Occurrence of regulated and non-regulated disinfection by-products in small drinking water systems

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HIGHLIGHTS

- The occurrence of 16 regulated and non-regulated DBPs were evaluated in 25 small systems.
- The gap between levels of regulated and non-regulated DBPs was comparable to that observed in large systems.
- All investigated DBPs followed a comparable seasonal evolution during the year.
- The seasonal fluctuations of DBPs were lower than those observed in large systems.
- Spatial variations of DBPs during distribution were high even if network sizes are small.

ABSTRACT

The occurrence of regulated and non-regulated disinfection by-products (DBPs) was investigated in the drinking water of small systems in two provinces in Canada, Newfoundland and Labrador (NL) and Quebec (QC), through an intensive sampling program. Sixteen DBPs were studied: four trihalomethanes (THMs), five haloacetic acids (HAAs), four haloacetonitriles (HANs), one halonitromethane, chloropikrin (CPK) and two haloketones (HKs). Average measured concentrations of these compounds were much higher than those reported in the literature for medium and large systems. The measured average value for THMs was $75 \mu g L^{-1}$ (Stdv = $69 \mu g L^{-1}$); HAAs, $77 \mu g L^{-1}$ (Stdv = $75 \mu g L^{-1}$); HANs, $2.5 \mu g L^{-1}$ (Stdv = $1.8 \mu g L^{-1}$); CPK, $0.4 \mu g L^{-1}$ (Stdv = $0.3 \mu g L^{-1}$) and HKs, $6.0 \mu g L^{-1}$ (Stdv = $4.5 \mu g L^{-1}$). The gap (some 10 times difference) between the average levels of regulated DBPs (THMs, HAAs) and non-regulated DBPs (HANs, CPK and HKs) is comparable to that observed in large systems where the occurrence of the same compounds has been reported. Generally, investigated DBPs followed a comparable seasonal evolution during the year: they decreased between the fall and winter and then increased to eventually reach a maximum in late summer. This trend was less observable in NL than in QC. However, observed seasonal fluctuations of DBPs were less considerable than those observed in medium and large systems located in similar temperate environments reported in the literature. Spatial variations from the plant to the extremities were high and comparable to those observed in large systems, which is surprising, considering the smaller size of distribution networks supplying small communities. Generally speaking, the results support the premise that problems associated with implementing treatment that removes DBP precursors in water submitted to chlorination can increase population exposure to these contaminants in small systems.

1. Introduction

Disinfection by-products (DBPs), generated by the reaction between a chemical disinfectant usually chlorine with organic matter, are an important concern for water supply, especially surface water supply, as they are generally rich in natural organic matter (Cedergren et al., 2002; Mouly et al., 2010).

DBPs constitute a large family of compounds presenting various levels of toxicological effects: more than 600 DBPs have been detected, but few have been identified (Richardson, 2011). Trihalomethanes (THMs) and haloacetic acids (HAAs) are the most prevalent DBPs in drinking water. Their formation is relatively well understood and their levels are regulated in various countries (in particular for THMs) (Singer, 2002; Richardson, 2011).

Recently, there has been an increased interest in investigating the presence of other DBPs, for example, haloacetonitriles (HANs), haloketones (HKs) and halonitromethanes (HNMs). In fact,
nitrogen DBPs (like HANs and HNMs) may have greater toxicological effects than HAAs and THMs (Muellner et al., 2007; Richardson et al., 2007), that can become a public health problem with the increasing use of alternative disinfectants as a way to reduce concentrations of regulated DBPs such as chloramines (Adams et al., 2005).

Most studies on the occurrence of regulated and non-regulated DBPs have been conducted in large systems (Rodriguez et al., 2004; Krasner et al., 2006; Goslan et al., 2009; Ye et al., 2009; Moufl et al., 2010). However, small water systems (i.e., serving 5,000 or fewer people) using surface waters may be more vulnerable to DBPs because of financial constraints, a relatively low capacity to implement adequate treatment technologies to remove contaminants, in particular DBP precursors, and an inability to hire qualified operators (Charrois et al., 2004; Coulibaly and Rodriguez, 2004; Edwards et al., 2012). There is presently very little information on the spatio-temporal variability of DBPs in the water of small communities. Only a few studies on the occurrence of regulated DBPs are available (Charrois et al., 2004; Tung and Xie, 2009). In many countries, as is the case in Canada, available data are particularly inexistant for non-regulated DBPs.

