1016/j.watres.2009.05.018*.

Allard et al. 2012. A. Sebastien, J. W. A. Charrois, C. A. Joll, and A. Heitz. '**Simultaneous Analysis of 10 Trihalomethanes at Nanogram per Liter Levels in Water Using Solid-Phase Microextraction and Gas Chromatography Mass-Spectrometry**'. Journal of Chromatography A, 1238 (2012), pp. 15-21. *https://doi.org/10.1016/j.chroma.2012.03.020*.

Allard and Gallard 2013. S. Allard, and H. Gallard. '**Abiotic Formation of Methyl Iodide on Synthetic Birnessite: A Mechanistic Study**'. Science of The Total Environment, 463-464 (2013), pp. 169-175. *https://doi.org/10.1016/j.scitotenv.2013.05.079*.

Allard et al. 2015. S. Allard, J. Tan, C. A. Joll, and U. von Gunten. '**Mechanistic Study on the Formation of Cl-/Br-/I-Trihalomethanes during Chlorination/Chloramination Combined with a Theoretical Cytotoxicity Evaluation**'. Environmental Science & Technology, 49 (18) (2015), pp. 11105-11114. *https://doi.org/10.1021/acs.est.5b02624*.

Amachi 2008. S. Amachi. '**Microbial Contribution to Global Iodine Cycling: Volatilization, Accumulation, Reduction, Oxidation and Sorption of Iodine**'. Microbes and Environments, 23 (4) (2008), pp. 269-276. *https://doi.org/10.1264/jsme2.me08548*.

Amachi et al. 2007. S. Amachi, K. Kimura, Y. Muramatsu, H. Shinoyama, and T. Fujii. '**Hydrogen Peroxide-Dependent Uptake of Iodine by Marine Flavobacteriaceae Bacterium Strain C-21**'. Applied and Environmental Microbiology, 73 (23) (2007), pp. 7536-7541. *https://doi.org/10.1128/AEM.01592-07*.

Aquaron et al. 1993. R. Aquaron, K. Zarrouck, M. El Jarari, R. Ababou, A. Talibi, and J. P. Ardissone. '**Endemic Goiter in Morocco (Skoura-Toundoute Areas in the High Atlas)**'. Journal of Endocrinological Investigation, 16 (1) (1993), pp. 9-14.

Awad et al. 2003. M. I. Awad, T. Oritani, and T. Ohsaka. '**Kinetic Studies on the Oxidation of Iodide by Peroxyacetic Acid**'. Inorganica Chimica Acta, 344 (2003), pp. 253-256. *https://doi.org/10.1016/S0020-1693(02)01337-3*.

Barber et al. 2015. L. B. Barber, M. L. Hladik, A. M. Vajda, K. C. Fitzgerald, and C. Douville. '**Impact of Wastewater Infrastructure Upgrades on the Urban Water Cycle: Reduction in Halogenated Reaction Byproducts Following Conversion from Chlorine Gas to Ultraviolet Light Disinfection**'. Science of The Total Environment, 529 (2015), pp. 264-274. *https://doi.org/10.1016/j.scitotenv.2015.04.112*.

Becker et al. 2013. W. Becker, B. Stanford, and E. J. Rosenfeldt. '**Guidance on Complying With Stage 2 D/DBP Regulation**'. Water Research Foundation, (2013). *http://www.waterrf.org/PublicReportLibrary/4427.pdf*.

Bell and Gelles 1951. R. P. Bell, and E. Gelles. ‘**608. The Halogen Cations in Aqueous Solution**’. Journal of the Chemical Society, (1951), pp. 2734-2740. *https://doi.org/10.1039/JR9510002734*.

Bellar et al. 1974. T. A. Bellar, J. J. Lichtenberg, and R. C. Kroner. **The Occurrence of Organohalides in Chlorinated Drinking Waters**. U.S. Environmental Protection Agency, Office of Research and Development, National Environmental Research Center. *https://doi.org/10.1002/j.1551-8833.1974.tb02129.x*.

Berliner 1951. E. Berliner. ‘**Kinetics of the Iodination of Phenol**’. Journal of the American Chemical Society, 73 (9) (1951), pp. 4307-4311. *https://doi.org/10.1021/ja01153a083*.

Bichsel 2000. Y. Bichsel. ‘**Behavior of Iodine Species in Oxidative Processes During Drinking Water Treatment**’. Doctoral Thesis (2000). *https://doi.org/10.3929/ethz-a-003861380*.

Bichsel and von Gunten 1999a. Y. Bichsel and U. von Gunten. ‘**Determination of Iodide and Iodate by Ion Chromatography with Postcolumn Reaction and UV/Visible Detection**’. Analytical Chemistry, 71 (1) (1999), pp. 34-38. *https://doi.org/10.1021/ac980658j*.

Bichsel and von Gunten 1999b. Y. Bichsel and U. von Gunten. ‘**Oxidation of Iodide and Hypoiodous Acid in the Disinfection of Natural Waters**’. Environmental Science & Technology, 33 (22) (1999), pp. 4040-4045. *https://doi.org/10.1021/es990336c*.

Bichsel and von Gunten 2000a. Y. Bichsel and U. von Gunten. ‘**Formation of Iodo-Trihalomethanes during Disinfection and Oxidation of Iodide-Containing Waters**’. Environmental Science & Technology, 34 (13) (2000), pp. 2784-2791. *https://doi.org/ 10.1021/es9914590*.

Bichsel and von Gunten 2000b. Y. Bichsel and U. von Gunten. ‘**Hypoiodous Acid: Kinetics of the Buffer-Catalyzed Disproportionation**’. Water Research, 34 (12) (2000), pp. 3197-3203. *https://doi.org/10.1016/S0043-1354(00)00077-4*.

Bond et al. 2015. T. Bond, M. R. Templeton, N. H. M. Kamal, N. Graham, and R. Kanda. ‘**Nitrogenous Disinfection Byproducts in English Drinking Water Supply Systems: Occurrence, Bromine Substitution and Correlation Analysis**’. Water Research, 85 (2015), pp. 85-94. *https://doi.org/10.1016/j.watres.2015.08.015*.

Bruchet et al. 2004. A. Bruchet, J. P. Duguet, and I. H. (Mel) Suffe. ‘**Role of Oxidants and Disinfectants on the Removal, Masking and Generation of Tastes and Odours**’. Reviews in Environmental Science and Biotechnology, 3 (1) (2004), pp. 33-41. *https://doi.org/10.1023/B:RESB.0000040011.81065.25*.

Burger and Liebhafsky 1973. J. D. Burger, and H. A. Liebhafsky. ‘**Thermodynamic Data for Aqueous Iodine Solutions at Various Temperatures. Exercise in Analytical Chemistry**’. Analytical Chemistry, 45 (3) (1973), pp. 600-602. *https://doi.org/10.1021/ac60325a044*.

Bürgi et al. 2001. H. Bürgi, T. H. Schaffner, and J. P. Seiler. ‘**The Toxicology of Iodate: A Review of the Literature**’. Thyroid, 11 (5) (2001), pp. 449-456. *https://doi.org/10.1089/105072501300176408*.

Buxton et al. 1988. G. V. Buxton, C. L. Greenstock, W. P. Helman, and A. B. Ross. ‘**Critical Review of Rate Constants for Reactions of Hydrated Electrons, Hydrogen Atoms and Hydroxyl Radicals ($^{\bullet}$OH/$^{\bullet}$O^{-}) in Aqueous Solution**’. Journal of Physical and Chemical Reference Data, 17 (2) (1988), pp. 513-886. *https://doi.org/10.1063/1.555805*.

Cancho et al. 2001. B. Cancho, C. Fabrellas, A. Diaz, and F. Ventura. ‘**Determination of the Odor Threshold Concentrations Of Iodinated Trihalomethanes in Drinking Water**’. Journal of Agricultural and Food Chemistry, 49 (4) (2001), pp. 1881-1884. *https://doi.org/10.1021/jf001252m*.

Cancho et al. 2000. B. Cancho, F. Ventura, M. T. Galceran, A. Diaz, and S. Ricart. ‘**Determination, Synthesis and Survey of Iodinated Trihalomethanes in Water Treatment Processes**’. Water Research, 34 (13) (2000), pp. 3380-3390. *https://doi.org/10.1016/S0043-1354(00)00079-8*.

Cancho et al. 1999. B. Cancho, F. Ventura, and M. T. Galceran. ‘**Solid-Phase Microextraction for the Determination of Iodinated Trihalomethanes in Drinking Water**’. Journal of Chromatography A, 841 (2) (1999), pp. 197-206. *https://doi.org/10.1016/S0021-9673(99)00320-9*.

Cardador et al. 2015. M. J. Cardador, J. Fernández-Salguero, and M. Gallego. ‘**Simultaneous Quantification of Trihalomethanes and Haloacetic Acids in Cheese by On-Line Static Headspace Gas Chromatography – Mass-Spectrometry**’. Journal of Chromatography A, 1408 (21) (2015), pp. 22-29. *https://doi.org/10.1016/j.chroma.2015.07.007*.

Cardador and Gallego 2016. M. J. Cardador and M. Gallego. ‘**Static Headspace–Gas Chromatography–Mass Spectrometry for the Simultaneous Determination of Trihalomethanes and Haloacetic Acids in Canned Vegetables**’. Journal of Chromatography A, 1454 (2016), pp. 9-14. *https://doi.org/10.1016/j.chroma.2016.05.080*.

Cardador et al. 2016. M. J. Cardador, M. Gallego, L. Cabezas, and J. Fernández-Salguero. '**Detection of Regulated Disinfection By-Products in Cheeses**'. Food Chemistry, 204 (2016), pp. 306-313. *https://doi.org/10.1016/j.foodchem.2016.02.146*.

Cardador et al. 2008. M. J. Cardador, A. Serrano, and M. Gallego. '**Simultaneous Liquid–Liquid Microextraction/Methylation for the Determination of Haloacetic Acids in Drinking Waters by Headspace Gas Chromatography**'. Journal of Chromatography A, 1209 (1-2) (2008), pp. 61-69. *https://doi.org/10.1016/j.chroma.2008.09.033*.

Carpenter 2015. L. J. Carpenter. '**BIOGEOCHEMICAL CYCLES | Iodine**'. In *Encyclopedia of Atmospheric Sciences (Second Edition)*, edited by G. R. North, J. Pyle, and F. Zhang, (2015), pp. 205-219. Oxford: Academic Press.

CELEX-EUR. 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, L 330, (41) (1998), pp. 32-54.

Chen et al. 2015. T-L. Chen, S-H. Tzing, and W-H. Ding. '**Rapid Screening of Haloacetamides in Water Using Salt-Assisted Liquid–Liquid Extraction Coupled Injection-Port Silylation Gas Chromatography–Mass Spectrometry**'. Journal of Chromatography A, 1422 (2015), pp. 340-344. *https://doi.org/10.1016/j.chroma.2015.10.047*.

Chinn et al. 2007. R. Chinn, T. Lee, S. W. Krasner, M. Dale, S. D. Richardson, J. Pressman, T. Speth, R. Miltner, and J. E. Simons. '**Solid-Phase Extraction of 35 DBPs with Analysis by GC/ECD and GC/MS**'. Proceedings of the American Water Works Association Water Quality Technology Conference; American Water Works Association: Denver, CO (2007).

Chu et al. 2012. W. Chu, N. Gao, D. Yin, S. W. Krasner, and M. R. Templeton. '**Trace Determination of 13 Haloacetamides in Drinking Water Using Liquid Chromatography Triple Quadrupole Mass Spectrometry with Atmospheric Pressure Chemical Ionization**'. Journal of Chromatography A, 1235 (2012), pp. 178-181. *https://doi.org/10.1016/j.chroma.2012.02.074*.

Chu et al. 2013. W. Chu, N. Gao, D. Yin, and S. W. Krasner. '**Formation and Speciation of Nine Haloacetamides, an Emerging Class of Nitrogenous DBPs, during Chlorination or Chloramination**'. Journal of Hazardous Materials, 260 (2013), pp. 806-812. *https://doi.org/10.1016/j.jhazmat.2013.06.044*.

Chu et al. 2014. W. Chu, N. Gao, D. Yin, S. W. Krasner, and W. A. Mitch. '**Impact of UV/H_2O_2/ Pre-Oxidation on the Formation of Haloacetamides and Other Nitrogenous Disinfection Byproducts during Chlorination**'. Environmental Science & Technology, 48 (20) (2014), pp. 12190-12198. *https://doi.org/10.1021/es502115x*.

Chu et al. 2016a. W. Chu, J. Hu, T. Bond, N. Gao, B. Xu, and D. Yin. '**Water Temperature Significantly Impacts the Formation of Iodinated Haloacetamides during Persulfate Oxidation**'. Water Research, 98 (2016), pp. 47-55. *https://doi.org/10.1016/j.watres.2016.04.002*.

Chu et al. 2016b. W. Chu, X. Li, T. Bond, N. Gao, and D. Yin. '**The Formation of Haloacetamides and Other Disinfection By-Products from Non-Nitrogenous Low-Molecular Weight Organic Acids during Chloramination**'. Chemical Engineering Journal, 285 (2016), pp. 164-171. *https://doi.org/10.1016/j.cej.2015.09.087*.

Chuang et al. 2019. Y-H. Chuang, A. Szczuka, and W. A. Mitch. '**Comparison of Toxicity-Weighted Disinfection Byproduct Concentrations in Potable Reuse Waters and Conventional Drinking Waters as a New Approach to Assessing the Quality of Advanced Treatment Train Waters**'. Environmental Science & Technology, 53 (7) (2019), pp. 3729-3738. *https://doi.org/10.1021/acs.est.8b06711*.

Cojocariu et al. 2016. C. Cojocariu, C. Postigo, S. D. Richardson, D. Barceló, and P. Silcock. '**Discovery of Emerging Disinfection By-Products in Water Using Gas Chromatography Coupled with Orbitrap-Based Mass Spectrometry**'. Application Note 10490 (2016). Thermo Scientific. *http://www.unitylabservices.eu/content/dam/tfs/ATG/CMD/cmd-documents/sci-res/app/ms/GC-MS/AN-10490-GC-MS-Disinfection-Byproducts-Water-AN10490-EN.pdf*.

Cortés and Marcos 2018. C. Cortés and R. Marcos. '**Genotoxicity of Disinfection Byproducts and Disinfected Waters: A Review of Recent Literature**'. Mutation Research/Genetic Toxicology and Environmental Mutagenesis, 831 (2018), pp. 1-12. *https://doi.org/10.1016/j.mrgentox.2018.04.005*.

Criquet et al. 2012. J. Criquet, S. Allard, E. Salhi, C. A. Joll, A. Heitz, and U. von Gunten. '**Iodate and Iodo-Trihalomethane Formation during Chlorination of Iodide-Containing Waters: Role of Bromide**'. Environmental Science & Technology, 46 (13) (2012), pp. 7350-7357. *https://doi.org/10.1021/es301301g*.

Criquet et al. 2015. J. Criquet, E. M. Rodriguez, S. Allard, S. Wellauer, E. Salhi, C. A. Joll, and U. von Gunten. '**Reaction of Bromine and Chlorine with Phenolic Compounds and Natural Organic Matter Extracts – Electrophilic Aromatic Substitution and Oxidation**'. Water Research, 85 (2015), pp. 476-486. *https://doi.org/10.1016/j.watres.2015.08.051*.

Day and Powell-Jackson 1972. T. K. Day and P. R. Powell-Jackson. '**Fluoride, water hardness and endemic goitre**'. The Lancet, 299 (7761) (1972), pp. 1135-1138. *https://doi.org/10.1016/s0140-6736(72)91370-0*.

Deane and Potter 1985. A. M. Deane and P. E. Potter. '**Proceedings of the Specialists' Workshop on Iodine Chemistry in Reactor Safety**'. INIS, 17 (19) (1985), 365 p.

Deborde and von Gunten 2008. M. Deborde and U. von Gunten. '**Reactions of Chlorine with Inorganic and Organic Compounds during Water Treatment—Kinetics and Mechanisms: A Critical Review**'. Water Research, 42 (1-2) (2008), pp. 13-51. *https://doi.org/10.1016/j.watres.2007.07.025*.

Dillon et al. 2015. G. Dillon, D. Shepherd, T. Hall, R. Gee, J. Parker, and P. Rumsby. '**CHAPTER 38: Risk-Based Approach to the Formation of DBPs of Concern in UK Drinking Water**'. In *Disinfection By-Products in Drinking Water*, edited by K. C. Thompson, S. Gillespie, E. Goslan. (2015), pp. 336-348. *https://doi.org/10.1039/9781782622710-00336*.

Ding and Zhang 2009. G. Ding and X. Zhang. '**A Picture of Polar Iodinated Disinfection Byproducts in Drinking Water by (UPLC/)ESI-TqMS**'. Environmental Science & Technology, 43 (24) (2009), pp. 9287-9293. *https://doi.org/10.1021/es901821a*.

Ding et al. 2013. H. Ding, L. Meng, H. Zhang, J. Yu, W. An, J. Hu, and M. Yang. '**Occurrence, Profiling and Prioritization of Halogenated Disinfection by-Products in Drinking Water of China**'. Environmental Science: Processes & Impacts, 15 (7) (2013), pp. 1424-1429. *https://doi.org/10.1039/c3em00110e*.

Dong et al. 2019a. H. Dong, Z. Qiang, and S. D. Richardson. '**Formation of Iodinated Disinfection Byproducts (I-DBPs) in Drinking Water: Emerging Concerns and Current Issues**'. Accounts of Chemical Research, 52 (4) (2019), pp. 896-905. *https://doi.org/10.1021/acs.accounts.8b00641*.

Dong et al. 2019b. Z. Dong, C. Jiang, J. Yang, X. Zhang, W. Dai, and P. Cai. '**Transformation of Iodide by Fe(II) Activated Peroxydisulfate**'. Journal of Hazardous Materials, 373 (2019), pp. 519-526. *https://doi.org/10.1016/j.jhazmat.2019.03.063*.

Duirk et al. 2011. S. E. Duirk, C. Lindell, C. C. Cornelison, J. Kormos, T. A. Ternes, M. Attene-Ramos, J. Osiol, E. D. Wagner, M. J. Plewa, and S. D. Richardson. '**Formation of Toxic Iodinated Disinfection By-Products from Compounds Used in Medical Imaging**'. Environmental Science & Technology, 45 (16) (2011), pp. 6845-6854. *https://doi.org/10.1021/es200983f*.

Eastman and Zimmermann 2000. C. J. Eastman and M. B. Zimmermann. '**The Iodine Deficiency Disorders**'. In *Endotext*, edited by L. J. De Groot, G. Chrousos, K. Dungan, K. R. Feingold, A. Grossman, J. M. Hershman, C. Koch, et al. South Dartmouth (MA): MDText.com, Inc (2000).

EPA 1998. EPA. '**National Primary Drinking Water Regulations: Disinfectants and Disinfection Byproducts**', final rule. Federal Register, 63 (241) (1998), pp. 69390-69376.

EPA 2016. EPA. '**Six-Year Review 3 Technical Support Document for Disinfectants/Disinfection Byproducts Rules**', (2016), 468 p.

Fabian and Gordon 1997. I. Fabian and G. Gordon. '**The Kinetics and Mechanism of the Chlorine Dioxide - Iodide Ion Reaction**'. Inorganic Chemistry, 36 (12) (1997), pp. 2494-2497. *https://doi.org/10.1021/ic961279g*.

Fang et al. 2018. C. Fang, S. W. Krasner, W. Chu, S. Ding, T. Zhao, and N. Gao. '**Formation and Speciation of Chlorinated, Brominated, and Iodinated Haloacetamides in Chloraminated Iodide-Containing Waters**'. Water Research, 145 (2018), pp. 103-112. *https://doi.org/10.1016/j.watres.2018.07.074*.

Farrenkopf et al. 1997. A. M. Farrenkopf, M. E. Dollhopf, S. N. Chadhain, G. W. Luther, and K. H. Nealson. '**Reduction of Iodate in Seawater during Arabian Sea Shipboard Incubations and in Laboratory Cultures of the Marine Bacterium Shewanella Putrefaciens Strain MR-4**'. Marine Chemistry, 57 (3) (1997), pp. 347-354. *https://doi.org/10.1016/S0304-4203(97)00039-X*.

Folgeman et al. 1989. K. D. Fogelman, D. M. Walker, and D. W. Margerum. '**Nonmetal Redox Kinetics: Hypochlorite and Hypochlorous Acid Reactions with Sulfite**'. Inorganic Chemistry, 28 (6) (1989), pp. 986-993. *https://doi.org/10.1021/ic00305a002*.

Fox et al. 2009. P. M. Fox, J. A. Davis, and G. W. Luther III. '**The Kinetics of Iodide Oxidation by the Manganese Oxide Mineral Birnessite**'. Geochimica et Cosmochimica Acta, 73 (10) (2009), pp. 2850-2861. *https://doi.org/10.1016/j.gca.2009.02.016*.

Francois 1987. R. Francois. '**The Influence of Humic Substances on the Geochemistry of Iodine in Nearshore and Hemipelagic Marine Sediments**'. Geochimica et Cosmochimica Acta, 51 (9) (1987), pp. 2417-2427. *https://doi.org/10.1016/0016-7037(87)90294-8*.

Frazey et al. 1998. P. A. Frazey, R. M. Barkley, and R. E. Sievers. '**Solid-Phase Microextraction with Temperature-Programmed Desorption for the Analysis of Iodination Disinfection Byproducts**'. Analytical Chemistry, 70 (3) (1998), pp. 638-644. *https://doi.org/10.1021/ac970395d*.

Fuge 2005. R. Fuge. '**Soils and Iodine Deficiency**'. In Essentials of Medical Geology: Impacts of the Natural Environment on Public Health, Academic Press, (2005), pp. 417-434.

Fuge and Johnson 2015. R. Fuge and C. C. Johnson. '**Iodine and Human Health, the Role of Environmental Geochemistry and Diet, a Review**'. Applied Geochemistry, 63 (2015), pp. 282-302. *https://doi.org/10.1016/j.apgeochem.2015.09.013*.

Fuge 1989. R. Fuge. '**Iodine in Waters: Possible Links with Endemic Goitre**'. Applied Geochemistry, 4 (2) (1989), pp. 203-208. *https://doi.org/10.1016/0883-2927(89)90051-6*.

Fukutomi and Gordon 1967. H. Fukutomi and G. Gordon. '**Kinetic Study of the Reaction between Chlorine Dioxide and Potassium Iodide in Aqueous Solution**'. Journal of the American Society, 89 (6) (1967), pp. 1362-1366. *https://doi.org/10.1021/ja00982a014*.

Gallard et al. 2009. H. Gallard, S. Allard, R. Nicolau, U. von Gunten, and J-P. Croué. '**Formation of Iodinated Organic Compounds by Oxidation of Iodide-Containing Waters with Manganese Dioxide**'. Environmental Science & Technology, 43 (18) (2009), pp. 7003-7009. *https://doi.org/10.1021/es9010338*.

Gan et al. 2015. W. Gan, V. K. Sharma, X. Zhang, L. Yang, and X. Yang. '**Investigation of Disinfection Byproducts Formation in Ferrate(VI) Pre-Oxidation of NOM and Its Model Compounds Followed by Chlorination**'. Journal of Hazardous Materials, 292 (2015), pp. 197-204. *https://doi.org/10.1016/j.jhazmat.2015.02.037*.

Garland et al. 1980. J. A. Garland, A. W. Elzerman, and S. A. Penkett. '**The Mechanism for Dry Deposition of Ozone to Seawater Surfaces**'. Journal of Geophysical Research: Oceans, 85 (C12) (1980), pp. 7488-7492. *https://doi.org/10.1029/JC085iC12p07488*.

Gerritsen and Margerum 1990. C. M. Gerritsen and D. W. Margerum. '**Non-Metal Redox Kinetics: Hypochlorite and Hypochlorous Acid Reactions with Cyanide**'. Inorganic Chemistry, 29 (15) (1990), pp. 2757-2762. *https://doi.org/10.1021/ic00340a010*.

Gilfedder et al. 2007. B. S. Gilfedder, M. Petri, and H. Biester. '**Iodine and Bromine Speciation in Snow and the Effect of Orographically Induced Precipitation**'. Atmospheric Chemistry and Physics, 7 (10) (2007), pp. 2661-2669. *https://doi.org/10.5194/acp-7-2661-2007*.

Gilfedder et al. 2009. B. S. Gilfedder, M. Petri, and H. Biester. '**Iodine Speciation and Cycling in Fresh Waters: A Case Study from a Humic Rich Headwater Lake (Mummelsee)**'. Journal of Limnology, 68 (2) (2009), pp. 396-408. *https://doi.org/10.4081/jlimnol.2009.396*.

Gilfedder et al. 2010. B. S. Gilfedder, M. Petri, M. Wessels, and H. Biester. '**An Iodine Mass-Balance for Lake Constance, Germany: Insights into Iodine Speciation Changes and Fluxes**'. Geochimica et Cosmochimica Acta, 74 (11) (2010), pp. 3090-3111. *https://doi.org/10.1016/j.gca.2010.03.008*.

Gong and Zhang 2013. T. Gong and X. Zhang. '**Determination of Iodide, Iodate and Organo-Iodine in Waters with a New Total Organic Iodine Measurement Approach**'. Water Research, 47 (17) (2013), pp. 6660-6669. *https://doi.org/10.1016/j.watres.2013.08.039*.

Gong and Zhang 2015. T. Gong and X. Zhang. '**Detection, Identification and Formation of New Iodinated Disinfection Byproducts in Chlorinated Saline Wastewater Effluents**'. Water Research, 68 (2015), pp. 77-86. *https://doi.org/10.1016/j.watres.2014.09.041*.

Goodyer and Behrens 2000. L. Goodyer and R. H. Behrens. '**Safety of Iodine Based Water Sterilization for Travelers**'. Journal of Travel Medicine, 7 (2000), p. 38. *https://doi.org/10.2310/7060.2000.00012*.

Goslan 2016. E. H. Goslan. '**Identification of High Risk Sites for Iodinated DBP Formation**'. Project DWI 70/2/291 Report. Department: School of Water, Energy and Environment, (2000). *http://dwi.defra.gov.uk/research/completed-research/reports/DWI70-2-291.pdf*.

Goslan et al. 2009. E. H. Goslan, S. W. Krasner, M. Bower, S. A. Rocks, P. Holmes, L. S. Levy, and S. A. Parsons. '**A Comparison of Disinfection By-Products Found in Chlorinated and Chloraminated Drinking Waters in Scotland**'. Water Research, 43 (18) (2009), pp. 4698-4706. *https://doi.org/10.1016/j.watres.2009.07.029*.

Grant et al. 1982. J. L. Grant, P. De Kepper, I. R. Epstein, K. Kustin, and M. Orban. '**Systematic Design of Chemical Oscillators. Part 9. Kinetics and Mechanism of the Oxidation of Iodine by Chlorite Ion**'. Inorganic Chemistry, 21 (6) (1982), pp. 2192-2196. *https://doi.org/10.1021/ic00136a012*.

Griffith and McKeown 1932. R. O. Griffith and A. McKeown. '**The Kinetics of the Iodine-Oxalate Reaction**'. *Transactions of the Faraday Society,* 28 (1932), pp. 752-760. *https://doi.org/10.1039/tf9322800752*.

Gruchlik et al. 2015. Y. Gruchlik, A Heitz, C. Joll, S. McDonald, J. Tan, L. Breckler, F. Bradder, G. Roeszler, and D. Halliwell. '**Novel Treatment Technologies for the Minimisation of Bromide and Iodide in Drinking Water**'. Final Report for ARC Linkage Project 100100285. CWQRC Report No: CWQRC-2015-004, 44 (2015).

Guan et al. 2017. C. Guan, J. Jiang, C. Luo, S. Pang, C. Jiang, J. Ma, Y. Jin, and J. Li. '**Transformation of Iodide by Carbon Nanotube Activated Peroxydisulfate and Formation of Iodoorganic Compounds in the Presence of Natural Organic Matter**'. Environmental Science & Technology, 51 (1) (2017), pp. 479-487. *https://doi.org/10.1021/acs.est.6b04158*.

Guo et al. 2014. W. Guo, Y. Shan, and X. Yang. '**Factors Affecting the Formation of Iodo-Trihalomethanes during Oxidation with Chlorine Dioxide**'. Journal of Hazardous Materials, 264 (2014), pp. 91-97. *https://doi.org/10.1016/j.jhazmat.2013.10.064*.

Han et al. 2017. J. Han, X. Zhang, J. Liu, X. Zhu, and T. Gong. '**Characterization of Halogenated DBPs and Identification of New DBPs Trihalomethanols in Chlorine Dioxide Treated Drinking Water with Multiple Extractions**'. Journal of Environmental Sciences, 58 (2017), pp. 83-92. *https://doi.org/10.1016/j.jes.2017.04.026*.

Hansson et al. 1987. R. C. Hansson, M. J. Henderson, P. Jack, and R. D. Taylor. '**Iodoform Taste Complaints in Chloramination**'. Water Research, 21 (10) (1987), pp. 1265-1271. *https://doi.org/10.1016/0043-1354(87)90179-5*.

Harkness et al. 2015. J. S. Harkness, G. S. Dwyer, N. R. Warner, K. M. Parker, W. A. Mitch, and A. Vengosh. '**Iodide, Bromide, and Ammonium in Hydraulic Fracturing and Oil and Gas Wastewaters: Environmental Implications**'. Environmental Science & Technology, 49 (3) (2015), pp. 1955-1963. *https://doi.org/10.1021/es504654n*.

Hassan et al. 2006. K. Z. A. Hassan, K. C. Bower, and C. M. Miller. '**Iron Oxide Enhanced Chlorine Decay and Disinfection By-Product Formation**'. Journal of Environmental Engineering, 132 (12) (2006), pp. 1609-1616. *https://doi.org/10.1061/(ASCE)0733-9372(2006)132:12(1609)*.

Hebert et al. 2010. A. Hebert, D. Forestier, D. Lenes, D. Benanou, S. Jacob, C. Arfi, L. Lambolez, and Y. Levi. '**Innovative Method for Prioritizing Emerging Disinfection By-Products (DBPs) in Drinking Water on the Basis of Their Potential Impact on Public Health**'. Water Research, 44 (10) (2010), pp. 3147-3165. *https://doi.org/10.1016/j.watres.2010.02.004*.

Heeb et al. 2014. M. B. Heeb, J. Criquet, S. Zimmermann-Steffens, and U. von Gunten. '**Oxidative Treatment of Bromide-Containing Waters: Formation of Bromine and Its Reactions with Inorganic and Organic Compounds – A Critical Review**'. Water Research, 48 (1) (2014), pp. 14-42. *https://doi.org/10.1016/j.watres.2013.08.030*.

Heeb et al. 2017. M. B. Heeb, I. Kristiana, D. Trogolo, J. S. Arey, and U. von Gunten. '**Formation and Reactivity of Inorganic and Organic Chloramines and Bromamines during Oxidative Water Treatment**'. Water Research, 110 (2017), pp. 91-101. *https://doi.org/10.1016/j.watres.2016.11.065*.

Hofman-Caris and Hofman 2019. R. Hofman-Caris and J. Hofman. '**Limitations of Conventional Drinking Water Technologies in Pollutant Removal**'. In *Applications of Advanced Oxidation Processes (AOPs) in Drinking Water Treatment*. Edited by A. Gil, L. A. Galeano, M. Á. Vicente. The Handbook of Environmental Chemistry, 67 (2019), pp. 21-51. *https://www.springer.com/fr/book/9783319768816*.

Hoigne and Bader 1994. J. Hoigne and H. Bader. '**Kinetics of Reactions of Chlorine Dioxide (OClO) in Water-I. Rate Constants for Inorganic and Organic Compounds**'. Water Research, 28 (1) (1994), pp. 45-55. *https://doi.org/10.1016/0043-1354(94)90118-X*.

Hoops et al. 2006. S. Hoops, S. Sahle, R. Gauges, C. Lee, J. Pahle, N. Simus, M. Singhal, L. Xu, P. Mendes, and U. Kummer. '**COPASI--a COmplex PAthway SImulator**'. Bioinformatics, 22 (24) (2006), pp. 3067-3074. *https://doi.org/10.1093/bioinformatics/btl485*.

Hou et al. 2009. X. Hou, A. Aldahan, S. P. Nielsen, and G. Possnert. '**Time Series of ^{129}I and ^{127}I Speciation in Precipitation from Denmark**'. Environmental Science & Technology, 43 (17) (2009), pp. 6522-6528. *https://doi.org/10.1021/es9012678*.

House 1962. D. A. House. '**Kinetics and Mechanism of Oxidations by Peroxydisulfate.**'. Chemical Reviews, 62 (3) (1962), pp. 185-203. *https://doi.org/10.1021/cr60217a001*.

Hrudey and Charrois 2012. S. E. Hrudey and J. WA. Charrois. **Disinfection By-Products and Human Health**. IWA Publishing, (2012).

Hu et al. 2017. J. Hu, H. Dong, J. Qu, and Z. Qiang. '**Enhanced Degradation of Iopamidol by Peroxymonosulfate Catalyzed by Two Pipe Corrosion Products (CuO and δ-MnO$_2$)**'. Water Research, 112 (2017), pp. 1-8. *https://doi.org/10.1016/j.watres.2017.01.025*.

Hu et al. 2018a. S. Hu, T. Gong, J. Ma, Y. Tao, and Q. Xian. '**Simultaneous Determination of Iodinated Haloacetic Acids and Aromatic Iodinated Disinfection Byproducts in Waters with a New SPE-HPLC-MS/MS Method**'. Chemosphere, 198 (2018), pp. 147-153. *https://doi.org/10.1016/j.chemosphere.2018.01.124*.

Hu et al. 2018b. S. Hu, T. Gong, Q. Xian, J. Wang, J. Ma, Z. Li, J. Yin, B. Zhang, and B. Xu. '**Formation of Iodinated Trihalomethanes and Haloacetic Acids from Aromatic Iodinated Disinfection Byproducts during Chloramination**'. Water Research, 147 (2018), pp. 254-263. *https://doi.org/10.1016/j.watres.2018.09.058*.

Hua and Reckhow 2007a. G. Hua and D. A. Reckhow. '**Comparison of Disinfection Byproduct Formation from Chlorine and Alternative Disinfectants**'. Water Research, 41 (8) (2007), pp. 1667-1678. *https://doi.org/10.1016/j.watres.2007.01.032*.

Hua and Reckhow 2005. G. Hua and D. A. Reckhow. '**Determination of TOCl, TOBr and TOI in Drinking Water by Pyrolysis and off-Line Ion Chromatography**'. Analytical and Bioanalytical Chemistry, 384 (2) (2005), pp. 495-504. *https://doi.org/10.1007/s00216-005-0214-3*.

Hua and Reckhow 2007b. G. Hua and D. A. Reckhow. '**Characterization of Disinfection Byproduct Precursors Based on Hydrophobicity and Molecular Size**'. Environmental Science & Technology, 41 (9) (2007), pp. 3309-3315. *https://doi.org/10.1021/es062178c*.

Hua and Reckhow 2008. G. Hua and D. A. Reckhow. '**DBP Formation during Chlorination and Chloramination: Effect of Reaction Time, pH, Dosage, and Temperature**'. Journal - American Water Works Association, 100 (8) (2008), pp. 82-95. *https://doi.org/10.1002/j.1551-8833.2008.tb09702.x*.

Hua et al. 2015. G. Hua, D. A. Reckhow, and I. Abusallout. '**Correlation between SUVA and DBP Formation during Chlorination and Chloramination of NOM Fractions from Different Sources**'. Chemosphere, 130 (2015), pp. 82-89. *https://doi.org/10.1016/j.chemosphere.2015.03.039*.

Hua et al. 2006. G. Hua, D. A. Reckhow, and J. Kim. '**Effect of Bromide and Iodide Ions on the Formation and Speciation of Disinfection Byproducts during Chlorination**'. Environmental Science & Technology, 40 (9) (2006), pp. 3050-3056. *https://doi.org/10.1021/es0519278*.

Huang et al. 2018. G. Huang, P. Jiang, L. K. Jmaiff Blackstock, D. Tian, and X-F. Li. '**Formation and Occurrence of Iodinated Tyrosyl Dipeptides in Disinfected Drinking Water**'. Environmental Science & Technology, 52 (7) (2018), pp. 4218-4226. *https://doi.org/10.1021/acs.est.7b06276*.

Huber et al. 2003. M. M. Huber, S. Canonica, G-Y. Park, and U. von Gunten. '**Oxidation of Pharmaceuticals during Ozonation and Advanced Oxidation Processes**'. Environmental Science & Technology, 37 (5) (2003), pp. 1016-1024. *https://doi.org/10.1021/es025896h*.

Huber et al. 2005. M. M. Huber, A. Göbel, A. Joss, N. Hermann, D. Löffler, C. S. McArdell, A. Ried, H. Siegrist, T. A. Ternes, and U. von Gunten. '**Oxidation of Pharmaceuticals during Ozonation of Municipal Wastewater Effluents: A Pilot Study**'. Environmental Science & Technology, 39 (11) (2005), pp. 4290-4299. *https://doi.org/10.1021/es048396s*.

Ingles and Israel 1948. O. G. Ingles and G. C. Israel. '**The Oxidation of Some Aldoses by Alkaline Solutions of Iodine**'. Journal of the Chemical Society, 174 (1948), pp. 810-814. *https://doi.org/ 10.1039/jr9480000810*.

Ioannou et al. 2016. P. Ioannou, P. Charisiadis, P. C. Andra, and K. C. Makris. '**Occurrence and Variability of Iodinated Trihalomethanes Concentrations within Two Drinking-Water Distribution Networks**'. Science of The Total Environment, 543 Part A (2016), pp. 505-513. *https://doi.org/10.1016/j.scitotenv.2015.10.031*.

Jafvert and Valentine 1992. C. T. Jafvert, and R. L. Valentine. '**Reaction Scheme for the Chlorination of Ammoniacal Water**'. Environmental Science & Technology, 26 (3) (1992), pp. 577-586. *https://doi.org/10.1021/es00027a022*.

Jeong et al. 2012. C. H. Jeong, E. D. Wagner, V. R. Siebert, S. Anduri, S. D. Richardson, E. J. Daiber, A. Bruce McKague, et al. '**Occurrence and Toxicity of Disinfection Byproducts in European Drinking Waters in Relation with the HIWATE Epidemiology Study**'. Environmental Science & Technology, 46 (21) (2012), pp. 12120-12128. *https://doi.org/10.1021/es3024226*.

Jette and King 1929. E. Jette and C. V. King. '**The Oxidation of Iodide Ion by Persulfate Ion. I. The Effect of Tri-Iodide Ion Formation on the Reaction Velocity**'. Journal of the American Chemical Society, 51 (4) (1929), pp. 1034-1047. *https://doi.org/10.1021/ja01379a008*.

Jiang et al. 2016. Y. Jiang, J. E. Goodwill, J. E. Tobiason, and D. A. Reckhow. '**Impacts of Ferrate Oxidation on Natural Organic Matter and Disinfection Byproduct Precursors**'. Water Research, 96 (2016), pp. 114-125. *https://doi.org/10.1016/j.watres.2016.03.052*.

Johanson 2000. K. J. Johanson. '**Iodine in Soil**'. Technical report TR-00-21. Swedish Nuclear Fuel and Waste Management Co., (2000). *http://www.skb.se/upload/publications/pdf/TR-00-21.pdf*.

Johnson 2003. C. C. Johnson. '**The Geochemistry of Iodine and Its Application to Environmental Strategies for Reducing the Risks from Iodine Deficiency Disorders (IDD)**'. Commissioned Report CR/03/057N. British Geological Survey, (2003). *http://nora.nerc.ac.uk/10724/1/CR03057N.pdf*.

Jones et al. 2012a. D. B. Jones, A. Saglam, H. Song, and T. Karanfil. '**The Impact of Bromide/Iodide Concentration and Ratio on Iodinated Trihalomethane Formation and Speciation**'. Water Research, 46 (1) (2012), pp. 11-20. *https://doi.org/10.1016/j.watres.2011.10.005*.

Jones et al. 2012b. D. B. Jones, H. Song, and T. Karanfil. '**The Effects of Selected Preoxidation Strategies on I-THM Formation and Speciation**'. Water Research, 46 (17) (2012), pp. 5491-5498. *https://doi.org/10.1016/j.watres.2012.07.018*.

Jones et al. 2011. D. B. Jones, A. Saglam, A. Triger, H. Song, and T. Karanfil. '**I-THM Formation and Speciation: Preformed Monochloramine versus Prechlorination Followed by Ammonia Addition**'. Environmental Science & Technology, 45 (24) (2011), pp. 10429-10437. *https://doi.org/10.1021/es202745t*.

Jones 2009. D. B. Jones. '**The Formation and Control of Iodinated Trihalomethanes in Drinking Water Treatment**'. All Theses. Paper 675 (2009). *https://tigerprints.clemson.edu/all_theses/675*.

Jortner et al. 1961. J. Jortner, R. Levine, M. Ottolenghi, and G. Stein. '**The Photochemistry of the Iodide Ion in Aqueous Solution**'. The Journal of Physical Chemistry, 65 (7) (1961), pp. 1232-1238. *https://doi.org/10.1021/j100825a033*.

Jortner et al. 1964. J. Jortner, M. Ottolenghi, and G. Stein. '**On the Photochemistry of Aqueous Solutions of Chloride, Bromide, and Iodide Ions**'. The Journal of Physical Chemistry, 68 (2) (1964), pp. 247-255. *https://doi.org/10.1021/j100784a005*.

Karpel Vel Leitner et al. 1998. N. Karpel Vel Leitner, J. Vessella, M. Dore, and B. Legube. '**Chlorination and Formation of Organoiodinated Compounds: The Important Role of Ammonia**'. Environmental Science & Technology, 32 (11) (1998), pp. 1680-1685. *https://doi.org/10.1021/es9706662*.

Kern and Kim 1965. D. M. Kern and C-H. Kim. '**Iodine Catalysis in the Chlorite-Iodide Reaction**'. Journal of the American Chemical Society, 87 (23) (1965), pp. 5309-5313. *https://doi.org/10.1021/ja00951a008*.

Khan et al. 2018. R. Khan, I. Hashmi, H. Nasir, S. Rasheed, and L. C. Campos. '**Incidence of Chlorination By-Products in an Institutional Drinking Water Distribution Network, Islamabad, Pakistan, Using Response Surface Methodology**'. Journal of Water, Sanitation and Hygiene for Development, 8 (4) (2018), pp. 740-751. *https://doi.org/10.2166/washdev.2018.047*.

Kim and Herrera 2010. E. J. Kim and J. E. Herrera. '**Characteristics of Lead Corrosion Scales Formed during Drinking Water Distribution and Their Potential Influence on the Release of Lead and Other Contaminants**'. Environmental Science & Technology, 44 (16) (2010), pp. 6054-6061. *https://doi.org/10.1021/es101328u*.

Kimura et al. 2019. S. Y. Kimura, A. A. Cuthbertson, J. D. Byer, and S. D. Richardson. '**The DBP Exposome: Development of a New Method to Simultaneously Quantify Priority Disinfection by-Products and Comprehensively Identify Unknowns**'. Water Research, 148 (2019), pp. 324-333. *https://doi.org/10.1016/j.watres.2018.10.057*.

King and Jacobs 1931. C. V. King and M. B. Jacobs. '**The Oxidation of Iodide Ion by Persulfate Ion. IV. Kinetics of the Reaction in Highly Dilute Aqueous Solution**'. Journal of the American Chemical Society, 53 (5) (1931), pp. 1704-1714. *https://doi.org/10.1021/ja01356a010*.

Korobova 2010. E. Korobova. '**Soil and Landscape Geochemical Factors Which Contribute to Iodine Spatial Distribution in the Main Environmental Components and Food Chain in the Central Russian Plain**'. Journal of Geochemical Exploration, Selenium and iodine anomalies in soils and health, 107 (2) (2010), pp. 180-192. *https://doi.org/10.1016/j.gexplo.2010.03.003*.

Kovalova et al. 2013. L. Kovalova, H. Siegrist, U. von Gunten, J. Eugster, M. Hagenbuch, A. Wittmer, R. Moser, and C. S. McArdell. '**Elimination of Micropollutants during Post-Treatment of Hospital Wastewater with Powdered Activated Carbon, Ozone, and UV**'. Environmental Science & Technology, 47 (14) (2013), pp. 7899-7908. *https://doi.org/10.1021/es400708w*.

Kralchevska et al. 2016. R. P. Kralchevska, V. K. Sharma, L. Machala, and R. Zboril. '**Ferrates (Fe^{VI}, Fe^{V}, and Fe^{IV}) Oxidation of Iodide: Formation of Triiodide**'. Chemosphere, 144 (2016), pp. 1156-1161. *https://doi.org/10.1016/j.chemosphere.2015.09.091*.

Krasner 2012. S. W. Krasner. '**Halogenated DBPs and Emerging Issues**'. In *Disinfection By-Products and Human Health*, IWA Publishing, (2012), pp. 59-71.

Krasner et al. 2004. S. W. Krasner, H. S. Weinberg, S. D. Richardson, S. J. Pastor, R. Chinn, M. J. Sclimenti, G. D. Onstad, and A. D. Thruston. '**Occurrence of a New Generation of Disinfection Byproducts**'. Environmental Science & Technology, 40 (23) (2004), pp. 7175-7185. *https://doi.org/10.1021/es060353j*.

Krasner 2009. S. W. Krasner. '**The Formation and Control of Emerging Disinfection By-Products of Health Concern**'. Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences, 367 (1904) (2009), pp. 4077-4095. *https://doi.org/10.1098/rsta.2009.0108*.

Krasner et al. 2006. S. W. Krasner, H. S. Weinberg, S. D. Richardson, S. J. Pastor, R. Chinn, M. J. Sclimenti, G. D. Onstad, and A. D. Thruston. '**Occurrence of a New Generation of Disinfection Byproducts**'. Environmental Science & Technology, 40 (23) (2006), pp. 7175-7185. *https://doi.org/10.1021/es060353j*.

Kristiana et al. 2009. I. Kristiana, H. Gallard, C. Joll, and J-P. Croué. '**The Formation of Halogen-Specific TOX from Chlorination and Chloramination of Natural Organic Matter Isolates**'. Water Research, 43 (17) (2009), pp. 4177-4186. *https://doi.org/10.1016/j.watres.2009.06.044*.

Kumar et al. 1986. K. Kumar, R. A. Day, and D. W. Margerum. '**Atom-Transfer Redox Kinetics: General-Acid-Assisted Oxidation of Iodide by Chloramines and Hypochlorite**'. Inorganic Chemistry, 25 (1986), pp. 4344-4350. *https://doi.org/10.1021/ic00244a012*.

Küpper et al. 2011. F. C. Küpper, M. C. Feiters, B. Olofsson, T. Kaiho, S. Yanagida, M. B. Zimmermann, L. J. Carpenter, et al. '**Commemorating Two Centuries of Iodine Research: An Interdisciplinary Overview of Current Research**'. Angewandte Chemie International Edition, 50 (49) (2011), pp. 11598-11620. *https://doi.org/10.1002/anie.201100028*.

Le Roux et al. 2015. J. Le Roux, N. Nada, M. T. Khan, and J-P. Croué. '**Tracing Disinfection Byproducts in Full-Scale Desalination Plants**'. Desalination, 359 (2015), pp. 141-148. *https://doi.org/10.1016/j.desal.2014.12.035*.

Lee and von Gunten 2010. Y. Lee and U. von Gunten. '**Oxidative Transformation of Micropollutants during Municipal Wastewater Treatment: Comparison of Kinetic Aspects of Selective (Chlorine, Chlorine Dioxide, Ferrate^{VI}, and Ozone) and Non-Selective Oxidants (Hydroxyl Radical)**'. Water Research, 44 (2) (2010), pp. 555-566. *https://doi.org/10.1016/j.watres.2009.11.045*.

Lee et al. 2014. Y. Lee, R. Kissner, and U. von Gunten. '**Reaction of Ferrate(VI) with ABTS and Self-Decay of Ferrate(VI): Kinetics and Mechanisms**'. Environmental Science & Technology, 48 (9) (2014), pp. 5154-5162. *https://doi.org/10.1021/es500804g*.

Lee et al. 2005. Y. Lee, J. Yoon, and U. von Gunten. '**Kinetics of the Oxidation of Phenols and Phenolic Endocrine Disruptors during Water Treatment with Ferrate (Fe(VI))**'. Environmental Science & Technology, 39 (22) (2005), pp. 8978-8984. *https://doi.org/10.1021/es051198w*.

Legrand et al. 2018. M. Legrand, J. R. McConnell, S. Preunkert, M. Arienzo, N. Chellman, K. Gleason, T. Sherwen, M. J. Evans, and L. J. Carpenter. '**Alpine Ice Evidence of a Three-Fold Increase in Atmospheric Iodine Deposition since 1950 in Europe Due to Increasing Oceanic Emissions**'. Proceedings of the National Academy of Sciences, 115 (48) (2018), pp. 12136-12141. *https://doi.org/10.1073/pnas.1809867115*.

Lengyel et al. 1996. I. Lengyel, J. Li, K. Kustin, and I. R. Epstein. '**Rate Constants for Reactions between Iodine- and Chlorine-Containing Species: A Detailed Mechanism of the Chlorine Dioxide/Chlorite-Iodide Reaction**'. Journal of the American Chemical Society, 118 (15) (1996), pp. 3708-3719. *https://doi.org/10.1021/ja953938e*.

Lente et al. 2009. G. Lente, J. Kalmár, Z. Baranyai, A. Kun, I. Kék, D. Bajusz, M. Takács, L. Veres, and I. Fábián. '**One- Versus Two-Electron Oxidation with Peroxomonosulfate Ion: Reactions with Iron(II), Vanadium(IV), Halide Ions, and Photoreaction with Cerium(III)**'. Inorganic Chemistry, 48 (4) (2009), pp. 1763-1773. *https://doi.org/10.1021/ic801569k*.

Li et al. 2018a. J. Li, J. Jiang, S-Y. Pang, Y. Zhou, Y. Gao, Y. Yang, S. Sun, G. Liu, J. Ma, C. Jiang, L. Wang. '**Transformation of Methylparaben by Aqueous Permanganate in the Presence of Iodide: Kinetics, Modeling, and Formation of Iodinated Aromatic Products**'. Water Research, 135 (2018), pp. 75-84. *https://doi.org/10.1016/j.watres.2018.02.014*.

Li et al. 2017. J. Li, J. Jiang, Y. Zhou, S-Y. Pang, Y. Gao, C. Jiang, J. Ma, Y. Jin, Y. Yang, G. Liu, L. Wang, C. Guan. '**Kinetics of Oxidation of Iodide (I^-) and Hypoiodous Acid (HOI) by Peroxymonosulfate (PMS) and Formation of Iodinated Products in the PMS/I^-/NOM System**'. Environmental Science & Technology Letters, 4 (2) (2017), pp. 76-82. *https://doi.org/10.1021/acs.estlett.6b00471*.

Li et al. 2018b. J. Li, S-Y. Pang, Y. Zhou, S. Sun, L. Wang, Z. Wang, Y. Gao, Y. Yang, and J. Jiang. '**Transformation of Bisphenol AF and Bisphenol S by Manganese Dioxide and Effect of Iodide**'. Water Research, 143 (2018), pp. 47-55. *https://doi.org/10.1016/j.watres.2018.06.029*.

Li and Mitch 2018. X-F. Li and W. A. Mitch. '**Drinking Water Disinfection Byproducts (DBPs) and Human Health Effects: Multidisciplinary Challenges and Opportunities**'. Environmental Science & Technology, 52 (4) (2018), pp. 1681-1689. *https://doi.org/10.1021/acs.est.7b05440*.

Lin et al. 2008. Y-P. Lin, M. P. Washburn, and R. L. Valentine. '**Reduction of Lead Oxide (PbO_2) by Iodide and Formation of Iodoform in the PbO_2/I^-/NOM System**'. Environmental Science & Technology, 42 (8) (2008), pp. 2919-2924. *https://doi.org/10.1021/es702797b*.

Liu et al. 2014. C. Liu, E. Salhi, J-P. Croué, and U. von Gunten. '**Chlorination of Iodide-Containing Waters in the Presence of CuO: Formation of Periodate**'. Environmental Science & Technology, 48 (22) (2014), pp. 13173–13180. *https://doi.org/10.1021/es5032079*.

Liu et al. 2019. J. Liu, J. Wang, J. Zhang, and Y. Mu. '**Iodo-Trihalomethanes Formation during Chlorination and Chloramination of Iodide-Containing Waters in the Presence of Cu^{2+}**'. Science of The Total Environment, 671 (2019), pp. 101-107. *https://doi.org/10.1016/j.scitotenv.2019.03.356*.

Liu et al. 2001. Q. Liu, L. M. Schurter, C. E. Muller, S. Aloisio, J. S. Francisco, and D. W. Margerum. '**Kinetics and Mechanisms of Aqueous Ozone Reactions with Bromide, Sulfite, Hydrogen Sulfite, Iodide, and Nitrite Ions**'. Inorganic Chemistry, 40 (17) (2001), pp. 4436-4442. *https://doi.org/10.1021/ic000919j*.

Liu et al. 2017. S. Liu, Z. Li, H. Dong, B. A. Goodman, and Z. Qiang. '**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 (2017), pp. 28-36. *https://doi.org/10.1016/j.jhazmat.2016.08.071*.

Longvah and Deosthale 1998. T. Longvah and Y. G. Deosthale. '**Iodine Content of Commonly Consumed Foods and Water from the Goitre-Endemic Northeast Region of India**'. Food Chemistry, 61 (3) (1998), pp. 327-331. *https://doi.org/10.1016/S0308-8146(97)00065-4*.

Luh and Mariñas 2014. J. Luh and B. J. Mariñas. '**Kinetics of Bromochloramine Formation and Decomposition**'. Environmental Science & Technology, 48 (5) (2014), pp. 2843-2852. *https://doi.org/10.1021/es4036754*.

Luo et al. 2014. Q. Luo, X. Chen, Z. Wei, X. Xu, D. Wang, and Z. Wang. '**Simultaneous and High-Throughput Analysis of Iodo-Trihalomethanes, Haloacetonitriles, and Halonitromethanes in Drinking Water Using Solid-Phase Microextraction/Gas Chromatography-Mass Spectrometry: An Optimization of Sample Preparation**'. Journal of Chromatography A, 1365 (2014), pp. 45-53. *https://doi.org/10.1016/j.chroma.2014.09.003*.

Luo et al. 2013. Q. Luo, D. Wang, Z. Wei, and Z. Wang. '**Optimized Chromatographic Conditions for Separation of Halogenated Acetic Acids by Ultra-Performance Liquid Chromatography–Electrospray Ionization-Mass Spectrometry**'. Journal of Chromatography A, 1277 (2013), pp. 26-34. *https://doi.org/10.1016/j.chroma.2012.12.046*.

Maberly et al. 1981. G. F. Maberly, C. J. Eastman, and J. M. Corcoran. '**Effect of Iodination of a Village Water-Supply on Goitre and Thyroid Function**'. The Lancet, 318 (8258) (1981), pp. 1270-1272. *https://doi.org/10.1016/S0140-6736%2881%2991502-6*.

Machek 2015. E. J. Machek Jr. '**Iopamidol as a Precursor to Iodinated Disinfection Byproduct (DBP) Formation as a Function of NOM Concentration, pH, and Chlorinated Oxidants**'. University of Akron, (2015), 294 p. *http://rave.ohiolink.edu/etdc/view?acc_num=akron1439045967*.

Magdeburg et al. 2014. A. Magdeburg, D. Stalter, M. Schlüsener, T. Ternes, and J. Oehlmann. '**Evaluating the Efficiency of Advanced Wastewater Treatment: Target Analysis of Organic Contaminants and (Geno-)Toxicity Assessment Tell a Different Story**'. Water Research, 50 (2014), pp. 35-47. *https://doi.org/10.1016/j.watres.2013.11.041*.

Martell 1976. A. Martell. **Critical Stability Constants: Inorganic Complexes**. Springer US, (1976), 257 p. *https://www.springer.com/gp/book/9781475755084*.

Martino et al. 2009. M. Martino, G. P. Mills, J. Woeljen, and P. S. Liss. '**A New Source of Volatile Organoiodine Compounds in Surface Seawater**'. Geophysical Research Letters, 36 (1) (2009). *http://dx.doi.org/10.1029/2008GL036334*.

McAskill 2010. J. B. McAskill. '**Novel Pathways for the Formation of Iodinated Disinfection By-Products**'. MS (Master of Science) thesis, University of Iowa, (2010), 104 p. *http://ir.uiowa.edu/etd/547*.

Mian et al. 2018. H. R. Mian, G. Hu, K. Hewage, M. J. Rodriguez, and R. Sadiq. '**Prioritization of Unregulated Disinfection By-Products in Drinking Water Distribution Systems for Human Health Risk Mitigation: A Critical Review**'. Water Research, 147 (2018), pp. 112-131. *https://doi.org/10.1016/j.watres.2018.09.054*.

Milenković and Stanisavljev 2012. M. C. Milenković and D. R. Stanisavljev. '**Role of Free Radicals in Modeling the Iodide–Peroxide Reaction Mechanism**'. The Journal of Physical Chemistry A, 116 (23) (2012), pp. 5541-5548. *https://doi.org/10.1021/jp303732u*.

Millero et al. 2008. F. J. Millero, R. Feistel, D. G. Wright, and T. J. Mc Dougall. '**The Composition of Standard Seawater and the Definition of the Reference-Composition Salinity Scale**'. Deep Sea Research Part I: Oceanographic Research Papers, 55 (1) (2008), pp. 50-72. *https://doi.org/10.1016/j.dsr.2007.10.001*.

Mishra et al. 2000. S. Mishra, V. Singh, A. Jain, and K. K. Verma. '**Determination of Iodide by Derivatization to 4-Iodo-N,N-Dimethylaniline and Gas Chromatography–Mass Spectrometry**'. The Analyst, 125 (3) (2000), pp. 459-464. *https://doi.org/10.1039/a908363d*.

Mohammad et al. 2010. A. M. Mohammad, M. I. Awad, and T. Ohsaka. '**Study of the Autocatalytic Chlorate–Triiodide Reaction in Acidic and Neutral Media**'. Journal of Advanced Research, 1 (3) (2010), pp. 209-214. *https://doi.org/10.1016/j.jare.2010.05.003*.

Montesinos and Gallego 2014. I. Montesinos and M. Gallego. '**How the Inclusion of Treated Water in Beverages Influences the Appearance of Halogenated Volatile Organic Compounds**'. Journal of Agricultural and Food Chemistry, 62 (42) (2014), pp. 10240-10247. *https://doi.org/10.1021/jf503431q*.

Moran et al. 2002. J. E. Moran, S. D. Oktay, and P. H. Santschi. '**Sources of Iodine and Iodine 129 in Rivers**'. Water Resources Research, 38 (8) (2002), pp 24-1-24-10. *https://doi.org/10.1029/2001WR000622*.

Morgan 1954. K. J. Morgan. '**Some Reactions of Inorganic Iodine Compounds**'. Quarterly Reviews, Chemical Society, 8 (2) (1954), pp. 123-146. *https://doi.org/10.1039/QR9540800123*.

Morris 1975. J. C. Morris. **Formation of Halogenated Organics by Chlorination of Water Supplies**. Research Reporting Series: Environmental Monitoring - US ENVIRONMENTAL MONITORING Series. Office of Research and Development, U.S. Environmental Protection Agency, (1975).

Nagy et al. 1988. J. C. Nagy, K. Kumar, and D. W. Margerum. '**Non-Metal Redox Kinetics: Oxidation of Iodide by Hypochlorous Acid and by Nitrogen Trichloride Measured by the Pulsed-Accelerated-Flow Method**'. Inorganic Chemistry, 27 (No. 16) (1988), pp. 2773-2780. *https://doi.org/10.1021/ic00289a007*.

National Research Council (US) Safe Drinking Water Committee 1987. National Research Council (US) Safe Drinking Water Committee. '**Disinfection Methods and Efficacy**'. In *Drinking Water and Health: Disinfectants and Disinfectant By-Products*, 7. National Academies Press (US), (1987). *https://www.ncbi.nlm.nih.gov/books/NBK217999/*.

Neal et al. 2007. C. Neal, M. Neal, H. Wickham, L. Hill, and S. Harman. '**Dissolved Iodine in Rainfall, Cloud, Stream and Groundwater in the Plynlimon Area of Mid-Wales**'. Hydrology and Earth System Sciences, 11 (1) (2007), pp. 283-293. *https://doi.org/10.5194/hess-11-283-2007*.

NHMRC and NRMMC 2011. NHMRC and NRMMC. **Australian Drinking Water Guidelines Paper 6 National Water Quality Management** Strategy, 1. National Health and Medical Research Council, National Resource Management Ministerial Council, Commonwealth of Australia, Canberra, (2011). *https://www.clearwater.asn.au/user-data/resource-files/Aust_drinking_water_guidelines.pdf*.

Ning and Graham 2008. B. Ning and N. J. D. Graham. '**Ozone Degradation of Iodinated Pharmaceutical Compounds**'. Journal of Environmental Engineering, 134 (12) (2008), 43 p. *https://doi.org/10.1061/(ASCE)0733-9372(2008)134:12(944)*.

Opoku-Agyeman 2010. B. Opoku-Agyeman. '**New Method for the Synthesis of Iodohydrins**'. Indiana University of Pennsylvania, (2010). *http://knowledge.library.iup.edu/cgi/viewcontent.cgi?article=1930&context=etd*.

Palmer et al. 1984. D. A. Palmer, R. W. Ramette, and R. E. Mesmer. '**Triiodide Ion Formation Equilibrium and Activity Coefficients in Aqueous Solution**'. Journal of Solution Chemistry, 13 (9) (1984), pp. 673-683. *https://doi.org/10.1007/BF00650374*.

Pan et al. 2016a. Y. Pan, W. Li, H. An, H. Cui, and Y. Wang. '**Formation and Occurrence of New Polar Iodinated Disinfection Byproducts in Drinking Water**'. Chemosphere, 144 (2016), pp. 2312-2320. *https://doi.org/10.1016/j.chemosphere.2015.11.012*.

Pan and Zhang 2013. Y. Pan and X. Zhang. '**Total Organic Iodine Measurement: A New Approach with UPLC/ESI-MS for off-Line Iodide Separation/Detection**'. Water Research, 47 (1) (2013), pp. 163-172. *https://doi.org/10.1016/j.watres.2012.09.040*.

Pan et al. 2016b. Y. Pan, X. Zhang, and Y. Li. '**Identification, Toxicity and Control of Iodinated Disinfection Byproducts in Cooking with Simulated Chlor(Am)Inated Tap Water and Iodized Table Salt**'. Water Research, 88 (2016), pp. 60-68. *https://doi.org/10.1016/j.watres.2015.10.002*.

Paquette and Ford 1985. J. Paquette and B. L. Ford. '**Iodine Chemistry in the +1 Oxidation State. I. The Electronic Spectra of OI^-, HOI, and H_2OI^+**'. Canadian Journal of Chemistry, 63 (9) (1985), pp. 2444-2448. *https://doi.org/10.1139/v85-404*.

Parsons et al. 2009. S. A. Parsons, E. H. Goslan, S. A. Rocks, P. Holmes, L. S. Levy, and S. W. Krasner. '**Study into the Formation of Disinfection By-Products of Chloramination, Potential Health Implications and Techniques for Minimisation**'. Cranfield University, (2009). *http://www.windsofjustice.org.uk/wp-content/uploads/2015/07/THMs-Cranfield-University.pdf.*

Pechtl et al. 2007. S. Pechtl, G. Schmitz, and R. von Glasow. '**Modelling Iodide? Iodate Speciation in Atmospheric Aerosol: Contributions of Inorganic and Organic Iodine Chemistry**'. Atmospheric Chemistry and Physics, 7 (5) (2007), pp. 1381-1393. *https://doi.org/10.5194/acp-7-1381-2007*.

Pejić et al. 2012. N. D. Pejić, J. P. Maksimović, S. M. Blagojević, S. R. Anić, Ž. D. Čupić, and L. Z. Kolar-Anić. '**Kinetic Analytical Method for Determination of Uric Acid in Human Urine Using Analyte Pulse Perturbation Technique**'. Journal of the Brazilian Chemical Society, 23 (8) (2012), pp. 1450-1459. *https://doi.org/10.1590/S0103-50532012005000006*.

Pérez Pavón et al. 2008. J. L. Pérez Pavón, S. Herrero-Martín, C. García Pinto, and B. Moreno Cordero. '**Determination of Trihalomethanes in Water Samples: A Review**'. Analytica Chimica Acta, 629 (1–2) (2008), pp. 6-23. *https://doi.org/10.1016/j.aca.2008.09.042*.

Plewa et al. 2004. M. J. Plewa, E. D. Wagner, S. D. Richardson, A. D. Thruston, Y-T. Woo, and A. B. McKague. '**Chemical and Biological Characterization of Newly Discovered Iodoacid Drinking Water Disinfection Byproducts**'. Environmental Science & Technology, 38 (18) (2004), pp. 4713-4722. *https://doi.org/10.1021/es049971v*.

Plewa et al. 2017. M. J. Plewa, E. D. Wagner, and S. D. Richardson. '**TIC-Tox: A Preliminary Discussion on Identifying the Forcing Agents of DBP-Mediated Toxicity of Disinfected Water**'. Journal of Environmental Sciences, 58 (2017), pp. 208-216. *https://doi.org/10.1016/j.jes.2017.04.014*.

Postigo et al. 2016. C. Postigo, C. I. Cojocariu, S. D. Richardson, P. J. Silcock, and D. Barceló. '**Characterization of Iodinated Disinfection By-Products in Chlorinated and Chloraminated Waters Using Orbitrap Based Gas Chromatography-Mass Spectrometry**'. Analytical and Bioanalytical Chemistry, 408 (13) (2016), pp. 3401-3411. *https://doi.org/10.1007/s00216-016-9435-x*.

Postigo et al. 2018a. C. Postigo, D. M. DeMarini, M. D. Armstrong, H. K. Liberatore, K. Lamann, S. Y. Kimura, A. A. Cuthbertson, S. H. Warren, S. D. Richardson, T. McDonald, Y. M. Sey, N. O. B. Ackerson, S. E. Duirk, and J. E. Simmons. '**Chlorination of Source Water Containing Iodinated X-Ray Contrast Media: Mutagenicity and Identification of New Iodinated Disinfection Byproducts**'. Environmental Science & Technology, 52 (22) (2018), pp. 13047-13056. *https://doi.org/10.1021/acs.est.8b04625*.

Postigo et al. 2018b. C. Postigo, P. Emiliano, D. Barceló, and F. Valero. '**Chemical Characterization and Relative Toxicity Assessment of Disinfection Byproduct Mixtures in a Large Drinking Water Supply Network**'. Journal of Hazardous Materials, 359 (2018), pp. 166-173. *https://doi.org/10.1016/j.jhazmat.2018.07.022*.

Postigo et al. 2017. C. Postigo, S. D. Richardson, and D. Barceló. '**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 (2017), pp. 127-134. *https://doi.org/10.1016/j.jes.2017.04.009*.

Preedy et al. 2009. V. R. Preedy, G. N. Burrow, and R. R. Watson. **Comprehensive Handbook of Iodine: Nutritional, Biochemical, Pathological and Therapeutic Aspects**. Academic Press. Elsevier, (2009).

Prütz et al. 2001. W. A. Prütz, R. Kissner, T. Nauser, and W. H. Koppenol. '**On the Oxidation of Cytochrome c by Hypohalous Acids**'. Archives of Biochemistry and Biophysics, 389 (1) (2001), pp. 110-122. *https://doi.org/10.1006/abbi.2001.2321*.

Rahman 2015. S. S. Rahman. '**Effect of pH and Temperature on Halogenated DBPs**'. Master of Science (MS), South Dakota State University, (2015), 54 p. *https://openprairie.sdstate.edu/etd/5*.

Rahn 1993. R. O. Rahn. '**Use of Potassium Iodide as a Chemical Actinometer**'. Photochemistry and Photobiology, 58 (6) (1993), pp. 874-880. *https://doi.org/10.1111/j.1751-1097.1993.tb04986.x*.

Rahn 1997. R. O. Rahn. '**Potassium Iodide as a Chemical Actinometer for 254 nm Radiation: Use of Iodate as an Electron Scavenger**'. Photochemistry and Photobiology, 66 (4) (1997), pp. 450-455. *https://doi.org/10.1111/j.1751-1097.1997.tb03172.x*.

Reckhow et al. 1990. D. A. Reckhow, P. C. Singer, and R. L. Malcolm. '**Chlorination of Humic Materials: Byproduct Formation and Chemical Interpretations**'. Environmental Science & Technology, 24 (11) (1990), pp. 1655-1664. *https://doi.org/10.1021/es00081a005*.

Richardson et al. 2008. S. D. Richardson, F. Fasano, J. J. Ellington, F. G. Crumley, K. M. Buettner, J. J. Evans, B. C. Blount, L. K. Silva, T. J. Waite, G. W. Luther. '**Occurrence and Mammalian Cell Toxicity of Iodinated Disinfection Byproducts in Drinking Water**'. Environmental Science & Technology, 42 (22) (2008), pp. 8330-8338. *https://doi.org/10.1021/es801169k*.

Richardson and Postigo 2016. S. D. Richardson and C. Postigo. '**A New Technique Helps to Uncover Unknown Peptides and Disinfection By-Products in Water**'. Journal of Environmental Sciences, 42 (2016), pp. 6-8. *https://doi.org/10.1016/j.jes.2016.03.008*.

Richardson 2008. S. D. Richardson. '**Environmental Mass Spectrometry: Emerging Contaminants and Current Issues**'. Analytical Chemistry, 80 (12) (2008), pp. 4373-4402. *https://doi.org/10.1021/ac800660d*.

Richardson and Kimura 2016. S. D. Richardson and S. Y. Kimura. '**Water Analysis: Emerging Contaminants and Current Issues**'. Analytical Chemistry, 88 (1) (2016), pp. 546-582. *https://doi.org/10.1021/acs.analchem.5b04493*.

Richardson and Ternes 2017. S. D. Richardson and T. A. Ternes. '**Water Analysis: Emerging Contaminants and Current Issues**'. Analytical Chemistry, 90 (1) (2017), pp. 398-428. *https://doi.org/10.1021/acs.analchem.7b04577*.

Risher et al. 2004. J. Risher, G. Diamond, S. G. Swarts, and R. Amata. '**Toxicological Profile for Iodine**'. U.S. Department of Health and Human Services / Agency for Toxic Substances and Disease Registry, (2004).

Rook 1974. J. J. Rook. '**Formation of Haloforms during Chlorination of Natural Waters**'. Water Treatment and Examination, 23 (1974), pp. 234-243.

Rose and Roberts (2019). M. R. Rose and A. L. Roberts. '**Iodination of Dimethenamid in Chloraminated Water: Active Iodinating Agents and Distinctions between Chlorination, Bromination, and Iodination**'. Environmental Science & Technology, 53 (2019), pp. 11764-11773. *https://doi.org/10.1021/acs.est.9b03645*.

Rougé et al. 2018. V. Rougé, S. Allard, J-P. Croué, and U. von Gunten. '**In Situ Formation of Free Chlorine During ClO_2 Treatment: Implications on the Formation of Disinfection Byproducts**'. Environmental Science & Technology, 52 (22) (2018), pp. 13421-13429. *https://doi.org/10.1021/acs.est.8b04415*.

Rumsby et al. 2009. P. Rumsby, L. Lawton, A. Ewence, L. Rockett, T. Hall, H. James, and P. Jackson. '**Review of the Current Toxicological and Occurrence Information Available on Iodinated Disinfection By-Products**'. DEFRA 7883.03 (DWI 70/2/233). WRc (2009).

Saiz-Lopez et al. 2014. A. Saiz-Lopez, R. P. Fernandez, C. Ordóñez, D. E. Kinnison, J. C. Gómez Martín, J-F. Lamarque, and S. Tilmes. '**Iodine Chemistry in the Troposphere and Its Effect on Ozone**'. Atmospheric Chemistry and Physics, 14 (23) (2014), pp. 13119-13143. *https://doi.org/10.5194/acp-14-13119-2014*.

Salhi and von Gunten 1999. E. Salhi and U. von Gunten. '**Simultaneous Determination of Bromide, Bromate and Nitrite in Low µg L^{-1} Levels by Ion Chromatography without Sample Pretreatment**'. Water Research, 33 (15) (1999), pp. 3239-3244. *https://doi.org/10.1016/S0043-1354(99)00053-6*.

Salminen et al. 2005. R. Salminen, M. J. Batista, M. Bidovec, and T. Tarvainen. '**Geochemical Atlas of Europe. Part 1 - Background Information, Methodology and Maps**'. Geological Survey of Finland, Otamedia Oy, Espoo, (2005). *http://weppi.gtk.fi/publ/foregsatlas/text/I.pdf*.

Sánchez-Polo et al. 2006. M. Sánchez-Polo, J. Rivera-Utrilla, E. Salhi, and U. von Gunten. '**Removal of Bromide and Iodide Anions from Drinking Water by Silver-Activated Carbon Aerogels**'. Journal of Colloid and Interface Science, 300 (1) (2006), pp. 437-441. *https://doi.org/10.1016/j.jcis.2006.03.037*.

Sarin et al. 2001. P. Sarin, V. L. Snoeyink, J. Bebee, W. M. Kriven, and J. A. Clement. '**Physico-Chemical Characteristics of Corrosion Scales in Old Iron Pipes**'. Water Research, 35 (12) (2001), pp. 2961-2969. *https://doi.org/10.1016/S0043-1354(00)00591-1*.

Sayess et al. 2017. R. Sayess, A. Khalil, M. Shah, D. A. Reckhow, and K. J. Godri Pollitt. '**Comparative Cytotoxicity of Six Iodinated Disinfection Byproducts on Nontransformed Epithelial Human Colon Cells**'. Environmental Science & Technology Letters, 4 (4) (2017), pp. 143-148. *https://doi.org/10.1021/acs.estlett.7b00064*.

Sayess and Reckhow 2017. R. Sayess and D. A. Reckhow. '**An Improved Method for Total Organic Iodine in Drinking Water**'. Water Research, 108 (2017), pp. 250-259. *https://doi.org/10.1016/j.watres.2016.10.079*.

Schlegel et al. 2006. M. L. Schlegel, P. Reiller, F. Mercier-Bion, N. Barré, and V. Moulin. '**Molecular Environment of Iodine in Naturally Iodinated Humic Substances: Insight from X-Ray Absorption Spectroscopy**'. Geochimica et Cosmochimica Acta, 70 (22) (2006), pp. 5536-5551. *https://doi.org/10.1016/j.gca.2006.08.026*.

Schulz et al. 2008. M. Schulz, D. Löffler, M. Wagner, and T. A. Ternes. '**Transformation of the X-Ray Contrast Medium Iopromide In Soil and Biological Wastewater Treatment**'. Environmental Science & Technology, 42 (19) (2008), pp. 7207-7217. *https://doi.org/10.1021/es800789r*.

Sedlak and von Gunten 2011. D. S. Sedlak and U. von Gunten. '**The Chlorine Dilemma**'. Science, 331 (6013) (2011).

Seki et al. 2013. M. Seki, J-i. Oikawa, T. Taguchi, T. Ohnuki, Y. Muramatsu, K. Sakamoto, and S. Amachi. '**Laccase-Catalyzed Oxidation of Iodide and Formation of Organically Bound Iodine in Soils**'. Environmental Science & Technology, 47 (1) (2013), pp. 390-397. *https://doi.org/10.1021/es303228n*.

Serrano et al. 2015. M. Serrano, I. Montesinos, M. J. Cardador, M. Silva, and M. Gallego. '**Seasonal Evaluation of the Presence of 46 Disinfection By-Products throughout a Drinking Water Treatment Plant**'. Science of The Total Environment, 517 (2015), pp. 246-258. *https://doi.org/10.1016/j.scitotenv.2015.02.070*.

Sfynia et al. 2017. C. Sfynia, T. Bond, N. Ganidi, R. Kanda, and M. R. Templeton. '**Predicting the Formation of Haloacetonitriles and Haloacetamides by Simulated Distribution System Tests**'. Procedia Engineering, XVIII

International Conference on Water Distribution Systems, WDSA2016, 186 (2017), pp. 186-192. *https://doi.org/10.1016/j.proeng.2017.03.226*.

Shackelford and Keith 1976. W. M. Shackelford and L. H. Keith. '**Frequency of Organic Compounds Identified in Water**'. Athens, Georgia: US Environmental Protection Agency, (1976). EPA-600/4-76-062.

Sha et al. 2015. A. D. Shah, Z-Q. Liu, E. Salhi, T. Höfer, and U. von Gunten. '**Peracetic Acid Oxidation of Saline Waters in the Absence and Presence of H_2O_2: Secondary Oxidant and Disinfection Byproduct Formation**'. Environmental Science & Technology, 49 (3) (2015), pp. 1698-1705. *https://doi.org/10.1021/es503920n*.

Sharma et al. 1999. S. K. Sharma, P. K. Chelleng, S. Gogoi, and J. Mahanta. '**Iodine Status of Food and Drinking Water of a Sub-Himalayan Zone of India**'. International Journal of Food Sciences and Nutrition, 50 (2) (1999), pp. 95-98. *https://doi.org/10.1080/096374899101274*.

Sharma 2011. V. K. Sharma. '**Oxidation of Inorganic Contaminants by Ferrates (VI, V, and IV)–Kinetics and Mechanisms: A Review**'. Journal of Environmental Management, 92 (4) (2011), pp. 1051-1073. *https://doi.org/10.1016/j.jenvman.2010.11.026*.

Sharma et al. 2015. V. K. Sharma, R. Zboril, and R. S. Varma. '**Ferrates: Greener Oxidants with Multimodal Action in Water Treatment Technologies**'. Accounts of Chemical Research, 48 (2) (2015), pp. 182-191. *https://doi.org/10.1021/ar5004219*.

Sherwen et al. 2016. T. Sherwen, M. J. Evans, L. J. Carpenter, S. J. Andrews, R. T. Lidster, B. Dix, T. K. Koenig, R. Sinreich, I. Ortega, R. Volkamer, A. Saiz-Lopez, C. Prados-Roman, A. S. Mahakan, and C. Ordonez. '**Iodine's Impact on Tropospheric Oxidants: A Global Model Study in GEOS-Chem**'. Atmospheric Chemistry and Physics, 16 (2016), pp. 1161-1186. *https://doi.org/10.5194/acp-16-1161-2016*.

Shetaya et al. 2012. W. H. Shetaya, S. D. Young, M. J. Watts, E. L. Ander, and E. H. Bailey. '**Iodine Dynamics in Soils**'. Geochimica et Cosmochimica Acta, 77 (2012), pp. 457-473. *https://doi.org/10.1016/j.gca.2011.10.034*.

Shi et al. 2012. H. Shi, X. Chen, Q. Wu, R. Mu, and F. Ma. '**Assessment and Removal of Emerging Water Contaminants**'. Journal of Environmental & Analytical Toxicology, 2 (7) (2012), 14 p. *https://doi.org/10.4172/2161-0525.S2-003*.

Shin et al. 1996. H-S. Shin, Y-S. Oh-Shin, J-H. Kim, and J-K. Ryu. '**Trace Level Determination of Iodide, Iodine and Iodate by Gas Chromatography-Mass Spectrometry**'. Journal of Chromatography A, 732 (2) (1996), pp. 327-333. *https://doi.org/10.1016/0021-9673(95)01281-8*.

Shin et al. 2018. J. Shin, U. von Gunten, D. A. Reckhow, S. Allard, and Y. Lee. '**Reactions of Ferrate(VI) with Iodide and Hypoiodous Acid: Kinetics, Pathways, and Implications for the Fate of Iodine during Water Treatment**'. Environmental Science & Technology, 52 (13) (2018), pp. 7458-7467. *https://doi.org/10.1021/acs.est.8b01565*.

Sigalla 1961. J. Sigalla. '**N° 52. — Les Réactions de l'iode En Solution Alcaline Un Nouveau Composé I_2OH^-**'. Journal de Chimie Physique, 58 (1961), pp. 602-605.

Smith et al. 2010. E. M. Smith, M. J. Plewa, C. L. Lindell, S. D. Richardson, and W. A. Mitch. '**Comparison of Byproduct Formation in Waters Treated with Chlorine and Iodine: Relevance to Point-of-Use Treatment**'. Environmental Science & Technology, 44 (22) (2010), pp. 8446-8452. *https://doi.org/10.1021/es102746u*.

Smyth and Johnson 2011. D. Smyth and C. C. Johnson. '**Distribution of Iodine in Soils of Northern Ireland**'. Geochemistry: Exploration, Environment, Analysis, 11 (1) (2011), pp. 25-39. *https://doi.org/10.1144/1467-7873/09-015*.

