

Application of a novel polarity method for the characterization of natural organic matter during water treatment

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ABSTRACT

The changes in polarity of natural organic matter (NOM) across a pilot-scale conventional water treatment plant with pre-ozonation and biofiltration were analyzed using the polarity rapid assessment method (PRAM). PRAM is a novel method developed for the characterization of the polarity of NOM in environmental samples. Characterization was accomplished by monitoring UV absorbance (UVA) at 254 nm. Under ambient conditions, the NOM was characterized as highly anionic (>50%), with limited hydrophobic (10–25%) and hydrophilic (0–10%) domains. Results showed temporal variations in the polarity of NOM entering the pilot plant. These differences were ascribed to changes in water blend and variability in NOM characteristics. Changes in NOM chemistry were also observed as the organics passed through the treatment process. Ozonation decreased the hydrophobicity and increased the polarity of NOM, confirming results from previous research utilizing resin fractionation methods. Coagulation, flocculation and biofiltration resulted in the decrease of the hydrophobic and hydrophilic character of the chromophoric NOM. However, these polarity changes varied during the pilot plant run, suggesting a more precise evaluation of NOM through unit operations is warranted.

Key words | natural organic matter (NOM), NOM characterization, ozonation, polarity rapid assessment method (PRAM)

INTRODUCTION

The accurate characterization of the properties of natural organic matter (NOM) in environmental samples continues to be a challenge. The stochastic nature associated with NOM formation and evolution, in addition to its complex chemical nature, complicates the analysis, inhibiting the creation of a single model that could be used to describe and predict its properties. At present, three models detailing the possible structure and associations of NOM in water are being debated (Ghosh & Schnitzer 1980; Wershaw 1999; Simpson *et al.* 2002; Sutton & Sposito 2005); however, no general conclusion has been reached.

NOM plays an important role in water treatment. For example, NOM serves as the main substrate for the formation of disinfection by-products (DBPs) (Singer 1999; Zhang & Minear 2002) and is one of the main components

associated with membrane fouling (Amy & Cho 1999; Taniguchi *et al.* 2003; Lee *et al.* 2004). Furthermore, NOM interferes with adsorption processes, reducing the capacity towards removal of chemicals of interest (Li *et al.* 2003). As a result, the transformations of NOM through water treatment processes have been of keen interest (Westerhoff *et al.* 1999; Marhaba & Van 2000; Marhaba *et al.* 2000; Buchanan *et al.* 2005; Sohn *et al.* 2007), especially when they are related to the formation of carcinogenic DBPs (Hwang 2001; Krasner *et al.* 2006; Yang *et al.* 2007).

Treatment processes that maximize NOM removal are of special interest even though complete removal is not attainable. In most cases, NOM is chemically and physically modified by unit operations (e.g. coagulation, filtration and oxidation) to minimize its effects. The specific

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physicochemical change depends on which treatment is applied. For example, ozonation increases NOM polarity by transforming hydrophobic compounds into hydrophilic compounds (Becker & O'Melia 1996; Westerhoff *et al.* 1999; Marhaba *et al.* 2000) and reduces its average molecular weight (Becker & O'Melia 1996; Vuorio *et al.* 1998; Hesse *et al.* 1999). Coagulation removes the high molecular weight fraction and is more effective with high specific ultraviolet absorbance (SUVA) waters (Edzwald 1993).

Advanced analytical techniques such as resin fractionation and mass spectrometry are commonly used to characterize NOM (Kitis *et al.* 2002; Leenheer & Croué 2003). These methods require extensive preparative steps to obtain solid phase extracts and limit the analysis to a select number of samples. However, these techniques are not available to the majority of utilities. This results in the need to use non-specific bulk parameters such as total organic carbon (TOC) and ultraviolet absorbance (UVA) to derive properties that can be used to study the effects of NOM during treatment processes.

From a utility's perspective, methods that could easily characterize the polarity of NOM are desirable, especially in order to predict reactivity. Current methods are time-consuming, reducing their applicability to routine monitoring. Recently, an alternative approach to the characterization of the polarity of NOM was presented (Rosario-Ortiz *et al.* 2007b). The polarity rapid assessment method (PRAM) permits NOM characterization under ambient conditions in less than one hour, allowing for the timely analysis of multiple samples. This approach enables researchers to evaluate the temporal variability of NOM characteristics and the effect of this variability on treatment systems.

