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Fate of Endocrine-Disruptor, Pharmaceutical, and Personal Care Product Chemicals during Simulated Drinking Water Treatment Processes

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The potential occurrence of endocrine-disrupting compounds (EDCs) as well as pharmaceuticals and personal care products (PPCPs) in drinking water supplies raises concern over the removal of these compounds by common drinking water treatment processes. Three drinking water supplies were spiked with 10 to 250 ng/L of 62 different EDC/PPCPs; one model water containing an NOM isolate was spiked with 49 different EDC/PPCPs. Compounds were detected by LC/MS/MS or GC/MS/MS. These test waters were subjected to bench-scale experimentation to simulate individual treatment processes in a water treatment plant (WTP). Aluminum sulfate and ferric chloride coagulants or chemical lime softening removed some polyaromatic hydrocarbons (PAHs) but removed <25% of most other EDC/PPCPs. Addition of 5 mg/L of powder activated carbon (PAC) with a 4-h contact time removed 50% to >98% of GC/MS/MS compounds (more volatile) and 10% to >95% of LC/MS/MS compounds (more polar); higher PAC dosages improved EDC/PPCP removal. EDC/PPCP percentage removal was independent of the initial compound concentration. Octanol−water partition coefficients served as a reasonable indicator of compound removal under controlled PAC test conditions, except for EDC/PPCPs that were protonated or deprotonated at the test pH and some that contained heterocyclic or aromatic nitrogen. Separate chlorine or ozone experiments decreased the EDC/PPCP initial concentration by <10% to >90%; EDC/PPCPs were likely transformed to oxidation byproducts. Ozone oxidized steroids containing phenolic moieties (estradiol, ethynylestradiol, or estrone) more efficiently than those without aromatic or phenolic moieties (androstenedione, progesterone, and testosterone). EDC/PPCP reactivity with oxidants were separated into three general groups: (1) compounds easily oxidized (>80% reacted) by chlorine are always oxidized at least as efficiently by ozone; (2) 6 of the ∼60 compounds (TCEP, BHC, chlordane, dieldrin, heptachlor epoxide, musk ketone) were poorly oxidized (<20% reacted) by chlorine or ozone; (3) compounds (24 of 60) reacting preferentially (higher removals) with ozone rather than chlorine. Conventional treatment (coagulation plus chlorination) would have low removal of many EDC/PPCPs, while addition of PAC and/or ozone could substantially improve their removals. Existing strategies that predict relative removals of herbicides, pesticides, and other organic pollutants by activated carbon or oxidation can be directly applied for the removal of many EDC/PPCPs, but these strategies need to be modified to account for charged (protonated bases or deprotonated acids) and aliphatic species. Some compounds (e.g., DEET, ibuprofen, gemfibrozil) had low removals unless ozonation was used. Other compounds had low removals by all the WTP processes considered (atrazine, iopromide, meprobamate, TCEP), and removal processes capable of removing these types of compounds should be investigated.

Introduction

Endocrine-disrupting compounds (EDCs) and pharmaceuticals and personal care products (PPCPs) have been detected in water supplies and wastewater effluents around the world (*1*−*4*). Some EDC/PPCPs exhibit adverse ecological impacts that have raised concern among public and regulatory groups about the fate of such compounds during potable water treatment and human exposure in drinking water (*3*, *5*−*13*). Some EDC/PPCPs are more polar than current regulated polyaromatic contaminants. This, coupled with occurrence at trace levels (parts per trillion), creates unique challenges for the analytical detection and assessment of removal performance by potable water treatment plant (WTP) processes (*3*, *12*). Drinking water treatment primarily relies upon adsorptive and oxidative processes to remove or transform organic materials. Recent studies for selected groups of EDC/PPCPs, pesticides, and herbicides indicate that coagulation, sedimentation, and filtration achieve minimal levels of removal (*13*−*16*). However, addition of common disinfectants (e.g., chlorine or ozone) can result in reaction and transformation of these compounds (*17*−*26*). For a small number of EDC/PPCPs, rate constants and oxidation mechanisms have been quantified (*18*, *19*, *21*, *24*, *27*−*30*). However, a comparison for the fate of a broad range of EDC/PPCPs at environmentally relevant concentrations across a number of common WTP processes (coagulation, activated carbon adsorption, chemical disinfection) is not currently available.

Chemical coagulation and softening aid in removing suspended solids (i.e., turbidity) from the water and aid in removing dissolved organic carbon (DOC). Chemical coagulation in water treatment usually employs aluminum- or iron-based salts, which precipitate as metal hydroxides. Chemical lime softening removes dissolved calcium and magnesium, using lime and soda ash to precipitate calcium carbonate at lower pH and magnesium hydroxide at pH > 11. Coagulation alone is generally not effective in removing trace-level organic pollutants (*31*, *32*).

Activated carbon adsorbs many organic pollutants (*33*). The USEPA identifies packed-bed granular activated carbon

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(GAC) as a "Best Available Technology" for treating numerous regulated organic pollutants. Powder activated carbon (PAC) effectively removes many problematic organic pollutants (e.g., taste and odor compounds, and some pesticides and herbicides). In GAC systems, adsorbed contaminant concentrations equilibrate with influent liquid-phase concentrations, whereas in traditional PAC applications the solid-phase contaminant concentrations approach equilibrium with reactor effluent liquid-phase concentrations. Traditional PAC applications add a PAC slurry at dosages of 1 to 25 mg/L to a solids-contact, or flocculation, chamber that has contact times of 0.5 to 5 h; removal of PAC (with adsorbed compounds) occurs during sedimentation and filtration processes (*34*). Sophisticated stand-alone systems using fluidized PAC reactors and recirculating PAC reactors coupled with ultrafiltration membrane systems both lead to long contact times between PAC and organics in the water, allowing full utilization of the PAC adsorption capacity (*35*−*37*). This paper considers traditional PAC addition to conventional treatment since it represents a relatively low-cost strategy for drinking water utilities to improve EDC/PPCP removal.

