Concentrations of disinfection by-products in swimming pool following modifications of the water treatment process: An exploratory study

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ABSTRACT

The formation and concentration of disinfection by-products (DBPs) in pool water and the ambient air vary according to the type of water treatment process used. This exploratory study was aimed at investigating the short-term impact of modifications of the water treatment process on traditional DBP levels (*e.g.*, trihalomethanes (THMs), chloramines) and emerging DBPs (*e.g.*, Halonitromethanes, Haloketones, NDMA) in swimming pool water and/or air. A sampling program was carried to understand the impact of the following changes made successively to the standard water treatment process: activation of ultraviolet (UV) photoreactor, halt of air stripping with continuation of air extraction from the buffer tank, halt of air stripping and suppression of air extraction from the buffer tank, suppression of the polyaluminium silicate sulfate (PASS) coagulant. UV caused a high increase of Halonitromethanes (8.4 fold), Haloketones (2.1 fold), and THMs in the water (1.7 fold) and, of THMs in the air (1.6 fold) and contributed to reducing the level of chloramines in the air (1.6 fold) and NDMA in the water (2.1 fold). The results highlight the positive impact of air stripping in reducing volatile contaminants. The PASS did not change the presence of DBPs, except for the THMs, which decrease slightly with the use of this coagulant. This study shows that modifications affecting the water treatment process can rapidly produce important and variable impacts on DBP levels in water and air and suggests that implementation of any water treatment process to reduce DBP levels should take into account the specific context of each swimming pool.

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Introduction

The evidence of the presence of contaminants of water and air at swimming pools by disinfection by-products (DBPs) resulting from water chlorination, such as chloramines (CAMs) trihalomethanes (THMs) and haloacetic acids (HAAs) have been the subject of several reviews (Jacobs et al., 2007; Teo et al., 2015; Chowdhury et al., 2014; Silva et al., 2012; Zwiener et al., 2007; Richardson et al., 2014; Manasfi et al., 2017). Their presence raises concerns about the respiratory effects on employees and bathers, and suspected carcinogenic and mutagenic potentials of these environments (Richardson et al., 2007; Plewa et al., 2008; Bougault et al., 2009; Liviac et al., 2010; Daiber et al., 2016; Hang et al., 2016). Even though there are several approaches to prevent the formation of these DBPs or their extraction from pool's air and water, only a limited number of studies have addressed the impact of those swimming pool water treatment processes on

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DBP contamination using a full-scale approach. Indeed, several reported studies consist of laboratory experiments, and explore the mechanistic aspect of the formation of certain groups of DBPs by looking at the chemistry of the precursors and the adjustment of various operational parameters (Glauner et al., 2005; Hansen et al., 2013a, 2013b; Soltermann et al., 2013; Weng et al., 2012). A number of other studies have compared the contamination profiles (speciation) and levels observed in various swimming pools according to water treatment type or various other characteristics (*e.g.*, disinfectant type) (Lee et al., 2010; Righi et al., 2014). It is uncommon to find case studies describing changes in contamination profiles and levels, under real conditions in a single swimming pool where the treatment processes are modified (Cassan et al., 2006). A relevant work with respect to the impact of the various procedures on the (types and levels) contamination portrait comes from France, where the issue of reducing CAM exposure, recognized as a source of occupational asthma in lifeguards, has led to numerous studies. In particular, they have investigated the implementation of stripping procedures and ultraviolet (UV) ray systems. Stripping procedures act through aeration to promote the extraction of CAM by volatilization (Gérardin et al., 1999, 2001, 2005b). The action of UV photoreactor, or dechloraminator, however, has not been as clear-cut as expected in terms of results and the advantages of these systems may be offset by an increased formation of THMs (Gérardin et al., 2005a). For instance, Hamel (2007) explored the impact of these means of reduction by enlarging the list of compounds studied to include CAMs and THMs in water and air, and enhanced the understanding of how operating conditions influence the formation of these DBPs. All these studies led ANSES (France's Agency for Food, Environmental and Occupational Health and Safety) to publish guidelines for authorization requests to implement treatment procedures, in particular, dechloraminators using UV, which must first undergo laboratory trials and experiments in a real environment (ANSES, 2015). Weng et al. (2012) and Hansen et al. (2013b) remark that the impact of UV rays on THMs is a source of contradiction in the literature. While the study by Cassan et al. (2006) reported an increase in levels of THMs following the installation of a UV system at a swimming pool, Beyer et al. (2004) observed the opposite effect at a swimming pool that had switched to a similar system and, at a third swimming pool, Kristensen et al. (2009), did not see any notable difference in THM levels whether the UV rays were on or off. Spiliotopulou et al. (2015) recently reported that DBPs are not formed in the UV reactor but during reactions that occur after chlorine addition and, that UV treatment followed by chlorination increased the formation of DBPs but that this impact was not observable on the long term with continuous UV treated water. The impact of UV rays continues to be the subject of laboratory investigations, providing an exclusive perspective on DBP levels in the water. These studies broaden the sample of DBPs studied and underscore the various consequences of operating conditions. Weng et al. (2012) documented the impact of UV rays on nitrogenous DBPs and concluded that UV rays appear to increase concentrations of some and decrease concentrations of others. Hansen et al. (2013b) assessed the impact of UV on 12 different compounds and considered that UV application could help lower levels of brominated haloacetonitriles (HANs) and brominated THMs. Using water samples from a distribution network,