The objective of this study was to present the applicability of using PRAM to understand polarity changes of NOM that occur during water treatment. For this, a pilot system simulating conventional treatment with pre-ozonation and biofiltration was studied. The pilot system had two different source water sources which varied in blend ratios, which also allowed the evaluation of changes in blend on the polarity of NOM. This study emphasizes the characterization under ambient conditions and the study of NOM polarity using this simple methodology.

EXPERIMENTAL

Pilot testing

A dual-train conventional treatment plant (rapid mix/flocculation/sedimentation) with pre-ozonation and biological filtration (O₃/BF) was used in this study. Figure 1 presents a diagram of the process. The system had a flow rate of 17 L/min (4.5 gal/min). The source water for the pilot plant was a mixture of Colorado River Water (CRW) and California State Project Water (SPW).

The water was ozonated before the rapid mix basin by an air-fed ozone generator (Trailgaz Labo, Trailgaz Ozone of America, Inc., Jenkintown, PA, USA). Ozone concentrations in the feed gas stream were measured with an ozone monitor (Model HC-12, PCI Ozone Corp., West Caldwell, NJ, USA) and recorded once per day. These measurements were combined with gas flow measurements to calculate the daily-applied ozone dose. Except for two days when the ozone generator was offline and one day when the dose was 1.7 mg/L, the applied ozone dose ranged from 1.1–1.3 mg/L, with an average of 1.2 mg/L and a standard deviation of 0.1 mg/L.

Chemical addition at the rapid mix included 2.0 mg/L polyaluminum chloride (PACL) (Sumalchlor 50, Summit Research Labs, Flemington, NJ, USA) and 2.0 mg/L cationic polymer (polydimethyldiallyl ammonium chloride, NS 3150, Neo Solutions, Inc., Beaver, PA, USA). The dual-media filters included 20 in. anthracite coal and 8 in. sand and were backwashed based on the following criteria: effluent turbidity (Hach 2100N Turbidimeter, Hach Company, Loveland, CO, USA) exceeding 0.3 nephelometric turbidity units, headloss greater than 1.8 m (6.0 ft) or filter runs longer than 24 h. The filtration rate was 2.3 L/min/m² (6.6 gal/min/ft²) and chlorine data were measured at the filter effluent (Hach Company CL-17 chlorine analyzer, Loveland, CO, USA). Chlorine and ammonium sulfate were added to the filter effluent to maintain a chloramine concentration of 1.5–2.5 mg/L to simulate full-scale disinfectant doses.

During testing, samples were collected from the pilot-plant influent (raw water), ozone effluent and filter effluent after chloramine addition (see Figure 1). Water quality samples were collected during three sampling events in November and December, 2003. These samples were

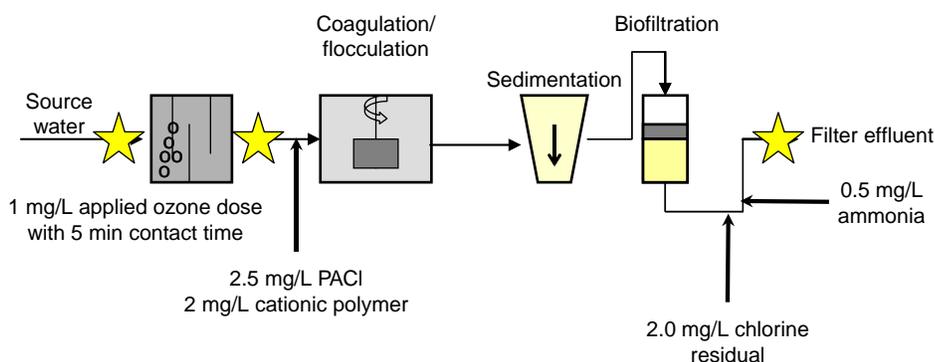


Figure 1 | O_3 /BF process diagram during pilot testing. The stars represent sampling locations. Grab samples were collected on three dates during the fall of 2003.

characterized according to *Standard Methods for the Examination of Water and Wastewater* (APHA *et al.* 1998).

Polarity characterization by PRAM

The specifics of PRAM have been described elsewhere (Rosario-Ortiz *et al.* 2007a,b). Briefly, the solid-phase extraction (SPE) cartridges used during this study (Extract Clean SPE kit catalog #210100, Alltech Associates, Deerfield, IL, USA) included non-polar (C18 and C2), polar (CN, diol and silica) and anionic exchangers (NH₂, SAX). Table 1 presents the SPE sorbents used and their respective molecular structures. The SPE cartridges were cleaned by flushing with Milli-Q to minimize interference with UVA at 254 nm. Water samples were run in duplicate through each cartridge and the UVA at 254 nm was obtained versus time. Light absorbance was measured using an ultraviolet (UV) spectrophotometer (Series 634 UV-Vis Spectrophotometer, Varian Instruments, Walnut Creek, CA, USA) with a 1 cm

path length micro-flow cell (Fisher Scientific, Pittsburgh, PA, USA). Flow through each SPE cartridge was maintained at 1.2 mL/min (except for CN, with a flow of 1.67 mL/min) using a syringe pump (KD Scientific Model 100, Holliston, MA, USA).