For some organic compounds, adsorptive removal by PAC may not be effective, but the compounds may be reactive with oxidants (*38*). During water treatment, chlorine or ozone addition disinfectants inactivate microbes, oxidize reduced metals, and oxidize organic material. Electron density effects of functional groups and degree of protonation affect the potential reactivity of organic compounds with oxidants (*25*, *39*, *40*). Electron-donating (e.g., hydroxyl, amine) or electron-withdrawing (e.g., carboxyl) functional groups lead to increasing and decreasing reactivity, respectively, for substituted aromatic rings (*40*). For example, free chlorine reacts rapidly with phenolic compounds, mainly through the reaction between HOCl and the deprotonated phenolate anion (*41*). This results in sequential chlorine addition to the aromatic ring followed by ring cleavage. The reactivity of the phenolic functional group likely explains the rapid transformation during chlorination of some estrogenic hormones (estradiol, ethynylestradiol, estriol, estrone) which contain phenolic moieties (*17*, *21*, *27*). The formation, fate, detection, and toxicity of oxidative byproducts from pesticides and EDC/PPCPs is of potential concern (*17*, *27*, *42*).

Several studies have investigated EDC or PPCP removal by ozone or chlorine, but direct comparisons are lacking between these two oxidants and a broad range of EDC/PPCPs under conditions typical of drinking water treatment facilities. A few examples of previous studies are presented; additional tabular data are provided as Supporting Information. The transformation of several amine-containing antibiotics, diclofenac, and caffeine were observed in laboratory experiments with chlorine (*14*, *24*, *43*). Ozonation of estrogenic chemicals is effective (*21*, *44*), but there are limited data on the reactivity of nonestrogen-based hormones (e.g., testosterone, progesterone, androstenedione);hence, these were included in our study. Ozonation significantly reduced concentrations of several estrogens, musk fragrances, and some pharmaceuticals (diclofenac, carbamazepine, and bezafibrate), but not clofibric acid (*45*, *46*). Removal of clofibric acid, ibuprofen, and diclofenac improved when ozonation was conducted in the presence of hydrogen peroxide (0.4−0.7 mg of H_2O_2/mg of O_3 dosed) (*26*, *47*). As ozone decays in water, the reactions produce hydroxyl ($HO^{\bullet}$) radicals. H_2O_2 addition increases the rate of molecular ozone decay (i.e., lower molecular ozone concentrations) but also increases $HO^{\bullet}$ concentrations. Molecular ozone is a selective electrophile that reacts with amines, phenols, and double bonds, whereas $HO^{\bullet}$ reacts less selectively with organic compounds (*25*, *48*, *49*). Due to the selective nature of ozone, micropollutant transformation may require the use of advanced oxidation processes (AOPs), such as O_3/H_2O_2 (*42*, *50*).

The objective of this paper is to compare relative removal for a large number of EDC/PPCPs spiked at environmentally relevant concentrations together into three natural waters or a model water by adsorptive processes (metal salt coagulation, lime softening, PAC addition) and oxidative processes (chlorine, ozone) under conditions (dosages, contact times) practiced in water treatment plants. EDC/PPCPs were selected on the basis of occurrence in aquatic systems, chemical properties, and ability to be analytically detected by recently developed liquid (LC/MS/MS) and gas (GC/MS/MS) mass spectroscopy methodologies (*1*−*4*). LC/MS/MS and GC/MS/MS analysis separated EDC/PPCPs largely on the basis of volatility (i.e., polarity). Several organic compounds (herbicides, pesticides, PAHs) currently regulated by the USEPA were included in this EDC/PPCPs study because (1) the compounds could be analyzed by the scheme employed; (2) comprehensive studies for the fate of all these compound mixtures of trace-level contaminants were limited and, when available, were generally at parts-per-billion rather than parts-per-trillion concentrations; (3) removal of these compounds serve as "markers" against which the removal of other EDC/PPCPs can be compared; and (4) the USEPA lists these compounds as potential endocrine disruptors. Trends are investigated between EDC/PPCP chemical properties and observed relative percentage removals for each individual water treatment process. The findings of this project can guide future research in selecting representative EDC/PPCPs for more detailed study of reaction mechanisms or fate in water systems.

Materials and Methods

EDC/PPCP Compounds. Table 1 identifies the EDC/PPCPs that were studied by spiking them together into four different source waters (see Supporting Information for EDC/PPCP chemical structures). All compounds were obtained from Sigma-Aldrich (St. Louis, MO) except atrazine and *N*,*N*-diethyl-*m*-toluamide (DEET) from Accustandard (New Haven, CT), fluoxetine and iopromide from the United States Pharmacopeia (Rockville, MD), and hydrocodone from Cerilliant (Round Rock, TX). Concentrated spiking solutions of the target compounds were prepared at high concentrations (10−250 mg/L) in methanol to minimize the volume of solvents introduced into experiments. Many compounds had low water solubility (e.g., PAHs) and therefore could not be spiked as neat standards. A small volume of the spiking solutions (50−250 μL) was injected into a 28-L stainless steel (SS) tank containing a source water. The added methanol increased the DOC concentration by approximately 0.7 mg/L. Methanol can increase ozone decay rates and affect $HO^{\bullet}$ concentrations, so every attempt was made to minimize its concentration. Acetone would have produced less of an effect on ozone chemistry, but it was not used in this study.