Shah et al. (2011) highlighted the contrasting effects of different types of lamps on HANs and Halonitromethanes (HNMs). Soltermann et al. (2013) documented the effects of UV on *N*-nitrosamines, pointing out that the degradation of these compounds depends on the UV dosage applied and on the initial concentrations of these compounds and their precursors. They concluded that this type of swimming pool water treatment could lead to the formation of *N*-nitrosodimethylamine (NDMA) that counterbalances and surpasses the expected level of degradation. Finally, whereas Afifi and Blatchley (2016) reported that continuous use of UV (low pressure of medium pressure) over a year resulted in lower levels of DBPs, more recently Cheema et al. (2017) showed that UV treatment results in short-term increase of several DBPs.

In a previous study we showed that the occurrence and speciation of traditional and emerging DBPs measured in air and water of a group of 41 public swimming pools varied highly from one pool to another (Tardif et al., 2015b, 2016). In addition, levels were relatively high compared to current guidelines or reference values.

The management of a swimming pool involves the use of several systems and processes whose normal operations may be momentarily altered by maintenance procedures or failures. This exploratory study was the opportunity to provide data for an initial assessment of what could be the short-term impact of modifications to water decontamination process on the occurrence of DBPs in water and air at a typical swimming pool during the following hours/days.

Specifically, we investigated the variations in the levels of DBPs before and after modifications to the water treatment process at full-scale conditions and compare the impact of such modifications with respect to the levels measured after 24 hr and 6 days. The exploratory study was conducted under controlled operational conditions based on various field successive sampling campaigns carried out in different locations within the pool (from the drinking water supplying the facility to the pool air environment) and in the presence of bathers.

1. Methodology

The facility under study is an indoor swimming pool in the greater Montréal region (Canada), which was built in 1990. The volume of the swimming pool is 875 m³, with a 463 m² L-shaped basin of a maximum depth of 3.6 m. Water in the pool flows out through overflow gutters and skimmers and a drain in the bottom. The pool has not been completely drained since its renovation in 2010. Used air is exhausted through vents on the back wall near the diving boards. Fresh air is introduced near the skylights at deck level and at the top of the skylight. More air circulates from baffles located on the ceiling.

The water treatment process includes the following procedures: 1) air stripping which consists of removing the volatile compounds from the water through desorbing with air in the buffer tank (SPENCER UB0055BZ-002 air booster with flow rate of 7–8.5 m³/min, in operation from 4:00 p.m. to 7:00 a.m. during the week and from 12:00 p.m. to 7:00 a.m. on the weekend, with ventilation 24 hr a day in the buffer tank), 2) flocculation (450 mL of polyaluminium silicate sulfate (PASS) per day, through three 10-min injections of 10 mL/min and one 15-min injection);

flocculation causes agglomeration of colloidal particles through the addition of a polymer (coagulant), 3) filtration (two sand filters with a flow rate of about 192 m^3/hr). 4) UV rays dosing, applied prior to chlorine addition, through medium pressure lamps (dose set at 61 mJ/cm^2, in operation 24 hr a day); recirculation of water at a flow rate varying from 3168 to 3632 L/min. Sodium hypochlorite was used for disinfection and pH was controlled by addition of muriatic acid.

The concentrations of DBPs in water of the swimming pool and in the air around it were measured in order to successively and systematically address the impact of the following four modifications (T1–T4) with respect to the baseline conditions (UV lamps off, air stripping plus air extraction, flocculation), as follows, T1: activation of UV photoreactor; T2: halt of air stripping with continuation of air extraction from the buffer tank; T3: halt of air stripping and suppression of air extraction from the buffer tank; T4: suppression of the polyaluminium silicate sulfate (PASS) coagulant. The filtration and chlorination processes were not modified. The general ventilation conditions remained the same throughout each campaign, i.e., the addition of 136 m^3 per minute of fresh outside air and the exhausting of 277 m^3 per minute of indoor air.