The UVA breakthrough curves were normalized to the initial conditions (i.e. breakthrough absorbance values were divided by the absorbance of the initial sample). Analysis of these breakthrough curves provided a measure of the total material adsorbed, described as a retention coefficient (*RC*) and defined as one minus the maximum breakthrough level (C_{\max}/C_0):

$$RC = 1 - \frac{C_{\max}}{C_0}$$

RC describes the fraction of UVA material that is adsorbed onto each of the SPE sorbents used and is a measure of the polarity. The maximum breakthrough level was obtained by averaging the normalized absorbance

Table 1 | Structure and primary interaction mechanism of SPE sorbents

Sorbent	Structure*	Primary interaction
C18	Si-O-(CH ₂) ₁₇ -CH ₃	Non-polar
C2	Si-O-CH ₂ -CH ₃	Non-polar
CN	Si-O-(CH ₂) ₃ CN	Polar
Silica	Si-OH	Polar
Diol	Si-O-(CH ₂) ₃ OCH(OH)CH ₂ (OH)	Polar
NH ₂ [†]	Si-O-(CH ₂) ₃ NH ₂	Anion exchange (weak)
SAX [‡]	SDB-(CH ₂) ₃ N ⁺ (CH ₃) ₃	Anion exchange (strong)

*Si: silica particle; SDB Styrene-divinylbenzene particle.

[†]Counter ion: OH.

[‡]Counter ion: Acetate.

values after a constant breakthrough level had been reached (Rosario-Ortiz *et al.* 2007b). Data are presented as *RC* obtained for each of the SPE sorbents used. Error bars represent the standard deviation from the mean calculated using propagation of error analysis and multiple evaluations of the error associated with UVA measurements and breakthrough profiles.

It should be noted the PRAM analysis was done only with UVA. However, previous comparisons between TOC and UVA at 254 nm based *RC* determinations have shown limited differences between both methods on water samples collected at other locations (Rosario-Ortiz *et al.* 2007b). As a result, the following analysis refers to chromophoric NOM fraction which may be the most important fraction for water treatment due to its reactivity, i.e. the UVA light absorption due to aromatics and functional groups. Each analysis was performed under ambient conditions because ionic strength and pH have been shown to significantly affect NOM polarity (Rosario-Ortiz *et al.* 2007b).

RESULTS AND DISCUSSION

Performance of the pilot plant

Table 2 presents the water quality parameters for grab samples collected on the three dates NOM polarity was analyzed during pilot plant testing. The raw water feeding

Table 2 | Water quality parameters for grab samples collected from the pilot plant

	%SPW	TOC (ppm)	UVA (cm ⁻¹)	SUVA (m ⁻¹ /ppm)	pH	TDS (ppm)
11/19/2003						
Raw water	76	2.5	0.058	2.32	8.10	345
Ozone effluent	76	2.4	0.042	1.75	n/a	-
Biofilter effluent	76	2.0	0.037	1.85	n/a	350
12/04/2003						
Raw water	80	2.4	0.059	2.43	n/a	331
Ozone effluent	80	2.4	0.049	2.01	n/a	-
Biofilter effluent	80	2.1	0.035	1.67	n/a	340
12/16/2003						
Raw water	92	2.6	0.067	2.57	8.11	298
Ozone effluent	92	2.6	0.058	2.23	n/a	-
Biofilter effluent	92	2.0	0.047	2.35	n/a	309

the pilot plant on the sample collection dates consisted of the SPW and CRW blends shown in Table 2. TOC and UVA at 254 nm for the pilot plant influent varied from 2.1–2.6 mg/L and 0.035–0.067 cm⁻¹, respectively. Removal of TOC during O₃/BF was 0.3–0.5 mg/L, which follows earlier research showing biofiltration removed 0.2–0.5 mg/L TOC in the form of biodegradable dissolved organic matter (Griffini *et al.* 1999). All samples showed similar reductions in UVA and thus comparable reductions in SUVA, but PRAM data revealed NOM polarity was statistically different on each date.