Source Waters. Three surface waters that provide raw water to WTPs and one model water were investigated (Table 2). The waters exhibit a range of DOC, pH, alkalinity, and conductance values. The three surface waters contained a few detectable EDC/PPCPs, but generally at less than 10 ng/L for any specific compound. One model water (SRW) was prepared by adding a natural organic matter (NOM) isolate, purchased from the International Humic Substances Society (IHSS, St. Paul, MN), to Nanopure water (NANOpure Infinity, Barnstead, IA) containing sodium bicarbonate (1 mM) as a pH buffer. IHSS isolated the NOM by reverse osmosis from the Suwannee River. Waters were stored in 28-L SS containers. Waters were filtered (ashed Whatman GF/F; 0.7-μm nominal pore size) to remove particulate matter prior to spiking EDC/PPCPs for chlorine, ozone, and PAC experiments. EDC/PPCPs were spiked to raw waters without filtration, for alum, ferric,

TABLE 1. List of EDC/PPCP Compounds Studied and Their Properties

compound name	CAS#[a]	use	MW	pK_a[b]	log K_{OW}[c]
		LC/MS/MS Analytes			
acetaminophen	103-90-2	analgesic	151.2	9.7 (9.4)	0.46
androstenedione	63-05-8	steroid[d]	286.2		2.75
atrazine	1912-24-9	herbicide[d]	215.1	<2 (1.6)	2.61
caffeine	58-08-2	stimulant	194.2	6.1	<0
carbamazepine	298-46-4	analgesic	236.3	<2	2.45
DEET	134-62-3	insect repellent	191.3	<2	2.18
diazepam	439-14-5	anti-anxiety	284.8	2.4, 1.5 (3.3)	2.82
diclofenac	15307-79-6	arthritis	318.1	(4.2)	0.70
dilantin	57410	anti-convulsant	252.3	(8.3)	2.47
erythromycin−H_2O	114-07-8	antibiotic	733.9	(8.8)	3.06
estradiol	50-28-2	steroid[d]	272.2	10.4	4.01
estriol	50-27-1	steroid[d]	288.4	10.4 & >15	2.45
estrone	53-16-7	steroid[d]	270.4	10.3 (10.5)	3.13
ethynylestradiol	57-63-6	birth control[d]	296.2	~10.5	3.67
fluoxetine	54910-89-3	anti-depressant	309.33		3.82
gemfibrozil	25812-30-0	anti-cholesterol	250.2	4.7	4.77
ibuprofen	15687-27-1	pain reliever	206.1	4.5 (4.9)	3.97
iopromide	73334-07-3	X-ray contrast media	790.9	<2 & >13	<0
meprobamate	57-53-4	anti-anxiety	218.3	<2	0.7
naproxen	22204-53-1	analgesic	230.1	4.5 (4.2)	3.18
oxybenzone	131-57-7	sunscreen	228.1		3.79
pentoxifylline	6493-05-6	blood viscosity control	278.1	6 & <2	0.29
progesterone	57-83-0	steroid[d]	314.2		3.87
sulfamethoxazole	723-46-6	antibiotic	253.1	2.1 & <2 (5.7)	0.89
tri(2-chloroethyl) phosphate (TCEP)	115-96-8	fire retardant	285.5		1.44
testosterone	58-22-0	steroid[d]	288.2	17.4	3.32
triclosan	3380-34-5	antibiotic	289.6	8 (7.9)	4.76
trimethoprim	738-70-5	antibiotic	290.1	6.3, 4.0, <2 (7.1)	0.91
		GC/MS/MS Analytes			
acenaphthene	83-32-9	PAH[d]	154.2		3.92
acenaphthylene	208-96-8	PAH[d]	152.2		3.94
aldrin	309-00-2	pesticide[d]	364.9		6.50
anthracene	120-12-7	PAH[d]	178.1		4.45
α-BHC	319-48-6	pesticide[d]	287.9		3.80
β-BHC	319-85-7	pesticide[d]	287.9		3.78
δ-BHC	319-86-8	pesticide[d]	287.9		4.14
γ-BHC	58-89-9	pesticide[d]	287.9		3.72
benz[*a*]anthracene	56-55-3	PAH[d]	228.3		5.76
benzo[*a*]pyrene	50-32-8	PAH[d]	252.1		6.13
benzo[*b*]fluoranthene	205-99-2	PAH[d]	252.3		5.78
benzo[*g*,*h*,*i*]perylene	191-24-2	PAH[d]	276.3		6.63
benzo[*k*]fluoranthene	207-08-9	PAH[d]	252.3		6.11
chrysene	218-01-9	PAH[d]	228.3		5.81
DDD	72-54-8	pesticide[d]	320.1		6.02
DDE	72-55-9	pesticide[d]	315.9		6.51
DDT	50-29-3	pesticide[d]	354.5		6.91
dibenz[*a*,*h*]anthracene	53-70-3	PAH[d]	278.4		6.75
dieldrin	60-57-1	pesticide[d]	380.9		5.40
endrin	72-20-8	pesticide[d]	380.9		5.20
fluoranthene	206-44-0	PAH[d]	202.3		5.16
fluorene	86-73-7	PAH[d]	166.2		4.18
galaxolide	55464-57-2	musk	258.2		5.90
α-chlordane	57-74-9	pesticide[d]	409.8		6.10
γ-chlordane	5566-34-7	pesticide[d]	409.8		7.00
heptachlor	76-44-8	pesticide[d]	373.3		6.10
heptachlor epoxide	1024-57-3	pesticide[d]	389.3		5.00
hydrocodone	125-29-1	pain relief	299.4		2.16
indeno[1,2,3-*cd*]pyrene	193-39-5	PAH[d]	276.3		6.70
methoxychlor	72-43-5	pesticide[d]	344		5.08
metolachlor	51218-45-2	pesticide[d]	283.8		3.13
mirex	2385-85-5	pesticide[d]	545.6		7.18
musk ketone	81-14-1	fragrance	294.3		4.31
naphthalene	91-20-3	PAH[d]	128.2		3.30
phenanthrene	85-01-8	PAH[d]	178.2		4.46
pyrene	129-00-0	PAH[d]	202.3		4.88