The purpose of this study is to improve knowledge on the occurrence of regulated and non-regulated DBPs in the drinking water of small communities in Canada. Accordingly, spatial evolution (inter-regions, intra-regions and along the distribution system) and temporal evolution (seasonally) of DBPs in water were investigated. For the first time (according to our knowledge), the spatial and temporal presence of non-regulated DBPs was investigated in the drinking water of small communities based on intensive and structured sampling programs.

2. Methodology

2.1. Case studies

Twenty-five small municipal systems were selected and studied in two provinces of Canada: Newfoundland and Labrador (NL) and Quebec (QC). Sampling campaigns in the systems were conducted monthly for one year between September 2010 and October 2011 (from September 2010 to September 2011 in NL and from October 2010 to October 2011 in QC). All systems were supplied by surface water sources and used chlorine as their main disinfectant (for primary and secondary disinfection). Systems in NL served a population varying from 330 to 2120 inhabitants. In QC, systems served a population varying from 1000 to 6220 inhabitants. Systems in NL did not present any prior treatment to chlorination, whereas in QC, systems had implemented one or more treatment processes prior to disinfection.

2.2. Sampling and analysis

During this study, 1500 samples were collected representing over 21000 data for numerous parameters. Water was sampled at source (RW) and in the water treatment plant (WTP) just after filtration and before chlorination. Various points were identified along the distribution system (DS) in order to collect water samples at different residence times (Table 1). Water was sampled at the beginning (DS1), middle (DS2) and end of the DS (DS3). In NL, systems had no treatment prior to chlorination, chlorination being the main treatment process. Thus, in NL, RW and WTP were represented by the same point. Samples were collected by water operators (in NL) and by U. Laval personnel (in QC). Samples were trained to follow equivalent sampling processes for both regions. Following field collection, samples were sent to the U. Laval laboratory for analysis.

Five families of DBPs were considered: THMs, HAAs and three families of non-regulated DBPs (HANs, HNMs, HKS). Four THMs (chloroform (TCM), bromodichloromethane (BDCM), dibromochloromethane (DBCM) and tribromomethane (TBM)), five HAAs (monochloroacetic acid (MCAA), monobromoacetic acid (MBAA), dichloroacetic acid (DCAA), trichloroacetic acid (TCAA) and dibromoacetic acid (DBAA)), four HANs (dichloroacetoniitre (DCAN), trichloroacetoniitre (TCAN), bromochloroacetoniitre (BCAN) and dibromoacetoniitre (DBAN)), one HNM (chloropiric (CPK)) and two HKS (1,1-dichloropropionate (DPC) and 1,1,1-trichloropropionate (TCP)) were analyzed during the study. The THM quantification limit was 3.6 μg L⁻¹ for TCM, 2.0 μg L⁻¹ for BDCM, 3.4 μg L⁻¹ for DBCM, 2.6 μg L⁻¹ for TBM. The HAA quantification limits were 1.0 μg L⁻¹ for all HAAs. The HAN, HNM and HK quantification limit were 0.01 μg L⁻¹ for all compounds. If the concentration of a DBP was lower than his LOQ, concentration was considered as null.

Many water characteristics were also measured. The sample strategy allowed us to obtain some 800 measures of each physical chemical parameter (turbidity, conductivity, UV absorbance at 254 nm (UV-254) and dissolved organic carbon (DOC), some 300 for bromide, 490 for pH, about 470 for temperature and about 1000 for free chlorine and each THM, HAA and non-regulated DBP under study. Bromide was analyzed by the MA.303-3.1 method (Centre d’expertise en analyse environnemental du Québec, 2009). Physico-chemical parameters were not sampled in every location because we only wanted to gain a general overview of water characteristics in the DS. Table 1 summarizes parameters measured at each sampling point. Details about analytical methods used are provided elsewhere (Mercier-Shanks et al., 2013).

2.3. Data analysis

Data were collected in a detailed Excel database for which all descriptive analyses were carried out. SYSTAT 13 Software Version No.13.1 was used for statistical analyses for this paper. Statistical analyses included Student’s t-test (for means comparison of water characteristics measurements between NL and QC) and ANOVA (for comparison of DBP mean levels between all systems in each region in order to detect a significant statistical difference in DBP levels between all systems belonging to a region), followed by a Games Howell test (for quantification of statistical differences between DBP levels in each system belonging to a region and identification of systems most or least correlated between each other).