NOM polarity of raw water

Figure 2 presents the PRAM results describing the *RC* obtained for the analysis of raw water during the three sampling events. The observed ranges in *RC* were 0.50–0.70 for the weak anion exchange (NH₂), 0.86–0.89 for the strong anion exchangers (SAX), 0.00–0.10 for the polar sorbents (CN, diol and silica) and 0.05–0.25 for non-polar sorbents (C18 and C2). The high *RC* for the anion exchangers were due to the negative charges associated with NOM that are caused by the de-protonation of carboxylic acids and phenolic groups at ambient pH (Stevenson 1994). These data support the interpretation that NOM at ambient pH is negatively charged with specific polar and non-polar domains, which are accessible to the functional groups on the SPE sorbents.

The *RC* for C18 and C2 were 0.15–0.25 and 0.05–0.12, respectively (Figure 2). Using a series of organic compounds as probes, recent studies showed how sorption onto the

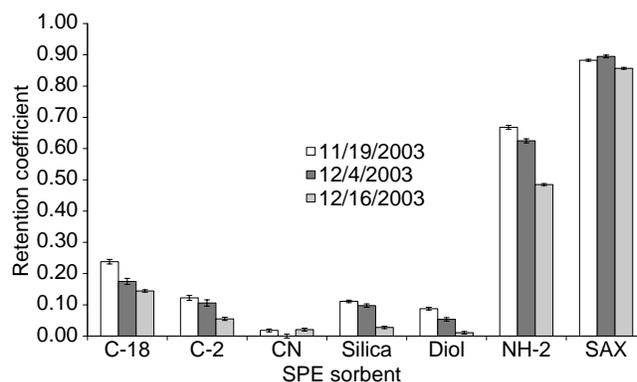


Figure 2 | PRAM results for three raw water samples collected from the pilot plant during six weeks of testing.

non-polar sorbents is related to hydrophobic surface area or non-polar character (Rosario-Ortiz *et al.* 2007b). Therefore, *RC* values obtained for the non-polar sorbents C2 and C18 are related to the overall hydrophobicity or non-polar surface area of the NOM. Differences in magnitude between the *RC* for C18 and C2 are due to the different capacities of both sorbents and the mode of interaction. The C18 sorbent consists of an octadecyl aliphatic chain that has a large surface area available for interaction (see Table 1). Furthermore, the hydrophobic octadecyl chain extends outside the surface of the silica particle, allowing greater access to the hydrophobic parts of the NOM than the shorter C2 chain. The C2 sorbent is an ethyl group with lower surface area and capacity for hydrophobic compounds. Furthermore, the shorter C2 chain suggests that interaction can also be influenced by secondary interactions with the silanol groups on the surface of the silica particle. These secondary interactions may be minimized due to the size of the NOM. Another alternative interpretation is the fact that C18 extends into the water phase, resulting in the ability to interact with the core of the NOM, something not possible with the C2 sorbent.

The *RC* for the polar sorbents (CN, silica and diol) were all less than 0.10 (Figure 2), indicating the overall polar surface area is less than 10%, based on UVA detection. The interaction mechanisms between NOM and these sorbents include dipole interactions and hydrogen bonding. However, the *RC* for polar SPE sorbents are apparently affected by the ability of water to saturate the active sites (Rosario-Ortiz *et al.* 2007b). The uncharged polar groups of the NOM (including ethers, quinones and aldehydes) must displace the water before sorption can occur by dipole or hydrogen bonding. Another theory proposes that the potential of silicic acid in water can act as a bridging agent between the polar functional groups within the NOM and the polar SPE sorbents (Leenheer 2006), but no research has yet explored this theory.

NH₂ is a weak anion exchanger and SAX is a strong anion exchanger. The differences arise from the molecular structures of these sorbents. NH₂, with a pKa of 9.8, will be positively charged at two units below its pKa and its capacity will depend on the pH of the water, including the pH of the water within the pores. The functional group of

SAX is a quaternary amine that is always charged (Table 1) at ambient pHs.

The pKa within the NOM is between 1.6 and 6 for carboxyl groups and 8 and 12 for phenolic groups (Thurman 1985). As a result, the anion exchange process with NH₂ and SAX probably targets carboxylic acids and a fraction of the phenolic functionalities. Due to its permanent charge, SAX removes a higher proportion of negatively charged compounds than NH₂, indicating less selectivity. The compounds that adsorb onto NH₂ do so less strongly than for SAX and may be moieties within the NOM with lower charge to radius ratios and, therefore, less anionic character. Additionally, changes to the pH of the solution within the pores could cause a reduction in the capacity of NH₂ to adsorb anionic moieties. This process is not a concern with SAX.