[a] Chemical Abstract Service Registry Number. [b] pK_a calculated from SPARC (*72*); pK_a values given for −OH, −COOH, or highest NH_x functional groups (pK_a values in parentheses are from literature reports). [c] The log K_{OW} value calculated by EPIWIN (EPI V3.10). [d] These identified compounds are known, probable, or suspected endocrine disruptors (*6*, *9*, *73*, *74*).

and lime experiments. Between 62 and 65 different EDC/PPCPs were together spiked into the three natural waters; due to logistical issues, only 49 different EDC/PPCPs were spiked into the model water (SRW). Control samples containing EDC/PPCP spikes in both filtered and nonfiltered water were conducted in at least triplicate; the cv's of these

TABLE 2. Summary of Water Quality and Oxidation Dosages

parameter	Source Description SRW IHSS isolate from the Suwannee River, GA	CRW Colorado River from Lake Mead, NV	ORW Ohio River near Louisville, KY	PVW Passaic River near Totowa, NJ
dissolved organic carbon [mg/L]	4.0	3.0	3.5	3.5
UV absorbance @ 254 nm [cm^{-1}]	0.192	0.048	0.08	0.09
SUVA [L mg^{-1} m^{-1}]	4.8	1.6	2.3	2.6
ambient pH	7.5	8.2	7.9	6.8
alkalinity [ppm as $CaCO_3$]	<10	140	128	38
hardness [ppm as $CaCO_3$]	0	307	103	80
conductance [μS/cm]	167	1100	338	407
number of EDC/PPCP spiked alum coagulation tests	49	62	65	62
initial pH	5.5 & 8.5	8.2	5.5 & 7.9	6.8
alum dosages [mg of Al^{3+}/L]	6.3 (78)[a]	4.7 (58)[a]	5.5 (68)[a]	6.3 (78)[a]
ferric coagulation tests				
initial pH	8.5	8.2	7.9	6.8
ferric dosages [mg of Fe^{3+}/L]	13	9.8	11.4	13.1
chemical softening tests				
lime dosage [ppm as $CaCO_3$]	72	320	182	180
soda-ash dosage [ppm as $CaCO_3$]	0	167	39	30
final adjusted pH	11.3	11.3	11.3	11.3
PAC adsorption tests				
PAC brands	AC800, WPM	WPM	AC800, WPM	WPM
PAC dosages [mg/L]	1, 5, 20 for WPM 5 for AC800	5	5	5
chlorination				
dose [mg of Cl_2/L]	6.75	3.5	2.8	3.8
ozonation				
dose(s) [mg/L]	4.0 (8.2)	2.5	3.5 (7.0)	3.0
dilution factor(s) for ozone stock	10% (20%)	5.2%	8.8% (17.5%)	6%

[a] Alum dosages in parentheses indicate alum concentrations as $Al_2(SO_4)_3 \cdot 18H_2O$.

were <15%, and these EDC/PPCP concentrations were used as the initial concentration in calculation of percentage EDC/PPCP removal during each experiment. Depending upon the source water and compound class, initial EDC/PPCP concentrations ranged from 10 to 250 ng/L.

Glassware. All glassware, supplies, and the SS drum were solvent-rinsed 3 times each using acetone, hexane, and methanol obtained from Burdick & Jackson (Muskegon, MI) or Sigma-Aldrich (St. Louis, MO). All control samples and oxidant experiments were conducted in 1-L silanized amber glass bottles (Eagle-Picher, Miami, OK) containing the test water. Some EDC/PPCP adsorption was observed without silanized glassware (*51*), and use of silanized glassware has been recommended elsewhere (*52*).

Chemical Treatments. Coagulation experiments were conducted as jar tests using a six-place gang stirrer (Phipps and Bird, Richmond, VA) (*53*); jars were 2-L silanized glass beakers filled with 1.5 L of EDC/PPCP-spiked source water. Chemicals (see below) were added via pipet during a rapid mixing step. Mixing conditions were 1 min of rapid mixing at 100 rpm, 20-min flocculation at 30 rpm, and 60 min of settling time. Experiments were conducted at a room temperature of approximately 20 °C. After settling, supernatant samples were collected and filtered (ashed Whatman GF/F) prior to LC/MS/MS and GC/MS/MS analysis.