Söderlund et al. 2011. M. Söderlund, M. Lusa, J. Lehto, M. Hakanen, K. Vaaramaa, and A.-M. Lahdenperä. '**Sorption of Iodine, Chlorine, Technetium and Cesium in Soil**'. POSIVA-WR--11-4. Posiva Oy (2011). *http://inis.iaea.org/Search/search.aspx?orig_q=RN:43068659*.

Spellman 2013. F. R. Spellman. **Handbook of Water and Wastewater Treatment Plant Operations**, Third Edition. CRC Press, (2013).

Standing Committee on Biocidal Products 2013. Standing Committee on Biocidal Products. '**Iodine (Including PVP-Iodine) Product Types 1, 3, 4, 22**'. Assessment Report, (2013). *http://dissemination.echa.europa.eu/Biocides/ActiveSubstances/1349-01/1349-01_Assessment_Report.pdf*.

Stanisavljev et al. 2011. D. R. Stanisavljev, M. C. Milenković, M. D. Mojović, and A. D. Popović-Bijelić. '**Oxygen Centered Radicals in Iodine Chemical Oscillators**'. The Journal of Physical Chemistry A, 115 (27) (2011), pp. 7955-7958. *https://doi.org/10.1021/jp203601w*.

Suess et al. 2019. E. Suess, F. Aemisegger, J. E. Sonke, M. Sprenger, H. Wernli, and L. H. E. Winkel. '**Marine versus Continental Sources of Iodine and Selenium in Rainfall at Two European High-Altitude Locations**'. Environmental Science & Technology, 53 (4) (2019), pp. 1905-1917. *https://doi.org/10.1021/acs.est.8b05533*.

Symons et al. 1998. J. M. Symons, R. Xia, G. E. Speitel Jr, A. C. Diehl, C. J. Hwang, S. W. Krasner, and S. E. Barrett. '**Factors Affecting Disinfection By-Product Formation During Chloramination**'. AWWA Research Foundation and American Water Works Association, (1998), 330 p.

Takaku et al. 1995. Y. Takaku, T. Shimamura, K. Masuda, and Y. Igarashi. '**Iodine Determination in Natural and Tap Water Using Inductively Coupled Plasma Mass Spectrometry**'. Analytical Sciences, 11 (5) (1995), pp. 823-827. *https://doi.org/10.2116/analsci.11.823*.

Tang et al. 2013. Q. Tang, Q. Xu, F. Zhang, Y. Huang, J. Liu, X. Wang, Y. Yang, and X. Liu. '**Geochemistry of Iodine-Rich Groundwater in the Taiyuan Basin of Central Shanxi Province, North China**'. Journal of Geochemical Exploration, Arsenic, Fluoride and Iodine in Groundwater of China, 135 (2013), pp. 117-123. *https://doi.org/10.1016/j.gexplo.2012.08.019*.

Terhalle et al. 2018. J. Terhalle, P. Kaiser, M. Jütte, J. Buss, S. Yasar, R. Marks, H. Uhlmann, T. C. Schmidt, and H. V. Lutze. '**Chlorine Dioxide—Pollutant Transformation and Formation of Hypochlorous Acid as a Secondary Oxidant**'. Environmental Science & Technology, 52 (17) (2018), pp. 9964-9971. *https://doi.org/10.1021/acs.est.8b01099*.

Ternes et al. 2003. T. A. Ternes, J. Stüber, N. Herrmann, D. McDowell, A. Ried, M. Kampmann, and B. Teiser. '**Ozonation: A Tool for Removal of Pharmaceuticals, Contrast Media and Musk Fragrances from Wastewater?**' Water Research, 37 (8) (2003), pp. 1976-1982. *https://doi.org/10.1016/S0043-1354(02)00570-5*.

Terryn et al. 2017. R. J. Terryn, C. A. Huerta-Aguilar, J. C. Baum, and V. K. Sharma. '**Fe^{VI}, Fe^{V}, and Fe^{IV} Oxidation of Cyanide: Elucidating the Mechanism Using Density Functional Theory Calculations**'. Chemical Engineering Journal, 330 (2017), pp. 1272-1278. *https://doi.org/10.1016/j.cej.2017.08.080*.

The European Commission 2018. The European Commission. ***ANNEXES to the Proposal for a Directive of the European Parliament and of the Council on the Quality of Water Intended for Human Consumption (Recast)***. (2018). *https://eur-lex.europa.eu/resource.html?uri=cellar:8c5065b2-074f-11e8-b8f5-01aa75ed71a1.0016.02/DOC_2&format=PDF*.

Tian et al. 2014. F-X. Tian, B. Xu, Y-L. Lin, C-Y. Hu, T-Y. Zhang, S-J. Xia, and N-Y. Gao. '**Photodegradation Kinetics of Iopamidol by UV Irradiation and Enhanced Formation of Iodinated Disinfection By-Products in Sequential Oxidation Processes**'. Water Research, 58 (2014), pp. 198-208. *https://doi.org/10.1016/j.watres.2014.03.069*.

Tian et al. 2017a. F-X. Tian, B. Xu, Y-L. Lin, C-Y. Hu, T-Y. Zhang, S-J. Xia, W-H. Chu, and N-Y. Gao. '**Chlor(Am)Ination of Iopamidol: Kinetics, Pathways and Disinfection by-Products Formation**'. Chemosphere, 184 (2017), pp. 489-497. *https://doi.org/10.1016/j.chemosphere.2017.06.012*.

Tian et al. 2017b. F-X. Tian, X-J. Hu, B. Xu, T-Y. Zhang, and Y-Q. Gao. '**Phototransformation of Iodate by UV Irradiation: Kinetics and Iodinated Trihalomethane Formation during Subsequent Chlor(Am)Ination**'. Journal of Hazardous Materials, 326 (2017), pp. 138-144. *https://doi.org/10.1016/j.jhazmat.2016.12.012*.

Trofe et al. 1980. T. W. Trofe, G. W. Jr. Inman, and J. D. Johnson. '**Kinetics of Monochloramine Decomposition in the Presence of Bromide**'. Environmental Science & Technology, 14 (5) (1980), pp. 544-549. *https://doi.org/10.1021/es60165a008*.

Troy and Margerum 1991. R. C. Troy and D. W. Margerum. '**Non-Metal Redox Kinetics: Hypobromite and Hypobromous Acid Reactions with Iodide and with Sulfite and the Hydrolysis of Bromosulfate**'. Inorganic Chemistry, 30 (18) (1991), pp. 3538-3543. *https://doi.org/10.1021/ic00018a028*.

Truesdalea and Jones 1996. V. W. Truesdalea and S. D. Jones. '**The Variation of Iodate and Total Iodine in Some UK Rainwaters during 1980–1981**'. Journal of Hydrology, 179 (1) (1996), pp. 67-86. *https://doi.org/10.1016/0022-1694(95)02873-0*.

Tugulea et al. 2018. A-C. Tugulea, R. Aranda-Rodriguez, D. Bérubé, M. Giddings, F. Lemieux, J. Hnatiw, L. Dabeka, and F. Breton. '**The Influence of Precursors and Treatment Process on the Formation of Iodo-THMs in Canadian Drinking Water**'. Water Research, 130 (2018), pp. 215-223. *https://doi.org/10.1016/j.watres.2017.11.055*.

Tugulea et al. 2015. A-C. Tugulea, R. Aranda-Rodriguez, M. Giddings, F. Lemieux, J. Hnatiw. '**Iodo-THM Formation During Chlorination of Source Waters With Naturally Occurring Ammonium and High Sodium Content**'. In *Disinfection By-Products in Drinking Water*, Edited by K. C. Thompson, S. Gillespie, E. Goslan. The Royal Society of Chemistry, 352 (2015), pp. 46-58.

Urbansky et al. 1997. E. T. Urbansky, B. T. Cooper, and D. W. Margerum. '**Disproportionation Kinetics of Hypoiodous Acid As Catalyzed and Suppressed by Acetic Acid–Acetate Buffer**'. Inorganic Chemistry, 36 (7) (1997), pp 1338-1344. *https://doi.org/10.1021/ic960745z*.

Valent and Ševčík 1998. I. Valent, and P. Ševčík. '**Simulations of the Iodine Interphase Transport Effect on the Oscillating Bray- Liebhafsky Reaction**'. The Journal of Physical Chemistry A, 102 (39) (1998), pp. 7576-7579. *https://doi.org/10.1021/jp982323b*.

Varga et al. 2010. D. Varga, I. Nagypál, and A. K. Horváth. '**Complex Kinetics of a Landolt-Type Reaction: The Later Phase of the Thiosulfate–Iodate Reaction**'. The Journal of Physical Chemistry A, 114 (18) (2010), pp. 5752-5758. *https://doi.org/10.1021/jp101458e*.

Vigneri et al. 1993. R. Vigneri, R. Catalfamo, V. Freni, A. Ippolito, G. L. La Rosa, and C. Regalbuto. '**Iodine Supplementation to a Public Water Supply: An Efficient Method for Correcting Iodine Deficiency**'. In *Iodine Deficiency in Europe: A Continuing Concern*. Edited by F. Delange, J. T. Dunn, D. Glinoer. NATO Science Series A: Life Sciences, 241 (1993), pp. 275-281.

Vikesland et al. 2013. P. J. Vikesland, E. M. Fiss, K. R. Wigginton, K. McNeill, and W. A. Arnold. '**Halogenation of Bisphenol-A, Triclosan, and Phenols in Chlorinated Waters Containing Iodide**'. Environmental Science & Technology, 47 (13) (2013), pp. 6764-6772. *https://doi.org/10.1021/es304927j*.

Vogt et al. 1999. R. Vogt, R. Sander, R. von Glasow, and P. J. Crutzen. '**Iodine Chemistry and Its Role in Halogen Activation and Ozone Loss in the Marine Boundary Layer: A Model Study**'. Journal of Atmospheric Chemistry, 32 (3) (1999), pp. 375-395. *https://doi.org/10.1023/A:100617990*.

Von Gunten 2003. U. von Gunten. '**Ozonation of Drinking Water: Part I. Oxidation Kinetics and Product Formation**'. Water Research, 37 (7) (2003), pp. 1443-1467. *https://doi.org/10.1016/S0043-1354(02)00457-8*.

Von Gunten 2003. U. von Gunten. '**Ozonation of Drinking Water: Part II. Disinfection and by-Product Formation in Presence of Bromide, Iodide or Chlorine**'. Water Research, 37 (7) (2003), pp. 1469-1487. *https://doi.org/10.1016/S0043-1354(02)00458-X*.

Von Gunten 2018. U. von Gunten. '**Oxidation Processes in Water Treatment: Are We on Track?**' Environmental Science & Technology, 52 (9) (2018), pp. 5062-5075. *https://doi.org/10.1021/acs.est.8b00586*.

Von Sonntag and von Gunten 2012. C. von Sonntag and U. von Gunten. **Chemistry of Ozone in Water and Waste-Water Treatment: From Basic Principles to Applications**. IWA Publishing, (2012). *http://dx.doi.org/10.2166/9781780400839*.

Voutchkova et al. 2014. D. Voutchkova, S. Kristiansen, B. Hansen, V. Ernstsen, B. Lyngby Sørensen, and K. H. Esbensen. '**Iodine Concentrations in Danish Groundwater: Historical Data Assessment 1933-2011**'. Environmental Geochemistry and Health, 36 (6) (2014), pp. 1151-1164. *https://doi.org/10.1007/s10653-014-9625-4*.

Wagner and Plewa 2017. E. D. Wagner and M. J. Plewa. '**CHO Cell Cytotoxicity and Genotoxicity Analyses of Disinfection By-Products: An Updated Review**'. Journal of Environmental Sciences, water treatment and disinfection by-products, 58 (2017), pp. 64-76. *https://doi.org/10.1016/j.jes.2017.04.021*.

Wang et al. 2017. L. Wang, D. Kong, Y. Ji, J. Lu, X. Yin, and Q. Zhou. '**Transformation of Iodide and Formation of Iodinated By-Products in Heat Activated Persulfate Oxidation Process**'. Chemosphere, 181 (2017), pp. 400-408. *https://doi.org/10.1016/j.chemosphere.2017.04.076*.

Wang et al. 2014a. S. Wang, W. Zheng, X. Liu, P. Xue, S. Jiang, D. Lu, Q. Zhang, G. He, J. Pi, M. E. Andersen, H. Tan, and W. Qu. '**Iodoacetic Acid Activates Nrf2-Mediated Antioxidant Response in Vitro and in Vivo**'. Environmental Science & Technology, 48 (22) (2014), pp. 13478-13488. *https://doi.org/10.1021/es502855x*.

Wang et al. 2015. T. Wang, W. Lin, X. Dai, L. Gao, B. Wang, and D. Quan. '**Determination of Iodate by HPLC-UV after On-Line Electrochemical Reduction to Iodide**'. Journal of Chromatographic Science, 53 (2) (2015), pp. 280-284. *https://doi.org/10.1093/chromsci/bmu053*.

Wang et al. 2016. X. Wang, J. Wang, Y. Zhang, Q. Shi, H. Zhang, Y. Zhang, and M. Yang. '**Characterization of Unknown Iodinated Disinfection Byproducts during Chlorination/Chloramination Using Ultrahigh Resolution Mass Spectrometry**'. Science of The Total Environment, 554–555 (2016), pp. 83-88. *https://doi.org/10.1016/j.scitotenv.2016.02.157*.

Wang et al. 2018. X. Wang, Y. Liu, Z. Huang, L. Wang, Y. Wang, Y. Li, J. Li, J. Qi, and J. Ma. '**Rapid Oxidation of Iodide and Hypoiodous Acid with Ferrate and No Formation of Iodoform and Monoiodoacetic Acid in the Ferrate/I$^-$/HA System**'. Water Research, 144 (2018), pp. 592-602. *https://doi.org/10.1016/j.watres.2018.07.061*.

Wang et al. 2019. X. Wang, Z. Wang, Y. Tang, D. Xiao, D. Zhang, Y. Huang, Y. Guo, and J. Liu. '**Oxidative Degradation of Iodinated X-Ray Contrast Media (Iomeprol and Iohexol) with Sulfate Radical: An Experimental and Theoretical Study**'. Chemical Engineering Journal, 368 (2019), pp. 999-1012. *https://doi.org/10.1016/j.cej.2019.02.194*.

Wang et al. 2014b. Z. Wang, B. Xu, Y-L. Lin, C-Y. Hu, F-X. Tian, T-Y. Zhang, and N-Y. Gao. '**A Comparison of Iodinated Trihalomethane Formation from Iodide and Iopamidol in the Presence of Organic Precursors during Monochloramination**'. Chemical Engineering Journal, 257 (2014), pp. 292-298. *https://doi.org/10.1016/j.cej.2014.07.084*.

Warner et al. 2000. J. A. Warner, W. H. Casey, and R. A. Dahlgren. '**Interaction Kinetics of I_2(aq) with Substituted Phenols and Humic Substances**'. Environmental Science & Technology, 34 (15) (2000), pp. 3180-3185. *https://doi.org/10.1021/es991228t*.

Watson et al. 2012. K. Watson, M. J. Farré, and N. Knight. '**Strategies for the Removal of Halides from Drinking Water Sources, and Their Applicability in Disinfection by-Product Minimisation: A Critical Review**'. Journal of Environmental Management, 110 (2012), pp. 276-298. *https://doi.org/10.1016/j.jenvman.2012.05.023*.

Wei et al. 2013a. X. Wei, X. Chen, X. Wang, W. Zheng, D. Zhang, D. Tian, S. Jiang, C. Nam Ong, G. He, and W. Qu. '**Occurrence of Regulated and Emerging Iodinated DBPs in the Shanghai Drinking Water**'. Edited by Zhi Zhou. PLoS ONE 8 (3): e59677 (2013). *https://doi.org/10.1371/journal.pone.0059677*.

Wei et al. 2013b. X. Wei, S. Wang, W. Zheng, X. Wang, X. Liu, S. Jiang, J. Pi, Y. Zheng, G. He, and W. Qu. '**Drinking Water Disinfection Byproduct Iodoacetic Acid Induces Tumorigenic Transformation of NIH3T3 Cells**'. Environmental Science & Technology, 47 (11) (2013), pp. 5913-5920. *https://doi.org/10.1021/es304786b*.

Wei et al. 2013c. Y. Wei, Y. Liu, L. Ma, H. Wang, J. Fan, X. Liu, and R-H. Dai. '**Speciation and Formation of Iodinated Trihalomethane from Microbially Derived Organic Matter during the Biological Treatment of Micro-Polluted Source Water**'. Chemosphere, 92 (11) (2013), pp. 1529-1535. *https://doi.org/10.1016/j.chemosphere.2013.04.019*.

Weinberg 2009. H. S. Weinberg. '**Modern Approaches to the Analysis of Disinfection By-Products in Drinking Water**'. Philosophical Transactions of the Royal Society A, 367 (1904) (2009), pp. 4097-4118. *https://doi.org/10.1098/rsta.2009.0130*.

Weinberg et al. 2002. H. S. Weinberg, S. W. Krasner, S. D. Richardson, and A. D. Thruston. '**The Occurrence of Disinfection By-Products (DBPs) of Health Concern in Drinking Water: Results of a Nationwide DBP Occurrence Study**'. U.S. Environmental Protection Agency, National Exposure Research Laboratory: Athens, GA. EPA/600/R-02/068, (2002).

Weinberg and Yamada 1998. H. S. Weinberg and Harumi Yamada. '**Post-Ion-Chromatography Derivatization for the Determination of Oxyhalides at Sub-PPB Levels in Drinking Water**'. Analytical Chemistry, 70 (1) (1998), pp. 1-6. *https://doi.org/10.1021/ac970651m*.

Wendel et al. 2014. F. M. Wendel, C. Lütke Eversloh, E. J. Machek, S. E. Duirk, M. J. Plewa, S. D. Richardson, and T. A. Ternes. '**Transformation of Iopamidol during Chlorination**'. Environmental Science & Technology, 48 (21) (2014), pp. 12689-12697. *https://doi.org/10.1021/es503609s*.

Westerhoff et al. 2004. P. Westerhoff, P. Chao, and H. Mash. '**Reactivity of Natural Organic Matter with Aqueous Chlorine and Bromine**'. Water Research, 38 (6) (2004), pp. 1502-1513. *https://doi.org/10.1016/j.watres.2003.12.014*.

Wever et al. 2018. R. Wever, B. E. Krenn, and R. Renirie. '**Marine Vanadium-Dependent Haloperoxidases, Their Isolation, Characterization, and Application**'. Methods in Enzymology, 605 (2018), pp. 141-201. *https://doi.org/10.1016/bs.mie.2018.02.026*.

Woo et al. 2018. B. Woo, J-H. Park, S. Kim, J. Lee, J-H. Choi, and H. Pyo. '**Determination of Six Iodotrihalomethanes in Drinking Water in Korea**'. Science of The Total Environment, 640–641 (2018), pp. 581-590. *https://doi.org/10.1016/j.scitotenv.2018.05.329*.

Wuilloud et al. 2003. R. G. Wuilloud, J. C. A. de Wuilloud, A. P. Vonderheide, and J. A. Caruso. '**Determination of Iodinated Phenol Species at Parts-per-Trillion Concentration Levels in Different Water Samples by Solid-Phase Microextraction/Offline GC-ICP-MS**'. Journal of Analytical Atomic Spectrometry, 18 (9) (2003), pp. 1119-1124. *https://doi.org/10.1039/B300911B*.

Xia et al. 2017. Y. Xia, Y-L. Lin, B. Xu, C-Y. Hu, Z-C. Gao, W-H. Chu, and N-Y. Gao. '**Iodinated Trihalomethane Formation during Chloramination of Iodate-Containing Waters in the Presence of Zero Valent Iron**'. Water Research, 124 (2017), pp. 219-226. *https://doi.org/10.1016/j.watres.2017.07.059*.

Xiao et al. 2007. W. Xiao, S. Hong, Z. Tang, S. Seal, and J. S. Taylor. '**Effects of Blending on Surface Characteristics of Copper Corrosion Products in Drinking Water Distribution Systems**'. Corrosion Science, 49 (2) (2007), pp. 449-468. *https://doi.org/10.1016/j.corsci.2006.04.018*.

Xie 2016. Y. F. Xie. **Disinfection Byproducts in Drinking Water: Formation, Analysis, and Control**. CRC Press, (2016).

Xu et al. 2016. C. Xu, S. Zhang, Y. Sugiyama, N. Ohte, Y-F. Ho, N. Fujitake, D. I. Kaplan, C. M. Yeager, K. Schwehr, and P. H. Santschi. '**Role of Natural Organic Matter on Iodine and ^{239,240}Pu Distribution and Mobility in Environmental Samples from the Northwestern Fukushima Prefecture, Japan**'. Journal of Environmental Radioactivity, 153 (2016), pp. 156-166. *https://doi.org/10.1016/j.jenvrad.2015.12.022*.

Xu 2014. L. Xu. '**Kinetics and Mechanism of the Reactions of Pentathionate Ion by Different Halogen-Containing Oxidizing Agents**'. Ph.D. dissertation, Pécs: University of Pécs Doctoral School of Chemistry (2014).

Xu et al. 2017. Z. Xu, X. Li, X. Hu, and D. Yin. '**Distribution and Relevance of Iodinated X-Ray Contrast Media and Iodinated Trihalomethanes in an Aquatic Environment**'. Chemosphere, 184 (2017), pp. 253-260. *https://doi.org/10.1016/j.chemosphere.2017.05.048*.

Xue et al. 2016. R. Xue, A. Donovan, H. Shi, J. Yang, B. Hua, E. Inniss, and T. Eichholz. '**Rapid Simultaneous Analysis of 17 Haloacetic Acids and Related Halogenated Water Contaminants by High-Performance Ion Chromatography-Tandem Mass Spectrometry**'. Analytical and Bioanalytical Chemistry, 408 (24) (2016), pp. 6613-6622. *https://doi.org/10.1007/s00216-016-9773-8*.

Yamaguchi 2011. M. Yamaguchi. '**Hemibonding of Hydroxyl Radical and Halide Anion in Aqueous Solution**'. The Journal of Physical Chemistry A, 115 (51) (2011), pp. 14620-14628. *https://doi.org/10.1021/jp2063386*.

Yamaguchi et al. 2006. N. Yamaguchi, M. Nakano, H. Tanida, H. Fujiwara, and N. Kihou. '**Redox Reaction of Iodine in Paddy Soil Investigated by Field Observation and the I K-Edge XANES Fingerprinting Method**'. Journal of Environmental Radioactivity, 86 (2) (2006), pp. 212-226. *https://doi.org/10.1016/j.jenvrad.2005.09.001*.

Yamaguchi et al. 2010. N. Yamaguchi, M. Nakano, R. Takamatsu, and H. Tanida. '**Inorganic Iodine Incorporation into Soil Organic Matter: Evidence from Iodine K-Edge X-Ray Absorption near-Edge Structure**'. Journal of Environmental Radioactivity, 101 (6) (2010), pp. 451-457. *https://doi.org/10.1016/j.jenvrad.2008.06.003*.

Yang and Zhang 2013. M. Yang and X. Zhang. '**Comparative Developmental Toxicity of New Aromatic Halogenated DBPs in a Chlorinated Saline Sewage Effluent to the Marine Polychaete Platynereis Dumerilii**'. Environmental Science & Technology, 47 (19) (2013), pp. 10868-10876. *https://doi.org/ 10.1021/es401841t*.

Yang et al. 2013. X. Yang, W. Guo, and W. Lee. '**Formation of Disinfection Byproducts upon Chlorine Dioxide Preoxidation Followed by Chlorination or Chloramination of Natural Organic Matter**'. Chemosphere, 91 (11) (2013), pp. 1477-1485. *https://doi.org/10.1016/j.chemosphere.2012.12.014*.

Ye et al. 2012. T. Ye, B. Xu, Y-L. Lin, C-Y. Hu, S-J. Xia, L. Lin, S. Andrew Mwakagenda, and N-Y. Gao. '**Formation of Iodinated Disinfection By-Products during Oxidation of Iodide-Containing Water with Potassium Permanganate**'. Journal of Hazardous Materials, 241–242 (2012), pp. 348-354. *https://doi.org/10.1016/j.jhazmat.2012.09.046*.

Yeo and Choi 2009. J. Yeo and W. Choi. '**Iodide-Mediated Photooxidation of Arsenite under 254 nm Irradiation**'. Environmental Science & Technology, 43 (10) (2009), pp. 3784-3788. *https://doi.org/10.1021/es900602n*.

Yiin and Margerum 1990. B. S. Yiin and D. W. Margerum. '**Nonmetal Redox Kinetics: Reactions of Iodine and Triiodide with Sulfite and Hydrogen Sulfite and the Hydrolysis of Iodosulfate**'. Inorganic Chemistry, 29 (8) (1990), pp. 1559-1564. *https://doi.org/10.1021/ic00333a023*.

Yu et al. 2015. W. Yu, L. Campos, T. Shi, G. Li, and N. Graham. '**Enhanced Removal of Manganese in Organic-Rich Surface Water by Combined Sodium Hypochlorite and Potassium Permanganate during Drinking Water Treatment**'. RSC Advances, 5 (35) (2015), pp. 27970-27977. *https://doi.org/10.1039/C5RA01643F*.

Zeng et al. 2016. T. Zeng, M. J. Plewa, and W. A. Mitch. '**N-Nitrosamines and Halogenated Disinfection Byproducts in U.S. Full Advanced Treatment Trains for Potable Reuse**'. Water Research, 101 (2016), pp. 176-186. *https://doi.org/10.1016/j.watres.2016.03.062*.

Zhai et al. 2014. H. Zhai, X. Zhang, X. Zhu, J. Liu, and M. Ji. '**Formation of Brominated Disinfection Byproducts during Chloramination of Drinking Water: New Polar Species and Overall Kinetics**'. Environmental Science & Technology, 48 (5) (2014), pp. 2579-2588. *https://doi.org/10.1021/es4034765*.

Zhang et al. 2013. E. Zhang, Y. Wang, Y. Qian, T. Ma, D. Zhang, H. Zhan, Z. Zhang, Y. Fei, and S. Wang. '**Iodine in Groundwater of the North China Plain: Spatial Patterns and Hydrogeochemical Processes of Enrichment**'. Journal of Geochemical Exploration, Arsenic, Fluoride and Iodine in Groundwater of China, 135 (2013), pp. 40-53. *https://doi.org/10.1016/j.gexplo.2012.11.016*.

Zhang et al. 2016a. J. Zhang, D-D. Chen, L. Li, W-W. Li, Y. Mu, and H-Q. Yu. '**Role of NOM Molecular Size on Iodo-Trihalomethane Formation during Chlorination and Chloramination**'. Water Research, 102 (2016), pp. 533-541. *https://doi.org/10.1016/j.watres.2016.07.007*.

Zhang et al. 2018. J. Zhang, J. Liu, C-S. He, C. Qian, and Y. Mu. '**Formation of Iodo-Trihalomethanes (I-THMs) during Disinfection with Chlorine or Chloramine: Impact of UV/H_2O_2 Pre-Oxidation**'. Science of The Total Environment, 640–641 (2018), pp. 764-771. *https://doi.org/10.1016/j.scitotenv.2018.05.306*.

Zhang et al. 2016b. M-S. Zhang, B. Xu, Z. Wang, T-Y. Zhang, and N-Y. Gao. '**Formation of Iodinated Trihalomethanes after Ferrate Pre-Oxidation during Chlorination and Chloramination of Iodide-Containing Water**'. Journal of the Taiwan Institute of Chemical Engineers, 60 (2016), pp. 453-459. *https://doi.org/10.1016/j.jtice.2015.11.007*.

Zhang et al. 2011. S. Zhang, J. Du, C. Xu, K. A. Schwehr, Y-F. Ho, H-P. Li, K. A. Roberts, D. I. Kaplan, R. Brinkmeyer, C. M. Yeager, H-S. Chang, and P. H. Santschi. '**Concentration-Dependent Mobility, Retardation, and Speciation of**

Iodine in Surface Sediment from the Savannah River Site'. Environmental Science & Technology, 45 (13) (2011), pp. 5543-5549. *https://doi.org/10.1021/es1040442*.

Zhang et al. 2010a. S. Zhang, K. A. Schwehr, Y-F. Ho, C. Xu, K. A. Roberts, D. I. Kaplan, R. Brinkmeyer, C. M. Yeager, and P. H. Santschi. '**A Novel Approach for the Simultaneous Determination of Iodide, Iodate and Organo-Iodide for ^{127}I and ^{129}I in Environmental Samples Using Gas Chromatography–Mass Spectrometry**'. Environmental Science & Technology, 44 (23) (2010), pp. 9042-9048. *https://doi.org/10.1021/es102047y*.

Zhang et al. 2016c. T-Y. Zhang, Y-L. Lin, A-Q. Wang, F-X. Tian, B. Xu, S-J. Xia, and N-Y. Gao. '**Formation of Iodinated Trihalomethanes during UV/Chloramination with Iodate as the Iodine Source**'. Water Research, 98 (2016), pp. 199-205. *https://doi.org/10.1016/j.watres.2016.04.012*.

Zhang et al. 2015. T-Y. Zhang, B. Xu, C-Y. Hu, Y-L. Lin, L. Lin, T. Ye, and F-X. Tian. '**A Comparison of Iodinated Trihalomethane Formation from Chlorine, Chlorine Dioxide and Potassium Permanganate Oxidation Processes**'. Water Research, 68 (2015), pp. 394-403. *https://doi.org/10.1016/j.watres.2014.09.040*.

Zhang and Minear 2002. X. Zhang and R. A Minear. '**Decomposition of Trihaloacetic Acids and Formation of the Corresponding Trihalomethanes in Drinking Water**'. Water Research, 36 (14) (2002), pp. 3665-3673. *https://doi.org/10.1016/S0043-1354(02)00072-6*.

Zhang et al. 2010b. Y. Zhang, Y. Zhang, and Y-P. Lin. '**Fast Detection of Lead Dioxide (PbO_2) in Chlorinated Drinking Water by a Two-Stage Iodometric Method**'. Environmental Science & Technology, 44 (4) (2010), pp. 1347-1352. *https://doi.org/10.1021/es902299b*.

Zhao et al. 2016. X. Zhao, E. Salhi, H. Liu, J. Ma, and U. von Gunten. '**Kinetic and Mechanistic Aspects of the Reactions of Iodide and Hypoiodous Acid with Permanganate: Oxidation and Disproportionation**'. Environmental Science & Technology, 50 (8) (2016), pp. 4358-4365. *https://doi.org/10.1021/acs.est.6b00320*.

Zhao et al. 2017. X. Zhao, J. Ma, and U. von Gunten. '**Reactions of Hypoiodous Acid with Model Compounds and the Formation of Iodoform in Absence/Presence of Permanganate**'. Water Research, 119 (2017), pp. 126-135. *https://doi.org/10.1016/j.watres.2017.04.033*.

Zhu and Zhang 2016. X. Zhu and X. Zhang. '**Modeling the Formation of TOCl, TOBr and TOI during Chlor(Am)Ination of Drinking Water**'. Water Research, 96 (2016), pp. 166-176. *https://doi.org/ 10.1016/j.watres.2016.03.051*.

10. Supporting information

Table I-S1: Total iodine concentrations in surface waters and groundwaters. Iodide concentrations are shaded in grey.

Country	Number of sites	Sources	Range (µg L^{-1})	Median (µg L^{-1})	References
Afghanistan	37	Water drawn for DW and irrigation	5.4 - 22.7		(Watts and Mitchell 2008)
mid-Wales, U.K.	4	Mineral waters	173 - 595		(Fuge and Johnson 1986)
Anadarko Basin, US	11	Brines	23 - 1400		(Collins 1969)
Argentine	5	Rivers	3.49 - 27.82		(Negri et al. 2013)
	9	Lakes	1.20 - 10.43		
Argentine	21	Rivers	16 - 95	30	(Watts et al. 2010)
	15	GW	52 - 395	110	
Australia	1	Lake (DW source)	50		(Hansson et al. 1987)
Australia	21	DW source waters	< 5 - 594	25.5	(Gruchlik et al. 2015)
Bahrain	15	Natural waters	(0.01 - 0.65) x 10^3	0.02 x 10^3	(Ali-Mohamed and Jamali 1989)
	9	Processed waters	(0.01 - 1.08) x 10^3	0.016 x 10^3	
British Columbia and Alberta, Canada	5	Streams and rivers	0.47 - 2.48		(Fuge 1989)
Canada	65	Source waters for DW treatment plants	< 0.018 - 131.36	0.222 (Summer) 0.075 (Winter)	(Tugulea et al. 2018)
			< 0.10 - 132	2.71 (Summer) 2.20 (Winter)	
Central Russia	24	River	0.40 - 11.40	3.18 (mean)	(Korobova 2010)
	8	Lake	0.70 - 33.00	5.88 (mean)	
	36	GW	0.30 - 21.00	3.72 (mean)	
	6	DW (non-artesian wells)	0.85 - 14.00	6.85 (mean)	
	10	DW (artesian wells)	2.10 - 23.00	9.45 (mean)	
China	34	Tap waters	0.6 - 9.9		(Lu et al. 2005)
China	19	Well or spring DW	0.6 - 84.8		
China	22	SW	1.4 - 39.9		
China	25	Rain and surface runoff	0.51 - 8.33	2.92	(Qin et al. 2014)
China	33	SW	0.08 - 4.13	0.53	(Qin et al. 2014)
China (sampling in 129 counties that were suspected to have high levels)	28857	Water supply sources and wells	(0.1 - 1.5) x 10^3		(Shen et al. 2011)
China	8	GW (for drinking, at 500 - 700 m depths)	187 - 805	287	(Andersen et al. 2009)
China	4390	shallow GW (< 100 - 150 m)	<1 - 1901	20	(Zhang et al. 2013)
	1708	Deep GW (> 100 - 150 m)	<1 - 1381	80	
China	155	Shallow GW (< 50 m)	2.7 - 4117	912	(Tang et al. 2013)
	734	medium-depth GW (50 - 200 m)	0.02 - 2782	331	
	61	deep wells (> 200 m)	6.4 - 2433	534	
China	4	tap waters	0.1 - 0.4		(Gong and Zhang

			4.7 - 10.6 (iodate)		2013)
			6.5 - 12.9		
Danemark	55	Tap waters	< 1.0 - 139	7.5	(Pedersen et al. 1999)
Danemark	41	Tap waters	2.1 - 30.2		(Rasmussen et al. 2000)
Danemark	22	Tap waters	0.7 - 140	9.4	(Andersen et al. 2002)
Danemark	2562	GW	<0.4 - 1220	5.4	(Voutchkova et al. 2014b)
England-Wales	3850	GW	> 3 at 1077 sites (including 38 DW sources)	1.0 - 15.1 (8 regions)	(Goslan 2016)
Europe	807	Stream waters	<0.01 - 104	0.33	(Salminen et al. 2005)
Europe	3	Rivers	0.9 - 4.2		(Moran et al. 2002)
Germany	1	Lake	1.29 (average)		(Gilfedder et al. 2009)
Germany, Austria, Swizerland	Many	rivers, lake	0.6 - 68 in most cases < 1 - 4		(Gilfedder et al. 2010)
			0.01 - 1.2 (iodate)		
			0 - 1.3		
		Precipitation	0.6 - 3.3 (iodine)		
			0.04 - 0.36 (iodate)		
			0.08 - 1.1		
India	287	From municipal supplies or springs and rivers that are used as DW sources	3.0 - 31.5 5.0 - 10.0 for 82% of the samples		(Longvah and Deosthale 1998)
India (sub-Himalayan zone)	108	Drinking water	0.76 - 3.01		(Sharma et al. 1999)
India (sub-Himalayan region)	14	Shallow tube wells (for DW)	3.42 - 13.25		(Bhattacharjee et al. 2013)
India	3	DW	1.8 - 2.6		(Chandra et al. 2006)
India	86	Tap water	20 - 150	93	(Basu et al. 2007)
India	35	SW and GW	7.21 - 43.22	11.7	(Kamavisdar and Patel 2002)
Italy (Covering nearly all the country)	157	Tap waters	0.34 - 27.1	2.9	(Dinelli et al. 2012)
Japan	22	Brines	(6 - 140) x 10^3		(Muramatsu et al. 2001)
Japan	42	Rivers and lakes	0.65 - 35.9	3.66-3.88	(Takaku et al. 1995)
Japan	4	Geothermal and hot springs	0.688 - 1.03 x 10^3		(Koh et al. 1988)
Morrocco, Agadir region, Ounein Valley	19	DW and water irrigation samples	0.5 - 35.2		(Johnson et al. 2002)
Morrocco (goitrous endemic areas (mountains)	3	DW	0.92 - 1.30	1.08	(Aquaron et al. 1993)
Nepal	17	Soil wells	< 1 for 13 sources		(Day and Powell-Jackson 1972)
New Zealand	35	public water supply centres	0.7 - 14.8		(Dean 1963)
Nigeria	23	SW and GW	Nd - 7.5	1.5 in non-goiter area (13/23 sites)	(Ubom and Tsuchiya 1988)
Scotland	7	Raw waters before DWT	1.62 - 6.84	2.87 Winter	(Parsons et al. 2009)
			1.53 - 8.13	2.61 Spring	

			3.61 - 12.1	6.8 Summer	
Spain	1	River (for DW)		3.2	(Cancho et al. 2000)
Sri Lanka	609	DW		13.5	(Balasuriya et al. 1992)
Sri Lanka	15	Shallow DW wells	3.3 - 84		(Fordyce et al. 2000)
Togo,	10	DW	1.0 - 9.2	2.8	(Jaffiol et al. 1992)
Togo (endemic goiter area)	2	DW		2 (average)	(Bilabina et al. 1994)
UK	97	Streams and rivers	0.4 - 15.6		(Fuge 1989)
	30	Springs and wells	1.18 - 14.0		
	12	Lakes	1.47 - 12.60		
USA	NR	Fresh water	Nd - 18	4 (mean)	(Safe Drinking Water Committee 1980)
USA	10	Stream and rivers	3.17 - 13.3		(Fuge 1989)
USA	39	Rivers	0.5 - 212.0	10.2	(Moran et al. 2002)
USA	23	Raw waters for DW	< 0.4 - 104.2	1.9	(Richardson et al. 2008)
USA	3	Lake	2.23 - 6.89		(Dorman and Steinberg 2010)
Wales	10	Streams	1.00 - 4.22	1.8	(Neal et al. 2007)
	6	GW	0.57 - 8.51	2.2	
		Rainfall	1.55 (average)		
World	72	Lakes and rivers	0.125 - 137		(Snyder and Fehn 2004)
France, Switzerland, Austria and Italy	16	Mineral waters	< 3 - 200	0.75	(Bichsel and von Gunten 1999)
			< 3 - 99 (iodate)	1.4	

DW: drinking water; GW: groundwater; SW: surface water

Table I-S2: ICM occurrence in fresh waters.

Country	Source	Iodixanol (ng L^{-1})	Iopamidol (ng L^{-1})	Iohexol (ng L^{-1})	Iomeprol (ng L^{-1})	Iopromide (ng L^{-1})	Diatrizoate (ng L^{-1})	Iot(h)alamic acid ((ng L^{-1})	Ioxit(h)alamic acid (ng L^{-1})	References
Germany	GW		<10-2400			<10-210	<10-170	<10-49	≤0.010	(Ternes and Hirsch 2000)
	Rivers and creeks		<10-2800		<10-89	<10-910	<10-100,000	<10-190	<10-80	
Germany	SW					1.6	2			(Putschew 2000)
	SW after bank filt.					<0.05	4			
	DW inf					<0.05	1.2			
Germany	channel			2000		8500	8000			(Putschew et al. 2001)
	Lake			500		2000-4000	1500-3000			
Germany	GW		<4.5-300				<3.6-1100			(Sacher et al. 2001)
France	Seine River		386		411	17	118	<LOQ	438	(Bruchet et al. 2005)
	borehole		208		85	<loq	100	<LOQ	92	
	DW eff		60		11	<loq	32	<LOQ	12	
Australia	GW					168				(Trenholm et al. 2006)
US	Ohio and Colorado rivers					2.2 and <1				
	DWs					<1 and 4.6				
Germany	Danube River (DW inf)		180-297	106-253	307-399	232-287	80-208			(Seitz et al. 2006a)
	DW eff		72-98	38-40	81-92	69-77	129-149			
Germany	Danube River		<40-520	<40-360	<40-480	<40-220	<40-580			(Seitz et al. 2006b)
Korea	SWs					20-361				(Kim et al. 2007)
	DW eff					<1				
Switzerland	GW wells		88				50			(Hollender et al. 2008)
Germany	SW		180	96	280					(Kormos et al. 2009)
	GW		470	<4	10					
	DW eff		244	<2	<1					
Germany	Rhine River			53	150					(Kormos et al. 2010)
	DW eff			5.5	31					
Germany	SW for DW		39-1230	<3-57	<3-1450	<3-120				(Kormos, 2011)
	DW eff		7-270	<3-5	<3-34	<3-21				
US	DWs (inf)		<10-2700	<10-120	<10	<10-25	< 10-93			(Duirk et al. 2011)
Germany	GW		<DL-79	<DL-187	<DL-1655	<DL - 39	<DL-4240	<DL-238	<DL-204	(Wolf et al. 2012)
Spain	Llobregat River (DW inf)			<30-341		<17-967	<17-84			(Boleda et al. 2013)
	DW			ND		<17-84	ND			
Israel	GW		<10-36,000	<10-180	<10-790	<10-250	<10-940	≤10	<10	(Zemann et al. 2014)
	SW		<10-78,000	< 10-1600	< 10-6900	<10-4500	<10 -860	< 10-23	< 10-51	
Switzerland	Rhine River		47-286	15-56	66-385	69-240	20-47	<LOQ	24-57	(Ens et al. 2014)
	GW		36-94	<LOQ	<LOQ	<LOQ	24-32	<LOQ	<LOQ	
	DW eff		17-43	<LOQ	<LOQ	<LOQ	16-22	<LOQ	<LOQ	
Serbia	SW					<4.7 –				(Petrović et al.

						75.2				2014)
	DW					6.8				
Germany	GW						960-1020			(Redeker et al. 2014)
Japan	River (DW inf)		<2-3300			<10-57				(Simazaki et al. 2015)
	DW eff		<2-2400							
Spain	LLobregat River	<2.5-129	8.9-65.7	26.1-165	23.6-1595	13.8-1785	4.8-27.2			(Zonja et al. 2015)
	Besos River	12.7-4050	18.2-394	221-1326	379-6100	30.6-836	57.5-200			
China	Tap		12	24	<LOD	5.7	3.7			(Li et al. 2015)
	DW inf		16	31	<LOD	8.5	6.4			
	DW eff		5.3	19.2	<LOD	4.4	6.3			
	SW		580	410	3.4	150	60			
Spain	Tap	ND	ND	0.5-5.0	<0.07-1.4	<0.05-1.0	<0.02-1.1			(Mendoza et al. 2016)
US	Chicago River		560-630			ND-160	Nd-55			(Fabbri et al. 2016)
Belgium	SW (DW inf)		<10-350			21-4200	<30-230			(Vlaamse Milieumaatschappij 2017, 2014–16)
China	DW eff			96.5-147	1.3-2.3	1.3-7.8	0.1-1.6			(Z. Xu et al. 2017)
Netherlands	Rhine River						80-620			(Sjerps et al. 2017)
	Meuse River						20-380			

DL: detection limit; DW: drinking water; eff: effluent; filt: filtered; GW: groundwater; inf: influent,; LOD: limit of detection; LOQ: limit of quantification; ND: not detected.

Table I-S3: Comparison of sensitivity of chromatographic methods for iodide and/or iodate determination with electrochemical, UV or mass spectrometry detection.

Sample	**Analytical Column and dimensions**	**Detection**	**DL (µg L^{-1})**	**References**
ION CHROMATOGRAPHY				
Not specified	Zeo-Karb 225 (800 x 3.2 mm)	Potentiometric (Ag)	40 **(iodide)**	(Franks and Pullen 1974)
Not specified	HPIC-AS1	DCA (Ag electrode)	10 **(iodide)**	(Rocklin and Johnson 1983)
	Wescan anion	Potentiometry (Ag-AgCl electrode or epoxy resin matrix electrode)	254 **(iodide)** (Ag-AgCl electrode) 6350 (other electrode)	(Suzuki et al. 1983)
Fresh water, sea water	Brownlee AX-MP (100 x 4.6 mm)	Potentiometry (iodide-selective electrode)	1.9 **(iodide)** can decrease by an order of magnitude in fresh water (pre-concentration).	(Butler and Gershey 1984)
Human serum	Spherisorb ODS (300 x 3.9 mm)	AMP (Ag)	4 **(iodide)**	(Hurst et al. 1984)
Serum and urine	Vydac 302-IC (250 x 4.6 mm)	UV (600 nm) after PCR with chloramine-T and 4,4'-bis(dimethylamino)diphenyl methane	1 **(iodide)**	(Buchberger and Winsauer 1985)
Not specified		potentiometric detection (copper wire electrode and Ag/AgCl reference Electrode)	30 **(iodide)** 500 **(iodate)**	(Haddad et al. 1985)
Cadmium sulfide, potassium fluoride, and iodized sodium chloride	AS2	DCA (Pt electrode)	10 **(iodide)**	(Han et al. 1987)
Human serum, food, water	Vydac 302 IC (250 x 4.6 mm)	Spectrophotometry After PCR with chloramine-T and 4,4'-bis(dimethylamino)diphenyl methane	0.004 **(iodide)** (5 mL injection for water) 1 **(iodide)** (other samples, 20 µL sample injection)	(Buchberger 1988)
Aqueous samples	Shodex I-524A	UV (226 nm)	10 **(iodide)**	(Bruins and Maurer 1989)
Concentrated salt solutions	TSKgel IC-Anion-PW (50 x 4.6 mm)	UV (226 nm)	5 **(iodide)**	(Ito and Sunahara 1990)
		AMP (GC)	5 **(iodide)**	
water	Vydac 302 IC (250 x 4.6 mm)	DCA (Ag electrode)	10 **(iodide)**	(Mehra and Frankenberger 1990)
Sea water	RP-C18 coated with cetyltrimethylammonium (150 x 4.6 mm)	DCA (GC electrode)	5 **(iodide)**	(Ito et al. 1991)
		UV	5 **(iodide)**, 10 **(iodate)**	
Sea water	PRP-X100 (150 x 4.1 mm)	UV (226 nm)	1.3 **(iodide)**	(McTaggart et al. 1994)
Lake, industrial and environmental waste waters	Spherisorb SAX (200 x 4.0 mm)	Electrochemical (GC electrode)	1 **(iodide)**	(Liu et al. 1995)
saline waters	IonPac ASII	UV (605 nm) after PCR with 4,4'-bis(dimethylamino)diphenyl methane in the presence of N-chlorosuccinimide	0.8 **(iodide)**	(Brandão et al. 1995)
Iodide standard solutions	IC-PAK A (50 mm x 4.6 mm) Brownlee PR-18 (100 x 4.6 mm)	DCA (Pt electrode)	127 **(iodide)**	(Chen and Hibbert 1997)
Mineral water		AMP (GC electrode)	10 **(iodide)**	(Yashin and Belyamova

				1998)
Urine and serum	Nova-Pak C18 (RP) coated with N-cetylpyridinium chloride (150 x 3.9 mm)	Potentiometry (Laboratory-made iodide ion-selective electrode)	1.47 **(iodide)**	(Almeida et al. 1997)
Sea water	TSK gel SAX (150 x 4.6 mm)	UV (226 nm)	0.2 **(iodide)**	(Ito 1997)
GW and soil	HPIC AS-11 *(25 mm)*	Conductivity (Ag electrode)	300 **(iodide)**	(Tucker and Flack 1998)
Sea water	AS4A-SC	Conductivity	100 **(iodide)**	(Chandramouleeswaran et al. 1998)
		UV (227 nm)	1 **(iodide)**	
Milk, other dairy products, and table salt	IC-A1	SBAWS conductivity	5 **(iodide)**	(Yang et al. 1998)
Pharmaceutical compounds	Carbon BI-01 (100 x 4.6 mm)	Conductivity	10 **(iodide)**	(Okamoto et al. 1998)
Soil and water	PRP-X100 (150 x 4.1 mm)	UV (230 nm)	40 **(iodide)**	(Papadoyannis et al. 1998)
DW	AS12 (250 x 4 mm)	UV (267 nm) after PCR with bromide under acidic conditions	0.05 **(iodate)**	(Weinberg and Yamada 1998)
Brine, table salt	AS11 (250 x 4 mm)	PAD (Ag electrode)	in the low µg L^{-1} range **(iodide)**	(Dionex (now part of Thermo Scientific) 1998)
Milk products			1 **(iodide)**	('Application Note 37, Determination of Iodide in Milk Products' 2000)
Sea water	TSKGel SAX (35 x 1 mm)	UV (226 nm)	0.2 **(iodide)**	(Ito 1999)
Mineral and DW	AS 11 (250 x 4 mm) for iodide AS 9 (250 x4 mm) for iodate	UV (249 nm) after PCR to IBr2- for iodide and UV (288 nm)after PCR to I3- for iodate	0.1 **(iodide, iodate)**	(Bichsel and von Gunten 1999)
DW	IC-Pak Anion HR (75 x 4.6 mm)	(API) MSD	30 **(iodate)** 60 **(iodide)**	(Buchberger and Ahrer 1999)
	Suppressor column + IC-Pak Anion HR (130 x 2 mm)	(API) MSD	0.5 **(iodate)**	
Urine	Luna RP 18 (C2) (150 × 4.6 mm)	DCA (Au electrode)	1 **(iodide)**	(Below and Kahlert 2001)
Electrolyte solution consisting of different concentrations of $CaCl_2$ and iodide and iodate	AS17 (50 x 4 mm)	PAD (Ag electrode)	0.6 **(iodide)**	(Hu et al. 2005)
Human urine	AS11 (250 × 4 mm)	DCA (Pt electrode)	0.5 **(iodide)**	(Cataldi et al. 2005)
		DCA (Ag electrode)	3.5 **(iodide)**	
		PAD (Ag electrode)	35 **(iodide)**	
Soil and sea water samples	AS16 (250 × 2 mm)	PAD (Ag electrode)	0.5 (disposable electrode); 2 (conventional) **(iodide)**	(Liang et al. 2005)
Natural waters	RP columns (graphite carbon and ODS), coated with fluorine-containing surfactant	conductivity	3300 **(iodide)**	(Helaleh et al. 2005)
GW	AS17 (50 x 4 mm)	conductivity ED50A electrochemical detector	19.9 **(iodide)** 21.7 **(iodate)**	(Hu and Moran 2005)
Infant formula samples	AS16 (250 ×2 mm)	PAD (Ag electrode)	5 **(iodide)**	(Cheng et al. 2005)
DW	AS16 (250 ×2 mm)	Electrospray ionization-MS	22 **(iodate)**	(Barron and Paull 2006)
DW	AS16 (250 ×2 mm)	Suppressed conductivity	1.4 **(iodate)**	(Barron and Paull 2006)
DW	Ion Pac9-HC (250 × 4 mm)	UV-vis (450 nm) after PCR (with 0.5 g L^{-1} o-dianisidine·2HCl (ODA) + 4.5	0.22 **(iodate)**	(Binghui et al. 2006)

		g L^{-1} KBr + 25% methanol + 5.6% nitric acid)		
hot spring water	TSKgel Super-IC-A/C-0.1 (150 x 6 mm)	conductometric	65 **(iodide)**	(Mori et al. 2006)
Milk	AS11 (250 × 4 mm)	DCA (Ag electrode)	6 **(iodide)**	(Melichercik et al. 2006)
Adsorbable organic iodide in SW	AS9-SC (250 × 4 mm)	PAD (Ag electrode)	0.04 **(iodide)**	(Bruggink et al. 2007)
Salts	Ionospher® A Chromsep anion-exchange LC-Varian®	DCA (Ag electrode)	7 **(iodide)**	(Malongo et al. 2008)
		DCA (SCPE)	0.5 **(iodide)**	
Seawater and Other Saline Matrices	AS20	Conductivity	15 **(iodide)**	(Hurum and Rohrer 2009)
		UV (223 nm)	15 **(iodide)**	
Seawater and Table Salt	Acclaim Mixed-Mode WAX-1 (150 x 2.1 mm)	UV (223 nm)	3.3 **(iodide)** 60 **(iodate)**	(Dionex Incorporation 2009)
NOM isolates	AS19 (250 x 4 mm)	Conductometric (Ag electrode)	8 **(iodide)** 10 **(iodate)**	(Gallard et al. 2009)
GW	AS 20	Conductivity (Ag electrode)	LOQ = 25 **(iodide)**	('Iodide Anion by Ion Chromatography, EPA Method 300' 2012)
Seawater	dilauryldimethylammonium bromide -coated monolithic ODS (50 × 4.6 mm and 100 × 4.6 mm) connected in series	UV (225 nm)	1.6 **(iodide)**	(Ito et al. 2012)
DW		DCA (Ag electrode)	0.003 **(iodide)**	(Voutchkova et al. 2014a)
		UV/VIS detector (287.8 nm)	0.005 **(iodate)**	
SW	first (4-mm AS16) and second dimension (0.4-mm AS20)	DCA (Ag electrode)	LOQ = 0.08 **(iodide)**	(Qin et al. 2014)
raw, ultraviolet- and ozone-treated aquacultural seawater	AS20 (250 x 4.0 mm)	(UV 226 nm)	1.2 **(iodide)**	(Rodriguez et al. 2016)
		LIQUID CHROMATOGRAPHY		
sea water, iodized salt, milk, and pharmaceuticals	Shim-pack ODS	UV (22 nm) (derivatization to 4-iodo-2,6-dimethylphenol)	5 **(iodide)**	(Verma et al. 1992)
Enteric coated potassium iodide tablets	C18 RP (250 x 4.6 mm)	Conductimetric	100 **(iodide)**	(Lookabaugh et al. 1987)
		Direct oxidative AMP ED (GC)	50 **(iodide)**	
		Dual-series AMP ED (GC)	40 **(iodide)**	
		UV (223 nm)	10 **(iodide)**	
freshwater and seawater samples	AS11 (250 x 4 mm)	UV (226 nm)	0.05 **iodide, iodate** (after reduction to iodide) in fresh water 0.5 **(total iodine)** (deionized water) after organic decomposition by dehydrohalogenation and reduction to iodide	(Schwehr and Santschi 2003)
bottled, natural and treated waters	Synergi Max-RP C12 (250 x 4.6 mm)	MS-MS	0.1 **(iodate)**	(Snyder et al. 2005)
Natural water	TSK-GEL-NH2-100	PEC-HPLC-UV (215 nm)	45? **(iodide)** (after reduction to	(Wang et al. 2015)

samples			iodide in the PECl	
Natural water samples	C8 DD (50 x 2.0 mm) C18 MGII (250 x 2.0 mm)	Amperometric	0.25 **(iodide)** Total iodine also determined after 30 min of UV irradiation and reduction to iodide.	(Takeda et al. 2016)
		UV–vis (450 nm) after PCR	0.61 **(iodate)**	
Mineral waters	IAM.PC.DD2 Regis HPLC (4.6 x 150 mm, 10 m,	HPLC-diode array detector	22.844 **(iodide)**	(Tatarczak-Michalewska et al. 2019)
GAS CHROMATOGRAPHY				
rainwater, DW, river water, seawater, oil brine, common salt, cow milk, and human blood serum	Neopentylglycol sebacate on Chromosorb G HP	ECD (derivatization to iodoacetone)	0.114 **(iodide)**	(Maros et al. 1989)
Serum and urine	Carbowax (21 m × 0.32 mm)	ECD (derivatization to 2-iodoethanol)	8 **(iodide)**	(Buchberger and Huebauer 1989)
DW	HP-5MS (30 m × 0.25 mm)	MSD (derivatization to 4-iodo-2,6-dimethylphenol)	0.5 **iodide, iodate**, iodine	(Shin et al. 1996)
Sea water	HP-5 (30 m × 0.25 mm)	MSD (derivatization to 4-iodo-N,N-dimethylaniline)	0.008 **(iodide)**	(Mishra et al. 2000)
GW and seawater	TR-5MS (30 m × 0.25 mm)	MSD (derivatization of iodine species to 4-iodo-N,N-dimethylaniline)	0.043 **(iodide)** 0.194 **(iodate)**	(Zhang et al. 2010b)

Ag: silver; AgCl: silver chloride; Au: gold; DCA: direct current amperometry; DL: detection limit; DW: drinking water; ECD: electrochemical detection; GC: glassy carbon; GW: groundwater; MSD: mass spectrometry detection; PAD: pulsed amperometric detection; PCR: post-column reaction; RP: reversed phase; SCPE: silver carbon paste electrode; SW: surface water.

Table I-S4: Comparison of sensitivity of chromatographic coupling methods for iodide, iodate determination with ICP-MS detection.

Samples	Method	Column	DL for I^- (ng L^{-1})	DL for IO_3^- (ng L^{-1})	Reference
DW	IC-ICP-MS	ICS-A23 and ICS-A13 in series (7.6 x 4.6 mm)	51	34	(Yamanaka et al. 1997)
DW	IC-ICP-MS	AS14 (4 mm)	2.1	7.2	(Dudoit and Pergantis 2001)
Water samples	IC-ICP-MS	AS16 (250 x 4 mm)	210	130	(Sacher et al. 2005)
GW	IC-ICP-MS	ICS-A23	25	25	(Yang et al. 2007)
mQ water	HPLC-ICP-MS	ICS-A23	25	35	(Wei et al. 2007)
Snow samples	IC-ICP-MS	AS16	30	30	(Gilfedder et al. 2007)
Rain, river water, brine, and soil solution	IC-ICP-MS	EXCELPAK ICS-A23	100-1000		(Yoshida et al. 2007)
Lake	IC-ICP-MS	AS16	30	30	(Gilfedder et al. 2008)
Tap water, urine	IC-ICP-MS	Hamilton PRP-X100 (150 x 4.1 mm)	2	1	(Wang and Jiang 2008)
DW, GW, SW, and swimming pool water	IC-ICP-MS	AS11-HC (250 x 4 mm)	700	330	(Shi and Adams 2009)
Aerosols using pure water	IC-ICP-MS or SEC-ICP-MS	ICS-A23	12	8	(Xu et al. 2010)
Bottled DW	HPLC-ICP-MS	ICS-A23	12	8	(Liu et al. 2011)
Seaweed and seawater	RP-HPLC-ICP-MS	Diamonsil C18 (2) (150 X 4.6 mm)	52	61	(Han et al. 2012)
Ice core samples	HPLC-IC-ICP-MS	AS16	5 ng kg^{-1}	7 ng kg^{-1}	(Spolaor et al. 2013)
Urine	RP-LC-ICP-MS	Zorbax Eclipse Plus C18 (12.5 x 4.6 mm)	46	47	(Han et al. 2014)
GW	IC-ICP-MS	AG19 (50 x 4 mm)	6	5	(Guo et al. 2016)
Snow samples	IC-ICP-MS	AS14 (250 x 4 mm)	23	73	(Gao et al. 2018)
Precipitation samples	LC-ICP-MS	HyperCarb (100 × 4.6 mm)	47	23	(Suess et al. 2019)

DL: detection limit; DW: drinking water; GW: groundwater; SW: surface water.

Table I-S5: ICP-MS methods for determination of total iodine or total organic iodine (TOI).

Iodine analysed	Samples	DL (µg L^{-1})	Reference
Total iodine	River, lake and tap water	0.01	(Takaku et al. 1995)
Total iodine	Solid samples	0.02	(Schnetger and Muramatsu 1996)
Total iodine	Soils	0.012	(Yamada et al. 1996)
Total iodine	Rock	0.2	(Muramatsu and Wedepohl 1998)
Total iodine	Surface, shallow waters	0.1	(Szidat et al. 2000)
Total iodine	River	0.01-0.05	(Tagami and Uchida 2005)
Total iodine	River water	0.01-0.04	(Tagami and Uchida 2006)
Total iodine	Bottled mineral water	0.01	(Birke et al. 2010)
Total iodine	Groundwater	0.5	(Brown et al. 2007)
Total iodine	Sea water	0.035	(Hansen et al. 2011)
Total iodine	Drinking water	0.19	(Goslan 2016)
TOI	Drinking water	0.95	(Sayess and Reckhow 2017)

DL: detection limit; TOI: total organic iodine.

Table I-S6: Cytotoxicity data for I-DBPs and chlorinated and brominated analogues.

Group/ Compound	Model	Exposure time (h)	Lowest Cytotoxic concentration (µM)	%C½ value (approx. LC50) (µM)	Reference
Iodoacids					
Iodoacetic Acid	Salmonella typhimurium strain TA100	210 min	<150	~250	(Cemeli et al. 2006)
Iodoacetic Acid	Salmonella typhimurium strain TA100	210 min	100	303	(Plewa et al. 2004)
Iodoacetic Acid	CHO-AS52 cells	72	0.5	2.95	(Plewa et al. 2004; Richardson et al. 2008)
Iodoacetic Acid	CHO-AS52 cells	72	0.5	4	(Cemeli et al. 2006)
Iodoacetic Acid	CHO-K1 cells	72	1.68	5.23	(Zhang et al. 2010a)
Iodoacetic Acid	binucleated TK6 cells	48	NA	<10 (total cyto)	(Liviac et al. 2010)
Iodoacetic Acid	CHO	3	35	80	(Hilliard et al. 1998) (as sodium iodoacetate)
Iodoacetic Acid	Caco-2 cells	4	NA	34	(Procházka et al. 2015)
Iodoacetic Acid	HepG2 cells, rat hepatocytes, and intact rats	24	2	8	(Wang et al. 2014)
Iodoacetic Acid	HepG2 cells	24	NA	12	(Hu et al. 2018)
Iodoacetic Acid	NIH3T3 cells	72	2.5	2.77	(Wei et al. 2013)
Iodoacetic Acid	Primary human lymphocytes	72	NS	NS	(Escobar-Hoyos et al. 2013)
Iodoacetic Acid	CCD 841 CoN Cell	12	NA	8.6	(Sayess et al. 2017)
Diiodoacetic acid	CHO-AS52 cells	72	100	332	(Richardson et al. 2008)
Diiodoacetic acid	HepG2 cells	24	NA	395	(Hu et al. 2018)
Diiodoacetic acid	CCD 841 CoN Cell	12	NA	954.7	(Sayess et al. 2017)
Bromoiodoacetic acid	CHO-AS52 cells	72	250	897	(Richardson et al. 2008)
Bromoiodoacetic acid	CCD 841 CoN Cell	12	NA	982.2	(Sayess et al. 2017)
Chloroiodoacetic acid	CHO-AS52 cells	72	170	304	(Wagner and Plewa 2017)
(Z)-3-bromo-3-iodopropenoic acid	CHO-AS52 cells	72	75	208	(Richardson et al. 2008)
(E)-3-bromo-3-iodopropenoic acid	CHO-AS52 cells	72	25	145	(Richardson et al. 2008)
(E)-3-bromo-2-iodopropenoic acid	CHO-AS52 cells	72	17.5	436	(Richardson et al. 2008)
(E)-2-iodo-3-methylbutenedioic acid	CHO-AS52 cells	72	700	944	(Richardson et al. 2008)
3,5-diiodosalicylic acid	HepG2 cells	24	NA	400	(Hu et al. 2018)
Bromoacetic acid	CHO-AS52 cells	72	2	10	(Plewa et al. 2010)
Bromoacetic acid	binucleated TK6 cells	48	NA	<20 (total cyto)	(Liviac et al. 2010)
Bromoacetic acid	S. typhimurium TA100 cells	210 min	NA	522	(Kargalioglu et al. 2002)
Bromoacetic acid	Salmonella typhimurium strain TA100	210 min	216	881	(Plewa et al. 2004)
Bromoacetic acid	CHO-AS52 cells	72	2	9.56	(Plewa et al. 2004)
Bromoacetic acid	CHO-AS52 cells	72	NA	8.9	(Plewa et al. 2002)
Bromoacetic acid	Caco-2 cells	4	NA	42	(Procházka et al. 2015)
Bromoacetic acid	CHO-K1 cells	72	4.5	26.9	(Zhang et al. 2010a)

Bromoacetic acid	Primary human lymphocytes	72	NS	NS	Escobar-Hoyes et al., 2013
Chloroacetic acid	CHO-AS52 cells	72	250	810	(Plewa et al. 2010)
Chloroacetic acid	binucleated TK6 cells	48	NA	<750(total cyto)	(Liviac et al. 2010)
Chloroacetic acid	CHO-AS52 cells	72	300	848	(Plewa et al. 2004)
Chloroacetic acid	CHO-AS52 cells	72	NA	944	(Plewa et al. 2002)
Chloroacetic acid	S. typhimurium TA100 cells	210 min	NA	16200	Karga
Chloroacetic acid	Caco-2 cells	4	NA	1200	(Procházka et al. 2015)
Chloroacetic acid	CHO-K1 cells	72	248	732	(Zhang et al. 2010a)
Chloroacetic acid	Primary human lymphocytes	72	NS	NS	(Escobar-Hoyos et al. 2013)
Dibromoacetic acid	CHO-AS52 cells	72	200	590	(Plewa et al. 2010)
Dibromoacetic acid	CHO-AS52 cells	72	NA	500	(Plewa et al. 2002)
Dibromoacetic acid	S. typhimurium TA100 cells	210 min	NA	15400	(Kargalioglu et al. 2002)
Dibromoacetic acid	CHO-K1 cells	72	359	641	(Zhang et al. 2010a)
Dichloroacetic acid	CHO-AS52 cells	72	2000	7300	(Plewa et al. 2010)
Dichloroacetic acid	CHO-AS52 cells	72	NA	11470	(Plewa et al. 2002)
Dichloroacetic acid	S. typhimurium TA100 cells	210 min	NA	74200	(Kargalioglu et al. 2002)
Dichloroacetic acid	CHO-K1 cells	72	3030	4230	(Zhang et al. 2010a)
Trichloroacetic acid	CHO-AS52 cells	72	400	2400	(Plewa et al. 2010)
Trichloroacetic acid	CHO-AS52 cells	72	NA	17520	(Plewa et al. 2002)
Trichloroacetic acid	CHO-K1 cells	72	2870	5440	(Zhang et al. 2010a)
Trichloroacetic acid	S. typhimurium TA100 cells	210 min	NA	42500	(Kargalioglu et al. 2002)
Iodomethanes					
Dibromoiodomethane	CHO-AS52 cells	72	1500	1900	(Richardson et al. 2008)
Dichloroiodomethane	CHO-AS52 cells	72	2000	4130	(Richardson et al. 2008)
Bromochloroiodomethane	CHO-AS52 cells	72	2200	2400	(Richardson et al. 2008)
Bromodiiodomethane	CHO-AS52 cells	72	1500	NA	(Richardson et al. 2008)
Chlorodiiodomethane	CHO-AS52 cells	72	1000	2410	(Richardson et al. 2008)
Iodoform	CHO-AS52 cells	72	10	66	(Richardson et al. 2008)
Iodoform	NIH3T3 cells	72	55	83.37	(Wei et al. 2013)
Iodoform	HepG2 cells	24	NA	1609	(Hu et al. 2018)
Trichloromethane	CHO-AS52 cells	72	NA	9620	(Wagner and Plewa 2017)cite (Plewa and Wagner 2009)
Trichloromethane	S. typhimurium TA100 cells	210 min	NA	28200	(Kargalioglu et al. 2002)
Bromodichloromethane	CHO-AS52 cells	72	NA	1150	(Wagner and Plewa 2017)
Dibromochloromethane	CHO-AS52 cells	72	NA	5350	(Wagner and Plewa 2017)
Tribromomethane	CHO-AS52 cells	72	NA	3960	(Wagner and Plewa 2017)
Tribromomethane	S. typhimurium TA100 cells	210 min	NA	12400	(Kargalioglu et al. 2002)
Iodonitriles					
Iodoacetonitrile	CHO-AS52 cells	72	0.1	3.3	(Muellner et al. 2007)
chloroiodoacetonitrile	NA		NA	NA	NA
bromoacetonitrile	CHO-AS52 cells	72	1	3.21	(Muellner et al. 2007)
chloroacetonitrile	CHO-AS52 cells	72	50	68.3	(Muellner et al. 2007)
Bromochloroacetonitrile	CHO-AS52 cells	72	7	8.46	(Muellner et al. 2007)

Iodoamides					
Iodoacetamide	CHO-AS52 cells	72	0.5	1.42	(Plewa et al. 2007)
Iodoacetamide	CCD 841 CoN cell	12	NA	39.1	(Sayess et al. 2017)
Iodoacetamide	LLC-PK1	6	10	NA	(Chen and Stevens 1991)
Iodoacetamide	HepG2 Cells	24	NA	14.57	(Hong et al. 2018)
Iodoacetamide	HepG2 Cells	48	NA	7.78	
Diiodoacetamide	CHO-AS52 cells	72	0.025	0.678	(Plewa et al. 2007)
Diiodoacetamide	HepG2 Cells	24	NA	5.8	(Hong et al. 2018)
Diiodoacetamide	HepG2 Cells	48	NA	2.48	
Bromoiodoacetamide	CHO-AS52 cells	72	2	3.81	(Plewa et al. 2007)
Bromoiodoacetamide	CCD 841 CoN cell	12	NA	136.3	(Sayess et al. 2017)
Bromoiodoacetamide	HepG2 Cells	24	NA	24.66	(Hong et al. 2018)
Bromoiodoacetamide	HepG2 Cells	48	NA	23.77	
Chloroiodoacetamide	CHO-AS52 cells	72	2	5.97	(Plewa et al. 2007)
Chloroiodoacetamide	CCD 841 CoN cell	12	NA	369	(Sayess et al. 2017)
Chloroiodoacetamide	HepG2 Cells	24	NA	28.48	(Hong et al. 2018)
Chloroiodoacetamide	HepG2 Cells	48	NA	27.29	
5-Amino-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2,4,6-triiodoisophthalamide	CHO-AS52 cells	72	900	1140	(Wendel et al. 2016)
2-(3,5-Bis((1,3-dihydroxypropan-2-yl)carbamoyl)-2,4,6-triiodophenoxy) propanoic acid	CHO-AS52 cells	72	NS	NA	
N1,N3-Bis(1,3-dihydroxypropan-2-yl)-2,4,6-triiodo-5-nitroisophthalamide	CHO-AS52 cells	72	900	934	
4-Chloro-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2,6-diiodo-5-nitroisophthalamide	CHO-AS52 cells	72	1000	1300	
4,6-Dichloro-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2-iodo-5-nitroisophthalamide	CHO-AS52 cells	72	250	823	
Bromoacetamide	CHO-AS52 cells	72	0.5	1.89	(Plewa et al. 2007)
Chloroacetamide	CHO-AS52 cells	72	75	148	(Plewa et al. 2007)
Dibromoacetamide	CHO-AS52 cells	72	2.5	12.2	(Plewa et al. 2007)
Dichloroacetamide	CHO-AS52 cells	72	800	1920	(Plewa et al. 2007)
Iodoaldehydes					
Iodobutanal	NA		NA	NA	NA
Iodoacetaldehyde	CHO-AS52 cells	72	5	6	(Jeong et al. 2015)
3,5-diiodo-4-hydroxybenzaldehyde	HepG2 cells	24	NA	962	(Hu et al. 2018)
Bromoacetaldehyde	CHO-AS52 cells	72	8	17.28	(Jeong et al. 2015)
Chloroacetaldehyde	CHO-AS52 cells	72	0.5	3.51	(Jeong et al. 2015)
iodophenols					
2-iodophenol	CHO-AS52 cells	72	150	601	(Liberatore et al. 2017)
4-iodophenol	HepG2 Cells	24	NA	490	(Gong et al. 2017)
4-iodophenol	CHO-AS52 cells	72	50	216	(Liberatore et al. 2017)
2,6-diiodo-4-nitrophenol	HepG2 Cells	24	NA	126	(Gong et al. 2017)
2,6-diiodo-4-nitrophenol	HepG2 Cells	24	NA	79	(Hu et al. 2018)
4-iodo-2-methylphenol	CHO-AS52 cells	72	25	163	(Liberatore et al. 2017)
4-Hydroxy-3,5-diiodo-1-	CHO-AS52 cells	72	33	33.2	(Wagner and Plewa

phenyl					2017)
2,4,6-triiodophenol	HepG2 Cells	24	NA	204	(Gong et al. 2017)
2,4,6-triiodophenol	HepG2 cells	24	NA	151	(Hu et al. 2018)
2,4,6-triiodophenol	CHO-AS52 cells	72	5	43.7	(Liberatore et al. 2017)
2,4,6-Triiodo-1-phenol	CHO-AS52 cells	72	60	70.1	(Wagner and Plewa 2017)
4-Hydroxy-3-iodo-1-phenolic acid	CHO-AS52 cells	72	150	318	
4-Hydroxy-3,5-diiodo-1-phenolic acid	CHO-AS52 cells	72	110	288	
4-Hydroxy-3-iodophenyl	CHO-AS52 cells	72	100	408	
Cyanogen halides					
Cyanogen bromide	CHO-AS52 cells	72	1	20.9	(Wagner and Plewa 2017)
Cyanogen chloride	CHO-AS52 cells	72	3000	3250	
Cyanogen iodide	CHO-AS52 cells	72	1	9.01	
Iopamidol high molecular weight DBPs					
5-Amino-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2,4,6-triiodoisophthalamide	CHO-AS52 cells	72	900	1440.7	(Wendel et al. 2016)
2-(3,5-Bis((1,3-dihydroxypropan-2-yl)carbamoyl)-2,4,6-triiodophenoxy) propanoic acid	CHO-AS52 cells	72	NS	NA	(Wendel et al. 2016)
N1,N3-Bis(1,3-dihydroxy-propan-2-yl)-2,4,6-triiodo-5-nitroisophthalamide	CHO-AS52 cells	72	900	933.7	(Wendel et al. 2016)
4-Chloro-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2,6-diiodo-5-nitroiso-phthalamide	CHO-AS52 cells	72	1000	1296.2	(Wendel et al. 2016)
4,6-Dichloro-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2-iodo-5-nitroisophthalamide	CHO-AS52 cells	72	250	832.3	(Wendel et al. 2016)
Other					
2,6-diiodo-1,4-benzoquinone	HepG2 cells	24	NA	182	(Hu et al. 2018)

LC50: lethal concentration for 50% of a population; NA: not applicable; NS: not statistically significant.

Table I-S7: Genotoxicity data for I-DBPs and their chlorinated and brominated analogues.

Group/ Compound	Model	Genetic endpoint	Lowest genotoxic concentra--tion (μM)	Concentra--tion that induces the midpoint in the average genotoxicity effect	Reference
Iodoacids					
Iodoacetic Acid	NIH3T3 cells	cell transformation assay	2	NA	(Wei et al. 2013)
Iodoacetic Acid	CHO	Chrom. Ab.	50	70	(Hilliard et al. 1998) (as sodium iodoacetate)
Iodoacetic Acid	HepG2 cells	SCGE	0.01	NA	(Zhang et al. 2012)
Iodoacetic Acid	CHO-AS52 cells	SCGE	5	8.7	(Plewa et al. 2004; Richardson et al. 2008)
Iodoacetic Acid	Primary human lymphocytes	SCGE	45	10.43	(Escobar-Hoyos et al. 2013)
Iodoacetic Acid	Nontransformed Human FHs Cells	SCGE	NA	21.9	(Attene-Ramos et al. 2010)
Diiodoacetic acid	CHO-AS52 cells	SCGE	1000	1980	(Richardson et al. 2008)
Bromoiodoacetic acid	CHO-AS52 cells	SCGE	2500	3160	(Richardson et al. 2008)
(Z)-3-bromo-3-iodopropenoic acid	CHO-AS52 cells	SCGE	NS	NS	(Richardson et al. 2008)
(E)-3-bromo-3-iodopropenoic acid	CHO-AS52 cells	SCGE	5000	6350	(Richardson et al. 2008)
(E)-3-bromo-2-iodopropenoic acid	CHO-AS52 cells	SCGE	7500	7580	(Richardson et al. 2008)
(E)-2-iodo-3-methylbutenedioic acid	CHO-AS52 cells	SCGE	6000	6000	(Richardson et al. 2008)
Bromoacetic acid	Nontransformed Human FHs Cells	SCGE	NA	56.5	(Attene-Ramos et al. 2010)
Bromoacetic acid	HepG2 cells		0.1	NA	(Zhang et al. 2012)
Bromoacetic acid	CHO-AS52 cells	SCGE	13	17	(Plewa et al. 2002; 2010)
Bromoacetic acid	Primary human lymphocytes	SCGE	8	12.07	(Escobar-Hoyos et al. 2013)
Chloroacetic acid	Nontransformed Human FHs Cells	SCGE	NA	3420	(Attene-Ramos et al. 2010)
Chloroacetic acid	HepG2 cells		NS	NA	(Zhang et al. 2012)
Chloroacetic acid	CHO-AS52 cells	SCGE	300	411	(Plewa et al. 2002; 2010)
Chloroacetic acid	Primary human lymphocytes	SCGE	730	805.8	(Escobar-Hoyos et al. 2013)
Dibromoacetic acid	CHO-AS52 cells	SCGE	750	1756	(Plewa et al. 2002; 2010)
Dibromoacetic acid	HepG2 cells	SCGE	1	NA	(Zhang et al. 2012)
Dichloroacetic acid	CHO-AS52 cells	SCGE	NS	NA	(Plewa et al. 2002)
Dichloroacetic acid	HepG2 cells	SCGE	10	NA	(Zhang et al. 2012)
Trichloroacetic acid	CHO-AS52 cells	SCGE	NS	NA	(Plewa et al. 2002)
Trichloroacetic acid	HepG2 cells	SCGE	100	NA	(Zhang et al. 2012)
Iodomethanes					
Dibromoiodomethane	CHO-AS52 cells	SCGE	NS	NS	(Richardson et al. 2008)
Dichloroiodomethane	CHO-AS52 cells	SCGE	NS	NS	(Richardson et al. 2008)

Bromochloroiodomethane	CHO-AS52 cells	SCGE	NS	NS	(Richardson et al. 2008)
Bromodiiodomethane	CHO-AS52 cells	SCGE	NS	NS	(Richardson et al. 2008)
Chlorodiiodomethane	CHO-AS52 cells	SCGE	2000	2950	(Richardson et al. 2008)
Iodoform	CHO-AS52 cells	SCGE	NS	NS	(Richardson et al. 2008)
Iodoform	SHE cells	Chrom. Ab.	negative		(Hikiba et al. 2005)
Iodoform	NIH3T3 cells	cell transformation assay	negative		(Wei et al. 2013)
Chloroiodomethane					
Chloroform	HepG2 cells	SCGE	10000	NA	(Zhang et al. 2012)
Bromodichloromethane	HepG2 cells	SCGE	1	NA	(Zhang et al. 2012)
Dibromochloromethane	HepG2 cells	SCGE	10	NA	(Zhang et al. 2012)
Bromoform	HepG2 cells	SCGE	1000	NA	(Zhang et al. 2012)
Iodonitriles					
Iodoacetonitrile	CHO-AS52 cells	SCGE	30	37.1	(Muellner et al. 2007)
bromoacetonitrile	CHO-AS52 cells	SCGE	40	38.5	(Muellner et al. 2007)
Ihloroacetonitrile	CHO-AS52 cells	SCGE	250	601	(Muellner et al. 2007)
Bromochloroacetonitrile	CHO-AS52 cells	SCGE	250	324	(Muellner et al. 2007)
Iodoamides					
Iodoacetamide	CHO-AS52 cells	SCGE	30	34.1	(Plewa et al. 2007)
Diiodoacetamide	CHO-AS52 cells	SCGE	25	33.9	(Plewa et al. 2007)
Bromoiodoacetamide	CHO-AS52 cells	SCGE	25	72.1	(Plewa et al. 2007)
Chloroiodoacetamide	CHO-AS52 cells	SCGE	200	302	(Plewa et al. 2007)
5-Amino-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2,4,6-triiodoisophthalamide	CHO-AS52 cells	SCGE	NS	NA	(Wendel et al. 2016)
2-(3,5-Bis((1,3-dihydroxypropan-2-yl)carbamoyl)-2,4,6-triiodophenoxy) propanoic acid	CHO-AS52 cells	SCGE	NS	NA	
N1,N3-Bis(1,3-dihydroxy-propan-2-yl)-2,4,6-triiodo-5-nitroisophthalamide	CHO-AS52 cells	SCGE	NS	NA	
4-Chloro-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2,6-diiodo-5-nitroiso-phthalamide	CHO-AS52 cells	SCGE	NS	NA	
4,6-Dichloro-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2-iodo-5-nitroisophthalamide	CHO-AS52 cells	SCGE	NS	NA	
Bromoacetamide	CHO-AS52 cells	SCGE	25	36.8	(Plewa et al. 2007)
Chloroacetamide	CHO-AS52 cells	SCGE	750	1380	(Plewa et al. 2007)
Dibromoacetamide	CHO-AS52 cells	SCGE	500	744	(Plewa et al. 2007)
Dichloroacetamide	CHO-AS52 cells	SCGE	NA	NS	(Plewa et al. 2007)
Iodoaldehydes					
Iodoacetaldehyde	CHO-AS52 cells	SCGE	900	1009	(Jeong et al. 2015)
Bromoacetaldehyde	CHO-AS52 cells	SCGE	200	381.2	(Jeong et al. 2015)
Chloroacetaldehyde	CHO-AS52 cells	SCGE	100	142.8	(Jeong et al. 2015)
Cyanogen halides					
Cyanogen bromide	CHO-AS52 cells	SCGE	500	500	(Wagner and Plewa 2017)
Cyanogen chloride	CHO-AS52 cells	SCGE	NS	NA	

Cyanogen iodide	CHO-AS52 cells	SCGE	200	214	
Iopamidol high molecular weight DBPs					
5-Amino-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2,4,6-triiodoisophthalamide	CHO-AS52 cells	SCGE	NA	NS	(Wendel et al. 2016)
2-(3,5-Bis((1,3-dihydroxypropan-2-yl)carbamoyl)-2,4,6-triiodophenoxy) propanoic acid	CHO-AS52 cells	SCGE	NA	NS	
N1,N3-Bis(1,3-dihydroxy-propan-2-yl)-2,4,6-triiodo-5-nitroisophthalamide	CHO-AS52 cells	SCGE	NA	NS	
4-Chloro-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2,6-diiodo-5-nitroiso-phthalamide	CHO-AS52 cells	SCGE	NA	NS	
4,6-Dichloro-N1,N3-bis(1,3-dihydroxypropan-2-yl)-2-iodo-5-nitroisophthalamide	CHO-AS52 cells	SCGE	NA	NS	

NA: not applicable; NS: not statistically significant.