The *RC* for NH₂ and SAX were 0.48–0.67 and 0.86–0.89, respectively (Figure 2). This indicates that the proportion of negatively charged chromophoric NOM is greater than approximately 50%. The differences, as discussed previously, are due to different capacities between both sorbents. However, the *RC*s for SAX should represent a better estimate for the overall proportion of charged moieties (carboxylic acids and phenolic groups) than NH₂.

Temporal variability of NOM

One important observation from Figure 2 is the apparent temporal variability in NOM polarity observed for the raw water samples. For example, the *RC* for C18 were 0.24, 0.17 and 0.14 for the three samples, with a standard error < 0.01. This variability could be ascribed to NOM changes caused by fluctuations in the CRW and SPW blends, which varied during testing (Table 2), as well as to temporal variability of the NOM. The relative importance of these two variables is difficult to evaluate with the limited number of samples analyzed. However, the last two samples had significant differences in NOM polarity in spite of having similar water blend ratios. This indicates part of the variability was caused by other environmental or geochemical effects involved in the creation and modification of the NOM, including differences in the chemistry within the water column.

Environmental factors are known to affect the concentration and physicochemical properties of NOM.

Rosario-Ortiz *et al.* showed samples collected at the same location during a six month period had different NOM properties and were different from samples collected at other streams (Rosario-Ortiz *et al.* 2007a). In another example, Sachse *et al.* evaluated the influence of catchment characteristics on the properties of NOM (Sachse *et al.* 2005). This research presents PRAM as a method to evaluate these differences and their effect on water treatment.

Evaluation of polarity changes caused by ozonation

Figure 3 presents the PRAM results obtained for the analysis of ozone effluent samples. The *RC* for C18 were between 0.11 and 0.18, a general decrease from the values between 0.14 and 0.24 for the raw waters. When taking the average of the three samples, this result indicates a reduction in the total hydrophobic area of approximately 15%. The *RC* for diol increased from an average of 0.05 for the raw water to an average of 0.13 for the ozone effluent, indicating an increase in the proportion of polar functional groups and overall polarity.

Another important observation from Figure 3 was the variation between the polarity of each sample after ozonation, which was different from that observed for the raw water (Figure 2). For example, the C18 *RC* for the raw water samples were 0.24, 0.17 and 0.14, with the errors showing each sample was significantly different. In the ozone effluent, these values were 0.17, 0.10 and 0.16, revealing

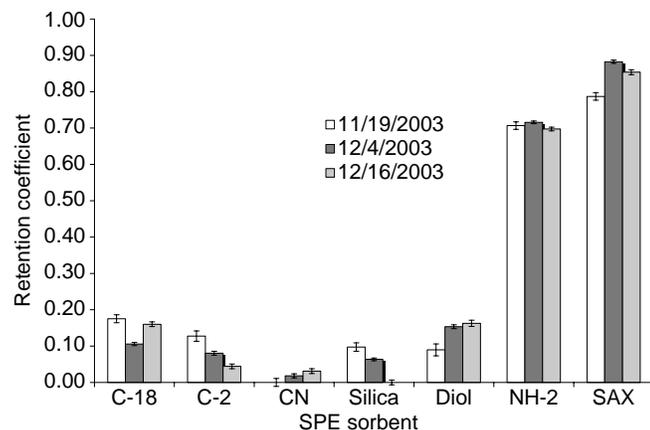


Figure 3 | PRAM results for the three ozone effluent samples collected from the pilot plant during six weeks of testing.

the first and last sample to be similar. This important finding, which requires additional study to determine its effect on treatment systems, indicates variability in the NOM entering the system and its reactivity during ozonation.

Changes in NOM polarity caused by ozone treatment were further investigated. Figure 4 shows the changes in *RC* between the ozone contactor influent and effluent for the three sampling events. A negative difference in *RC* for a sorbent indicates ozonation decreased the NOM polarity fraction that interacts with that sorbent. The opposite is true for a positive change in the *RC* value. This simple mathematical manipulation can be used to follow NOM changes across a treatment process and helps clarify the physicochemical transformations that occur.

As Figure 4 shows, the C18 *RC* values decreased for two of the three samples and practically remain unchanged for the last sample. These results indicate NOM oxidation reduced hydrophobicity, which agrees with other reports that used resin fractionation for NOM analysis through ozone treatment (Westerhoff *et al.* 1999). However, a similar trend was not observed for C2. Since the capacity of C2 is less than that of C18 for the adsorption of hydrophobic compounds, the data suggest hydrophobicity changed within the core of the NOM and beyond the reach of the shorter C2 chain.