1.1. Water and air contaminants

Traditional DBPs [CAMs (monochloramine (MCAM), dichloramine (DCAM), trichloramine (TCAM)); THMs (chloroform (TCM), chlorodibromomethane (CDBM), dichlorobromomethane (DCBM), tribromomethane (TBM)), HAAs (monochloroacetic acid (MCAA), Monobromoacetic acid (MBAA), dichloroacetic acid (DCAA), trichloroacetic acid (TCAA), bromochloroacetic acid (BCAA), Dibromoacetic acid (DBAA))] and emerging DBPs [[HANs (trichloroacetonitrile (TCAN), dichloroacetonitrile (DCAN), dibromoacetonitrile (DBAN), bromochloroacetonitrile (BCAN)), HNMs (chloropicrin (CPK)); haloketones (HKs) (11DCPone, 111TCPone), NDMA] were analyzed at the laboratory of the Research Chair in Drinking Water of Université Laval in Québec City, whereas the levels of THMs in air were measured at the laboratory at the *Unité d'inhalation expérimentale* of the Department of Environmental and Occupational Health of the Université de Montréal. The analyses of the CAMs in air samples were subcontracted to the laboratory of the Environmental Service of the City of Montréal. Details pertaining to the sampling and analytical methods used are described in recent publications and reports (Tardif et al., 2015b, 2016). The method used for CAMs in air was based on the one developed by Héry et al. (1994) and it consists of pumping air at a rate of approximately 1 L/min for 120 min through a device consisting of a Teflon filter that captured the particulate pollution (droplets of chlorinated compounds) that could interfere with the dosage, and then through two cellulose filters impregnated with sodium carbonate and diarsenic trioxide. These two filters were desorbed with doubly distilled water. After percolation over an ion exchange resin, the desorbate was analyzed using ion chromatography.

In addition, the pH, temperature, residual chlorine, free chlorine and monochloramine were measured in pool water through direct readings onsite. Details pertaining to the methods are described in a recent report (Tardif et al., 2015b).

1.2. Design of the sampling campaign

The sampling campaign was carried out in two phases (fall of 2013 and winter of 2014) (Table 1). The first sampling took place on Monday morning to get the baseline configuration, before modifying the treatment process the next day (Tuesday). Samples were collected on Wednesday, 24 hr after modification. The team returned the following Monday to pick up samples, seven days after modification. The return to the baseline situation (normal conditions) took place after sampling on that Monday. The process was repeated at least 7 days later (Table 1), in order to study a new configuration and to re-establish a new baseline involving the deactivation of the UV photoreactor. Two days of sampling were added (January 21 and March 12) to measure the levels of DBPs in the treatment configuration that would have normally been in operation if the interventions due to the project had not taken place, i.e., with all the treatment devices in operation (situation equivalent to the baseline situation with the UV rays activated). These two samplings would also take into account the "normal" contamination of the site during the week before starting the series of programmed modifications and one week after the reestablishment of the normal water treatment process used throughout the winter. Each sampling visit lasted approximately 3 hr and took place during the morning.

A series of major problems were noted by the maintenance crew two days before the last sampling (March 12): there was no air extraction from the buffering tank because of a breakdown in part of the ventilation duct, which caused the contaminants to outgas into the pool area while air stripping continued; there was no supply of PASS (empty barrel); free chlorine was absent because of a problem with the pumps, without anyone knowing

Table 1 – Calendar of the sampling campaign [*].			
	Situation		Sampling campaign
T1	Activation of UV rays	Before	November 4 (115) [**]
		After	November 6 and 12 (81–97)
T2	Deactivation of stripping	Before	February 24 (101)
		After	February 26 and March 4 (84–109)
T3	Deactivation of stripping and of ventilation in the buffer tank	Before	February 10 (97)
		After	February 12 and 18 (80–72)
T4	Flocculation halted	Before	January 27 (82)
		After	January 29 (122)
NORMAL1	Normal (complete treatment)		January 21 (104)
NORMAL2	System failure		March 12 (88)

[*] For detailed description of the modifications and sampling procedure see Section 2.2.
[**] Number of bathers (Mean ± SD): 95 ± 22.

when the outage began and how long it lasted. This situation gave us the opportunity, particularly relevant in the scope of the study, to document the repercussions on DBP contamination in the "worst-case" scenario (a breakdown of almost the entire water treatment process).

In the water, the traditional DBPs (*i.e.*, THMs, HAAs) were systematically measured at two different points in the basin (A, B), in the middle of the visit (Fig. 1). A few minutes before that, another sample for each of the two compounds was collected from a specially installed faucet at the end of the treatment process, just before the water inlet into the pool. During the days after modifications were made to the water treatment process, two samples (for measuring each compound) were taken by the crew responsible for pool maintenance at the same point, before the facility opened. A sample was collected at the faucet to measure the concentration of traditional DBPs in the drinking water distribution network supplying the swimming pool. With respect to emerging DBPs (eDBPs: HANs, HNMs, HKs, and NDMA), a single sample was taken in the basin, below the lifeguard chair corresponding to point A, in the middle of the visit (Fig. 1).

During each morning of sampling, the CAMs and THMs in the air were measured simultaneously at three points (A, B, C) around the pool, by taking samples of air at the breathing height of a man standing beside the pool (Fig. 1). To do this, the pumps were positioned by the lifeguard chairs at approximately 150 cm above the surface of the water. A more detailed description of the methodology, including analytical methods, is available in a recent report (Tardif et al., 2015a).

1.3. Statistical analyses

SAS (SAS Institute Inc., 2009) software was used to verify the normality of data distributions with the Shapiro–Wilk test and then proceeding, after controlling for the equality of variables,

with a *t* test on two independent samples to compare the concentrations of THMs and CAMs measured at point C and those measured on average at points A and B. The threshold of statistical significance was set at 0.1.