3. Results and discussion

3.1. Portrait of DBP occurrence in small systems

Because there are regulations in QC that mandate water utilities supplied by surface waters to remove turbidity and NOM (mainly through filtration), the water submitted to chlorination (WTP) is of much higher quality in QC than in NL (data available in Supplementary section). The gap in levels of DBP precursor indicators (UV-254, DOC, SUVA and bromide) in water before disinfection explains why the DBP concentrations in small water systems (SWS) under study are significantly different between the two regions. In fact, the levels of these precursors have an impact on disinfectant demand and the potential for DBP formation. Table 2 presents the concentration distribution of all studied DBPs in SWS for both regions during the study period, as well as their 5% and 95% percentiles and coefficient of variation (CV).

It is important to note the gap between levels of regulated DBPs (THMs and HAAs) and non-regulated DBPs (HANs, HKS and CPK) in both regions. Both THM and HAA average concentrations were...
more than 10 times higher than each family of non-regulated DBPs. Such marked differences are observable in other studies. Regulated DBPs were 13 times higher in water from U.S. systems (Krasner et al., 2006) and 65 times higher in water from systems in Scotland (Goslan et al., 2009).

Table 3 resumes the DBP levels reported elsewhere in several studies dealing mainly with large and medium systems in China (Ye et al., 2009), Taiwan (Chang et al., 2010), Athens (Golfinopoulos et al., 2003), France (Mouly et al., 2010), the U.S. (Krasner et al., 2006) and Canada, in the greater Québec City area (Rodriguez et al., 2003). Results show that the occurrence of THMs and HAAs in SWS in NL or in QC was higher than those reported in larger systems (Table 3). Also, THMs and HAAs (except MBAA for which the level was basically lower than the quantification limit) presented significantly different concentrations in NL and QC. In fact, the annual average levels for THM and HAA were about three times higher in NL systems than in QC systems. In small systems of both provinces, the main THM observed was TCM and the two main HAAs observed were DCAA and TCAA. These compounds are also the compounds most observed in larger systems (for example, Golfinopoulos et al., 2003; Rodriguez et al., 2003; Ye et al., 2009; Chang et al., 2010). Also, the annual average level of TCAA was higher than the DCAA level in both regions.

Table 3
Levels of regulated and non-regulated DBPs observed in the literature.

<table>
<thead>
<tr>
<th>Compounds</th>
<th>Average THM level (µg L⁻¹)</th>
<th>Average HAA level (µg L⁻¹)</th>
<th>Average HAN level (µg L⁻¹)</th>
<th>Average CPK level (µg L⁻¹)</th>
<th>Average HK level (µg L⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average level in NL</td>
<td>125</td>
<td>129</td>
<td>3.1</td>
<td>0.5</td>
<td>9.4</td>
</tr>
<tr>
<td>Average level in QC</td>
<td>40</td>
<td>40</td>
<td>2.1</td>
<td>0.4</td>
<td>3.9</td>
</tr>
<tr>
<td>Chang et al. (2010)</td>
<td>14</td>
<td>8.3</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Goslan et al. (2009)</td>
<td>74</td>
<td>20</td>
<td>1.4</td>
<td>0.1</td>
<td>–</td>
</tr>
<tr>
<td>Ye et al. (2009)</td>
<td>12</td>
<td>7.4</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Krasner et al. (2006)</td>
<td>31</td>
<td>34</td>
<td>3.0</td>
<td>0.2</td>
<td>2.0</td>
</tr>
<tr>
<td>Rodriguez et al. (2004)</td>
<td>44</td>
<td>38</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Golfinopoulos et al. (2003)</td>
<td>22</td>
<td>19</td>
<td>0.1</td>
<td>0.1</td>
<td>0.5</td>
</tr>
</tbody>
</table>
Concerning the non-regulated DBPs, HANs, CPK and HKs also presented significantly different concentrations between the two regions (Table 2). The annual average levels for HAN and CPK were around 40% and 20% higher in NL than in QC, respectively. HAN levels in SWS of both regions were higher than the levels reported in other systems in Athens (Golfinopoulos et al., 2003) and Scotland (Goslan et al., 2009) and comparable to occurrences found in the U.S. (Krasner et al., 2006) (Table 3). CPK occurrence in both regions was higher than in Scotland (Goslan et al., 2009) and the U.S. (Krasner et al., 2006) (Table 3). Also, the annual average level for HK was two times higher in NL than in QC. The levels of regulated and non-regulated DBPs in SWS were higher than those observed in larger systems in Athens (Golfinopoulos et al., 2003) and others in the U.S. (Krasner et al., 2006) (Table 3). Finally, as shown in Table 2, the significant difference between brominated DBPs between SWS of the two regions, due mainly to BDCM, might be explained primarily by the higher levels of bromide in raw waters in NL than in QC.