Table I-S8: Parameters used in the kinetic model discussed in the main text.

Reactions	**Rate constants (mol L^{-1})**	***pK_a***
$BrO^- + I^- \rightarrow IO^- + Br^-$	6.8×10^5	
$ClO^- + Br^- \rightarrow BrO^- + Cl^-$	9×10^{-4}	
$ClO^- + I^- \rightarrow IO^- + Cl^-$	30	
$BrO^- + H^+ = HOBr$		8.8
$HOBr + I^- \rightarrow HOI + Br$	5×10^9	
$HOCl + Br- \rightarrow HOBr + Cl^-$	1550	
$ClO^- + H^+ = HOCl$		7.53
$HOCl + I^- \rightarrow HOI + Cl^-$	4.3×10^8	
$HOI + ClO^- \rightarrow IO_2^-$	52	
$HOI + HOCl \rightarrow IO_2^-$	8.2	
$IO^- + H^+ = HOI$		10.4
$IO^- + BrO^- \rightarrow IO_2^-$	1.8×10^3	
$IO^- + HOBr \rightarrow IO_2^-$	1.9×10^6	
$IO_2^- + BrO^- \rightarrow IO_3^-$	4×10^4	
$IO_2^- + ClO^- \rightarrow IO_3^-$	1000	
$IO_2^- + HOBr \rightarrow IO_3^-$	4×10^7	
$IO_2^- + HOCl \rightarrow IO_3^-$	160	
$IO_2^- + HOI \rightarrow IO_3^-$	300	
$BrO^- + PhO^- \rightarrow 0.2 * Ph\text{-}Br + 0.8 * Br^-$	3.5×10^4	
$HOBr + PhO^- \rightarrow 0.2 * Ph\text{-}Br + 0.8 * Br^-$	6.6×10^7	
$HOCl + PhO^- \rightarrow 0.2 * Ph\text{-}Cl + 0.8 * Cl^-$	2.19×10^4	
$PhO^- + HOI \rightarrow 0.2 * Ph\text{-}I + 0.8 * I^-$	2×10^6	
$PhOH + HOBr \rightarrow 0.2 * Ph\text{-}Br + 0.8 * Br^-$	1	
$HOCl + PhOH \rightarrow 0.2 * Ph\text{-}Cl + 0.8 * Cl^-$	0.36	
$PhOH + HOI \rightarrow 0.2 * Ph\text{-}I + 0.8 * I^-$	100	
$PhO^- + H^+ = PhOH$		10.0

11. Bibliography for supporting information

Ali-Mohamed and Jamali 1989. A. Y. Ali-Mohamed and S. G. A. Jamali. '**Potentiometric Determination of Fluoride and Iodide in Natural and Processed Waters of Bahrain Using Ion-Selective Electrodes**'. Water Research, 23 (5) (1989), pp. 659-662. *https://doi.org/10.1016/0043-1354(89)90033-X*.

Almeida et al. 1997. A. A. Almeida, X. Jun, and J. L. F. C. Lima. '**Ion Chromatographic Determination of Iodide in Urine and Serum Using a Tubular Ion-Selective Electrode Based on a Homogeneous Crystalline Membrane**'. Mikrochimica Acta, 127 (1-2) (1997), pp. 55-60. *https://doi.org/10.1007/BF01243164*.

Andersen et al. 2009. S. Andersen, H. Guan, W. Teng, and P. Laurberg. '**Speciation of Iodine in High Iodine Groundwater in China Associated with Goitre and Hypothyroidism**'. Biological Trace Element Research, 128 (2) (2009), pp. 95-103. *https://doi.org/10.1007/s12011-008-8257-x*.

Andersen et al. 2002. S. Andersen, S. B. Petersen, and P. Laurberg. '**Iodine in Drinking Water in Denmark Is Bound in Humic Substances**'. European Journal of Endocrinology, 147 (5) (2002), pp. 663-670. *https://doi.org/10.1530/eje.0.1470663*.

'**Application Note 37, Determination of Iodide in Milk Products**'. 2000. Dionex Corporation.

Aquaron et al. 1993. R. Aquaron, K. Zarrouck, M. El Jarari, R. Ababou, A. Talibi, and J. P. Ardissone. '**Endemic Goiter in Morocco (Skoura-Toundoute Areas in the High Atlas)**'. Journal of Endocrinological Investigation, 16 (1) (1993), pp. 9-14. *https://doi.org/10.1007/BF03345821*.

Attene-Ramos et al. 2010. M. S. Attene-Ramos, E. D. Wagner, and M. J. Plewa. '**Comparative Human Cell Toxicogenomic Analysis of Monohaloacetic Acid Drinking Water Disinfection Byproducts**'. Environmental Science & Technology, 44 (19) (2010), pp. 7206-7212. *https://doi.org/10.1021/es1000193*.

Balasuriya et al. 1992. S. Balasuriya, P. A. J. Perera, K. B. Herath, S. L. Katugampola, and M. A. Fernando. '**Role of Iodine Content of Drinking Water in the Aetiology of Goitre in Sri Lanka**'. The Ceylon Journal of Medical Science, 35 (2) (1992), pp. 45-51. *http://archive.cmb.ac.lk:8080/research/handle/70130/3485*.

Barron and Paull 2006. L. Barron and B. Paull. '**Simultaneous Determination of Trace Oxyhalides and Haloacetic Acids Using Suppressed Ion Chromatography-Electrospray Mass Spectrometry**'. Talanta, 69 (3) (2006), pp. 621-630. *https://doi.org/10.1016/j.talanta.2005.10.032*.

Basu et al. 2007. S. Basu, B. Mohanty, S. Sarkar, and G. S. Kumar. '**Estimation of Iodine Levels in Drinking Water in Puducherry District**'. Biomedical Research, 18 (3) (2007), pp. 171-173.

Below and Kahlert 2001. H. Below and H. Kahlert. '**Determination of Iodide in Urine by Ion-Pair Chromatography with Electrochemical Detection**'. Fresenius' Journal of Analytical Chemistry, 371 (4) (2001), pp. 431-436. *https://doi.org/10.1007/s002160101080*.

Bhattacharjee et al. 2013. A. Bhattacharjee, A. K. Chandra, T. Malik, H. K. Tiwari, and A. Mishra. '**Iodine Content in Edible Salt and Drinking Water in Sub-Himalayan Tarai Region of Eastern Uttar Pradesh, India**'. International Journal of Biological and Medical Research, 4 (1) (2013), pp. 2834-2838. *http://14.139.206.50:8080/jspui/handle/1/4705*.

Bichsel and von Gunten 1999. Y. Bichsel and U. von Gunten. '**Determination of Iodide and Iodate by Ion Chromatography with Postcolumn Reaction and UV/Visible Detection**'. Analytical Chemistry, 71 (1) (1999), pp. 34-38. *https://doi.org/10.1021/ac980658j*.

Bilabina et al. 1994. I. Bilabina, M. Brazier, H. Bour, A. Doh, and G. Desmet. '**Evaluation of Iodide Deficiency in Togo Using an Optimized Potentiometric Method for Iodide Estimation in Urine**'. Annales De Biologie Clinique, 52 (4) (1994), pp. 261-264.

Bingui et al. 2006. Z. Binghui, Z. Zhixiong, and Y. Jing. '**Ion Chromatographic Determination of Trace Iodate, Chlorite, Chlorate, Bromide, Bromate and Nitrite in Drinking Water Using Suppressed Conductivity Detection and Visible Detection**'. Journal of Chromatography A, 1118 (1) (2006), pp. 106-110. *https://doi.org/10.1016/j.chroma.2006.01.139*.

Birke et al. 2010. M. Birke, C. Reimann, A. Demetriades, U. Rauch, H. Lorenz, B. Harazim, and W. Glatte. '**Determination of Major and Trace Elements in European Bottled Mineral Water — Analytical Methods**'. Journal of Geochemical Exploration, 107 (3) (2010), pp. 217-226. *https://doi.org/10.1016/j.gexplo.2010.05.005*.

Boleda et al. 2013. M. R. Boleda, M. T. Galceran, and F. Ventura. '**Validation and Uncertainty Estimation of a Multiresidue Method for Pharmaceuticals in Surface and Treated Waters by Liquid Chromatography–Tandem Mass Spectrometry**'. Journal of Chromatography A, 1286 (2013), pp. 146-158. *https://doi.org/10.1016/j.chroma.2013.02.077*.

Brandão et al. 1995. A. C. M. Brandão, W. W. Buchberger, E. C. V. Butler, P. A. Fagan, and P. R. Haddad. '**Matrix-Elimination Ion Chromatography with Post-Column Reaction Detection for the Determination of Iodide in Saline Waters**'. Journal of Chromatography A, 706 (1-2) (1995), pp. 271-275. *https://doi.org/10.1016/0021-9673(94)01207-U*.

Brown et al. 2007. C. F. Brown, K. N. Geiszler, and M. J. Lindberg. '**Analysis of ^{129}I in Groundwater Samples: Direct and Quantitative Results below the Drinking Water Standard**'. Applied Geochemistry, 22 (3) (2007), pp. 648-655. *https://doi.org/10.1016/j.apgeochem.2006.12.010*.

Bruchet et al. 2005. A. Bruchet, C. Hochereau, C. Picard, V. Decottignies, J. M. Rodrigues, and M-L. Janex-Habibi. '**Analysis of Drugs and Personal Care Products in French Source and Drinking Waters: The Analytical Challenge and Examples of Application**'. Water Science and Technology, 52 (8) (2005), pp. 53-61. *https://doi.org/10.2166/wst.2005.0224*.

Bruggink et al. 2007. C. Bruggink, W. van Rossum, E. Spijkerman, and E. van Beelen. '**Iodide Analysis by Anion-Exchange Chromatography and Pulsed Amperometric Detection in Surface Water and Adsorbable Organic Iodide**'. Journal of Chromatography A, 1144 (2) (2007), pp. 170-174. *https://doi.org/10.1016/j.chroma.2007.01.083*.

Bruins and Maurer 1989. J. Bruins and W. Maurer. '**Zur direkten Bestimmung von Iodidspuren mittels Ionen-Chromatographie in Wässern mit hohem Salzgehalt**'. Fresenius' Zeitschrift für analytische Chemie, 334 (1989), pp. 122-125. *https://doi.org/10.1007/BF00476670*.

Buchberger 1988. W. Buchberger. '**Determination of Iodide and Bromide by Ion Chromatography with Post-Column Reaction Detection**'. Journal of Chromatography, 439 (1) (1988), pp. 129-135. *https://doi.org/10.1016/s0021-9673(01)81681-2*.

Buchberger and Winsauer 1985. W. Buchberger, and K. Winsauer. '**Determination of Traces of Iodide in Serum and Urine by Ion Chromatography**'. Microchimica Acta, 87 (5-6) (1985), pp. 347-352. *https://doi.org/10.1007/BF01196833*.

Buchberger and Ahrer 1999. W. Buchberger and W. Ahrer. '**Combination of Suppressed and Non-Suppressed Ion Chromatography with Atmospheric Pressure Ionization Mass Spectrometry for the Determination of Anions**'. Journal of Chromatography A*,* 850 (1-2) (1999), pp. 99-106. *https://doi.org/10.1016/S0021-9673(99)00273-3*.

Buchberger and Huebauer 1989. W. Buchberger and U. Huebauer. '**Selective Determination of Bromide and Iodide in Serum and Urine by Gas Chromatography**'. Mikrochimica Acta, 99 (1-2) (1989), pp. 137-142. *https://doi.org/10.1007/BF01242799*.

Butler and Gershey 1984. E. C. V. Butler and R. M. Gershey. '**Application of Ion-Exchange Chromatography with an Ion-Selective Electrode Detector to Iodine Determination in Natural Waters**'. Analytica Chimica Acta, 164 (1984), pp. 153-161. *https://doi.org/10.1016/S0003-2670(00)85627-5*.

Cancho et al. 2000. B. Cancho, F. Ventura, M. T. Galceran, A. Diaz, and S. Ricart. '**Determination, Synthesis and Survey of Iodinated Trihalomethanes in Water Treatment Processes**'. Water Research, 34 (13) (2000), pp. 3380-3390. *https://doi.org/10.1016/S0043-1354(00)00079-8*.

Cataldi et al. 2005. T. Cataldi, A. Rubino, M. C. Laviola, and R. Ciriello. '**Comparison of Silver, Gold and Modified Platinum Electrodes for the Electrochemical Detection of Iodide in Urine Samples Following Ion Chromatography**'. Journal of Chromatography B, 827 (2) (2005), pp. 224-231. *https://doi.org/10.1016/j.jchromb.2005.09.017*.

Cemeli et al. 2006. E. Cemeli, E. D. Wagner, D. Anderson, S. D. Richardson, and M. J. Plewa. '**Modulation of the Cytotoxicity and Genotoxicity of the Drinking Water Disinfection Byproduct Iodoacetic Acid by Suppressors of Oxidative Stress**'. Environmental Science & Technology, 40 (6) (2006), pp. 1878-1883. *https://doi.org/10.1021/es051602r*.

Chandra et al. 2006. A. K. Chandra, L. H. Singh, S. Tripathy, A. Debnath, and J. Khanam. '**Iodine Nutritional Status of Children in North East India**'. The Indian Journal of Pediatrics, 73 (9) (2006), pp. 795-798. *https://doi.org/10.1007/BF02790388*.

Chandramouleeswaran et al. 1998. S. Chandramouleeswaran, B. Vijayalakshmi, S. Kartihkeyan, T. P. Rao, and C. S. P. Iyer. '**Ion-Chromatographic Determination of Iodide in Sea Water with UV Detection**'. Mikrochimica Acta 128 (1-2) (1998), pp. 75-77. *https://doi.org/10.1007/BF01242193*.

Chen and Stevens 1991. Q. Chen and J. L. Stevens. '**Inhibition of Iodoacetamide and t-Butylhydroperoxide Toxicity in LLC-PK1 Cells by Antioxidants: A Role for Lipid Peroxidation in Alkylation Induced Cytotoxicity**'. Archives of Biochemistry and Biophysics, 284 (2) (1991), pp. 422-430. *https://doi.org/10.1016/0003-9861(91)90318-D*.

Chen and Hibbert 1997. Z. Chen and D. B. Hibbert. '**Simultaneous Amperometric and Potentiometric Detection of Inorganic Anions in Flow Systems Using Platinum and Silver/Silver Chloride Electrodes**'. Analytica Chimica Acta, 350 (1-2) (1997), pp. 1-6. *https://doi.org/10.1016/S0003-2670(97)00328-0*.

Cheng et al. 2005. J. Cheng, P. Jandik, and N. Avdalovic. '**Pulsed Amperometric Detection of Sulfide, Cyanide, Iodide, Thiosulfate, Bromide and Thiocyanate with Microfabricated Disposable Silver Working Electrodes in Ion**

Chromatography'. Analytica Chimica Acta, 536 (1-2) (2005), pp. 267-274. *https://doi.org/10.1016/j.aca.2004.12.039*.

Collins 1969. A. G. Collins. '**Chemistry of Some Anadarko Basin Brines Containing High Concentrations of Iodide**'. Chemical Geology, Geochemistry of Subsurface Brines, 4 (1-2) (1969), pp. 169-187. *https://doi.org/10.1016/0009-2541(69)90044-8*.

Day and Powell-Jackson 1972. T. K. Day and P. R. Powell-Jackson. '**Fluoride, water hardness and endemic goitre**'. The Lancet, 299 (7761) (1972), pp. 1135-1138. *https://doi.org/10.1016/S0140-6736(72)91370-0*.

Dean 1963. G. A. Dean. '**The Iodine Content of Some New Zealand Drinking Waters with a Note on the Contribution from Sea Spray to the Iodine in Rain**'. New Zealand Journal of Science 6 (2) (1963), pp. 208-214'. *https://doi.org/10.1016/0011-7471(64)90564-9*.

Dinelli et al. 2012. E. Dinelli, A. Lima, S. Albanese, M. Birke, D. Cicchella, L. Giaccio, P. Valera, and B. De Vivo. '**Major and Trace Elements in Tap Water from Italy**'. Journal of Geochemical Exploration, 112 (2012), pp. 54-75. *https://doi.org/10.1016/j.gexplo.2011.07.009*.

Dionex Incorporation 2009. '**Determination of Iodide and Iodate in Seawater and Iodized Table Salt by HPLC with UV Detection**'. Application Note 236. *http://dionex.su/en-us/webdocs/81306-AN236-LC-Iodide-Seawater-15Sept2009-LPN2312.pdf*.

Dionex (now part of Thermo Scientific) Application Update 122. 1998. '**Application Update 122. The Determination of Iodide in Brine**'. Dionex Corporation, (1998). *https://assets.thermofisher.com/TFS-Assets/CMD/Application-Notes/4185-AU122_LPN034075-01.pdf*.

Dorman and Steinberg 2010. J. W. Dorman and S. M. Steinberg. '**Analysis of Iodide and Iodate in Lake Mead, Nevada Using a Headspace Derivatization Gas Chromatography–Mass Spectrometry**'. Environmental Monitoring and Assessment, 161 (1–4) (2010), pp. 229-236. *https://doi.org/10.1007/s10661-008-0740*-5.

Dudoit and Pergantis 2001. A. Dudoit and S. A. Pergantis. '**Ion Chromatography in Series with Conductivity Detection and Inductively Coupled Plasma-Mass Spectrometry for the Determination of Nine Halogen, Metalloid and Non-Metal Species in Drinking Water**'. Journal of Analytical Atomic Spectrometry, 16 (6) (2001), pp. 575-580. *https://doi.org/10.1039/B100783L*.

Duirk et al. 2011. S. E. Duirk, C. Lindell, C. C. Cornelison, J. Kormos, T. A. Ternes, M. Attene-Ramos, J. Osiol, E. D. Wagner, M. J. Plewa, and S. D. Richardson. '**Formation of Toxic Iodinated Disinfection By-Products from Compounds Used in Medical Imaging**'. Environmental Science & Technology, 45 (16) (2011), pp. 6845-6854. *https://doi.org/10.1021/es200983f*.

Ens et al. 2014. W. Ens, F. Senner, B. Gygax, and G. Schlotterbeck. '**Development, Validation, and Application of a Novel LC-MS/MS Trace Analysis Method for the Simultaneous Quantification of Seven Iodinated X-Ray Contrast Media and Three Artificial Sweeteners in Surface, Ground, and Drinking Water**'. Analytical and Bioanalytical Chemistry, 406 (12) (2014), pp. 2789-2798. *https://doi.org/10.1007/s00216-014-7712-0*.

Escobar-Hoyos et al. 2013. L. F. Escobar-Hoyos, L. S. Hoyos-Giraldo, E. Londoño-Velasco, I. Reyes-Carvajal, D. Saavedra-Trujillo, S. Carvajal-Varona, A. Sánchez-Gómez, E. D. Wagner, and M. J. Plewa. '**Genotoxic and Clastogenic Effects of Monohaloacetic Acid Drinking Water Disinfection By-Products in Primary Human Lymphocytes**'. Water Research 47 (10) (2013), pp. 3282-3290. *https://doi.org/10.1016/j.watres.2013.02.052*.

Fabbri et al. 2016. D. Fabbri, P. Calza, D. Dalmasso, P. Chiarelli, V. Santoro, and C. Medana. '**Iodinated X-Ray Contrast Agents: Photoinduced Transformation and Monitoring in Surface Water**'. Science of The Total Environment, 572 (2016), pp. 340-351. *https://doi.org/10.1016/j.scitotenv.2016.08.003*.

Fordyce et al. 2000. F. M. Fordyce, C. C. Johnson, U. R.B. Navaratna, J. D. Appleton, and C. B. Dissanayake. '**Selenium and Iodine in Soil, Rice and Drinking Water in Relation to Endemic Goitre in Sri Lanka**'. Science of The Total Environment, 263 (1–3) (2000), pp. 127-141. *https://doi.org/10.1016/S0048-9697(00)00684-7*.

Franks and Pullen 1974. M. C. Franks and D. L. Pullen. '**A Technique for the Determination of Trace Anions by the Combination of a Potentiometric Sensor and Liquid Chromatography, with Particular Reference to the Determination of Halides**'. Analyst, 99 (1181) (1974), pp. 503-514. *https://doi.org/10.1039/AN9749900503*.

Fuge 1989. R. Fuge. '**Iodine in Waters: Possible Links with Endemic Goitre**'. Applied Geochemistry, 4 (2) (1989), pp. 203-208. *https://doi.org/10.1016/0883-2927(89)90051-6*.

Fuge and Johnson 1986. R. Fuge and C. Johnson. '**The Geochemistry of Iodine**'. Environmental Geochemistry and Health, 8 (2) (1986), pp. 31-54. *https://doi.org/10.1007/BF02311063*.

Gallard et al. 2009. H. Gallard, S. Allard, R. Nicolau, U. von Gunten, and J-P. Croué. '**Formation of Iodinated Organic Compounds by Oxidation of Iodide-Containing Waters with Manganese Dioxide**'. Environmental Science & Technology, 43 (18) (2009), pp. 7003-7009. *https://doi.org/10.1021/es9010338*.

Gao et al. 2018. Y. Gao, C. Yang, J. Ma, and M. Yin. '**Characteristics of the Trace Elements and Arsenic, Iodine and Bromine Species in Snow in East-Central China**'. Atmospheric Environment, 174 (2018), pp. 43-53. *https://doi.org/10.1016/j.atmosenv.2017.11.015*.

Gilfedder et al. 2007. B. S. Gilfedder, M. Petri, and H. Biester. '**Iodine and Bromine Speciation in Snow and the Effect of Orographically Induced Precipitation**'. Atmospheric Chemistry and Physics, 7 (10) (2007), pp. 2661-2669. *https://doi.org/10.5194/acp-7-2661-2007*.

Gilfedder et al. 2008. B. S. Gilfedder, M. Petri, and H. Biester. '**Iodine Speciation and Cycling in Limnic Systems: Observations from a Humic Rich Headwater Lake (Mummelsee)**'. Biogeosciences Discussions, 5 (2008), pp. 25-64. *https://doi.org/10.5194/bgd-5-25-2008*.

Gilfedder et al. 2009. B. S. Gilfedder, M. Petri, and H. Biester. '**Iodine Speciation and Cycling in Fresh Waters: A Case Study from a Humic Rich Headwater Lake (Mummelsee)**'. Journal of Limnology, 68 (2) (2009), pp. 396-408. *https://doi.org/10.4081/jlimnol.2009.396*.

Gilfedder et al. 2010. B. S. Gilfedder, M. Petri, M. Wessels, and H. Biester. '**An Iodine Mass-Balance for Lake Constance, Germany: Insights into Iodine Speciation Changes and Fluxes**'. Geochimica et Cosmochimica Acta, 74 (11) (2010), pp. 3090-3111. *https://doi.org/10.1016/j.gca.2010.03.008*.

Gong et al. 2017. T. Gong, Y. Tao, X. Zhang, S. Hu, J. Yin, Q. Xian, J. Ma, and B. Xu. '**Transformation among Aromatic Iodinated Disinfection Byproducts in the Presence of Monochloramine: From Monoiodophenol to Triiodophenol and Diiodonitrophenol**'. Environmental Science & Technology, 51 (18) (2017), pp. 10562-10571. *https://doi.org/10.1021/acs.est.7b03323*.

Gong and Zhang 2013. T. Gong and X. Zhang. '**Determination of Iodide, Iodate and Organo-Iodine in Waters with a New Total Organic Iodine Measurement Approach**'. Water Research, 47 (17) (2013), pp. 6660-6669. *https://doi.org/10.1016/j.watres.2013.08.039*.

Goslan 2016. E. H. Goslan. '**Identification of High Risk Sites for Iodinated DBP Formation**'. Project DWI 70/2/291 Report. Department: School of Water, Energy and Environment, (2016). *http://dwi.defra.gov.uk/research/completed-research/reports/DWI70-2-291.pdf*.

Gruchlik et al. 2015. Y. Gruchlik, A. Heitz, C. Joll, U. von Gunten, S. Allard, J. Criquet, S. McDonald, J. Tan, L. Breckler, F. Bradder, G. Roeszler, and D. Halliwell. '**Novel Treatment Technologies for the Minimisation of Bromide and Iodide in Drinking Water**'. Final Report for ARC Linkage Project 100100285. CWQRC Report No: CWQRC-2015-004, 44, (2015).

Guo et al. 2016. W. Guo, Y. Wang, J. Li, Y. Peng, L. Jin, Q. Guo, and S. Hu. '**Rapid and Sensitive Speciation of Inorganic Iodine in Groundwater by IC-ICP-MS With Ar-CH_4 Mixed Plasma**'. Atomic Spectroscopy, 37 (1) (2016), pp. 7-12.

Haddad et al. 1985. P. R. Haddad, P. W. Alexander, and M. Trojanowicz. '**Ion Chromatography of Inorganic Anions with Potentiometric Detection Using a Metallic Copper Electrode**'. Journal of Chromatography A, 321 (1985), pp. 363-374. *https://doi.org/10.1016/S0021-9673(01)90454-6*.

Han et al. 2014. C. Han, J. Sun, H. Cheng, J. Liu, and Z. Xu. '**Speciation Analysis of Urine Iodine by Ion-Pair Reversed-Phase Liquid Chromatography and Inductively Coupled Plasma Mass Spectrometry**'. Analytical Methods, 6 (14) (2014), pp. 5369-5375. *https://doi.org/10.1039/C4AY00396A*.

Han et al. 1987. K. Han, W. F. Koch, and K. W. Pratt. '**Improved Procedure for the Determination of Iodide by Ion Chromatography with Electrochemical Detection**'. Analytical Chemistry, 59 (5) (1987), pp. 731-736. *https://doi.org/10.1021/ac00132a011*.

Han et al. 2012. X. Han, L. Cao, H. Cheng, J. Liu, and Z. Xu. '**Determination of Iodine Species in Seaweed and Seawater Samples Using Ion-Pair Reversed Phase High Performance Liquid Chromatography Coupled with Inductively Coupled Plasma Mass Spectrometry**'. Analytical Methods, 4 (10) (2012), pp. 3471-3477. *https://doi.org/10.1039/c2ay25871d*.

Hansen et al. 2011. V. Hansen, P. Yi, X. Hou, A. Aldahan, P. Roos, and G. Possnert. '**Iodide and Iodate (^{129}I and ^{127}I) in Surface Water of the Baltic Sea, Kattegat and Skagerrak**'. Science of The Total Environment, 412-413 (2011), pp. 296-303. *https://doi.org/10.1016/j.scitotenv.2011.10.001*.

Hansson et al. 1987. R. C. Hansson, M. J. Henderson, P. Jack, and R. D. Taylor. '**Iodoform Taste Complaints in Chloramination**'. Water Research, 21 (10) (1987), pp. 1265-1271. *https://doi.org/10.1016/0043-1354(87)90179-5*.

Helaleh et al. 2005. M. I. H. Helaleh, A. Al-Omair, K. Tanaka, and M. Mori. '**Ion Chromatography of Common Anions by Use of a Reversed-Phase Column Dynamically Coated with Fluorine-Containing Surfactant**'. Acta Chromatographica, (15) (2005), pp. 247-257.

Hikiba et al. 2005. H. Hikiba, E. Watanabe, J. C. Barrett, and T. Tsutsui. ‘**Ability of Fourteen Chemical Agents Used in Dental Practice to Induce Chromosome Aberrations in Syrian Hamster Embryo Cells**’. Journal of Pharmacological Sciences, 97 (1) (2005), pp. 146-152. *https://doi.org/10.1254/jphs.FPJ04044X*.

Hilliard et al. 1998. C. A. Hilliard, M. J. Armstrong, C. I. Bradt, R. B. Hill, S. K. Greenwood, and S. M. Galloway. ‘**Chromosome Aberrations in Vitro Related to Cytotoxicity of Nonmutagenic Chemicals and Metabolic Poisons**’. Environmental and Molecular Mutagenesis, 31 (4) (1998), pp. 316-326. *https://doi.org/10.1002/(SICI)1098-2280(1998)31:4<316::AID-EM3>3.0.CO;2-G*.

Hollender et al. 2008. J. Hollender, H. Singer, and C. S. McArdell. ‘**Polar Organic Micropollutants In The Water Cycle**’. In *Dangerous Pollutants (Xenobiotics) in Urban Water Cycle*. Edited by P. Hlavinek, O. Bonacci, J. Marsalek, and I. Mahrikova. NATO Science for Peace and Security Series, (2008), pp. 103-116.

Hong et al. 2018. H. Hong, H. Wu, J. Chen, B. Wu, H. Yu, B. Yan, and Y. Liang. ‘**Cytotoxicity Induced by Iodinated Haloacetamides via ROS Accumulation and Apoptosis in HepG-2 cells**’. Environmental Pollution, 242 Part A (2018), pp. 191-197. *https://doi.org/10.1016/j.envpol.2018.06.090*.

Hu and Moran 2005. Q. Hu and J. E. Moran. ‘**Simultaneous Analyses and Applications of Multiple Fluorobenzoate and Halide Tracers in Hydrologic Studies**’. Hydrological Processes, 19 (14) (2005), pp. 2671-2687. *https://doi.org/10.1002/hyp.5780*.

Hu et al. 2005. Q. Hu, P. Zhao, J. E. Moran, and J. C. Seaman. ‘**Sorption and Transport of Iodine Species in Sediments from the Savannah River and Hanford Sites**’. Journal of Contaminant Hydrology, 78 (3) (2005), pp. 185-205. *https://doi.org/10.1016/j.jconhyd.2005.05.007*.

Hu et al. 2018. S. Hu, T. Gong, Q. Xian, J. Wang, J. Ma, Z. Li, J. Yin, B. Zhang, and B. Xu. ‘**Formation of Iodinated Trihalomethanes and Haloacetic Acids from Aromatic Iodinated Disinfection Byproducts during Chloramination**’. Water Research, 147 (2018), pp. 254-263. *https://doi.org/10.1016/j.watres.2018.09.058*.

Hurst et al. 1984. W. Hurst, J. W. Stefovic, and W. J. White. ‘**HPLC Determination of Iodide in Scrum Using Paired Ion Chromatography with Electrochemical Detection**’. Journal of Liquid Chromatography, 7 (10) (1984), pp. 2021-2130. *https://doi.org/10.1080/01483918408068854*.

Hurum and Rohrer 2009. D. Hurum and J. Rohrer. ‘**Determination of Iodide in Seawater and Other Saline Matrices Using a Reagent-Free Ion Chromatography System with Suppressed Conductivity and UV Detections**’. Application Note 239. Dionex Incorporation (now part of Thermo Fisher Scientific). (2009). Available online: *https://assets.thermofisher.com/TFS-Assets/CMD/Application-Notes/AN-239-IC-Iodide-Seawater-AN71348-EN.pdf*.

‘Iodide Anion by Ion Chromatography, EPA Method 300’ 2012. '**Iodide by Ion Chromatography**'. SOP: WPCL-AA-041 CDPR, (2012), pp. 1-11. *https://www.cdpr.ca.gov/docs/emon/pubs/anl_methds/iodide_by_ion_chromatography.pdf*.

Ito 1997. K. Ito. ‘**Determination of Iodide in Seawater by Ion Chromatography**’. Analytical Chemistry, 69 (17) (1997), pp. 3628-3632. *https://doi.org/10.1021/ac9700787*.

Ito 1999. K. Ito. ‘**Semi-Micro Ion Chromatography of Iodide in Seawater**’. Journal of Chromatography A, 830 (1) (1999), pp. 211-217. *https://doi.org/10.1016/S0021-9673(98)00910-8*.

Ito et al. 2012. K. Ito, R. Nomura, T. Fujii, M. Tanaka, T. Tsumura, H. Shibata, and T. Hirokawa. ‘**Determination of Nitrite, Nitrate, Bromide, and Iodide in Seawater by Ion Chromatography with UV Detection Using Dilauryldimethylammonium-Coated Monolithic ODS Columns and Sodium Chloride as an Eluent**’. Analytical and Bioanalytical Chemistry, 404 (8) (2012), pp. 2513-2517. *https://doi.org/10.1007/s00216-012-6405-9*.

Ito et al. 1991. K. Ito, E. Shoto, and H. Sunahara. ‘**Ion Chromatography of Inorganic Iodine Species Using C18 Reversed-Phase Columns Coated with Cetyltrimethylammonium**’. Journal of Chromatography A, 549 (1991), pp. 265-272. *https://doi.org/10.1016/S0021-9673(00)91438-9*.

Ito and Sunahara 1990. K. Ito and H. Sunahara. ‘**Ion Chromatography with Ultraviolet and Amperometric Detection for Iodide and Thiocyanate in Concentrated Salt Solutions**’. Journal of Chromatography A, 502 (1990), pp. 121-129. *https://doi.org/10.1016/S0021-9673(01)89569-8*.

Jafioll et al. 1992. C. Jaffiol, N. Perezi, R. Baylet, L. Baldet, M. Chapat, and H. Lapinski. ‘**Endemic goiter in the La Kara region (Togo). Analysis of etiologic factors**’. Bulletin De l’Academie Nationale De Medecine, 176 (4) (1992), pp. 557-565; discussion 566-567.

Jeong et al. 2015. C. H. Jeong, C. Postigo, S. D. Richardson, J. E. Simmons, S. Y. Kimura, B. J. Mariñas, D. Barceló, P. Liang, E. D. Wagner, and M. J. Plewa. ‘**Occurrence and Comparative Toxicity of Haloacetaldehyde Disinfection Byproducts in Drinking Water**’. Environmental Science & Technology, 49 (23) (2015), pp. 13749-13759. *https://doi.org/10.1021/es506358x*.

Johnson et al. 2002. C. C. Johnson, M. H. Strutt, M. Mreunas, and M. Mounir. '**Iodine in the Environment of the High Atlas Mountain Area of Morocco**.' British Geological Survey Commissioned Report CR/02/196N, (2002).

Kamavisdar and Patel. 2002. A. Kamavisdar and R. M. Patel. '**Flow Injection Spectrophotometric Determination of Iodide in Environmental Samples**'. Microchimica Acta, 140 (1-2) (2002), pp. 119-124. *https://doi.org/10.1007/s00604-002-0901-0*.

Kargalioglu et al. 2002. Y. Kargalioglu, B. J. McMillan, R. A. Minear, and M. J. Plewa. '**Analysis of the Cytotoxicity and Mutagenicity of Drinking Water Disinfection By-Products In Salmonella Typhimurium**'. Teratogenesis, Carcinogenesis, and Mutagenesis, 22 (2) (2002), pp. 113-128. *https://doi.org/10.1002/tcm.10010*.

Kim et al. 2007. S. D. Kim, J. Cho, I. S. Kim, B. J. Vanderford, and S. A. Snyder. '**Occurrence and Removal of Pharmaceuticals and Endocrine Disruptors in South Korean Surface, Drinking, and Waste Waters**'. Water Research, 41 (5) (2007), pp. 1013-1021. *https://doi.org/10.1016/j.watres.2006.06.034*.

Koh et al. 1988. T. Koh, M. Ono, and I. Makino. '**Spectrophotometric Determination of Iodide at the 10^{-6} mol L^{-1} Level by Solvent Extraction with Methylene Blue**'. Analyst, 113 (6) (1988), pp. 945-948. *https://doi.org/10.1039/an9881300945*.

Kormos et al. 2010. J. L. Kormos, M. Schulz, H-P. E. Kohler, and T. A. Ternes. '**Biotransformation of Selected Iodinated X-Ray Contrast Media and Characterization of Microbial Transformation Pathways**'. Environmental Science & Technology, 44 (13) (2010), pp. 4998-5007. *https://doi.org/10.1021/es1007214*.

Kormos et al. 2011. J. L. Kormos, M. Schulz, and T. A. Ternes. '**Occurrence of Iodinated X-Ray Contrast Media and Their Biotransformation Products in the Urban Water Cycle**'. Environmental Science & Technology, 45 (20) (2011), pp. 8723-8732. *https://doi.org/10.1021/es2018187*.

Kormos et al. 2009. J. L. Kormos, M. Schulz, M. Wagner, and T. A. Ternes. '**Multistep Approach for the Structural Identification of Biotransformation Products of Iodinated X-Ray Contrast Media by Liquid Chromatography/Hybrid Triple Quadrupole Linear Ion Trap Mass Spectrometry and ^{1}H and ^{13}C Nuclear Magnetic Resonance**'. Analytical Chemistry, 81 (22) (2009), pp. 9216-9224. *https://doi.org/10.1021/ac9011717*.

Korobova 2010. E. Korobova. '**Soil and Landscape Geochemical Factors Which Contribute to Iodine Spatial Distribution in the Main Environmental Components and Food Chain in the Central Russian Plain**'. Journal of Geochemical Exploration, Selenium and iodine anomalies in soils and health, 107 (2) (2010), pp. 180-192. *https://doi.org/10.1016/j.gexplo.2010.03.003*.

Li et al. 2015. X. Li, J. Hu, D. Yin, and X. Hu. '**Solid-Phase Extraction Coupled with Ultra High Performance Liquid Chromatography and Electrospray Tandem Mass Spectrometry for the Highly Sensitive Determination of Five Iodinated X-Ray Contrast Media in Environmental Water Samples**'. Journal of Separation Science, 38 (11) (2015), pp. 1998-2005. *https://doi.org/10.1002/jssc.201401296*.

Liang et al. 2005. L. Liang, Y. Cai, S. Mou, and J. Cheng. '**Comparisons of Disposable and Conventional Silver Working Electrode for the Determination of Iodide Using High-Performance Anion-Exchange Chromatography with Pulsed Amperometric Detection**'. Journal of Chromatography A, 1085 (1) (2005), pp. 37-41. *https://doi.org/10.1016/j.chroma.2004.12.060*.

Liberatore et al. 2017. H. K. Liberatore, M. J. Plewa, E. D. Wagner, J. M. VanBriesen, D. B. Burnett, L. H. Cizmas, and S. D. Richardson. '**Identification and Comparative Mammalian Cell Cytotoxicity of New Iodo-Phenolic Disinfection Byproducts in Chloraminated Oil and Gas Wastewaters**'. Environmental Science & Technology Letters, 4 (11) (2017), pp. 475-480. *https://doi.org/10.1021/acs.estlett.7b00468*.

Liu et al. 1995. A. Liu, L. Xu, L. Tao, S. Dong, and E. Wang. '**Electrocatalytic Oxidation and Ion Chromatographic Detection of Br^{-}, I^{-}, SO_3^{2-}, $S_2O_3^{2-}$ and SCN^{-} at a Platinum Particle-Based Glassy Carbon Modified Electrode**'. Journal of Chromatography A, 699 (1-2) (1995), pp. 39-47. *https://doi.org/10.1016/0021-9673(95)00032-I*.

Liu et al. 2011. W. Liu, H. Yang, B. Li, and S. Xu. '**Determination of Bromine and Iodine Speciation in Drinking Water Using High Performance Liquid Chromatography-Inductively Coupled Plasma-Mass Spectrometry**'. Geostandards and Geoanalytical Research, 35 (1) (2011), pp. 69-74. *https://doi.org/10.1111/j.1751-908X.2010.00033.x*.

Liviac et al. 2010. D. Liviac, A. Creus, and R. Marcos. '**Genotoxicity Testing of Three Monohaloacetic Acids in TK6 Cells Using the Cytokinesis-Block Micronucleus Assay**'. Mutagenesis, 25 (5) (2010), pp. 505-509. *https://doi.org/10.1093/mutage/geq034*.

Longvah and Deosthale 1998. T. Longvah and Y. G. Deosthale. '**Iodine Content of Commonly Consumed Foods and Water from the Goitre-Endemic Northeast Region of India**'. Food Chemistry, 61 (3) (1998), pp. 327-331. *https://doi.org/10.1016/S0308-8146(97)00065-4*.

Lookabaugh et al. 1987. M. Lookabaugh, I. S. Krull, and W. R. LaCourse. '**Determination of Iodide and Thiosulfate by Paired-Ion, Reversed-Phase High-Performance Liquid Chromatography with Ultraviolet Absorbance, Electrochemical,**

and Conductimetric Detection'. Journal of Chromatography, 387 (1987), pp. 301-312. *https://doi.org/10.1016/s0021-9673(01)94533-9*.

Lu et al. 2005. Y-L. Lu, N-J. Wang, L. Zhu, G-X. Wang, H. Wu, L. Kuang, and W-M. Zhu. '**Investigation of Iodine Concentration in Salt, Water and Soil along the Coast of Zhejiang, China**'. Journal of Zhejiang University-SCIENCE B, 6 (12) (2005), pp. 1200-1205. *https://doi.org/10.1631/jzus.2005.B1200*.

Malongo et al. 2008. T. K. Malongo, S. Patris, P. Macours, F. Cotton, J. Nsangu, and J-M. Kauffmann. '**Highly Sensitive Determination of Iodide by Ion Chromatography with Amperometric Detection at a Silver-Based Carbon Paste Electrode**'. Talanta, 76 (3) (2008), pp. 540-547. *https://doi.org/10.1016/j.talanta.2008.03.053*.

Maros et al. 1989. L. Maros, M. Kaldy, and S. Igaz. '**Simultaneous Determination of Bromide and Iodide as Acetone Derivatives by Gas Chromatography and Electron Capture Detection in Natural Waters and Biological Fluids**'. Analytical Chemistry, 61 (7) (1989), pp. 733-735. *https://doi.org/10.1021/ac00182a018*.

McTaggart et al. 1994. A. R. McTaggart, E. C. V. Butler, P. R. Haddad, and J. H. Middleton. '**Iodide and Iodate Concentrations in Eastern Australian Subtropical Waters, with Iodide by Ion Chromatography**'. Marine Chemistry, 47 (2) (1994), pp. 159-172. *https://doi.org/10.1016/0304-4203(94)90106-6*.

Mehra and Frankenberger 1990. H. C. Mehra, and W. T. Frankenberger Jr. '**Simultaneous Determination of Cyanide, Chloride, Bromide, and Iodide in Environmental Samples Using Ion Chromatography with Amperometric Detection**'. Microchemical Journal, 41 (1) (1990), pp. 93-97. *https://doi.org/10.1016/0026-265X(90)90100-J*.

Melichercik et al. 2006. J. Melichercik, L. Szijarto, and A. R. Hill. '**Comparison of Ion-Specific Electrode and High Performance Liquid Chromatography Methods for the Determination of Iodide in Milk**'. Journal of Dairy Science, 89 (3) (2006), pp. 934-937. *https://doi.org/10.3168/jds.S0022-0302(06)72158-0*.

Mendoza et al. 2016. A. Mendoza, B. Zonja, N. Mastroianni, N. Negreira, M. López de Alda, S. Pérez, D. Barceló, A. Gil, and Y. Valcárcel. '**Drugs of Abuse, Cytostatic Drugs and Iodinated Contrast Media in Tap Water from the Madrid Region (Central Spain): A Case Study to Analyse Their Occurrence and Human Health Risk Characterization**'. Environment International, 86 (2016), pp. 107-118. *https://doi.org/10.1016/j.envint.2015.11.001*.

Mishra et al. 2000. S. Mishra, V. Singh, A. Jain, and K. K. Verma. '**Determination of Iodide by Derivatization to 4-Iodo-N,N-Dimethylaniline and Gas Chromatography–Mass Spectrometry**'. The Analyst, 125 (3) (2000), pp. 459-464. *https://doi.org/10.1039/a908363d*.

Moran et al. 2002. J. E. Moran, S. D. Oktay, and P. H. Santschi. '**Sources of Iodine and Iodine 129 in Rivers**'. Water Resources Research, 38 (8) (2002), pp **24**-1-**24**-10. *https://doi.org/10.1029/2001WR000622*.

Mori et al. 2006. M. Mori, K. Tanaka, T. Satori, M. Ikedo, W. Hu, and H. Itabashi. '**Influence of Acidic Eluent for Retention Behaviors of Common Anions and Cations by Ion-Exclusion/Cation-Exchange Chromatography on a Weakly Acidic Cation-Exchange Resin in the H^+-Form**'. Journal of Chromatography A, 1118 (1) (2006), pp. 51-55. *https://doi.org/10.1016/j.chroma.2006.02.083*.

Muellner et al. 2007. M. G. Muellner, E. D. Wagner, K. McCalla, S. D. Richardson, Y-T. Woo, and M. J. Plewa. '**Haloacetonitriles vs. Regulated Haloacetic Acids: Are Nitrogen-Containing DBPs More Toxic?**' Environmental Science & Technology, 41 (2) (2007), pp. 645-651. *https://doi.org/10.1021/es0617441*.

Muramatsu et al. 2001. Y. Muramatsu, U. Fehn, and S. Yoshida. '**Recycling of Iodine in Fore-Arc Areas: Evidence from the Iodine Brines in Chiba, Japan**'. Earth and Planetary Science Letters, 192 (4) (2001), pp. 583-593. *https://doi.org/10.1016/S0012-821X(01)00483-6*.

Muramatsu and Wedepohl 2001. Y. Muramatsu and K. H. Wedepohl. '**The Distribution of Iodine in the Earth's Crust**'. Chemical Geology, 147 (3-4) (2001), pp. 201-216. *https://doi.org/10.1016/S0009-2541(98)00013-8*.

Neal et al. 2007. C. Neal, M. Neal, H. Wickham, L. Hill, and S. Harman. '**Dissolved Iodine in Rainfall, Cloud, Stream and Groundwater in the Plynlimon Area of Mid-Wales**'. Hydrology and Earth System Sciences, 11 (1) (2007), pp. 283-293. *https://doi.org/10.5194/hess-11-283-2007*.

Negri et al. 2013. A. E. Negri, J. O. Fernández Niello, A. Wallner, A. Arazi, L. K. Fifield, and S. G. Tims. '**^{129}I Dispersion in Argentina: Concentrations in Fresh and Marine Water and Deposition Fluences in Patagonia**'. Environmental Science & Technology, 47 (17) (2013), pp. 9693-9698. *https://doi.org/10.1021/es400610h*.

Okamoto et al. 1998. T. Okamoto, A. Isozaki, and H. Nagashima. '**Studies on Elution Conditions for the Determination of Anions by Supressed Ion-Interaction Chromatography Using a Graphitized Carbon Column**'. Journal of Chromatography A, 800 (2) (1998), pp. 239-245. *https://doi.org/10.1016/S0021-9673(97)01148-5*.

Papadoyannis et al. 1998. I. N. Papadoyannis, V. F. Samanidou, and K. V. Moutsis. '**Determination of Silver Iodide by High Pressure Ion Chromatography in Soil and Water Matrices After Solid Phase Extraction**'. Journal of Liquid Chromatography & Related Technologies, 21 (3) (1998), pp. 361-379. *https://doi.org/10.1080/10826079808000496*.

Parsons et al. 2009. S. A. Parsons, E. H. Goslan, S. A. Rocks, P. Holmes, L. S. Levy, and S. W. Krasner. '**Study into the Formation of Disinfection By-Products of Chloramination, Potential Health Implications and Techniques for Minimisation**'. Cranfield University, (2009). *http://www.windsofjustice.org.uk/wp-content/uploads/2015/07/THMs-Cranfield-University.pdf.*

Pedersen et. al. 1999. K. M. Pedersen, P. Laurberg, S. Nøhr, A. Jørgensen, and S. Andersen. '**Iodine in Drinking Water Varies by More than 100-Fold in Denmark. Importance for Iodine Content of Infant Formulas**'. European Journal of Endocrinology, 140 (5) (1999), pp. 400-403. *https://doi.org/10.1530/eje.0.1400400*.

Petrović et al. 2014. M. Petrović, B. Škrbić, J. Živančev, L. Ferrando-Climent, and D. Barceló. '**Determination of 81 Pharmaceutical Drugs by High Performance Liquid Chromatography Coupled to Mass Spectrometry with Hybrid Triple Quadrupole–Linear Ion Trap in Different Types of Water in Serbia**'. Science of The Total Environment, 468-469 (2014), pp. 415-428. *https://doi.org/10.1016/j.scitotenv.2013.08.079*.

Plewa et al. 2002. M. J. Plewa, Y. Kargalioglu, D. Vankerk, R. A. Minear, and E. D. Wagner. '**Mammalian Cell Cytotoxicity and Genotoxicity Analysis of Drinking Water Disinfection By-Products**'. Environmental and Molecular Mutagenesis, 40 (2) (2002), pp. 134-142. *https://doi.org/10.1002/em.10092*.

Plewa et al. 2007. M. J. Plewa, M. G. Muellner, S. D. Richardson, F. Fasano, K. M. Buettner, Y-T. Woo, A. B. McKague, and E. D. Wagner. '**Occurrence, Synthesis, and Mammalian Cell Cytotoxicity and Genotoxicity of Haloacetamides: An Emerging Class of Nitrogenous Drinking Water Disinfection Byproducts**'. Environmental Science & Technology, 42 (3) (2007), pp. 955-961. *https://doi.org/10.1021/es071754h*.

Plewa et al. 2010. M. J. Plewa, J. E. Simmons, S. D. Richardson, and E. D. Wagner. '**Mammalian Cell Cytotoxicity and Genotoxicity of the Haloacetic Acids, a Major Class of Drinking Water Disinfection by-Products**'. Environmental and Molecular Mutagenesis, 51 (8-9) (2010), pp. 871-878. *https://doi.org/10.1002/em.20585*.

Plewa et al. 2004. M. J. Plewa, E. D. Wagner, S. D. Richardson, A. D. Thruston, Y-T. Woo, and A. B. McKague. '**Chemical and Biological Characterization of Newly Discovered Iodoacid Drinking Water Disinfection Byproducts**'. Environmental Science & Technology, 38 (18) (2004), pp. 4713-4722. *https://doi.org/10.1021/es049971v*.

Procházka et al. 2015. E. Procházka, B. I. Escher, M. J. Plewa, and F. D. L. Leusch. '***In Vitro* Cytotoxicity and Adaptive Stress Responses to Selected Haloacetic Acid and Halobenzoquinone Water Disinfection Byproducts**'. Chemical Research in Toxicology, 28 (10) (2015), pp. 2059-2068. *https://doi.org/10.1021/acs.chemrestox.5b00283*.

Putschew et al. 2000. A. Putschew, S. Wischnack, and M. Jekel. '**Occurrence of Triiodinated X-Ray Contrast Agents in the Aquatic Environment**'. The Science of The Total Environment, 255 (1-3) (2000), pp. 129-134. *https://doi.org/10.1016/S0048-9697(00)00461-7*.

Putschew et al. 2001. A. Putschew, S. Schittko, and M. Jekel. '**Quantification of Triiodinated Benzene Derivatives and X-Ray Contrast Media in Water Samples by Liquid Chromatography–Electrospray Tandem Mass Spectrometry**'. Journal of Chromatography A, 930 (1-2) (2001), pp. 127-134. *https://doi.org/10.1016/S0021-9673(01)01186-4*.

Qin et al. 2014. X. Qin, T. Zhang, Z. Gan, and H. Sun. '**Spatial Distribution of Perchlorate, Iodide and Thiocyanate in the Aquatic Environment of Tianjin, China: Environmental Source Analysis**'. Chemosphere, 111 (2014), pp. 201-208. *https://doi.org/10.1016/j.chemosphere.2014.03.082*.

Rasmussen et al. 2000. L. Rasmussen, E. H. Larsen, and L. Ovesen. '**Iodine Content in Drinking Water and Other Beverages in Denmark**'. European Journal of Clinical Nutrition, 54 (2000), pp. 57-60. *https://doi.org/10.1038/sj.ejcn.1600893*.

Redeker et al. 2014. M. Redeker, A. Wick, B. Meermann, and T. A. Ternes. '**Removal of the Iodinated X-Ray Contrast Medium Diatrizoate by Anaerobic Transformation**'. Environmental Science & Technology, 48 (17) (2014), pp. 10145-10154. *https://doi.org/10.1021/es5014714*.

Richardson et al. 2008. S. D. Richardson, F. Fasano, J. J. Ellington, F. G. Crumley, K. M. Buettner, J. J. Evans, B. C. Blount, L. K. Silva, T. J. Waite, G. W. Luther, A. B. McKague, R. J. Miltner, E. D. Wagner, and M. J. Plewa. '**Occurrence and Mammalian Cell Toxicity of Iodinated Disinfection Byproducts in Drinking Water**'. Environmental Science & Technology, 42 (22) (2008), pp. 8330-8338. *https://doi.org/10.1021/es801169k*.

Rocklin and Johnson 1983. R. D. Rocklin and E. L. Johnson. '**Determination of Cyanide, Sulfide, Iodide, and Bromide by Ion Chromatography with Electrochemical Detection**'. Analytical Chemistry 55 (1) (1983), pp. 4-7. *https://doi.org/10.1021/ac00252a005*.

Rodriguez et al. 2016. E. S. Rodriguez, A. N. Setiawan, S. Pope, P. R. Haddad, P. N. Nesterenko, and B. Paull. '**A Simple and Sensitive Method for the Determination of Iodide and Iodate in Raw, Ultraviolet- and Ozone-Treated Aquacultural Seawater Samples Using Ion Chromatography Coupled to an Ultraviolet Detector**'. Analytical Methods, 8 (28) (2016), pp. 5587-5595. *https://doi.org/10.1039/C6AY01381C*.

Sacher et al. 2001. F. Sacher, F. T. Lange, H-J. Brauch, and I. Blankenhorn. '**Pharmaceuticals in Groundwaters Analytical Methods and Results of a Monitoring Program in Baden-Württemberg, Germany**'. Journal of Chromatography A, 12 (2001), pp. 199-210. *https://doi.org/10.1016/S0021-9673(01)01266-3*.

Sacher et al. 2005. F. Sacher, B. Raue, and H-J. Brauch. '**Analysis of Iodinated X-Ray Contrast Agents in Water Samples by Ion Chromatography and Inductively-Coupled Plasma Mass Spectrometry**'. Journal of Chromatography A, 1085 (1) (2005), pp. 117-123. *https://doi.org/10.1016/j.chroma.2005.01.031*.

Safe Drinking Water Committee 1980. National Research Council (US) Safe Drinking Water Committee. '**The Contribution of Drinking Water to Mineral Nutrition in Humans**'. National Academies Press (US), (1980). *https://www.ncbi.nlm.nih.gov/books/NBK216589/*.

Salminen et al. 2005. R. Salminen, M. J. Batista, M. Bidovec, and T. Tarvainen. '**Geochemical Atlas of Europe. Part 1 - Background Information, Methodology and Maps**'. Geological Survey of Finland, Otamedia Oy, Espoo, (2005). *http://weppi.gtk.fi/publ/foregsatlas/text/I.pdf*.

Sayess et al. 2017. R. Sayess, A. Khalil, M. Shah, D. A. Reckhow, and K. J. Godri Pollitt. '**Comparative Cytotoxicity of Six Iodinated Disinfection Byproducts on Nontransformed Epithelial Human Colon Cells**'. Environmental Science & Technology Letters, 4 (4) (2017), pp. 143-148. *https://doi.org/10.1021/acs.estlett.7b00064*.

Sayess and Reckhow 2017. R. Sayess and D. A. Reckhow. '**An Improved Method for Total Organic Iodine in Drinking Water**'. Water Research, 108 (2017), pp. 250-259. *https://doi.org/10.1016/j.watres.2016.10.079*.

Schnetger and Muramatsu 1996. B. Schnetger, and Y. Muramatsu. '**Determination of Halogens, with Special Reference to Iodine, in Geological and Biological Samples Using Pyrohydrolysis for Preparation and Inductively Coupled Plasma Mass Spectrometry and Ion Chromatography for Measurement**'. The Analyst, 121 (11) (1996), pp. 1627-1631. *https://doi.org/10.1039/AN9962101627*.

Schwehr and Santschi 2003. K. A. Schwehr, and P. H. Santschi. '**Sensitive Determination of Iodine Species, Including Organo-Iodine, for Freshwater and Seawater Samples Using High Performance Liquid Chromatography and Spectrophotometric Detection**'. Analytica Chimica Acta, 482 (1) (2003), pp. 59-71. *https://doi.org/10.1016/S0003-2670(03)00197-1*.

Seitz et al. 2006a. W. Seitz, J-Q. Jiang, W. H. Weber, B. J. Lloyd, M. Maier, and D. Maier. '**Removal of Iodinated X-Ray Contrast Media During Drinking Water Treatment**'. Environmental Chemistry, 3 (1) (2006), pp. 35-39. *https://doi.org/10.1071/EN05036*.

Seitz et al. 2006b. W. Seitz, W. H. Weber, J-Q. Jiang, B. J. Lloyd, M. Maier, D. Maier, and W. Schulz. '**Monitoring of Iodinated X-Ray Contrast Media in Surface Water**'. Chemosphere, 64 (8) (2006), pp. 1318-1324. *https://doi.org/10.1016/j.chemosphere.2005.12.030*.

Sharma et al. 1999. S. K. Sharma, P. K. Chelleng, S. Gogoi, and J. Mahanta. '**Iodine Status of Food and Drinking Water of a Sub-Himalayan Zone of India**'. International Journal of Food Sciences and Nutrition, 50 (2) (1999), pp. 95-98. *https://doi.org/10.1080/096374899101274*.

Shen et al. 2011. H. Shen, S. Liu, D. Sun, S. Zhang, X. Su, Y. Shen, and H. Han. '**Geographical Distribution of Drinking-Water with High Iodine Level and Association between High Iodine Level in Drinking-Water and Goitre: A Chinese National Investigation**'. British Journal of Nutrition, 106 (2) (2011), pp. 243-247. *https://doi.org/10.1017/S0007114511000055*.

Shi and Adams 2009. H. Shi and C. Adams. '**Rapid IC–ICP/MS Method for Simultaneous Analysis of Iodoacetic Acids, Bromoacetic Acids, Bromate, and Other Related Halogenated Compounds in Water**'. Talanta, 79 (2) (2009), pp. 523-527. *https://doi.org/10.1016/j.talanta.2009.04.037*.

Shin et al. 1996. H-S. Shin, Y-S. Oh-Shin, J-H. Kim, and J-K. Ryu. '**Trace Level Determination of Iodide, Iodine and Iodate by Gas Chromatography-Mass Spectrometry**'. Journal of Chromatography A, 732 (2) (1996), pp. 327-333. *https://doi.org/10.1016/0021-9673(95)01281-8*.

Simazaki et al. 2015. D. Simazaki, R. Kubota, T. Suzuki, M. Akiba, T. Nishimura, and S. Kunikane. '**Occurrence of Selected Pharmaceuticals at Drinking Water Purification Plants in Japan and Implications for Human Health**'. Water Research, 76 (2015), pp. 187-200. *https://doi.org/10.1016/j.watres.2015.02.059*.

Sjerps et al. 2017. R. M. A. Sjerps, T. L. ter Laak, and G. J. J. G. Zwolsman. '**Projected Impact of Climate Change and Chemical Emissions on the Water Quality of the European Rivers Rhine and Meuse: A Drinking Water Perspective**'. Science of The Total Environment, 601-602 (2017), pp. 1682-1694. *https://doi.org/10.1016/j.scitotenv.2017.05.250*.

Snyder and Fehn 2004. G. Snyder and U. Fehn. '**Global Distribution of ^{129}I in Rivers and Lakes: Implications for Iodine Cycling in Surface Reservoirs**'. Nuclear Instruments and Methods in Physics Research Section B: Beam

Interactions with Materials and Atoms, Proceedings of the Ninth International Conference on Accelerator Mass Spectrometry, 223-224 (2004), pp. 579-586. *https://doi.org/10.1016/j.nimb.2004.04.107*.

Snyder et al. 2005. S. A. Snyder, B. J. Vanderford, and D. J. Rexing. '**Trace Analysis of Bromate, Chlorate, Iodate, and Perchlorate in Natural and Bottled Waters**'. Environmental Science & Technology, 39 (12) (2005), pp. 4586-4593. *https://doi.org/10.1021/es047935q*.

Spolaor et al. 2013. A. Spolaor, P. Vallelonga, J. Gabrieli, N. Kehrwald, C. Turetta, G. Cozzi, L. Poto, J. M. C. Plane, C. Boutron, and C. Barbante. '**Speciation Analysis of Iodine and Bromine at Picogram-per-Gram Levels in Polar Ice**'. Analytical and Bioanalytical Chemistry, 405 (2-3) (2013), pp. 647-654. *https://doi.org/10.1007/s00216-012-5806-0*.

Suess et al. 2019. E. Suess, F. Aemisegger, J. E. Sonke, M. Sprenger, H. Wernli, and L. H. E. Winkel. '**Marine versus Continental Sources of Iodine and Selenium in Rainfall at Two European High-Altitude Locations**'. Environmental Science & Technology, 53 (4) (2019), pp. 1905-1917. *https://doi.org/10.1021/acs.est.8b05533*.

Suzuki et al. 1983. K. Suzuki, H. Aruga, H. Ishiwada, T. Oshima, H. Inoue, and T. Shirai. '**Determinatioii of Anions with a Potentiometric Detector for Ion Chromatography**'. Bunseki Kagaku, 32 (1983), pp. 585-590. *https://doi.org/10.2116/bunsekikagaku.32.10_585*.

Szidat et al. 2000. S. Szidat, A. Schmidt, J. Handl, D. Jakob, W. Botsch, R. Michel, H-A. Synal, C. Schnabel, M. Suter, J. M. López-Gutiérrez, and W. Städe. '**Iodine-129: Sample Preparation, Quality Control and Analyses of Pre-Nuclear Materials and of Natural Waters from Lower Saxony, Germany**'. Nuclear Instruments and Methods in Physics Research Section B: Beam Interactions with Materials and Atoms, 172 (1-4) (2000), pp. 699-710. *https://doi.org/10.1016/S0168-583X(00)00377-3*.

Tagami and Ushida 2005. K. Tagami, and S. Uchida. '**Sample Storage Conditions and Holding Times for the Determination of Total Iodine in Natural Water Samples by ICP-MS**', Atomic Spectroscopy, 26 (6) (2005), pp. 209-214.

Tagami and Ushida 2006. K. Tagami, and S. Uchida. '**Concentrations of Chlorine, Bromine and Iodine in Japanese Rivers**'. Chemosphere, 65 (11) (2006), pp. 2358-2365. *https://doi.org/10.1016/j.chemosphere.2006.04.077*.

Takaku et al. 1995. Y. Takaku, T. Shimamura, K. Masuda, and Y. Igarashi. '**Iodine Determination in Natural and Tap Water Using Inductively Coupled Plasma Mass Spectrometry**'. Analytical Sciences, 11 (5) (1995), pp. 823-827. *https://doi.org/10.2116/analsci.11.823*.

Takeda et al. 2016. A. Takeda, H. Tsukada, Y. Takaku, N. Satta, M. Baba, T. Shibata, H. Hasegawa, Y. Unno, and S. Hisamatsu. '**Determination of Iodide, Iodate and Total Iodine in Natural Water Samples by HPLC with Amperometric and Spectrophotometric Detection, and Off-Line UV Irradiation**'. Analytical Sciences, 32 (8) (2016), pp. 839-845. *https://doi.org/10.2116/analsci.32.839*.

Tang et al. 2013. Q. Tang, Q. Xu, F. Zhang, Y. Huang, J. Liu, X. Wang, Y. Yang, and X. Liu. '**Geochemistry of Iodine-Rich Groundwater in the Taiyuan Basin of Central Shanxi Province, North China**'. Journal of Geochemical Exploration, Arsenic, Fluoride and Iodine in Groundwater of China, 135 (2013), pp. 117-123. *https://doi.org/10.1016/j.gexplo.2012.08.019*.

Tatarczak-Michalewska et al. 2019. M. Tatarczak-Michalewska, J. Flieger, J. Kawka, W. Flieger, and E. Blicharska. '**HPLC-DAD Determination of Iodide in Mineral Waters on Phosphatidylcholine Column**'. Molecules, 24 (7) (2019), pp. 1243-1256. *https://doi.org/10.3390/molecules24071243*.

Ternes and Hirsch 2000. T. A. Ternes, and R. Hirsch. '**Occurrence and Behavior of X-Ray Contrast Media in Sewage Facilities and the Aquatic Environment**'. Environmental Science & Technology, 34 (13) (2000), pp. 2741-2748. *https://doi.org/10.1021/es991118m*.

Trenholm et al. 2006. R. A. Trenholm, B. J. Vanderford, J. C. Holady, D. J. Rexing, and S. A. Snyder. '**Broad Range Analysis of Endocrine Disruptors and Pharmaceuticals Using Gas Chromatography and Liquid Chromatography Tandem Mass Spectrometry**'. Chemosphere, 65 (11) (2006), pp. 1990-1998. *https://doi.org/10.1016/j.chemosphere.2006.07.004*.

Tucker and Flack 1998. H. L. Tucker and R. W. Flack. '**Determination of Iodide in Ground Water and Soil by Ion Chromatography**'. Journal of Chromatography A, 804 (162) (1998), pp. 131-135. *https://doi.org/10.1016/S0021-9673(98)00113-7*.

Tugulea et al. 2018. A-C. Tugulea, R. Aranda-Rodriguez, D. Bérubé, M. Giddings, F. Lemieux, J. Hnatiw, L. Dabeka, and F. Breton. '**The Influence of Precursors and Treatment Process on the Formation of Iodo-THMs in Canadian Drinking Water**'. Water Research, 130 (2018), pp. 215-223. *https://doi.org/10.1016/j.watres.2017.11.055*.

Ubom and Tsuchiya 1988. G. A. Ubom and Y. Tsuchiya. '**Determination of Iodide in Natural Water by Ion Chromatography**'. Water Research, 22 (11) (1988), pp. 1455-1458. *https://doi.org/10.1016/0043-1354(88)90104-2*.

Verma et al. 1992. K. K. Verma, A. Jain, and A. Verma. '**Determination of Iodide by High-Performance Liquid Chromatography after Precolumn Derivatization**'. Analytical Chemistry, 64 (13) (1992), pp. 1484-1489. *https://doi.org/10.1021/ac00037a030*.

Vlaamse Milieumaatschappij. 2017. '**Medicijnen in de waterketen. Resultaten van verkennend onderzoek in de periode 2014-2016**'.

Voutchkova et al. 2014a. D. D. Voutchkova, V. Ernstsen, B. Hansen, B. L. Sørensen, C. Zhang, and S. M. Kristiansen. '**Assessment of Spatial Variation in Drinking Water Iodine and Its Implications for Dietary Intake: A New Conceptual Model for Denmark**'. Science of The Total Environment, 493 (2014), pp. 432-444. *https://doi.org/10.1016/j.scitotenv.2014.06.008*.

Voutchkova et al. 2014b. D. D. Voutchkova, S. M. Kristiansen, B. Hansen, V. Ernstsen, B. L. Sørensen, and K. H. Esbensen. '**Iodine Concentrations in Danish Groundwater: Historical Data Assessment 1933-2011**'. Environmental Geochemistry and Health, 36 (6) (2014), pp. 1151-1164. *https://doi.org/10.1007/s10653-014-9625-4*.

Wagner and Plewa 2017. E. D. Wagner and M. J. Plewa. '**CHO Cell Cytotoxicity and Genotoxicity Analyses of Disinfection By-Products: An Updated Review**'. Journal of Environmental Sciences, water treatment and disinfection by-products, 58 (2017), pp. 64-76. *https://doi.org/10.1016/j.jes.2017.04.021*.

Wang and Jiang 2008. K-E. Wang and S-J. Jiang. '**Determination of Iodine and Bromine Compounds by Ion Chromatography/Dynamic Reaction Cell Inductively Coupled Plasma Mass Spectrometry**'. Analytical Sciences, 24 (4) (2008), pp. 509-514. *https://doi.org/10.2116/analsci.24.509*.

Wang et al. 2014. S. Wang, W. Zheng, X. Liu, P. Xue, S. Jiang, D. Lu, Q. Zhang, G. He, J. Pi, M. E. Andersen, H. Tan, and W. Qu. '**Iodoacetic Acid Activates Nrf2-Mediated Antioxidant Response *in Vitro* and *in Vivo***'. Environmental Science & Technology, 48 (22) (2014), pp. 13478-13488. *https://doi.org/10.1021/es502855x*.

Wang et al. 2015. T. Wang, W. Lin, X. Dai, L. Gao, B. Wang, and D. Quan. '**Determination of Iodate by HPLC-UV after On-Line Electrochemical Reduction to Iodide**'. Journal of Chromatographic Science, 53 (2) (2015), pp. 280-284. *https://doi.org/10.1093/chromsci/bmu053*.

Watts and Mitchell 2008. M. J. Watts and C. J. Mitchell. '**A Pilot Study on Iodine in Soils of Greater Kabul and Nangarhar Provinces of Afghanistan**'. Environmental Geochemistry and Health, 31 (4) (2008), pp. 503-509. *https://doi.org/10.1007/s10653-008-9202-9*.

Watts et al. 2010. M. J. Watts, J. O'Reilly, A. Maricelli, A. Coleman, E. L. Ander, and N. I. Ward. '**A Snapshot of Environmental Iodine and Selenium in La Pampa and San Juan Provinces of Argentina**'. Journal of Geochemical Exploration, 107 (2) (2010), pp. 87-93. *https://doi.org/10.1016/j.gexplo.2009.11.002*.

Wei et al. 2007. L. Wei, Y. Hong-Xia, L. Bing, C. Deng-Yun, and Z. Hui-Juan. '**Speciation Stabilities of Iodine in Underground Water by High Performance Liquid Chromatography-Inductively Coupled Plasma Mass Spectrometry**'. Chinese Journal of Analytical Chemistry, 35 (4) (2007), pp. 571-573. *https://doi.org/10.1016/S1872-2040(07)60047-4*.

Wei et al. 2013. X. Wei, S. Wang, W. Zheng, X. Wang, X. Liu, S. Jiang, J. Pi, Y. Zheng, G. He, and W. Qu. '**Drinking Water Disinfection Byproduct Iodoacetic Acid Induces Tumorigenic Transformation of NIH3T3 Cells**'. Environmental Science & Technology, 47 (11) (2013), pp. 5913-5920. *https://doi.org/10.1021/es304786b*.

Weinberg and Yamada 1998. H. S. Weinberg and H. Yamada. '**Post-Ion-Chromatography Derivatization for the Determination of Oxyhalides at Sub-PPB Levels in Drinking Water**'. Analytical Chemistry, 70 (1) (1998), pp. 1-6. *https://doi.org/10.1021/ac970651m*.

Wendel et al. 2016. F. M. Wendel, T. A. Ternes, S. D. Richardson, S. E. Duirk, J. A. Pals, E. D. Wagner, and M. J. Plewa. '**Comparative Toxicity of High-Molecular Weight Iopamidol Disinfection Byproducts**'. Environmental Science & Technology Letters, 3 (3) (2016), pp. 81-84. *https://doi.org/10.1021/acs.estlett.6b00037*.

Wolf et al. 2012. L. Wolf, C. Zwiener, and M. Zemann. '**Tracking Artificial Sweeteners and Pharmaceuticals Introduced into Urban Groundwater by Leaking Sewer Networks**'. Science of The Total Environment, 430 (2012), pp. 8-19. *https://doi.org/10.1016/j.scitotenv.2012.04.059*.

Xu et al. 2010. S-Q. Xu, Z-Q. Xie, W. Liu, H-X. Yang, and B. Li. '**Extraction and Determination of Total Bromine, Iodine, and Their Species in Atmospheric Aerosol**'. Chinese Journal of Analytical Chemistry, 38 (2) (2010), pp. 219-224. *https://doi.org/10.1016/S1872-2040(09)60026-8*.

Xu et al. 2017. Z. Xu, X. Li, X. Hu, and D. Yin. '**Distribution and Relevance of Iodinated X-Ray Contrast Media and Iodinated Trihalomethanes in an Aquatic Environment**'. Chemosphere, 184 (2017), pp. 253-260. *https://doi.org/10.1016/j.chemosphere.2017.05.048*.

Yamada et al. 1996. H. Yamada, T. Kiriyama, and K. Yonebayashi. '**Determination of Total Iodine in Soils by Inductively Coupled Plasma Mass Spectrometry**'. Soil Science and Plant Nutrition, 42 (4) (1996), pp. 859-866. *https://doi.org/10.1080/00380768.1996.10416633*.

Yamanaka et al. 1997. M. Yamanaka, T. Sakai, H. Kumagai, and Y. Inoue. '**Specific Determination of Bromate and Iodate in Ozonized Water by Ion Chromatography with Postcolumn Derivatization and Inductively-Coupled Plasma Mass Spectrometry**'. Journal of Chromatography A, 789 (1-2) (1997), pp. 259-265. *https://doi.org/10.1016/S0021-9673(97)00518-9*.

Yang et al. 2007. H. Yang, W. Liu, B. Li, H. Zhang, X. Liu, and D. Chen. '**Speciation Analysis for Iodine in Groundwater Using High Performance Liquid Chromatography-Inductively Coupled Plasma-Mass Spectrometry (HPLC-ICP-MS)**'. Geostandards and Geoanalytical Research, 31 (4) (2007), pp. 345-351. *https://doi.org/10.1111/j.1751-908X.2007.00132.x*.

Yang et al. 1998. X. Yang, P. Chen, W. Wei, and S. Yao. '**Determination of Iodide by Ion Chromatography with Series Bulk Acoustic Wave Detection**'. Microchemical Journal, 58 (1) (1998), pp. 58-66. *https://doi.org/10.1006/mchj.1997.1514*.

Yashin and Belyamova 1998. A. Y. Yashin and T. T. Belyamova. '**Simultaneous Determination of Sulfide, Iodide, and Rhodanide by Ion Chromatography with the Use of an Amperometric Detector**'. Journal of Analytical Chemistry, 53 (4) (1998), pp. 394-396.

Yoshida et al. 2007. S. Yoshida, Y. Muramatsu, S. Katou, and H. Sekimoto. '**Determination of the Chemical Forms of Iodine with IC-ICP-MS and Its Application to Environmental Samples**'. Journal of Radioanalytical and Nuclear Chemistry, 273 (1) (2007), pp. 211-214. *https://doi.org/10.1007/s10967-007-0738-4*.

Zemann et al. 2014. M. Zemann, L. Wolf, A. Pöschko, N. Schmidt, A. Sawarieh, N. Seder, A. Tiehm, H. Hötzl, and N. Goldscheider. '**Sources and Processes Affecting the Spatio-Temporal Distribution of Pharmaceuticals and X-Ray Contrast Media in the Water Resources of the Lower Jordan Valley, Jordan**'. Science of The Total Environment, 488-489 (2014), pp. 100-114. *https://doi.org/10.1016/j.scitotenv.2014.04.063*.

Zhang et al. 2013. E. Zhang, Y. Wang, Y. Qian, T. Ma, D. Zhang, H. Zhan, Z. Zhang, Y. Fei, and S. Wang. '**Iodine in Groundwater of the North China Plain: Spatial Patterns and Hydrogeochemical Processes of Enrichment**'. Journal of Geochemical Exploration, Arsenic, Fluoride and Iodine in Groundwater of China, 135 (2013), pp. 40-53. *https://doi.org/10.1016/j.gexplo.2012.11.016*.

Zhang et al. 2012. L. Zhang, L. Xu, Q. Zeng, S-H. Zhang, H. Xie, A-L. Liu, and W-Q. Lu. '**Comparison of DNA Damage in Human-Derived Hepatoma Line (HepG2) Exposed to the Fifteen Drinking Water Disinfection Byproducts Using the Single Cell Gel Electrophoresis Assay**'. Mutation Research/Genetic Toxicology and Environmental Mutagenesis, 741 (1-2) (2012), pp. 89-94. *https://doi.org/10.1016/j.mrgentox.2011.11.004*.

Zhang et al. 2010a. S-H. Zhang, D-Y. Miao, A-L. Liu, L. Zhang, W. Wei, H. Xie, and W-Q. Lu. '**Assessment of the Cytotoxicity and Genotoxicity of Haloacetic Acids Using Microplate-Based Cytotoxicity Test and CHO/*HGPRT* Gene Mutation Assay**'. Mutation Research/Genetic Toxicology and Environmental Mutagenesis, 703 (2) (2010), pp. 174-179. *https://doi.org/10.1016/j.mrgentox.2010.08.014*.

Zhang et al. 2010b. S. Zhang, K. A. Schwehr, Y-F. Ho, C. Xu, K. A. Roberts, D. I. Kaplan, R. Brinkmeyer, C. M. Yeager, and P. H. Santschi. '**A Novel Approach for the Simultaneous Determination of Iodide, Iodate and Organo-Iodide for ^{127}I and ^{129}I in Environmental Samples Using Gas Chromatography–Mass Spectrometry**'. Environmental Science & Technology, 44 (23) (2010), pp. 9042-9048. *https://doi.org/10.1021/es102047y*.

Zonja et al. 2015. B. Zonja, A. Delgado, S. Pérez, and D. Barceló. '**LC-HRMS Suspect Screening for Detection-Based Prioritization of Iodinated Contrast Media Photodegradates in Surface Waters**'. Environmental Science & Technology, 49 (6) (2015), pp. 3464-3472. *https://doi.org/10.1021/es505250q*.

II. Instrumentation, reagents and material

II. Instrumentation, reagents and material

1. Reagents and material

1.1. Glassware

All the glassware was cleaned in a 0.5% diluted DECON solution for several hours, rinced thoroughly with Milli-Q water (mQ), cleaned with 0.5% diluted HCl and rinsed thoroughly with mQ once more. Non-volumetric glassware was dried at 100 °C in an oven and all glassware openings were covered with aluminium foil. The 25 and 50 mL pyrex reagent bottles were additionally calcinated for 24 h at 450 °C. Additionally, all the glassware used for chlorine at some point (flasks for dilutions, tubes, reagent bottles), were thoroughly rinsed with a 50 mg L^{-1} chlorine solution for at least one hour.

2 mL GC amber glass vials with Teflon-lined screw caps were used for stock standard, and working standard storage. 25 mL Pyrex-reagent bottles were used for storage of other reagents as well as 2-bromopropionic acid. Hamilton syringes were used to transfer standards contained in methanol (MeOH). Scorex Accura micropipettes of 20 to 200 µL and 100 to 1000 µL, as well as a macropipette of 1 to 10 mL were used for the different dilutions of aqueous solutions.

Clear glass 20 mL HS vials were delivered by Perkin Elmer, with 20 mm crimp butyl/PTFE pharmafix standard caps obtained from Interchim. 20 mm, 22, 6 OD, 20 mL clear crimp headspace (HS) vials, together with combination seals of aluminium cap, plain with centre hole, Pharma-Fix-Septum, 47° shore, 1, 3.0 mm were delivered by Perkin Elmer. All method development and validation analyses were performed with new HS vials which were discarded after a single use.

The 20 mm crimper and decapper used for HS vial sealing and opening were delivered by GRACE Alltech. A vial holder rack was used for both HS vial sample preparation and upright sealing.

1.2. Standards and reagent provenance

4 different classes of DBPs were studied and are presented with some of their main characteristics in Table II-1.