2. Results

Average values for the physico-chemical parameters are the following: temperature (28.4 ± 0.9 °C), free chlorine (1.4 ± 0.4 mg/L), total chlorine (2.1 ± 0.5 mg/L), monochloramine (0.21 ± 0.07 mg/L), pH (7.61 ± 0.15) For the 10 visits the average number of bathers was 95 ± 22. Currently, there are no guidelines or standards in Québec or Canada for DBPs in water or air for indoor swimming pools. In Quebec the recommendation for pH, chloramine and free chlorine in swimming pool water are 7.2–7.8, ≤0.5 mg/L and 0.8–2.0 mg/L, respectively.

With respect to DBPs, the results showed typical contamination of predominantly chlorinated DBPs. TCM represents more than 96% of total THMs (37 ± 9 µg/L), TCAA and DCAA representing around 70% and 26% of total HAAs (248 ± 36 µg/L), and DCAN represents 93% of HANs, respectively in water. Similarly, TCM represents 96% of total THMs (241 ± 9 µg/m³) in air. Given this invariable profile of speciation, the following sections only take into account the results for the total quantity of each class of DBP, without detailing the specific results by compound for each one.

2.1. Impact of treatment modifications on DBPs in the air

2.1.1. CAMs
Fig. 2 presents the average levels of CAMs (mainly TCAM) measured in the air around the pool before and after each modification to the treatment process. A reduction in levels of
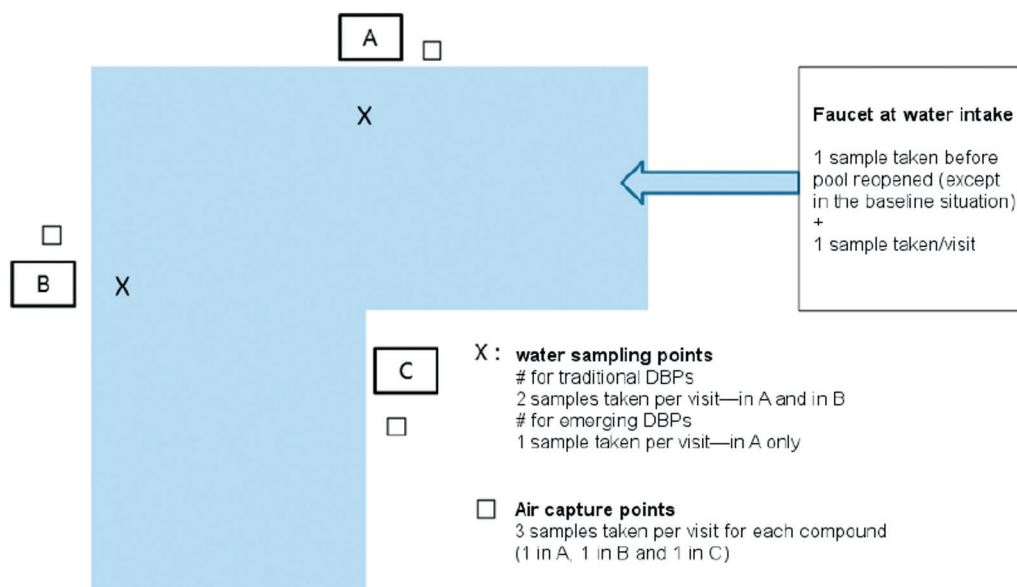

Fig. 1 – Diagram of sampling plan around the swimming pool. Water: for traditional disinfection by-products (DBPs), 1 samples were collected in points A and B in the middle of the visit (n = 2). For emerging DBPs, a single sample was taken in A only in the middle of the visit. Air: 3 samples were taken in points A, B and C during the visit (sampling duration: chloramines (CAMs): 120 min), trihalomethanes (THMs) (95 min).

CAMs (1.2–1.6 fold) was observed in the air when the UV photoreactor was operating (T1). Interestingly, a similar decrease was observed in the levels of combined chloride in water (1.4–1.7 fold). On the contrary, the suppression of stripping produced important increases in CAM levels after 6 days ((T2): 1.7 fold; (T3): 1.4 fold). Whether PASS (T4) is used or not does not appear to have any effect.

2.1.2. THMs

Fig. 3 presents the average levels of THMs measured in the air around the pool before and after each modification to the treatment process. Contrary to the case for CAMs, activation of UV rays (T1) led to an increase in THM levels in the air (1.3 fold) after 6 days. On the other hand, as is the case for CAMs, maintaining air extraction from the buffer tank appears to delay the increase in levels of contaminants in the air around the main swimming basin when air stripping is halted ((T2): 2.2 fold; (T3): 2.0 fold). Suppression of PASS (T4) appears to result in a slight decrease in levels of THM contamination in the air.

2.2. Impact of treatment modifications on DBPs in the water

2.2.1. Traditional DBPs

As shown in Fig. 4 activation of UV photoreactor (T1) produced high levels of THMs after 6 days (up to 1.7 fold) close to what was observed in air. Likewise, suppression of stripping also resulted in increased levels of THMs ((T2): 1.24 fold; (T3): 1.4 fold) whereas suppression of PASS (T4) produced reduced levels (1.2 fold) after 24 hr.