In addition to the high inter-regional differences in DBP levels described above, there were also considerable intra-regional disparities between SWS within each region. In Table 2, these disparities are represented by the high values of CV for each family (all higher than 54%) and the 5% and 95% percentile values illustrating the large range of DBP levels. Moreover, statistical analyses on intra-regional differences in DBP levels based on an ANOVA, followed by a Games Howell test with SYSTAT were carried out (not presented in this paper) on all DBP measurements made in each system within a region. These analyses provided information on the statistical differences between DBP levels in each region and identified systems presenting comparable distributions of DBP levels within a region. Results showed that very few systems presented a comparable distribution of DBP levels in each region. Such differences were related to characteristics of water quality before disinfection.

These results suggest that the influence of water characteristics before chlorination on DBP levels in the DS is particularly meaningful in SWS supplied by surface waters. They may experience problems implementing treatment technologies to remove contaminants (DBP precursors in particular). The difficulty to remove DBP precursors may explain why the levels of regulated and non-regulated DBPs in small systems supplied by surface waters are higher than those observed in larger systems.

3.2. Temporal variations of DBP levels

Physical and chemical characteristics of raw water usually vary within a year, especially in NL and QC, where winters are long and very cold, and summers are comparatively short and hot (Rodriguez et al., 2003; Environment Canada, 2014). Consequently, seasonal variations of water temperature are considerable. And because seasonal changes occur very rapidly, there is sudden watershed runoff associated with snowmelt in the spring and relatively rapid decay of vegetation during the fall, a source of NOM in water (Rodriguez et al., 2003).

Figs. 1 and 2 illustrate the influence of seasonal climate variation on regulated and non-regulated DBP occurrence in SWS of NL and QC. In general, THM and HAA levels followed water temperature variations in both regions. Globally, levels decreased between the fall and winter (January–March), then increased until summer (July–September) to eventually reach a maximum in late summer. However this trend was less observable in NL than in QC and less observable for several DBPs, for example average HAA levels appeared highest in January in NL due to high levels in a few systems. Temporal variations of THM and HAA levels in each region were considerable. However, fluctuations (calculated by the ratio $(\max - \min)/\min \times 100$) appeared higher in QC than in NL. Average THM concentrations fluctuated by 90% in 13 months in QC and by 66% in NL. HAA concentrations mostly doubled in QC and fluctuated by 48% in NL. These temporal fluctuations were lower than those observed in a large system of the Québec City region (Rodriguez et al., 2004). In fact, using our formula, the temporal fluctuations of THMs and HAAs in Rodriguez’s study were respectively 980% and 770% for levels measured at the end of the system. Such differences in variations between both regions might...
be explained by higher variations of both water temperature and DBP precursor indicators in WTP water in QC than in RW in NL. In fact, average water temperatures varied from 6 °C to 20 °C in NL and from 2 °C to 19 °C in QC. Moreover, THM and HAA levels also followed DOC level variations in both regions (Figs. 1 and 2). In fact, average levels of DOC in water before chlorination varied during the sampling campaign from 5.5 mg L\(^{-1}\) to 8.8 mg L\(^{-1}\) (variation of 60%) in NL systems and from 1.5 mg L\(^{-1}\) to 3.4 mg L\(^{-1}\) in QC systems (variation of 130%). The higher variation in QC WTP waters might be explained, in part, by the diversity of treatment processes in place in QC. It may also be worth noting that the DCAA level was higher than the TCAA level in QC in winter (data not shown), as already observed in a previous study with a large system (Rodriguez et al., 2004). There could be different possible explanations. NOM characteristics of water may change over the year, as DCAA and TCAA have different precursors (Reckhow et al., 1985; Rodriguez et al., 2004). Also, the pH of the treated water was higher in winter (5.3–7.6, 5–95% percentiles) than in summer (5.0–7.5, 5–95% percentiles) in QC. In fact, TCAA formation in chlorinated waters is higher at lower pH, whereas DCAA formation is not as affected by pH (Stevens et al., 1989). However, as the difference in pH between seasons is low, it is unlikely that pH difference is the explanation. Finally, the possible biodegradation of DCAA (discussed later in this paper) could change the preponderance of these two HAAs in the DS.