Table II-1: Analytical figures of merit for the DBPs analysed.

Compound	Formula	Acronym	MW (g mol^{-1})	Solubility (g L^{-1})	Boiling point (°C)	log Kow	pK_a
Trihalomethanes							
THM4							
Trichloromethane (chloroform)	$CHCl_3$	TCM	119.4	8.0	61	1.97	
Bromodichloromethane	$CHBrCl_2$	BDCM	163.8	4.0	90	2.0	
Dibromochloromethane	$CHBr_2Cl$	DBCM	208.3	2.7	120	2.16	
Tribromomethane (bromoform)	$CHBr_3$	TBM	252.7	3.1	149	2.4	
I-THMs							
Dichloroiodomethane	$CHCl_2I$	DCIM	210.8	0.717 (a)	132	2.03 (a)	
Bromochloroiodomethane	$CHBrClI$	BCIM	255.3	0.346 (a)	157 (b)	2.11 (a)	
Dibromoiodomethane	$CHBr_2I$	DBIM	299.7	0.162 (a)	186 (b)	2.20 (a)	
Chlorodiiodomethane	$CHClI_2$	CDIM	302.3	0.082 (a)	191 (b)	2.53 (a)	
Bromodiiodomethane	$CHBrI_2$	BDIM	346.7	0.038 (a)	222 (b)	2.62 (a)	
Triiodomethane (iodoform)	CHI_3	TIM	393.7	0.100	218	3.03 (a)	
Haloacetic acids							
HAA5							
Chloroacetic acid	$ClCH_2COOH$	MCAA	94.5	miscible	189	0.22	2.87
Bromoacetic acid	$BrCH_2COOH$	MBAA	138.9		208	0.41	2.89
Dichloroacetic acid	$Cl_2CHCOOH$	DCAA	128.9		194	0.92	1.26
Dibromoacetic acid	$Br_2CHCOOH$	DBAA	217.8		232-234	0.70	1.48
Trichloroacetic acid	$Cl_3CHCOOH$	TCAA	163.4		195 -198	1.33	0.51
Other HAAs							
Bromochloroacetic acid	$BrClCHCOOH$	BCAA	173.4	miscible	215	0.61	1.4
Bromodichloroacetic acid	$BrCl_2CHCOOH$	BDCAA	207.8		-	1.53 (est)	0.03
Dibromochloroacetic acid	$Br_2ClCHCOOH$	DBCAA	252.3		-	1.62 (est)	0.03
Tribromoacetic acid	$Br_3CHCOOH$	TBAA	296.7		245	1.71 (est)	0.72
Iodoacetic acid	ICH_2COOH	(M)IAA	185.9		decomposes	0.85	3.18
Haloacetamides							
Chloroacetamide	$ClCH_2CONH_2$	CAcAm	93.5	miscible	225	-0.53	
Bromoacetamide	$BrCH_2CONH_2$	BAcAm	138.0		-	-	
Dichloroacetamide	$Cl_2CHCONH_2$	DCAcAm	128.0		234	-	
Trichloroacetamide	Cl_3CCONH_2	TCAcAm	162.4		240	-	
Haloacetonitriles							
Chloroacetonitrile	$ClCH_2CN$	(M)CAN	75.5	miscible	126	0.45	
Bromoacetonitrile	$BrCH_2CN$	(M)BAN	120.0		150	0.20 (est)	
Iodoacetonitrile	ICH_2CN	M)IAN	167.0		185	-	

References: Pubchem Open database
(a) Environ. Sci. Technol. 2018 52(22): 13047-13056. doi: 10.1021/acs.est.8b04625.
(b) References: ChemSrc Open database
(est): estimated

200 and 2000 µg mL^{-1} THM4 mix standards in MeOH were purchased from AccuStandard. 1,2-dibromopropane was purchased from Interchim. 100 to 250 mg pure standards for 5 of the 6 ITHMs were purchased from Accustandard (resold by Interchim). Bromochloroiodomethane was provided by Dr. S. Allard (Curtin University, Australia). HANs were bought from Combi-Blocks resold by Interchim and HAcAms were purchased from Tokyo Chemical Industry Europe via Interchim or Combi-Blocks. MIAA was purchased from AccuStandard. The 2000 µg mL^{-1} HAA9 mix in methyl-tert-butyl ether as well as the derivatisation reagent, dimethylsulfate (DMS) were obtained from Sigma-Aldrich, France. The ion-pairing agent, tetrabutylammonium hydrogen sulfate (TBA-HSO_4) was purchased from Merk, and 2-bromopropionic acid from AccuStandard.

Anhydrous sodium sulfate was purchased from Sigma-Aldrich.

1.2.1. Standard and reagent preparation

Standards and blanks were prepared from ultrapure water (mQ) at 18 mΩ using an ultrapure water system (Merk).

1.2.1.1. Haloacetic acids

A stock standard solution containing 1 g L^{-1} MIAA and intermediate concentration standard solutions (160 mg L^{-1}) containing all 10 HAAs were prepared in MeOH. Further dilutions for working standards were prepared daily or weekly in mQ water. The internal standard (2-bromopropionic acid) was prepared in MeOH at a concentration of 3.2 g L^{-1}. Further dilutions were done in mQ water. The working solution was prepared in mQ water at a concentration of 320 µg mL^{-1}. A 0.5 M TBA-HSO_4 solution was prepared in mQ water.

1.2.1.2. Trihalomethanes

Stock standard solutions containing each iodinated THM at a concentration close to 1 g L^{-1} were prepared in MeOH. An intermediate mix solution containing all 6 I-THM at 40 mg L^{-1} was then prepared in MeOH. Intermediate concentration THM4 standard solutions (16 and 160 mg L^{-1}) were prepared with the I-THM mix solution (1/10 or 1/100 dilution factors) in MeOH. Working standard solutions were prepared daily or weekly in mQ water.

The internal standard solution (1,2-dibromopropane) was prepared in MeOH at a concentration of 320 mg L^{-1} and the working solution in mQ water at a concentration of 6.4 mg L^{-1}.

The sodium sulfate (Na_2SO_4) was heated in a muffle furnace at 450 °C for up to 4 hours to remove phthalates and other potentially interfering organic substances.

1.2.2. Conservation

All stock standard solutions were stored frozen at -20 °C. Working standards were left to reach room temperature (around 20 °C) before use to prevent overspiking. Working standards and reagents were kept in the fridge at 4 (±3) °C for 24 and 168 hours respectively. All volatile DBP standards or methanol-containing standards as well as the 1,2-dibromopropane were kept in single use small vials before discard.

Volatiles can be lost to headspace very easily every time a standard is opened to the atmosphere (Kolb and Ettre 2006). To avoid this, all vials containing MeOH and /or volatile DBPs were filled to the top without any headspace. While preparing standards and HS samples, care was taken not to shake open vials nor leave them open for too long.

1.3. Sample preparation

THMs were analysed by placing 10 mL of water samples containing 1,2-dibromopropane as internal standard (6.4 µg L^{-1}) (spiked as a small 100 µL aliquot) in a HS glass vial containing 4 g (2.8 M) of sodium sulfate. Generally the calibration range went from 60 ng L^{-1} to 60 µg L^{-1} for the THM4, and 100 times lower for the I-THMs. The vials were immediately sealed and stirred in a vortex mixer for at least 1 min or until complete salt dissolution if some salt crystals were still present.

For HAAs, the method development is detailed in Chap. III. Briefly, 9.84 mL of water samples spiked with bromopropionic acid (internal standard) were placed in a HS glass vial containing 4 g (2.8 M) of sodium sulfate. Then, 100 µL of a 0.5 M concentration of an ion-pairing agent (tetrabutylammonium hydrogensulfate, 2.3 µmol as aqueous solution) and 60 µL of derivatization reagent (dimethylsulfate) were added. Generally the calibration range went from 5 ng L^{-1} to 40 µg L^{-1}. The vials were immediately sealed and stirred in a vortex mixer for at least a 1 min or until complete salt dissolution if some salt crystals were still present, before being placed unto the carrousel.

2. Headspace-trap-gas chromatography – mass spectrometry for disinfection by-product analysis

2.1. Instrument

2.1.1. General configuration

The analytical instrument used was a Trace™ 1300 GC coupled with an ISQ-LT Single Quadrupole Mass Spectrometer (Thermo Scientific®) operating in Full Scan and then in Single Ion Monitoring (SIM) mode. A ZB-5MS (Phenomenex®) column (30 m × 0.25 mm ID, 1 µm film thickness) was used for gas chromatographic separations. The GC-MS was directly connected to a Turbomatrix Headspace 40 Trap (Perkin Elmer® Turbomatrix Air Monitoring Trap M0413628), bypassing the inlet split of the GC injector. By installing the fused silica tubing in this way, the GC carrier gas is supplied by the HS and the incoming HS sample is not split or diluted in any way.

2.1.2. Headspace-trap principle

The principle of headspace - trap is similar to classical static headspace, but after vial pressurisation, the vapour is fully vented through an adsorbent trap (Tipler 2013b). The analytes adsorbed to the trap are desorbed by rapid heating of the trap to high temperatures while passing the carrier gas flow through the trap to the column (Barani et al. 2006). In this way, the amount of sample vapour entering the GC column can be increased by a factor of up to 100 times (Tipler 2013b). The other advantages of such a trap is that it separates the volatile analytes of interest from the excess of the diluted headspace gas (Kolb and Ettre 2006), while purging water, nitrogen and oxygen from the trap, thus significantly reducing the amount of these compounds from entering the GC-MS instruments (Marotta 2017). Schulz et al. (2007) used an HS trap method to enrich and focus volatile constituents from spirits prior to separation and found detection limits and extraction yields 35-55 times lower than those observed with direct static headspace.

2.2. Turbomatrix HS 40 Trap instrument description

The Turbomatrix HS 40 Trap instrument (coupled to a GC-MS) for the analysis of volatile compounds is an autosampler for up to 40 vials that can be used to determine volatile organic compounds (VOCs) present in several matrices (Barani et al. 2006).

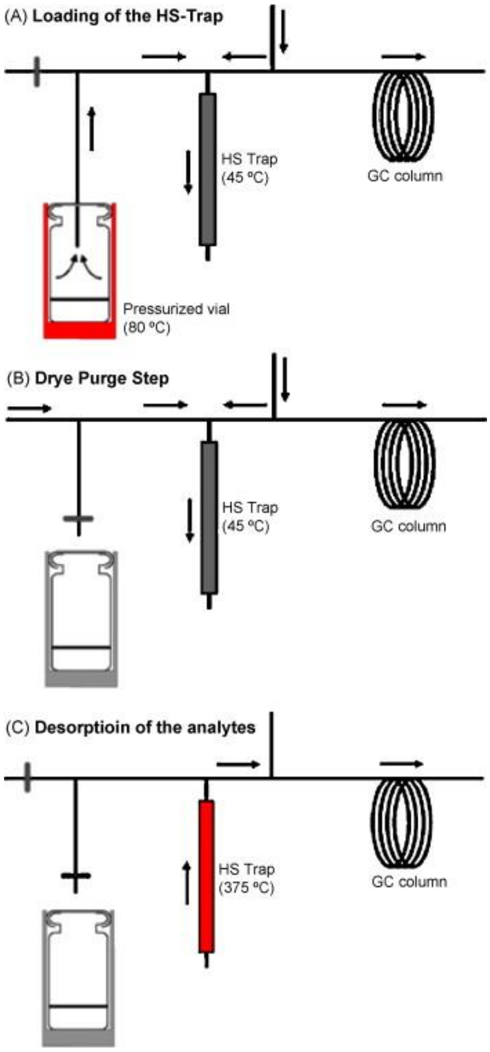

Figure II-1: Principle of sample preparation with headspace (HS) trap. The loading is accomplished by pressurizing the sample vials and allowing the pressure to decay through the cooled adsorbent trap (A). A drying step removes moisture from the sample (B). After thermal desorption, the analytes are transported by the carrier gas into the GC column for separation (C) (from Schulz et al. 2007).

The analysis sequence is detailed in the following steps. Throughout all of the steps before desorption/injection, the transfer line and analytical column are pneumatically isolated (Figure II-1), constantly supplied with carrier gas avoiding any column pressure change ('TurboMatrix Headspace Sampler and HS 40/110 Trap User's Guide' 2008).

2.2.1. Equilibration

The vial is heated in an oven at 60 °C for a 10 min (THMs) or 20 min period (HAAs) (Table II-2), defined by the sample characteristics in order to reach equilibrium conditions between the sample and its vapours in the headspace (Barani et al. 2006).

2.2.2. Pressurization

After the equilibration phase, 1 min vial pressurization is done by a needle which pierces the septum and allows the carrier gas at a pre-set pressure to enter the vial to set the internal pressure to 40 psi (Barani et al. 2006).

2.2.3. Trap load

After vial pressurization, the pressurized headspace vapour inside the vial is allowed to vent for 1.3 min – which is sufficient to allow the pressure in the vial to decay to its lowest value – through an adsorbent trap which retains the analytes (Figure II-1A). An isolating flow of carrier gas keeps the headspace vapour out of the GC column during this step. This pressurization/loading process may be repeated up to 4 times to vent up to 99% of the headspace vapour through the adsorbent trap (Tipler 2013a). This option could be valuable for enhancing the sensitivity in trace analyses. However, increasing the number of vial extractions leads to increasing amounts of water adsorbed on the trap, thus requiring more extensive drying prior to trap desorption (Røen et al. 2010) and thus only 1 cycle was chosen in our method.

2.2.4. Trap dry-purge

After loading the trap, the trap is then purged with carrier gas (Figure II-1B) for 4 min to eliminate water to protect the capillary column and lower the baseline (Barani et al. 2006).

2.2.5. Trap desorb and trap hold

After dry purge completion, the trap temperature is rapidly increased to the desired high value (‘trap desorption temperature’, Table II-2) to desorb the trapped analytes. The temperature is then kept at that value for a specified amount of time to clean the trap and avoid any possible carry-over. As soon as the trap is heated, the column isolation is turned off, the flow of carrier gas is reversed, transporting the analytes to the GC column and the GC run begins (Barani et al. 2006).

Table II-2: Headspace trap parameters.

Parameters	THMs	HAAs
Sample volume	10 mL	10 mL
thermostatting	10 min, 60 °C	20 min, 60 °C
needle temperature	90 °C	90 °C
transfer line temperature	100 °C	100 °C
trap load temperature	42 °C	42 °C
trap desorption temperature	220 °C	200 °C
number of cycles	1	1
pressurization time	1 min	1 min
decay time	1.3 min	1.3 min
Dry purge time	4 min	4 min
desorption time	0.5 min	0.5 min
trap hold	13 min	10 min
column pressure	15.6 psi	15.6 psi
vial pressure	40 psi	40 psi
desorption pressure	15.6 psi	15.6psi

2.3. Headspace-trap GC-MS methodology

2.3.1. GC-MS conditions

Initially the MS was operated in scan mode with an m/z scan ranging from 46 to 400. Once the standard peaks were identified along with their retention times, the MS was operated in SIM mode with 29 ms dwell times.

Table II-3: GC conditions.

Parameter	Conditions
Column	HP-5MS column (30 µm × 250 µm I.D. × 0.25 µm)
Inlet conditions	Splitless, 200 °C. Column flow rate: 1.0 mL min^{-1}
GC temperature program	40 °C for 3 min, 20 °C min^{-1} to 60 °C, hold 3 min. 5°C min^{-1} to 100 °C, and 25 °C min^{-1} to 200 °C, hold 5 min.
Ion source temperature	250 °C
MS transfer line temperature	290 °C
Full scan window	Start at 1.5 min, from m/z 46 to 400

2.3.2. Data acquisition and processing

2.3.2.1. Order of analysis

Before each batch of samples is run, the trap is cleaned at 300 °C for 30 minutes. Then a matrix blank to test for contamination or interference is run followed by the external standards, more blanks to check for carry-over, the samples, and finally a medium concentrated standard is repeated after the samples (or if there are many samples, after approximately 10 samples). The calibration standards should not deviate from the calibration curve by more than 20 percent. Acceptable recoveries are 80 to 120 percent of the true value for the repeated standard(s).

2.3.2.2. Internal standard count

The ratio between the peak area corresponding to the quantification fragment and the peak area of the internal standard 1,2-dibromopropane were used for quantification.

In order to check for HS vial leaks due to poor crimping of the vial caps, the area count of the internal standard was monitored. Samples with low IS area counts where leaks are suspected were thus discarded.

2.3.2.3. Software

The HS-trap system was controlled by an internal graphical user interface, while the GC and MS were controlled by the Xcalibur Quan software (ThermoFisher). The Xcalibur Quan software (ThermoFisher) is used for method development and running the analysis while TraceFinder (ThermoFisher) is used for peak integration, calibration, and quantification.

Excel and SigmaPlot softwares were used for establishing graphs and correlations.

3. Sample analysis

3.1. NOM characterization

3.1.1. DOC analysis

Glass tubes with 30 mL of directly filtered samples (with 0.45 µm PES filters) and 150 µL of sodium azide (6.5 g L^{-1}) were collected for TOC and chromatographic analysis. The concentration of dissolved organic carbon (DOC) was determined using a TOC-meter-Shimadzu analyser (TOC-V_{SCH} combustion catalytic oxidation) after filtration through a 0.45 µm PES filter.

3.1.2. Natural organic matter

NOM size fractionation was performed using high performance liquid chromatography (HPLC) (Agilent® 1260 Infinity II) equipped with a size exclusion column (Agilent® Bio SEC-5 column, 100 Å; 300 mm x 7.8 mm, 5 µm particle size), with an exclusion limit range of 100 to approximatively 100,000 Da. A phosphate buffer eluent was used: 1 g L^{-1}; pH 6.8 and ionic strength of 0.039 M (Zhou et al. 2000; Song et al. 2010). Other parameters used during HPLC are an injection volume of 100 µL, a flow rate of 1.0 mL min^{-1}, a pressure of around 80 bar and a temperature of 23 °C. The HPSEC is coupled to both a diode array detector equipped with a 60 mm high sensitivity cell. The eluting dissolved organic matter fractions were detected using UV absorbance at 254 nm. Samples with organic carbon concentrations exceeding 5 mgC L^{-1} were diluted before analysis to avoid fouling of the column and signal saturation.

Some of the samples were characterized by size exclusion chromatography coupled to organic carbon detection (LC-OCD) at the certified laboratory "Het Waterlaboratorium (HWL)" (Haarlem, Noord-Holland) according to the LC-OCD procedure, described by Huber et al. (2011).

3.1.3. Iodine and bromine speciation

3.1.3.1. Inorganic species

Iodide, iodate, bromide and bromate were measured using a liquid chromatograph coupled to an inductively coupled plasma mass spectrometer (IC-ICP-MS).

Separation was done by a Dionex IonPac AS16 (2 x 250 mm) analytical column with an AG16 (2 x 50 mm) guard column using NaOH 20 mM gradient eluent (Table II-3). ICP-MS (Agilent 7900) conditions were set at an RF Power of 1550W, no gas mode (collision cell), with an aerosol dilution (0.1 L min^{-1}), and bromine and iodine were determined through their ^{79}Br and ^{127}I isotopes, respectively. The quantification limit was 0.2 µg L^{-1} for all 4 species. Calibration curves were prepared in mQ water from 1 g L^{-1} liquid standards of iodide, bromide, bromate and solid KIO_3. 6-point calibrations with concentrations from 5 to 200 µg L^{-1} were used for bromide and from 0.5 to 20 µg L^{-1} for the three other inorganic compounds.

Table II-4: HPLC conditions for iodine and bromine speciation by HPLC-ICP-MS.

Injection volume (µL)	25
Elution flow rate (µL min^{-1})	400
Elution gradient	10 mM NaOH held for 3 min, increased to 20 mM in 1 min, held at 20 mM for 8 min, decreased back to 10 mM in 30 s, held for another 5 minutes for equilibration (total run time: 17.50 min).
Targets retention times	IO_3^-: 2 min 50 s; Br^-: 5 min 20 s; I^-: 12 min 10 s

3.1.3.2. Total iodine and bromine

Total iodine and bromine in the water samples were determined by ICP-MS (Varian 820 MS) in 0.5% tetramethylammonium hydroxide matrixes with ^{129}Xe as internal standard (Balaram et al. 2012). Analyses were performed using He at 100 mL min^{-1} at the collision reaction interface (skimmer cone) to handle potential polyatomic interferences. The quantification limit level was 0.5 µg L^{-1} for both iodine and bromine. Iodine and bromine standards were prepared from liquid standards of 1 g L^{-1} bromide and iodide respectively. The standard calibration curve went from 0.5 to 200 µg L^{-1} for both bromine and iodine.

3.1.3.3. Total organic bromine and iodine

The total organic bromine and iodine have been assessed by:

TOBr = [total bromine] – [Br^-] – [BrO_3^-]

TOI = [total iodine] – [I^-] – [IO_3^-]

Bromate was never detected in any drinking water samples (<0.2 µg L^{-1}).

3.2. Trihalomethane formation potential tests

The THMFP test was carried out using a 7-day chlorine test procedure according to the Standard Methods 5710B (Standard method, 2017) with minor modifications.

3.2.1. Reagents

A phosphate buffer was prepared with 68.1 g KH_2PO_4 and 11.7 g NaOH in 1 L water, in order to get a pH of 7 after a 1/50 dilution in the target sample.

A standardised stock hypochlorite solution (10-15% NaOCl) was used (§ 3.3.1) to prepare a stock 5000 mg L^{-1} sodium hypochlorite in mQ water.

3.2.2. 4-hour test

735 µL phosphate buffer (1/50 dilution to obtain a pH of 7.0 ± 0.2) and 735 µL stock sodium hypochlorite solution were added into a 25 mL vial, filled completely (36.50 mL actual volume) with the sample and tightly closed with a PTFE-lined screw cap and buffered by a phosphate solution and chlorinated. They were then stored in the dark for 4 h at 25 °C.

The free chlorine residual was then measured using the N-diethyl-p-phenylenediamine (DPD) colorimetric method, and the chlorine demand (D_{Cl}) in mgCl_2 L^{-1} was calculated as the difference between the initial concentration (100 mgCl_2 L^{-1}) and the residual chlorine concentration.

3.2.3. 7-day test

The volume (V_D) of stock 5000 mg L^{-1} sodium hypochlorite required for the 7-day test was then calculated according to the following formula:

$$V_D = \frac{Dcl+3}{5} \times \frac{Vs}{100}$$

With V_s, the sample volume to be analysed in mL.

From the chlorine consumed during the 4-hour test, this standard formula gives an estimation of the excess of chlorine to add in order to reach 3 to 5 mg L^{-1} residual free chlorine in the sample after 1 week. However, as this is an estimation, to increase the likelihood of achieving the desired chlorine residual concentration (3 to 5 mg L^{-1}) at the end of the 7-day reaction period, several sample portions were dosed to provide a range of chlorine concentrations, with each initial chlorine dose differing in increments of 2 $mgCl_2$ L^{-1}.

A reagent blank was also prepared for quality control of the reagent solutions by adding 1 mL chlorine dosing solution to 50 mL phosphate buffer, mixing and completely filling a 25-mL vial, immediately closed with a PTFE-lined screw cap. The samples and the blank were then incubated at 25 ± 2 °C for 7 days, after which the free chlorine residual of each sample were immediately measured once more.

3.2.4. Sample quenching

After the residual free chlorine measurements, the samples with chlorine residuals between 3 and 5 mg L^{-1} at the end of the chlorination test were immediately quenched using 1 mL sodium thiosulfate (0.45 mg L^{-1}) and the samples were stored at 4 °C in the dark until the DBPs were analysed by HS-trap-GC-MS.

145 µL of the quenching agent was also added to another reagent bottle with 735 µL of the reacted reagent blank and immediately filled with mQ water and closed. After mixing, a portion of this reagent blank was measured for THMs. The sum of all THMs in the reagent blank should be less than 5 µg L^{-1} TTHM as $\mu gCHCl_3$ L^{-1}.

3.3. Chlorine analysis

3.3.1. UV for stock solution

Either the standardized stock hypochlorite solution (10-15% NaOCl) or the stock 5000 mg L^{-1} sodium hypochlorite solution were diluted by respectively 1/1000 or 1/50 to reach approximately 100 mg L^{-1}. The absorbance of the solution was then measured with a Carry 100 spectrophotometer at 292 nm. Two high precision spectrophotometer cells made of Quartz SUPRASIL 10 mm Light Path were used.

The cells were cleaned with a Hellemax solution and rinsed with mQ water before use. At 292 nm, the molar absorption coefficient of OCl^- is 357 M^{-1} cm^{-1} (Feng et al. 2007).

3.3.2. N-diethyl-p-phenylenediamine (DPD) colorimetric method for low chlorine concentrations

3.3.2.1. Reagents

Another phosphate buffer was prepared, with 24 g anhydrous disodium hydrogen phosphate (Na_2HPO_4), 46 g anhydrous potassium dihydrogen phosphate (KH_2PO_4) and 800 mg disodium ethylenediaminetetraacetate dehydrate (EDTA) in 1 L mQ water.

The DPD indicator solution was prepared with 6 mL of concentrated sulfuric acid solution, 200 mg EDTA (to improve stability by retarding oxidation of the solution) and 1 g DPD oxalate in a brown-glass bottle.

3.3.2.2. Calibration and free chlorine measurements

The 5 g L^{-1} concentrated solution was diluted successively to 100 and then 10 $mgCl_2$ L^{-1} to prepare a 5-point calibration curve with 0.2, 0.4, 0.6, 0.8 and 1.0 $mgCl_2$ L^{-1} before each measurement campaign. In a tube, 9 mL of mQ water were added, followed by 0.5 mL of phosphate buffer, 0.5 mL of DPD solution and 1 mL of sample (which amounts to a 1/10 dilution of the sample). The absorbance of the calibration standards and the samples were measured at 515 nm.

4. Water matrix

4.1. De Blankaart drinking water treatment plant (Belgium)

The water samples used in this study were sourced from a drinking water treatment plant, located in the West of Flanders in Belgium and designed for a production of 40,000 m^3 day^{-1}. The raw water is drawn from the river IJzer (approximately 80%) and from surrounding polders (approximately 20%) (De Watergroep 2015). The water is characterized by a high NOM content, with an average DOC content of 13 mgC L^{-1} (Schoutteten 2018), combined with a high alkalinity and hardness. During the summer months, algae blooms occur in the 3 million m^3 reservoir for the raw water storage due to a high phosphate concentration (Verdickt et al. 2012). The Belgian part of the hydrographic basin of the IJzer river covers an area holding 110,000 inhabitants and is largely agrarian: intensive farming and intensive cattle breeding (Baert et al. 1996).

The water is treated by a conventional treatment train consisting of biological ammonia oxidation, enhanced coagulation with ferric chloride, decantation, rapid sand filtration, granular activated carbon (GAC) filtration and disinfection with NaOCl (Figure II-2). A pH adjustment is performed before and after coagulation by sulfuric acid addition and sodium hydroxide respectively. An intermediate chlorination is performed before sand filtration in order to enhance manganese oxidation, which in its turn is removed during sand filtration.

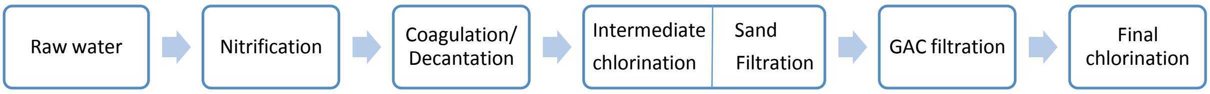

Figure II-2: Flow chart of the water treatment process at De Blankaart.

Due to the combination of high DOC concentration and relatively high alkalinity, the water requires large amounts of flocculant and sulfuric acid during coagulation (Verdickt et al. 2012). De Watergroep has been studying the possibility of replacing the enhanced coagulation and decantation treatment step by a combination of ion exchange (IEX), coagulation and flotation. A fluidized ion exchange pilot plant has been in operation since the end of 2015 on a scale of 50 m^3 h^{-1}, with resin PPA860S (Verdickt and Schoutteten 2018).

4.2. Ion exchange column test samples

A small column experiment was performed as it is easier than using a jar-test to translate the results to the full scale. The four resins tested were Purolite PPA860S (France), Amberlite IRA410, Amberlite IRA958 and Dowex TAN-1 from Inaqua (Germany). Three of the resins are strong base type I resins with a quaternary ammonium group, and one is a strong base type II resin where a methyl group is replaced with an alcohol group.

Their main properties are presented in Table II-5.

Table II-5: Properties of the four anion exchange resins selected.

Trade Name	**Resin Type**	**Matrix/ Structure**	**Functional Group**	**Mean Particle Size (µm)**	**Water Content (%)**	**Total Exchange Capacity (eq.L^{-1})**	**Manufacturer**
TAN-1	MP	Styrene-DVB	Quaternary amine	420-1200	70-82	0.7	Dowex
IRA410	Gel	Styrene-DVB	Dimethyl ethanol ammonium	600-750	45-51	≥1.25	Rohm and Haas
IRA958	MP	Crosslinked acrylic	Quaternary ammonium	630-850	66-72	≥0.80	Rohm and Haas
PPA860S	MP	Polyacrylic crosslinked with DVB	Quaternary ammonium	740	66-72	0.8	Purolite

DVB: divinylbenzene; MP: macroporous; References: https://www.lenntech.com/Data-sheets

The experiment took place in October 2018. Four parallel transparent PVC columns, with an internal diameter of 8.64 cm, were used for performing breakthrough experiments. The resins were pretreated extensively to avoid possible leaching of residual monomers, and to avoid irreversible adsorption of NOM occurring on virgin resins. The procedure consisted of a jar pretreatment and a dummy loading run in the columns. For the jar pretreatment, the following regeneration procedure was conducted three times consecutively: regeneration using 3 Bed Volumes (BV) of a 10% NaCl brine followed by decantation, and a triple rinsing with 3 BV of tap water followed by decantation.

After the jar pretreatment procedure, the resins were transferred inside the columns, and a dummy loading run was performed using 500 BV of filtered raw water at an empty bed contact time (EBCT) of 1.73 min, followed by a regeneration with 1 BV of 10% NaCl at an EBCT of 30 minutes and a rinse with 1 BV of tap water at an EBCT of 30 minutes. The latter regeneration procedure coincides with practices in IEX pilot scale research conducted at De Watergroep (Verdickt et al. 2012). After this extensive pretreatment procedure, the resins were considered to be suitable to perform a representative IEX cycle.

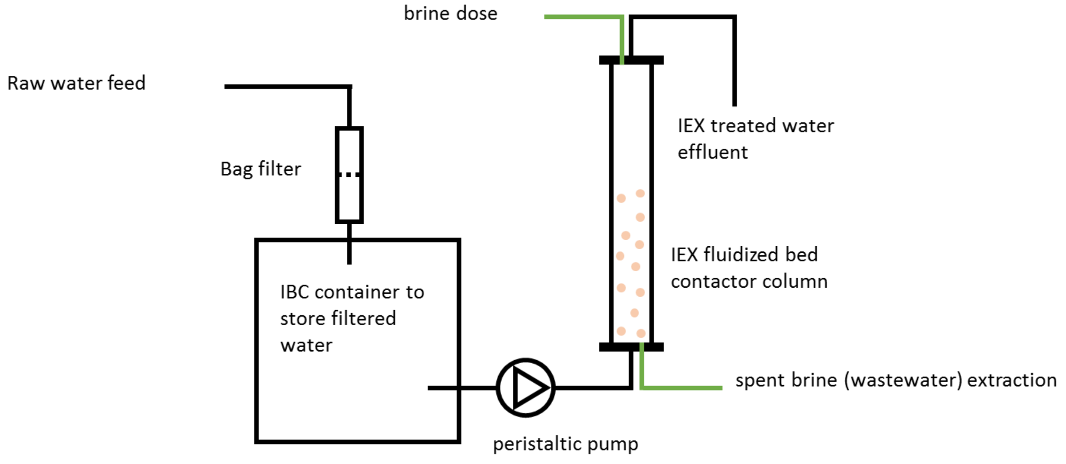

Figure II-3: IEX column test scheme. The lines in green are the regeneration lines.

The breakthrough experiments were run in fluidized bed modus (i.e. bottom – up, see Figure II-3) using raw water from the De Blankaart reservoir, which had been prefiltered over a 25 µm bag filter (Filtration CVBA, Drongen, Belgium), at an EBCT of 1.73 min. Samples were taken at 0 and every recurring 50 BV, until 500 BV of raw water were treated. To simulate the average treated water quality which would be obtained after 500 BV, a mixed effluent sample was created by mixing equal volumes (208 mL) of each of the 11 taken samples for each resin.

4.3. Water sampling

Samples were collected from the full scale or the pilot plant. Amber glass bottles of 1 L or 2.5 L were used for the collection of samples for chlorination tests and DBP analysis. A quenching agent (ascorbic acid or sodium thiosulfate) was added in excess to the samples which already contained chlorine. Samples were filtered through 0.45 µm Polyethersulfone (PES) filters within a few hours after sampling. Glass tubes with 30 mL of directly filtered (0.45 µm PES filters) samples and 150 µL of sodium azide (6.5 g L^{-1}) were collected for TOC and HPSEC analysis. Finally, samples were stored at 4 °C until the chlorination experiments or analysis.

5. Bibliography

Baert et al. 1996. R. Baert, R. Loontiens, and M. Devos. '**Potassium and Nitrate Pollution of Surface Water in the Catchment Area of the "Blankaert" Water Production Centre in Flanders (Belgium)**'. In *Proceedings _ 4th International Conference On Environmental Compliance And Enforcement*, 2 (1996), pp. 635-642. Chiang Mai, Thailand.

Balaram et al. 2012. V. Balaram, M. Satyanarananan, D.V. Avdeev, N. Berdnikov, P. Roy, S. S. Sawant, K. S. V. Subramanyam, K. V. Anjaiah, C. T. Kamala, R. Mathur, and B. Dasaram. '**Use of Xenon as Internal Standard for the Accurate Determination of Trace Elements in Water Samples by ICP-MS**'. Atomic Spectroscopy, 33 (2012), pp. 41-47.

Barani et al. 2006. F. Barani, N. Dell'Amico, L. Griffone, M. Santoro, and C. Tarabella. '**Determination of Volatile Organic Compounds by Headspace Trap**'. Journal of Chromatographic Science, 44 (10) (2006), pp. 625-630. *https://doi.org/10.1093/chromsci/44.10.625*.

De Watergroep. 2015. 'Annual Report'. Activity report De Watergroep 2015.

Feng et al. 2007. Y. Feng, D. W. Smith, and J. R. Bolton. '**Photolysis of Aqueous Free Chlorine Species (HOCl and OCl^{-}) with 254 nm Ultraviolet Light**'. Journal of Environmental Engineering and Science, 6 (8) (2007), pp. 277-284. *https://doi.org/10.1139/s06-052*.

Huber et al. 2011. S. A. Huber, A. Balz, M. Abert, and W. Pronk. '**Characterisation of Aquatic Humic and Non-Humic Matter with Size-Exclusion Chromatography – Organic Carbon Detection – Organic Nitrogen Detection (LC-OCD-OND)**'. Water Research, 45 (2) (2011), pp. 879-885. *https://doi.org/10.1016/j.watres.2010.09.023*.

Kolb and Ettre 2006. B. Kolb and L. S. Ettre. *Static Headspace–Gas Chromatography: Theory and Practice*. John Wiley & Sons, Inc (2006).

Marotta 2017. L. Marotta. '**The Benefit of Headspace Trap Coupled with GC–MS for the Determination of Trace Levels of BTEX Compounds in Drinking Water**'. The Column, 13 (6) (2017), pp. 16-19. *https://doi.org/10.1016/j.trac.2011.08.011*.

Røen et al. 2010. B. T. Røen, E. Unneberg, J. Aa. Tørnes, and E. Lundanes. '**Headspace-Trap Gas Chromatography–Mass Spectrometry for Determination of Sulphur Mustard and Related Compounds in Soil**'. Journal of Chromatography A, 1217 (14) (2010), pp. 2171-2178. *https://doi.org/10.1016/j.chroma.2010.01.088*.

Schoutteten 2018. K. Schoutteten. '**Combination of Ion Exchange and Coagulation for Removal of Natural Organic Matter**'. TU Delft, Delft University of Technology, The Netherlands. *http://doc2cs.com/videos/*.

Schulz et al. 2007. K. Schulz, J. Dreßler, E-M. Sohnius, and D. W. Lachenmeier. '**Determination of Volatile Constituents in Spirits Using Headspace Trap Technology**'. Journal of Chromatography A, 1145 (1–2) (2007), pp. 204-209. *https://doi.org/10.1016/j.chroma.2007.01.082*.

Song et al. 2010. J. Song, W. Huang, P. Peng, B. Xiao, and Y. Ma. '**Humic Acid Molecular Weight Estimation by High-Performance Size-Exclusion Chromatography with Ultraviolet Absorbance Detection and Refractive Index Detection**'. Soil Science Society of America Journal, 74 (6) (2010), pp. 2013-2020. *https://doi.org/10.2136/sssaj2010.0011*.

Standard method, 2017. Standard Methods For the Examination of Water and Wastewater (5710). '**Formation of trihalomethanes and other disinfection by-products**'. (2017). *https://doi.org/10.2105/SMWW.2882.112*.

Tipler 2013a. A. Tipler. '**An Introduction to Headspace Sampling in Gas Chromatography - Fundamentals and Theory**'. (2013).

Tipler 2013b. A. Tipler. '**Gas Chromatography/ Mass Spectrometry**'. (2013).

'TurboMatrix Headspace Sampler and HS 40/110 Trap User's Guide'. 2008. *PerkinElmer*, 357.

Verdickt et al. 2012. L. Verdickt, W. Closset, V. D'Haeseleer, and J. Cromphout. '**Applicability of Ion Exchange for NOM Removal from a Sulfate-Rich Surface Water Incorporating Full Reuse of the Brine**'. Water Science and Technology: Water Supply, 12 (6) (2012), pp. 878-887. *https://doi.org/10.2166/ws.2012.065*.

Verdickt and Schoutteten 2018. L. Verdickt and K. Schoutteten. '**Combining Ion Exchange and Coagulation/Flotation for Enhanced Natural Organic Matter Removal in Drinking Water Treatment**'. Presented at the IWA World Water Congress, Tokyo (2018).

Zhou et al. 2000. Q. Zhou, S. E. Cabaniss, and P. A. Maurice. '**Considerations in the Use of High-Pressure Size Exclusion Chromatography (HPSEC) for Determining Molecular Weights of Aquatic Humic Substances**'. Water Research, 34 (14) (2000), pp. 3505-3514. *https://doi.org/10.1016/S0043-1354(00)00115-9*.

III. Development of analytical methods

III. Development of analytical methods

1. Review of existing headspace-gas chromatography methods for disinfection by-products / assessment of headspace-trap gas chromatography – mass spectrometry feasibility

1.1. Introduction

1.1.1. Gas chromatography – mass spectrometry for disinfection by-products

DBPs are most frequently identified and quantified using gas chromatography (GC) (Kinani et al. 2016), which is used for organic compounds that can be volatilized without thermal decomposition, usually with both a molar mass lower than 500 g mol^{-1} and a boiling point lower than 400 °C (Kusch 2018). In GC analysis, the sample is vaporized and injected onto the head of a chromatographic column in a stream of carrier gas. The components of the vaporized sample are fractioned between the inert gaseous mobile phase (usually helium, nitrogen, or hydrogen) and the stationary phase, which can be either solid or liquid supported by an inert solid matrix (Falaki 2019). The elution speed of the various analytes mainly relies on their interaction with the stationary phase (Chary and Fernandez-Alba 2012).

Currently GC coupled to mass spectrometry (MS) still dominates the analysis of a wide variety of volatile, semivolatile, and nonpolar compounds in environmental samples (Lambropoulou 2013). The separation of the compounds by GC is followed by identification based on molecule fragmentation, with electron ionization or electron impact (EI), the most common ionization mode for determining DBPs; either in full scan (FS) mode to attain more information for the identification and structural confirmation of target DBPs, or in selected ion monitoring (SIM) or selected reaction monitoring (SRM) modes for their quantitative determination (Kinani et al. 2016).

However, most GC-MS analytical methods are unable to directly analyse samples from complex environmental matrices (Lambropoulou 2013; Falaki 2019). In the case of aqueous samples, most stationary phases are intolerant towards repetitive injections of high amounts of water, while standard injection liners are incompatible with the huge gas expansion of water (van Boxtel et al. 2015). Therefore one or more pretreatment steps prior to GC-MS determination of analytes of interest are still mandatory in many applications (Falaki 2019).

1.1.2. Headspace extraction

Analysing a gas extract of volatile compounds, from a liquid sample placed in a closed vessel and heated at known temperature, by gas chromatography is called headspace gas chromatography (HS-GC). There are two main types of HS-GC; the most common technique is a single step static headspace extraction (SHE) or simply named headspace, where the sample is in equilibrium with the gas phase in a closed vessel. The other main technique is referred to as dynamic headspace extraction, gas-phase stripping, or purge and trap (PT) where the volatile compounds are stripped by a continuous flow of an inert purge and accumulated in a cryogenic or sorbent trap (Kolb and Ettre 2006; Koning, Janssen, and Brinkman 2009).

The main advantages from using HS are sample preparation simplification for the analysis of VOCs in complex matrices, and clean sample injection as the needle never penetrates the surface of dirty liquid samples (Kolb and Ettre, 2006). Different modifications have been developed to the static HS method, such as the use of headspace solid-phase microextraction (HS-SPME) (Pérez Pavón et al. 2008a; Rubio and Pérez-Bendito 2009), and HS traps (Ross 2012).

1.2. Feasibility of analysis method for trihalomethanes

Many HS methods exist for THMs, some of which are presented in Table III-1. These include purge and trap GC-MS (Ekdahl and Abrahamsson 1997; Chen and Her 2001; Lee et al. 2001; Golfinopoulos et al. 2001; Nikolaou 2002; Zoccolillo et al. 2005; Culea et al. 2006; Lara-Gonzalo et al. 2008; Ruiz-Bevia et al. 2009; Prakash et al. 2009; Ikem 2010), PT-GC-electron capture detector (ECD) (Zygmunt 1996; Ekdahl and Abrahamsson 1997; Allonier 2000; Vikesland et al. 2007) or PT-GC with atomic emission detection (AED) (Campillo et al. 2004), static headspace (HS) with either GC-MS (Golfinopoulos et al. 2001; Nikolaou 2002; Safarova et al. 2004; Culea et al. 2006; Caro et al. 2007; Pérez Pavón et al. 2008b; Rosero et al. 2012; Montesinos and Gallego 2013a) or HS-GC-ECD (Kuivinen and Johnsson 1999; Gallard and von Gunten 2002) or HS-SPME followed by GC-MS (Stack et al. 2000; Cardinali et al. 2004; Nakamura and Daishima 2005; San Juan et al. 2007; Niri et al. 2008; Joll et al. 2010) or HS-SPME-ECD (Cho et al. 2003; Antoniou et al. 2006; Bahri and Driss 2010; Sá et al. 2011) and HS-SPME-µECD (Rosero et al. 2012). However all these have only focussed on the regulated THM4 as well as other compounds in several cases. Few have included the ITHMs. Cancho-Grande et al. (2000) were the earliest to study HS-GC-ECD and purge and trap GC-MS for I-THMs but found LLE more suitable than HS and PT to recover all I-THMs quantitatively. Weinberg et al. (2002) included I-THM analysis by an EPA method using dynamic purge and trap in a US nationwide drinking water study. Allard et al. (2012) developed a new HS-SPME-GC-MS method for simultaneous analysis of all 10 THMs using a programmable temperature vaporizing (PTV) inlet with good method performance. In the food industry, a HS-GC-MS method was developed by Montesinos and Gallego (2014) for all 10 THMs in soft drinks and fruit juices, while Cardador et al. (2015, 2016) and Cardador and Gallego (2016a,b) analysed all 10 THMs in cheeses and in both liquid and solid phases of canned vegetables.

Table III-1: GC-MS methods for THMs in drinking water or other food matrices using headspace techniques.

Analytical method	THMs analysed	Detection limits (ng L^{-1})	References
PT-GC-MS	THM4	NR	(Cancho et al. 2000)
PT-GC-MS	THM4	50-100	(Golfinopoulos et al. 2001)
PT-GC-MS	THM4	50-100	(Lee et al. 2001)
PT-GC-MS	$CHCl_3$, $CHBr_3$	2-59	(Martínez et al. 2002)
PT-GC-MS	THM4	10-50	(Nikolaou 2002)
PT-GC-MS	THM4	In the order of ng L^{-1}	(Zoccolillo et al. 2005)
PT-GC-MS	THM4	1×10^3	(Culea et al. 2006)
PT-GC-MS	THM4	40-200	(Lara-Gonzalo et al. 2008)
PT-GC-MS	THM4	50-100	(Prakash et al. 2009)
PT-GC-MS	THM4	NR	(Ikem 2010)
HS-SPME-GC-MS	THM4	$(1\text{-}2.8) \times 10^3$	(Stack et al. 2000)
HS-SPME-GC-MS	THM4	0.43-6	(San Juan et al. 2007)
HS-SPME-GC-MS	THM4	5-6.3	(Kim et al. 2012)
HS-SPME-GC-MS	THM4	8-20	(Allard et al. 2012)
	I-THMs	1-8	
HS-GC-MS	THM4	50-400	(Nikolaou 2002)
HS-GC-MS	THM4	<100	(Culea et al. 2006)
HS-GC-MS	THM4	23-102	(Rosero et al. 2012)
HS-GC-MS	THM4	10-20	(Montesinos and Gallego 2013)
HS-GC-MS	THM4	10-20	(Montesinos and Gallego 2014)
HS-GC-MS	I-THMs	20-100	
HS-GC-MS	THM4	50-100	(Cardador et al. 2015)
	I-THMs	120-500	
HS-GC-MS	THM4	10-20	(Cardador and Gallego 2016b)
	I-THMs	20-150	

1.3. Feasibility of analysis method for haloacetic acids

Delgado et al. (2014) describe in a chapter of a relatively recent book the different analytical methods available in order to analyse HAAs in waters intended for human consumption. In the following paragraphs, methods based on HS-GC-MS are discussed.

1.3.1. Challenges with haloacetic acid analysis by gas chromatography

Gas chromatography is most commonly used for analyzing HAAs. However, HAAs are challenging to study by GC due to their low volatility and high polarity. Hence the existing methods to analyse the

HAAs require a derivatisation step to their respective methyl esters to make them suitable for GC (Cardador et al., 2008).

Several methods are currently approved by the EPA for compliance monitoring of HAA5 in drinking waters:

- EPA Method 552.1,
- EPA Method 552.2,
- EPA Method 552.3,
- EPA Method 557,
- Standard Method 6251B (22nd edition),
- Standard Method online 6251B-07,
- Standard Method online 6251B-94 (EPA 2016).

EPA Methods 552.1 and Standard Method 6251B can also be used to determine BCAA concentrations, while EPA Methods 552.2, 552.3 and 557 can analyse the HAA9 (EPA 2016). These methods use methyl tert-butyl ether (MTBE) or anion exchange resins to extract HAAs from water samples; diazomethane or acidic methanol (derivatization) to convert HAAs into methyl esters, and analysis by gas chromatography-electron capture detection (GC-ECD). However, these methods have the following limitations:

- (i) a low sensitivity detection for monochloroacetic acids (MCAA),
- (ii) a susceptibility to chromatographic interference,
- (iii) identification problems due to drifting in retention time,
- (iv) long GC run times,
- And (v) poor or no response for brominated trihaloacetic acids (tri-HAAs) (Xie 2001; Waseem and Abdullah 2010).

1.3.2. Headspace for haloacetic acid analysis

While HS is usually used for volatile compounds, HS-GC can also be used as an alternative way to quantify some nonvolatile compounds if they can be quantitatively transferred to related volatile substances by related reactions (Chai et al. 2003; van Boxtel et al. 2015).

1.3.2.1. Thermal decarboxylation

There are several HS method analytical possibilities for HAAs. Thermal decarboxylation of TCAA to $CHCl_3$ has been used for HS-GC determination of TCAA in various matrices (Christensen and Rasmussen 1988; Plümacher and Renner 1993; Reeves et al. 2000; Xie et al. 2018).

$$CCl_3COOH \longrightarrow CHCl_3 + CO_2$$

The other tri-HAAs (*i.e.*) TBAA, BDCAA and CDBAA which are difficult to analyse by conventional GC methods could theoretically also be thermally decarboxylated and measured as $CHBr_3$, $CHBrCl_2$, $CHBr_2Cl$ (Zhang and Minear 2002; Ma and Chiang 2005). However the mono-HAAs and di-HAAs cannot be measured in the same way, as their thermal degradation is a hydrolysis process (Lifongo et al. 2010). Hence, for analysis of HAAs in water samples by HS techniques, methylation of the HAAs to their less soluble methyl esters has been the method of choice.

1.3.2.2. Methylation

1.3.2.2.1. Headspace gas chromatography methods

Methyl esters of haloacetic acids are less soluble in water than free acids, giving headspace methods analytical possibilities. Neitzel et al. (1998) were the first to develop a HS-GC-MS technique using dimethylsulfate (DMS) as a methylating agent to analyse 3 HAAs (MCAA, DCAA, TCAA) directly in water. Using tetrabutylammonium hydrogensulfate (TBA-HSO_4) as an ion-pairing agent for the in-situ methylation reaction (Figure III-1) greatly increased the yield of the reaction (Neitzel et al. 1998).

$$\text{R-COOH} + (CH_3CH_2CH_2CH_2)_4N(HSO_4) \rightleftarrows \text{R-COO}^-N^+(CH_3CH_2CH_2CH_2)_4 + H_2SO_4$$

$$2\ \text{R-COO}^-N^+(CH_3CH_2CH_2CH_2)_4 + (CH_3)_2SO_4 \rightleftarrows 2\ \text{R-COO-}CH_3 + 2((CH_3CH_2CH_2CH_2)_4N^+)SO_4^{2-}$$

Figure III-1: HAA methylation by DMS in the presence of TBA-HSO_4

Cardador et al. (2008) have improved the HS-GC-MS method one step further for the HAA9 by adding a micro-volume of n-pentane (LLME step) which protects the esters and increases their volatility by co-evaporation of n-pentane and the methylated esters. Additionally, THMs were also analysed simultaneously. In the absence of n-pentane only four HAA methyl esters (MCAA, MBAA, DCAA, and TCAA methyl esters) were extracted (Cardador et al. 2008). For the past decade, they have been optimizing the method for various food matrices (Cardador and Gallego 2012, 2017), including the analysis of MIAA (Cardador and Gallego 2015; Cardador and Gallego 2016a) or 4 I-HAAs as well (Cardador et al. 2015; Cardador and Gallego 2016b; Cardador et al. 2016).

1.3.2.2.2. Headspace solid-phase microextraction gas chromatography methods

Also in 1999 a method for the analysis of six HAAs using HS-SPME-GC-MS was first developed (Sarrión et al. 1999), using acid-catalysed ethylation to obtain low detection limits (Table III-2), and good sensitivity for the HAA6 with LODs 2 to 20 times lower than those obtained with the EPA method 552.2, but still with a significant cost of time and labor (Sarrión et al. 2000). In 2000 they proposed a new method with direct derivatization of HAAs in water by DMS and were able to analyse all the HAA9. (Sá et al. (2012) also determined 9 HAAs after derivatization to their respective methylated esters with DMS, using the commercial fiber CAR-PDMS and HS-SPME-GC-ECD. Cardador and Gallego (2010) attempted to combine liquid-liquid microextraction (LLME) – using n-pentane as extraction solvent, and-HS-SPME – using a CAR-PDMS SPME fiber and with derivatization by DMS in the presence of TBA-HSO_4. However they concluded that the method was far less sensitive than the more simple LLME HS-GC-MS technique and inadequate for the levels of HAAs in drinking water samples.

Table III-2: HS-GC methods (using derivatization) to monitor HAAs (mainly in water samples).

HAAs	**Matrix**	**Derivatisation reagent**	**Extraction technique**	**Detection**	**LOD ($\mu g\ L^{-1}$)**	**RSD (%)**	**References**
MCAA, DCAA, TCAA	Water	DMS	HS	MS	1-10	≤ 6.6	(Neitzel et al. 1998)
DCAA, TCAA	Blood and solid tissues	Acidic methanol	HS	MS	5 and 10	-	(Muralidhara and Bruckner 1999)
HAA9	Water	DMS	LLME-HS	MS	0.02-0.40	≤ 10.4	(Cardador et al. 2008)
13 HAAs	Liquid phase of canned vegetables	DMS	LLME-HS	MS	0.02-0.50	≤ 10.5	(Cardador and Gallego 2016b)
HAA6	Water	Acidic ethanol	HS-SPME	IT-MS	0.01-0.20	≤ 7.9	(Sarrión et al. 1999)
HAA9	Water	DMS	HS-SPME	IT-MS	0.01-0.40	≤ 10.9	(Sarrión et al. 2000)
HAA9	Water	DMS	LLME-HS-SPME	MS	0.3-15	≤ 12.7	(Cardador and Gallego 2010)
HAA9	Water	Acidic methanol	LLME-HS-SPME	ECD	0.2-16	-	(Wu et al. 2002)
TCAA	Urine	Acidic methanol	LLME-HS-SPME	ECD	0.015	-	(Rastkari et al. 2012)
HAA9	Water	Acidic methanol	LLME-HS-SPME	ECD	0.02-0.7	≤ 12	(Hammami and Driss 2013)
6 HAAs	Water	DMS	HS-SPME	ECD	0.029-0.283	≤ 20	(Sá et al. 2012)
DCAA, TCAA	Water	Acidic methanol	HS	ECD	3 and 0.5	≤ 21.3	(Wang and Wong 2005)
HAA5	Water	DMS	HS	ECD	0.5	-	(Ghoochani et al. 2013)
HAA6 + MIAA	Water	DMS	HS-trap	MS	0.005-0.10	≤ 20	This work

DMS: dimethylsulfate, ECD: electron capture detection, HS: headspace, LLME: liquid-liquid microextraction, LOD: limit of detection, MS: mass spectrometry, RSD: relative standard deviation

Some authors have also combined the utilization of LLE using MTBE as extractant solvent to HS-SPME-GC-ECD, with acidic methanol for derivatization (table III-2). The most sensitive method among them is the one by Hammami and Driss (2013). Nevertheless, the sample preparation is more complex as it involves first LLE with evaporation of the extraction solvent to dryness, before HS-SPME derivatization of HAAs into their methyl esters with acidic methanol and finally extraction.

1.3.3. Conclusion

The esterification processes used to transform HAAs in their respective methylated esters, make these compounds less polar, more stable and volatile. However, the thermal instability of HAAs is an important problem that must be controlled throughout the process since there might be degradation. While there exist several HS-GC methods for the determination of HAAs with low detection limits and adequate precision performance (Table III-2), there are currently no HS-trap GC-MS method described in the literature for the analysis of HAAs in drinking water.

1.4. Feasibility for haloacetonitriles and haloacetamides

1.4.1. Feasibility for haloacetonitriles

Several LLE, HS and HS-SPME-GC-MS methods exist for their analysis with many able to analyse THMs in the same run. Better limits of detection are achieved with HS and HS-SPME than with conventional LLE pre-treatment related methods (Kristiana et al. 2012). It is worth noting that in several HS studies MCAN and MBAN have significantly higher detection limits than other haloacetonitriles (Kristiana et al. 2012; Montesinos and Gallego 2013). Kristiana et al. (2012), explained this by the lower K_{ow} values of MCAN (0.14) and MBAN (0.25) compared to BCAN (1.23), DBAN (1.30), DCAN (1.53) and TCAN (2.54) at 25 °C. However Luo et al. (2014) found lower values for MCAN and MBAN than for all the other HANs they studied in their HS-SPME-GC-MS method.

1.4.2. Feasibility for haloacetamides

HAcAms analysis is particularly complex due to their low volatility and mono- and di-halogenated HAcAms having significantly different polarity to their trihalogenated analogues. The use of GC-MS is limited or of no use when the target DBPs have high molecular masses, are non-volatile or very polar (Shi et al. 2012). Previous studies have indicated that the organic solvents methyl tert-butyl ether (MTBE), n-hexane, and ethyl acetate can be used as extraction solvents towards the detection and identification of HAcAms by GC-MS (Chu et al., 2010; Liew et al., 2012). HAcAms analysis has focused on DCaCAm and TCAcAm because of high LODs for the mono-HAAms (Liew et al. 2012) and because DCAcAm is the main HAcAm found in drinking water (Krasner et al. 2006).

To our knowledge, there is no existing HS method for HAcAm analysis. Ding and Chu (2017) in their review on the recent advances in the analysis of nitrogenous disinfection by-products wrote that HS and PT are not suitable for the concentration of HAcAms, because of their high boiling points.

1.5. Conclusion

Highly water soluble analytes such as the HAAs and the HAcAms are more challenging to analyse by HS-GC than the more volatile THMs and HANs, due to poor partitioning into headspace. Thus, they are expected to require extraction in a volatile solvent and/or derivatisation to a more volatile form for acceptable method performance. HAcAms analysis by HS has never been achieved before and they are not suitable for direct HS analysis.

Most of the methods developed to date for the determination of HAAs in waters have two serious drawbacks: the time required for the sample treatment and the degradation of the most unstable species to other DBPs (Cardador et al., 2008). In the case of the Cardador et al. (2015, 2016) studies, what is highly interesting is that they were able to combine the analysis of the 10 THMs as well as 13 HAAs. Thus, the initial step was to attempt to combine HAA and THM analysis by HS-trap-GC-MS.

From that premise, the aims of this work were to first develop a fast derivatisation/extraction method to analyse HAAs – possibly in combination with ITHMs – by HS-trap and attain low quantification limits, and to develop a method for the other N-DBPs.

2. Initial attempts to combine THM and HAA analysis

Briefly the mechanism was, each HAA in an aqueous medium forms an ion-pair with TBA-HSO_4 enabling the transfer of HAAs from the aqueous phase into the organic n-pentane phase where they react with DMS to produce methyl haloacetates (Cardador et al., 2008).

Sample preparation was based on the procedure described in Cardador et al. (2008, 2015, 2016):

In a 22 mL glass HS vial were placed:

- 4 g (2.8 M) of Na_2SO_4, 9.6 mL water samples or mQ water containing the spiked HAAs,
- 6.4 µg L^{-1} of 1,2-dibromopropane,
- 10 µg L^{-1} of 2-bromopropionic acid,
- 100 µL of a 0.5 M concentration of an ion pairing agent (TBA-HSO_4, 2.3 µmol as aqueous solution), 150 µl of derivatisation reagent (DMS),
- and 150 µL of n-pentane were added.

The vials were immediately sealed and stirred in a Vortex mixer for 3 min in order to carry out the methylation process. Finally, the vial was placed in the 40-space autosampler carrousel from which the robotic arm took each one and introduced it in the HS oven.

THM and HAA methyl ester separation through the chromatographic column was performed by using an appropriate program of temperatures described in Chapter II.

2.1. n-pentane, a problematic solvent

An important n-pentane and DMS solvent front is clearly visible on the first chromatogram (Figure III-2a)), hindering early peak detection as illustrated in the case of the chloroform and bromodichloromethane peaks (Figure III-2c)), which are greatly affected (poor peak resolutions and intensities).

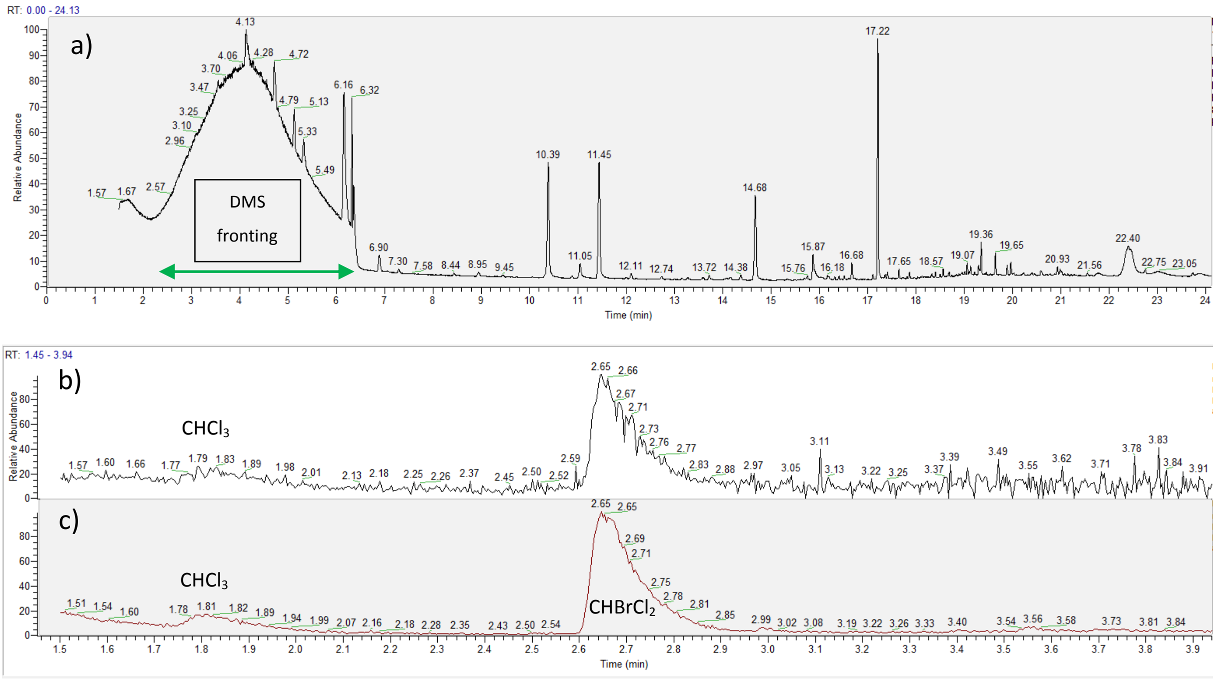

Figure III-2: An example of the n-pentane issue for chloroform, 20 µg L^{-1} trihalomethanes, 50 µg L^{-1} TBA-HSO$_4$, 50 µg L^{-1} DMS, 150 µL n-pentane. a): Full scan 40-400, where the DMS solvent front is clearly visible, b): SIM m/z 47 (characteristic of $CHCl_3$), and c) m/z 85 (characteristic of $CHCl_3$ and $CHBrCl_2$).

n-pentane also impacted on the detection of HAAs. With increasing n-pentane levels, peak areas decreased for all HAAs especially for the early eluting peaks MCAA and MBAA - that were not detectable after adding a tiny amount of n-pentane - as well as DCAA. This is probably due to the presence of both the extractant (n-pentane) and the derivatisation reagent DMS competing with the active sites of the fibre, which can reduce the sorption of the analytes on the fibre as well as its lifetime (Cardador and Gallego 2010). According to Cardador et al. (2010), in HS-SPME-GC-MS, by decreasing the temperature to 45 °C, less n-pentane was volatilized, therefore reducing the competition with the esters in the fibre and/or the affinity of the analytes for the fibre coating. The thermostatting temperature was tested at 45, 55 and 60 °C using 80 µL DMS and 150 µL n-pentane but this did not lead to any significant improvement in the peaks.

The problem is more related to the column temperature program. Indeed, due to the low boiling point of n-pentane (36.1 °C), when starting the GC column at 40 °C, the solvent does not recondense in the column inlet. In the absence of cryogenic units, it was not possible to start the column temperature 10 to 40 °C below that temperature to condense the solvent and prevent tailing peaks.

By using an important split-ratio, Cardador et al, (2010) were probably able to reduce the amount of n-pentane reaching their column, however only splitless injections are possible in our HS-trap configuration (see Chapter II).

Eventually it was decided to work without n-pentane for THMs as well as for HAAs as even small amounts (10 µL) decreased peak area size for MCAA and MBAA.

2.2. In the absence of n-pentane

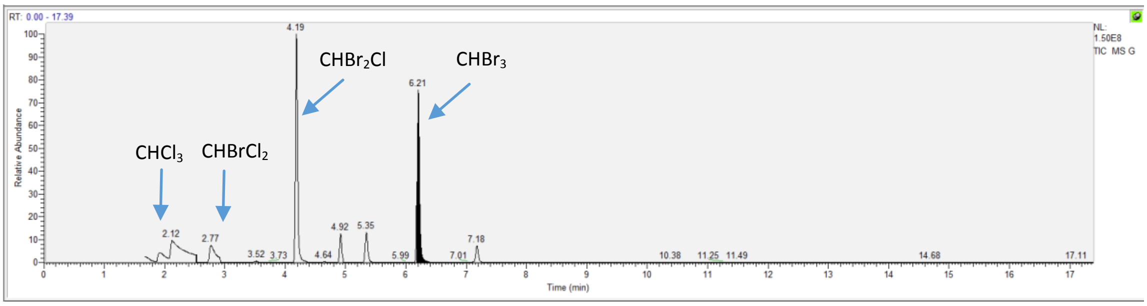

Figure III-3: 20 µg L^{-1} haloacetic acid standard without trihalomethanes added.

In the absence of n-pentane, huge THM formation from HAA thermodegradation was clearly visible especially $CHBr_2Cl$ and $CHBr_3$, but also $CHCl_3$ and $CHBrCl_2$. Hence, it is not possible to analyse THMs and HAAs in the same run as the degradation of trihaloacetic acids, artificially increases THM concentrations during the analysis.

3. Trihalomethane method

3.1. Sample preparation

3.1.1. Salt addition

Adding salt to an HS vial often optimizes the extraction, according to (Montesinos and Gallego (2013), the best results for THM4 among different salts (potassium chloride, sodium chloride and anhydrous sodium sulfate) is with sodium sulfate.

Allard et al. (2012) found that for all 8 of the 10 THMs the extraction efficiency of all analytes increased with the sodium sulfate concentration even above the salt saturation of the solution at 5.5 g. The two exceptions were $CHCl_3$ and $CHCl_2Br$ where the maximum was found around 2.5 g.

The solubility of sodium sulfate is highly dependent on temperature, with a sharp increase in the solubility up to 32.4 °C, followed by a slight decrease in solubility at higher temperatures (Bharmoria et al. 2014). When totally solubilized, samples can even be put in the fridge, whereas if one salt crystal remains, when the lab temperature drops slightly (such as in the evening), some of the salt will re-crystallize. Additionally, if some salt remains after 3 minutes of vortexing, it will not entirely dissolve during the thermostatting.

As the lab temperature varies slightly thus the salt saturation as well. Because the increase in efficiency was not the same for all THMs it was preferred to maintain the salt addition always below the saturation point for better correction by the internal standard, therefore it was fixed at 4 g (40% w/v), and sometimes at 3 g when the solubility was too low (see part V.2).

3.1.2. Solvent addition

Organic modifiers are often used to accelerate volatilization of analytes from the water in HS-GC-MS. n-pentane was previously shown to be problematic for THMs. Other organic modifiers such as MTBE and n-hexane have been shown to improve the volatilization of brominated THMs (Montesinos and Gallego 2013), but were not used as the THMs are already sufficiently volatile and non-polar and to protect the column.

3.2. Headspace-trap conditions

3.2.1. Thermostatting

Montesinos and Gallego (2013, 2014) as well as Cardador et al. (2015a) found that peak signals increased with temperature up to 70 or 80 °C and with a minimum of 10 minute equilibration time, above which conditions the signals remained constant. However, important HAA degradation could take place at temperatures ≥ 60 °C as shown in Table III-4, which could be an issue in real samples containing both THMs and HAAs.

Zhang and Minear (2002) calculated the decomposition rates of 30 µg L^{-1} of the brominated tri-HAAs in water at pH7 at 4 different temperatures (4, 23, 36 and 50 °C). Table III-4 presents the degradation percentages of the brominated tri-HAAs by extrapolating the equations from the following figure:

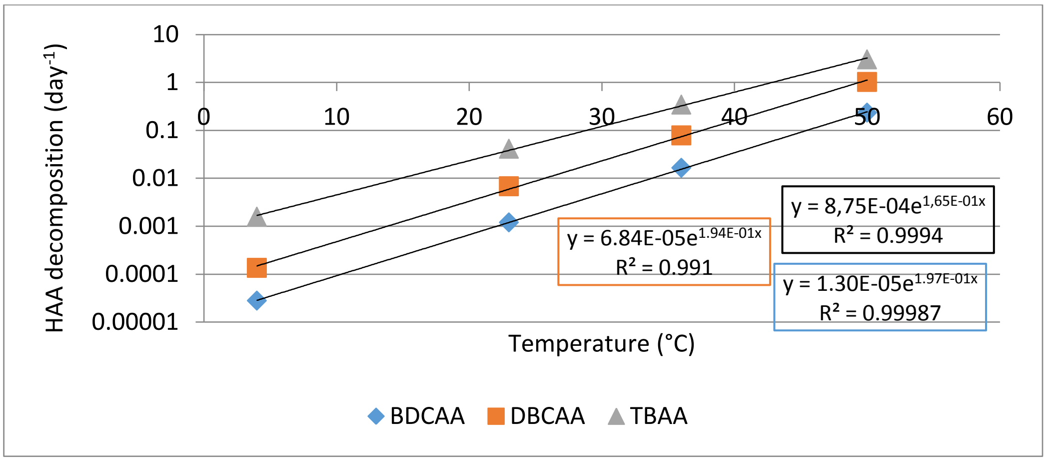

Figure III-4: Correlation between the decomposition rates for the brominated tri-HAAs and the water temperatures (figure built using the decomposition rates given by Zhang and Minear (2002)).

Thus to limit the degradation, the temperature was left at 60 °C and the heating period at 10 minutes.

Table III-4: Estimation of brominated tri-HAA decarboxylation, using data from Zhang and Minear (2002). (In grey, from extrapolated decomposition rates).

Temperatures	36°C	50°C	60°C	70°C	80°C
HAAs	% loss (calculated)		% loss (extrapolated)		
10 minutes					
TBAA	0	2	12	84	100
DBCAA	0	1	6	46	100
BDCAA	0	0	1	9	61
20 minutes					
TBAA	0	4	26	100	100
DBCAA	0	1	11	100	100
BDCAA	0	0	2	19	100

3.2.2. Desorption

In this study, desorption temperatures of 180, 200 and 220 °C were evaluated. The responses were found to be relatively similar at 200 and 220 °C, even for the highest molecular weight compounds ($CHBrI_2$ and CHI_3). Hence, the temperature was fixed at 220 °C to limit carry-over.

3.3. Results

3.3.1. Chromatographic separations

Chromatographic separations for all the THMs were achieved in less than 18 minutes. All the analytes and internal standard peaks were well separated, with adequate gaussian peaks except for $CHCl_3$, which eluted the earliest with some tailing. However, the precision and linearity remained acceptable (see Chapter II).

3.3.2. Method validation

3.3.2.1. Precision

The precision of the method was evaluated by determining the repeatability. The repeatability was done with five samples at 6 µg L^{-1} of each regulated THMs and 60 ng L^{-1} of I-THMs in one day. RSDs were always below 10%.

3.3.2.2. Linearity

The linear range and the r^2 for each compound were studied. The linearity was investigated up to 300 µg L^{-1} for THM4 and up to 5 µg L^{-1} for the I-THMs. For the highest molecular weight compounds and iodoform in particular, the calibration curves became sometimes exponential at the higher concentration, be it with external or internal calibration. One of the possible answers could be the presence of excess amounts of MeOH from the stock standards at high standard concentrations. For example, a solution containing 16 mg L^{-1} THM4 diluted in mQ to 40 µg L^{-1} leads to 25 µL of MeOH added in the HS vial.

Possible issues from the presence of excess MeOH:

- MeOH could perhaps increase the solubility of the less soluble compounds,
- MeOH could perhaps compete for space in the gas phase, changing the partitioning coefficient,
- MeOH could perhaps compete for the sites on the trap with the more volatile compounds.

Therefore high concentrations of MeOH are one of the most common problems in purge-and-trap analysis by creating signal response anomalies ('Bulletin 916 Purge-and-Trap System Guide' 2019). More concentrated standards of THM4 were thus bought (2 g L^{-1} instead of 0.2 g L^{-1}), allowing the preparation of working standards with less MeOH traces.

For the THM4, the calibration curves were linear up to 60 µg L^{-1}, but at 300 µg L^{-1} there was a slight decrease by 15% compared to the expected value calculated by the linear regression for $CHBr_3$ and $CHBr_2Cl$, and tailing of $CHBrCl_2$ leading to part of the peak exceeding the RT and not being totally measured (-30%). Hence, the upper range for the THM4 was fixed at 60 µg L^{-1}.

3.3.2.3. Carry-over

Some carry-over took place in the ng L^{-1} range, which could potentially affect the measurements of the lower concentrations of the I-THMs, especially for the heaviest iodinated THMs, mainly CHI_3, $CHBrI_2$ and $CHClI_2$.

Table III-5: Carry-over for I-THMs.

Compound	Concentration studied (ng L^{-1})	Carry-over (ng L^{-1})	Concentration studied (ng L^{-1})	Carry-over (ng L^{-1})
CHBrClI	348	No	870	No
$CHBrI_2$	300	Negligible	1200	8
$CHClI_2$	300	1	768	1.6
$CHBr_2I$	340	0.8	800	1.3
$CHCl_2I$	290	Negligible	1160	0.9
CHI_3	300	0.75	1200	20

3.3.2.4. Haloacetic acid interferences

25 µg L^{-1} HAAs were spiked in Saint-Amand mineral water and analysed with the method for THMs. The percentage of THM formation was calculated by assuming 100% to be the spiked HAA concentration, and calculating the peak area of the decomposition THM for each tri-HAA. Less than 1% $CHCl_3$, $CHBr_2Cl$ and $CHBrCl_2$ were formed while 6.5% $CHBr_3$ was produced.

4. Haloacetic acid method development

4.1. Sample preparation (without n-pentane)

4.1.1. Dimethylsulfate volume

Nearly all the studies reported in Table III-2, that used dimethylsulfate (DMS) as a methylation agent, added amounts close to 100 µL. More specifically, Sarrion et al. (2000) studied the effect of the DMS concentration by adding a range of different volumes up to 200 µL and found optimal volumes between 60 and 100 µL with 100 µL the best for most of the HAAs except for MBAA, which was at 60 µL. Surprisingly in our case, when first attempting to add 100 µL of DMS to HS vials containing the HAA standards, the excess of DMS was extracted on the fibre appearing in the chromatogram. This lead to a shift in RTs and broadening of the peaks for some of the analytes. None of the previous studies reported any such issue with the DMS excess, except for Cardador and Gallego (2010) who explained their problem with DMS fronting in the case of HS-SPME-GC/MS to be due to the presence of n-pentane enhancing volatilization of DMS.

A test with a range of DMS volumes was thus carried out from 50 to 100 µL in increments of 5 µL. Surprisingly no such fronting was found, this is because DMS half-life in water at pH 7 and 25 °C is only of 1.15 hours, producing methanol and sulfuric acid (PubChem). This means that the excess DMS, in the absence of any solvents, will degrade rapidly in water. As all samples had been prepared in one go and placed on the carrousel, there had been sufficient time (several hours) to hydrolyse the DMS in excess for the samples with high amounts of DMS. Thus leaving a sufficient amount of time for excess DMS to decrease should remove the fronting problem in the first tubes.

Therefore, a period of 3 hours and 30 minutes (see part on methyl ester stability) was chosen between the time the samples were prepared and the analysis. The amount of DMS was increased from 50 µL up to 150 µL in increments of 5 µL. With such conditions, volumes of 70 µL and more lead to DMS excess and poor peak shapes. To ensure complete methylation of the analytes without too much DMS excess that can reach the trap and column, a volume of 60 µL of DMS was chosen.

4.1.2. Salt addition parameter

The effect of the amount of sodium sulfate on the extraction of HAA methyl esters was studied to the water from 0 to 50% w/v addition in duplicates, using 25 µg L^{-1} HAAs with 100 µL of TBA-HSO$_4$, 60 µL of DMS. All samples were entirely dissolved except for those at 5 g, where some salt crystals remained.

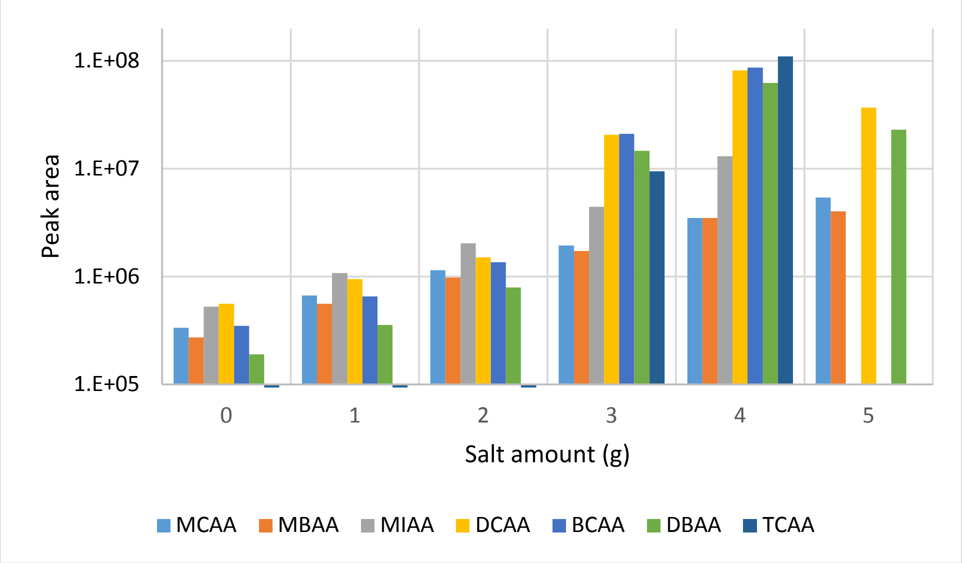

Figure III-5: Evolution of the HAA signal with the salt added to the HS vials.

Different behaviours were observed for mono-, di- and tri-HAAs. All three mono-HAAs (MCAA, MBAA, MIAA) showed an 2-fold increase in signal at every 1 g salt increase from 0 to 4 g. The di-HAAs also showed a doubling of the signal from 0 to 1 g and from 1 to 2 g. However from 2 to 3 g the signal increases by approximately 15 times for DCAA, BCAA, and DBAA respectively, and increased by approximately 4 times from 3 to 4 g with the same trend of slightly higher increase for the more brominated species. TCAA dropped tenfold in signal count at 1 g, before increasing back to the same level at 2 g. At 3 g, the signal increase by 360 times for TCAA and by 12 at 4 g.

Unfortunately, at 5 g, peak areas dropped for all HAAs except for MCAA and MBAA. MCAA the earliest to elute showed a slight increase in the RT and some tailing. The following peaks for 2-bromopropionic acid, DCAA and MBAA showed poor "roundish" shapes with an important shift in their RTs, typical signs of the presence of excess DMS in the column. MIAA, TCAA peaks were no longer visible, but seemed to have suffered huge RT shifts out of the usual RT peak range. The RTs for the following HAAs (BCAA, DBAA) had considerably shifted as well. The signal for several fragment ions were completely drowned, notably those at m/z 127 and 129 for BCAA. Overall, it seems as if an increase from 4 to 5 g allows more DMS to reach the column and negatively impact the analysis.

It is worth noting that both (Sarrión et al. 2000) and (Cardador et al. 2008) were able to use 5 g of salt which they found gave the highest signal for 10 mL samples. However in order to use 5 g, one would need to i) leave the samples for a longer time between preparation and analysis to hydrolyze DMS and ii) make certain all the salt is dissolved at 5 g for better repeatability, since the different HAAs behave differently. For these reasons, it was decided not to increase the salt content and to use only 4 g.

Sodium sulfate of higher granulometry (Sigma-Aldricht), was easier to vortex than the salts from other suppliers (Merk and VWR) which quickly formed a solid block at the bottom of the vial, extremely difficult and long to dissolve without heating. Also only with this salt were the 3 brominated tri-HAAs detectable at 40% salt content.

4.1.3. Tetrabutylammonium hydrogen sulfate addition

The effect of the amount of ion-pairing agent on the derivatization of 25 µg L^{-1} HAAs was studied by adding up to 200 µL (6.6 µmol) of TBA-HSO_4 (0.5 M) to the water and using 60 µL of DMS.