2.2.1.2. HAAs. Overall, no clear impacts of the modifications on the average levels of HAAs were observed (Fig. 5). Indeed, UV rays (T1) as well as suppression of stripping or PASS had no noteworthy effect on HAA levels.

2.2.2. Emerging DBPs

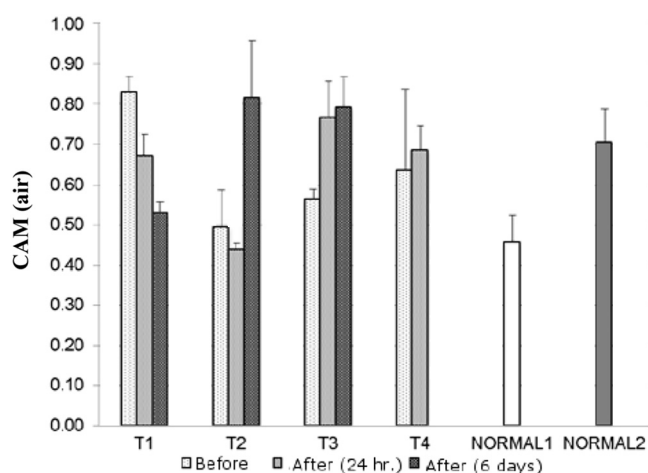Table 2 presents the average levels of HANs (mainly DCAN) measured in the pool water before and after each modification

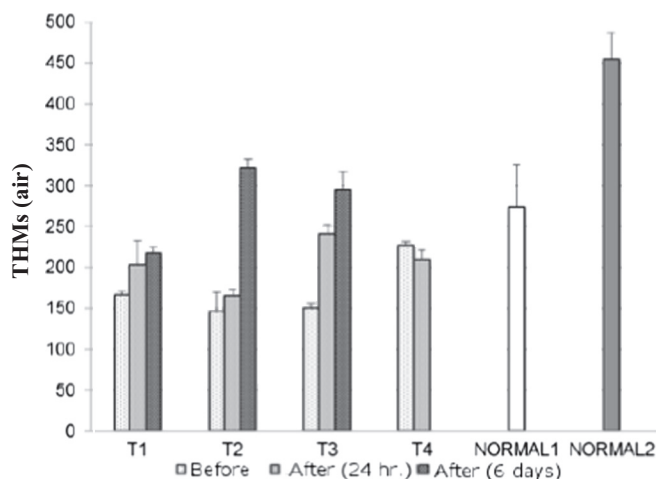**Fig. 3 – Concentrations of THMs ($\mu g/m^3$) in the air around the pool before and after each modification to the water treatment process (n = 2, ± Standard Deviation (SD)).**

to the process during the two campaigns. Whereas UV (T1) increased levels (1.4 fold) after 24 hr, halt of stripping and air extraction (T3) produced lower levels (0.7 fold).

The average levels of HNMs, which consist exclusively of CPK, measured in the pool water, before and after each modification to the treatment process during the two campaigns were importantly increased after UV activation (8.4 fold). Other modifications to treatment have no effect (Table 2).

As with the HNMs, UV rays caused (2.1 fold) a marked increase in levels of the only HK detected (111 TCPone) while the other treatment processes do not appear to have notable effects on those levels.

With respect to NDMA, UV rays appear to play a significant role in lowering levels (from 2.1 fold to complete elimination after 6 days). On the contrary, the absence of stripping appears to increase concentrations of this compound in the water ((T2): 2.13 fold; (T3): 1.44 fold). Finally halting the flocculation process resulted in higher levels after 24 hr.
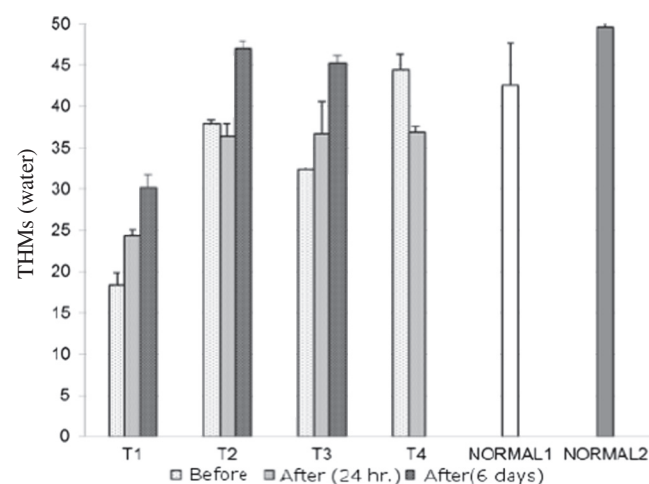

Fig. 2 – Concentrations of CAMs (mg/m³) in the air around the pool before and after each modification to the water treatment process (n = 3, ±Standard Deviation (SD)).