Two coagulants were used: aluminum sulfate (alum, $Al_2(SO_4)_3 \cdot 14H_2O$) and ferric chloride (ferric, $FeCl_3 \cdot 6H_2O$) (Fisher Scientific). An alum dose (10 mg of alum/mg of TOC) was selected on the basis of the total organic carbon (TOC) concentration of the water and represents the high end of dosages that may be used to meet the USEPA Interim Enhanced Surface Water Treatment Rule (*54*). Ferric dosages were based on adding equivalent dosages of ferric and aluminum ions. Calcium hydroxide ($Ca(OH)_2$) and soda ash (Na_2CO_3) simulated chemical softening treatments (*55*). The applied lime and soda ash dosages for excess-lime softening conditions were calculated on the basis of initial pH, alkalinity, and carbonic acid concentration. After lime and soda ash addition, the pH was adjusted to 11.3 ± 0.2 by adding a sodium hydroxide solution. For some waters, a pH 9 chemical softening test was conducted in which $Ca(OH)_2$ was added to raise the pH to 9; this represents many WTPs that practice "partial" softening. Table 2 summarizes the chemical dosages for individual experiments.

Powder Activated Carbon Experiments. Two activated carbon brands, AC800 (Acticarb, Dunnellon, FL) and WPM (Calgon Carbon Corp., Pittsburgh, PA) were selected because they exhibited different removal capability for bisphenol A (common plasticizer), 17β-estradiol (natural estrogen), and 17α-ethynyl estradiol (synthetic estrogen) in screening tests with six different PAC brands (*56*). PAC was hydrated for 24 h in Nanopure water prior to use and added to the samples as a slurry (1000 mg/L). Table 2 summarizes PAC dosages. PAC experiments were performed using the six-place gang stirrer (mixing at 100 rpm); 2-L glass beakers filled with 1.5 L of source water served as reactors. A contact time of 4 h was followed by 1 h of settling. Supernatant was collected and filtered (ashed Whatman GF/F) to remove residual PAC.

Limited experimentation was also conducted with one water (CRW) and distilled water spiked only with tritium (^{3}H)-labeled 17β-estradiol (E2) to investigate the effects of contact time and initial E2 concentration. 17β-Estradiol [2,4-^{3}H] purchased from Sigma-Aldrich (St. Louis, MO) with an activity of 16.8 Ci/mmol at a concentration of 1.0 mCi/mL in ethanol, equivalent to 16.2 μg/mL, was used.

Analysis. Analysis was performed using gas chromatography with tandem mass spectrometric detection (GC/MS/MS) for more volatile compounds (e.g., organochlorine pesticides and polycyclic aromatic hydrocarbons) and liquid chromatography with tandem mass spectrometric detection (LC/MS/MS) for more polar and less volatile compounds (Table 1). These extraction and analytical methods have been published previously (*51*, *57*). Briefly, the method began with the automated extraction (Oasis HLB solid phase, methanol/

MTBE extraction) of a 1-L water sample. The resulting extract is concentrated to 1 mL and split equally for GC and LC analyses. The extract was used directly in LC analyses, whereas the extract was liquid−liquid-extracted (NaCl added then 3× extracted using DEM in hexane) to remove polar/nonvolatile compounds prior to GC analysis (*57*). Stable isotopically labeled surrogates (anthracene, DDE, diazepam, testosterone, caffeine, atrazine, estradiol) and internal standards were added to each sample prior to extraction in order to determine the efficiency of extraction and/or losses due to matrix suppression. Surrogate recovery rarely varied by more than 15%, therefore, data were not adjusted for surrogate recovery. LC/MS/MS compounds were analyzed using an Applied Biosystems API 4000 triple quadrupole with an Agilent 1100 liquid chromatograph. Retention times and basic information on the LC/MS/MS procedure and GC/MS/MS analysis performed using both GC-ion trap and GC-triple quadrupole systems are detailed elsewhere (*51*, *57*). The minimum reporting level (MRL) is 1 ng/L and 10 ng/L for all LC/MS/MS and GC/MS/MS compounds, respectively.

DOC was measured using a Shimadzu TOC5050 analyzer (high-temperature combustion at 680 °C, Shimadzu, Japan). A VWR conductivity meter (model 2052) and a Beckman pH meter (model 511201) were calibrated prior to each use. Alkalinity was measured (Hach Digital Titrator model 16900) by colorimetric titration. Radioactivity was measured by a liquid scintillation counter (GMI, Beckman LS6500, Albertville, MN). For scintillation measurements, 4 mL of scintillation liquid (ScintiSafe Plus, Fisher Scientific) was pipetted into a standard glass vial and 100 μL of the sample was added. The method detection limit (MDL) was 0.005 nM (1.4 ng/L).

Chlorination. Stock free chlorine solutions (1200 mg/L) were prepared daily from 5% sodium hypochlorite (Fisher Scientific, USA) in Nanopure water. Aliquots of the stock solution were transferred to 1-L silanized amber glass bottles containing the water sample; zero headspace was maintained in the bottle. Free chlorine residuals were measured by the DPD Method using a Hach DR4000 spectrophotometer (USEPA-approved Hach Method #8021, Hach Company, Loveland, CO).

Ozonation. Stock liquid ozone solutions (40−50 mgO_3/L) were prepared daily by passing gaseous ozone (OREC model V5-0, Phoenix, AZ), fed by pure oxygen, through a gas-washing bottle containing pH 5 phosphate-buffered Nanopure water, followed by chilled Nanopure water at 4 °C (*58*, *59*). Dissolved ozone concentrations in the stock solution were measured spectrophotometrically (258 nm; $\epsilon_{O_3} = 3150$ M^{-1} cm^{-1}). Aliquots of the stock liquid ozone solution were transferred via a glass syringe to 1-L silanized amber glass bottles containing the water sample; zero headspace was maintained in the bottle. Prior to ozone dosing, a predetermined volume of water sample was removed from a full 1-L silanized amber glass bottle so that it could accommodate the liquid ozone solution. Consequently, samples were diluted by the ozone stock solution prepared in Nanopure water by 7.5 to 10%, depending upon the ozone dose applied (3−4 mg/L); the dilution was up to 20% for a high ozone dose of 8.2 mg/L in one experiment (SRW). EDC/PPCP concentrations reported for ozonation experiments account for dilution. Dissolved ozone residuals in source waters were measured by the indigo method (Standard Method 4500-O3) using a Hach DR4000 spectrophotometer. Hydrogen peroxide (H_2O_2) was added (0.025 mg of H_2O_2/mg of O_3) just prior to ozone stock solution addition in select experiments to increase $HO^{\bullet}$ concentrations. Working stock solutions of H_2O_2 (30 mg/L; Fisher Scientific) were prepared in Nanopure water.