Seasonal variations of HANs, CPK and HKs in SWS under study in both regions were also considerable (Fig. 2). Non-regulated DBPs followed the same temporal evolution as THMs and HAAs. During the sampling campaign, average HAN concentrations almost doubled in QC and almost tripled in NL from winter to summer. CPK concentrations almost doubled in QC and fluctuated by 63% in NL. Finally, HKs concentrations fluctuated by 86% in QC and by 73% in NL. Seasonal fluctuations were higher than those observed during a Canadian study that investigated DCAN (no fluctuation), CPK (no fluctuation), DCP (fluctuation of 25% in the system, using our formula) and TCP (fluctuation of 8% in the system, using our formula) in mostly medium and large systems (Williams et al., 1997).

### 3.3. Spatial variations of DBP occurrence within the distribution systems

Table 4 and Supplementary Fig. 1 present the portrait of the spatial variations for average concentrations of free residual chlorine, THMs and HAAs within the DS. In Supplementary Fig. 1, these variations were calculated using the ratio of DBP concentrations in DS1. On average, free chlorine levels decreased regularly along the DS in SWS of both regions. For QC systems (Table 4), THM concentration increased regularly along the DS. This type of evolution has also been observed in different studies with medium and large distribution systems (Chen and Weisel, 1998; Williams et al., 1998; Rodriguez and Sérodes, 2001; Mouly et al., 2010). In NL (Table 4) average THM concentrations increased and then stabilized at the end of the DS. HAA concentrations in QC and NL increased at the beginning of the DS and then decreased. This evolution pattern along the DS has also been observed in different studies (Rodriguez et al., 2004; Speight and Singer, 2005). This decrease was due mainly to the drop in DCAA concentration along the DS (annual average decrease of 12% in QC and 21% in NL between DS1 and DS3). This drop was particularly important in summer when the concentration of DCAA decreased by an average of 23% in QC and 40% in NL between DS1 and DS3. This profile is most likely due to the degradation of DCAA by biofilm (Bayless and Andrews, 2008). In fact, between DS1 and DS3, the degradation of DCAA might counterbalance and even exceed the potential formation of this compound along the DS when free chlorine is decreasing. This can occur even though the residual disinfection level is relatively high in DS1, especially in NL. Concerning non-regulated DBPs (Table 5 and Supplementary Fig. 2) in both regions, HAN average levels increased regularly along the DS.

The CPK average concentration stabilized at the end of the DS in NL, whereas it slowly decreased in QC (Table 5). Even if this result might suggest CPK degradation in QC exceeds the potential formation of this compound along the DS, as already observed in the literature (Lebel et al., 1997), its decrease is too low to clearly interpret a trend here, due to the low level measured.

HK levels evolved differently along the DS in NL and QC. In fact in NL, the average HK concentration decreased slowly between DS2 and DS3, whereas in QC, it increased. The difference was due to dissimilar variations of specific HK species, namely DCP and TCP (data not shown). In both regions, average levels of DCP decreased along the DS, whereas TCP levels increased. The main explanation is that DCP would be gradually oxidized into TCP (Bougeard et al., 2010; Mercier-Shanks et al., 2013), of which chloroform is one hydrolysis product (Yang et al., 2007). Since the average decrease of DCP in NL small systems was higher than in QC, DCP oxidized more easily into TCP due to a higher residual chlorine level (Table 4). TCP would be gradually hydrolyzed into chloroform and induce a global HK decrease. On the other hand, DCP oxidation in QC would be lower due to a lower residual disinfectant level than in NL (Table 4), and possibly a lower TCP hydrolyze which would induce a global HK increase.

### 3.4. Spatio-temporal variations of DBP occurrence

Values of standard deviations in Tables 4 and 5 suggest that spatial variations of DBP occurrence within a region varied seasonally and intra-regionally (from system to system) during the study period. Fig. 3 illustrates the seasonal and spatial evolution of free chlorine and regulated and non-regulated DBPs during summer and winter in NL and QC. Variations in both regions differed according to seasons. First, the drop in free chlorine in both regions is higher in summer than in winter, due to a higher water temperature and chlorine demand. Also, the decrease in HAAs is only observable in summer, under low chlorine conditions and mostly due to the DCAA decrease. It reinforces the hypothesis of a possible degradation of DCAA by biofilm in summer. This influence of season (temporal) and the location of the system (spatial) has already
been observed in a previous study in Canada (Rodriguez et al., 2004).