Fig. 4 – Concentrations of THMs ($\mu g/L$) in the pool water before and after each modification to the water treatment process (n = 4, ± Standard Deviation (SD)).
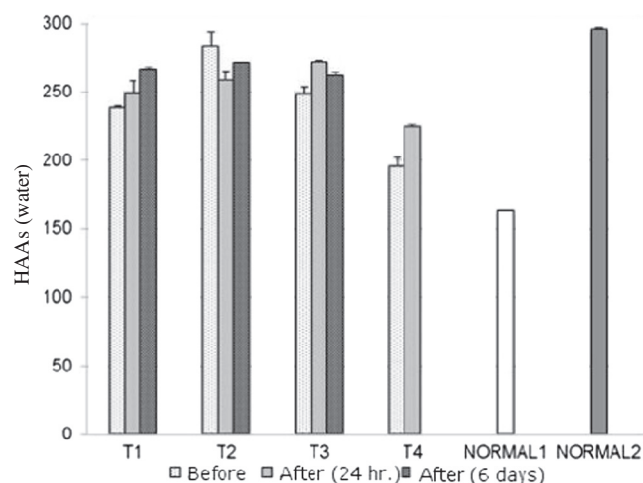
Fig. 5 – Concentrations of haloacetic acids (HAAs) (μg/L) in the pool water before and after each modification to the water treatment process (n = 2, ±Standard Deviation (SD)).

2.3. Spatial and temporal variations on contamination levels

2.3.1. Contamination of water at the source (water distribution system) versus contamination of water in the pool

Water samples were collected at one of the faucets at the facility to compare the THM and HAA levels originally present in the drinking water in the distribution system supplying the facility with those measured in the swimming pool water. The results clearly showed higher levels of these contaminants in the swimming pool. In the water from the distribution system, we found constant concentrations of HAAs throughout the duration of the campaign, averaging 16 (±2.7 μg/L) with no differences in levels between fall and winter. The highest levels in the pool water reached values almost 20 times higher than those in the water distribution system. THM levels in the drinking water distribution system were 18.5 (±4.2 μg/L) on average. There was, however, a difference between the average level in samples taken in the fall during that campaign (12.4 ± 0.6 μg/L, $n = 3$) and those taken during the winter (20.3 ± 2.8 μg/L, $n = 10$) (1.63 fold increase). The THM levels are higher and more variable in the swimming pool water than in the distribution system water, with an average concentration of 37.0 (±9 μg/L). As well, in the swimming pool, almost similar difference is noted this time between the levels measured during the fall and those measured

during the winter (24.3 ± 5.4 μg/L and 40.9 ± 5.7 μg/L) (1.68 fold increase). Overall, for both fall and winter, results show that the levels of THMs in the pool water were 2 fold higher compared to levels measured in the distribution system.

2.3.2. Contamination of water before it enters the pool (after treatment) versus contamination of water in the pool

Water samples ($n = 31$) were taken at a water intake (point P) in the mechanical room, installed between the end of the water treatment process and the outlet into the pool, to compare the levels of THMs and HAAs measured there with those measured in the pool. Out of all of these samples, 10 (corresponding to the 10 visits) were taken on different days, before the pool opened to bathers. The other samples were taken during each of our visits, with bathers in the pool. Fig. 6 presents the correlations between levels of THM and HAA measured at point P before the pool opened for each visit ($n = 10$), with no bathers, and those measured a few hours later with bathers present. Comparable levels between the two samples were observed, with ratios between the second measurement and the first of between 0.86 and 1.12 for THMs, and 0.84 and 1.05 for HAAs. Fig. 7 presents the correlations between the levels of THMs and HAAs measured at point P and those measured at the same time in the pool after it opened, with bathers present. Here again, there is no major difference between the measurements taken at these two sampling points, with ratios between measurement at the basin and measurement at point P alternating between 0.94 and 1.3 for the THMs and 0.93 and 1.17 for the HAAs. Overall, the results show that the water treatment process for this pool does not much change the levels of DBPs present in the water returning to the pool compare to the levels in the pools, no matter the presence or not of bathers at least after the opening of the pool.

2.4. Spatial variations of DBPs between sampling points

In the water, no major difference is observed between the THM and HAA measurements taken between the two sampling points, A and B (Fig. 1). In the air, THM levels at sampling point C tend to be higher than the averages of levels measured at points A and B (in 15 points of comparison out of 19), but this result is not statistically significant ($p = 0.7$). It is the same for measurements of CAM (21 points of comparison), but with a statistically significant difference of almost 0.10 μg/m^3 ($p = 0.09$), and a 17% increase, on average. The proximity of sampling point C to the

Emerging DBPs	T1[c]			T2			T3			T4		Normal1[d]	Normal2
	t0[b]	t1	t2	t0	t1	t2	t0	t1	t2	t0	t1		
HKs (μg/L)	3.1	6.0	6.5	2.5	2.3	2.3	1.9	1.9	2.2	1.8	1.9	3.4	4.7
Halonitromethanes (HNMs) (μg/L)	0.17	1.3	1.4	0.22	0.20	0.22	0.16	0.20	0.22	0.17	0.15	0.94	1.42
HANs (μg/L)	24.8	35	30.1	24.2	24.9	25.4	34.3	22.7	23.7	21.5	21.2	23.4	26.7
NDMA (ng/L)	12.8	6	<1.6	11.9	25.4	15.1	10.5	9.8	15.2	<1.6	3.7	4.6	<1.6

Table 2 – Concentrations [a] of emergent DBPs in the pool water before and after each modification to the treatment process.