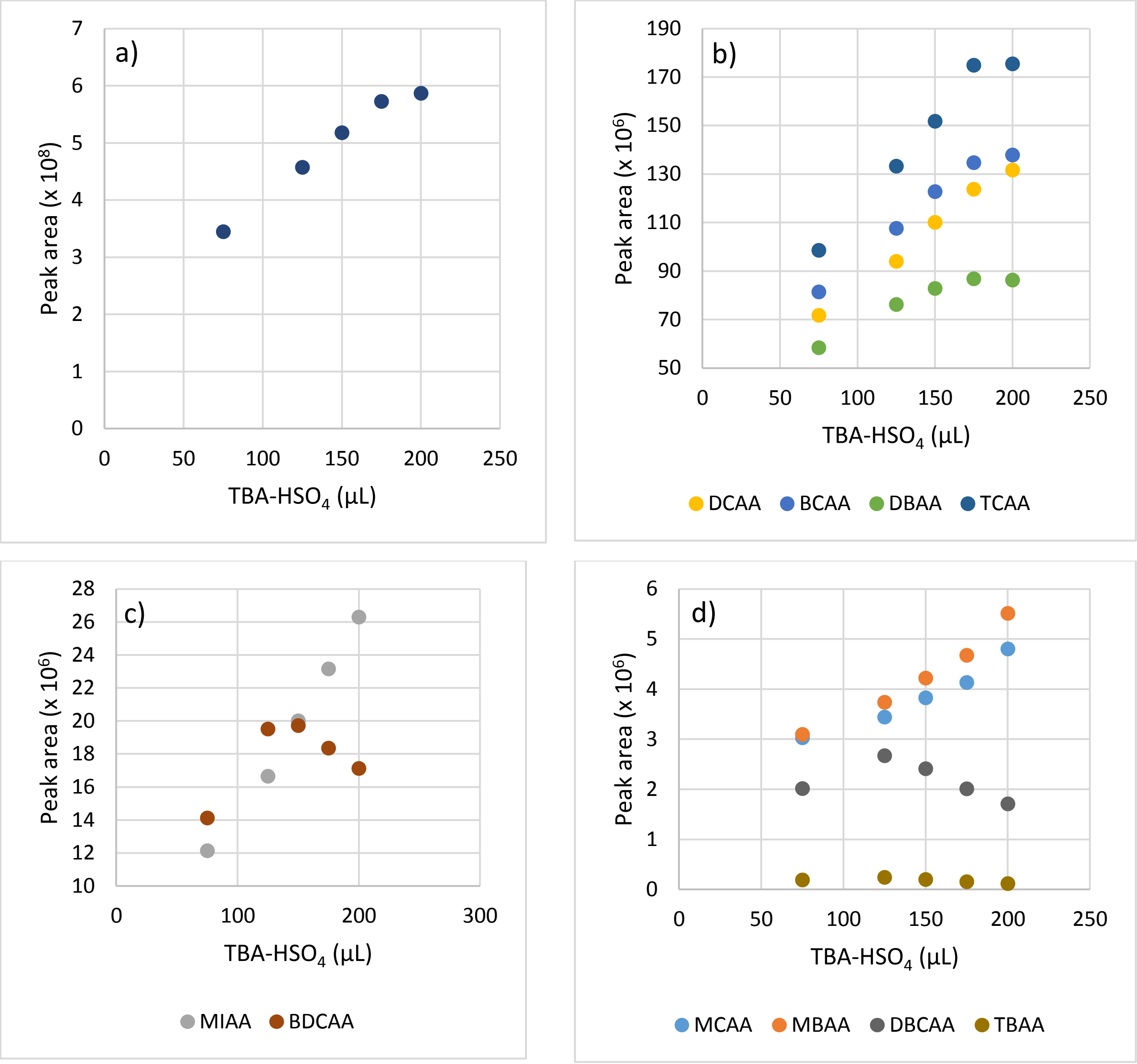

Figure III-6: Influence of the volume of ion-pairing agent on a) the sum of signals for the 9 HAAs, b), c) and d) individual HAAs.

For MBAA and MCAA, the signals increased from 50 to 200 with positive quadratic curves. The signals increased initially in a linear manner for MIAA, DCAA, BCAA, TCAA, DBAA, with some reaching maximums for the highest volumes of TBA-HSO_4 tested. Increasing the volumes lead to initially higher peak area signals at 125 µL or 150 µL for the brominated tri-HAAs then linear decrease onwards.

Hence 150 µL may be a good compromise between increasing the peak signal for 7 of the HAAs while not losing too much in signal for the brominated tri-HAAs. However, as the brominated tri-HAAs esters greatly suffer from hydrolysis and are tricky to analyse anyway, it may be more interesting to use 200 µL of 0.5 M TBA-HSO_4 for HAA analysis for several reasons: mainly boosting the signal for MCAA and MBAA that have the poorest limits of detection and also to increase the repeatability of the analysis by lowering the impact of the volume of TBA-HSO_4 on the signal variation.

4.1.4. Methyl ester stability: influence of pH and time

Cardador et al. (2008) could only detect MCAA, MBAA, DCAA, and TCAA by HS-GC-MS in the absence of n-pentane due to methyl ester hydrolysis. Sarrión et al. (2000) and Sá et al. (2012) on the other hand did not report any stability issues for the 9 brominated and chlorinated methyl esters in aqueous samples, with good RSDs in particular for Sarrión et al. (2000).

However with the HS-trap using an autosampler, many samples can remain for hours on the autosampler tray, awaiting analysis, thus compounds which are not stable in an aqueous media will degrade with the last samples to be analysed experiencing the most hydrolysis.

Thus the effect of pH on methyl ester stability in the aqueous matrix, was studied by adding sulfuric acid to the water to set the pH at different values (2.6, 5.4, and 7) (Figure III-6).

Figure III-7: From top to bottom, HAAs peak area signal evolution from the moment the first run was started pH 2.6, 5.4, and 7 with 100 µL of TBA-HSO4, 60 µL of DMS, 4 g of salt.

At pH 2.6, brominated tri-HAAs methyl esters degrade and no peaks are seen except for DBCAA, which was detected 5 times in the first 7 standards but with huge non repeatable variations.

At all three pHs studied, the highest peak areas for the more stable compounds is actually for the 2nd or 3rd vial, thus after approximately 3 hours at lab temperature before analysis. With increasing time before analysis, the signal decreased except for the mono-HAAs.

4.2. Headspace trap - GC parameters

4.2.1. Choice of internal standard

Initially 1,2-dibromopropane was chosen as the internal standard and 2-bromopropionic acid as surrogate. However 1,2-dibromopropane peak area signal increased with HAA concentrations and was thus discarded. 2-bromopropionc acid was used as the internal standard instead, as is recommended by Sarrion et al. (2000) who experienced the same phenomenon.

4.2.2. Desorb temperature

The desorb temperature was tested at 175, 200, 225, 250 and 275 °C. The carry-over decreased from 175 to 200 °C but no further at higher temperature. So a desorb temperature of 200 °C was chosen.

5. Attempts at combining trihalomethane with haloacetonitrile analysis

5.1. Parameter optimisation

5.1.1. Salt addition

According to (Montesinos and Gallego (2013), the best results for HANs among different salts (potassium chloride, sodium chloride and anhydrous sodium sulfate) is with sodium sulfate. Similarly to most of the THMs, peak signals increase with salt addition, even above the saturation point (Luo et al. 2014). For the reasons already previously stated for THMs, the salt amount was kept at 4 g.

5.1.2. Solvent addition

Peak signal for HANs have been found to increase with the addition of microvolumes of MTBE (Montesinos and Gallego 2013; Cardador and Gallego 2017). No solvent was added in our study to avoid solvent fronting by HS-trap GC-MS.

5.1.3. pH

The hydrolysis rate of HANs increases with an increasing pH value (Glezer et al. 1999). It also depends on the number and type of halogen atoms in the molecule, with the following hierarchy of hydrolysis rates: k_H *TCAN* > k_H *BDCAN* > k_H *DCAN* > k_H *BCAN* > k_H *DBAN* > k_H *MCAN* > k_H *MBAN* (Yu and Reckhow 2015). The hydrolysis of haloacetonitriles under basic conditions yields the corresponding haloacetamides, which can be further hydrolyzed into the corresponding haloacetic acids. Therefore HANs should be preserved in weak acid solutions between sampling and analysis (Glezer et al. 1999).

HANs can be analysed by HS between pH 2.6 and 5.6 (HANs), above which the signals decrease due to base-catalysed degradation (Montesinos and Gallego 2013). A pH of 5.3 was used for the subsequent tests.

5.1.4. Thermostatting

Montesinos and Gallego (2013) found an optimal heating of the HS vials at 80 °C, for 18 minutes. Such high conditions were not used to limit potential degradation of HAAs in real samples.

5.1.5. Desorption

A previous study found 220 °C an optimal temperature for desorption of HANs from SPME fibers (Luo et al. 2014), hence this parameter was not modified.

5.2. Results

$CHCl_3$ and CAN elute with very close RTs (1.9 min for chloroform and 2.34 min for CAN). Thus when a high concentration of $CHCl_3$ is used compared to CAN, drowning of m/z 48 and 50, two of the few possible m/z fragments (Figure III-8a)). While m/z 75 and 77 can be used, there is some tailing occurring. Bromoacetonitrile is also impacted for m/z 79, while m/z 119 and 121 can be used.

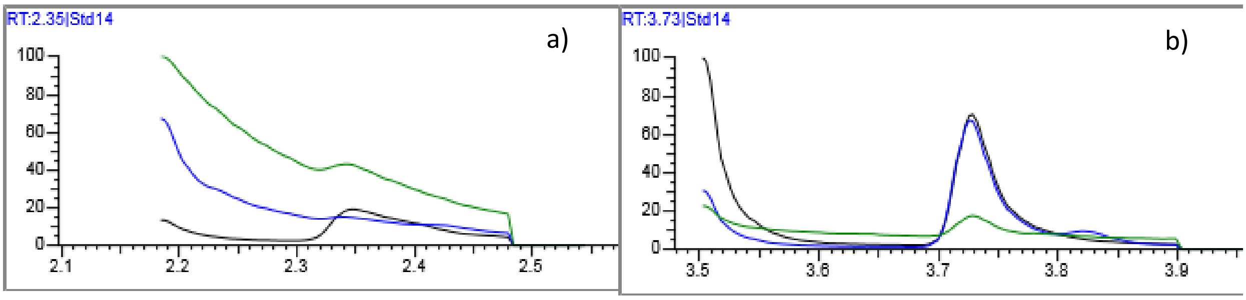

Figure III-8: THM/HAN mix standard a) CAN peak m/z = 48 (green), 50 (blue), 75 (black), b) BAN peak m/z = 79 (green), 119 (black) and 121 (blue).

To allow better elution and separation of these early eluters, a lower starting temperature is required. Several studies using similar columns start the oven temperature program at 35, 34 or even

30 °C. However, in our lab it is not realistically possible to lower the initial oven temperature to 30 or 35 °C, to allow better separation of these two eluting compounds. Indeed, at lower temperatures, the oven will take much longer to cool down after each run. One way of cooling the initial temperature is by using cryogenic cooling. Kristiana et al. (2012) used a cryogenic unit with liquid CO_2 to optimize their HS-SPME-GC-MS method by testing different initial temperatures (0, 10 and 35 °C) on the same column – ZB-5 Phenomenex® column (30 m × 0.25 mm ID, 1 µm film thickness) – as in our study. They reached Gaussian peak shapes for all HANs at 0 °C initial temperature. Another possibility of improving separation of these two peaks is by switching to a thicker column which would give better results for early eluters (but which may increase run time and carry-over for late eluters).

6. Attempts to analyze haloacetamides

HAcAms have similar stability to HANs and they can hydrolyse to their corresponding HAAs via base catalysis in neutral and basic conditions (Chu et al. 2010). A comparison on the stability of DCAcAm and TCAcAm at pH 4, 5, 6 and 7 showed they were the most stable at pH 5, for up to seven days (Chu 2009). Therefore water samples need to be acidified at the time of sampling to a pH between 5 and 6 to prevent the base-catalysed hydrolysis of HAcAms (Chu et al. 2010). Hence, a pH of 5.5 was chosen for the following experiments in mQ water.

The GC oven conditions were the following: 40 °C for 11 min, 25 °C/min to 285 °C, hold 4 min.

6.1. Initial attempts by HS-trap GC-MS

Analysis of aqueous standards of HAcAms was tested in the presence of Na_2SO_4, but no peaks were detected. Small aliquots (50-150 µL) of polar volatile solvents (MeOH, MTBE, ethyl acetate) were then also added to try and enhance the volatility of HAcAms but no peaks were detected either.

Hence traditional equilibrium headspace was abandoned for full evaporation headspace.

6.2. Full evaporation technique

6.2.1. Aqueous matrix

10 µL of a 1 ppm HAcAm sample in mQ water was injected in a HS vial and fully evaporated. All 4 HAcAms tested were detectable, but detection limits were high (no peaks detectable at 10 µg L^{-1}). Thus, analytes required concentration before analysis.

Some tailing was visible, especially for the first two compounds, probably due to their high polarity as the GC column is nearly non-polar. It is difficult to avoid some degree of tailing for polar peaks that strongly interact with surface imperfection.

6.2.2. Liquid-liquid extraction in ethyl acetate

Ethyl acetate LLE was carried in the presence of Na_2SO_4. The latest eluting compounds BAcAm, DCAcAm and TCAcAm were detectable with LOQ < 10 µg L^{-1}. CAcAm on the other hand is not detectable, probably because the low quantifying and confirming ions are drowned by ethyl acetate.

6.2.3. Initial oven temperature variation

Increasing the temperature from 40 to 60 °C meant peaks eluted faster, reducing the run time, but the signal decreased. The better results at 40 °C, can be attributed to ethyl acetate focussing. Without cryogenic focussing, it is not possible to go much lower than 40 °C, because of the lab temperature and slower oven cooling time at low temperature, which greatly increases run time.

7. Discussion: Disinfection by-product stability issues leading to uncertainties in measurements

7.1. Choice of quenching agents

7.1.1. For haloacetamides

(Chu et al. 2010, 2013) have investigated the effects of four quenching agents: sodium thiosulfate, sodium sulfite, ammonium chloride, and ascorbic acid, on the stabilities of HAcAms. Ammonium chloride was the best at preventing the decay of DCAcAm and TCAcAm. And while ascorbic acid showed a better capacity to prevent the decay of DCAcAm and TCAcAm than sodium thiosulfate and sodium sulfite (Chu et al. 2010), a relatively large (>> stoichiometric) amount of all three quenching agents all degraded HAcAm compounds to some degree, especially the bromine-containing HAcAms (Chu et al., 2013).

7.1.2. For haloacetonitriles

Ammonium chloride is recommended in USEPA method 551.1 as a dechlorination agent when analysing HANs (Hodgeson et al. 1995). While ammonium chloride showed no adverse effect on the stability of HANs, it is considered to be a non-ideal quenching agent, only suitable for samples containing a free chlorine residual (Kristiana et al. 2014). Ascorbic acid does not significantly affect the stability of HANs (Urbansky 1999; Kristiana et al. 2014), while sodium sulfite and sodium thiosulfate have been found to degrade HANs (Nikolaou et al. 2000; Kristiana et al. 2014).

7.1.3. For trihalomethanes

Ascorbic acid and sodium thiosulfate were found to be suitable quenching agents for all ten THMs, while sodium sulfite degrades $CHClI_2$, $CHBrI_2$ and CHI_3 in raw water (Cancho et al. 2000).

7.1.4. Ascorbic acid issue for haloacetic acid methylation

A study investigated the applicability of five quenching agents: sodium sulfite, sodium arsenite, sodium borohydride, ascorbic acid and ammonium chloride, for the analysis of seven different categories of chlorinated and brominated DBPs (including THMs, HAAs, and HANs) in drinking water. Ascorbic acid was recommended for the analysis of most organic chlorinated and brominated DBPs while sodium sulfite was suitable for the analysis of inorganic DBPs (Kristiana et al. 2014).

However ascorbic acid can also react with DMS in our method thus the pH has to be adjusted (as the pK_a for ascorbic acid is of 4.76 and higher than the pK_as for HAAs (0.03 -3.18). If HAA methyl esters are too unstable in acidic aqueous media, ascorbic acid may not be the right quenching agent.

US EPA methods use ammonium chloride forming chloramines, which are postulated to preserve HAAs from microbial degradation. However small amounts of HAAs continue being formed during sample storage (Pepich et al. 2004).

Table III-6: Summary of recommended quenching agents for different DBP classes.

DBP class	Recommended quenching agent
Trihalomethanes	Ascorbic acid, sodium thiosulfate, sodium arsenite
Haloacetic acids	Ascorbic acid, sodium thiosulfate
Haloacetonitriles	Ascorbic acid, ammonium chloride
Haloacetamides	Ammonium chloride, ascorbic acid

7.2. Thermodegradation and trihalomethane overestimation during analysis

As seen previously, while THMs are stable in drinking water for a wide pH range and at ambient temperatures, several classes of compounds, which are less stable can degrade and form THMs. The major interferences in the case of Headspace GC-MS analysis of THMs come from HAAs and HANs Takahashi et al. (2003).

7.2.1. Haloacetic acid degradation to trihalomethanes by the headspace conditions

Several studies have reported that HAA hydrolysis via decarboxylation will lead to the formation of carbon dioxide and their corresponding THMs (Zhang and Minear 2002; Takahashi et al. 2003; Cardador et al. 2008; Lifongo et al. 2010; Cardador et al. 2015). Decarboxylation of haloacetic acids is in the order tri > di > mono- substituted acids (Lifongo et al. 2010). The reactions are first-order, temperature dependent, and independent of the initial HAA concentration (Zhang and Minear 2002; Lifongo et al. 2010), with rate constants increasing exponentially from 4 °C to 50 °C for brominated tri-HAAs (Zhang and Minear 2002).

During GC analysis, HAA breakdown is what leads to lower formation potentials for HAAs and overestimation of THM concentrations (Alexandrou et al. 2018). Takahashi et al. (2003) studied the effect of heating conditions on the formation of THMs from THAAs during straight-forward HS-GC-MS of water (without any kind of derivatisation nor solvent addition). After 30 minutes at 60 °C, 2% TCAA, 12% DBCAA, 21% BDCAA and 100% TBAA degraded to their respective THMs. Further tests showed neither pH, nor heating time had any influence on Tri-HAA degradation when heating at 60 °C.

7.2.2. Degradation of other compound by the GC-MS conditions

Other volatile DBPs may be released from the solution during the extraction (Alexandrou et al. 2017). Bromopicrin forms bromoform at GC injection temperatures above 170 °C. Other trihalonitro compounds, such as dibromochloronitromethane, bromodichloronitromethane, and chloropicrin, also decompose extensively (about 50%) at an injection port temperature of 250 °C, forming several products including the other THM4.

Some degradation of these trihalonitro compounds can also occur in the transfer line, especially when at or above 250 °C (Chen et al. 2002).

7.3. Conclusion

Temperature and pH are important for the hydrolysis kinetics of THMs, HAAs, HANs and HAcAms. To preserve sample stability and avoid degradation of HAAs, HANs and HAcAms, it is necessary to keep samples in cold and in slightly acidic conditions (pH 5-6) before and during thermostatting.

High inlet temperatures (200-250 °C) may lead to overestimation of THMs, while lower temperatures will increase carry-over of the less volatile compounds such as the heaviest I-THMs.

8. Possible future improvements, using a programmable temperature vaporizing inlet or a cryogenic cooling system

8.1. Possible installation

The cryofocusing option can be ordered for the Turbomatrix HS 40, a water adsorption trap is required to remove water from the sample and prevent ice formation and blockage of the GC column. However, the cryofocusing option is for the use with HS only (without trap). Nevertheless, it allows more HS gas volume and thus up to 50 times lower detection limit for static HS without trap ('TurboMatrix Headspace Sampler and HS 40/110 Trap User's Guide' 2008). Increasing the sensitivity would allow the use of lower thermostatting temperature and hence reduce the risk of (or actual) thermodegradation of sensitive compounds compared to the use of the trap (Kolb and Ettre 2006b).

8.2. Possible improvements

8.2.1. For better separation in the column

A low initial temperature and a suitable GC oven ramp program would then be applicable to improve separation of early eluters such as chloroform and HANs and suppress band broadening.

8.2.2. For potential use of n-pentane to analyse haloacetic acids

a) By focusing and removing the excess n-pentane,
b) The presence of n-pentane will stop the time-related methyl ester hydrolysis,
c) Thus reducing the sample preparation time by allowing the preparation of all samples together instead of every 36.7 minutes, making it more user-friendly,
d) And also enabling better analysis and less thermal degradation of brominated tri-HAAs,
e) Therefore potentially being able to simultaneously analyse THMs and HAAs in one single run.

9. Conclusion

Two HS-trap GC-MS methods were successfully developed for the analysis of 10 THMs and 7 HAAs. These methods were then implemented for 2 studies on a drinking water treatment plant (Chapter IV and Chapter V).

The method for THMs is generally more sensitive than prior methods while using conditions, which do not lead to overestimation of the THM4 concentrations.

The method for HAAs does not allow the analysis of the more thermodegradable brominated tri-HAAs.

HANs elute early which leads to poor peak shapes and drowning of the signal. However, this can be improved in a future GC setup, with a new multimode injector (MMI – Multimode Inlet – Agilent).

10. Bibliography

Alexandrou et al. 2018. L. D. Alexandrou, B. J. Meehan, and O. A. H. Jones. '**Regulated and Emerging Disinfection By-Products in Recycled Waters**'. Science of The Total Environment, 637–638 (2018), pp. 1607-1616. *https://doi.org/10.1016/j.scitotenv.2018.04.391*.

Alexandrou et al. 2018. L. D. Alexandrou, B. J. Meehan, P. D. Morrison, and O. A. H. Jones. '**A New Method for the Fast Analysis of Trihalomethanes in Tap and Recycled Waters Using Headspace Gas Chromatography with Micro-Electron Capture Detection**'. International Journal of Environmental Research and Public Health, 14 (5) (2018), 9 p. *https://doi.org/10.3390/ijerph14050527*.

Allard et al. 2012. A. Sebastien, J. W.A. Charrois, C. A. Joll, and A. Heitz. '**Simultaneous Analysis of 10 Trihalomethanes at Nanogram per Liter Levels in Water Using Solid-Phase Microextraction and Gas Chromatography Mass-Spectrometry**'. Journal of Chromatography A, 1238 (2012), pp. 15-21. *https://doi.org/10.1016/j.chroma.2012.03.020*.

Allonier 2000. A. Allonier. '**Determination of Trihalomethanes in Chlorinated Sea Water Samples Using a Purge-and-Trap System Coupled to Gas Chromatography**'. Talanta, 51 (3) (2000), pp. 467-477. *https://doi.org/10.1016/S0039-9140(99)00296-9*.

Antoniou et al. 2006. C. V. Antoniou, E. E. Koukouraki, and E. Diamadopoulos. '**Determination of Chlorinated Volatile Organic Compounds in Water and Municipal Wastewater Using Headspace–Solid Phase Microextraction–Gas Chromatography**'. Journal of Chromatography A, 1132 (1–2) (2006), pp. 310-314. *https://doi.org/10.1016/j.chroma.2006.08.082*.

Bahri and Driss 2010. M. Bahri, and M.R. Driss. '**Development of Solid-Phase Microextraction for the Determination of Trihalomethanes in Drinking Water from Bizerte, Tunisia**'. Desalination, 250 (1) (2010), pp. 414-417. *https://doi.org/10.1016/j.desal.2009.09.067*.

Bharmoria et al. 2014. P. Bharmoria, P. Singh Gehlot, H. Gupta, and A. Kumar. '**Temperature-Dependent Solubility Transition of Na_2SO_4 in Water and the Effect of NaCl Therein: Solution Structures and Salt Water Dynamics**'. The Journal of Physical Chemistry B, 118 (44) (2014), pp. 12734-12742. *https://doi.org/10.1021/jp507949h*.

'Bulletin 916 Purge-and-Trap System Guide' 2019. Helpful guides. Merk (2019). *https://www.sigmaaldrich.com/analytical-chromatography/literature/guides.html*.

Campillo et al. 2004. N. Campillo, P. Viñas, I. López-García, N. Aguinaga, and M. Hernández-Córdoba. '**Purge-and-Trap Capillary Gas Chromatography with Atomic Emission Detection for Volatile Halogenated Organic Compounds Determination in Waters and Beverages**'. Journal of Chromatography A, 1035 (1) (2004), pp. 1-8. *https://doi.org/10.1016/j.chroma.2004.02.029*.

Cancho et al. 2000. B. Cancho, F. Ventura, M. T. Galceran, A. Diaz, and S. Ricart. '**Determination, Synthesis and Survey of Iodinated Trihalomethanes in Water Treatment Processes**'. Water Research, 34 (13) (2000), pp. 3380-3390. *https://doi.org/10.1016/S0043-1354(00)00079-8*.

Cardador and Gallego 2010. M. J. Cardador and M. Gallego. '**Optimisation and Comparison of Several Microextraction/Methylation Methods for Determining Haloacetic Acids in Water Using Gas Chromatography**'. Analytical and Bioanalytical Chemistry, 396 (3) (2010), pp. 1331-1343. *https://doi.org/10.1007/s00216-009-3281-z*.

Cardador et al. 2008. M. J. Cardador, A. Serrano, and M. Gallego. '**Simultaneous Liquid–Liquid Microextraction/Methylation for the Determination of Haloacetic Acids in Drinking Waters by Headspace Gas Chromatography**'. Journal of Chromatography A, 1209 (1-2) (2008), pp. 61-69. *https://doi.org/10.1016/j.chroma.2008.09.033*.

Cardador et al. 2015. M. J. Cardador, J. Fernández-Salguero, and M. Gallego. '**Simultaneous Quantification of Trihalomethanes and Haloacetic Acids in Cheese by On-Line Static Headspace Gas Chromatography – Mass-Spectrometry**'. Journal of Chromatography A, 1408 (21) (2015), pp. 22-29. *https://doi.org/10.1016/j.chroma.2015.07.007*.

Cardador and Gallego 2012. M. J. Cardador and M. Gallego. '**Effect of the Chlorinated Washing of Minimally Processed Vegetables on the Generation of Haloacetic Acids**'. Journal of Agricultural and Food Chemistry, 60 (29) (2012), pp. 7326-7332. *https://doi.org/10.1021/jf302591u*.

Cardador and Gallego 2015. M. J. Cardador and M. Gallego. '**Haloacetic Acids Content of Fruit Juices and Soft Drinks**'. Food Chemistry, 173 (2015), pp. 685-693. *https://doi.org/10.1016/j.foodchem.2014.10.105*.

Cardador and Gallego 2016a. M. J. Cardador and M. Gallego. '**Origin of Haloacetic Acids in Milk and Dairy Products**'. Food Chemistry, 196 (2016), pp. 750-756. *https://doi.org/10.1016/j.foodchem.2015.10.011*.

Cardador and Gallego 2016b. M. J. Cardador and M. Gallego. '**Static Headspace–Gas Chromatography–Mass Spectrometry for the Simultaneous Determination of Trihalomethanes and Haloacetic Acids in Canned Vegetables**'. Journal of Chromatography A, 1454 (2016), pp. 9-14. *https://doi.org/10.1016/j.chroma.2016.05.080*.

Cardador and Gallego 2017. M. J. Cardador and M. Gallego. '**Simultaneous Determination of 14 Disinfection By-Products in Meat Products Using Microwave-Assisted Extraction and Static Headspace Coupled to Gas Chromatography–Mass Spectrometry**'. Journal of Chromatography A, 1509 (2017), pp. 9-15. *https://doi.org/10.1016/j.chroma.2017.06.028*.

Cardador et al. 2016. M. J. Cardador, M. Gallego, L. Cabezas, and J. Fernández-Salguero. '**Detection of Regulated Disinfection By-Products in Cheeses**'. Food Chemistry, 204 (2016), pp. 306-313. *https://doi.org/10.1016/j.foodchem.2016.02.146*.

Cardinali et al. 2004. F. L. Cardinali, D. L. Ashley, J. C. Morrow, D. M. Moll, and B. C. Blount. '**Measurement of Trihalomethanes and Methyl Tertiary-Butyl Ether in Tap Water Using Solid-Phase Microextraction GC-MS**'. Journal of Chromatographic Science, 42 (4) (2004), pp. 200-206. *https://doi.org/10.1093/chromsci/42.4.200*.

Caro et al. 2007. J. Caro, A. Serrano, and M. Gallego. '**Direct Screening and Confirmation of Priority Volatile Organic Pollutants in Drinking Water**'. Journal of Chromatography A, 1138 (1–2) (2007), pp. 244-250. *https://doi.org/10.1016/j.chroma.2006.10.054*.

Chai et al. 2003. X-S. Chai, Q. X. Hou, J. Y. Zhu, S-L. Chen, S. F. Wang, and L. Lucia. '**Carboxyl Groups in Wood Fibers. 1. Determination of Carboxyl Groups by Headspace Gas Chromatography**'. Industrial & Engineering Chemistry Research, 42 (22) (2003), pp. 5440-5444. *https://doi.org/10.1021/ie020972a*.

Chary and Fernandez-Alba 2012. N. S. Chary and A. R. Fernandez-Alba. '**Determination of Volatile Organic Compounds in Drinking and Environmental Waters**'. TrAC Trends in Analytical Chemistry, 32 (2012), pp. 60-75. *https://doi.org/10.1016/j.trac.2011.08.011*.

Chen et al. 2002. P. H. Chen, S. D. Richardson, S. W. Krasner, G. Majetich, and G. L. Glish. '**Hydrogen Abstraction and Decomposition of Bromopicrin and Other Trihalogenated Disinfection Byproducts by GC/MS**'. Environmental Science & Technology, 36 (15) (2002), pp. 3362-3371. *https://doi.org/10.1021/es0205582*.

Chen and Her 2001. T-C. Chen and G-R. Her. '**On-Line Monitoring of Trihalomethanes in Drinking Water Using Continuous-Flow Purge and Cryofocusing Gas Chromatography–Mass Spectrometry**'. Journal of Chromatography A, 927 (1–2) (2001), pp. 229-235. *https://doi.org/10.1016/S0021-9673(01)01072-X*.

Cho et al. 2003. D-H. Cho, S-H. Kong, and S-G. Oh. '**Analysis of Trihalomethanes in Drinking Water Using Headspace-SPME Technique with Gas Chromatography**'. Water Research, 37 (2), pp. 402-408. *https://doi.org/10.1016/S0043-1354(02)00285-3*.

Christensen et al. 1988. J. M. Christensen, K. Rasmussen, and B. Køppen. '**Automatic headspace gas chromatographic method for the simultaneous determination of trichloroethylene and metabolites blood and urine**'. Journal of Chromatography, 442 (1998), pp. 317-323. *https://doi.org/10.1016/S0021-9673(00)94479-0*.

Chu et al. 2009. W. Chu, N. Gao, and Y. Deng. '**Stability of Newfound Nitrogenous Disinfection By-Products Haloacetamides in Drinking Water**'. Chinese Journal of Organic Chemistry, 29 (2009), pp. 1569-1574.

Chu et al. 2010a. W. Chu, N. Gao, and Y. Deng. '**Formation of Haloacetamides during Chlorination of Dissolved Organic Nitrogen Aspartic Acid**'. Journal of Hazardous Materials, 173 (1) (2010), pp. 82-86. *https://doi.org/10.1016/j.jhazmat.2009.08.051*.

Chu et al. 2010b. W. Chu, N. Gao, Y. Deng, and S. W. Krasner. '**Precursors of Dichloroacetamide, an Emerging Nitrogenous DBP Formed during Chlorination or Chloramination**'. Environmental Science & Technology, 44 (10) (2010), pp. 3908-3912. *https://doi.org/10.1021/es100397x*.

Chu et al. 2013. W. Chu, N. Gao, D. Yin, and S. W. Krasner. '**Formation and Speciation of Nine Haloacetamides, an Emerging Class of Nitrogenous DBPs, during Chlorination or Chloramination**'. Journal of Hazardous Materials, 260 (2013), pp. 806-812. *https://doi.org/10.1016/j.jhazmat.2013.06.044*.

Culea et al. 2006. M. Culea, O. Cozar, and D. Ristoiu. '**Methods Validation for the Determination of Trihalomethanes in Drinking Water**'. Journal of Mass Spectrometry, 41 (12) (2006), pp. 1594-1597. *https://doi.org/10.1002/jms.1149*.

Delgado et al. 2014. B. Delgado, P. Rocío-Bautista, V. Pino, and J. H. Ayala. '**Analytical Methods to Monitor Haloacetic Acids in Waters Intended for Human Consumption**'. In *Drinking Water and Water Management: New Research*, (2014), pp. 105-131. Nova Science Publishers, Inc. Mason Giannino.

Ekdahl and Abrahamsson 1997. A. Ekdahl and K. Abrahamsson. '**A Simple and Sensitive Method for the Determination of Volatile Halogenated Organic Compounds in Sea Water in the amol L^{-1} to pmol L^{-1} Range**'. Analytica Chimica Acta, 357 (1997), pp. 197-209. *https://doi.org/10.1016/S0003-2670(97)00566-7*.

EPA 2016. EPA. 'Six-Year Review 3 Technical Support Document for Disinfectants/Disinfection Byproducts Rules', (2016), 468 p.

Falaki 2019. F. Falaki. '**Sample Preparation Techniques for Gas Chromatography**'. In *Gas Chromatography - Derivatization, Sample Preparation, Application*, P. Kusch, IntechOpen (2019). *https://doi.org/10.5772/intechopen.84259*.

Gallard and von Gunten 2002. H. Gallard and U. von Gunten. '**Chlorination of Natural Organic Matter: Kinetics of Chlorination and of THM Formation**'. Water Research, 36 (1) (2002), pp. 65-74. *https://doi.org/10.1016/S0043-1354(01)00187-7*.

Ghoochani et al. 2013. M. Ghoochani, N. Rastkari, R. Nabizadeh Nodehi, A. Mahvi, S. Nasseri, and S. Nazmara. '**Study on the TOC Concentration in Raw Water and HAAs in Tehran's Water Treatment Plant Outlet**'. Journal of Environmental Health Science and Engineering, 11 (28) (2013), 6 p. *https://doi.org/10.1186/2052-336X-11-28*.

Glezer et al. 1999. V. Glezer, B. Harris, N. Tal, B. Iosefzon, and O. Lev. '**Hydrolysis of Haloacetonitriles: LINEAR FREE ENERGY RELATIONSHIP, Kinetics and Products**'. Water Research, 33 (8) (1999), pp. 1938-1948. *https://doi.org/10.1016/S0043-1354(98)00361-3*.

Golfinopoulos et al. 2001. S. K. Golfinopoulos, T. D. Lekkas, and A. D. Nikolaou. '**Comparison of Methods for Determination of Volatile Organic Compounds in Drinking Water**'. Chemosphere, 45 (3) (2001), pp. 275-284. *https://doi.org/10.1016/S0045-6535(00)00553-1*.

Hammami and Driss 2013. B. Hammami and M. R. Driss. '**Development of Dry Derivatization and Headspace Solid-Phase Microextraction Technique for the GC-ECD Determination of Haloacetic Acids in Tap Water**'. Journal of Analytical Chemistry, 68 (8) (2013), pp. 671-679. *https://doi.org/10.1134/S1061934813080054*.

Hodgeson et al. 1995. J. W. Hodgeson, A. L. Cohen, and D. J. Munch. '**EPA Method 551.1: Determination of Chlorination Disinfection Byproducts, Chlorinated Solvents, and Halogenated Pesticides/Herbicides in Drinking Water by Liquid-Liquid Extraction and Gas Chromatography With Electron-Capture Detection**'.

Ikem 2010. A. Ikem. '**Measurement of Volatile Organic Compounds in Bottled and Tap Waters by Purge and Trap GC–MS: Are Drinking Water Types Different?**' Journal of Food Composition and Analysis, 23 (1) (2010), pp. 70-77. *https://doi.org/10.1016/j.jfca.2009.05.005*.

Joll et al. 2010. C. A. Joll, M. J. Alessandrino, and A. Heitz. '**Disinfection By-Products from Halogenation of Aqueous Solutions of Terpenoids**'. Water Research, 44 (1) (2010), pp. 232-242. *https://doi.org/10.1016/j.watres.2009.09.003*.

Kim et al. 2012. T-S. Kim, H. Suk-Young, Jong-Eun Kim, Hyun-Hee Lim, and Ho-Sang Shin. '**Simultaneous Determination of 37 Volatile Organic Compounds at ng/L Concentration Level in Surface Water by HS-SPME-GC/MS**'. Analytical Science and Technology, 25 (1) (2012), pp. 39-49. *https://doi.org/10.5806/AST.2012.25.1.039*.

Kinani et al. 2016. A. Kinani, S. Kinani, and S. Bouchonnet. '**Formation and Determination of Organohalogen By-Products in Water. Part III. Characterization and Quantitative Approaches**'. TrAC Trends in Analytical Chemistry, 85 (2016), pp. 295-305. *https://doi.org/10.1016/j.trac.2016.09.013*.

Kolb and Ettre 2006. B. Kolb and L. S. Ettre. 2006~~a~~. '**Static Headspace–Gas Chromatography: Theory and Practice**'. John Wiley & Sons, Inc.

Koning et al. 2009. S. Koning, H-G Janssen, and U. A. Th. Brinkman. '**Modern Methods of Sample Preparation for GC Analysis**'. Chromatographia, 69 (S1) (2009), pp. 33-78. *https://doi.org/10.1365/s10337-008-0937-3*.

Krasner et al. 2006. S. W. Krasner, H. S. Weinberg, S. D. Richardson, S. J. Pastor, R. Chinn, M. J. Sclimenti, G. D. Onstad, and A. D. Thruston. '**Occurrence of a New Generation of Disinfection Byproducts**'. Environmental Science & Technology, 40 (23) (2006), pp. 7175-7185. *https://doi.org/10.1021/es060353j*.

Kristiana et al. 2012~~a~~. I. Kristiana, Ina, C. Joll, and A. Heitz. '**Analysis of Halonitriles in Drinking Water Using Solid-Phase Microextraction and Gas Chromatography–Mass Spectrometry**'. Journal of Chromatography A, 1225 (2012), pp. 45-54. *https://doi.org/10.1016/j.chroma.2012.01.005*.

Kristiana et al. 2014. I. Kristiana, A. Lethorn, C. Joll, and A. Heitz. '**To Add or Not to Add: The Use of Quenching Agents for the Analysis of Disinfection by-Products in Water Samples**'. Water Research, 59 (2014), pp. 90-98. *https://doi.org/10.1016/j.watres.2014.04.006*.

Kuivinen and Johnsson 1999. J. Kuivinen and H. Johnsson. '**Determination of Trihalomethanes and Some Chlorinated Solvents in Drinking Water by Headspace Technique with Capillary Column Gas-Chromatography**'. Water Research, 33 (5) (1999), pp. 1201-1208. *https://doi.org/10.1016/S0043-1354(98)00311-X*.

Kusch 2018. P. Kusch. '**Introductory Chapter: Gas Chromatography - The Most Versatile Analytical Technique**'. In *Gas Chromatography - Derivatization, Sample Preparation, Application*. IntechOpen (2018). *https://doi.org/10.5772/intechopen.81693*.

Lambropoulou 2013. D. A. Lambropoulou. '**Microextraction Techniques Coupled to Advanced GC–MS Techniques for Analysis of Environmental Samples**'. Comprehensive Analytical Chemistry, 61 (2013) , pp. 23-54. *https://doi.org/10.1016/B978-0-444-62623-3.00002-2*.

Lara-Gonzalo et al. 2008. A. Lara-Gonzalo, J. Enrique Sánchez-Uría, E. Segovia-García, and A. Sanz-Medel. '**Critical Comparison of Automated Purge and Trap and Solid-Phase Microextraction for Routine Determination of Volatile Organic Compounds in Drinking Waters by GC–MS**'. Talanta, 74 (5) (2008), pp. 1455-1462. *https://doi.org/10.1016/j.talanta.2007.09.036*.

Lee et al. 2001. K-J. Lee, H. Pyo, S-J. Park, E-A. Yoo, and D-W. Lee. 'A **Study on Purge Efficiency in Purge and Trap Analysis of VOCs in Water**'. Bulletin- Korean Chemical Society, 22 (2) (2001), pp. 171-178.

Liew et al. 2012. D. Liew, K. L. Linge, C. A. Joll, A. Heitz, and J. W. A. Charrois. '**Determination of Halonitromethanes and Haloacetamides: An Evaluation of Sample Preservation and Analyte Stability in Drinking Water**'. Journal of Chromatography A, 1241 (2012), pp. 117-122. *https://doi.org/10.1016/j.chroma.2012.04.037*.

Lifongo et al. 2010. L. L. Lifongo, D. J. Bowden, and P. Brimblecombe. '**Thermal Degradation of Haloacetic Acids in Water**'. International Journal of Physical Sciences, 5 (6) (2010), pp. 738-747.

Luo et al. 2014. Q. Luo, X. Chen, Z. Wei, X. Xu, D. Wang, and Z. Wang. '**Simultaneous and High-Throughput Analysis of Iodo-Trihalomethanes, Haloacetonitriles, and Halonitromethanes in Drinking Water Using Solid-Phase Microextraction/Gas Chromatography-Mass Spectrometry: An Optimization of Sample Preparation**'. Journal of Chromatography A, 1365 (2014), pp. 45-53. *https://doi.org/10.1016/j.chroma.2014.09.003*.

Ma and Chiang 2005. Y-C. Ma, and C-Y. Chiang. '**Evaluation of the Effects of Various Gas Chromatographic Parameters on Haloacetic Acids Disinfection By-Products Analysis**'. Journal of Chromatography A, 1076 (1) (2005), pp. 216-219. *https://doi.org/10.1016/j.chroma.2005.03.129*.

Martínez et al. 2002. E. Martínez, S. Lacorte, I. Llobet, P. Viana, and D. Barceló. '**Multicomponent Analysis of Volatile Organic Compounds in Water by Automated Purge and Trap Coupled to Gas Chromatography–Mass Spectrometry**'. Journal of Chromatography A, 959 (1–2) (2002), pp. 181-190. *https://doi.org/10.1016/S0021-9673(02)00439-9*.

Montesinos and Gallego 2013~~a~~. I. Montesinos and M. Gallego. '**Speciation of Common Volatile Halogenated Disinfection By-Products in Tap Water under Different Oxidising Agents**'. Journal of Chromatography A, 1310 (2013), pp. 113-120. *https://doi.org/10.1016/j.chroma.2013.08.036*.

Montesinos and Gallego 2014. I. Montesinos and M. Gallego. '**How the Inclusion of Treated Water in Beverages Influences the Appearance of Halogenated Volatile Organic Compounds**'. Journal of Agricultural and Food Chemistry, 62 (42) (2014), pp. 10240-10247. *https://doi.org/10.1021/jf503431q*.

Muralidhara and Bruckner 1999. S. Muralidhara and J. V. Bruckner. '**Simple Method for Rapid Measurement of Trichloroethylene and Its Major Metabolites in Biological Samples**'. Journal of Chromatography B: Biomedical Sciences and Applications, 732 (1) (1999), pp. 145-153. *https://doi.org/10.1016/S0378-4347(99)00282-0*.

Nakamura and Daishima 2005. S. Nakamura and S. Daishima. '**Simultaneous Determination of 22 Volatile Organic Compounds, Methyl-Tert-Butyl Ether, 1,4-Dioxane, 2-Methylisoborneol and Geosmin in Water by Headspace Solid Phase Microextraction-Gas Chromatography–Mass Spectrometry**'. Analytica Chimica Acta, 548 (1–2) (2005), pp. 79-85. *https://doi.org/10.1016/j.aca.2005.05.077*.

Neitzel et al. 1998~~a~~. P. L. Neitzel, W. Walther, and W. Nestler. '**In-Situ Methylation of Strongly Polar Organic Acids in Natural Waters Supported by Ion-Pairing Agents for Headspace GC-MSD Analysis**'. Fresenius' Journal of Analytical Chemistry, 361 (3) (1998), pp. 318-323. *https://doi.org/10.1007/s002160050893*.

Nikolaou 2002. A. Nikolaou. '**Application of Different Analytical Methods for Determination of Volatile Chlorination By-Products in Drinking Water**'. Talanta, 56 (4) (2002), pp. 717-726. *https://doi.org/10.1016/S0039-9140(01)00613-0*.

Nikoalou et al. 2000. A. D. Nikolaou, S. K. Golfinopoulos, M. N. Kostopoulou, and T. D. Lekkas. '**Decomposition of Dihaloacetonitriles in Water Solutions and Fortified Drinking Water Samples**'. Chemosphere, 41 (8) (2000), pp. 1149-1154. *https://doi.org/10.1016/S0045-6535(00)00025-4*.

Niri et al. 2008. V. H. Niri, L. Bragg, and J. Pawliszyn. '**Fast Analysis of Volatile Organic Compounds and Disinfection By-Products in Drinking Water Using Solid-Phase Microextraction–Gas Chromatography/Time-of-Flight Mass Spectrometry**'. Journal of Chromatography A, 1201 (2) (2008), pp. 222-227. *https://doi.org/10.1016/j.chroma.2008.03.062*.

Pepich et al. 2004. B. V. Pepich, M. M. Domino, T. A. Dattilio, P. S. Fair, and D. J. Munch. '**Validating Sample Preservation Techniques and Holding Times for the Approved Compliance Monitoring Methods for Haloacetic Acids under the US EPA's Stage 1 D/DBP Rule**'. Water Research, 38 (4) (2004), pp. 895-902. *https://doi.org/10.1016/j.watres.2003.09.040*.

Pérez Pavón et al. 2008a. J. L. Pérez Pavón, S. Herrero-Martín, C. García Pinto, and B. Moreno Cordero. '**Determination of Trihalomethanes in Water Samples: A Review**'. Analytica Chimica Acta, 629 (1–2) (2008), pp. 6-23. *https://doi.org/10.1016/j.aca.2008.09.042*.

Pérez Pavón et al. 2008b. J. L. Pérez Pavón, S. Herrero-Martín, C. García Pinto, and B. Moreno Cordero. '**Headspace–Programmed Temperature Vaporizer–Fast Gas Chromatography–Mass Spectrometry Coupling for the Determination of Trihalomethanes in Water**'. Journal of Chromatography A, 1194 (1) (2008), pp. 103-110. *https://doi.org/10.1016/j.chroma.2008.04.037*.

Plümacher and Renner 1993. J. Plümacher and I. Renner. '**Determination of Volatile Chlorinated Hydrocarbons and Trichloroacetic Acid in Conifer Needles by Headspace Gas-Chromatography**'. Fresenius' Journal of Analytical Chemistry, 347 (3–4) (1993), pp. 129-135. *https://doi.org/10.1007/BF00322844*.

Prakash et al. 2009. B. Prakash, A. D. Zaffiro, M. Zimmermann, D. J. Munch, and B. V. Pepich. '**Method 524.3: Measurement of Purgeable Organic Compounds in Water by Capillary Column Gas Chromatography/Mass Spectrometry**'. EPA Document # EPA 815-B-09-009 (2009).

Rasanen et al. 2010. I. Rasanen, J. Viinamaki, E. Vuori, and I. Ojanpera. '**Headspace In-Tube Extraction Gas Chromatography-Mass Spectrometry for the Analysis of Hydroxylic Methyl-Derivatized and Volatile Organic Compounds in Blood and Urine**'. Journal of Analytical Toxicology, 34 (3) (2010), pp. 113-121. *https://doi.org/10.1093/jat/34.3.113*.

Rastkari et al. 2012. N. Rastkari, R. Ahmadkhaniha, and M. Yunesian. '**Simultaneous Determination of Trichloroethylene, Perchloroethylene and Trichloroacetic Acid in Human Urine Using Solid-Phase Microextraction Fibre Coated with Single-Walled Carbon Nanotubes**'. International Journal of Environmental Analytical Chemistry, 92 (14) (2012), pp. 1650-1665. *https://doi.org/10.1080/03067319.2011.581367*.

Reeves et al. 2000. N. M. Reeves, M. R. Heal, and J. N. Cape. '**A New Method for the Determination of Trichloroacetic Acid in Spruce Foliage and Other Environmental Media**'. Journal of Environmental Monitoring, 2 (5) (2000), pp. 447-450. *https://doi.org/10.1039/b003247f*.

Rosero et al. 2012. M. M. Rosero, M. Aguirre, D. Pezo, G. Taborda, C. Dussán, and C. Nerin. '**Solventless Microextraction Techniques for Determination of Trihalomethanes by Gas Chromatography in Drinking Water**'. Water, Air, & Soil Pollution, 223 (2), pp. 667-678. *https://doi.org/10.1007/s11270-011-0891-9*.

Ross 2012. C. F. Ross. '**2.02 – Headspace Analysis**'. In *Comprehensive Sampling and Sample Preparation*, Analytical Techniques for Scientists, edited by J. Pawliszyn, 2 (2012), pp. 27-50.

Rubio and Pérez-Bendito 2009. S. Rubio and D. Pérez-Bendito. '**Recent Advances in Environmental Analysis**'. Analytical Chemistry, 81 (12) (2009), pp. 4601-4622. *https://doi.org/10.1021/ac900738s*.

Ruiz-Bevia et al. 2009. F. Ruiz-Bevia, M. J. Fernandez-Torres, and M. P. Blasco-Alemany. '**Purge Efficiency in the Determination of Trihalomethanes in Water by Purge-and-Trap Gas Chromatography**'. Analytica Chimica Acta, 632 (2) (2009), pp. 304-314. *https://doi.org/10.1016/j.aca.2008.11.022*.

Sá et al. 2012. C. S. A. Sá, R. A. R. Boaventura, and I. B. Pereira. '**Analysis of Haloacetic Acids in Water and Air (Aerosols) from Indoor Swimming Pools Using HS-SPME/GC/ECD**'. Journal of Environmental Science and Health, Part A, 47 (2) (2012), pp. 176-183. *https://doi.org/10.1080/10934529.2012.640246*.

Sá et al. 2011. C. S. A. Sá, R. A. R. Boaventura, and I. B. Pereira. '**Analysis of Trihalomethanes in Water and Air from Indoor Swimming Pools Using HS-SPME/GC/ECD**'. Journal of Environmental Science and Health, Part A, 46 (4) (2011), pp. 355-363. *https://doi.org/10.1080/10934529.2011.542385*.

Safarova et al. 2004. V. I. Safarova, S. V. Sapelnikova, E. V. Djazhenko, G. I. Teplova, G. F. Shajdulina, and F. Kh. Kudasheva. '**Gas Chromatography–Mass Spectrometry with Headspace for the Analysis of Volatile Organic Compounds in Waste Water**'. Journal of Chromatography B, 800 (1–2) (2004), pp. 325-330. *https://doi.org/10.1016/j.jchromb.2003.10.070*.

San Juan et al. 2007. P. M. San Juan, J. David Carrillo, and M. Teresa Tena. '**Fibre Selection Based on an Overall Analytical Feature Comparison for the Solid-Phase Microextraction of Trihalomethanes from Drinking Water**'. Journal of Chromatography A, 1139 (1) (2007), pp. 27-35. *https://doi.org/10.1016/j.chroma.2006.10.084*.

Sarrión et al. 2000. M. N. Sarrión, F. J. Santos, and M. T. Galceran. '**In Situ Derivatization/Solid-Phase Microextraction for the Determination of Haloacetic Acids in Water**'. Analytical Chemistry 72 (20) (2000), pp. 4865-4873. *https://doi.org/10.1021/ac000479d*.

Sarrión et al. 1999. M. N. Sarrión, F. J. Santos, and M. T. Galceran. '**Solid-Phase Microextraction Coupled with Gas Chromatography–Ion Trap Mass Spectrometry for the Analysis of Haloacetic Acids in Water**'. Journal of Chromatography A, 859 (2) (1999), pp. 159-171. *https://doi.org/10.1016/S0021-9673(99)00873-0*.

Stack et al. 2000. M. A. Stack, G. Fitzgerald, S. O'Connell, and K. J. James. '**Measurement of Trihalomethanes in Potable and Recreational Waters Using Solid Phase Micro Extraction with Gas Chromatography-Mass Spectrometry**'. Chemosphere, 41 (11) (2000), pp. 1821-1826. *https://doi.org/10.1016/S0045-6535(00)00047-3*.

Takahashi et al. 2003. Y. Takahashi, S. Onodera, M. Morita, and Y. Terao. '**A Problem in the Determination of Trihalomethane by Headspace-Gas Chromatography/Mass Spectrometry**' Journal of Health Science, 49 (1) (2003), pp. 1-7. *https://doi.org/10.1248/jhs.49.1*.

'TurboMatrix Headspace Sampler and HS 40/110 Trap User's Guide'. 2008. *PerkinElmer*, 357.

Urbansky 1999. E. T. Urbansky. '**Ascorbic Acid Treatment to Reduce Residual Halogen-Based Oxidants Prior to the Determination of Halogenated Disinfection Byproducts in Potable Water**'. Journal of Environmental Monitoring, 1 (1999), pp. 471-476. *https://doi.org/10.1039/A904574K*.

van Boxtel et al. 2015. N. van Boxtel, K. Wolfs, A. Van Schepdael, and E. Adams. '**Application of Acetone Acetals as Water Scavengers and Derivatization Agents Prior to the Gas Chromatographic Analysis of Polar Residual Solvents in Aqueous Samples**'. Journal of Chromatography A, 1425 (2015), pp. 62-72. *https://doi.org/10.1016/j.chroma.2015.11.020*.

Vikesland et al. 2007. P. J. Vikesland, N. G. Love, K. Chandran, E. Matthew Fiss, R. Rebodos, and A. E. Zaklikowski. '**Experimental Methods and Procedure**'. In *Seasonal Chlorination Practices and Impacts to Chloraminating Utilities*, 168 p. American Water Works Research Foundation.

Wang and Wong 2005. Y. H. Wang and P. K. Wong. 2005. '**Determination of Dichloroacetic Acid and Trichloroacetic Acid in Drinking Water by Acidic Methanol Esterification and Headspace Gas Chromatography**'. Water Research, 39 (9) (2005), pp. 1844-1848. *https://doi.org/10.1016/j.watres.2005.02.010*.

Waseem and Abdullah 2010. S. Waseem and M. P. Abdullah. '**Comparative study of sample preparation techniques coupled to GC for the analysis of halogenated acetic acids (HAAs) acids in tap water**'. Journal of Chromatographic Science, 48 (3) (2010), pp. 188-193. *https://doi.org/10.1093/chromsci/48.3.188*.

Weinberg et al. 2002. H. S. Weinberg, S. W. Krasner, S. D. Richardson, and A. D. Thruston. '**The Occurrence of Disinfection By-Products (DBPs) of Health Concern in Drinking Water: Results of a Nationwide DBP Occurrence Study**'. U.S. Environmental Protection Agency, National Exposure Research Laboratory: Athens, GA. EPA/600/R-02/068, (2002).

Wu et al. 2002. F. Wu, W. Gabryelski, and K. Froese. 2002. '**Improved Gas Chromatography Methods for Micro-Volume Analysis of Haloacetic Acids in Water and Biological Matrices**'. The Analyst, 127 (10) (2002), pp. 1318-1323. *https://doi.org/10.1039/b204574e*.

Xie et al. 2018. W-Q. Xie, Y-X. Gong, and K-X. Yu. 2018. '**Utilizing Two Detectors in the Measurement of Trichloroacetic Acid in Human Urine by Reaction Headspace Gas Chromatography**'. Biomedical Chromatography, 32 (10) (2018) e4288. *https://doi.org/10.1002/bmc.4288*.

Xie 2001. Y. Xie. '**Analyzing Haloacetic Acids Using Gas Chromatography/Mass Spectrometry**'. Water Research, 35 (6) (2001), pp. 1599-1602. *https://doi.org/10.1016/S0043-1354(00)00397-3*.

Yu and Reckhow 2015. Y. Yu and D. A. Reckhow. '**Kinetic Analysis of Haloacetonitrile Stability in Drinking Waters**'. Environmental Science & Technology, 49 (18) (2015), pp. 11028-11036. *https://doi.org/10.1021/acs.est.5b02772*.

Zhang and Minear 2002. X. Zhang and R. A Minear. '**Decomposition of Trihaloacetic Acids and Formation of the Corresponding Trihalomethanes in Drinking Water**'. Water Research, 36 (14) (2002), pp. 3665-3673. *https://doi.org/10.1016/S0043-1354(02)00072-6*.

Zocolillo et al. 2005. L. Zoccolillo, L. Amendola, C. Cafaro, and S. Insogna. '**Improved Analysis of Volatile Halogenated Hydrocarbons in Water by Purge-and-Trap with Gas Chromatography and Mass Spectrometric Detection**'. Journal of Chromatography A, 1077 (2) (2005), pp. 181-187. *https://doi.org/10.1016/j.chroma.2005.04.076*.

Zygmunt et al. 1996. B. Zygmunt. '**Determination of Trihalomethanes in Aqueous Samples by Means of a Purge-and-Trap System with on-Sorbent Focusing Coupled to Gas Chromatography with Electron-Capture Detection**'. Journal of Chromatography A, 7 (1) (1996), pp.157-163. https://doi.org/10.1016/0021-9673(95)00892-6.

IV. Formation and removal of disinfection by-products in a full scale drinking water treatment plant

IV. Formation and removal of disinfection by-products in a full scale drinking water treatment plant

Henry MacKeown[1], Junias Adusei Gyamfi[1,2], Klaas Schoutteten[3,4], David Dumoulin[1], Liesbeth Verdickt[4], Baghdad Ouddane[1], Justine Criquet[1]*

[1]Univ. Lille CNRS, UMR 8516 – LASIR, Equipe Physico-Chimie de l'Environnement, F-59000, Lille, France.

[2]Delft University of Technology, Faculty of Civil Engineering and Geosciences, Department of Water Management, PO Box 5048, GA Delft, 2600, The Netherlands.

[3]Ghent University, Faculty of Bioscience Engineering, Department of Green Chemistry and Technology, Coupure Links 653, 9000 Gent, Belgium.

[4]De Watergroep, Vooruitgangstraat 189, 1030 Brussels, Belgium.

*Corresponding author: justine.criquet@univ-lille.fr

Abstract:

In this case study, high sensitivity simple methods for the analysis of trihalomethanes (THM4), iodinated-trihalomethanes (I-THMs), haloacetic acids (HAAs), bromide, iodide and iodate have been developed. A one-step procedure for the analysis of haloacetic acids by headspace GC-MS provides good reproducibility and low limits of quantification (≤ 50 ng L^{-1}). These methods were applied to characterize the formation of DBPs in a full scale drinking water treatment plant. In this treatment plant, the incorporation of bromine into THMs increases throughout the water treatment line, due to the formation of bromine reactive species favored by the decrease of competition between DOC and bromide towards chlorine. A linear correlation has been observed between the bromine incorporation factor and the Br^-/DOC mass ratio. The conversion of iodine to iodate by chlorination occurs in this water due to the relatively high bromide concentration. Moreover, a higher formation of iodate compared to iodide levels in the raw water is observed indicating a degradation of organic iodinated compounds. The formation of I-THMs was constant in terms of quantity and speciation between campaigns despite fluctuating concentrations of DOC and total iodine in the raw water. A preferential removal of DBPs formed by the intermediate chlorination in the order I-DBPs>Br-DBPs>Cl-DBPs occurs during the subsequent activated carbon filtration. The removal rates range from 25 to 36% for the regulated THM4, from 82 to 93% for the ∑I-THMs and 95% for haloacetic acids. The assessment of the relative toxicity shows that despite a much lower concentration of HAAs (less than 10% of the total mass of measured DBPs) compared to THMs, these compounds are responsible for 75% of the relative cytotoxicity of the treated water. Bromoacetic acid on its own accounts for more than 60% of the overall toxicity of the 17 compounds included in this study.

Keywords:

DBPs, chlorination, activated carbon, trihalomethanes, haloacetic acids, natural organic matter.

1. Introduction

Chlorine is the main chemical disinfectant used in drinking water treatment, especially as a final step to maintain a residual concentration in the distribution network, avoiding bacterial regrowth. Pre-chlorination or intermediate chlorination is also used to oxidize iron or manganese or to avoid biofouling of filters. However, chlorine also reacts with natural organic matter (NOM) and halides to form a variety of disinfection by-products (DBPs) (Sedlak and von Gunten 2011). The removal of NOM is effective to control the DBP formation, however, literature shows that the efficiency of physical-chemical processes for NOM removal is highly variable and that the same treatment process can lead to different treatment efficiencies when applied to different water types (Bond et al. 2011). Coagulation can effectively remove the humic substances with a high hydrophobic character but will be much less effective for the hydrophilic fractions; then activated carbon could be useful for the removal of these remaining fractions (Bond et al. 2011). Activated carbon (GAC) is considered as an advanced treatment for NOM removal through adsorption and/or biodegradation (Yang et al. 2010; Chili et al. 2012; Fu et al. 2017) for the control of DBP formation. However, only few studies deal with the interest of GAC for the removal of DBPs preformed from pre-chlorination or intermediate chlorination (Tung et al. 2006; Kim and Kang 2008; Cuthbertson et al. 2019). Pre-chlorination has been shown as a detrimental process in terms of DBP formation. However, Cuthbertson et al. (2019) recently showed the reduction of the relative toxicity of the pre-chlorinated water by GAC.

The concentration and properties of aqueous NOM have a significant influence on the formation of DBPs during disinfection. The different reactive fractions of NOM can be present in a wide range of concentrations and preponderance within the same geographical area as for example the hydrophobic fraction representing from 30 to 84% of the raw water NOM in 30 resources across Scotland (Golea et al. 2017). In addition, the level of bromide and iodide in water plays an important role in DBP formation, inducing the formation of a mix of chlorinated, brominated and iodinated DBPs (Bichsel 2000; Jones et al. 2011; Hua et Reckhow 2012; Roccaro et al. 2014; Zhang et al. 2015; Allard et al. 2015). Bromide also plays a role in the speciation of iodine species by the enhancement of iodate formation, a safe end-product for iodide, hence reducing the risk of formation of I-DBPs during chlorination (Criquet et al. 2012). Considering the toxicity, iodinated DBPs (I-DBPs) are usually more cytotoxic and genotoxic than the brominated analogues which already are considered as more toxic than the chlorinated ones (Richardson et al. 2007). First identified in drinking waters in 1976 (Shackelford et Keith 1976), iodinated THMs (I-THMs) were the first I-DBPs studied due to early concerns about medicinal taste and odours in drinking water (Hansson et al. 1987) with the lowest odour threshold at 0.03 µg L^{-1} for iodoform (Cancho et al.

2001). I-THMs are the most important class of I-DBPs in terms of occurrence, sometimes even exceeding the THM4 (Richardson et al. 2008; Allard et al. 2015; Tugulea et al. 2015, 2018).

In this study, we investigated the removal of DBP precursors in a full scale water treatment plant in relation to bromine and iodine incorporation and DBP formation. A high sensitivity new method for the analysis of THMs and HAAs is described. An assessment of individual DBP removal by GAC has been carried out at higher levels of halides than what could be found in literature. This study is the first to report the effectiveness of I-THM removal by GAC in a full scale plant and to relate it to physical-chemical properties of DBPs. Finally, a comparison of theoretical relative toxicity from regulated and unregulated DBPs analyzed in this study is presented.

2. Material and Methods

2.1 Water resource, drinking water treatment plant and sample collection

The water samples used in this study were sourced from a drinking water treatment plant, located in the West of Flanders in Belgium and designed for a production of 40,000 m^3 day^{-1}. The raw water is drawn from the river IJzer (approximately 80%) and from surrounding lowlands (approximately 20%) (De Watergroep 2015), and stored in a 3 million m^3 reservoir. The water is characterized by a high NOM content, with an average DOC content of 12 mgC L^{-1}, combined with a high alkalinity and hardness. The Belgian part of the hydrographic basin of the IJzer river covers an area holding 110,000 inhabitants, and a largely agrarian activity, *i.e.* intensive farming and intensive cattle breeding (Baert et al. 1996). The water is treated by a conventional treatment train consisting of biological ammonia oxidation, enhanced coagulation with ferric chloride, decantation, rapid sand filtration, granular activated carbon (GAC) filtration and disinfection with sodium hypochlorite. A pH adjustment is performed before and after coagulation by adding sulfuric acid and sodium hydroxide respectively. An intermediate chlorination is performed before sand filtration in order to enhance manganese oxidation, which is then removed during sand filtration.

Samples were collected from the reservoir (raw water) and after each treatment step in the spring and summer of 2018. Three different campaigns were performed, two campaigns (A and B) for the determination of the THM formation potential (THMFP) and the characterization of precursors, and one (campaign C) for the determination of THMs and HAAs (Table IV-S1). Amber glass bottles of 2.5 L were used for the collection of samples to perform chlorination tests and for determination of initial DBP concentrations. A quenching agent (sodium thiosulfate for THMs (4.65 mg L^{-1}) and ascorbic acid (5 mg L^{-1}) for HAAs) was added in excess to the sand filtrate, GAC filtrate and final water, as these three water samples contain or potentially contain some residual free chlorine. Samples were filtered through 0.45 µm polyethersulfone (PES) membrane filters within a few hours after sampling. In addition, 30 mL samples were taken for dissolved organic carbon (DOC) and size exclusion chromatography (HPSEC) analysis. These were taken at the same sampling locations using a glass syringe, immediately filtered using 0.45 µm PES syringe filters (after discarding the first 5 mL of filtrate), and spiked with 150 µL of sodium azide (6.5 g L^{-1}). Finally, all samples were stored at 4 °C until the chlorination experiments or analysis.

The concentration of dissolved organic carbon was determined using a TOC analyzer (Shimadzu® VCSH). NOM size fractionation was performed using high performance liquid chromatography (Agilent® 1260 Infinity II) equipped with a size exclusion column (Agilent® Bio SEC-5 column, 100 Å; 300 mm x 7.8 mm, 5 µm particle size), with an exclusion limit range of 100 to approximatively 100,000 Da. A phosphate buffer eluent was used: 1 g L^{-1}; pH 6.8 and ionic strength of 0.039 M (Zhou et al. 2000; Song et al. 2010). Other parameters used during exclusion chromatography are an injection volume of 100 µL, a flow rate of 1.0 mL min^{-1}, a pressure of around 80 bar and a temperature of 23 °C. The HPSEC is coupled to a diode array detector equipped with a 60 mm high sensitivity cell. The eluting dissolved organic matter fractions were detected using UV absorbance at 254 nm. Samples with organic carbon concentrations exceeding 5 mgC L^{-1} were diluted before analysis to avoid fouling of the column and signal saturation.

2.2 Development of high sensitivity methods

A headspace-trap method has been developed for the analysis of the 10 THMs. The analytical instrument used was a Trace™ 1300 GC coupled with an ISQ-LT single quadrupole mass spectrometer (Thermo Scientific®) operating in SIM mode. A ZB-5MS (Phenomenex®) column (30 m × 0.25 mm ID, 1 µm film thickness) was used for gas chromatographic separations. The headspace trap injector (Perkin Elmer® TurboMatrix Headspace 40 Trap) was directly connected to the GC-MS bypassing the inlet split of the GC injector. For THMs analysis, 10 mL of the water samples were transferred into 22 mL glass vials containing 4 g of sodium sulfate (2.8 M), and were spiked with 1,2-dibromopropane as internal standard (6.4 µg L^{-1}). The vials were immediately sealed and stirred in a vortex mixer until complete salt dissolution. This method ensured low quantification limits for the 10 THMs included in this study, ranging from 0.4 ng L^{-1} ($CHBr_2I$) to 2 ng L^{-1} (CHI_3) for I-THMs and 3 ng L^{-1} for THM4 (Table IV-1).

Haloacetic acids (HAA5 + iodoacetic and bromochloroacetic acids) were analyzed by a one-step procedure. HAAs in aqueous medium forms an ion-pair with tetrabutylammonium hydrogensulfate and react with dimethylsulfate to produce methyl haloacetates which are directly volatized in the head-space. 9.84 mL of the water samples were spiked with bromopropionic acid (10 µg L^{-1}) as internal standard, acidified to pH 2.6 using sulfuric acid, and placed in a 22 mL glass vial containing 4 g

(2.8 M) of sodium sulfate. 100 µL of a 0.5 M ion pairing agent (tetrabutylammonium hydrogensulfate, final amount of 2.3 µmol) and 60 µL of derivatization reagent (dimethylsulfate) were then added. The vials were immediately sealed and stirred before being placed into an autosampler. The analysis was performed after 2 to 4 hours of reaction time, this time should not exceed 5 hours to avoid the hydrolysis of methylester. The limit of quantification ranged from 5 ng L^{-1} (iodoacetic acid) to 50 ng L^{-1} (chloroacetic acid) (Table IV-1). The Headspace-trap-GC-MS operating parameters are specified in the Supporting Information (Tables IV-S2 to IV-S4).

Table IV-1: Disinfection by-products analyzed, formulae and limits of quantification

Compound	Formula	Limit of quantification (ng L^{-1})
Trihalomethanes		
THM4		
Trichloromethane (chloroform)	$CHCl_3$	3
Bromodichloromethane	$CHBrCl_2$	3
Dibromochloromethane	$CHBr_2Cl$	3
Tribromomethane (bromoform)	$CHBr_3$	3
I-THMs		
Dichloroiodomethane	$CHCl_2I$	0.4
Bromochloroiodomethane	$CHBrClI$	1
Dibromoiodomethane	$CHBr_2I$	0.2
Chlorodiiodomethane	$CHClI_2$	0.5
Bromodiiodomethane	$CHBrI_2$	0.8
Triiodomethane (iodoform)	CHI_3	2
Haloacetic acids		
HAA5		
Chloroacetic acid	$ClCH_2COOH$	50
Bromoacetic acid	$BrCH_2COOH$	25
Dichloroacetic acid	$Cl_2CHCOOH$	10
Dibromoacetic acid	$Br_2CHCOOH$	10
Trichloroacetic acid	$Cl_3CHCOOH$	10
others		
Bromochloroacetic acid	$BrClCHCOOH$	10
Iodoacetic acid	ICH_2COOH	5
Inorganic species		
Bromide	Br^-	200
Iodide	I^-	200
Iodate	IO_3^-	200

Iodide, iodate and bromide were measured (on quenched samples) using a liquid chromatograph coupled to an inductively coupled plasma mass spectrometer (HPLC-ICP-MS). Separation was done by a Dionex IonPac AS16 (2 x 250 mm) analytical column with an AG16 (2 x 50 mm) guard column using NaOH 20 mM gradient eluent (Table VI-S5). ICP-MS (Agilent 7900) conditions were set at an RF Power of 1550W, no gas mode (collision cell), with an aerosol dilution (0.1 L min^{-1}), and bromine and iodine were determined through their ^{79}Br and ^{127}I isotopes, respectively. The quantification limit was 0.2 µg L^{-1} for all 4 species. Total iodine and bromine in the water samples were determined by ICP-MS (Varian 820 MS) in 0.5% tetramethylammonium hydroxide matrixes with ^{129}Xe as internal standard (Balaram et al. 2012). Analyses were performed using He at 100 mL min^{-1} at the collision reaction interface (skimmer cone) to handle potential polyatomic interferences. The quantification limit level was 0.5 µg L^{-1} for both iodine and bromine.

2.3. Chlorination tests

The THMFP tests were carried out using a 7-day chlorination procedure according to the Standard Methods 5710 for the formation of trihalomethanes and other by-products (Standard method, 2017) with minor modifications. The samples were placed in either 25 or 50 mL glass reagent bottles with PTFE lined screw caps, buffered with a phosphate solution at pH 7.0 ± 0.2 and chlorinated using a stock solution of 5000 mg L^{-1} sodium hypochlorite at different doses in order to ensure samples with free chlorine residuals between 3 and 5 mg L^{-1} at the end of the chlorination test. Samples were incubated in a thermostatically controlled cabinet at 25 ± 2 °C for 7 days, after which the free chlorine residual of each sample was measured using the *N*, *N*-diethyl-*p*-phenylenediamine (DPD) colorimetric method (Rodier et al. 2016). In the samples containing a free chlorine residual between 3 and 5 mg L^{-1}, the chlorine residual was quenched using sodium thiosulfate and the samples were stored at 4 °C in the dark until the THMs were analyzed. Note that for the sample after nitrification (campaign A), no bottle having a free chlorine residual between 3 and 5 mg L^{-1} was obtained, therefore the bottle with the closest chlorine residual (*i.e.* 1.9 $mgCl_2$ L^{-1}) was selected. Results are presented as ΔTHMFP corresponding to the difference between THM concentrations after chlorination tests and THMs already formed in the treatment line (Standard method, 2017).

3. Results and discussion

3.1. THMFP *vs.* NOM fractions

The DOC concentration decreases from 8.5 to around 2.5 mgC L^{-1} for campaign A and from 6.7 to 2.2 mgC L^{-1} for campaign B along the treatment line, corresponding to a removal of around 70% of DOC for both campaigns. The biggest removal occurs during the coagulation step – from 57 to 65% – which are high removal efficiencies when comparing to the literature (14-66% (Liang et Singer 2003; Boyer et Singer 2005; Vasyukova et al. 2013)). The DOC decrease in subsequent steps is quite low with less than 12% of the initial DOC content removed during activated carbon filtration (Figure IV-1a).

The removal of THM precursors is slightly higher than the DOC removal with a decrease of the THMFP of around 80% for the entire treatment (Figure IV-S1). The THMFP/DOC ratio is around 70 µg THM/mgC in the raw waters for both campaigns A and B, despite the different DOC concentrations of 8.5 and 6.7 mgC L^{-1}, for campaigns A and B, respectively (Figure IV-S2). This result is in agreement with the literature showing good correlation between THMFP and DOC in raw waters with THMFP/DOC ratios usually in the 27 to 167 µg THM/mgC range (Table IV-S6). As such, the DOC in this surface water has an intermediate reactivity towards the formation of DBPs compared to the full range of NOM characterized in literature. The THMFP/DOC ratio (Figure IV-S2) decreases progressively along the treatment line (around 35% for both campaigns), thus the THMFP is more reduced than the DOC during treatment. The same behavior has been observed in several studies (Table IV-S6). For example, Golea et al. (2017) show a 50% decrease of the THMFP/DOC ratio during water treatment. This reflects a more preferential removal of the highly reactive organic THM precursors by the water treatment process – such as reactive hydrophobic fractions by coagulation-decantation – or the progressive transformation of NOM towards less reactive moieties as the DOC is not significantly removed during nitrification nor in the filtration steps.

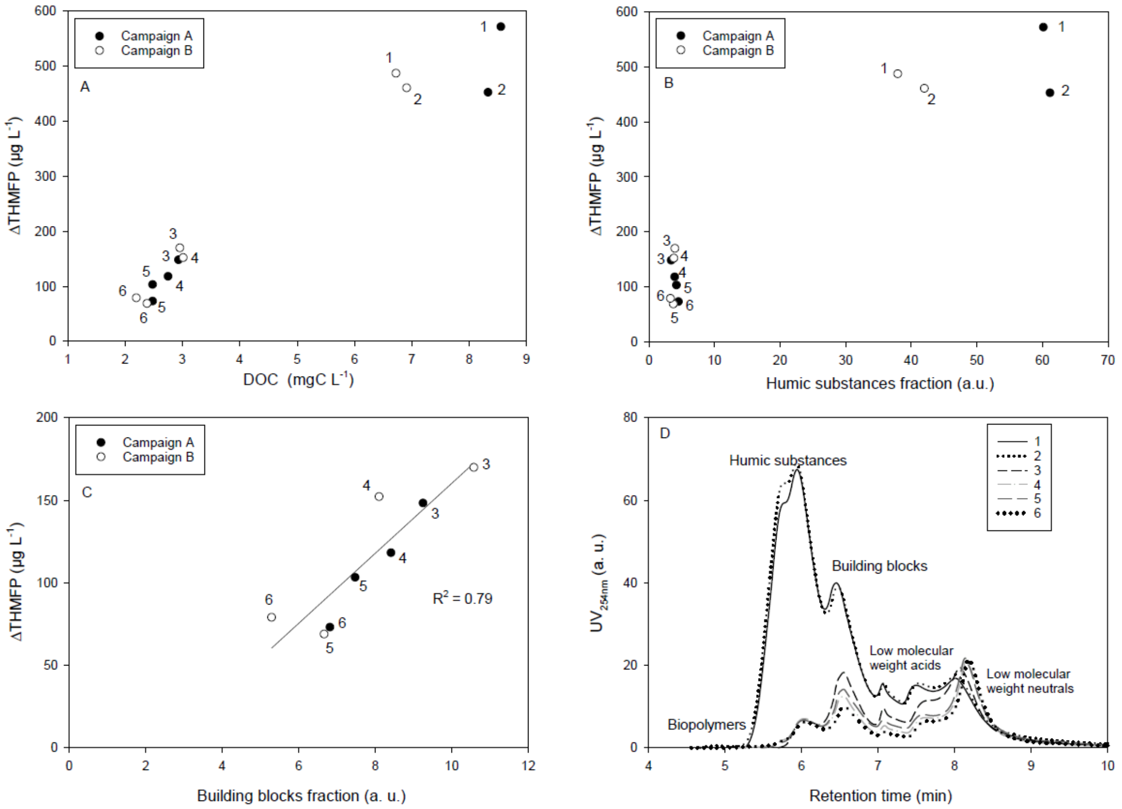

Figure IV-1: Evolution of the ΔTHMFP ($[THM]_{FP}$ – $[THM]_0$) in waters sampled after the different treatment steps in function of (A) the DOC content (B) the fraction of humic substances and (C) the fraction of building blocks for treatment steps after coagulation. (D) Size exclusion chromatograms with UV detection (254 nm) for campaign A (campaigns B and C in Figure S3). 1- raw water, 2- after nitrification, 3- after coagulation-decantation, 4- after intermediate chlorination and sand filtration, 5- after GAC filtration, 6- after final chlorination. (a.u.: arbitrary unit).

Considering the speciation of the THM4 formed in the THMFP tests (Figure IV-S1), $CHCl_3$ is the main THM formed at all treatment steps. Of the four regulated THMs, the three brominated species were present in relatively low concentrations, accounting for 12-51% (mass weighted) of the THM4 despite a consequent level of bromide in raw water $[Br^-]$ = 140-170 µg L^{-1}. $CHBrCl_2$ was found to be the main brominated THM species present, being formed at all treatment steps. In the standardized protocol of the formation potential tests, chlorine is introduced in a large excess explaining the predominance of chloroform and the extremely low I-THMs levels (<0.1 µg L^{-1}). In the chlorinated final water from the drinking water treatment plant (residual = 0.3 ± 0.1 mgCl_2 L^{-1}) the main species found were $CHBrCl_2$ and $CHBr_2Cl$. This difference in detected THM species is due to the higher rate constants observed for reactive bromine species compared to reactive chlorine species (Criquet et al. 2015), thus the NOM precursor sites are preferentially brominated at low chlorine doses. The THM4 in the final water ranged from 26 up to 34 µg L^{-1}, well below the THM regulation level of 100 µg L^{-1} imposed by the European Union.

In Figure IV-1, a similar trend is shown between the reductions of the ΔTHMFP with DOC (a) and the humic substance fraction measured using SEC-UV (b). This fraction, corresponding to compounds of relatively high molecular weight, is the main fraction removed by the coagulation step (reduction of around 90% of its UV absorbance in the SEC chromatogram; Figure IV-1d). These results are in good agreement with previous studies which have found good correlation with this particular fraction (Vasyukova et al. 2013; Li et al. 2014; Golea et al. 2017). Assuming that the UV absorbance properties of humic substances (HS) are not significantly changed during coagulation, a lower reduction of THMFP (70%) compared to humic substances (90%) is observed, underlining the role of other organic fractions in addition to HS as THM precursors. In addition, the DBPFP was further reduced while the humic substances were not removed by the following treatment steps (Figure IV-1b). A proportional decrease of the THMFP *vs*. the building-block fraction was then observed during the two sampling campaigns (Figure IV-1c). Good correlations were obtained for the steps after coagulation/decantation (r^2 = 0.79, Figure IV-1c) when the remaining humic substances were no longer removed.

3.2. Bromine incorporation

The formation of bromine-containing DBPs is of particular interest as they are generally more toxic than chlorine-containing compounds (Plewa et al. 2004). The bromine incorporation factor (BIF), first developed by Gould et al. (1983), describes the molar contribution of all brominated species (Eq.1, with THMs in molar concentrations), and can range from 0 to 3, with values closer to 3 representing a more brominated THM sample.

$$BIF\ (THMs) = \frac{[CHBrCl_2]+2\,[CHBr_2Cl]+3[CHBr_3]}{[CHCl3]+\,[CHBrCl_2]+[CHBr_2Cl]+[CHBr_3]} \qquad \text{Eq. 1}$$

The incorporation of bromine into THMs after the THMFP tests increases along the treatment line (Figure IV-2). The BIF after the THMFP test is low for the raw water and after the nitrification step (around 0.15), then increases after coagulation to approximately 0.4, and increases further after GAC filtration (from 0.5 to 0.65 for campaigns A and B, respectively). It has to be noted that the bromide concentration decreases after sand filtration due to the intermediate chlorination step, but increases once more after GAC filtration indicating bromide release while DOC is slightly removed (Figure IV-3). This could be due to microbial degradation of some brominated compounds adsorbed on GAC.