The *RC* values obtained for diol increased during the last two samples and remain unchanged for the first one. This change corresponds to an increase of the overall proportion of polar functional groups, confirming previous research demonstrating ozonation causes an increase in the overall NOM polarity. This increase is also observed for the NH₂ *RC* values. The trends for both diol and NH₂ are similar, although the magnitude of the increase is larger for NH₂. This suggests that the same groups responsible for increase in diol adsorption may be causing an increase in anionic character as measured by NH₂. Ongoing investigations are examining the different mechanisms of NH₂ and diol to explain this result. On the other hand, SAX does not follow the increase demonstrated by NH₂. For the first sample, SAX describes a decrease in the proportion of negative charges compared to the other two samples in which no change is observed. Additional work in this area will elucidate these differences and their significance.

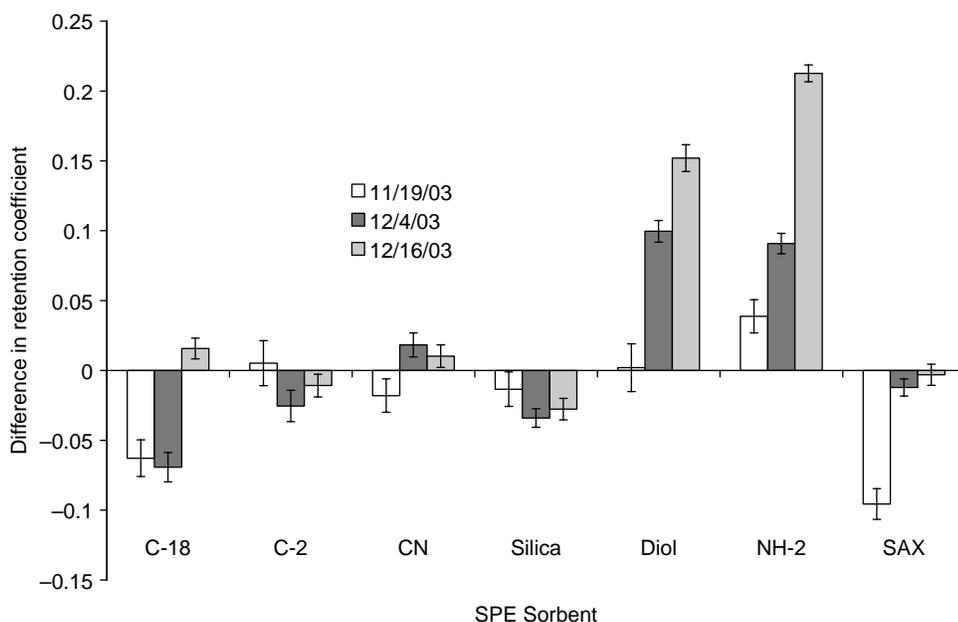


Figure 4 | Difference in retention coefficient between raw water and ozone effluent. A positive difference indicates an increase in the specific polarity fraction. A negative difference indicates a decrease in the specific polarity fraction by removal or chemical transformation.

These results generally agree with previous observations on the effect of ozone on NOM, although this study detected differences within all the samples, a result not previously seen. The vast majority of studies examining the changes in the polarity of NOM are done using resin fractionation techniques, limiting the number of samples analyzed due to experimental time and costs. As discussed previously, the variability may depend not only on variable water blends but also on other environmental factors.

Analysis of complete treatment

Figure 5 presents the polarity results for the filter effluent samples. The ranges in *RC* values were 0.01–0.09 for C18 and <0.05 for the other polar and non-polar SPE sorbents. The C18 *RC* values for the first two samples were within experimental error of each other and different from the third sample. This variability is dissimilar from the polarity fluctuations observed in the raw water samples, which all had different *RC*. For the other polar and non-polar sorbents (C2, CN, silica and diol), the *RC* values were within experimental error, with the exception of the CN *RC* for the last sample. The *RC* values for NH₂ were 0.39–0.45 while the *RC* for SAX were >0.80. The NH₂ *RC* values were

within experimental error for the first two samples and different from the third sample, but the SAX *RC* were all different. These results suggest that the fraction of negatively charged moieties which interact with NH₂ are more sensitive to oxidation than those which interact with SAX.