DBPs: disinfection by-products; HKs: haloketones; HANs: haloacetonitriles; NDMA: N-nitrosodimethylamine.

[a] Concentrations corresponding to a single sample collected in the middle of the visit.

[b] t0: before; t1: after 24 hr; t2: after 6 days.

[c] T1, T2, T3 and T4: modifications (see text).

[d] Normal1: normal operation (all systems in function)/Normal2: system failure.

buffer tank may explain the greater contamination level in that part of the pool area.

3. Discussion

This study afforded an exceptional opportunity to gather data on the impact of various modifications in the water treatment process of a typical swimming pool on water and air contamination by traditional and emerging DBPs, under full-scale controlled operational conditions. Being able to investigate all these contexts within a single swimming pool facility minimized the variable environmental conditions that exist from one pool to another, which were broadly outlined in a previous research (Tardif et al., 2015b, 2016). We believe that data such as what we gathered, using an experimental systematic approach, at full-scale conditions, and covering a wide range of compounds are undeniably relevant in considering the highly desirable and strongly supported goal of reducing exposure levels to DBPs in swimming pools.

3.1. Considerations related to levels of contamination

It is important to note that the type of contamination found in the swimming pool studied in the scope of this project is typical in terms of species present and levels, in that the predominant DBPs are chlorinated compounds (and not brominated compounds). TCM thus makes up most of the THMs measured, and TCAA (and to a much lesser extent, DCAA) makes up most of the HAAs, as has often been reported in the literature

(a)

(b)

Fig. 6 – Correlations (n = 10) between THMs (a) and HAAs Acides haloacétiques (AHA) (b) levels measured at the water intake (point P) before the pool opened (without bathers) and a few hours later (with bathers present).

(Bessonneau et al., 2011; Teo et al., 2015; Cimetiere and De Laat, 2014; Catto et al., 2012; Simard et al., 2013). The results confirm a significant increase in contamination levels in the pool water compared to levels at the source (up to 20 times more HAAs and two times more THMs). The substantial accumulation of non-volatile HAAs in the water and, conversely, the high volatility of THMs explain this difference (Kanan and Karanfil, 2011).

With respect to CAMs in the air, all levels measured are above or equal to the value of 0.3 mg/m^3 recommended by Parrat et al. (2012) and ANSES (2012) to minimize the health impact of these compounds. The concentrations of THMs in the water are either higher or lower than the various reference values or guidelines in force internationally (e.g., 20 μg/L in Germany, 30 μg/L in Switzerland, 100 μg/L in the United Kingdom, Finland and Denmark). Finally, it is noteworthy that out of 21 samples of NDMA, although none exceeded the maximum concentration level acceptable for drinking water recommended by Health Canada (40 ng/L), and the World Health Organization (100 ng/L), 12 exceed the standard in force for drinking water in Ontario, Canada (9 ng/L) (Soltermann et al., 2013).