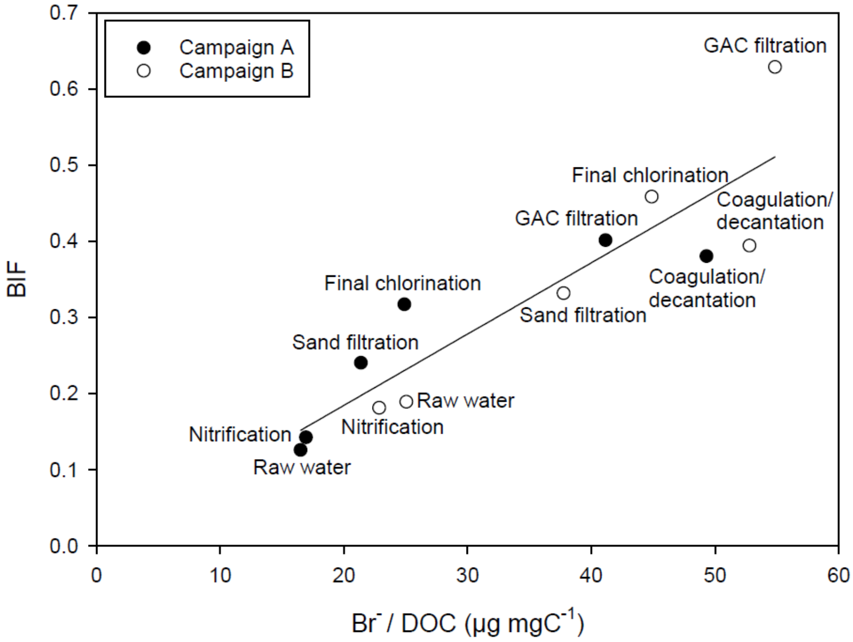

Figure IV-2: Correlation between the bromine incorporation factor (BIF) from ΔTHMFP vs. Br-/DOC in the treatment line including sampling campaigns A and B.

The BIF for ΔTHMFP was found to increase linearly in function of the Br^-/DOC (Figure IV-2). Effectively, a higher Br^-/DOC ratio (usually the result of DOC removal during treatment) favors reaction of chlorine with Br^- instead of DOC, resulting in the formation of reactive bromine species (which in turn react with NOM to form brominated THMs, resulting in a higher BIF). The maximum BIF value (0.63) is reached after GAC filtration, where the Br^-/DOC ratio is highest. Even though this value represents a quite moderate bromine incorporation into THMs, one should keep in mind that the formation potential protocol favors reactive chlorine species over bromine species. The samples from both A and B campaigns show a single correlation between the BIF and Br^-/DOC ratio (Figure IV-2), however when correcting the Br^-/DOC ratio for the amount of Cl_2 consumed, different correlations for campaigns A and B are found (Figure IV-S6). In addition, adjusting the Br^-/DOC ratio with consumed chlorine increases the linearity of the established correlations with BIF, as indicated by regression coefficients of 0.99 and 0.91 for campaigns A and B respectively (Figure IV-S6 – with correction for Cl_2 consumed), and regression coefficients of 0.80 and 0.83 for campaigns A and B respectively (Figure IV-2, regression lines not shown – without correction for Cl_2 consumed). The different slopes could be due to the higher bromide concentration in the raw water for the campaign B (170 µg L^{-1}) compared to campaign A (140 µg L^{-1}) in conjunction with lower DOC values (6.7 and 8.5 mgC L^{-1} respectively for campaigns B and A). Then, the competition between NOM and Br^- is in favor of a higher formation of bromine active species in campaign B.

The BIFs of the treated water having a chlorine residual around 0.3 $mgCl_2$ L^{-1} (sampled from the water treatment works without further treatment except chlorine residual quenching), ranged from 1.1 to 1.4 for campaigns A and B respectively, which is much higher than the ones observed for the THMFP. This matches BIF values found by Hong et al. (2013), ranging between 0.86 and 1.42 for bromide values between 109 and 209 µg L^{-1}. The removal of bromide ions from the raw water to the final chlorination ranges from 37 to 56%, which is due to incorporation of bromide into the organic matrix through the chlorination steps. During final chlorination, 20 to 40% of the bromide content is incorporated into NOM (Figure IV-S5). This result is in agreement with the literature showing from 10 to 40% of halogen incorporated into NOM (Criquet et al. 2015; Zhao et al. 2017).

3.3. Iodine speciation and I-DBP formation

The level of iodide ion found in the raw water during this survey highly varied, with concentrations between < 0.2 and 18.2 µg L^{-1}. These concentrations are however in line with the range of concentrations usually found in surface waters (Snyder et Fehn 2004; Salminen et al. 2005). During campaign A (Figure IV-3), iodide was not detected and iodate was below the quantification limit (< 0.2 µg L^{-1}) in the raw water, however, some I-THMs and mainly iodate were found after intermediate chlorination (see below). Then, a part of iodine could certainly be present in the raw water as incorporated in the organic matrix. Analysis of total iodine by ICP-MS during campaign B confirmed this fact, with only 17% of the total iodine present as iodide in the raw water (total iodine = 33.3 µg L^{-1}; $[I^-]$ = 5.6 µg L^{-1} and $[IO_3^-]$ < 0.2 µg L^{-1}) which is in the same range as that found by Gilfedder et al. (2009) (15% of iodine as inorganic iodine in a lake). The absence of supplementary peaks on the HPLC-ICP-MS chromatogram suggests that the iodine is bound to non-ionic organic species. Regular monitoring since 2017 shows the presence of iopromide and iomeprol up to 0.3 µg L^{-1} in the raw water. However, these target substances could represent only a small part of the iodinated anthropogenic substances present in this water; no extensive study has been performed on iodine speciation and importance of the organic fraction. In addition, other sources of organic iodine may originate from the surrounding agricultural activities, and may include iodinated cattle feed supplements (Moreno-Reyes et al. 2011), as well as iodophor medication, iodine-containing sterilizers of milking equipment, teat dips and udder washes, or pesticides and fungicides containing iodine (Scientific Committee on Food; Health and Consumer Protection Directorate-General

2002). One should also consider that some authors suggest that the major fraction of iodine in a number of natural fresh waters is bound to high molecular weight humic substances (Gilfedder et al. 2009). This complexation could then interfere with the analysis of iodide. Iodate was near absent (< 0.2 µg L^{-1}) in raw water and all treatment steps before intermediate chlorination.

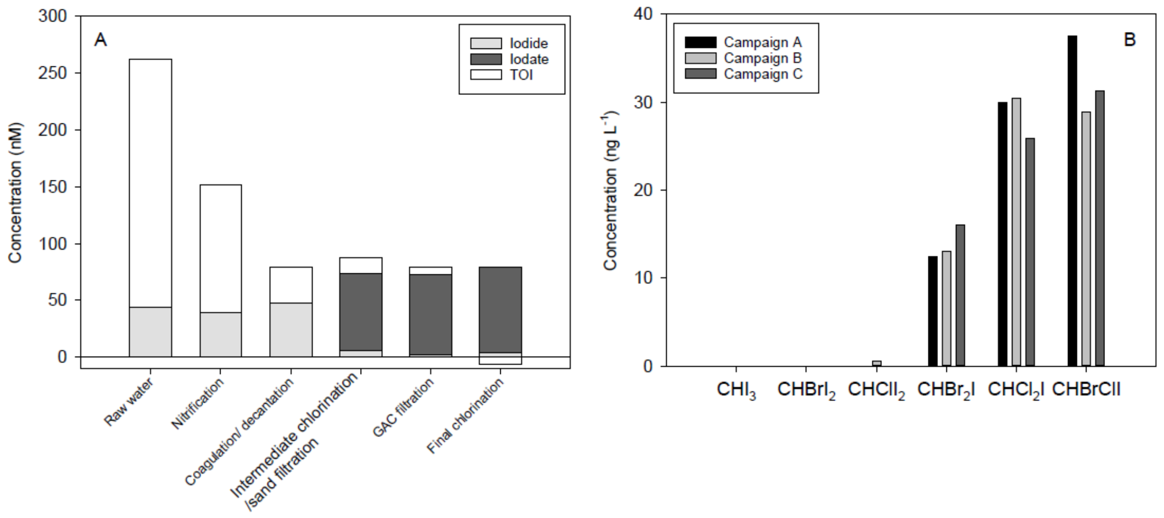

Figure IV-3: A) Iodine speciation after each treatment step of the water treatment plant (campaign B) and B) I-THMs formation after final chlorination for the 3 campaigns. Iodide and iodate were analyzed by IC-ICP-MS. Total Organic Iodine was assessed by the difference between total iodine measured by ICP-MS and inorganic iodine species (iodide and iodate).

The speciation of iodine and I-THMs formation has been assessed directly on the (filtered) water samples from the treatment plant without further experiments involving chemical modification (such as a DBPFP test). In fact, the DBPFP test is not suitable for I-DBPs, as the large chlorine dose would favor the formation of iodate and I-THMs would hardly be detected. The organic iodine (TOI) has been assessed by the difference between the total iodine concentration obtained by ICP-MS and the inorganic species, *i.e.* I^- and iodate contents determined by HPLC-ICP-MS. According to Figure IV-3, a part of organic iodine is removed after nitrification while the concentration of iodide stays unchanged. This behavior is not fully explained, especially because the DOC remains unchanged at this step. It is possible that organic iodine is incorporated into the biomass. A second decrease of organic iodine without significant change of iodide is observed during coagulation suggesting that a part of the organic iodine is bound to the high molecular weight NOM, which constitutes the major fraction removed during this process (Figure IV-1d). After the intermediate chlorination step, most of the iodine is converted into iodate. In fact, the quantity of iodate formed is higher than the initial iodide present in the water before chlorination, indicating that a part of the organic iodine also contributes to the formation of iodate. This behavior has been observed for the 3 campaigns (Figure IV-3). This high degree of transformation is certainly due to the relatively high concentration of bromide, which plays a role of a catalyst in iodide oxidation to iodate during chlorination (Criquet et al. 2012). Interestingly, the iodate concentrations varies from 5 to 13 µg L^{-1} in the treated water between sampling campaigns, however despite this variation a surprisingly stable I-THMs formation and speciation is observed (Figure IV-3). The same reproducibility between the campaigns is observed for the intermediate chlorination step. The sum of I-THMs formed ranges from 73 to 80 ng L^{-1} in the distributed water which corresponds to less than 0.6% (on a mass basis) of the total THMs formed. The major I-THMs formed are the CHBrClI and $CHCl_2I$ followed by $CHBr_2I$, while no iodoform has been observed. These compounds correspond to the mono-iodinated compounds which are predominant for low iodide levels (Zhang et al. 2015). Iodoacetic acid reached a concentration of around 20 ng L^{-1} after the intermediate chlorination step but was below detection level (< 2 ng L^{-1}) in the final water (Figure IV-4b).

3.4. Impact of GAC filtration on DBPs: THMs, I-THMs and HAAs

The presence of an intermediate chlorination step in this water treatment plant allows to assess the efficiency of the full-scale activated carbon filtration to remove DBPs. While two previous studies found that some I-THMs were removed by GAC filtration (Cancho et al. 2000; Cuthbertson et al. 2019), scarce information on the removal efficiency of I-DBPs in a full scale drinking water treatment plant is available. THM4 were formed (from 15 up to 28 µg L^{-1}) and are present in the treatment steps after intermediate chlorination. Some iodinated THMs were formed as well, adding up to a total of 270-312 ng L^{-1} of I-THMs, as well as HAAs reaching 8.4 µg L^{-1} (campaign C) for the sum of the 7 HAAs analyzed (Figure IV-4).

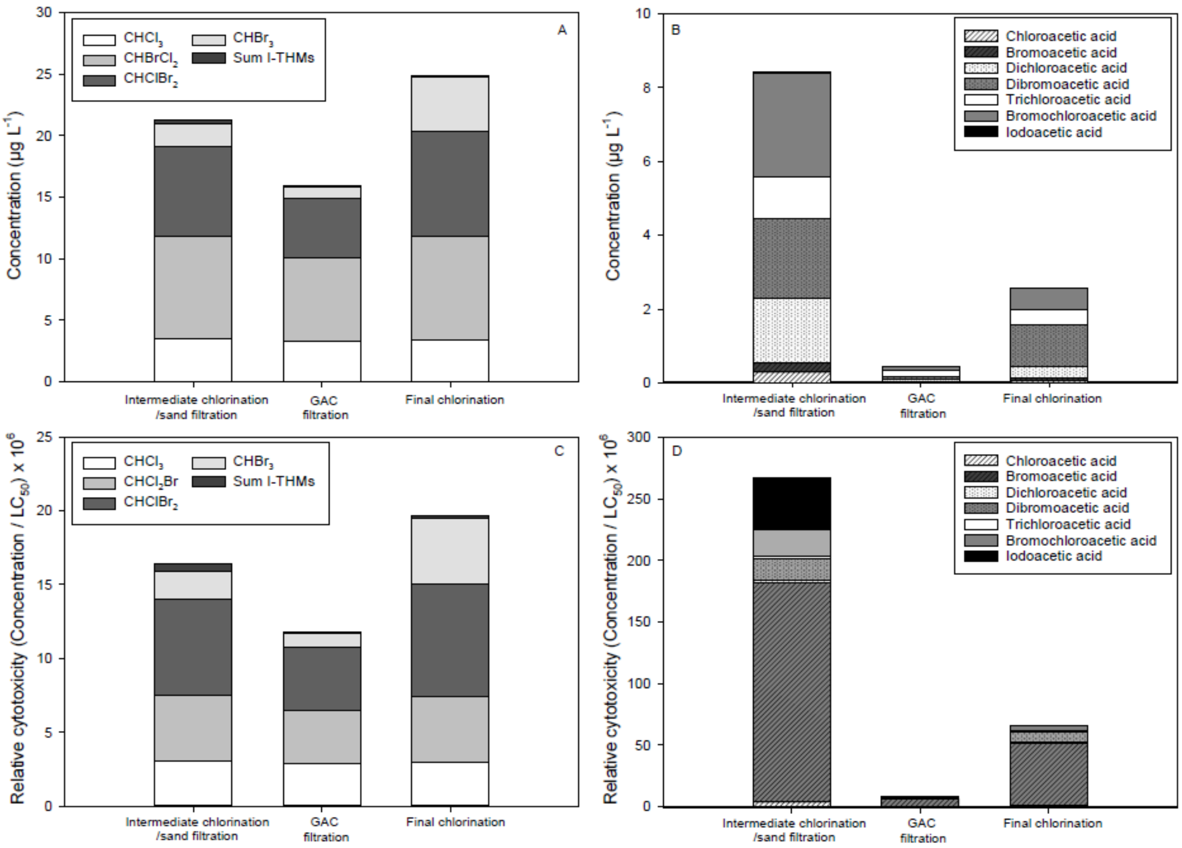

Figure IV-4: Concentrations of (a) THM4, sum of I-THMs and (b) HAAs in the water treatment plant (campaign C) after intermediate chlorination-sand filtration, activated carbon filtration (GAC) and final chlorination steps. Relative cytotoxicity of (c) THMs and (d) HAAs in the treated water for the same treatment steps. The relative cytotoxicity has been calculated using the LC50 related to each compounds (see paragraph 3.5).

The concentrations after activated carbon filtration showed a decrease in all analyzed DBPs. The removal based on the total mass ranged from 25 to 36% for the THM4 (3 campaigns); from 82 to 93% for the ∑I-THMs (3 campaigns) and 95% for HAAs (campaign C). Considering the THM4, $CHBr_3$ shows the highest removal (58% on average) followed by $CHBr_2Cl$ (43%), $CHBrCl_2$ (18%) and $CHCl_3$ (15%). This indicates that the removal of the highly brominated THMs is significantly higher than the chlorinated ones. This is also corroborated by Speth and Miltner (1990), who investigated the activated carbon adsorption behavior of different halogenated compounds including THM4, and reported increasing Freundlich adsorption constants with increasing bromine content. The removal efficiency increases with increasing molecular weight and increasing hydrophobicity (Figure IV-5). In addition, I-THMs show much higher removal, ranging between 77 and 99% on average for each compound. The same trend of increased removal with increasing molecular weight is observed ($CHClI_2$ showing the best removal – note that iodoform and $CHBrI_2$ were not detected, Figure IV-5). A positive relationship has been found between these two parameters ($R^2 = 0.83$, Figure IV-5), however I-THMs seems to have their own proportional increase with the molecular weight. Considering the hydrophobicity of the compounds, THM4 and I-THMs are in a similar range of K_{ow} (Table IV-S7), and then this parameter could only explain the variation within iodinated or non-iodinated compound families (Figure IV-5), therefore other factors such as biodegradation (Cuthbertson et al. 2019) or halogen bonding (Politzer et al., 2007; Anderson et al. 2018) could have impacted the THM removal on GAC. The latter process involves interactions with basic sites present on the activated carbon surface (Tessmer et al. 1997), and its bond strength increases in the order of Cl < Br < I (two references above). In comparison to other commercially available activated carbon types, the GAC type used in this drinking water treatment plant (Filtrasorb F400, Calgon Carbon – Chemviron and Norit GAC1240) indeed contains a substantial amount of basic sites on its surface.

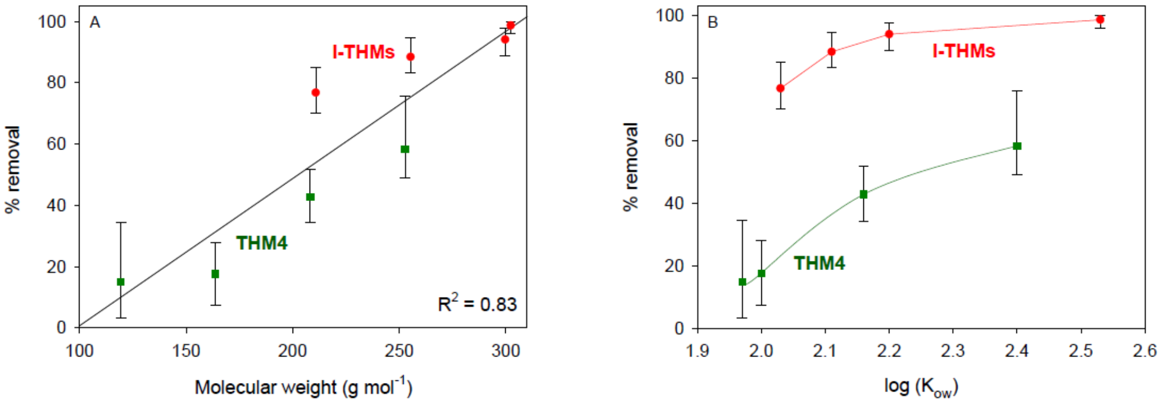

Figure IV-5: Percentage of trihalomethane removal through the granular activated carbon filtration expressed in function of molecular weight and hydrophobicity (values and references in Table IV-S8) Error bars represent the deviation of data from 3 campaigns.

HAAs show high removal efficiency by activated carbon filtration, from 87% (trichloroacetic acid) up to 97% (bromo- and dibromoacetic acids). The overall removal, based on total mass of the 7 HAAs analyzed in this study is around 95% (Table IV-S7, the same removal efficiency is obtained considering the HAA5). Tung et al. (2006) showed that adsorption plays a minor role in HAA removal and GAC filters have a lower adsorption capacity for mono-and dihaloacetic acids than for the regulated THMs. On the other hand, Kim et Kang (2008) showed that the removal of HAA5 by adsorption was high at the early stage of GAC operation and better than for THM4, with breakthrough occurring later. The GAC filtration here consists of 6 filters with 4 to 5 filters working in parallel during the campaigns. The average time since reactivation varied from 305 to 350 days for the 3 campaigns corresponding to bed volumes from 19000 to 21150. The contact time also varied between campaigns from 16 to 32 minutes. Considering the different varying parameters it is difficult to assess the main factor modifying the DBP removal. However, $CHCl_3$ and $CHCl_2Br$ showing the lowest adsorption on the one hand and $CHBr_2I$, $CHClI_2$ and CHBrClI showing the highest removal on the other hand, show the same variation with the different parameter (data not shown). More contrasted parameters would be necessary to assess the key parameters of adsorption, halogen bonding and/or biodegradation and to explain the higher variability of THM4 removal. The latter process can in fact play an important role especially considering the different removal efficiency of compounds showing the same hydrophobicity.

3.5. Relative toxicity assessment

A final chlorination step is performed after activated carbon filtration. The chlorine dose is controlled to achieve a target value of 0.3 $mgCl_2\ L^{-1}$ in the treated water leaving the plant. The median concentration of THM4 in the treated water is 25 $\mu g\ L^{-1}$ (n = 135, from routine monitoring analysis results from January 2016 until May 2019). In this study, the THM4 reached a concentration ranging from 25 to 34 $\mu g\ L^{-1}$ for the 3 campaigns which is far below the EU and US guidelines (respectively 100 (CELEX-EUR 1998) and 80 $\mu g\ L^{-1}$ (EPA 2016)). The highest concentration was found during campaign A, which coincides with the highest DOC concentration. The quantity of THM4 present in the water after final chlorination is always higher than the quantity present after the intermediate chlorination. The increase is however limited, *i.e.* from 4 to 11 $\mu g\ L^{-1}$. The concentration of THM4 corrected of the chlorine dose lower in the final step is indeed lower than during the intermediate chlorination. In addition, the quantity of I-THMs and HAAs formed during final chlorination is much lower compared to the quantity formed during intermediate chlorination. The occurrence of both I-THMs and HAAs is a factor 3 to 4 lower after postchlorination compared to intermediate chlorination. The low formation of I-THMs can be explained by the conversion of iodide into iodate, while the organic HAA-precursors has to be supposed to be the limiting parameter of HAA formation. Cuthbertson et al. (2019) on the contrary showed a chlorine controlled HAA formation, i.e. a continuous increase of HAA formation with the increasing chlorine dose.

In order to prioritize the different DBPs, regardless of the non-toxicity of the produced water and the compliance with the national and supranational quality guidelines, an assessment of relative toxicity has been performed using the mammalian cell cytotoxicity data as reported in Table IV-S8. In this approach, the concentration of each DBP is divided by its respective $LC_{50} \times 10^6$ (lethal concentration for 50% of a population) (Wagner and Plewa 2017), as the contribution of a DBP to toxicity is a function of both concentration and toxic potency. The relative cytotoxicity of THMs and HAAs has been summed to compare the toxicity of these two families.

Considering THMs, the general trend of cytotoxicity is similar to the species-specific concentration (Figures IV-4a and IV-4c). The LC_{50}s of the regulated THM4 are between 4.0×10^{-3} and 1.2×10^{-2} M, and despite exhibiting the highest concentration only the relative toxicity of $CHBrCl_2$, having the highest LC_{50}, is reduced compared with the other species. Considering I-THMs, their low concentrations and the absence of iodoform (most cytotoxic I-THM, $LC_{50} = 4.47 \times 10^{-5}$ M), induce a low contribution of only 0.48-0.58% to the overall THM cytotoxicity in final waters. The GAC filtration showing high removal of I-THMs contributes significantly to reduce the impact of these species to the relative cytotoxicity, as their contribution has been seen to account for up to 4.5% (campaign B) of the overall cytotoxicity of the THMs formed from the intermediate chlorination.

Compared to THMs, HAAs show much lower LC_{50} values, indicating their higher relative toxicity. This is especially the case for bromoacetic acid (9.6 x 10^{-6} M) and iodoacetic acid (2.95 x 10^{-6} M). The main HAAs present (dichloroacetic acid, dibromoacetic acid, trichloroacetic acid and bromochloroacetic acid) do not contribute much to the total relative cytotoxicity of HAAs (Figure 4d). On the contrary, bromoacetic acid contributes the most with 80% of the relative cytotoxicity of HAAs in the final water. Considering the intermediate chlorination, iodoacetic acid (showing the lowest LC_{50} value) contributes to 16% of the relative cytotoxicity despite a concentration of at least a factor 10 lower than the other HAAs (23 ng L^{-1}). However, the iodoacetic acid is removed well from the water by the GAC filtration and is no further formed in the final chlorination. The GAC filtration reduces the total relative toxicity by a factor higher than 10.

As stated in paragraph 3.5, the HAAs formed during intermediate chlorination are removed well in the subsequent GAC filtration (95%), while the DOC reduction only accounts for around 10% of the initial DOC. Then it could be questioned if the HAAs would have been much higher in the final water without intermediate chlorination, as their NOM precursors may not be specifically removed by the GAC (Cuthbertson et al. 2019). Remarkably, HAAs represent less than 10% of the total mass of measured DBPs (*i.e.* THMs + HAAs) but account for more than 75% of the relative cytotoxicity of the treated water. Bromoacetic acid by itself accounts for more than 60% of the overall toxicity of the 17 compounds included in this study.

It is clear that currently unregulated DBPs (*i.e.* HAAs or I-DBPs) can have a higher contribution to the relative toxicity of chlorinated waters than currently regulated DBPs (*i.e.* THM4). However THM4 serve as an indicator parameter for DBPs in general in the European drinking water directive. Drinking water companies should apply a risk based approach to minimize formation of DBPs by removing precursors in the treatment and by limiting the amount of disinfectant used, however without compromising the disinfection efficiency (EPRS 2019). Nonetheless, the WHO (2017) recently commended to include 9 chlorinated and brominated HAAs in the drinking water legislation, as these substances are formed to a higher extent under acidic conditions. Together with THM4, these 9 HAAs should then serve as indicator parameter for all DBPs formed and all water types.

4. Conclusion

THMFP tests performed on samples taken from a drinking water treatment plant in addition to the characterization of organic matter give some valuable information on the DBP formation and the removal of NOM fractions during the different steps of water treatment. First, the majority of DOC, especially high molecular weight fractions, are removed during coagulation, which was accompanied by the highest decrease of THMFP. However, a further decrease of the THMFP is seen with the subsequent removal of the building block fraction despite the relatively low removal of the overall DOC. The possible incorporation of bromine into THMs is seen to increase throughout the water treatment plant, since the reduction of DOC concentration results in a lower competition for chlorine to react with bromide, forming bromine reactive species.

Despite large modifications of water quality in terms of DOC and iodine concentrations, the formation of I-THMs was constant in terms of quantity and speciation between campaigns. A high conversion of iodine into iodate by chlorination is observed in this water certainly due to the relatively high bromide level which catalyzes iodate formation. A degradation of organic iodinated compounds is observed in this water treatment plant with a larger formation of iodate compared to iodide levels in the raw water.

In this water treatment plant, the activated carbon filtration is effective to remove the HAAs and I-THMs (> 80%) formed during intermediate chlorination but to a lesser extent for THM4 (< 40%). A preferential removal in the order I-DBPs>Br-DBPs>Cl-DBPs is observed with no clear trend related to the compound hydrophobicity suggesting other factors such as biodegradation or halogen bonding are also governing DBP removal on GAC. The amount of DBPs formed per quantity of DOC decreases through the treatment line showing the preferential removal of the highly reactive precursors. This is especially the case for HAAs that are not formed in a high extent during the final chlorination step. From a theoretical assessment, the relative cytotoxicity of HAAs is more important than the one of THMs, even considering their much lower concentrations. Bromoacetic acid accounts for 60% of the relative cytotoxicity of THMs and HAAs formed in the distributed water. These results would support the importance of inclusion of HAAs in the future European regulation considering their higher toxicity, their different behavior and formation compared to THMs presently used as a surrogate for all DBPs.

Acknowledgments

This work was supported by Interreg 2 Seas program via DOC2C's project. Authors also thank De Watergroep, the Region "Hauts-de-France", CPER Climibio and Chevreul Institute for support. Asmae Adnane and Veronique Alaimo are thanked for their technical support.

5. Bibliography

Allard et al. 2015. S. Allard, J. Tan, C. A. Joll and U. von Gunten. '**Mechanistic Study on the Formation of Cl-/Br-/I-Trihalomethanes during Chlorination/Chloramination Combined with a Theoretical Cytotoxicity Evaluation**'. Environmental Science & Technology, 49 (18) (2015), pp. 11105-11114. *https://doi.org/10.1021/acs.est.5b02624*.

Anderson et al. 2018. N. L. Anderson, F. W. Aquino, A. E. Raeber, X. Chen and B. W. Wong '**Halogen bonding interactions: revised benchmarks and a new assessment of exchange vs dispersion**' Journal of Chemical Theory and Computation, 14 (1) (2018), pp. 180-190. *https://doi.org/ 10.1021/acs.jctc.7b01078*.

Baert and Devos 1996. R. Baert, R. Loontiens and M. Devos. '**Potassium and Nitrate Pollution of Surface Water in the Catchment Area of the "Blankaert" Water Production Centre in Flanders (Belgium)**'. In *Proceedings _ 4th International Conference On Environmental Compliance And Enforcement*, Chiang Mai, Thailand, (1996), pp. 2:635-642.

Balaram et al. 2012. V. Balaram, M. Satyanarananan, D. V. Avdeev, N. Berdnikov, P. Roy, S. S. Sawant, K. Subramanyam, et al. '**Use of Xenon as Internal Standard for the Accurate Determination of Trace Elements in Water Samples by ICP-MS**'. Atomic Spectroscopy, 33 (2012), pp. 41-47.

Bichsel 2000. Y. Bichsel. '**Behavior of Iodine Species in Oxidative Processes During Drinking Water Treatment'**. Doctoral thesis Diss. ETH, No. 13429.

Bond et al. 2011. T. Bond, E. H. Goslan, S. A. Parsons and B. Jefferson (2011) '**Treatment of disinfection by-product precursors**'. *Environmental Technology*, 32 (1) (2011), pp. 1-25. *https://doi.org/10.1080/09593330.2010.495138*.

Boyer and Singer 2005. T. H. Boyer and P. C. Singer. '**Bench-Scale Testing of a Magnetic Ion Exchange Resin for Removal of Disinfection by-Product Precursors**'. Water Research, 39 (7) (2005), pp. 1265-1276. *https://doi.org/10.1016/j.watres.2005.01.002*.

Cancho et al. 2001. B. Cancho, C. Fabrellas, A. Diaz and F. Ventura. '**Determination of the Odor Threshold Concentrations Of Iodinated Trihalomethanes in Drinking Water**'. Journal of Agricultural and Food Chemistry, 49 (4) (2001), pp. 1881-1884. *https://doi.org/10.1021/jf001252m*.

Cancho et al. 2000. B. Cancho, F. Ventura, M. T. Galceran, A. Diaz and S. Ricart. '**Determination, Synthesis and Survey of Iodinated Trihalomethanes in Water Treatment Processes**'. Water Research, 34 (13) (2000), pp. 3380-3390. *https://doi.org/10.1016/S0043-1354(00)00079-8*.

CELEX-EUR. 1998. CELEX-EUR. « **Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption** ». Official Journal of the European Communities, L 330, (41) (1998), pp. 32-54.

Chili et al. 2012. C. A. Chili, P. Westerhoff, and A. Ghosh. '**GAC Removal of Organic Nitrogen and Other DBP Precursors**'. Journal AWWA, 104 (7) (2012), pp. E406-415. *https://doi.org/10.5942/jawwa.2012.104.0090*.

Criquet et al. 2012. J. Criquet, S. Allard, E. Salhi, C. A. Joll, A. Heitz and U. von Gunten. '**Iodate and Iodo-Trihalomethane Formation during Chlorination of Iodide-Containing Waters: Role of Bromide**'. Environmental Science & Technology, 46 (13) (2012), pp. 7350- 7357. *https://doi.org/10.1021/es301301g*.

Criquet et al. 2015. J. Criquet, E. M. Rodriguez, S. Allard, S. Wellauer, E. Salhi, C. A. Joll and U. von Gunten. '**Reaction of Bromine and Chlorine with Phenolic Compounds and Natural Organic Matter Extracts – Electrophilic Aromatic Substitution and Oxidation**'. Water Research, 85 (2015), pp. 476-486. *https://doi.org/10.1016/j.watres.2015.08.051*.

Cuthbertson et al. 2019. A. A. Cuthbertson, S. Kimura, H. K. Liberatore, R. S. Summers, D. Knappe, Benjamin D. Stanford, J. C. Maness, R. Mulhern, M. Selbes and S. D. Richardson. '**Does GAC with Chlorination Produce Safer Drinking Water? From DBPs and TOX to Calculated Toxicity**'. Environmental Science & Technology, 53 (2019), pp. 5987-5999. *https://doi.org/10.1021/acs.est.9b00023*.

De Watergroep 2015. De Watergroep Annual activity report De Watergroep: '**Activiteitenverslag 2015**' (2015). Accessible on line: *https://www.dewatergroep.be/nl-be/over-de-watergroep/wat-doen-we/jaarverslagen*.

EPRS 2019. EPRS European Parliamentary Research Service. '**Revision of the Drinking Water Directive**' 15-04-2019. Accessible on line: *http://www.europarl.europa.eu/thinktank/en/document.html?reference=EPRS_BRI(2018)625179*.

EPA 2016. EPA. '**Six-Year Review 3 Technical Support Document for Disinfectants/Disinfection Byproducts Rules**'. (2016), 468 p.

Gilfedder et al. 2009. B. S. Gilfedder, M. Petri and H. Biester. '**Iodine Speciation and Cycling in Fresh Waters: A Case Study from a Humic Rich Headwater Lake (Mummelsee)**'. Journal of Limnology, 68 (2) (2009), pp. 396-408. *https://doi.org/10.4081/jlimnol.2009.396*.

Golea et al. 2017. D. M. Golea, A. Upton, P. Jarvis, G. Moore, S. Sutherland, S. A. Parsons and S. J. Judd. '**THM and HAA formation from NOM in raw and treated surface waters**'. Water Research, 112 (2017), pp. 226-235. *https://doi.org/10.1016/j.watres.2017.01.051*.

Hansson et al. 1987. R. C. Hansson, M. J. Henderson, P. Jack and R. D. Taylor. '**Iodoform Taste Complaints in Chloramination**'. Water Research, 21 (10) (1987), pp. 1265-1271. *https://doi.org/10.1016/0043-1354(87)90179-5*.

Hong et al. 2013. H. Hong, Y. Xiong, M. Ruan, F. Liao, H. Lin and Y. Liang. '**Factors affecting THMs, HAAs and HNMs formation of Jin Lan Reservoir water exposed to chlorine and monochloramine**'. Science of The Total Environment, 444 (2013), pp. 196-204. *https://doi.org/10.1016/j.scitotenv.2012.11.086*.

Hua and Reckhow 2012. G. Hua and D. A. Reckhow. '**Evaluation of Bromine Substitution Factors of DBPs during Chlorination and Chloramination**'. Water Research, 46 (13) (2012), pp. 4208-4216. *https://doi.org/10.1016/j.watres.2012.05.031*.

Jones et al. 2011. D. B. Jones, A. Saglam, A. Triger, H. Song and T. Karanfil. '**I-THM Formation and Speciation: Preformed Monochloramine versus Prechlorination Followed by Ammonia Addition**'. Environmental Science & Technology, 45 (24) (2011), pp. 10429-10437. *https://doi.org/10.1021/es202745t*.

Kim and Kang 2008. J. Kim and B. Kang. '**DBPs Removal in GAC Filter-Adsorber**'. Water Research, 42 (1‑2) (2008), pp. 145-152. *https://doi.org/10.1016/j.watres.2007.07.040*.

Fu et al. 2017. J. Fu, W-N. Lee, C. Coleman, K. Nowack, J. Carter, and C-H. Huang. '**Removal of Disinfection Byproduct (DBP) Precursors in Water by Two-Stage Biofiltration Treatment**'. Water Research, 123 (2017), pp. 224-235. *https://doi.org/10.1016/j.watres.2017.06.073*.

Li et al. 2014. A. Li, X. Zhao, R. Mao, H. Liu and J. Qu. '**Characterization of Dissolved Organic Matter from Surface Waters with Low to High Dissolved Organic Carbon and the Related Disinfection Byproduct Formation Potential**'. Journal of Hazardous Materials, 271 (2014), pp. 228-235. *https://doi.org/10.1016/j.jhazmat.2014.02.009*.

Liang and Singer 2003. L. Liang and P. C. Singer. '**Factors Influencing the Formation and Relative Distribution of Haloacetic Acids and Trihalomethanes in Drinking Water**'. Environmental Science & Technology, 37 (13) (2003), pp. 2920-2928. *https://doi.org/10.1021/es026230q*.

Moreno-Reyes et al. 2011. R. Moreno-Reyes, H. Van Oyen and S. Vandevijvere. '**Optimization of Iodine Intake in Belgium**'. Annales d'Endocrinologie, 72 (2) (2011), pp. 158-161. *https://doi.org/10.1016/j.ando.2011.03.021*.

Plewa et al. 2004. M. J. Plewa, E. D. Wagner, S. D. Richardson, A. D. Thruston, Y-T. Woo and A. B. McKague. '**Chemical and Biological Characterization of Newly Discovered Iodoacid Drinking Water Disinfection Byproducts**'. Environmental Science & Technology, 38 (18) (2004), pp. 4713-4722. *https://doi.org/10.1021/es049971v*.

Politzer et al. 2007. P. Politzer, P. Lane, M.C. Concha, Y. Ma and J. S. Murray. '**An overview of halogen bonding**'. Journal of Molecular Modeling, 13 (2) (2007), pp. 305-311. *https://doi.org/ 10.1007/s00894-006-0154-7*.

Richardson et al. 2007. S. D. Richardson, M. J. Plewa, E. D. Wagner, R. Schoeny and D. M. DeMarini. '**Occurrence, genotoxicity, and carcinogenicity of regulated and emerging disinfection by-products in drinking water: A review and roadmap for research**'. Mutation Research, 636 (1-3) (2007), pp. 178-242. *https://doi.org/10.1016/j.mrrev.2007.09.001*.

Richardson et al. 2008. S. D. Richardson, F. Fasano, J. J. Ellington, F. G. Crumley, K. M. Buettner, J. J. Evans, B. C. Blount, L. K. Silva, T. J. Waite, G. W. Luther, A. B. McKague, R. J. Miltner, E. D. Wagner and M. J. Plewa. '**Occurrence and Mammalian Cell Toxicity of Iodinated Disinfection Byproducts in Drinking Water**'. Environmental Science & Technology, 42 (22) (2008), pp. 8330-8338. *https://doi.org/10.1021/es801169k*.

Roccaro et al. 2014. P. Roccaro, G. V. Korshin, D. Cook, C. W. K. Chow and M. Drikas. '**Effects of pH on the Speciation Coefficients in Models of Bromide Influence on the Formation of Trihalomethanes and Haloacetic Acids**'. Water Research, 62 (2014), pp. 117-126. *https://doi.org/10.1016/j.watres.2014.05.024*.

Rodier et al. 2016. J. Rodier, B. Legube and N. Merlet. '**L'analyse de l'eau**'. Technique et ingénierie, Dunod (10e édition) (2016), 1600 p.

Salminen et al. 2005. R. Salminen, M. J. Batista, M. Bidovec and T. Tarvainen. '**Geochemical Atlas of Europe. Part 1 - Background Information, Methodology and Maps**'. Geological Survey of Finland, Otamedia Oy, Espoo, (2005). *http://weppi.gtk.fi/publ/foregsatlas/text/I.pdf*.

Scientific Committee on Food; Health and Consumer Protection Directorate-General 2002. Scientific Committee on Food; Health and Consumer Protection Directorate-General. '**Opinion of the Scientific Committee on Food on the**

Tolerable Upper Intake Level of iodine'. SCF/CS/NUT/UPPLEV/26 Final. Brussels: European Commission (2002), 25 p.

Sedlak and von Gunten 2011. D. S. Sedlak and U. von Gunten. '**The Chlorine Dilemma**'. Science, 331 (6013) (2011), pp. 42-43. *https://doi.org/10.1126/science.1196397*.

Shackelford and Keith 1976. W. M. Shackelford and L. H. Keith. '**Frequency of Organic Compounds Identified in Water**'. Athens, Georgia: US Environmental Protection Agency, (1976). EPA-600/4-76-062.

Snyder and Fehn 2004. G. Snyder and U. Fehn. '**Global Distribution of ^{129}I in Rivers and Lakes: Implications for Iodine Cycling in Surface Reservoirs**'. Nuclear Instruments and Methods in Physics Research Section B: Beam Interactions with Materials and Atoms, Proceedings of the Ninth International Conference on Accelerator Mass Spectrometry, 223-224 (2004), pp. 579-586. *https://doi.org/10.1016/j.nimb.2004.04.107*.

Song et al. 2013. J. Song, W. Huang, P. Peng, B. Xiao and Y. Ma. '**Humic Acid Molecular Weight Estimation by High-Performance Size-Exclusion Chromatography with Ultraviolet Absorbance Detection and Refractive Index Detection**'. Soil Science Society of America Journal, 74 (6) (2013), pp. 2013-2020. *https://doi.org/10.2136/sssaj2010.0011*.

Speth and Miltner 1990. T. F. Speth and R. J. Miltner. '**Technical note: Adsorption capacity of GAC for synthetic organics**'. Journal AWWA, 82 (2) (1990), pp. 72-75. *https://doi.org/10.1002/j.1551-8833.1990.tb06922.x*.

Standard method, 2017. Standard Methods For the Examination of Water and Wastewater (5710). '**Formation of trihalomethanes and other disinfection by-products**'. (2017). *https://doi.org/10.2105/SMWW.2882.112*.

Tessmer et al. 1997. C. H. Tessmer, D. Radisav, V. Lois and J. Uranowski. '**Impact of Oxygen-Containing Surface Functional Groups on Activated Carbon Adsorption of Phenols**'. Environmental Science & Technology, 31 (7) (1997), pp. 1872-1878. *https://doi.org/10.1021/es960474r*.

Tugulea et al. 2018. A-C. Tugulea, R. Aranda-Rodriguez, D. Bérubé, M. Giddings, F. Lemieux, J. Hnatiw, L. Dabeka, and F. Breton. '**The Influence of Precursors and Treatment Process on the Formation of Iodo-THMs in Canadian Drinking Water**'. Water Research, 130 (2018), pp. 215-223. *https://doi.org/10.1016/j.watres.2017.11.055*.

Tugulea et al. 2015. A-C. Tugulea, R. Aranda-Rodriguez, M. Giddings, F. Lemieux and J. Hnatiw. '**Iodo-THM Formation During Chlorination of Source Waters With Naturally Occurring Ammonium and High Sodium Content**'. In *Disinfection By-Products in Drinking Water*, Special Publication No. 352, (2015), pp. 46-58. The Royal Society of Chemistry. K. C. Thompson, S. Gillespie, E. Goslan. *https://doi.org/10.1039/9781782622710-00046*.

Tung et al. 2006. H. H. Tung, R. F. Unz, and Y. F. Xie. '**HAA Removal by GAC Adsorption**'. American Water Works Association, 98 (6) (2006), pp. 107-112. *https://doi.org/10.1002/j.1551-8833.2006.tb07687.x*.

Vasyukova et al. 2013. E. Vasyukova, R. Proft, J. Jousten, I. Slavik, and W. Uhl. '**Removal of Natural Organic Matter and Trihalomethane Formation Potential in a Full-Scale Drinking Water Treatment Plant**'. Water Science and Technology: Water Supply, 13 (4) (2013), pp. 1099-1108. *https://doi.org/10.2166/ws.2013.095*.

Wagner and Plewa 2017. E. D. Wagner and M. J. Plewa. '**CHO Cell Cytotoxicity and Genotoxicity Analyses of Disinfection By-Products: An Updated Review**'. Journal of Environmental Sciences, 58 (2017), pp. 64-76. *https://doi.org/10.1016/j.jes.2017.04.021*.

WHO (2017). World Health Organization - Regional office for Europe. '**Support to the revision of Annex I Council Directive 98/83/EC on the Quality of Water Intended for Human Consumption (Drinking Water Directive) Recommendations.**' (2017), 240 p. *https://ec.europa.eu/environment/water/water-drink/pdf/WHO_parameter_report.pdf*.

Yang et al. 2010. J-S. Yang, D-X. Yuan, and T-P. Weng. '**Pilot Study of Drinking Water Treatment with GAC, O3/BAC and Membrane Processes in Kinmen Island, Taiwan'**. Desalination, 263 (1–3) (2010), pp. 271-278. *https://doi.org/10.1016/j.desal.2010.06.069*.

Zhang et al. 2015. T-Y. Zhang, B. Xu, C-Y. Hu, Y-L. Lin, L. Lin, T. Ye, and F-X. Tian. '**A Comparison of Iodinated Trihalomethane Formation from Chlorine, Chlorine Dioxide and Potassium Permanganate Oxidation Processes**'. Water Research, 68 (2015), pp. 394-403. *https://doi.org/10.1016/j.watres.2014.09.040*.

Zhao et al. 2017. X. Zhao, J. Ma and U. von Gunten. '**Reactions of Hypoiodous Acid with Model Compounds and the Formation of Iodoform in Absence/Presence of Permanganate**'. Water Research, 119 (2017), pp. 126-135. *https://doi.org/10.1016/j.watres.2017.04.033*.

Zhou et al. 2000. Q. Zhou, S. E. Cabaniss and P. A. Maurice. '**Considerations in the Use of High-Pressure Size Exclusion Chromatography (HPSEC) for Determining Molecular Weights of Aquatic Humic Substances**'. Water Research, 34 (14) (2000), pp. 3505-3514. *https://doi.org/10.1016/S0043-1354(00)00115-9*.

6. Supporting information

Table IV-S1: Information relative to the campaigns performed on the drinking water treatment plant.

Campaigns	A	B	C
Date	10th of April 2018	24th of June 2018	5th of June 2018
Analysis of treated waters	X	X	X
DOC characterization	X	X	X
DBP Formation potential	X	X	
THMs	X	X	X
HAAs			X
Bromide, iodide, iodate	X	X	X
Total iodine		X	

Table IV-S2: Analytical figures of merit for the determination of HAAs and THMs by HS-trap GC-MS in SIM mode.

Compound	Formula	Limit of quantification (ng L^{-1})	Retention time (min)	m/z *
Trihalomethanes				
THM4				
Trichloromethane (chloroform)	$CHCl_3$	3	1.89	47, $\underline{83}$, 85
Bromodichloromethane	$CHBrCl_2$	3	2.73	83, $\underline{85}$, 129
Dibromochloromethane	$CHBr_2Cl$	3	4.17	127, $\underline{129}$, 131
Tribromomethane (bromoform)	$CHBr_3$	3	6.18	171, $\underline{173}$
I-THMs				
Dichloroiodomethane	$CHCl_2I$	0.4	4.75	83, $\underline{85}$, 127
Bromochloroiodomethane	CHBrClI	1	7.21	$\underline{127}$, 129, 131
Dibromoiodomethane	$CHBr_2I$	0.2	10.38	171, $\underline{173}$, 300
Chlorodiiodomethane	$CHClI_2$	0.5	11.42	$\underline{175}$, 177, 302
Bromodiiodomethane	$CHBrI_2$	0.8	14.66	219, $\underline{221}$, 348
Triiodomethane (iodoform)	CHI_3	2	17.2	140, $\underline{267}$, 394
Haloacetic acids				
HAA5				
Chloroacetic acid	$ClCH_2COOH$	50	3.51	$\underline{108}$, 79, 59
Bromoacetic acid	$BrCH_2COOH$	25	4.67	$\underline{152}$, 93, 95
Dichloroacetic acid	$Cl_2CHCOOH$	10	4.91	$\underline{85}$, 83, 59
Dibromoacetic acid	$Br_2CHCOOH$	10	9.97	171, 59, $\underline{173}$
Trichloroacetic acid	$Cl_3CHCOOH$	10	6.94	$\underline{117}$, 59, 119
Others				
Bromochloroacetic acid	BrClCHCOOH	10	7.16	$\underline{129}$, 59, 127
Iodoacetic acid	ICH_2COOH	5	7.10	$\underline{200}$, 169, 141
Internal standards				
1,2-dibromopropane (for THM analysis)			5.35	93, $\underline{121}$, 123
2-Bromopropionic acid (for HAA analysis)			5.38	$\underline{87}$, 109, 135

*Mass used for quantification underlined

Table IV-S3: GC conditions for THMs and for HAAs.

Parameter	Conditions
Column	ZB-5MS Phenomenex® column (30 m × 0.25 mm 1 µm film thickness)
Inlet conditions	Splitless, 200 °C. Column flow rate: controlled by the HS column pressure*
GC temperature program	40 °C for 3 min, 20 °C/min to 60 °C, hold 3 min. 5 °C/min to 100 °C, and 25 °C/min to 200 °C, hold 5 min.
Ion source temperature	250 °C
MS transfer line temperature	290 °C

** The GC-MS was directly connected with the TurboMatrix Headspace 40 Trap, thus bypassing the inlet split of the GC injector. By installing the fused silica tubing in this way, the GC carrier gas is supplied by the HS and the incoming HS sample is not split or diluted in any way.*

Table IV-S4: Headspace trap parameters.

Parameter	**Optimum conditions**
Sample volume	10 mL
thermostatting	10 min, 60 °C
needle temperature	90 °C
transfer line temperature	100 °C
trap load temperature	42 °C
trap desorption temperature	220 °C
number of cycles	1
pressurization time	1 min
decay time	1.3 min
Dry purge time	4 min
desorption time	0.5 min
trap hold	13 min
column pressure	15.6 psi
vial pressure	40 psi
desorption pressure	15.6 psi

For HAAs, each sample should be analyzed 3.5 hours after preparation (allowing sufficient time to hydrolyze the excess of dimethylsulfate) but no more than 5 hours later, to limit methyl ester hydrolysis.

Table IV-S5: HPLC conditions for iodine and bromine speciation by HPLC-ICP-MS.

Injection Volume (µL)	25
Elution flow rate (µL min^{-1})	400
Elution gradient	10 mM NaOH held for 3 min, increased to 20 mM in 1 min, held at 20 mM for 8 min, decreased back to 10 mM in 30 s, held for another 5 minutes for equilibration (total run time : 17.50 minutes).
Targets retention times	IO_3^-: 2 min 50 s Br^-: 5 min 20 s I^-: 12 min 10 s

Table IV-S6: THMFP/DOC measured in both raw and treated waters with the same THMFP test in the literature.

Chlorination conditions	THMFP/DOC raw waters (µg THM mgC^{-1})	THMFP/DOC treated waters (µg THM mgC^{-1})	
7 days, pH 7, 25 °C 3-5 mg L^{-1} free chlorine residual	27.3	21.64-25.53	(Teksoy et al. 2008)
	48.2		(Tubić et al. 2013)
	59-167		(Pifer and Fairey 2014)
	95 (average)	76 (average)	(Yang et al. 2015)
	43-145		(Delpla and Rodriguez 2016)
	149	71	(Golea et al. 2017)

Table IV-S7: Average retention by the activated carbon filtration and standard deviation observed for the 3 campaigns of measurement for THMs and percentage obtained for campaign B for HAAs. Chemical and Physical properties of THMs and HAA when available.

Species	**Average removal during GAC filtration (%)**	**Relative standard deviation (3 campaigns) (%)**	**MW (g mol^{-1})**	**Solubility (g L^{-1})**	**log K$_{ow}$**
Trihalomethanes					
$CHCl_3$	15	17	119.4	8.0	1.97
$CHBrCl_2$	18	10	163.8	4.0	2.0
$CHBr_2Cl$	43	9	208.3	2.7	2.16
$CHBr_3$	58	15	252.7	3.1	2.4
∑ THM4	**31**	**6**			
$CHCl_2I$	77	8	210.8	0.717 (a)	2.03 (a)
CHBrClI	88	6	255.3	0.346 (a)	2.11 (a)
$CHBr_2I$	94	5	299.7	0.162 (a)	2.20 (a)
$CHClI_2$	96	2	302.3	0.082 (a)	2.53 (a)
$CHBrI_2$	Not present	-	346.7	0.038 (a)	2.62 (a)
CHI_3	Not present	-	393.7	0.100	3.03 (a)
∑ I-THMs	**87**	**6**			
Haloacetic acids					
Chloroacetic acid	91	-	94.5	miscible	0.22
Bromoacetic acid	97	-	138.9		0.41
Dichloroacetic acid	96	-	128.9		0.92
Dibromoacetic acid	97	-	217.8		0.70
Trichloroacetic acid	85	-	163.4		1.33
Bromochloroacetic acid	96	-	173.4		0.61
Iodoacetic acid	>91 (b)	-	185.9		0.85
∑ HAAs	95	-			

References: Pubchem Open database
(a) Environ. Sci. Technol. 2018 52(22): 13047-13056. doi: 10.1021/acs.est.8b04625.
(b) Calculated with the limit of quantification

Table IV-S8: Individual LC 50 chronic cytotoxicity (Lethal Concentration for 50% of a population) from Wagner and Plewa, 2017).

Individual cytotoxicity					
Trihalomethanes	**LC50 (M)**	**LC50 (µg L^{-1})**	**Haloacetic acids**	**LC50 (M)**	**LC50 (µg L^{-1})**
$CHBr_3$	3.96×10^{-3}	15.7	Chloroacetic acid	8.10×10^{-4}	8.57
$CHBr_2Cl$	5.36×10^{-3}	25.7	Bromoacetic acid	9.60×10^{-6}	0.07
$CHBrCl_2$	1.15×10^{-2}	70.2	Dichloroacetic acid	7.30×10^{-3}	56.6
$CHCl_3$	9.62×10^{-3}	80.6	Trichloroacetic acid	2.40×10^{-3}	14.7
CHI_3	4.47×10^{-5}	0.11			
$CHBrI_2$	1.91×10^{-3}	5.5	Dibromoacetic acid	5.90×10^{-4}	2.7
$CHClI_2$	2.40×10^{-3}	7.9	Iodoacetic acid	2.95×10^{-6}	0.02
$CHBr_2I$	1.40×10^{-3}	4.7	Bromochloroacetic acid	7.78×10^{-4}	4.5
$CHCl_2I$	4.13×10^{-3}	19.6			
$CHBrClI$	2.42×10^{-3}	9.5			

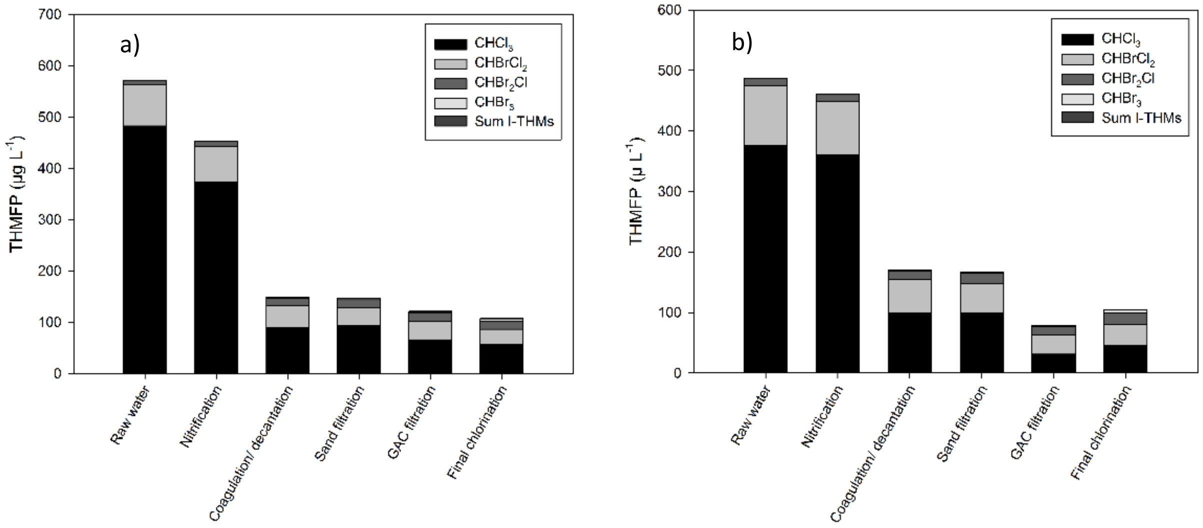

Figure IV-S1: THMFP reduction across the treatment line. a) campaign A and b) campaign B.

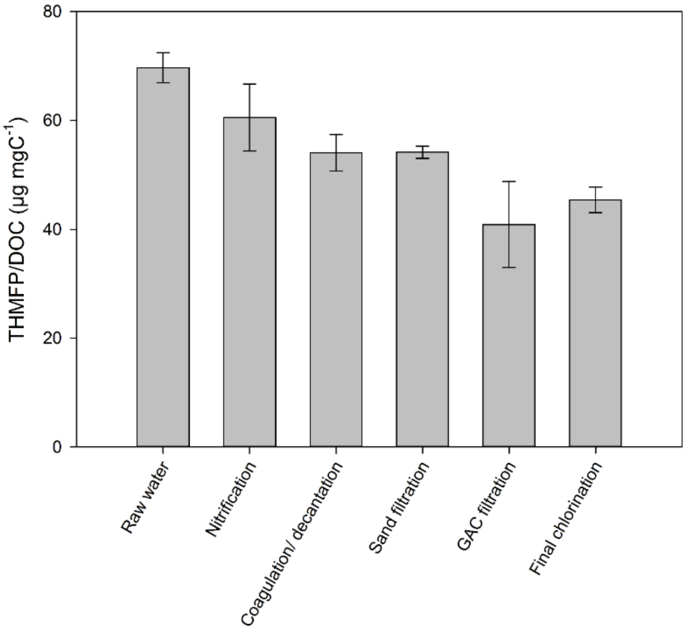

Figure IV-S2: Average evolution of the THMFP/DOC across the treatment line for campaigns A and B.

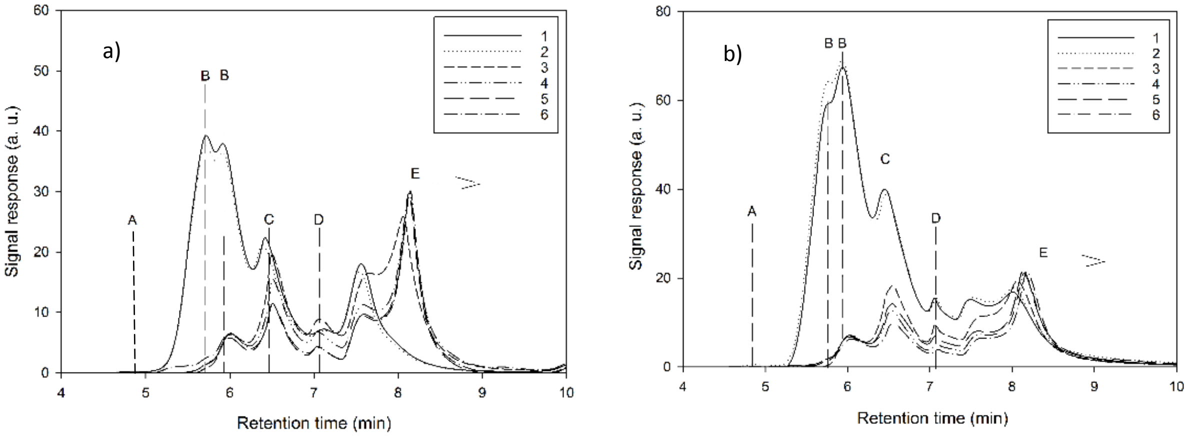

Figure IV-S3: Chromatograms of waters samples obtained by size exclusion chromatography with UV detection at 254 nm. a) campaign C, and b) campaign B. 1: raw water, 2: after nitrification, 3: after coagulation-decantation, 4: after intermediate chlorination and sand filtration, 5: after GAC filtration, 6: after final chlorination. A: biopolymers, B: humic substances, C: building blocks, D: low molecular weight acids, E: low molecular weight neutrals.

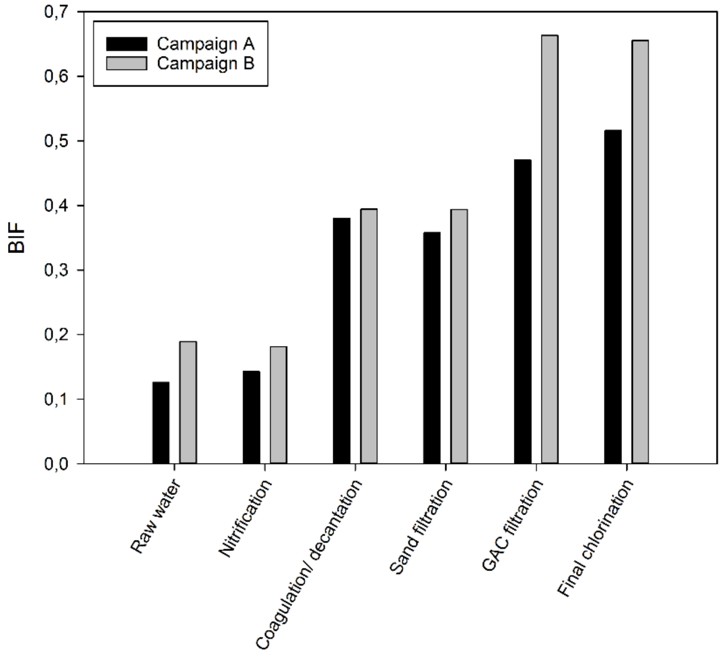

Figure IV-S4: Bromine incorporation factor (BIF) evolution along the treatment line.

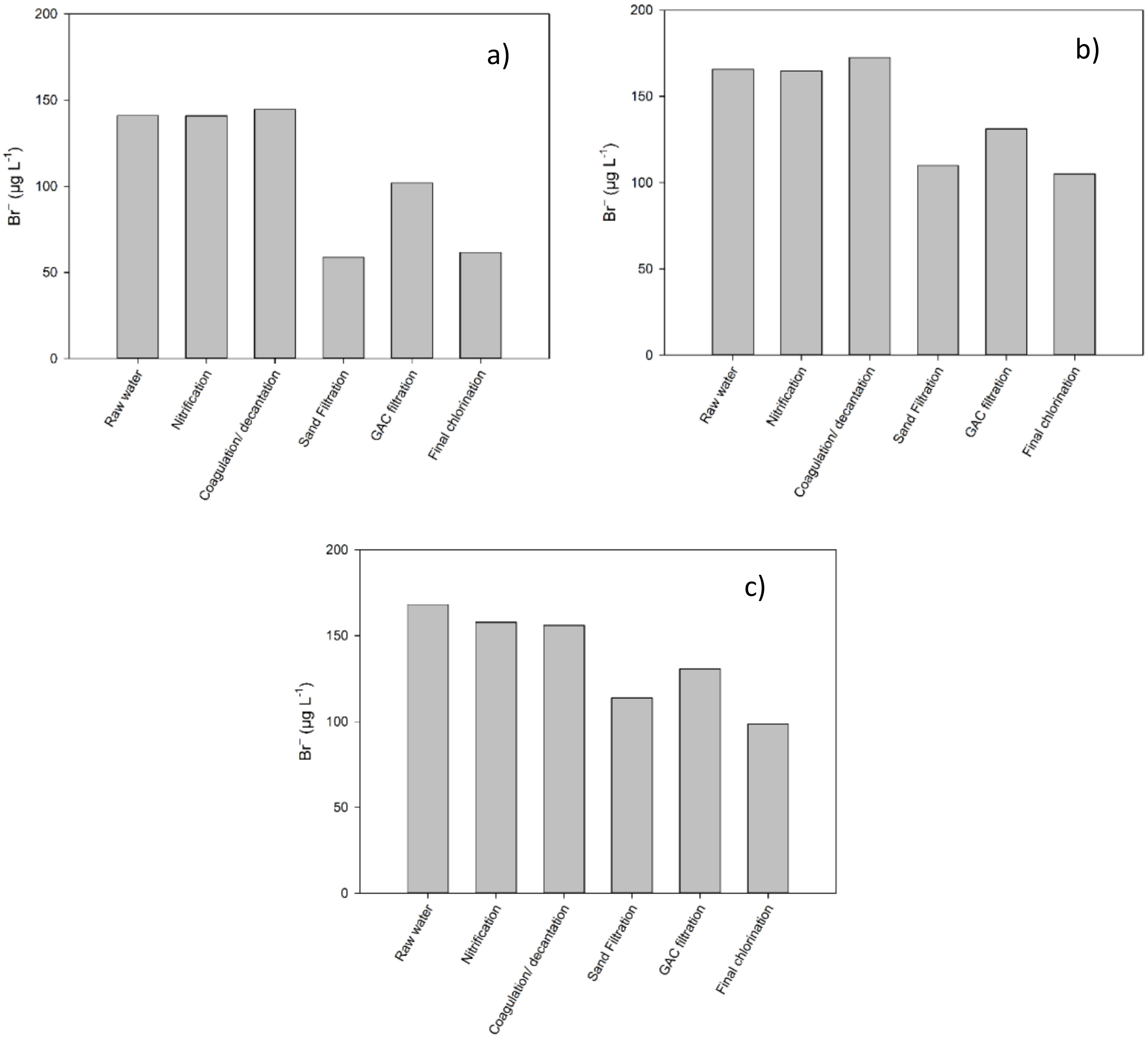

Figure IV-S5: Bromide evolution after the different treatment steps. a) campaign A, b) campaign B, c) campaign C.

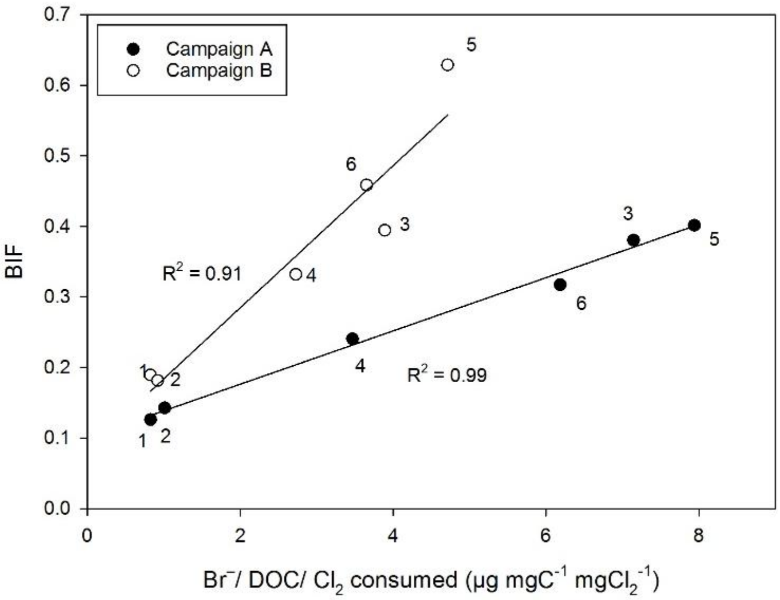

Figure IV-S6: Evolution of the bromine incorporation factor (BIF) for ΔTHMFP as a function of the Br^-/DOC concentration ratio and chlorine consumption. 1: Raw water, 2: after nitrification, 3: after coagulation-decantation, 4: after intermediate chlorination and sand filtration, 5: after GAC filtration, 6: after final chlorination.

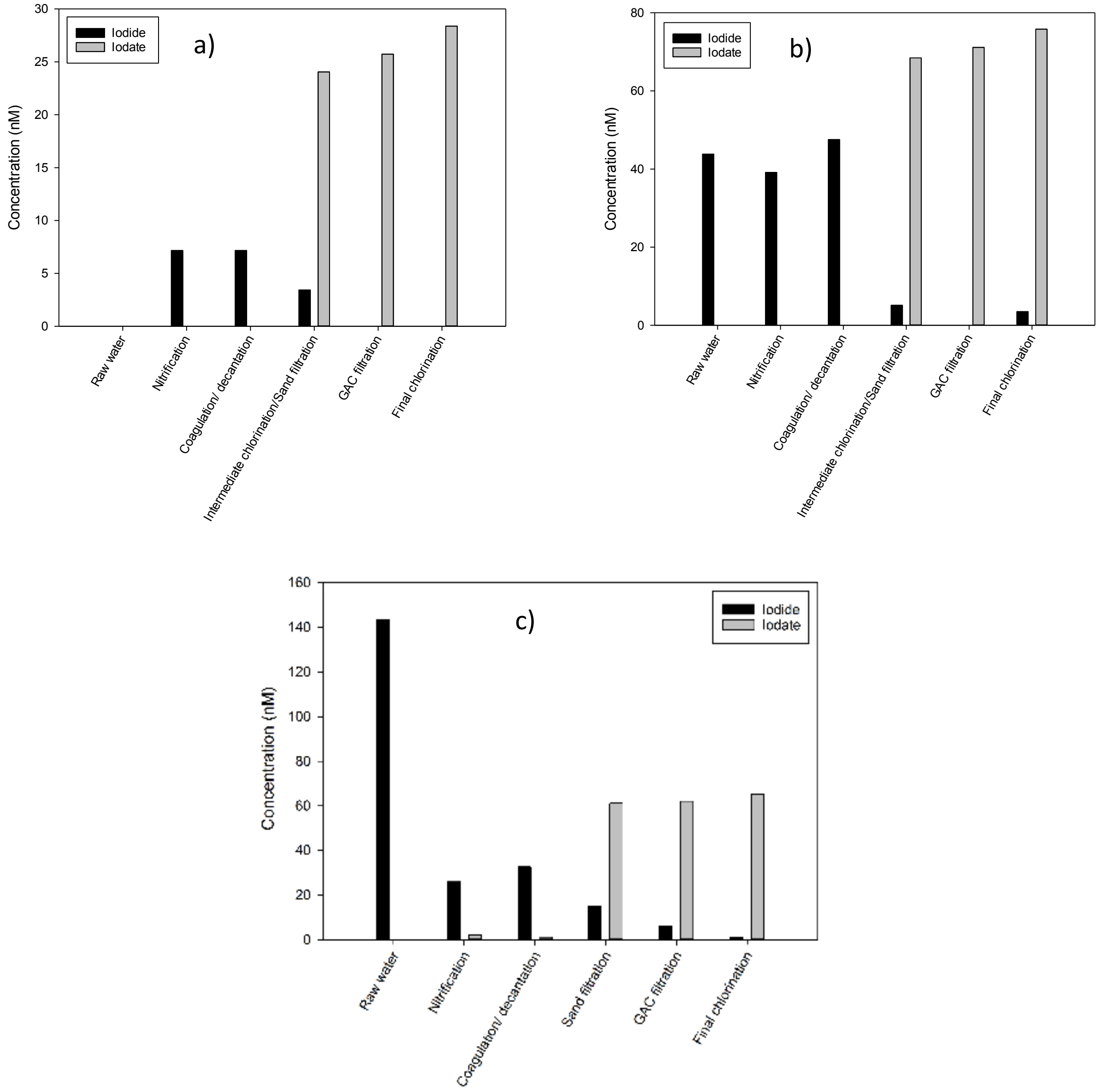

Figure IV-S7: Iodide and iodate evolution across the treatment line. a) campaign A, b) campaign B, c) campaign C.

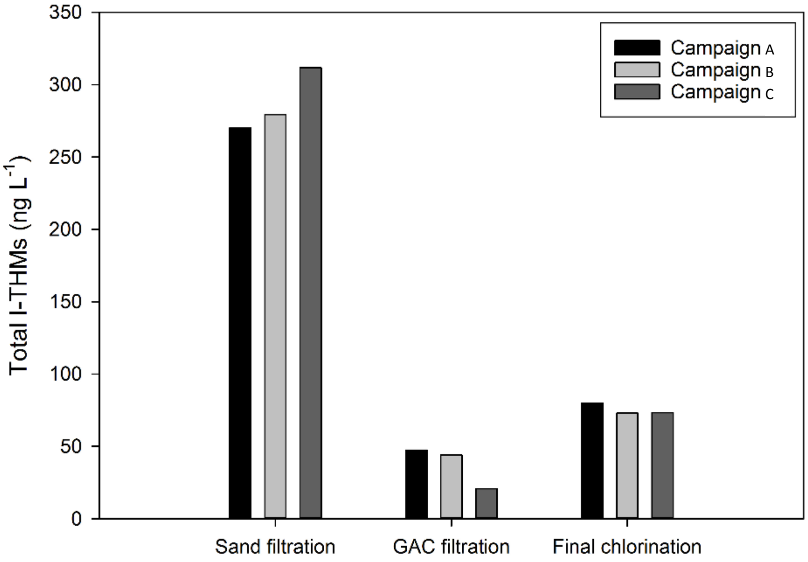

Figure IV-S8: Total I-THMs in the last 3 treatment steps for all 3 campaigns.

7. Bibliography for supporting information

Bauman and Stenstrom 1990. L. C. Bauman, and M. K. Stenstrom. '**Removal of Organohalogens and Organohalogen Precursors in Reclaimed Wastewater—I**'. Water Research, 24 (8) (1990), pp. 949-955. *https://doi.org/10.1016/0043-1354(90)90116-N*.

Delpa and Rodriguez 2016. I. Delpla, and M. J. Rodriguez. '**Experimental Disinfection By-Product Formation Potential Following Rainfall Events**'. Water Research, 104 (2016), pp. 340-348. *https://doi.org/10.1016/j.watres.2016.08.031*.

Golea et al. 2017. D. M. Golea, A. Upton, P. Jarvis, G. Moore, S. Sutherland, S. A. Parsons, and S. J. Judd. '**THM and HAA Formation from NOM in Raw and Treated Surface Waters**'. Water Research, 112 (2017), pp. 226-235. *https://doi.org/10.1016/j.watres.2017.01.051*.

Hyung Kim and Yu 2005. M. Hyung Kim and M. J. Yu. '**Characterization of NOM in the Han River and Evaluation of Treatability Using UF–NF Membrane**'. Environmental Research, 97 (1) (2005), pp. 116-123. *https://doi.org/10.1016/j.envres.2004.07.012*.

Musikavong et al. 2005. C. Musikavong, S. Wattanachira, T. Marhaba, and P. Pavasant. '**Reduction of Organic Matter and Trihalomethane Formation Potential in Reclaimed Water from Treated Industrial Estate Wastewater by Coagulation**'. Journal of Hazardous Materials, 127 (1–3) (2005), pp. 48-57. *https://doi.org/10.1016/j.jhazmat.2005.06.042*.

Pifer and Fairey 2014. A. D. Pifer and J. L. Fairey. '**Suitability of Organic Matter Surrogates to Predict Trihalomethane Formation in Drinking Water Sources**'. Environmental Engineering Science, 31 (3) (2014), pp. 117-126. *https://doi.org/10.1089/ees.2013.0247*.

Rakram and Wattanachira 2014. P. Rakruam and S. Wattanachira. '**Reduction of DOM Fractions and Their Trihalomethane Formation Potential in Surface River Water by In-Line Coagulation with Ceramic Membrane Filtration**'. Journal of Environmental Sciences, 26 (3) (2014), pp. 529-536. *https://doi.org/10.1016/S1001-0742(13)60471-4*.

Singer et al. 1982. P. C. Singer, J. J. Barry III, G. M. Palen, and A. E. Scrivner. '**Trihalomethane Formation in North Carolina Drinking Waters**'. Water Resources Research Institute of the University of North Carolina (1982), 127 p.

Teksoy et al. 2008. A. Teksoy, U. Alkan, and H. S. Başkaya. '**Influence of the Treatment Process Combinations on the Formation of THM Species in Water**'. Separation and Purification Technology, 61 (3) (2008), pp. 447-454. *https://doi.org/10.1016/j.seppur.2007.12.008*.

Tubić et al. 2013. A. Tubić, J. Agbaba, B. Dalmacija, J. Molnar, S. Maletić, M. Watson, and S. U. Perović. '**Insight into Changes during Coagulation in NOM Reactivity for Trihalomethanes and Haloacetic Acids Formation**'. Journal of Environmental Management, 118 (2013), pp. 153-160. *https://doi.org/10.1016/j.jenvman.2012.11.046*.

Yang et al. 2010. J-S. Yang, D-X. Yuan, and T-P. Weng. '**Pilot Study of Drinking Water Treatment with GAC, O3/BAC and Membrane Processes in Kinmen Island, Taiwan**'. Desalination, 263 (1–3) (2010), pp. 271-278. *https://doi.org/10.1016/j.desal.2010.06.069*.

Yang et al. 2015. L. Yang, D. Kim, H. Uzun, T. Karanfil, and J. Hur. '**Assessing Trihalomethanes (THMs) and N-Nitrosodimethylamine (NDMA) Formation Potentials in Drinking Water Treatment Plants Using Fluorescence Spectroscopy and Parallel Factor Analysis**'. Chemosphere, 121 (2015), pp. 84-91. *https://doi.org/10.1016/j.chemosphere.2014.11.033*.

V. Comparison of different ion exchange resins for the removal of disinfection by-products precursors

V. Comparison of different ion exchange resins for the removal of disinfection by-products precursors

With contributions from Klaas Schoutteten[1,2] Junias Adusei Gyamfi[3]

[1]Ghent University, Faculty of Bioscience Engineering, Department of Green Chemistry and Technology, Coupure Links 653, 9000 Gent, Belgium.

[2]De Watergroep, Vooruitgangstraat 189, 1030 Brussels, Belgium.

[3]Univ. Lille CNRS, UMR 8516 – LASIR, Equipe Physico-Chimie de l'Environnement, F-59000, Lille, France.

1. Introduction

The most commonly applied methods for removal of NOM from water are coagulation (Matilainen et al. 2010), adsorption (Sillanpää and Bhatnagar 2015), membrane filtration (Sillanpää et al. 2015), advanced oxidation processes (AOPs) (Matilainen et al. 2010). Ion exchange processes (IEX) – where negatively charged NOM fractions are removed by reversible exchange of a counter ion, usually chloride, from the anionic resin surface (Bolto et al. 2004) – were introduced more recently, with Levchuk et al. (2018) presenting a summary of the studies on NOM removal by ion exchange in the last 10 to 15 years in a recent review. The first industrial applications considered the magnetic ion exchange resin (MIEX) early in the 2000s. More recently, other types of processes such as suspended ion exchange (SIX) (Galjaard, et al. 2010) and fluidized ion exchange (FIX) (Cornelissen et al. 2009) have been explored as well.

The raw waters at De Blankaart treatment plant have relatively high concentrations of DOC, usually in the 10-15 mg L^{-1} range. The conventional coagulation/decantation treatment step at the plant thus requires very high and costly amounts of coagulant in order to lower the DOC before further treatment steps. An IEX pilot plant has been in operation since 2015 to test the feasibility of adding an IEX treatment step in the treatment line.

Ion exchange is indeed of interest as a pre-treatment before coagulation as it has been shown to significantly reduce the coagulant dose required (by 50 to more than 80%) (Singer and Bilyk 2002; Fearing et al. 2004; Grefte et al. 2013), as well as increase the floc size and strength (Jarvis et al. 2008). IEX is also more efficient than coagulation for the removal of charged organic compounds (Bolto et al. 2002). Using IEX in combination with coagulation, can thus remove both a higher amount of total DOC (Humbert et al. 2008), and a broader spectrum of NOM precursors, with better removal of hydrophilic and low molecular weight fractions (LMW) fractions more difficult to remove using conventional coagulation treatment (Singer and Bilyk 2002; Drikas et al. 2003; Fearing et al. 2004; Allpike et al. 2005; Tan and Kilduff 2007; Mergen et al. 2009) and hence enhance reduction in DBP formation potential (DBPFP) (Singer and Bilyk 2002; Boyer and Singer 2005; Watson et al. 2015). Furthermore, anionic resins can also remove bromide from certain water sources with low alkalinity (Singer and Bilyk 2002), thereby potentially reducing the formation of brominated DBPs during disinfection.

De Watergroep was interested in comparing 4 different ion exchange resins for the optimisation of the pilot plant. In the past, the removal efficiency of NOM by IEX resins was mainly studied through measuring color and NOM characteristics such as DOC, TOC, UV absorbance at 254 nm (Croué et al. 1999; Bolto et al. 2002; Humbert et al. 2005) and more recently NOM fraction removal (Grefte et al. 2013). While these studies offer useful guidance in understanding the removal of NOM fractions, less has been done on DBP formation potential, with studies focusing on THMFP and HAAFP (Tan et al. 2005; Boyer and Singer 2005; Brezinski et al. 2019). In this study, the efficiency of the four resins to remove NOM was compared in terms of DOC, UV absorbance and NOM fraction removal. Following ion exchange fluidized bed column tests, DBP formation potential tests were carried out to compare precursor removal and bromine and iodine incorporation under different halide concentration scenarios.

2. Material and Methods

2.1. Water resource and pilot plant

2.1.1. Raw water

The raw water is drawn from the river Iser (approximately 80%) (De Watergroep 2015) downstream of the town of Ypres and two wastewater treatment plants and from surrounding lowlands (approximately 20%) (De Watergroep 2015), and stored in a 3 million m³ reservoir.

The water is characterized by a high NOM content, with an average DOC content of 12 mgC L^{-1}, combined with a high alkalinity and hardness. The Belgian part of the hydrographic basin of the Iser river covers an area holding 110,000 inhabitants, and a largely agrarian activity, *i.e.* intensive farming and intensive cattle breeding (Baert et al. 1996).