Oxidant Quenching. Control studies were conducted to select an oxidant quenching reagent that would not affect the stability of EDC/PPCPs or their recovery. Control studies compared LC/MS/MS compound recoveries for three common reagents used to quench residual chlorine (100 mg/L sodium thiosulfate ($Na_2S_2O_3$), 50 mg/L ammonium chloride (NH_4Cl), and 50 mg/L ascorbic acid ($C_6H_8O_6$)). Deionized water spiked with a combination of 26 EDC/PPCP compounds at initial concentrations of approximately 100 ng/L was placed in glass vials. Quenching reagents were added to each vial; a control set of vials contained no quenching reagents. The vials were held at 4 °C for 5, 13, and 14 days prior to solid-phase extraction. $Na_2S_2O_3$ addition had detrimental effects (<80% recovery) on trimethoprim, erythromycin−H_2O, fluoxetine, atrazine, diazepam, progesterone, and diclofenac. NH_4Cl addition resulted in 90−110% recoveries of all 26 compounds. Addition of ascorbic acid yielded 85−115% recoveries of 25 compounds, but only 48% recovery of erythromycin−H_2O. On the basis of good recoveries with NH_4Cl and ascorbic acid, either was considered acceptable. Ascorbic acid is routinely used to quench chlorine residuals during sampling for pesticides (*60*), so it was selected for our study and added after 24 h of contact time for chlorine experiments in PVW and CRW. Because details of quenching requirements were not finalized at the time of experimentation, quenching reagents were not added to free chlorine experiments with SRW or ORW, thus slightly longer contact times than in PVW or CRW experiments occurred with for SRW and ORW during the free chlorine experiments.

EDC/PPCP Sample Preservation. Separate studies indicated that many EDC/PPCPs spiked into surface waters may not be biologically stable over even a few sample holding days (*51*, *61*). Sulfuric acid addition to pH 2 was required to prevent EDC/PPCP degradation; minimal hydrolysis occurred. Therefore, after each experiment using the three surface waters, samples were acidified with concentrated sulfuric acid. At the time of the model water experiment (SRW), these stability tests had not been completed, and sulfuric acid was not added, but samples were stored in the dark at 4 °C and extracted within one week; no significant degradation was expected.

Results

Chemical Treatments. Figure 1 presents data from PVW in a control sample (no coagulant added) and after three different chemical treatments. Error bars on the control sample indicate high reproducibility in quantifying EDC/PPCP concentrations; similar levels of reproducibility were observed throughout this study. EDC/PPCP control concentrations ranged from 30 to 150 ng/L. Discussion throughout the remainder of this paper will mainly compare percentage removal ($[1 - C/C_0]\times100\%$) of EDC/PPCPs to simplify comparison between removal of different compounds; C_0 and C are EDC/PPCP concentrations in the control sample and after experimental treatment, respectively. In Figure 1, percentage removals by alum coagulation are also presented on the x-axis in parentheses. Removals greater than 20% were considered statistically significant. In PVW, only 2 of the 28 compounds analyzed by LC/MS/MS exhibited >20% removal during alum coagulation (hydrocodone, 24%; erythromycin−H_2O, 33%). In contrast, 12 of 32 of the GC/MS/MS compounds exhibited >20% removal during alum coagulation; the highest removals were for PAHs. For most EDC/PPCPs, ferric coagulation achieved comparable removals as equivalent alum dosages.

For many compounds, differences between source water composition or differences in EDC/PPCP spiked concentrations did not affect the percentage removal of EDC/PPCPs. Among the few exceptions were the hormones androstenedione, progesterone, and testosterone, which were better removed in ORW than in the other waters. This may be due