3.2. Considerations on the impact of the modifications in the water treatment process

In light of the overall results, it appears, unsurprisingly, that the modifications have different effects according to the type of DBP compounds. This suggests that the choice of a water treatment process for a facility should be made by taking into account the goal pursued in terms of decontamination of water and air (e.g., reducing the levels in general or targeting some DBPs in particular). UV may have a major impact on the DBP levels and is still difficult to predict the duration and extent of this impact. They produce increases in eDBPs, in particular, HNMs and HKs, as well as THMs, in the water. The increases of the latter, which are known to be quite volatile, have subsequent repercussions on THM levels in the air. On the other hand, the activation of the UV photoreactor, in this case, contributes to sharply decreasing the amount of CAM in the air and NDMA in the water. The positive impact of UV on CAMs, and the opposite effect they appear to have in producing higher levels of THMs, have already been demonstrated in previous investigations (Cassan et al., 2006; Gérardin et al., 2005a: Cimetiere and De Laat, 2014). With respect to NDMA, our results should be compared with those of Soltermann et al. (2013), who reported that UV rays may cause an increase in levels of NDMA in swimming pool water, since our study shows a decrease of this contaminant as previously reported (Walse and Mitch, 2008; Krasner et al., 2013). As expected, the results underscore the positive impact of air stripping on reducing volatile contaminants in the air, while, unsurprisingly, halting mechanical mixing leads to increased levels of contaminants in the ambient air. The comparison of treatment modifications T2 and T3 (with and without air extraction in the buffer tank) reveals the buffering effect that the ventilation system could have in the event of failure of the stripping device, by, quite logically, temporarily slowing down an increase of air contamination by DBPs. The use of PASS does not appear to have an effect on the presence (quantity and speciation) of DBPs. However there was a rapid deterioration of water clarity in the hours and days following the suspension of
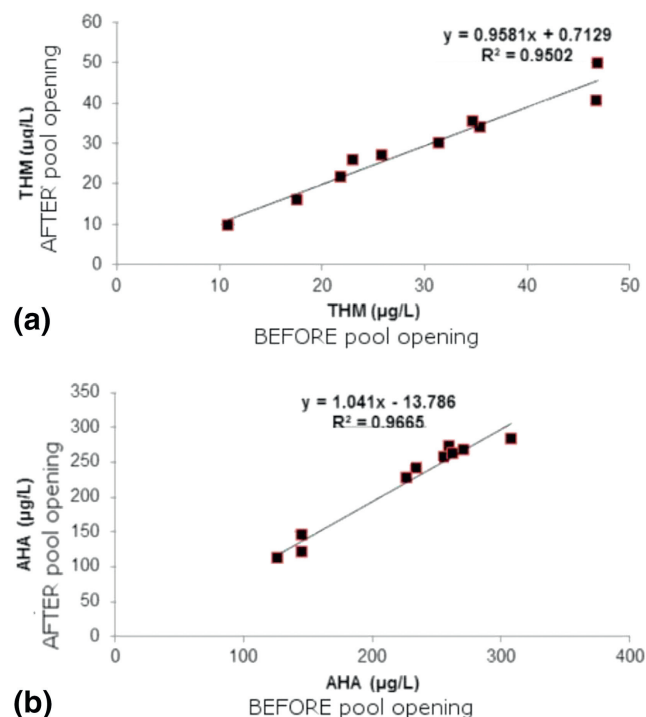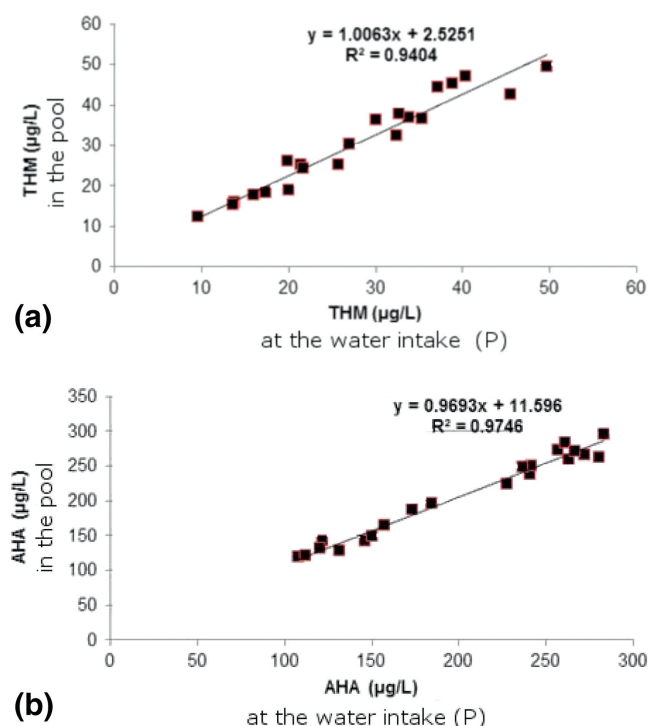
(a)

$y = 1.0063x + 2.5251$
$R^2 = 0.9404$

(b)

Fig. 7 – Correlations (*n* = 21) between THMs (a) and HAAs Acides haloacétiques (AHA) (b) levels measured at the water intake (point P) and those measured at the same time in the pool (average of the two sampling points). Levels measured in the presence of bathers.

PASS injection, and even after it was restarted, we were unable to collect all the data we needed (notably, after one week without flocculation).

4. Conclusions

Based on an original experimental design, and tested under real conditions at full-scale, this project generated new and useful data to better understand the dynamic of DBP contamination in swimming pool water. This case study revealed contamination by potentially high levels of CAMs in the air, and to a lesser extent, of NDMA in the water. Our current knowledge indicates that these two contaminants, in particular, may potentially expose workers and bathers to health risks. In addition to UV, air stripping appears to have a positive effect on reducing DBP levels of CAMs (air) and NDMA (water). The isolated breakdown of one of the devices in the treatment system and the additional breakdowns of several others had an almost immediate impact.

More generally, the investigation results will also raise stakeholders' awareness of the need to make choices about treatment processes that take into account the specific context of each swimming pool and the related issues, in this case, contamination by DBPs. Of course, as this was an exploratory study, the results cannot be used to validate (or invalidate) the various processes examined here in other contexts. In fact, in the specific context of the swimming pool

under study, the assessment of these impacts should be explored further with a more comprehensive design involving multiple samplings carried out over a longer period of time and additional data in order to draw firmer conclusions. Studies of this type are undoubtedly extremely relevant with respect to the issue of DBP exposure and are to be recommended. Finally, it should be kept in mind that any changes in DBP speciation and/or levels in air or in water of a swimming pool resulting from such modifications are likely to affect exposure of bathers and workers and therefore the importance of the health risks associated with such DBP exposure.

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