2.1.2. Ion exchange pilot

Due to the combination of high DOC concentration and relatively high alkalinity, the water requires large amounts of flocculant and sulfuric acid during coagulation (Verdickt et al. 2012). De Watergroep has been studying the possibility of replacing the enhanced coagulation and decantation treatment step by a combination of IEX, coagulation and flotation (Figure V-1). A fluidized ion exchange pilot plant has been in operation since the end of 2015 on a scale of 50 m^3 h^{-1}, with resin PPA860S (Verdickt and Schoutteten 2018).

2.1.3. Column experiment and ion exchange resins

A column experiment was performed, as it is easier using a column than a jar-test to translate the results to the full scale. The four resins tested were Purolite PPA860S, Amberlite IRA410, Amberlite IRA958 and Dowex TAN-1. Three of the resins are strong base type I resins with a quaternary ammonium group, and one is a strong base type II resin where a methyl group is replaced with an alcohol group. Their main properties are presented in Table V-1.

Table V-1: Properties of the four anion exchange resins selected.

Trade Name	**Resin Type**	**Matrix/ Structure**	**Functional Group**	**Mean Particle Size (µm)**	**Water Content (%)**	**Total Exchange Capacity (eq.L^{-1})**	**Manufacturer**
TAN-1	MP	Styrene-DVB	Quaternary amine	420-1200	70-82	0.7	Dowex
IRA410	Gel	Styrene-DVB	Dimethyl ethanol ammonium	600-750	45-51	≥1.25	Rohm and Haas
IRA958	MP	Crosslinked acrylic	Quaternary ammonium	630-850	66-72	≥0.80	Rohm and Haas
PPA860S	MP	Polyacrylic crosslinked with DVB	Quaternary ammonium	740	66-72	0.8	Purolite

DVB: divinylbenzene; MP: macroporous; Reference: https://www.lenntech.com/Data-sheets

The experiment took place in October 2018. Four parallel transparent PVC columns, with an internal diameter of 86.4 mm, were used for performing breakthrough experiments. The resins were pre-treated extensively to avoid possible leaching of residual monomers, and to avoid irreversible adsorption of NOM occurring on virgin resins. The procedure consisted of a jar pre-treatment and a dummy loading run in the columns. For the jar pre-treatment, the following regeneration procedure was conducted three times consecutively: regeneration using 3 bed volumes (BV) of a 10% NaCl brine followed by decantation, and a triple rinsing with 3 BV of tap water followed by decantation.

After the jar pretreatment procedure, the resins were transferred inside the columns, and a dummy loading run was performed using 500 BV of filtered raw water at an empty bed contact time (EBCT) of 1.73 min, followed by a regeneration with 1 BV of 10% NaCl at an EBCT of 30 minutes and a rinse with 1 BV of tap water at an EBCT of 30 minutes. The latter regeneration procedure coincides with practices in IEX pilot scale research conducted at De Watergroep (Verdickt et al. 2012). After this extensive pretreatment procedure, the resins were considered to be suitable to perform a representative IEX cycle.

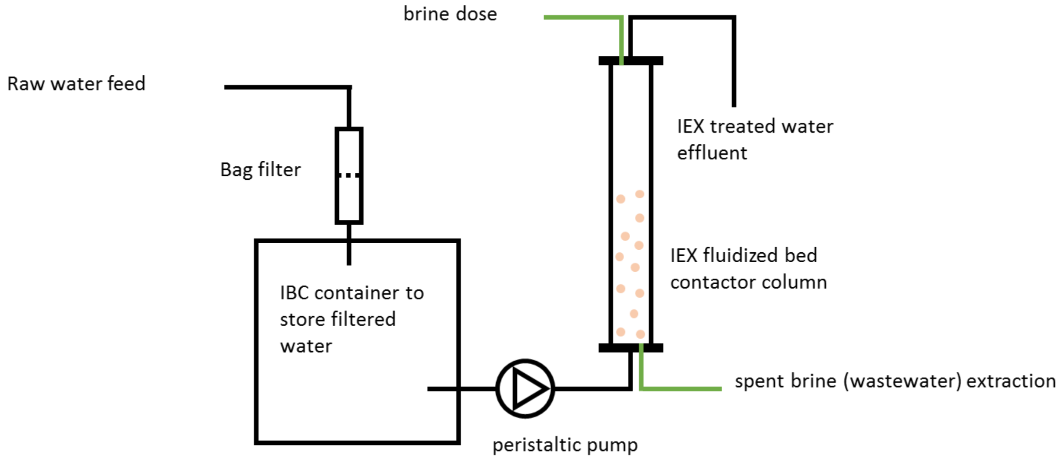

Figure V-1: IEX column test scheme. In green are the regeneration lines.

The breakthrough experiments were run in fluidized bed modus (*i.e.* bottom – up, see Figure V-1) using raw water from the De Blankaart reservoir which had been prefiltered over a 25 µm bag filter (Filtration CVBA, Drongen, Belgium), at an EBCT of 1.73 min. Samples were taken at 0 BV and every recurring 50 BV, until 500 BV of raw water were treated. To simulate the average treated water quality which would be obtained after 500 BV, a mixed effluent sample was created by mixing equal volumes (208 mL) of each of the 11 taken samples for each resin, for the DBP experiments.

2.2. Sample collection and analysis

The sample collection and analysis were performed according to the procedures detailed in Chapter II.

2.3. Chlorination tests

The THM_{7d} tests (using a 7-day chlorination procedure) presented in Chapter II were used to compared the different treated waters.

In addition, to avoid the bias by the large amount of chlorine used in the 7-day tests on iodine speciation, chlorination experiments with different halide concentrations have been performed with a lower chlorine dose, *i.e.* 0.3 mg Cl_2 L^{-1} residual after a contact time of 24 h.

The THM_{24h} and HAA_{24h} formation potential tests were carried out in the same manner, except that the chlorine dose was lower so as to reach a chlorine residual of 0.3 mg L^{-1} and the incubation time and temperature were of 24 h and 20 °C respectively. The following concentrations and ratios of bromide and iodide have been used as well as the samples with no spiking (Table V-2).

Table V-2: Range of iodide and bromide concentrations after spiking.

I^- (µg L^{-1})	7	63	63	63	63	254
Br^- (µg L^{-1})	228	228	400	800	2200	228
Br^-/I^- (molar ratio)	50	5.7	10	20	55	1.4

3. Effect of ion exchange treatment on NOM and inorganic compounds

3.1. Raw water composition

The DOC of the raw water was relatively high, around 12 mgC L^{-1} (Table V-4), but at a typical level for this site. The SUVA was moderate, around 2 (Table V-6) indicating the presence of hydrophilic and low molecular weight compounds (Edzwald and Tobiason 1999), as well as some hydrophobic and high molecular weight compounds.

3.1.1. Natural organic matter fractions

LC-OCD fractionation indicated that humic substances (HS) dominated (40%), followed by hydrolysates of biopolymers, building blocks (BB) and low molecular weight neutrals (LMWN), these 3 fractions each representing approximately 14% of the DOC. No low molecular weight acids (LMWA) were found in the feed water by LC-OCD. Additionally, around 15% of the DOC is not eluted through the column presumably due to hydrophobic interactions; this represents the so-called hydrophobic organic carbon fraction (HOC).

The results of fractionation performed on a higher resolution column by LASIR (Chapter II 3.1.2.), but only with UV detection, are in agreement with these results except for the slight presence of absorbing low molecular weight acids in the water. The biopolymer fraction was however not detected as this fraction usually does not absorb in the UV range (Huber et al. 2011).

3.1.2. Inorganic compounds

The further characterization of the water samples has been performed by De Watergroep. The mineralization of the water is relatively high, mainly with calcium, chloride, sulfate and bicarbonate, while low nitrate and iron levels were detected (Table V-3). The concentration of ammonia in the raw water was 0.16 mg L^{-1}.

Table V-3: Evolution of some of the main cations and anions.

	Br^-	I^-	Cl^-	Na^+	NH_4^+	NO_2^-	NO_3^-	Ca^{2+}	SO_4^{2-}	HCO_3^-
	µg L^{-1}	µg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}
Raw water	195	7.2	147	105	0.16	0.01	-	71.9	104	247
IRA410	196	1.1	264	123	0.17	0.03	1	71.3	11	227
IRA958	228	7.0	305	185	0.17	0.06	-	73.1	58	243
TAN-1	215	3.2	270	147	0.28	0.03	-	71.9	52	232
PPA860S	208	5.9	220	112	0.26	0.05	-	72.1	49	226

3.2. Effect of ion exchange on water characteristics

3.2.1. Dissolved organic carbon removal

The treatment of raw water by ion exchange removed from 23 up to 41% of the dissolved organic carbon content depending on the type of resin (Table V-4). The best removal is obtained with the type II strong base resin IRA410, a styrene divinyl benzene type with a dimethyl ethanol ammonium functional group. This resin has the best total exchange capacity according to the manufacturer information (Table V-1). This parameter however does not explain the difference of removal rates obtained for the 3 other resins, which are all type I resins with a quaternary ammonium group.

Table V-4: DOC measurements by Het Laboratorium.

Water samples	DOC	
	mgC L^{-1}	% decrease
Raw water	11.8	
IRA410	6.9	41
IRA958	9.1	23
TAN-1	7.8	34
PPA860S	7.5	37

The DOC removal by such resins is usually in the 40-70% range (See below section 3.3.1).

3.2.1.1. Removal of NOM fractions

Considering the removal of the different fractions of NOM, the humic substance fraction is the main one removed by the different IEX resins (from 41 to 72%), followed by the building block fraction (12-41%) and LMWN fraction (13-36%) (Figure V-2 and Table V-5). After IEX treatment, the concentration of the biopolymer fraction was found to slightly increase by 6-32% (see Table V-5), which may have been caused by the release of polymeric IEX material (Cornelissen et al. 2008).

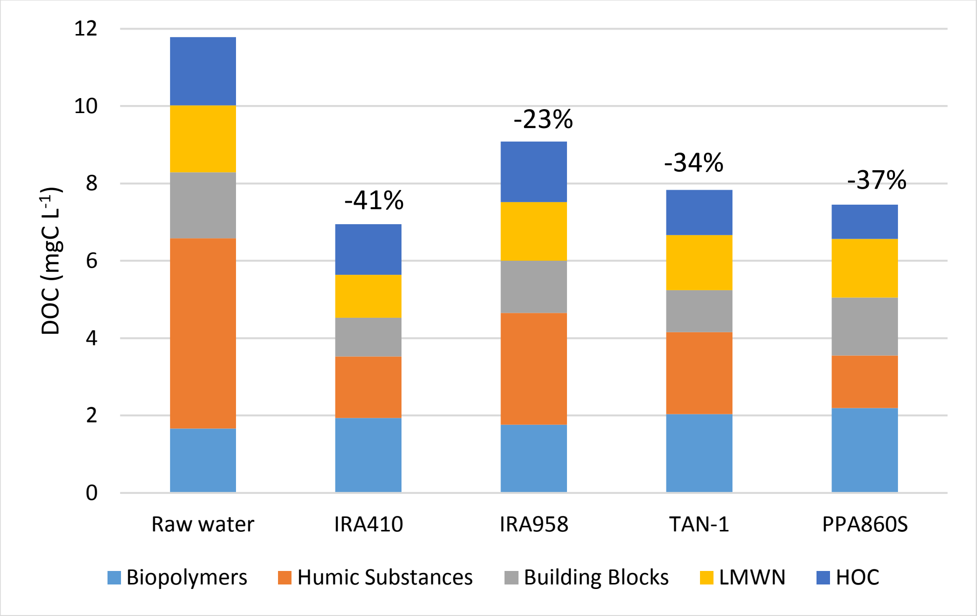

Figure V-2: Representation of NOM fraction removal by the 4 resins.

These results are in agreement with previous studies that show effective removal of high charge density organic fractions such as HS and less effective removal of BB, LMWN, and no removal of biopolymers due to size exclusion (Bond et al. 2010; Cornelissen et al. 2010; Huber et al. 2011; Grefte et al. 2013). Nevertheless, in the future treatment plant scheme, if SIX is added before coagulation, this biopolymer fraction can be expected to be greatly removed by the post-coagulation step (Galjaard et al. 2018).

Table V-5: Percentage of removal of the different NOM fractions by the four resins based on LC-OCD results.

NOM fraction	IRA410	IRA958	TAN-1	PPA860S
Biopolymers	-17%	-6%	-22%	-32%
Humic substances	68%	41%	57%	72%
Building blocks	41%	21%	36%	12%
Low molecular weight neutrals	36%	13%	18%	13%
Hydrophobic organic carbon	26%	11%	34%	50%

Removal of HS and BB fractions by the anionic resins can mainly be explained by ionic interactions. Removal of the LMWN could possibly be explained by physical adsorption onto the resin material (Cornelissen et al. 2008).

The most effective resin types for the removal of HS are Purolite PPA860S (72% removal) and Amberlite IRA410 (68%). The most effective removal of BB and LMWN fractions is with resin type Amberlite IRA410 (41% and 36% removal respectively).

Although the IRA410 removed much of the HS fraction, the molecular weight of this fraction remained approximately the same (-5% around 500 g mol^{-1}). The removed part of the HS seems to be both low and high molecular weight compounds. The molecular weight of HS of waters treated by the other resins decreased by 15-22% with the lowest MW for PPA860S (400 g mol^{-1}), demonstrating the removal of higher molecular weights HS.

3.2.1.2. Removal of UV absorbing compounds

The removal in UV254 absorbing compounds (40-66%) followed the same trend as the DOC removal, with IRA410 the most efficient and IRA958 the least efficient amongst the 4 resins (Figure V-3).

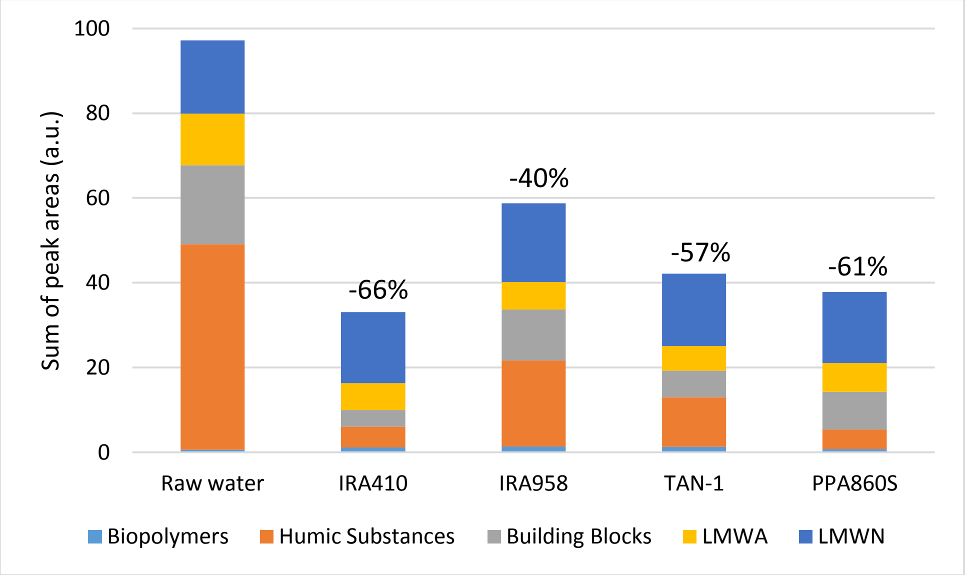

Figure V-3: Decrease in peak areas of UV absorbing compounds by the 4 resins.

Table V-6: Percentage decrease of the UV254 absorbing compounds in the different NOM fractions, overall UV254 and SUVA by the four resins.

NOM fraction	IRA410	IRA958	TAN-1	PPA860S
Biopolymer	-78%	-118%	-104%	-27%
Humic substance	90%	58%	76%	91%
Building blocks	79%	35%	67%	52%
Low molecular weight acids	48%	47%	52%	44%
Low molecular weight neutrals	3%	-7%	1%	3%
UV254	77%	47%	65%	68%
SUVA	62%	32%	47%	50%

The main UV254 absorbing fraction removed was the HS fraction (58-91%), followed by the BB fraction (35-79%), and the LMWA fraction (44-52%), while the LMWN were not removed (Table V-6). A slight increase in the biopolymer peak areas was also observed here.

It can be seen from Table V-6 that resins PPA860S, IRA410, and TAN-1 removed more (approximately 70%) of the UV254 absorbing organics than IRA958 (approximately 45%).

The SUVA was lowered by the IEX treatment system by 32-62%, indicating preferential removal of higher-SUVA components. The final SUVA values (0.75-1.34) indicate that the remaining NOM is mainly hydrophilic (Edzwald and Tobiason 1999).

3.2.2. Removal of cations and anions

The cation removal for all the resins was indeed low or inexistent as these resins are designed for the exchange of anions. Considering iron, while its concentration is quite low in the raw water (15 µg L^{-1}), the concentrations after treatment are systematically below that level (by 13-33%); its retention possibly happens by complexation with humic substances (Cornelissen et al. 2010). Conversely, the sodium concentrations increased in the range (6-76%), which can be probably linked to the incomplete washing of the resins after regeneration by NaCl.

Table V-7 shows a significant increase in Cl^- concentrations after ion exchange treatment for all resins. However, considering the increase of sodium concentrations (0.28-3.47 meq L^{-1}), and assuming that the same amount of Cl^- would originate from the washing of the resin, a substantial part of chloride increase is then not directly linked to the ion or NOM exchange. The amount of chloride attributable to the ion exchange is highly variable between resin types (0.99-2.53 meq L^{-1}) (Table V-7).

Table V-7: Change in ion concentration presented on an equivalent base.

	Na^+		Cl^-		SO_4^{2-}		HCO_3^-		Cl^- exchanged with DOC	DOC
Resin	C (mg L^{-1})	Δ (meq L^{-1})	C (mg L^{-1})	Δ (meq L^{-1})	C (mg L^{-1})	Δ (meq L^{-1})	C (mg L^{-1})	Δ (meq L^{-1})	Δ (meq L^{-1})	charge density (meq gC^{-1})
Raw water	105	-	147	-	104	-	247	-	-	-
IRA410	123	0.77	264	3.30	11	-1.94	227	-0,34	0.25	52
IRA958	185	3.47	305	4.46	58	-0.96	243	-0,08	-0.05	-19
TAN-1	147	1.80	270	3.47	52	-1.08	232	-0,26	0.32	82
PPA860S	112	0.28	220	2.06	49	-1.15	226	-0,36	0.27	63

C: concentration; Δ: change in concentration after IEX treatment

Indeed, sulfate is an important competing ion in De Blankaart waters, with 44 to 89% sulfate removal for all 4 resins (Table V-7). Sulfate ions are smaller than NOM molecules, and hence the ion exchange process is kinetically faster for this ion (Cornelissen et al. 2010). This explains why a major part of the practical exchange capacity of the resin goes to the removal of sulfate (0.96-1.94 meq L^{-1}, which represents 64-97% of the actual ion exchange), and another small part to bicarbonate (0.08-0.36 meq L^{-1}). To conclude, only 10-19% (0.25-0.32 meq L^{-1}) of the total amount of chloride exchanged could be attributed to NOM removal for 3 of the 4 resins, so the charge density of the organic matter can be estimated to be around 52-82 meq gC^{-1}. These values are higher than those reported, ranging usually between 8 and 11 meq gC^{-1}, by Boyer and Singer (2006), Cromphout et al. (2008), and – for the same water source with MIEX resin – Verdickt et al. (2012).

For IRA958, the calculated amount of chloride exchanged for NOM was negative (-0.05 meq L^{-1}) and close to zero, likely due to the uncertainties on the analytical measurements of the ion concentrations. Verliefde et al. (2009) also found a negative amount of chloride exchanged for NOM (-0.18 meq L^{-1}), for PPA860S.

Finally, the concentration of chloride is higher than the European quality guideline of 250 mg L^{-1} for waters treated by three of the four resins in the conditions used here. This parameter would have to be carefully studied for the upscaling of this process as the increasing chloride concentration could increase the corrosive character of treated waters.

Bromide can be removed on MIEX resins, as has been shown in several studies (Watson et al. 2015). However, for De Blankaart waters, bromide was not removed and even increased by up to 17%, perhaps due to some bromide contamination during resin regeneration (bromide being a common contaminant of commercially available NaCl). The bromide anions may also have been outcompeted for binding sites by the other anions (Watson et al. 2015).

Iodide adsorption onto the resin was also observed (18-85%) for IRA410, TAN-1 and PPA860S (Table V-3), which has not been previously reported in the literature for these 3 resins. IRA958 was not able to significantly remove iodide (3% reduction).

3.3. Comparison to literature

3.3.1. Previous studies on the 4 resins

Relatively few studies exist on the 4 resins which were used in this study, except for IRA958. They are grouped in Table V-8. Some studies on Purolite A860, similar to Purolite PPA860S are presented as well.

Table V-8: Summary of studies investigating at least one of the 4 resins.

Water	Resin	Scale	pH	DOC mgC L^{-1}	UV m^{-1}	SUVA L m^{-1} mg^{-1}	SO_4^- mg L^{-1}	Removal % (parameter)	Reference
Reconstituted aquatic waters	IRA410	batch	7	4-5 (TOC)	NR	NR	NR	77 (UV254)	(Bolto et al. 2002, 2004)
SW	IRA410	column	NR	11.8	23	2	104	41 (DOC) 77 (UV) 62 (SUVA)	This experiment
Reconstituted aquatic waters	IRA958	batch	7	TOC = 4-5	NR	NR	NR	73 and 90 (UV254)	(Bolto et al. 2002, 2004)
Synthetic water	IRA958	batch	7.70-7.83	9.2-9.9	NR	NR	NR	90 (DOC)	(Boyer and Singer 2008)
SW	IRA958	batch	7-7.9	5.6-6.7	14.0-16.0	2.2-2.9	13-29	80 (DOC) 59 (SUVA)	(Humbert et al. 2008)
								47 (DOC) 21 (SUVA)	
								13 (DOC) 3 (SUVA)	
WW effluent	IRA958	column	7	8.53 (TOC)	12.5	1.47	NR	10 (SUVA) 47 (UV) 42 (TOC)	(Kim and Dempsey 2010)
SW	IRA958	batch	NR	NR	NR	NR	NR	≈73 (DOC)	(Gan et al. 2013)
SW	IRA958	column	NR	11.8	23	2	104	23 (DOC) 47 (UV) 32 (SUVA)	This experiment
Filtered SW	TAN-1	batch	5.65	4 (TOC)	19.5	NR	NR	70 (TOC)	(Brezinskiet al. 2019)
	TAN-1	batch		7.4-27.5	NR	NR	NR	25 (DOC) 29 (UV) 6 (SUVA)	(Graf et al. 2014)
SW	TAN-1	batch	7.8	5.6	NR	NR	27.2	57.8 (DOC) 23.2 (SO4) 8.8 (NO3)	(Liu 2017)
SW	TAN-1	column	nr	11.8	23	2	104	34 (DOC) 65 (UV) 47 (SUVA)	This experiment
SWs	A860	batch	6.9-7.6	1.3-5.8	NR	NR	NR	37-63 (DOC)	(Dixit 2017)
Filtered SW	A860	batch	5.65	4 (TOC)	19.5	NR	NR	68 (TOC)	(Brezinskiet al. 2019)
SW	A860 FL		NR	NR	NR	NR	NR	60 (UV254)	(Cornelissen et al. 2009)
SW	PPA860S	column	7.7-8	3.1	NR	3.23	30	71 (TOC) 68 (DOC) 93 (SO4)	(Verliefde et al. 2009)
SW	PPA860S	column	8.1	2.4	2.4	1.0	25.2	73 (TOC), 74 (DOC) 83 (UVA), 37 (SUVA) 100 (HS), 90 (BB), 57-60 (biopolymer, LMWN and LMWA), 99 (SO4)	(Sadmani 2014)
Reconstituted aquatic waters	PPA860	batch	NR	8.7 (TOC)	NR	NR	NR	35-40 (biopolymers), 40–67 (HS), 20-33 (BB), 12-13 (neutrals), 33–41 (DOC) 38–56 (UV)	(Bazri et al. 2016)
SW	PPA860S	column	NR	11.8	23	2	104	37 (DOC) 68 (UV) 50 (SUVA)	This experiment

DOC: dissolved organic carbon; NR: not reported; SW: surface waters; TOC: total organic carbon.

While most of the studies had DOC removal rates in the 40-70% range with their experimental batch or column test conditions (Table V-8), in some cases, lower removal rates due to competition with sulfates and nitrates were observed. Dixit (2017) compared DOC removal for Purolite A860 at different initial sulfate concentrations, and observed nearly 50% reduction in DOC removal when increasing the sulfate level from 0 mg L^{-1} to the highest concentration of 100 mg L^{-1}, which is comparable to the concentration in the raw water for De Blankaart in this study. Sadmani (2014) found PPA860S to efficiently remove most major NOM fractions, including biopolymers, while in this study BB were poorly removed and the biopolymer concentration increased.

3.3.2. The influence of resin parameters

In the literature, generally none of these 4 resins have been investigated together for the removal of NOM by IEX (Table V-8) with the exception of two studies, including the one by Bolto et al. (2002), in which the removal of NOM isolates by 19 types of strong resins and 5 low base resins was compared based on UV254 absorber removal. Nevertheless, comparison studies on other commercial anionic exchange resins have shown that NOM removal efficiencies vary considerably depending on the water type, anion exchange resin, and experimental conditions such as the contact time (Croué et al. 1999; Humbert et al. 2005; Tan et al. 2005). The non-removable NOM fraction – which probably correlates with the amount of uncharged species in the NOM (Bolto et al. 2002) – may vary between < 10% and 40% (Fettig 1999). However, according to our results UV254 overestimates the NOM removal compared to the organic carbon measurement.

3.3.2.1. Gel vs. macroporous structure

Bolto et al. (2002) observed that macroporous resins should remove more NOM than gel resins, through a better diffusion of organic anions. Tan et al. (2005) found the opposite due to a higher swelling capacity of the gel-type resins in water. In our case the structure does seem to have some influence on the results, as IRA410 (gel) had a slightly better DOC removal rate compared to the other 3 macroporous resins. Also the gel structure may explain why despite similar HS removal rates, IRA410 and PPA860S did not target the same MW in that fraction (see 3.2.2), with the macroporous resin (PPA860S) able to target higher MW HS.

3.3.2.2. Water content and functional groups

The best-performing IEX resin IRA410 (in terms of NOM removal) consisted of a styrene resin with the lowest water content and highest exchanging capacity.

According to Humbert et al. (2005), resins with a styrene structure display a greater affinity for aromatic compounds than resins with an acrylic structure. Cornelissen et al. (2008) on the other hand found no difference in NOM or NOM fraction removal for styrene or acrylic resins, while Boyer and Singer (2005) found polyacrylic resins to be more efficient at NOM removal than the more hydrophobic polystyrene resins which are thus less open in water. This leads to lower water content, so it is difficult to dissociate from the water content parameter when comparing two resins with different structures and different water content.

However previous studies by Bolto et al. (2002) and Cornelissen et al. (2008) have observed better sorption of UV-absorbing organics for high resin water content and low resin capacity. Indeed, a resin with high moisture has less dry matter, therefore less active groups and less capacity, but on the other hand, such a resin may provide a more open structure for large ions into its structure (Cornelissen et al. 2008).

In the only comparison between IRA410 and IRA958 resins, Bolto et al. (2004) found equal or better reduction of UV254 for IRA958 compared to IRA410. Hence the poor performance of IRA958 cannot be explained by its water content, especially compared to the resin PPA860S which has the same water content.

When the ionic strength of the water is high, contraction of the NOM will reduce size exclusion of DOC by polystyrene resins (Croué et al. 1999). This may help in explaining the better removal rate for styrene resin IRA410 compared to acrylic resin IRA958.

To summarize, the small number of resins tested and the large number of resin characteristics which may influence their performance render it difficult to explain the NOM removal results in our study, especially the poor performance of IRA958 compared to the very similar resin PPA860S.

4. Reduction of the DBP formation potential

The 4 resins remove different amounts of overall DOC but also different amounts of NOM fraction. DBP formation potential tests were performed to evaluate the impact of NOM removal by each resin on the formation of disinfection by-products.

4.1. THM_{7d}, THM_{24h} and HAA_{24h} speciation

Two different trihalomethane formation potentials have been performed (THM_{7d} and THM_{24h}). The 7-day test is the classical procedure to account for the total formation of DBPs, however as shown in chapter IV, the high chlorine dose of this test is not suitable for the assessment of iodinated DBPs. Then, the THM_{7d} has only been performed on the raw and IEX treated waters without increasing halide concentrations. A lower dose (residual of free chlorine of 0.3 $mgCl_2\ L^{-1}$) and contact time (24 h) have been used to assess the evolution of THM10 and HAAs after IEX treatment. The following figure V-4 shows the DBP formation potential decrease through the treatment of water with the four IEX resins.

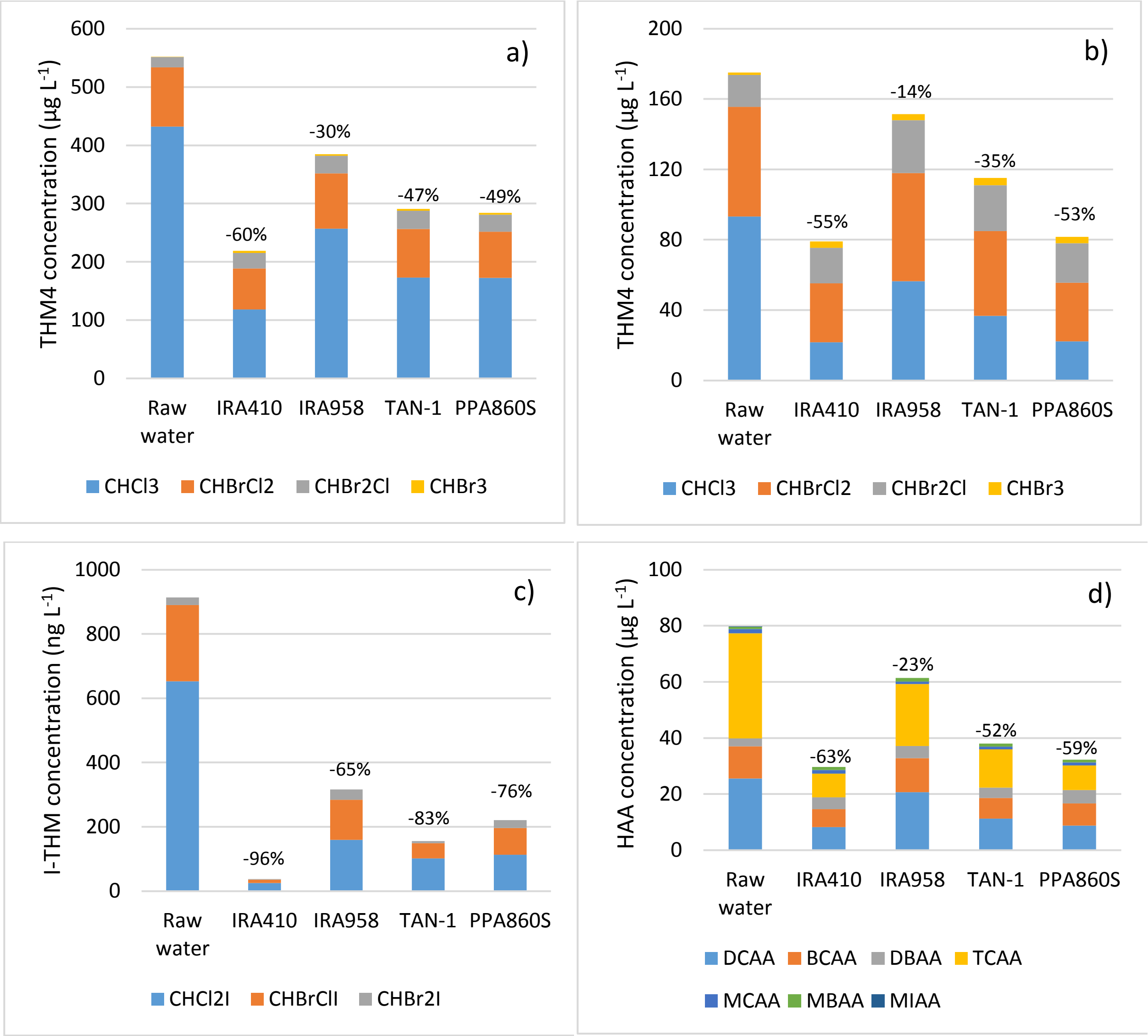

Figure V-4: Evolution of the DBPFPs for the different samples (raw and treated by the 4 different resins) a) THM_{7d}, b) THM_{24h}, c) $I\text{-}THM_{24h}$, d) HAA_{24h}.

4.1.1. THM_{7d}

The formation potential of the THM4 after 7 days (THM_{7d}) of the raw water is around 550 µg L^{-1} (Figure V-4a)), which is in the same range as noticed for previous experiments (Chapter IV). $CHCl_3$ is the main THM formed in all waters after the 7-day formation potential test. The reduction in THM_{7d} compared to the raw water is in the range 30-60%.

The order of THMFP reduction efficiency for the resins was: IRA958 < TAN-1 ≈ PPA860S < IRA410.

4.1.2. THM_{24h} and HAA_{24h}

4.1.2.1. THM4

Considering the formation of THM4 after 24h (Figure V-4b)), $CHBrCl_2$ is the major compound, due to the lower chlorine dose used leading to a higher bromide to chlorine ratio. Compared to the 7-day test, not only are the formation potentials 2.5-3.5 times lower after the 24-h test (Table V-9), but the reduction in THMFP is also lower, in the range 14-55%, with the exception of the PPA860S waters. This may be due to some fractions that have not reacted yet after 24 h or that have formed other unstable DBPs which can end up as THMs after a longer period of chlorination.

Table V-9: Comparison of the THM formation after 24 hours and after 7 days

	IRA410	IRA958	TAN-1	PPA860S	Raw water
THM_{7d} (µg L^{-1})	219	385	291	284	552
THM_{24h} (µg L^{-1})	79	152	115	82	176
THM_{7d}/THM_{24h}	2.8	2.5	2.5	3.5	3.1

The order of removal efficiency for the resins was: IRA958 < TAN-1 < PPA860S ≈ IRA410.

4.1.2.2. I-THMs

The major I-THMs formed are $CHCl_2I$ followed by CHBrClI and $CHBr_2I$, while no iodoform has been observed. These compounds correspond to the mono-iodinated compounds which are predominant for low iodide levels (Zhang et al. 2015). Compared to the THM4 the reduction in I-THM is 2-10 times higher (Table V-10), in the range 76-96% (Figure V-4c)).

While all resins reduced the I-THM FP more than the THM4 FP, IRA410 resin, in particular is much more efficient at limiting I-THM formation (Table V-10).

Table V-10: Comparison of the reduction in THM4 and I-THMs by the different IEX (compared to the raw water)

Reduction factor	IRA410	IRA958	TAN-1	PPA860S
THM4	2.21	1.16	1.52	2.14
ITHMs	21.47	2.9	6.93	4.16
ITHMs/THM4	9.69	2.51	4.55	1.94

The order of removal efficiency for the resins was: IRA958 < PPA860S < TAN-1 << IRA410.

4.1.2.3. HAAs

Compared to the THMs the reduction in HAAFP is also higher, in the range 23-63% based on the 24-hour HAA formation potential tests. DCAA, TCAA and BCAA are the main HAAs formed, representing 78 to 93% of the measured HAAs. The mono-HAAs are at much lower levels, as is usually the case in the literature (Villanueva et al. 2003; Ye et al. 2009; Wei et al. 2010; Bougeard et al. 2010; Zhang et al. 2010; Villanueva et al. 2012; Ghoochani et al. 2013; Bond et al. 2014; Hong et al. 2017; Zhou et al. 2019).

The order of removal efficiency for the resins was: IRA958 < TAN-1 < PPA860S ≈ IRA410.

4.2. Correlation between THMFP and DOC, UV and SUVA

The decrease in THM4 and HAA FPs followed the trends of DOC and SUVA reductions in the samples, with the highest removal for IRA410 and the least for IRA958.

4.2.1. THM$_{7d}$

A strong correlation ($r^2 > 0.99$) was observed between THM$_{7d}$ of the 5 waters and DOC as well as UV254 and SUVA (Figure V-5). The Figure V-5a) shows higher reduction of the THMFP than of the DOC, thus the different IEX resins are more efficient at removing THM precursors than overall DOC, with a 10-33% decrease of the THMFP/DOC ratio. Figure V-5b) shows higher reduction of UV254 than reduction in THMFP, demonstrating that a fraction of the precursors do not absorb UV, and come from more hydrophilic fractions. According to the curve, if the UV absorbance was equal to zero, approximately 140 µg L^{-1} of THMs would still theoretically be formed. Figure V-5c) shows almost a direct proportionality between THMFP and SUVA parameters which implies that the decrease in THMFP after treatment is mainly due to the removal of aromatic high molecular weight compounds. Non-HS and low UV absorbing organic matter which do contribute substantially to the formation of THMs are not removed as efficiently by the treatment.

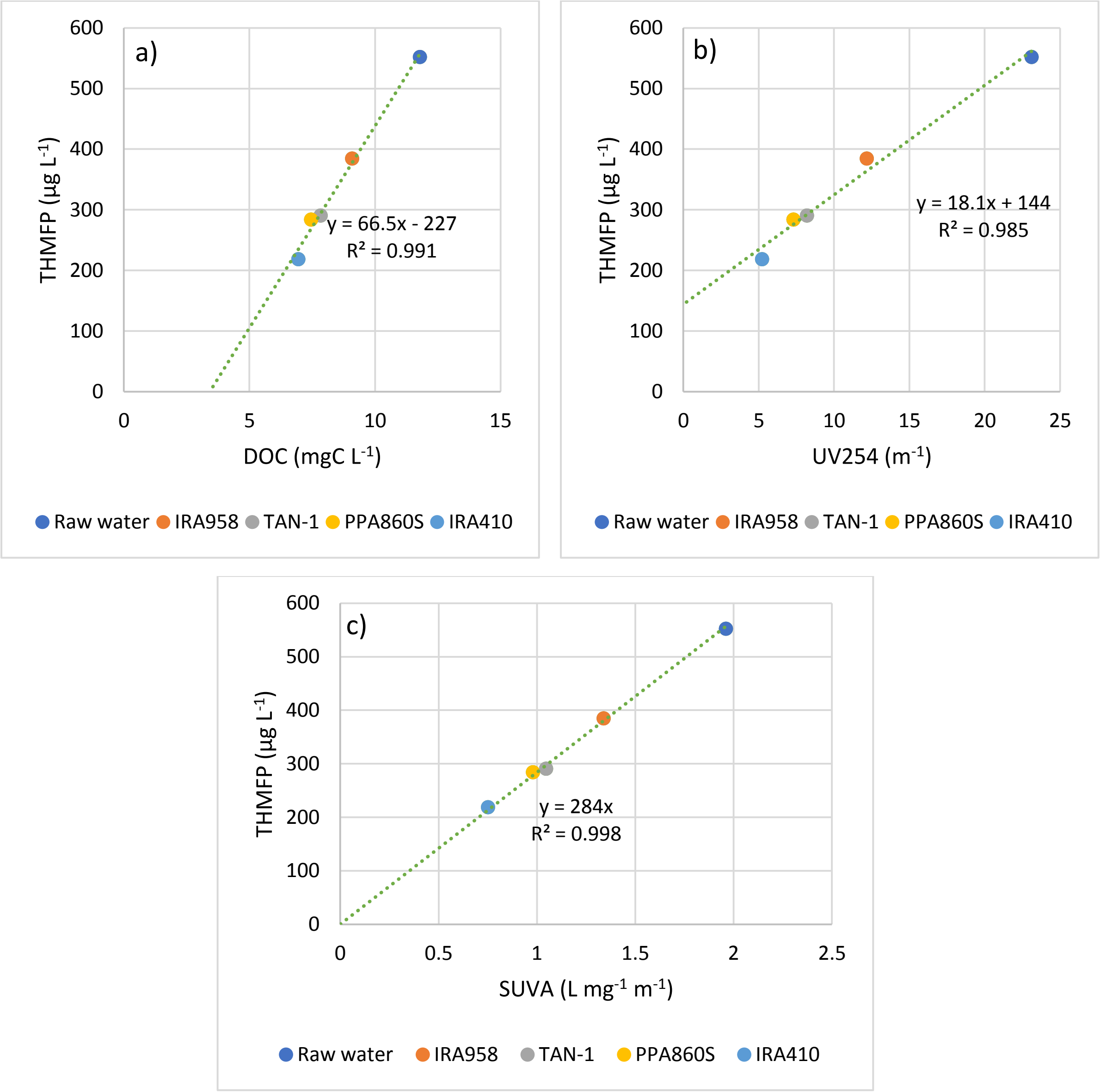

Figure V-5: Correlations between the THM$_{7d}$ for the different samples (raw and treated by the 4 different resins) and a) the DOC, b) the UV254 and c) the SUVA.

While earlier studies also observed strong correlations between DBP formations and SUVA values (Kitis et al. 2001; Kitis et al. 2002; Kitis et al. 2004), most were for high-SUVA NOM fractions obtained by various physicochemical separations. Interestingly, in another study by Ates et al. (2007), no strong correlations between SUVA and THM formations and SUVA and HAA formations were found in low-SUVA waters (treated by IEX or GAC) and containing lower NOM fractions. Zhai et al. (2017) also found poorer correlations for SUVA between 0.5 and 2 than above 2 for HAAs and THMs. The difference may be due to the poor removal of the hydrophilic fractions for De Blankaart waters. Indeed while some non-UV absorbing compounds also contribute as precursors to THM formation, mainly the UV-absorbing precursors are removed, leading to the good correlation between SUVA reduction and THMFP reduction.

Overall, these results show that the reduction in THMFP by IEX treatment can be estimated by simple UVA absorbance measurements for the De Blankaart waters, but some more measurements may be required to make sure this correlation remains true with changing raw water characteristics.

4.2.2. THM_{24h} and HAA_{24h}

The formation potential of THMs and HAAs after the 24-h test also correlated with the DOC, UV and SUVA (R^2 from 0.8 to 0.9 Figure V-S1), but the correlations were not as strong as for THM_{7d}, possibly due to:

- The preferential removal of more or less reactive precursors by the different resins;
- The higher importance of HOBr kinetics with precursors at higher bromide/chlorine ratios compared to the 7-day test (see 5.1.2. for a more complete explanation), such that the small differences in initial bromide concentrations in these non-spiked samples leads to different amounts of DBPs formed. The correlation between SUVA and THMs was due to the correlation between $CHCl_3$ (the dominant THM) and SUVA (Figure V-S5), so an increase in brominated species changes the overall correlation of THM to SUVA;
- Or due to the formation of other non-measured DBPs.

4.3. THM and HAA (24-h test) evolution

In order to identify the capacity of the resins to remove specifically the DBP precursors, the NOM fractions have been considered. The THMs and in a lower extend the HAAs are not directly linked to the humic acid fraction removal (Figure V-S2). The IRA958 resin shows a particularly low THM precursor removal, as the THM formation potential after 24h is only decreased by around 14% compared to 23% removal of DOC and 41.3 % of the humic acid fraction. On the other hand, the THMFP decrease for the resins IRA410 and PPA860S are higher (around 55%) than the removal of DOC (around 40%). This high degree of THMFP reduction compared to the total DOC removal is certainly due to the high removal of the humic acid fraction for these two resins (around 70%) (Figure V-6).

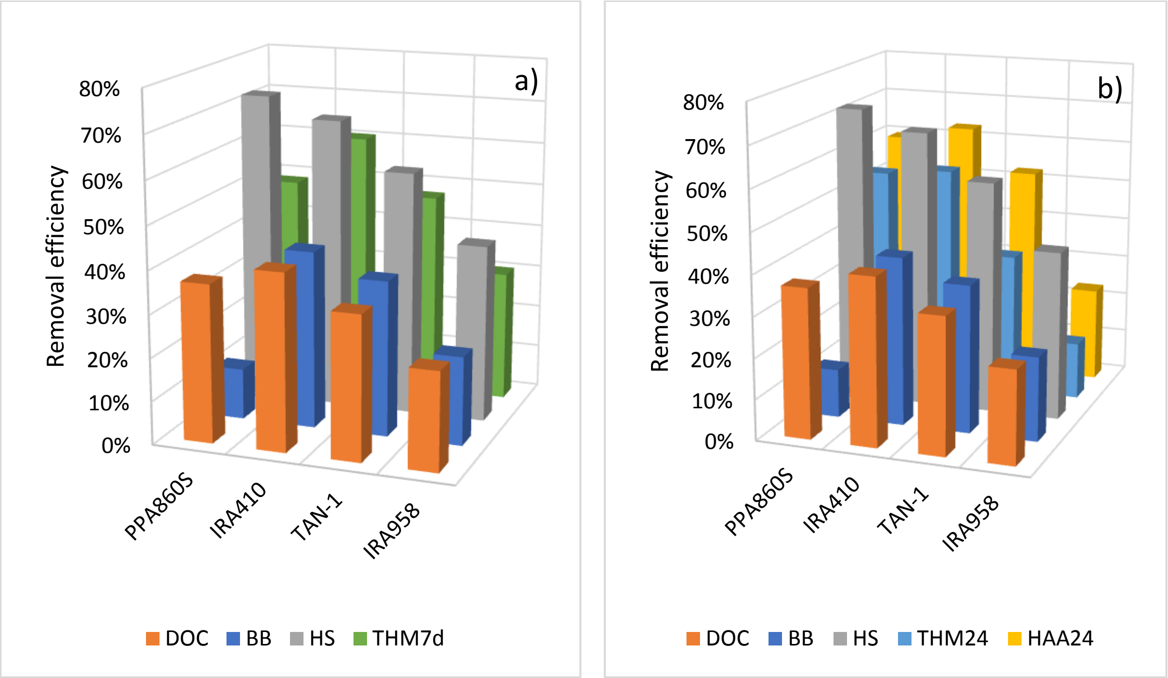

Figure V-6: Removal efficiencies of DOC, BB, HS and a) THM$_{7d}$, b) THM$_{24h}$ and HAA$_{24h}$.

Considering HAAs, the HAA formation decrease is slightly higher compared to THMs for PPA860S (59%) and IRA410 (63%) resins (compared to around 55% for the THMs). This difference is much higher when considering TAN-1 (52% for HAAs and 34% for THMs) and IRA958 (23% for HAAs and 14% for THMs). The PPA860 resin has a particularly low building block removal efficiency (12%), which does not change substantially its capacity to decrease the DBP FP counterbalanced by the good efficiency for HS.

4.4. I-THM assessment with initial iodide concentrations

The trend in I-THM formation reduction is a bit different to the trends for THM4, HAAs or SUVA, with TAN-1 lowering more the DBP formation potential than PPA860S. That is due to the lower iodide concentration for water treated with TAN-1 resin compared to PPA860S. Indeed, the I-THM concentration is correlated either to the initial iodide or to the initial iodine concentration (Figure V-7) for all 4 treated waters.

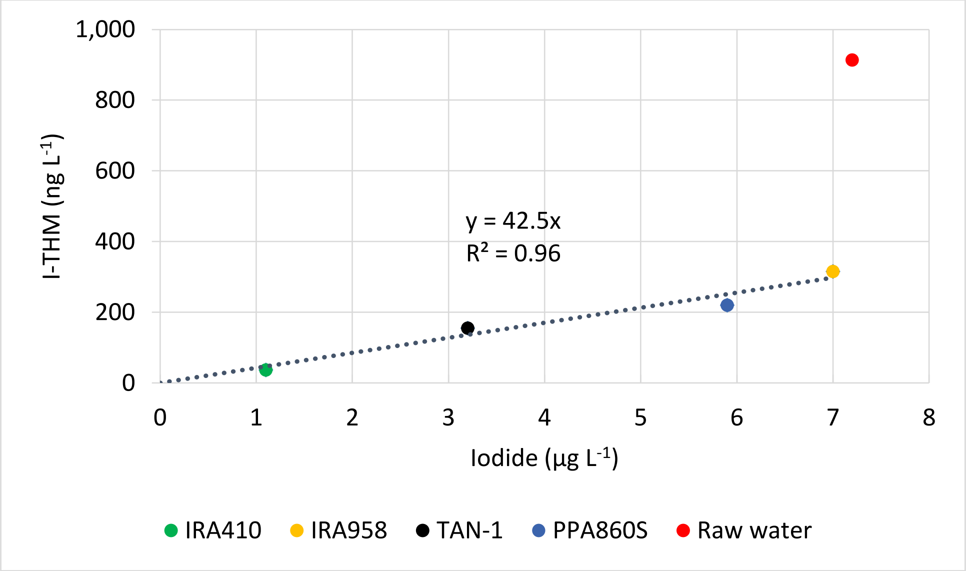

Figure V-7: Evolution of the I-THM concentration as a function of the initial iodide concentration.

However, the raw water sample is different and cannot be correlated to the initial iodide concentration, presumably due to different physico-chemical characteristics such as the total iodine concentration. The content of ammonia in the water does not explain neither this behaviour as the raw water shows the same concentration as the waters treated by the IRA410 and IRA958 resins (around 0.17 mg L^{-1}) and lower concentration than the water treated by PPA860S and TAN-1 resins (around 0.27 mg L^{-1}). Thus, all the resins by lowering iodide and total iodine concentrations or by removing specific precursors show a beneficial effect, lowering the concentration of I-DBPs.

5. Assessment of the IEX resins to reduce the formation of DBPs under different halide concentration conditions

5.1. Increasing bromide concentrations

The formation of bromine-containing DBPs is of particular interest as they are generally more toxic than chlorine-containing compounds (Plewa et al. 2004). To better understand the influence of bromide, the concentration of bromide was varied from 2.5 to 27.5 µM (228 to 2200 µg L^{-1}) while the iodide concentration was kept constant (0.5 µM *i.e.* 63 µg L^{-1}). This led to molar bromide/iodide ratios ranging from 5.7 to 50, comparable to a recent study by Allard et al. (2015). In order to compare the efficiency of the different resins in the same halide concentration conditions, the concentrations of bromide and iodide in the raw water and the different IEX treated waters have been equalized.

5.1.1. THM4 formation and speciation

The sum of THM4 increases with the bromide concentration (Figure V-8) for the raw (x 1.7) and IEX treated waters (x 1.9-2.3).

However, when considering the molar concentrations, the amount of THMs is also constantly increased (x 1.4) for the 4 IEX treated waters with the increasing bromide concentration but not in raw water (Figure V-S3a)). In fact, the amount of THM4 is constant, around 1.2 µM for the raw water up to 800 µg L^{-1} of bromide and only slightly increases to 1.3 µM for the highest tested concentration (2200 µg L^{-1}). The THM4 speciation is however greatly modified towards the brominated compounds.

Figure V-8: THM4 evolution as a function of initial bromide concentration for the 5 samples (iodide = 63 µg L^{-1}) a) Raw water, b) IRA410, c) IRA958, d) TAN-1, e) PPA860S. The same figure with molar concentration is presented in the supporting information (Figure V-S3).

At a bromide to iodide molar ratio of 5.7 (I^- = 63 µg L^{-1}), with bromide concentrations already quite high (228 µg L^{-1}), the main THM4 were found to be a mix of $CHBrCl_2$, $CHCl_3$ and $CHBr_2Cl$ for the 4 resins, and $CHCl_3$ and $CHBrCl_2$ for the raw water. Some $CHBr_3$ was also formed, especially for IRA410 but was a minor species.

When the bromide concentration was increased to a bromide/iodide ratio of 10, the raw water contained mainly a mix of $CHBrCl_2$, $CHCl_3$ and $CHBr_2Cl$. The 4 resin samples contained mainly $CHBrCl_2$ and $CHBrCl_2$.

When the bromide/iodide ratio was increased to 20, TAN-1 and IRA958 samples contained a mix of $CHBrCl_2$, $CHBr_2Cl$ and $CHBr_3$, with $CHBr_2Cl$ slightly predominant, while PPA860 and IRA410 contained mainly $CHBr_2Cl$ and $CHBr_3$. The raw water sample contained mainly $CHBrCl_2$ and $CHBr_2Cl$.

When the bromide/iodide ratio was increased to 50, $CHBr_3$ was predominant in all 5 samples, followed by $CHBr_2Cl$, especially in the raw water.

Bromine reacts more with hydrophilic compounds than with hydrophobic compounds (Awad et al. 2016), which may be the reason for an overall increase in THM formation from the treated waters which contain less hydrophobic compounds than the raw water. For bromide levels above 800 µg L^{-1}, the water treated with IRA958 produces even more THMs than the raw water despite a substantial decrease of the DOC (23%).

5.1.1.1. Bromine incorporation factor

The bromine incorporation factor (BIF), first developed by Gould et al. (1983), describes the molar contribution of all brominated species (Eq.1, with THM4 in molar concentrations), and can range from 0 to 3, with values closer to 3 representing a more brominated THM sample.

$$BIF\ (THM4) = \frac{[CHBrCl_2]+2\,[CHBr_2Cl]+3[CHBr_3]}{[CHCl3]+\,[CHBrCl_2]+[CHBr_2Cl]+[CHBr_3]} \qquad \text{Eq. 1}$$

The calculated BIF values were plotted against the bromine concentration (Figure V-9a)).

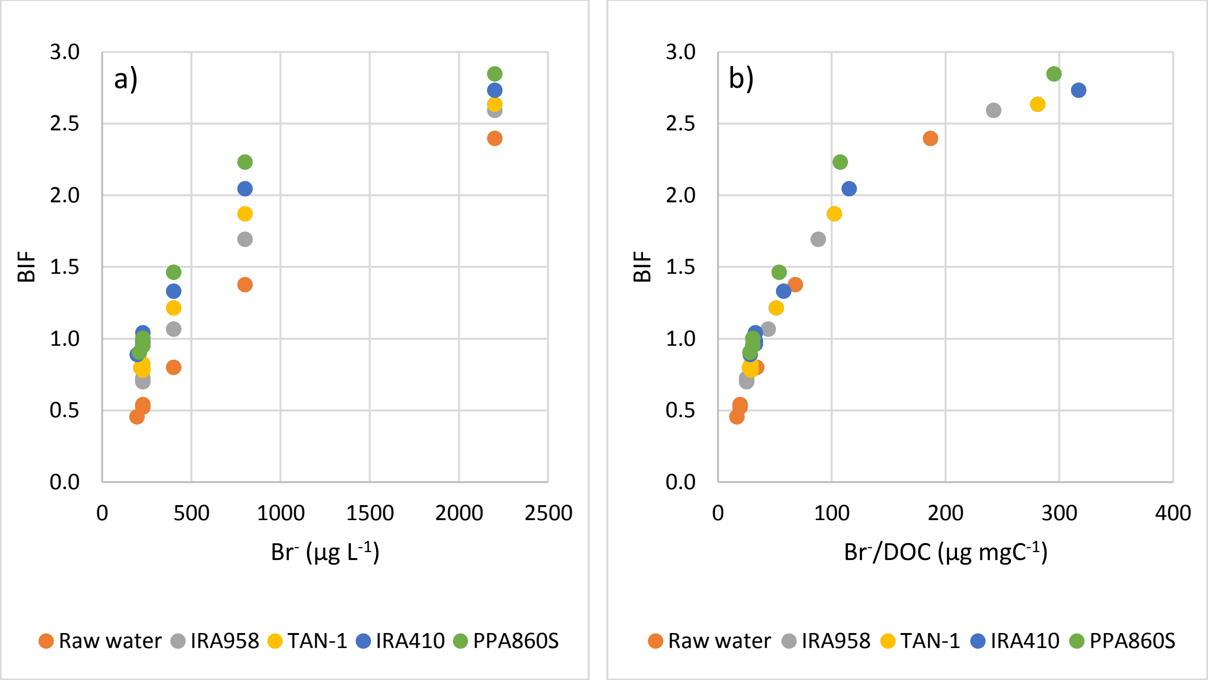

Figure V-9: BIF evolution for all the tested samples as a function of a) bromide, b) bromide to DOC ratio.

In general, higher levels of bromide resulted in higher BIF values. More specifically, for each treated water, BIF and Br^- concentrations were more or less linearly correlated at first before converging towards the maximum of 3. However, the BIF was sample-specific for the 5 different waters, at each bromide level, the BIF followed the order: Raw water < IRA958 < TAN-1 < IRA410 < PPA860S (Figure V-9a).

This indicated that besides the bromide concentration, the NOM properties were also important for bromine incorporation into DBP precursors.

Indeed, the BIFs – regardless of the water samples – were found to increase in function of the Br^-/DOC (Figures V-9b)) or Br^-/SUVA. Effectively, a higher Br^-/DOC ratio (usually the result of DOC removal during treatment) favors the reaction of chlorine with Br^- instead of with DOC, resulting in the formation of reactive bromine species. This explains why the maximum BIF values at each bromide concentration are reached for IRA410 and Purolite PPA860S, as they were the samples with the lowest DOCs (Table V-4). Above a Br^-/DOC ratio of approximately 60 µg mgC^{-1}, the curve is no longer linear, due to progressive saturation of the THM precursors with bromide ions. This is also seen in the case of $CHBr_3$ which is correlated with the Br^-/DOC ratio for Br^- up to 800 µg L^{-1} (Figure V-10) with, for each bromide level, the $CHBr_3$ concentrations following the same trend Raw water < IRA958 < TAN-1 < IRA410 < PPA860S, but which is not correlated anymore at the highest bromide concentration for the resins (not enough NOM precursors available). This explains why at the highest bromide level, the higher $CHBr_3$ concentrations in the samples followed a different order: IRA958 < Raw water < TAN-1 < PPA860S < IRA410.

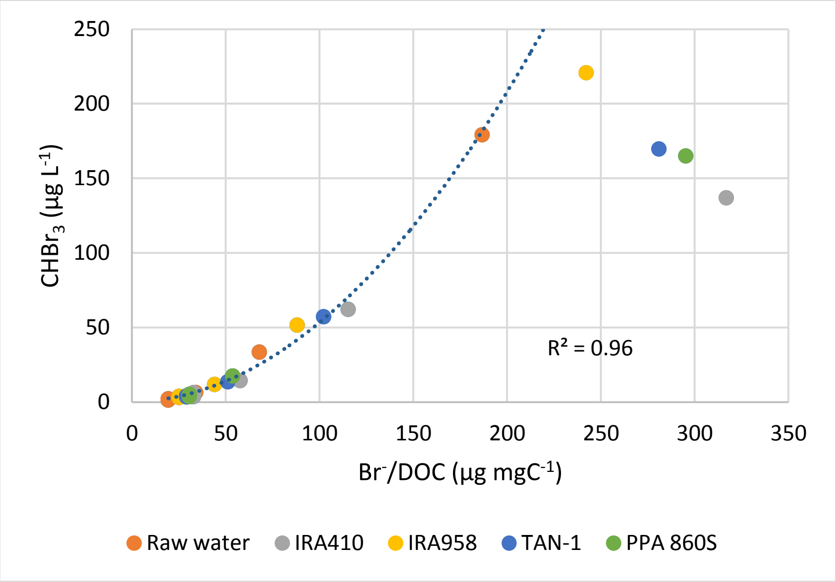

Figure V-10: Bromoform evolution for all the tested samples as a function of the bromide to DOC ratio.

5.1.2. I-THM formation and speciation

In this section, the concentration of iodide has been kept constant (63 µg L^{-1}). 87-100% of the total I-THMs measured consisted of mono-I-THMs (CH_2ClI, CHBrClI and $CHBr_2I$) at all Br^- concentrations (Figure V-11). Very low $CHClI_2$ levels were measured at the lowest Br^- concentration (Br^-/I^- molar ratio of 5.7; Br^- = 228 µg L^{-1}), this compounds is not detectable anymore when increasing the bromide concentration. Very low $CHBrI_2$ levels were detected at the highest Br^-/I^- ratio for some resins.

At a bromide to iodide molar ratio of 5.7, with bromide concentrations already quite high (228 µg L^{-1}), the main I-THM was found to be $CHCl_2I$ for all samples, however closely followed by CHBrClI (Figure V-11). Some $CHBr_2I$ was also formed, especially for PPA860S but was a more minor species, especially for raw water. When the bromide concentration was increased to a bromide/iodide ratio of 10, more CHBrClI and less $CHCl_2I$ were formed, both dominating the speciation (Figure V-11). $CHBr_2I$ concentrations increased but the species remained minor, except for PPA860S where all three mono-I-THMs were formed in the same range.

When the bromide/iodide ratio was increased to 20, $CHCl_2I$ became minor for raw water and PPA860S where CHBrClI and $CHBr_2I$ became the major species (Figure V-11). For the other waters CHBrClI dominated while $CHCl_2I$ and $CHBr_2I$ accounted for similar fractions.

When the bromide/iodide ratio was increased to 50, Br^-/I-THMs clearly became the major species with highest concentrations of $CHBr_2I$ (Figure V-11). This is consistent with a previous study by Allard et al. (2015).

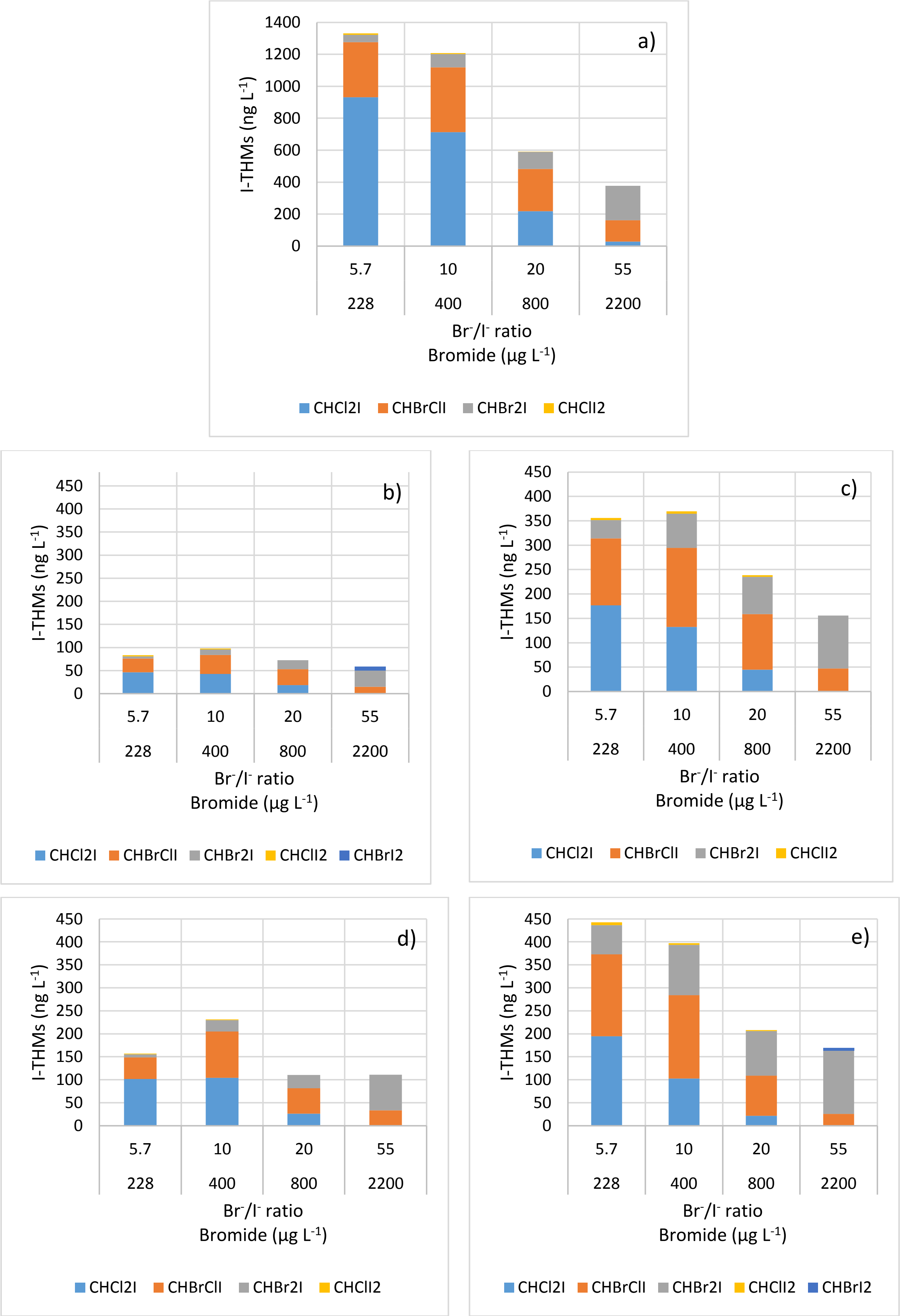

Figure V-11: I-THM evolution as a function of initial bromide concentration for the 5 samples (iodide = 63 µg L^{-1}) a) Raw water, b) IRA410, c) IRA958, d) TAN-1, e) PPA860S.

Overall when increasing the Br^- to I^- ratio from 5.7 to 10, an increase in I-THMs was observed for 2 resins TAN-1 (+32%) and IRA410 (+13%), due to more increase in brominated I-THMs than decrease in $CHCl_2I$ (especially in the case of TAN-1 where $CHCl_2I$ formation remained the same). However, the decrease in $CHCl_2I$ led to an overall I-THM decrease for raw and PAA860S-treated waters by approximately 10%. At higher Br^-/I^- ratios, I-THMs levels decreased for all samples.

The reduction efficiency in I-THMs by IEX (the ratio between the I-THMs formed after IEX and the I-THM formed in raw water) dropped when increasing the Br^-/I^- ratio (Figure V-12). This is due to a higher reduction in I-THM formation in the raw than in the treated waters, at increasing bromide concentrations. Indeed, the higher competition of DOC for reactive bromine in raw water than in the treated waters at the lower bromide concentrations is less apparent at higher bromide concentrations.

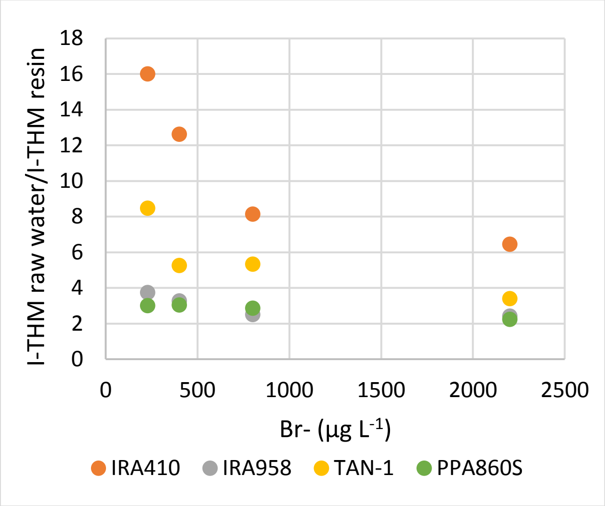

Figure V-12: Evolution of the 'I-THM in raw water / I-THM in treated water' ratio as function of the bromide concentration.

5.1.3. HAA formation and speciation

The sum of the 7 HAAs remains more or less stable with increasing bromide concentrations, but the molar concentrations decrease (Figure V-S4). This is most certainly due to the trihaloacetic acids (tri-HAAs) speciating towards more brominated tri-HAAs which could not be measured in this study. Indeed, TCAA decreased with increasing bromide concentrations, from being a major HAA species (24-45% of the mass of the measured HAAs at Br^-=228 µg L^{-1}, to ≤2 % at the highest Br^- concentration. IRA958 and raw water samples contained the highest amounts and weight percentages of TCAA.

When looking only at di-HAAs (DCAA, BCAA and TCAA), there appears to be a slight increase in the sum of HAAs (Figure V-13) with increasing bromide concentrations.

But this seems to be due to the increase in MW of the species, because the same graphs but in molar concentration show no increase (Figure V-S4), due to more substitution than new species formation amongst di-HAAs (Sawade et al. 2016). At a bromide to iodide molar ratio of 5.7, DCAA was the main di-HAA, closely followed by, BCAA and DBAA. DCAA decreased rapidly with increasing bromide concentrations, while BCAA initially increased then decreased. DBAA increased to become the major species. At Br$^-$ = 2200 µg L^{-1}, DBAA represents 69 to 83% of the total HAAs (75-92% of the di-HAAs), with the highest proportion for PPA860S.

No MIAA was detected except in raw water (several dozens of ng L^{-1}) where the concentration rapidly decreased with increasing Br$^-$ concentrations. Low levels of MCAA decreased and low levels of MBAA increased with the bromide concentration.

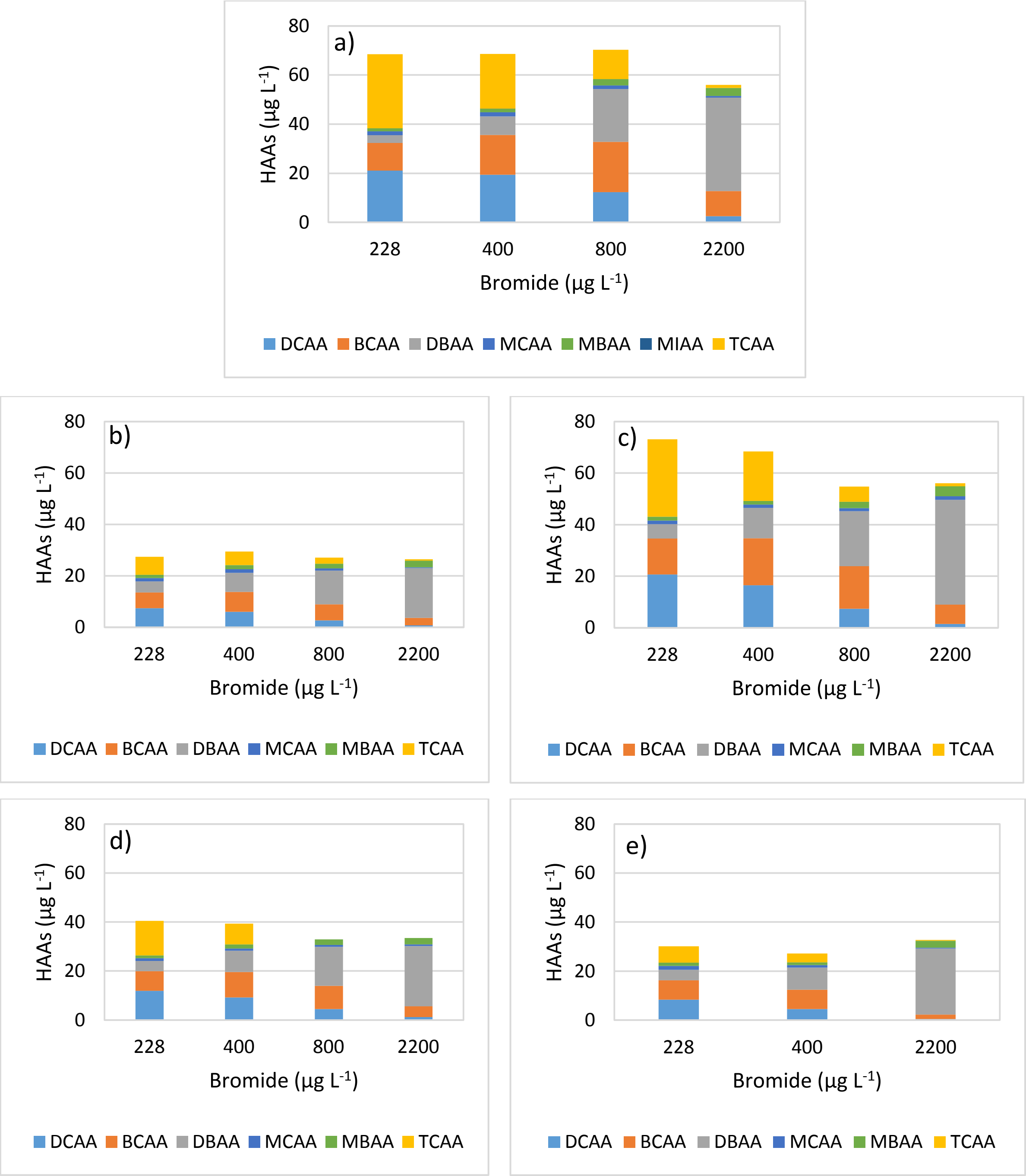

Figure V-13: HAA evolution as a function of initial bromide concentration for the 5 samples (iodide = 63 µg L^{-1}) a) Raw water, b) IRA410, c) IRA958, d) TAN-1, e) PPA860S.

Bromine incorporation factor

As was the case for the THMs (Figure V-9), the BIF was found to initially increase linearly in function of the Br^-, the Br^-/DOC ratio (Figure V-14).

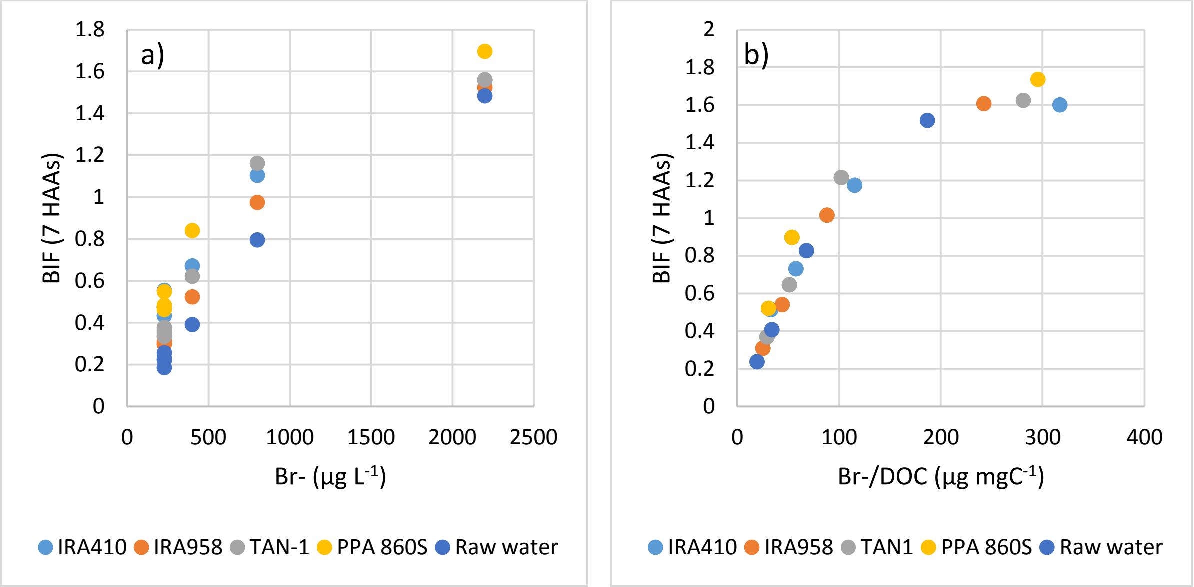

Figure V-14: HAA BIF evolution for all the samples (with iodide = 63 µg L^{-1}) as function of a) bromide, b) bromide to DOC ratio.

The BIF converges towards a maximum of approximately 1.6-1.7, but the BIF is skewered, because the 3 brominated tri-HAAs – which would increased the BIF – could not be measured in this study.

The BIF was sample-specific for the 5 different waters, at each bromide level, the BIF followed the order: Raw water < IRA958 < TAN-1 < IRA410 < PPA860S. Overall, the Br^-/DOC ratio seems a better indicator than the bromide concentration for the incorporation of bromine in HAAs.

Unlike for $CHBr_3$ (Figure V-10), while the most brominated compound measured here (DBAA) is correlated to the Br^-/DOC ratio (Figure V-15), it is sample specific, with IRA958 and raw water samples forming similar amounts of DBAA at all Br^- levels. TAN-1 and PPA860S also formed similar amounts off DBAA.

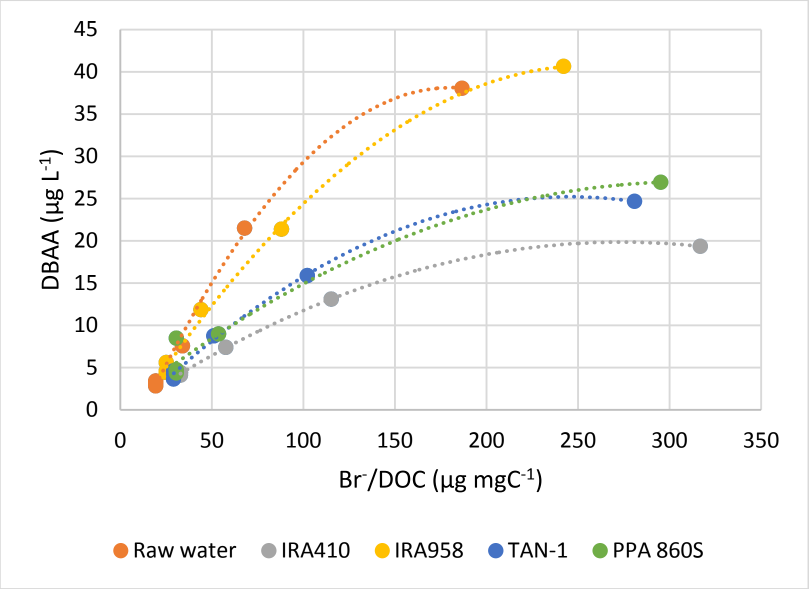

Figure V-15: Dibromoacetic acid evolution for all the tested samples as a function of bromide to DOC ratio.

While $CHBr_3$ levels were higher in the order: Raw water < IRA958 < TAN-1 < IRA410 < PPA860S, DBAA was formed higher in the order IRA410 < TAN-1 ≈ PPA860S < IRA958 ≈ Raw water, which is also different to the BIF order for di-HAAs: Raw water < IRA958 < TAN-1 < IRA410 < PPA860S.

5.2. Increasing iodide concentrations

The same experiment has been performed with increasing iodide concentrations (at 0.06, 0.5 and 2 µM, corresponding to 7, 63 and 254 µg L^{-1}). The concentration of bromide was kept constant to the minimum background level possible (228 µg L^{-1}) (Figure V-16).

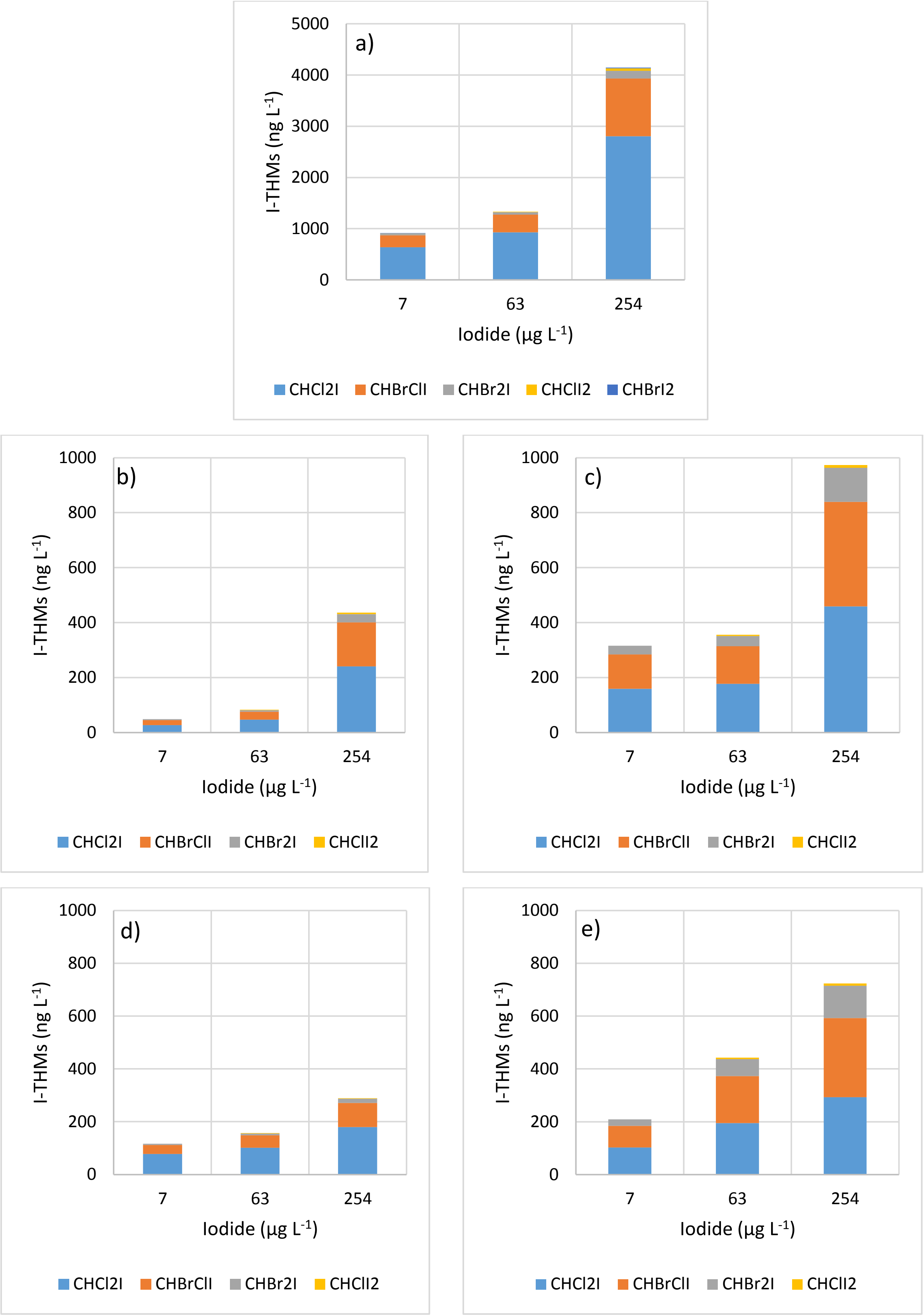

Figure V-16: I-THM evolution as a function of initial iodide concentration for the 5 samples (bromide = 228 µg L^{-1}) a) Raw water, b) IRA410, c) IRA958, d) TAN-1, e) PPA860S.