Finally, HK variations are different between summer and winter. In summer, HK level decreases and in winter it increases, especially in NL. This observation strengthens the idea that once the majority of the HK precursors have reacted, especially in summer when the temperature is high and in NL where free chlorine level is low, DCP would gradually be oxidized into TCP using

<table>
<thead>
<tr>
<th>Region</th>
<th>Location</th>
<th>HAN annual average (µg L⁻¹)</th>
<th>Standard deviation (µg L⁻¹)</th>
<th>CPK annual average (µg L⁻¹)</th>
<th>Standard deviation (µg L⁻¹)</th>
<th>HK annual average (µg L⁻¹)</th>
<th>Standard deviation (µg L⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>NL</td>
<td>DS1</td>
<td>2.7</td>
<td>1.8</td>
<td>0.5</td>
<td>0.3</td>
<td>9.3</td>
<td>5.2</td>
</tr>
<tr>
<td></td>
<td>DS2</td>
<td>3.1</td>
<td>2.1</td>
<td>0.5</td>
<td>0.3</td>
<td>9.5</td>
<td>4.9</td>
</tr>
<tr>
<td></td>
<td>DS3</td>
<td>3.4</td>
<td>2.4</td>
<td>0.5</td>
<td>0.4</td>
<td>9.2</td>
<td>5.0</td>
</tr>
<tr>
<td>QC</td>
<td>DS1</td>
<td>2.0</td>
<td>1.3</td>
<td>0.4</td>
<td>0.4</td>
<td>3.7</td>
<td>2.5</td>
</tr>
<tr>
<td></td>
<td>DS2</td>
<td>2.2</td>
<td>1.4</td>
<td>0.4</td>
<td>0.3</td>
<td>3.9</td>
<td>2.6</td>
</tr>
<tr>
<td></td>
<td>DS3</td>
<td>2.3</td>
<td>1.5</td>
<td>0.4</td>
<td>0.3</td>
<td>4.2</td>
<td>2.4</td>
</tr>
</tbody>
</table>

Fig. 3. Variation of average DBP occurrence along the DS in NL and in QC in: (a)–(f) summer (July–September) and (g)–(l): winter (January–March).

Table 5
Spatial variation of non-regulated DBP concentration in NL and QC.
residual free chlorine. This spatio-temporal evolution has already been observed in a previous study in Canada (Mercier-Shanks et al., 2013). At the same time, TCP would be hydrolyzed into chloroform in SWS, which decreased the total HK level and increased the level of THM at the end of the network. On the contrary, HAN and CPK variations are comparable between summer and winter.

Finally, all these observations are made regionally. In fact, standard deviations (represented by errors bars in Fig. 3) within regions are high for all seasons. In addition to temporal and spatial variations, it is important to take into account specific SWS characteristics which have a distinct influence on DBP occurrence. In fact, DBP variations are different, depending on site-specific distribution system conditions, especially for HAs, as DCAA degradation depends on biofilm presence in the SWS.

4. Conclusions

This study succeeded in enhancing knowledge of the spatio-temporal occurrence of regulated DBPs and HANs, HKs, and CPK in SWS. This knowledge is necessary to better appraise the particular characteristics of such systems regarding their high DBP levels in comparison to medium and larger systems (for which said issues have been documented extensively, particularly for regulated DBPs).

Also, potential biological degradation of DCAA, widely observed in large systems, appears surprising for SWS considering the size of the networks. It suggests that water residence times may be comparable, in some cases, to residence times in large systems (perhaps due to stagnation or to a poor hydraulic management).

Generally speaking, observations made in various selected SWS concurred with those made regionally. However, analyses of data for the SWS under study demonstrated that local characteristics of water quality, climate, treatment, operations and distribution systems make the occurrence of DBPs in SWS site-specific. Because of the limited human and technical resources of SWS, the variability of such local characteristics is higher than in medium and large systems. The results may prove useful to authorities for establishing investment priorities to improve SWS capacities to deliver safe water. They may also serve in epidemiological or risk analysis studies to better assess population exposure to potentially harmful DBPs.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.chemosphere.2014.08.002.

References