Analysis of Figures 2 and 5 reveals the differences in NOM polarity between the raw water and filter effluent samples. The average C18 *RC* values decreased from 0.19 to 0.06 during treatment, representing a 68% reduction in hydrophobicity. Results for the raw water samples indicate

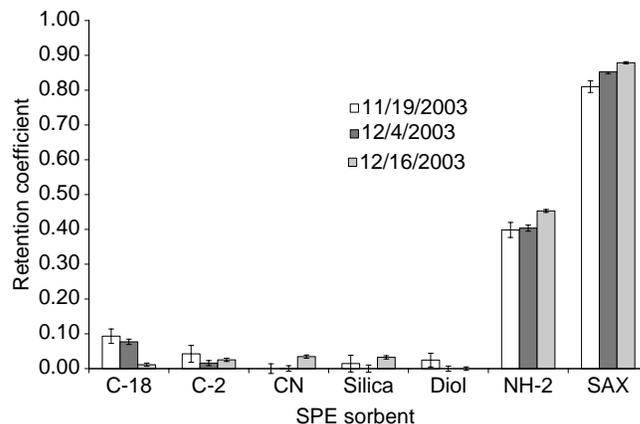


Figure 5 | PRAM results for the three filter effluent samples collected from the pilot plant during six weeks of testing.

the C18 *RC* were different from each other, whereas the C18 *RC* for the first two filter effluent samples were within experimental error, showing treatment reduced NOM variability. A similar decrease was observed for the average C2 *RC* values, which declined from 0.09 to 0.03 for the raw water and filter effluent, respectively. This change represented a 66% reduction in hydrophobic character and was similar to the C18 data.

The average *RC* values for silica and diol decreased from 0.08 and 0.05, respectively, to nearly 0 for both sorbents. This result indicates treatment eliminated interaction between the sorbents and most of the polar functional groups. However, it is impossible to conclude whether these groups were removed or chemically modified, since the analysis is based on UVA detection.

The average NH_2 *RC* decreased from 0.59 to 0.40. This change corresponds to a 32% decrease in the proportion of chemical groups that interact with this sorbent. For SAX, the *RC* values remained similar (0.88 and 0.85) between raw and filter effluent.

The changes in NOM polarity between the ozone effluent and the filter effluent show significant decreases in *RC* for the NH_2 anion exchanger, diol, as well as for the non-polar materials C18 and C2 (Figure 6). Significant

reductions in *RC* were observed for diol and NH_2 . Overall, the changes in TOC were minimal, which may indicate that small polar compounds were removed.

Alternative analysis with three sorbents

In this paper, the evaluation of NOM polarity using PRAM included seven sorbents, allowing a multidimensional characterization of NOM. In applications that only need a general assessment of polarity trends, the C18, diol and SAX sorbents could be used to measure hydrophobicity, hydrophilicity and anionic character.

Figure 7 presents the sum of the C18, diol and SAX *RC* values for the sample collected on 16 December 2003. This figure shows the overall hydrophobicity of the material decreases across the treatment process. Ozonation decreased the non-polar character of NOM while increasing its polarity. The rest of the treatment process (coagulation, flocculation, sedimentation and biofiltration) decreased both non-polar and polar functional moieties of the NOM. Conversely, the proportion of anionic charges measured by the SAX resin remained relatively constant. These results could not be extrapolated to the overall mass of the NOM by TOC measurement due to limitations with