For all waters, the I-THM concentrations increase as the initial iodide concentration is increased. Interestingly the speciation amongst the monoiodinated THMs did not vary much with the iodide concentration for each sample, but for a slight increase in importance of the brominated species. At high iodide concentrations of 63 and 254 µg L^{-1}, in addition to the three monoiodinated THMs, some traces of $CHClI_2$ (mostly < 10 ng L^{-1}) were detected in all the samples. No CHI_3 was detected which is in agreement with a study by Hua et al. (2006) where much higher amounts of iodide were needed to form a substantial amount of CHI_3.

The rate of increase is not the same for all waters (Table V-11). Thus, the correlation which seemed to exist between the initial iodide concentration and the I-THM formation for the waters with iodide spiking (Figure V-7) does not exist here after spiking, with different iodide levels in each water despite identical initial iodide concentrations (7, 63 or 254 µg L^{-1}). In addition, the ratio between the formation of I-THMs in the raw water and each treated water varies in function of iodide concentrations. In fact, this ratio decreases greatly for IRA410 while it increases for TAN-1 and to a lower extent for IRA958 and PPA860S.

Table V-11: I-THM raw water/I-THM treated water.

Iodide (µg L^{-1})	IRA410	IRA958	TAN-1	PPA860S
7	19.0	2.89	7.85	4.31
63	16.0	3.74	8.47	3.01
254	9.49	4.26	14.3	5.72

Considering the THM4 for the raw water in this study, a slight decrease was observed with the increasing concentration of iodide (a decrease by 13% at 2 µM of iodide compared to 0.06 µM). However, the initial iodide concentration has no impact on the THM4 for all resin samples (Figure V-17). Hua et al. (2006) also found no impact of adding 2 µM of iodide to some raw waters on the THM4.

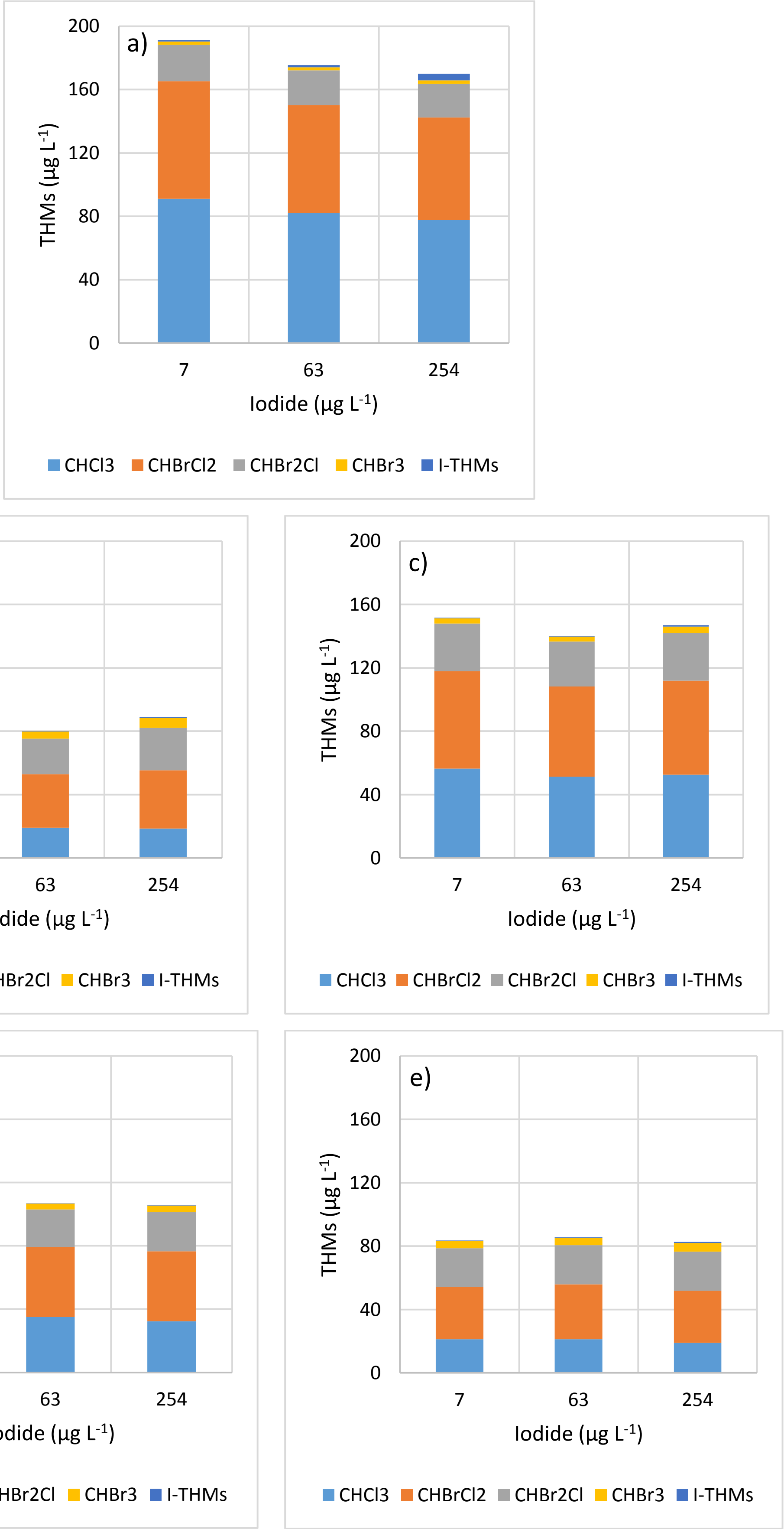

Figure V-17: THM evolution as a function of initial iodide concentration for the 5 samples (bromide = 228 µg L^{-1}) a) Raw water, b) IRA410, c) IRA958, d) TAN-1, e) PPA860S.

Another way to visualize the impact of iodide on I-THM formation is to use the iodine incorporation factor (Figure V-18).

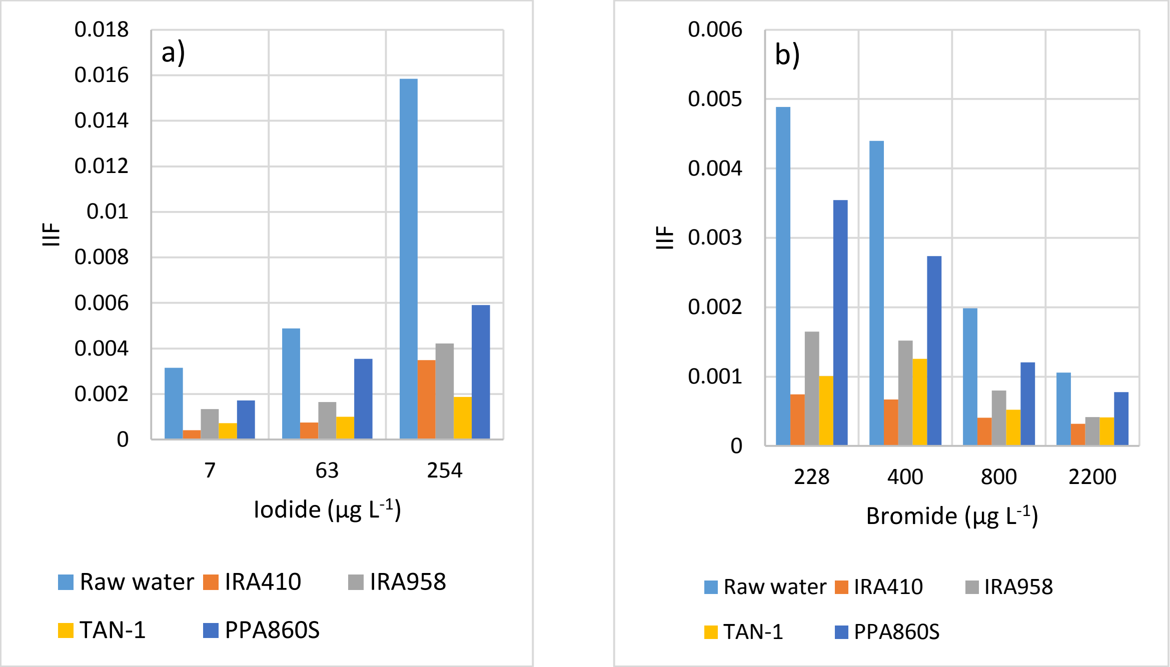

Figure V-18: Evolution of the IIF with a) the initial iodide concentration (for a constant initial bromide concentration of 228 µg L^{-1}) and b) the initial bromide concentration (for a constant initial iodide concentration of 63 µg L^{-1}).

The iodine incorporation factor increases with the initial iodide concentration and decreases with increasing initial bromide concentrations (Figure V-18). The different water samples do not follow this trend in the same way, with the water treated by TAN-1 having the highest IIF amongst treated waters at a low initial iodide concentration (7 µg L^{-1}) but the water treated by PPA860S giving the highest IIF for iodide concentrations equal or above 63 µg L^{-1}, regardless of the level of spiking with bromide.

6. Correlation between THMs and HAAs

6.1. The bromine substitution factor

To assess and compare the impact of bromide on halogen speciation of DBPs, the bromine substitution factor (BSF) (Hua, Reckhow, and Kim 2006) was calculated for each DBP class and subclass studied (THMs, di-HAAs, HAAs), according to equations (1), (2), and (3) respectively. This corresponds to the normalization of the BIF parameter calculated previously and is useful in order to compare different compounds.

$$BSF\ (THMs) = \frac{[CHBrCl_2]+2\,[CHBr_2Cl]+3[CHBr_3]}{3*([CHCl3]+[CHBrCl_2]+[CHBr_2Cl]+[CHBr_3])} \qquad \text{Eq. 1}$$

$$BSF\ (di-HAAs) = \frac{[BCAA]+2\,[DBAA]}{2*([DCAA]+[BCAA]+[DBAA])} \qquad \text{Eq. 2}$$

$$BSF\ (HAAs) = \frac{[BCAA]+[MBAA]+2\,[DBAA]}{[MCAA]+[MBAA]+2*([DCAA]+[BCAA]+[DBAA])+3*[TCAA]} \qquad \text{Eq. 3}$$

The molar concentrations were used and the BSF values can vary from 0 (fully chlorinated) to 1 (fully brominated) neglecting the iodinated compounds. The Figure V-19 presents the relationship between the incorporation of bromine into THMs and HAAs and di-HAAs from all the data acquired for the different bromide and iodide levels and for the different resins.

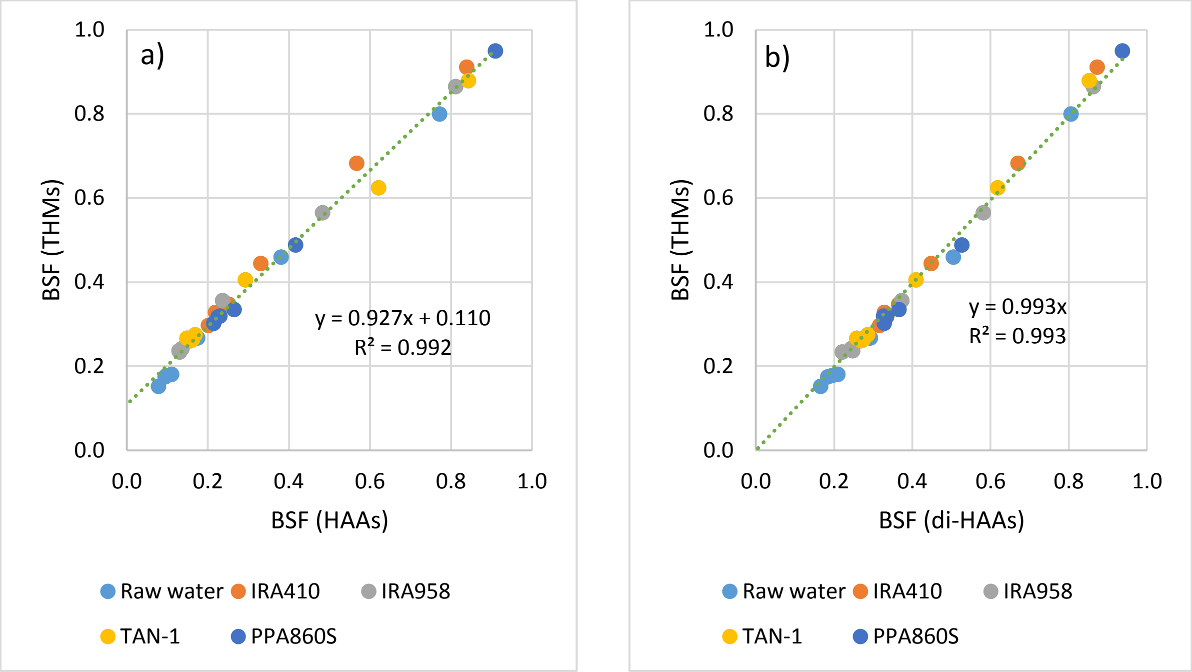

Figure V-19: Relationship between BSF for THMs and a) BSF for HAAs (7 HAAs), b) BSF for di-HAAs.

The similar BSFs for THM4 and di-HAAs (not presented) indicates that, for both DBP classes, their organic precursors have similar affinities for reaction with bromine or chlorine. The high linear correlation between the BSF for di-HAAs and the BSF for THMs (r^2 >0.99, Figure V-19) demonstrates that di-HAAs and THMs had the same bromine substitution, which agreed with previous studies (Hua et al. 2006; Hua and Reckhow 2012; Tan et al. 2016; Hong et al. 2017).

6.2. Statistical analysis

The relationships of the DBP classes and individual DBP species for all the samples (n=33) were examined by the Pearson correlation coefficient (r), and the statistical significance verified by the two-tailed paired-samples t-tests (Table V-12). The following correlations are considered statistically significant ($p < 0.05$).

Table V-12: Correlation between various THMs and HAAs.

	MCAA	MBAA	DCAA	BCAA	DBAA	TCAA	$CHCl_3$	$CHBrCl_2$	$CHBr_2Cl$	$CHBr_3$	di-HAAs	HAAs
MCAA												
MBAA	-0.22											
DCAA	0.55	-0.55										
BCAA	0.59	-0.04										
DBAA	-0.34	0.95	-0.59									
TCAA	0.44	-0.49	0.98									
$CHCl_3$	0.45	-0.56	0.96	0.46	-0.58	0.97						
$CHBrCl_2$	0.54	-0.50	0.87	0.81	-0.50	0.80	0.80					
$CHBr_2Cl$	-0.06	0.65	-0.34	0.43	0.71	-0.34						
$CHBr_3$	-0.49	0.89	-0.64	-0.35	0.93	-0.53						
di-HAAs	0.28	0.50	0.36	0.74	0.51	0.35		0.43	0.60			
HAAs	0.48	0.04	0.79	0.78	0.02	0.78	0.73					
THMs	-0.15	0.75	-0.07	0.27	0.81	0.05	-0.02	0.00	0.67	0.73	0.81	0.52

6.2.1. THM4 vs. di-HAAs

As the brominated tri-HAAs were not measured in this study, THMs and HAAs cannot be correlated (R^2 = 0.10, not presented) when spiking with bromide as the weight of the HAAs is underestimated, except perhaps for the lowest bromide concentrations (R^2 = 0.95, not presented). When comparing THMFP and di-HAAFP, a relatively strong correlation was found (Table V-12), but poorly linear (Figures V-20a) and b)), due to the increase in total THM concentration at higher initial bromide concentration, while the di-HAA concentration remains approximately the same, hence increasing the ratio of THMs to di-HAAs. Thus THMs are a poor surrogate for HAA formation in De Blankaart waters treated by different resins when increasing the initial bromide concentration.

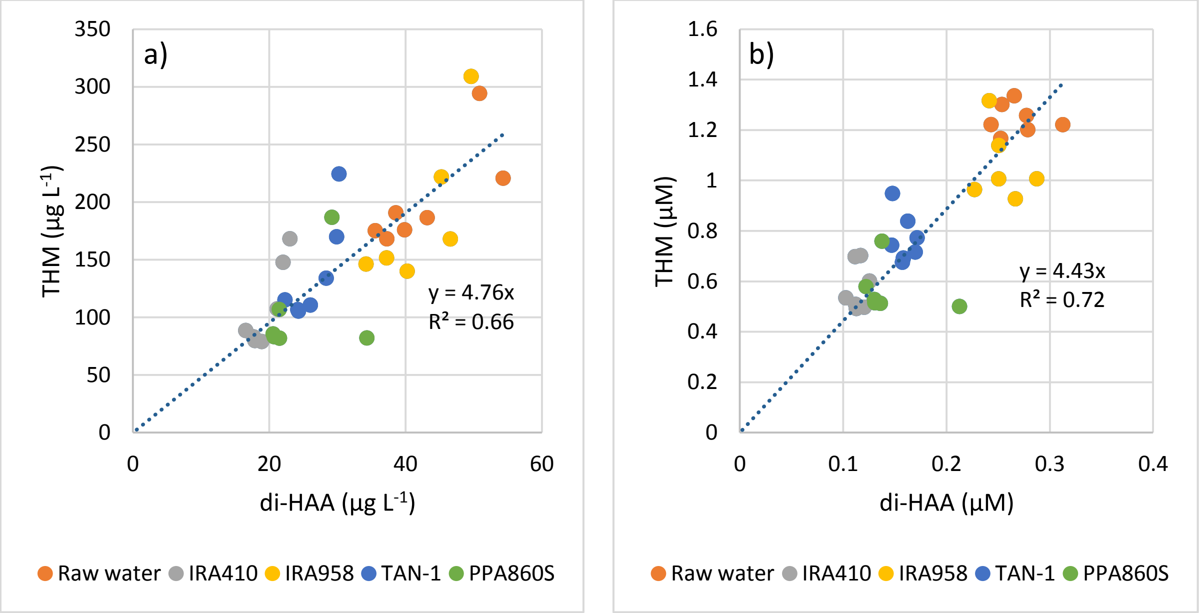

Figure V-20: Correlation between THMs and di-HAAs for all samples a) in mass concentrations, and b) in molar concentrations.

6.2.2. Other correlations

6.2.2.1. Between individual HAAs

In this study, TCAA and DCAA were produced at roughly the same concentrations (at a ratio of approximately 1, Figure V-21), which is slightly surprising since previous studies have suggested that DCAA and TCAA may have different precursors and reaction pathways (Hua and Reckhow 2007; Hong et al. 2009; Zeng and Arnold 2014; Hua et al. 2015). The good correlation between TCAA and DCAA in this study may be due to the use of the same raw water, with more or less equal removal of the different TCAA and DCAA precursors for each resin. Although, the repetition of experiments at different halide levels on the same 5 waters could artificially increase the correlation.

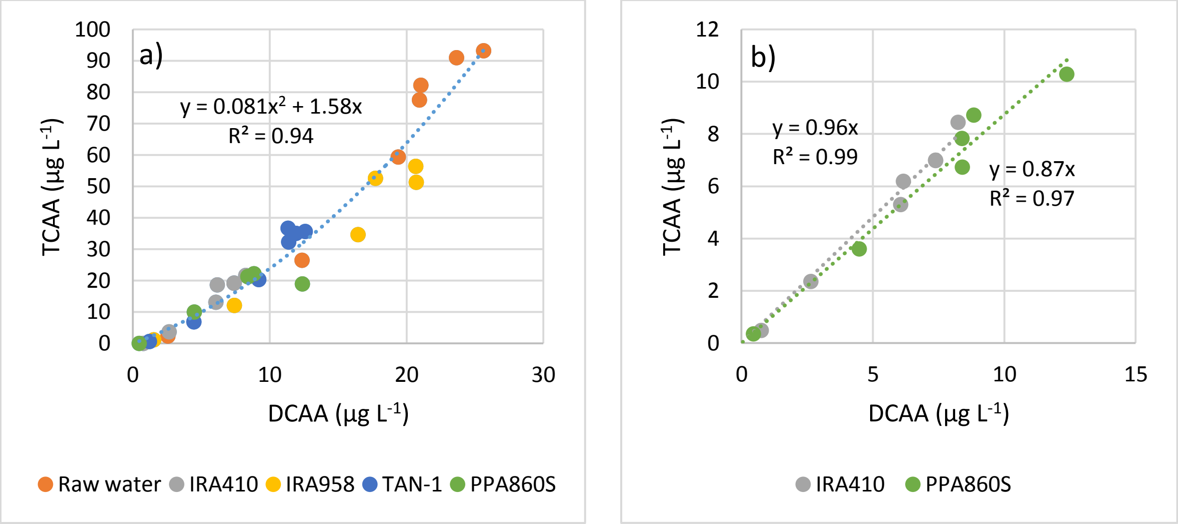

Figure V-21: a) Correlation between TCAA and DCAA for all samples b) Correlation between TCAA and DCAA for IRA410 and PPA860S.

When looking into more details, the curves were not quite linear for TCAA to DCAA ratio for raw waters and IRA958, with the ratio increasing towards the formation of TCAA for concentration higher than 10 µg L^{-1} , *i.e.* for increasing bromide concentrations (Figure V-21a)). For IRA410 and PPA860S, the TCAA to DCAA ratio remained the same for all samples (Figure V-21b)). This difference can be explained by the influence of the Br^-/DOC ratio with increase in the ratio for the lowest Br^-/DOC.

6.2.2.2. Between individual THMs and individual HAAs

In a previous study, the more brominated THMs (bromoform and dibromochloromethane) were those found with the highest predictive capacity, fully or partially, explaining the concentrations of eight of the nine HAAs examined, including chlorinated HAAs such as DCAA (Villanueva et al. 2003). While the correlation between $CHCl_3$ and TCAA was not strong (r = 0.658) (Villanueva et al. 2003).

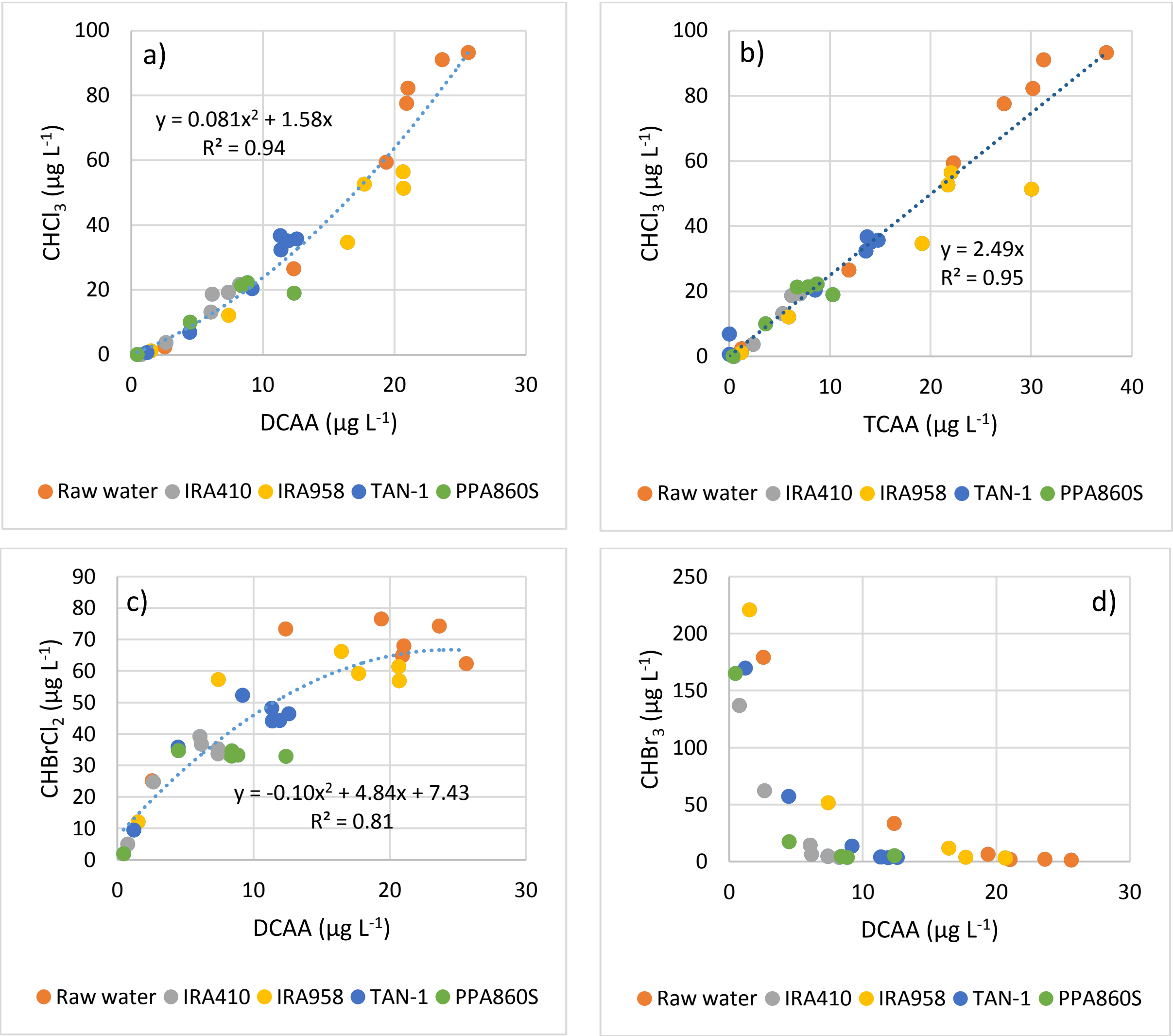

Figure V-22: Correlation between individual THMs and individual HAAs: a) $CHCl_3$ and DCAA, b) $CHCl_3$ and TCAA, c) $CHBrCl_2$ and DCAA, and d) $CHBr_3$ and DCAA.

In this study on the other hand, the best THM predictor for several chlorinated HAAs was chloroform, with $CHCl_3$ highly correlated to TCAA and DCAA ($r > 0.9$, Figure V-22). $CHCl_3$ and TCAA in particular were linearly correlated (Figure V-22b)), especially in non-spiked waters ($r^2 = 0.998$, not presented), indicating possible common precursors (Bougeard et al. 2010).

The difference between both studies lies on the fact that Villanueva et al. (2003) analysed different raw surface water samples without spiking with bromide, whereas this study used the same source but treated by different resins before halide spiking. The brominated tri-HAAs would be expected to be better correlated to bromoform and dibromochloromethane, in particular for dibromochloromethane and dibromochloroacetic acid for which Villanueva et al. (2003) found the highest correlation.

7. Conclusion

The raw water sample contained a mixture of hydrophobic and hydrophilic compounds, while the treated waters contained mainly hydrophilic compounds, which explained the reduced formation potentials in THMs and HAAs.

It can be observed that IRA410 is overall the best resin, followed by PPA860S in terms of UV254 absorbance, DOC and SUVA254 reduction. The IRA410 resin decreases about 70% of UV254, which can be attributed to the aromatic NOM fraction. The DOC is reduced by 41% and the reduction of SUVA254 is at 62%. However a major part of the practical exchange capacity of the resin goes to the unwanted removal of sulfate (> 64%), whereas only 10-19% is used effectively for DOC removal.

Additionally an improvement of the regeneration of the resins is required if the IRA410 is to be used in the full scale plant, as the chloride concentration in the treated water exceeded the drinking water limit.

The only resin for which the treated water did not exceed the chloride concentration limit was the PPA860S resin. However, while the formation potential tests showed IRA410 and PPA860S as the best resins for reduction of THM and HAA formation, PPA860S is not as good at reducing the formation potential for I-THMs and care should be taken if the iodide concentrations increase in the source water in the future.

8. Bibliography

Standard method, 2017. Standard Methods For the Examination of Water and Wastewater (5710). '**Formation of trihalomethanes and other disinfection by-products**'. (2017). *https://doi.org/10.2105/SMWW.2882.112*.

Allard et al. 2015. S. Allard, J. Tan, C. A. Joll, and U. von Gunten. '**Mechanistic Study on the Formation of Cl-/Br-/I-Trihalomethanes during Chlorination/Chloramination Combined with a Theoretical Cytotoxicity Evaluation**'. Environmental Science & Technology, 49 (18) (2015), pp. 11105-11114. *https://doi.org/10.1021/acs.est.5b02624*.

Allpike et al. 2005. B. P. Allpike, A. Heitz, C. A. Joll, R. I. Kagi, G. Abbt-Braun, F. H. Frimmel, T. Brinkmann, N. Her, and G. Amy. '**Size Exclusion Chromatography To Characterize DOC Removal in Drinking Water Treatment**'. Environmental Science & Technology, 39 (7) (2005), pp. 2334-2342. *https://doi.org/10.1021/es0496468*.

Ates et al. 2007. N. Ates, M. Kitis, and U. Yetis. '**Formation of Chlorination By-Products in Waters with Low SUVA—Correlations with SUVA and Differential UV Spectroscopy**'. Water Research, 41 (18) (2007), pp. 4139-4348. *https://doi.org/10.1016/j.watres.2007.05.042*.

Awad et al. 2016. J. Awad, J. van Leeuwen, C. Chow, M. Drikas, R. J. Smernik, D. J. Chittleborough, and E. Bestland. '**Characterization of Dissolved Organic Matter for Prediction of Trihalomethane Formation Potential in Surface and Sub-Surface Waters**'. Journal of Hazardous Materials, 308 (2016), pp. 430-439. *https://doi.org/10.1016/j.jhazmat.2016.01.030*.

Baert et al. 1996. R. Baert, R. Loontiens, and M. Devos. '**Potassium and Nitrate Pollution of Surface Water in the Catchment Area of the "Blankaert" Water Production Centre in Flanders (Belgium)**'. In *Proceedings _ 4th International Conference On Environmental Compliance And Enforcement*, (2) (1996), pp. 635-642. Chiang Mai, Thailand.

Balaram et al. 2012. V. Balaram, M. Satyanarananan, D. V. Avdeev, N. Berdnikov, P. Roy, S. S. Sawant, K. S. V. Subramanyam, K. V. Anjaiah, C. T. Kamala, R. Mathur, and B. Dasaram. '**Use of Xenon as Internal Standard for the Accurate Determination of Trace Elements in Water Samples by ICP-MS**'. Atomic Spectroscopy, 33 (2012), pp. 41-47.

Bazri et al. 2016. M. M. Bazri, B. Martijn, J. Kroesbergen, and M. Mohseni. '**Impact of Anionic Ion Exchange Resins on NOM Fractions: Effect on N-DBPs and C-DBPs Precursors**'. Chemosphere, 144 (2016), pp. 1988-1995. *https://doi.org/10.1016/j.chemosphere.2015.10.086*.

Bolto et al. 2002. B. Bolto, D. Dixon, R. Eldridge, S. King, and K. Linge. '**Removal of Natural Organic Matter by Ion Exchange**'. Water Research, 36 (20) (2002), pp. 5057-5065. *https://doi.org/10.1016/S0043-1354(02)00231-2*.

Bolto et al. 2004. B. Bolto, D. Dixon, and R. Eldridge. '**Ion Exchange for the Removal of Natural Organic Matter**'. Reactive and Functional Polymers, 60 (2004), pp. 171-182. *https://doi.org/10.1016/j.reactfunctpolym.2004.02.021*.

Bond et al. 2010. T. Bond, E.H. Goslan, S. A. Parsons, and B. Jefferson. '**Disinfection By-Product Formation of Natural Organic Matter Surrogates and Treatment by Coagulation, MIEX® and Nanofiltration**'. Water Research, 44 (5) (2010), pp. 1645-1653. *https://doi.org/10.1016/j.watres.2009.11.018*.

Bond et al. 2014. T. Bond, J. Huang, N. J. D. Graham, and M. R. Templeton. '**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 (2014), pp. 469-479. *https://doi.org/10.1016/j.scitotenv.2013.09.106*.

Bougeard et al. 2010. C. M. M. Bougeard, E. H. Goslan, B. Jefferson, and S. A. Parsons. '**Comparison of the Disinfection By-Product Formation Potential of Treated Waters Exposed to Chlorine and Monochloramine**'. Water Research, 44 (3) (2010), pp. 729-740. *https://doi.org/10.1016/j.watres.2009.10.008*.

Boyer and Singer 2005. T. H. Boyer and P. C. Singer. '**Bench-Scale Testing of a Magnetic Ion Exchange Resin for Removal of Disinfection by-Product Precursors**'. Water Research, 39 (7) (2005), pp. 1265-1276. *https://doi.org/10.1016/j.watres.2005.01.002*.

Boyer and Singer 2006. T. H. Boyer and P. C. Singer. '**A Pilot-Scale Evaluation of Magnetic Ion Exchange Treatment for Removal of Natural Organic Material and Inorganic Anions**'. Water Research, 40 (15), pp. 2865-2876. *https://doi.org/10.1016/j.watres.2006.05.022*.

Boyer and Singer 2008. T. H. Boyer and P. C. Singer. '**Stoichiometry of Removal of Natural Organic Matter by Ion Exchange**'. Environmental Science & Technology, 42 (2) (2008), pp. 608-613. *https://doi.org/10.1021/es071940n*.

Brezinksi et al. 2019. K. Brezinski, B. Gorczyca, and M. Sadrnourmohammadi. '**Ion-Exchange for Trihalomethane Control in Potable Water Treatment – a Municipal Water Treatment Case Study in Rainy River, Ontario, Canada**'. Water Quality Research Journal, 54 (2) (2019), pp. 142-160. *https://doi.org/10.2166/wqrj.2018.134*.

Clark et al. 2010. J. M. Clark, S. H. Bottrell, C. D. Evans, D. T. Monteith, R. Bartlett, R. Rose, R. J. Newton, and P. J. Chapman. '**The Importance of the Relationship between Scale and Process in Understanding Long-Term DOC Dynamics**'. Science of The Total Environment, 408 (13) (2010), pp. 2768-2775. *https://doi.org/10.1016/j.scitotenv.2010.02.046*.

Cornelissen et al. 2009. E.R. Cornelissen, E. F. Beerendonk, M. N. Nederlof, J. P. van der Hoek, and L. P. Wessels. '**Fluidized Ion Exchange (FIX) to Control NOM Fouling in Ultrafiltration**'. Desalination, 236 (1–3) (2009), pp. 334-3341. *https://doi.org/10.1016/j.desal.2007.10.084*.

Cornelissen et al. 2010. E. R. Cornelissen, D. Chasseriaud, W. G. Siegers, E. F. Beerendonk, and D. van der Kooij. '**Effect of Anionic Fluidized Ion Exchange (FIX) Pre-Treatment on Nanofiltration (NF) Membrane Fouling**'. Water Research, 44 (10) (2010), pp. 3283-3293. *https://doi.org/10.1016/j.watres.2010.03.007*.

Cornelissen et al. 2008. E. R. Cornelissen, N. Moreau, W. G. Siegers, A. J. Abrahamse, L. C. Rietveld, A. Grefte, M. Dignum, G. Amy, and L. P. Wessels. '**Selection of Anionic Exchange Resins for Removal of Natural Organic Matter (NOM) Fractions**'. Water Research, 42 (1–2) (2008), pp. 413-423. *https://doi.org/10.1016/j.watres.2007.07.033*.

Cromphout et al. 2008. J. Cromphout, L. Verdickt, E. Martin, R. Vanhoucke, and T. Vanhullebusch. '**Comparison between Magnetic Ion Exchange Resin-Ultrafiltration and Enhanced Coagulation–Filtration for the Treatment of an NOM Loaded Surface Water**'. Water Science & Technology: Water Supply, 8 (2008). *https://doi.org/10.2166/ws.2008.147*.

Croué et al. 1999. J-P. Croué, D. Violleau, C. Bodaire, and B. Legube. '**Removal of Hydrophobic and Hydrophilic Constituents by Anion Exchange Resin**'. Water Science and Technology, 40 (9) (1999), pp. 207-214. *https://doi.org/10.1016/S0273-1223(99)00658-7*.

De Watergroep. 2015. 'Annual Report'. Activity report De Watergroep 2015.

Dixit 2017. F. Dixit. '**Anion Exchange Resins for the Removal of Microcystins from Surface Water**'. Submitted for the degree of Master of Applied Science at the University of British Columbia (2017). *https://doi.org/10.14288/1.0355879*.

Drikas et al. 2003. M. Drikas, C. W. K. Chow, and D. Cook. '**The Impact of Recalcitrant Organic Character on Disinfection Stability, Trihalomethane Formation and Bacterial Regrowth: An Evaluation of Magnetic Ion Exchange Resin (MIEX®) and Alum Coagulation**'. Journal of Water Supply: Research and Technology-Aqua, 52 (7) (2003), pp. 475-487. *https://doi.org/10.2166/aqua.2003.0043*.

Edzwald and Tobiason 1999.J. K. Edzwald and J. E. Tobiason. '**Enhanced Coagulation: US Requirements and a Broader View**'. Water Science and Technology, 40 (9) (1999), pp. 63-70. *https://doi.org/10.1016/S0273-1223(99)00641-1*.

Evans et al. 2005. C. D. Evans, D. T. Monteith, and D. M. Cooper. '**Long-Term Increases in Surface Water Dissolved Organic Carbon: Observations, Possible Causes and Environmental Impacts**'. Environmental Pollution, 137 (1) (2005), pp. 55-71. *https://doi.org/10.1016/j.envpol.2004.12.031*.

Fearing et al. 2004. D. A. Fearing, J. Banks, D. Wilson, P. H. Hillis, A. T. Campbell, and S. A. Parsons. '**NOM Control Options: The next Generation**'. Water Science and Technology: Water Supply, 4 (4) (2004), pp. 139-145. *https://doi.org/10.2166/ws.2004.0071*.

Fettig 1999. J. Fettig. '**Removal of Humic Substances by Adsorption/Ion Exchange**'. Water Science and Technology, 40 (9) (1999), pp. 173-182. *https://doi.org/10.1016/S0273-1223(99)00654-X*.

Galjaard et al. 2018. G. Galjaard, E. Koreman, D. Metcalfe, G. Moore, and P. Ericsson. '**NOM-Removal by the SIX®-Process**'. Water Practice and Technology, 13 (3) (2018), pp. 524-541. *https://doi.org/10.2166/wpt.2018.072*.

Galjaard et al. 2010. G. Galjaard, P. C. Kamp, and E. Koreman. '**SIX®: A New Resin Treatment Technology for Drinking Water**'. Water Practice & Technology, 4 (2010), 4 p. *https://doi.org/10.2166/wpt.2009.076*.

Gan et al. 2013. X. Gan, T. Karanfil, S. S. Kaplan Bekaroglu, and J. Shan. '**The Control of N-DBP and C-DBP Precursors with MIEX®**'. Water Research, 47 (3) (2013), pp. 1344-1352. *https://doi.org/10.1016/j.watres.2012.11.049*.

Ghoochani et al. 2013. M. Ghoochani, N. Rastkari, R. Nabizadeh Nodehi, A. Mahvi, S. Nasseri, and S. Nazmara. '**Study on the TOC Concentration in Raw Water and HAAs in Tehran's Water Treatment Plant Outlet**'. Journal of Environmental Health Science and Engineering, 11 (1) (2013), pp. 28. *https://doi.org/10.1186/2052-336X-11-28*.

Goslan et al. 2009. E. H. Goslan, S. W. Krasner, M. Bower, S. A. Rocks, P. Holmes, L. S. Levy, and S. A. Parsons. '**A Comparison of Disinfection By-Products Found in Chlorinated and Chloraminated Drinking Waters in Scotland**'. Water Research, 43 (18) (2009), pp. 4698-4706. *https://doi.org/10.1016/j.watres.2009.07.029*.

Graf et al. 2014. K. C. Graf, D. A. Cornwell and T. H. Boyer. '**Removal of Dissolved Organic Carbon from Surface Water by Anion Exchange and Adsorption: Bench-Scale Testing to Simulate a Two-Stage Countercurrent Process**'. Separation and Purification Technology, 122 (2014), pp. 523-532. *https://doi.org/10.1016/j.seppur.2013.12.012*.

Grefte et al. 2013. A. Grefte, M. Dignum, E. R. Cornelissen, and L. C. Rietveld. '**Natural Organic Matter Removal by Ion Exchange at Different Positions in the Drinking Water Treatment Lane**'. Drinking Water Engineering and Science, 6 (1) (2013), pp. 1-10. *https://doi.org/10.5194/dwes-6-1-2013*.

Hong et al. 2009. H. C. Hong, M. H. Wong and Y. Liang. '**Amino Acids as Precursors of Trihalomethane and Haloacetic Acid Formation During Chlorination**'. Archives of Environmental Contamination and Toxicology, 56 (4) (2009), pp. 638-645. *https://doi.org/10.1007/s00244-008-9216-4*.

Hong et al. 2017. H. Hong, X. Yan, X. Song, Y. Qin, H. Sun, H. Lin, J. Chen, and Y. Liang. '**Bromine Incorporation into Five DBP Classes upon Chlorination of Water with Extremely Low SUVA Values**'. Science of The Total Environment, 590–591 (2017), pp. 720-728. *https://doi.org/10.1016/j.scitotenv.2017.03.032*.

Hua and Reckhow 2007. G. Hua and D. A. Reckhow. '**Characterization of Disinfection Byproduct Precursors Based on Hydrophobicity and Molecular Size**'. Environmental Science & Technology, 41 (9) (2007), pp. 3309-3315. *http://pubs.acs.org/doi/abs/10.1021/es062178c*.

Hua and Reckhow 2012. G. Hua and D. A. Reckhow. '**Evaluation of Bromine Substitution Factors of DBPs during Chlorination and Chloramination**'. Water Research, 46 (13) (2012), pp. 4208-4216. *https://doi.org/10.1016/j.watres.2012.05.031*.

Hua et al. 2015. G. Hua, D. A. Reckhow, and I. Abusallout. '**Correlation between SUVA and DBP Formation during Chlorination and Chloramination of NOM Fractions from Different Sources**'. Chemosphere, 130 (2015), pp. 82-89. *https://doi.org/10.1016/j.chemosphere.2015.03.039*.

Hu et al. 2006. G. Hua, D. A. Reckhow, and J. Kim. '**Effect of Bromide and Iodide Ions on the Formation and Speciation of Disinfection Byproducts during Chlorination**'. Environmental Science & Technology, 40 (9) (2006), pp. 3050-3056. *https://doi.org/10.1021/es0519278*.

Huber et al. 2011. S. A. Huber, A. Balz, M. Abert, and W. Pronk. '**Characterisation of Aquatic Humic and Non-Humic Matter with Size-Exclusion Chromatography – Organic Carbon Detection – Organic Nitrogen Detection (LC-OCD-OND)**'. Water Research, 45 (2) (2011), pp. 879-885. *https://doi.org/10.1016/j.watres.2010.09.023*.

Humbert et al. 2005. H. Humbert, H. Gallard, H. Suty, and J-P. Croué. '**Performance of Selected Anion Exchange Resins for the Treatment of a High DOC Content Surface Water**'. Water Research, 39 (9) (2005), pp. 1699-1708. *https://doi.org/10.1016/j.watres.2005.02.008*.

Humbert et al. 2008. H. Humbert, H. Gallard, H. Suty, and J-P. Croué. '**Natural Organic Matter (NOM) and Pesticides Removal Using a Combination of Ion Exchange Resin and Powdered Activated Carbon (PAC)**'. Water Research, 42 (6–7) (2008), pp. 1635-1643. *https://doi.org/10.1016/j.watres.2007.10.012*.

Jarvis et al. 2008. P. Jarvis, M. Mergen, J. Banks, B. McIntosh, S. A. Parsons, and B. Jefferson. '**Pilot Scale Comparison of Enhanced Coagulation with Magnetic Resin Plus Coagulation Systems**'. Environmental Science & Technology, 42 (4) (2008), pp. 1276-1282. *https://doi.org/10.1021/es071566r*.

Kalscheur et al. 2012. K. N. Kalscheur, R. R. Penskar, A. D. Daley, S. M. Pechauer, J. J. Kelly, C. G. Peterson, and K. A. Gray. '**Effects of Anthropogenic Inputs on the Organic Quality of Urbanized Streams**'. Water Research, 46 (8) (2012), pp. 2515-2524. *https://doi.org/10.1016/j.watres.2012.01.043*.

Kim and Dempsey 2010. H-C. Kim and B. A. Dempsey. '**Removal of Organic Acids from EfOM Using Anion Exchange Resins and Consequent Reduction of Fouling in UF and MF**'. Journal of Membrane Science, 364 (1–2) (2010), pp. 325-330. *https://doi.org/10.1016/j.memsci.2010.08.032*.

Kitis et al. 2001. M. Kitis, T. Karanfil, J. E. Kilduff, and A. Wigton. '**The Reactivity of Natural Organic Matter to Disinfection By-Products Formation and Its Relation to Specific Ultraviolet Absorbance**'. Water Science and Technology, 43 (2) (2001), pp. 9-16. *https://doi.org/10.2166/wst.2001.0067*.

Kitis et al. 2004. M. Kitis, T. Karanfil, J. E. Kilduff. '**The Reactivity of Dissolved Organic Matter for Disinfection By-Product Formation**'. Turkish Journal of Engineering and Environmental Sciences, 28 (3) (2004), pp. 167-179.

Kitis et al. 2002. M. Kitis, T. Karanfil, A. Wigton, and J. E. Kilduff. '**Probing Reactivity of Dissolved Organic Matter for Disinfection By-Product Formation Using XAD-8 Resin Adsorption and Ultrafiltration Fractionation**'. Water Research, 36 (15) (2002), pp. 3834-3848. *https://doi.org/10.1016/S0043-1354(02)00094-5*.

Lee et al. 2004. N. Lee, G. Amy, J-P. Croué, and H. Buisson. '**Identification and Understanding of Fouling in Low-Pressure Membrane (MF/UF) Filtration by Natural Organic Matter (NOM)**'. Water Research, 38 (20) 2004), pp. 4511-4523. *https://doi.org/10.1016/j.watres.2004.08.013*.

Levchuk et al. 2018. I. Levchuk, J. J. Rueda Márquez, and M. Sillanpää. '**Removal of Natural Organic Matter (NOM) from Water by Ion Exchange – A Review**'. Chemosphere, 192 (2018), pp. 90-104. *https://doi.org/10.1016/j.chemosphere.2017.10.101*.

Li and Mitch 2018. X-F. Li and W. A. Mitch. '**Drinking Water Disinfection Byproducts (DBPs) and Human Health Effects: Multidisciplinary Challenges and Opportunities**'. Environmental Science & Technology, 52 (4) (2018), pp. 1681-1689. *https://doi.org/10.1021/acs.est.7b05440*.

Liu 2017. C. Liu. '**Removal of Perfluorinated Compounds in Drinking Water Treatment: A Study of Ion Exchange Resins and Magnetic Nanoparticles**'. For the degree of Doctor of Philosophy, Waterloo, Ontario, Canada: University of Waterloo, (2017). *https://uwspace.uwaterloo.ca/handle/10012/12660*.

Matilainen et al. 2010. A. Matilainen, M. Vepsäläinen, and M. Sillanpää. '**Natural Organic Matter Removal by Coagulation during Drinking Water Treatment: A Review**'. Advances in Colloid and Interface Science, 159 (2) (2010), pp. 189-197. *https://doi.org/10.1016/j.cis.2010.06.007*.

Mergen et al. 2009. M. R. D. Mergen, B. J. Adams, G. M. Vero, T. A. Price, S. A. Parsons, B. Jefferson, and P. Jarvis. '**Characterisation of Natural Organic Matter (NOM) Removed by Magnetic Ion Exchange Resin (MIEX® Resin)**'. Water Science and Technology: Water Supply, 9 (2) (2009), pp. 199-205. *https://doi.org/10.2166/ws.2009.301*.

Pelekani and Sneyink 1999. C. Pelekani, C, and V.L. Snoeyink. '**Competitive Adsorption in Natural Water: Role of Activated Carbon Pore Size**'. Water Research, 33 (5) (1999), pp. 1209-1219. *https://doi.org/10.1016/S0043-1354(98)00329-7*.

Plewa et al. 2004. M. J. Plewa, E. D. Wagner, S. D. Richardson, A. D. Thruston, Y-T Woo, and A. B. McKague. 2004. '**Chemical and Biological Characterization of Newly Discovered Iodoacid Drinking Water Disinfection Byproducts**'. Environmental Science & Technology, 38 (18), pp. 4713-4722. *https://doi.org/10.1021/es049971v*.

Rodier et al. 2016. J. Rodier, B. Legube, and N. Merlet. 'L'analyse de l'eau'. In , 10e édition, 1600. Technique et Ingénierie, Dunod.

Ruggiero et al. 2006. A. Ruggiero, A. G. Solimini, and G. Carchini. '**Effects of a Waste Water Treatment Plant on Organic Matter Dynamics and Ecosystem Functioning in a Mediterranean Stream**'. Annales de Limnologie - International Journal of Limnology, 42 (2) (2006), pp. 97-107. *https://doi.org/10.1051/limn/2006014*.

Sadmani 2014. A. Sadmani. '**The Rejection of Pharmaceutically Active and Endocrine Disrupting Compounds via Nanofiltration as a Function of Natural Water Components**'. For the degree of Doctor of Philosophy, University of Toronto (2014).

Sawade et al. 2016. E. Sawade, R. Fabris, A. Humpage, and M. Drikas. '**Effect of Increasing Bromide Concentration on Toxicity in Treated Drinking Water**'. Journal of Water and Health, 14 (2) (2016), pp. 183-191. *https://doi.org/10.2166/wh.2015.127*.

Sedlak and von Gunten 2011. D.S. Sedlak and U. von Gunten U. 2011. 'The Chlorine Dilemma'. *Science*, 2011, sec. Vol. 331, Issue 6013.

Sillanpää and Bhatnagar 2015. M. Sillanpää and A. Bhatnagar. '**Chapter 7 - NOM Removal by Adsorption**'. In *Natural Organic Matter in Water*, edited by Mika Sillanpää, (2015), pp. 213-238. Butterworth-Heinemann. *https://doi.org/10.1016/B978-0-12-801503-2.00007-0*.

Sillanpää et al. 2015. M. Sillanpää, S. Metsämuuronen, and M. Mänttäri. '**Chapter 5 - Membranes**'. In *Natural Organic Matter in Water*, edited by Mika Sillanpää, (2015) pp. 113-157. Butterworth-Heinemann. *https://doi.org/10.1016/B978-0-12-801503-2.00005-7*.

Singer and Bilyk 2002. P. C. Singer and K. Bilyk. '**Enhanced Coagulation Using a Magnetic Ion Exchange Resin**'. Water Research, 36 (16) (2002), pp. 4009-4022. *https://doi.org/10.1016/S0043-1354(02)00115-X*.

Song et al. 2010. J. Song, W. Huang, P. Peng, B. Xiao, and Y. Ma. 2010. '**Humic Acid Molecular Weight Estimation by High-Performance Size-Exclusion Chromatography with Ultraviolet Absorbance Detection and Refractive Index Detection**'. Soil Science Society of America Journal, 74 (6) (2010), pp. 2013-2020. *https://doi.org/10.2136/sssaj2010.0011*.

Tan et al. 2016. J. Tan, S. Allard, Y. Gruchlik, S. McDonald, C.A. Joll, and A. Heitz. '**Impact of Bromide on Halogen Incorporation into Organic Moieties in Chlorinated Drinking Water Treatment and Distribution Systems**'. Science of The Total Environment, 541 (2016), pp. 1572-1580. *https://doi.org/10.1016/j.scitotenv.2015.10.043*.

Tan and Kilduff 2017. Y. Tan and J. E. Kilduff. '**Factors Affecting Selectivity during Dissolved Organic Matter Removal by Anion-Exchange Resins**'. Water Research, 41 (18) (2017), pp. 4211-4221. *https://doi.org/10.1016/j.watres.2007.05.050*.

Tan et al. 2005. Y. Tan, J. E. Kilduff, M. Kitis, and T. Karanfil. '**Dissolved Organic Matter Removal and Disinfection Byproduct Formation Control Using Ion Exchange**'. Desalination, Seminar in Environmental Science and Technology: Evaluation of Alternative Water Treatment Systems for Obtaining Safe Water, 176 (1) (2005), pp. 189-200. *https://doi.org/10.1016/j.desal.2004.10.019*.

Teixeira and Sousa 2013. M. R. Teixeira and V. S. Sousa. '**Fouling of Nanofiltration Membrane: Effects of NOM Molecular Weight and Microcystins**'. Desalination, 315 (2013), pp.149-155. *https://doi.org/10.1016/j.desal.2012.03.012*.

Verdickt et al. 2012. L. Verdickt, W. Closset, V. D'Haeseleer, and J. Cromphout. '**Applicability of Ion Exchange for NOM Removal from a Sulfate-Rich Surface Water Incorporating Full Reuse of the Brine**'. Water Science and Technology: Water Supply, 12 (6) (2012), pp. 878-887. *https://doi.org/10.2166/ws.2012.065*.

Verdickt and Schoutteten 2018. L. Verdickt and K. Schoutteten. '**Combining Ion Exchange and Coagulation/Flotation for Enhanced Natural Organic Matter Removal in Drinking Water Treatment**'. Presented at the IWA World Water Congress, Tokyo (2018).

Verliefde et al. 2008. A.R.D. Verliefde, E.R. Cornelissen, S.G.J. Heijman, I. Petrinic, T. Luxbacher, G.L. Amy, B. Van der Bruggen, and J.C. van Dijk. '**Influence of Membrane Fouling by (Pretreated) Surface Water on Rejection of Pharmaceutically Active Compounds (PhACs) by Nanofiltration Membranes**'. Journal of Membrane Science, 330 (1–2) (2008), pp. 90-103. *https://doi.org/10.1016/j.memsci.2008.12.039*.

Villanueva et al. 2003. C. M. Villanueva, M. Kogevinas, and J. O. Grimalt. '**Haloacetic Acids and Trihalomethanes in Finished Drinking Waters from Heterogeneous Sources**'. Water Research, 37 (4) (2003), pp. 953-958. *https://doi.org/10.1016/S0043-1354(02)00411-6*.

Villanueva et al. 2012. C. M. Villanueva, G. Castaño-Vinyals, V. Moreno, G. Carrasco-Turigas, N. Aragonés, E. Boldo, E. Ardanaz, et al. 2012. '**Concentrations and Correlations of Disinfection By-Products in Municipal Drinking Water from an Exposure Assessment Perspective**'. Environmental Research, 114 (2012), pp. 1-11. *https://doi.org/10.1016/j.envres.2012.02.002*.

Watson et al. 2015. K. Watson, M. J. Farré, and N. Knight. '**Enhanced Coagulation with Powdered Activated Carbon or MIEX® Secondary Treatment: A Comparison of Disinfection by-Product Formation and Precursor Removal**'. Water Research, 68 (2015), pp. 454-466. *https://doi.org/10.1016/j.watres.2014.09.042*.

Wei et al. 2010. J. Wei, B. Ye, W. Wang, L. Yang, J. Tao, and Z. Hang. '**Spatial and Temporal Evaluations of Disinfection By-Products in Drinking Water Distribution Systems in Beijing, China**'. Science of The Total Environment, 408 (20) (2010), pp. 4600-4606. *https://doi.org/10.1016/j.scitotenv.2010.06.053*.

Worral et al. 2004. F. Worrall, R. Harriman, C. D. Evans, C.D. Watts, J. Adamson, C. Neal, E. Tipping, et al. 2004. '**Trends in Dissolved Organic Carbon in UK Rivers and Lakes**'. Biogeochemistry, 70 (3) (2004), pp. 369-402. *https://doi.org/10.1007/s10533-004-8131-7*.

Xie 2016. Y. F. Xie. **Disinfection Byproducts in Drinking Water: Formation, Analysis, and Control**. CRC Press, (2016).

Yang et al. 2015. L. Yang, D. H. Han, B-M. Lee, and J. Hur. '**Characterizing Treated Wastewaters of Different Industries Using Clustered Fluorescence EEM–PARAFAC and FT-IR Spectroscopy: Implications for Downstream Impact and Source Identification**'. Chemosphere, 127 (2015), pp. 222-228. *https://doi.org/10.1016/j.chemosphere.2015.02.028*.

Ye et al. 2009. B. Ye, W. Wang, L. Yang, J. Wei, and X. E. 2009. '**Factors Influencing Disinfection By-Products Formation in Drinking Water of Six Cities in China**'. Journal of Hazardous Materials, 171 (1–3) (2009), pp. 147-152. *https://doi.org/10.1016/j.jhazmat.2009.05.117*.

Zeng and Arnold 2014. T. Zeng and W. A. Arnold. '**Clustering Chlorine Reactivity of Haloacetic Acid Precursors in Inland Lakes**'. Environmental Science & Technology, 48 (1) (2014), pp. 139-148. *https://doi.org/10.1021/es403766n*.

Zhai et al. 2017. H. Zhai, X. He, Y. Zhang, T. Du, A. S. Adeleye, and Y. Li. '**Disinfection Byproduct Formation in Drinking Water Sources: A Case Study of Yuqiao Reservoir**'. Chemosphere, 181 (2017), pp. 224-231. *https://doi.org/10.1016/j.chemosphere.2017.04.028*.

Zhang et al. 2015. T-Y. Zhang, B. Xu, C-Y. Hu, Y-L Lin, L. Lin, T. Ye, and F-X. Tian. '**A Comparison of Iodinated Trihalomethane Formation from Chlorine, Chlorine Dioxide and Potassium Permanganate Oxidation Processes**'. Water Research, 68 (2015), pp. 394-403. *https://doi.org/10.1016/j.watres.2014.09.040*.

Zhang et al. 2010. Y. Zhang, C. Collins, N. Graham, M. R. Templeton, J. Huang, and M. Nieuwenhuijsen. '**Speciation and Variation in the Occurrence of Haloacetic Acids in Three Water Supply Systems in England: Speciation and Variation in Occurrence of HAAs**'. Water and Environment Journal, 24 (3) (2010), pp. 237-245. *https://doi.org/10.1111/j.1747-6593.2009.00200.x*.

Zhou et al. 2000. Q. Zhou, S. E. Cabaniss, and P. A. Maurice. '**Considerations in the Use of High-Pressure Size Exclusion Chromatography (HPSEC) for Determining Molecular Weights of Aquatic Humic Substances**'. Water Research, 34 (14) (2000), pp. 3505-3514. *https://doi.org/10.1016/S0043-1354(00)00115-9*.

Zhou et al. 2019. X. Zhou, L. Zheng, S. Chen, H. Du, B. M. Gakoko Raphael, Q. Song, F. Wu, J. Chen, H. Lin, and H. Hong. '**Factors Influencing DBPs Occurrence in Tap Water of Jinhua Region in Zhejiang Province, China**'. Ecotoxicology and Environmental Safety, 171 (2019), pp. 813-822. *https://doi.org/10.1016/j.ecoenv.2018.12.106*.

Zularisam et al. 2007. A.W. Zularisam, A.F. Ismail, M.R. Salim, M. Sakinah, and O. Hiroaki. '**Fabrication, Fouling and Foulant Analyses of Asymmetric Polysulfone (PSF) Ultrafiltration Membrane Fouled with Natural Organic Matter (NOM) Source Waters**'. Journal of Membrane Science 299 (1–2) (2007), pp. 97-113. *https://doi.org/10.1016/j.memsci.2007.04.030*.

9. Supporting information

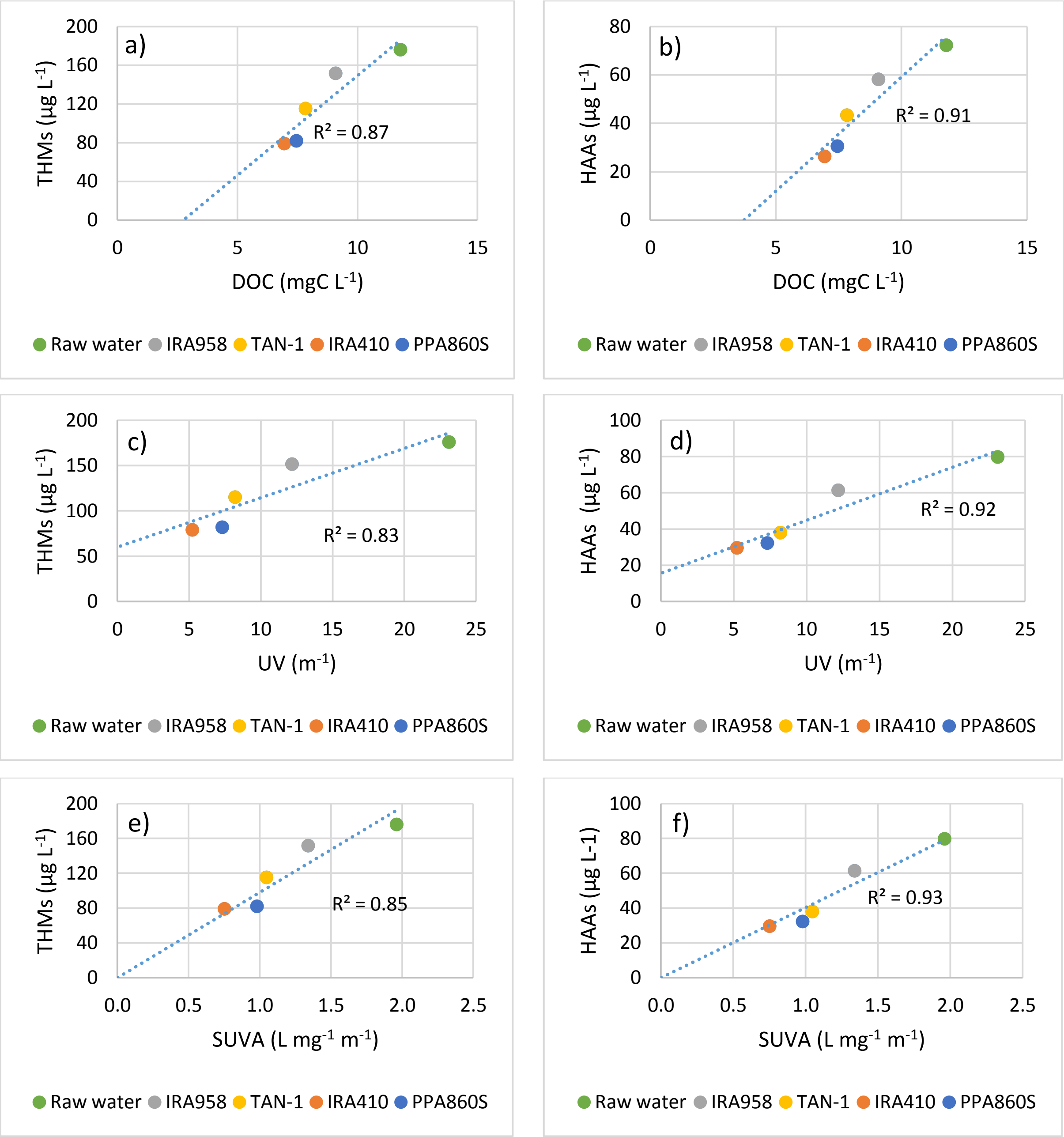

Figure V-S1: Correlation between the DBPFP for the different samples (raw and treated by the 4 different resins) and a) the DOC for THM$_{24h}$, b) the DOC for HAA$_{24h}$, c) the UV for THM$_{24h}$, d) the UV for HAA$_{24h}$, e) the SUVA for THM$_{24h}$, f) the SUVA for HAA$_{24h}$.

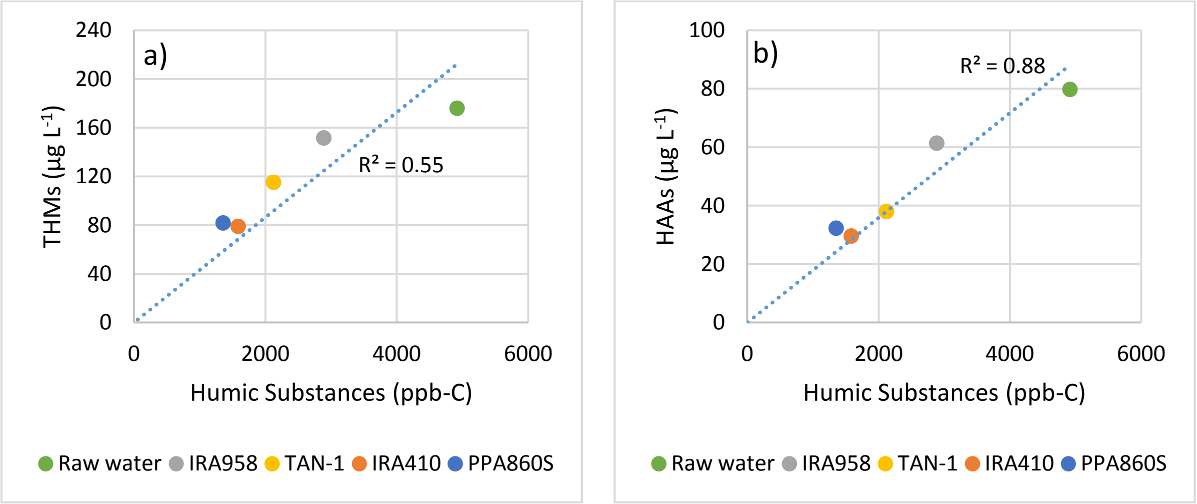

Figure V-S2: Correlation between the 24-h DBPFP for the different samples (raw and treated by the 4 different resins) and the HS fraction a) THMs and b) HAAs.

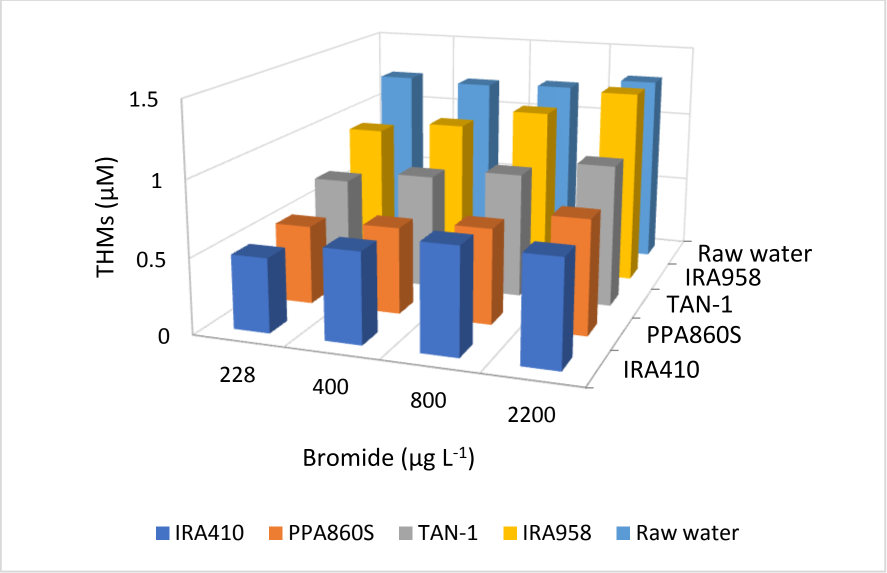

Figure V-S3: THM evolution in molarity as a function of initial bromide concentration for the 5 samples (Iodide = 63 µg L^{-1}).

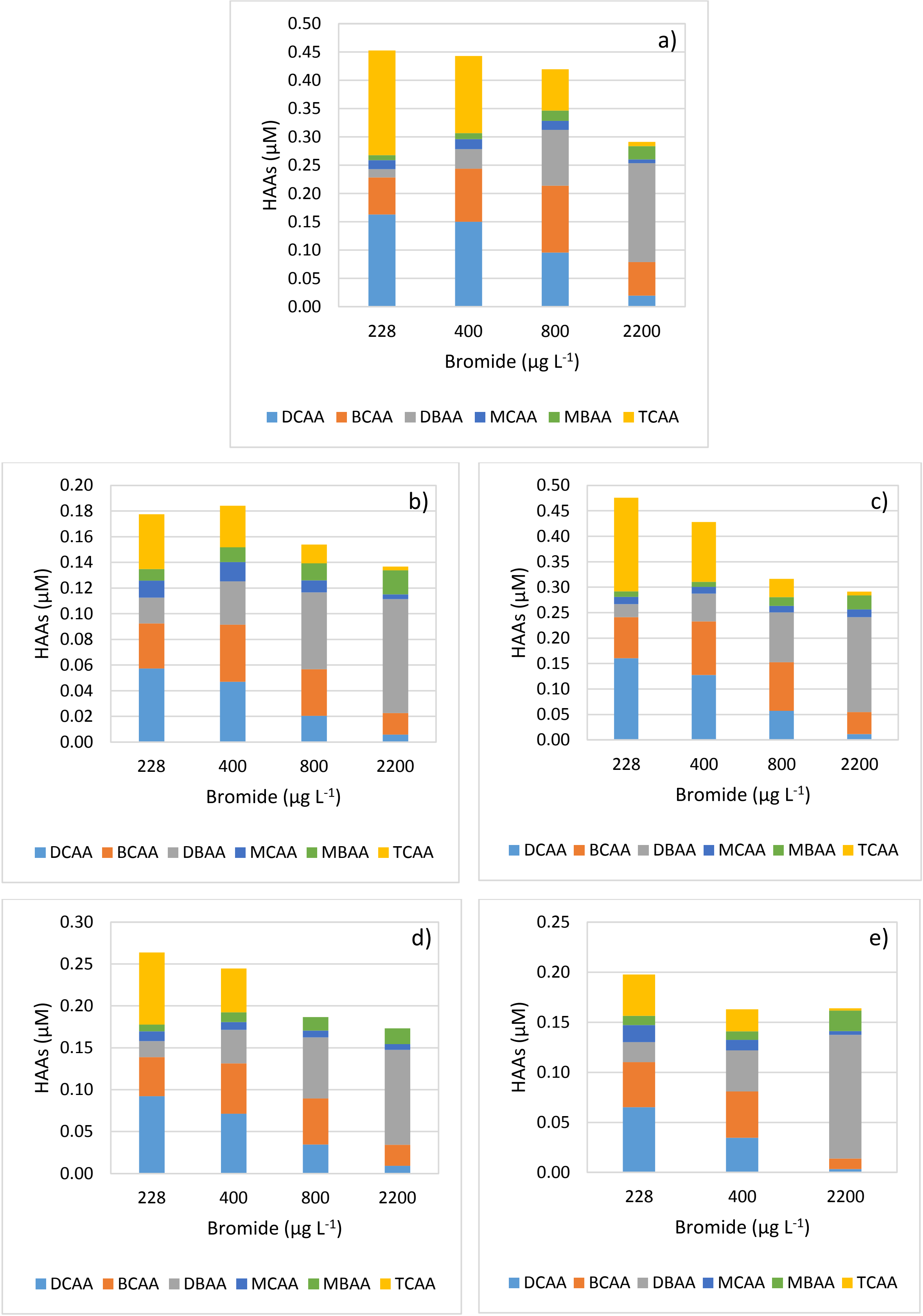

Figure V-S4: HAA molar concentration evolution as a function of initial bromide concentration for the 5 samples (iodide = 63 µg L^{-1}) a) Raw water, b) IRA410, c) IRA958, d) TAN-1, e) PPA860S.

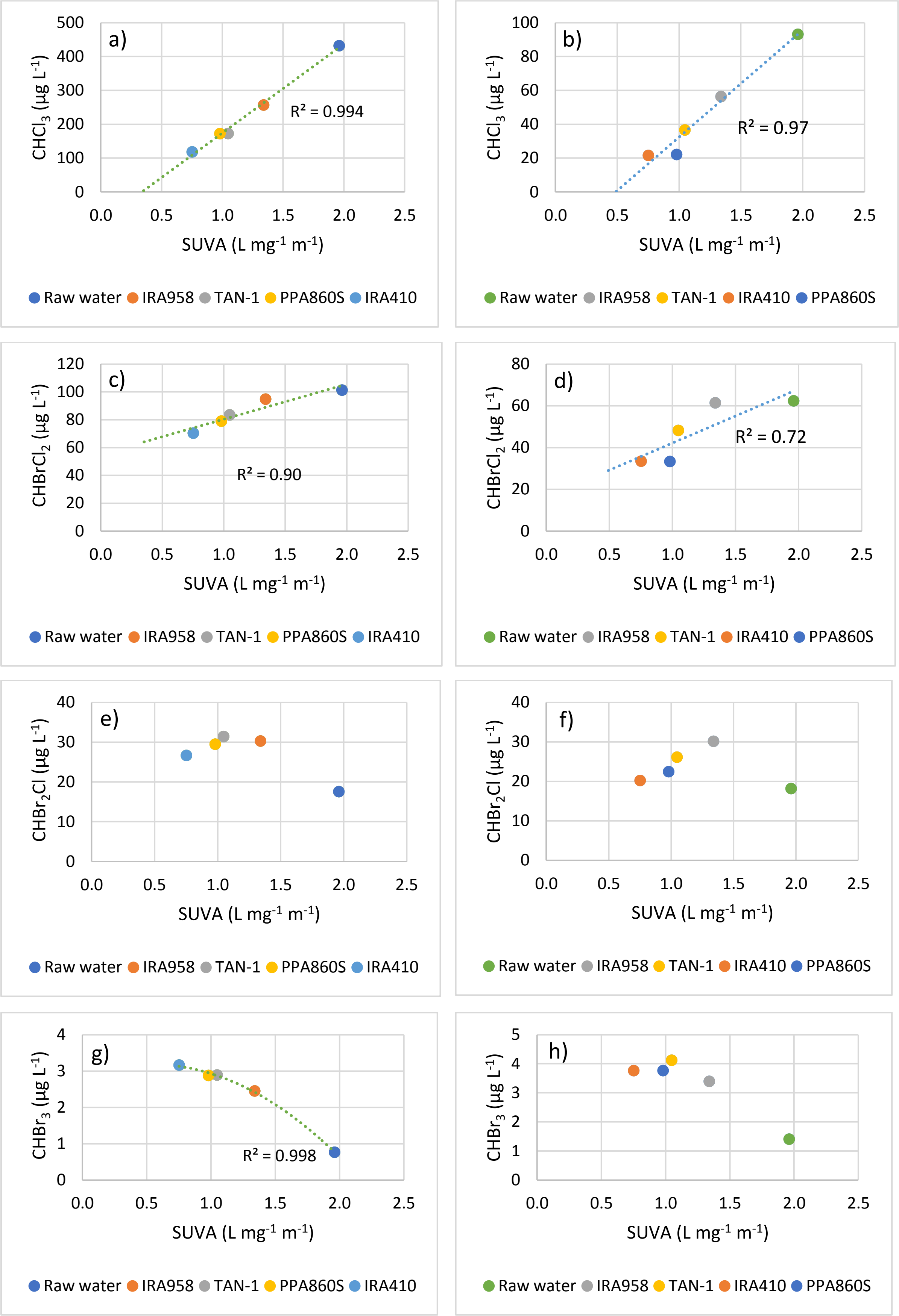

Figure V-S5: Evolution of the individual 7-day THMFP as function of the SUVA: a) chloroform, c) bromodichloromethane, e) dibromochloromethane, g) bromoform; and evolution of the individual THM_{24h} as function of the SUVA: b) chloroform, d) bromodichloromethane, f) dibromochloromethane, h) bromoform.

General conclusion

General conclusion

As part of the DOC2C's project, the objective of this work was to assess the formation of DBPs in the 2 seas area, by looking at the benefits of DOC removal by the different treatments of surface waters used by the project partners on DBP formation and speciation.

The review on the fate of iodide in drinking water treatment gives a comprehensive overview of the potential formation of reactive iodine species during oxidative water treatment of iodide-containing waters with various oxidants and their reaction with dissolved inorganic and organic compounds. Iodine reactive species are similar to chlorine and bromine in terms of speciation, mechanism of reaction or NOM incorporation. The main difference is the possible formation of iodate, safe end-product during chlorination. The fewer available rate constants do not allow a solid conclusion in terms of reactivity, which however appears generally to be intermediate between chlorine and bromine. The iodinated disinfection by-products are favored by chloramination process compared to chlorination due to the impossible formation of iodate by chloramines. Overall it remains difficult to compare occurrence and speciation of I-DBPs in real waters due to the lack of occurrence studies which also measure all the main parameters which can influence the incorporation of I^- (the oxidant type, I^-, Br^- and NOM concentrations, NOM type, solution pH as well as ammonium concentration). A better understanding would enable better mitigation of I-DBP formation, especially through processes which favor the formation of iodate, in particular controlled ozonation without formation of bromate or applying a substantial pre-chlorination contact time before chloramination. This is especially important as I-DBPs are known to have generally higher toxicity than their chlorinated and brominated analogues. Iodoacetic acid and iodoacetamids exibit particularly strong cytotoxicity.

During this PhD, the analysis of DBPs has been set up at the LASIR laboratory, 2 headspace-trap gas chromatography methods were developed: one for the 10 trihalomethanes – with a focus on iodinated species – and one for the haloacetic acids. Low detection limits were achieved compared to previous gas chromatography methods. Both methods have the main advantage of directly analysing water samples without any extraction required. The method for the THMs is simple, fast, and without the use of toxic solvents. The method for the HAAs, while effective, is constricted by time, as the analytes degrade relatively quickly in a water matrix. Additionally the 3 brominated tri-HAAs – are not measurable by this method. This method was not suitable for the analysis of haloacetonitriles nor haloacetamides, then, a new multimode injector (MMI – Multimode Inlet – Agilent) has been purchased to be set up on the recent GC-MS-MS instrument. The implementation of a temperature ramp on the injector will enable a better separation of these early eluters and will avoid any thermal degradation.

Two DBPFP studies were done in 2018 on a drinking water treatment plant in Belgium, belonging to De Watergroep, one of the partners of the DOC2C's project. The first study performed on samples taken from the full scale treatment plant give some valuable information on the relation between DBP formation and the removal of NOM fractions during the different steps of water treatment at this plant. Monitoring of iodine and inorganic iodine species showed very high conversion of iodine into iodate certainly due to the relatively high bromide level which catalyzes iodate formation. A degradation of organic iodinated compounds is observed in this water treatment plant with a larger formation of iodate compared to iodide levels in the raw water. Relative cytotoxicity estimations support the importance of inclusion of HAAs in the future European regulation considering their higher toxicity – despite their lower concentrations in the final waters of this treatment plant, their different behavior and formation compared to THMs presently used as a surrogate for all DBPs.

The second study, comparing four ion-exchange resins in terms of disinfection by-product precursor removal, showed low DOC removal compared to the literature, due to high competition with inorganic anions such as sulfate. Reductions in THM and HAA formation potentials based on 24-hour tests were higher and followed the same trend. Only one resin had chloride concentrations in the treated waters lower than the European limit. However, despite being one of the two best resins for THMFP and HAAFP reduction, this resin was not as effective for I-THM removal and care should be taken if the iodide concentrations increase in the source water in the future.

In addition to what is presented in this manuscript, a substantial work has been performed in collaboration with Victoria Hawkes for her Master Thesis from the University of New Hampshire "Mechanism of by-product formation from different water matrices by chlorination and chloramination" (Sept. 2019). DBPFPs have been determined on waters from the drinking water treatment plant of Bangui (Central African Republic). This study showed that the treatment line, while quite old, removes efficiently the high molecular weight fractions of DOC and that the maximum formation of disinfection by-products is moderate in the treated water, demonstrating that the distributed water would certainly respect the drinking water guidelines in terms of disinfection by-products. Finally, DBP formation potentials have also been determined on a drinking water resource in Northern France which showed a substantial presence of iodide (32.8 $\mu g\ L^{-1}$) and the formation of iodoform which is rarely shown in the occurrence studies. The implementation of the nanofiltration process on this water source in Summer 2019, originally to reduce hardness, will also permit to deliver a water of a better quality in terms of disinfection by-products, taste and odours to the consumers.