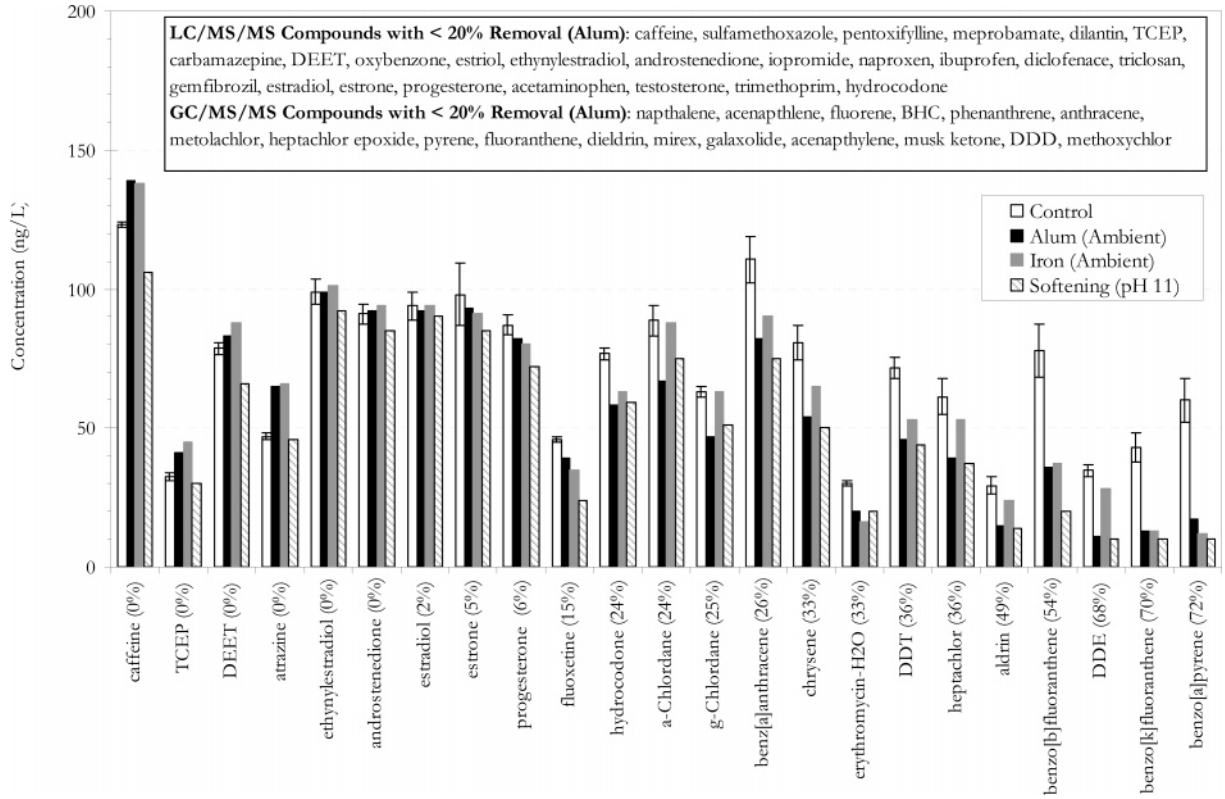

FIGURE 1. Concentrations of select compounds before (control) and after chemical treatments in PVW at ambient pH (6.8). Values in parentheses are percentage removal after alum treatment; C_0 is based upon EDC/PPCP spiked concentration in nonfiltered source water. Caption identifies other compounds analyzed by LC/MS/MS or GC/MS/MS methods with <20% removal during alum coagulation experiments (PVW) at ambient pH. Error bars represent the standard deviation ($n = 5$).

to experimental accuracy, but more likely EDC/PPCPs partitioned onto the suspended sediment particulate materials present in the natural water (ORW); the model water (SRW) did not contain suspended sediment materials. EDC/PPCP removal could occur if these compounds partition onto the particulate matter initially present in the source water or partition/adsorb onto metal (hydr)oxide or carbonate precipitates formed during coagulation. Overall, the percentage removal averaged across all four source waters at ambient pH for EDC/PPCPs detected by LC/MS/MS was only 6% ± 8% (range: 0−28%; $n = 28$), and 26% ± 24% (range: 0−83%; $n = 34$) for EDC/PPCPs detected by GC/MS/MS. Among GC/MS/MS compounds, the more hydrophobic PAHs were removed better (60−80%), as were some of the hydrophobic pesticides (25−50% removal). GC/MS/MS compounds with log K_{OW} greater than 6.5 also had removals >20%. This suggests removal by partitioning onto particulates in suspension or onto precipitated solids that had adsorbed DOC. Differences in removals between ambient pH and pH 5.5 were within experimental reproducibility (20%), except for removal of PAHs and DDT in SRW which tended to be removed better at higher pH levels. Fluoxetine was also removed better at higher coagulation pH levels; this may be attributed to an amine group in its structure and subsequent interaction with metal hydroxide precipitate surfaces.

Chemical lime softening achieved comparable EDC/PPCP removal as alum or ferric coagulation within experimental reproducibility (20%) and were probably attributed to the same removal mechanisms (sorption onto turbidity and precipitated solids). To minimize base hydrolysis, samples were filtered and the pH was reduced immediately after chemical softening treatment (~4 h). Base hydrolysis was not specifically studied in DOC-free water.

Overall, chemical precipitation processes achieve minimal removal of most EDC/PPCP compounds examined, with the highest removals observed for PAHs. The neutral semivolatile compounds analyzed by GC/MS/MS had higher removals compared against more polar compounds that required LC/MS/MS analysis. The two aliphatic compounds in this study (TCEP and meprobamate) had among the lowest removals, and neutral compounds with higher log K_{OW} exhibited better removal and suggests that EDC/PPCP hydrophobicity may be an indicator for removal potential.

Powder Activated Carbon Addition. Experiments were conducted with ^{3}H-E2 and PAC to quantify effects of PAC dosage (0.01−25 mg/L), PAC kinetics (1, 4, 24 h), and initial ^{3}H-E2 concentrations. At a 1 mg/L PAC dosage (AC800) in CRW, ^{3}H-E2 removal was 32, 58, and 84% after contact times of 1, 4, and 24 h, respectively. Increasing PAC dosage improved ^{3}H-E2 removal, and the effect of contact time became less significant at PAC dosages of 5 and 25 mg/L. PAC dosages of 5 mg/L for all source waters, and 1 and 20 mg/L for SRW, with a 4-h contact time were selected for experiments with the entire suite of EDC/PPCPs because of these ^{3}H-E2 results and because they represent conditions typically employed at WTPs (*34*). Furthermore, these dosages would also achieve partial, but not complete, EDC/PPCP removal, allowing relationships between compound structure and removal potential to be explored.