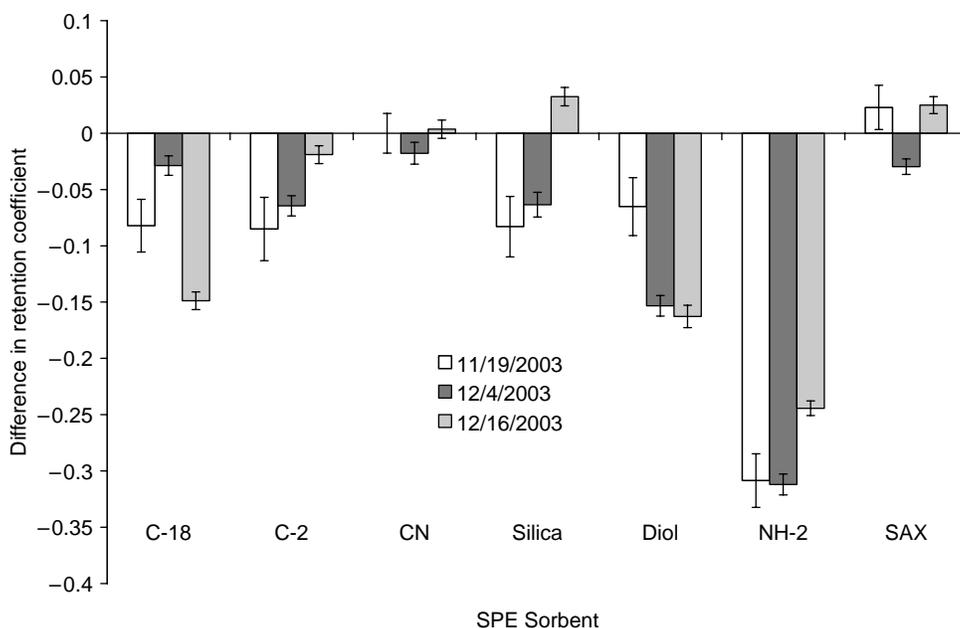


Figure 6 | Difference in retention coefficient between ozone effluent and filter effluent. A positive difference indicates an increase in the specific polarity fraction. A negative difference indicates a decrease in the specific polarity fraction by removal or chemical transformation.

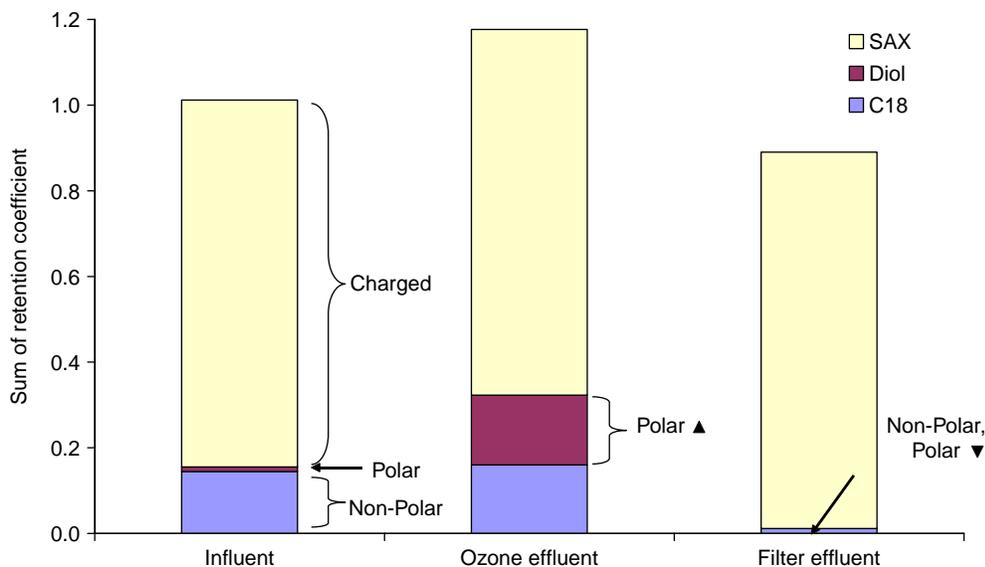


Figure 7 | Change in RC through the treatment for representative sorbents. The three sorbents represent hydrophobic (C18), hydrophilic (diol) and anionic (SAX) character of the NOM.

mass detection due to contamination of the SPE cartridges. Ongoing efforts in this area are focused on the selection and optimization of cleaning procedures which will reduce background carbon contamination. The use of these three sorbents could be an easy indicator of how a specific treatment changes NOM polarity.

CONCLUSION

This paper presents the first time PRAM has been used for the characterization of the polarity of NOM across a treatment system. The application of PRAM offers the advantage of allowing the characterization of numerous samples, resulting in the evaluation of variability of NOM. This is advantageous for drinking water utilities interested in understanding the variations in NOM in their water supplies and how these variations affect treatment systems at ambient pH. Furthermore, PRAM is a viable alternative for polarity characterization when use of resin fractionation is restricted.

In this study, PRAM was utilized to characterize the polarity of NOM and how it is modified by conventional treatment with pre-ozonation and biofiltration. The NOM was characterized by high anionic character (>50%) with 10–25% hydrophobic and 0–10% hydrophilic. The influent

NOM was shown to vary. This variation was also observed in the way the NOM changed through treatment. Ozonation resulted in a clear decrease in hydrophobic character and an increase in hydrophilic character. This agrees with previous investigations using resin fractionation. However, this is the first time that the same behavior has been observed under ambient conditions.

Additionally, the results suggest any generalizations on the short-term behavior of NOM through water treatment plants, especially if variable blends are used, may be ill-founded. The effects of this variability should be evaluated, especially how it may relate to formation of DBPs and other issues within the water industry. Furthermore, NOM polarity should be analyzed at ambient conditions to avoid artificial modification of NOM characteristics. Future studies will emphasize the use of mass-dependent RC values such as TOC in order to evaluate both chromophoric and non-chromophoric components of the NOM.

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