Over 2 orders of magnitude in initial ^{3}H-E2 concentration, removal of ^{3}H-E2 by PAC in Nanopure water and CRW was nearly independent of initial concentration (Table 3), which is consistent with other trace-contaminant studies (*34*, *62*). Data in Table 3 also demonstrate the effects of natural water matrix constituents (mostly DOC) on trace-level ^{3}H-E2 removal by PAC. CRW had approximately 45% less ^{3}H-E2 adsorption capacity than Nanopure water. Therefore, in experiments using a suite of EDC/PPCPs, small differences in initial EDC/PPCP concentration between source waters probably is not important, but the amount and characteristics

TABLE 3. Percentage Removal (%) of ^{3}H-17β-Estradiol after 4-Hour Contact with PAC (1 mg/L WPM) in Buffered Nanopure Water and a Surface Water (CRW)

	percentage removal (%) after PAC treatment for variable initial ^{3}H-17β-estradiol concentration				
water source	6.8 ng/L	27 ng/L	135 ng/L	270 ng/L	1360 ng/L
Nanopure water (buffered at pH 8.2)	71 ± 10	90 ± 1	83 ± 4	77 ± 5	77 ± 3
CRW (pH 8.2)	39 ± 7	51 ± 4	43 ± 9	46 ± 5	45 ± 3

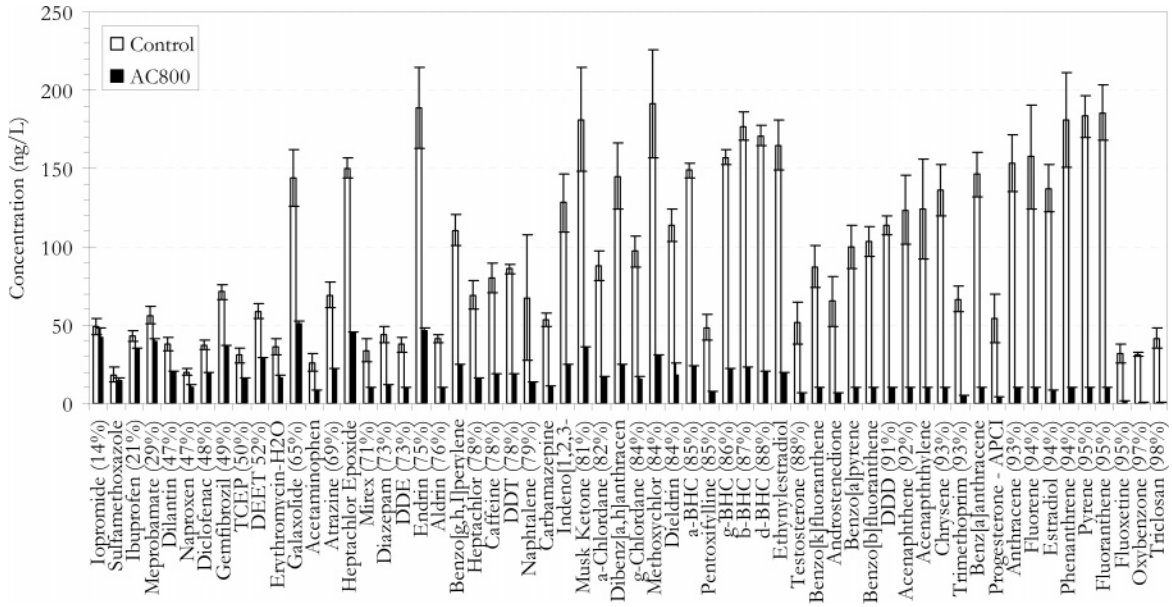

FIGURE 2. EDC/PPCP concentrations in control and PAC-treated samples (ORW; 4-h contact time; AC800). The *x*-axis indicates ranking from lowest removal (left) to highest removal (right). Values in parentheses are percentage removal after AC800 treatment; C_0 is based upon EDC/PPCP spiked concentration in filtered source water. Error bars for the control sample are one standard deviation ($n = 5$) and for the AC800 sample are the difference between the average of two duplicate samples.

of DOC in the source waters may be important for EDC/PPCP removal.

PAC addition resulted in partial removal of nearly all EDC/PPCP compounds spiked into the source waters. Figure 2 shows EDC/PPCP concentrations in control (no PAC) and post-PAC (5 mg/L of AC800; 4-h contact time) for ORW experiments. Compounds listed on the *x*-axis are ranked from lowest percentage removal (left) to highest removal (right). Concentrations in control samples ranged from 18 to 191 ng/L for the different EDC/PPCPs, and from <1 to 50 ng/L after contact with PAC. The relative removal of the EDC/PPCPs by PAC may be indicative of removal in a continuous flow GAC application. For example, diclofenac removal was lower than carbamazepine (Figure 2) (*18*).

While each of the EDC/PPCP spiked concentrations varied slightly among experiments with the four source waters, discussions above related to ^{3}H-E2 suggest that the percentage removal of EDC/PPCPs should be independent of initial concentration. Figure 3 summarizes the average percentage removal for all four waters. The average standard deviation of percentage removal among the four waters was only 10%. However, percentage removals were consistently higher in SRW experiments. This could be due to not using sulfuric acid to preserve samples during SRW experiments (see Analysis section) or associated with differences in organic matter characteristics between the high SUVA of SRW and the lower SUVA of other waters, which affected competition for adsorption sites (Table 2). Increasing SUVA correlates with higher aromatic carbon content, hydrophobicity, and molecular weight of DOC, all of which can affect removal of trace-level contaminants by PAC (*63*). In addition, SRW contained the lowest total EDC/PPCP spiked concentration (1789 ng/L) compared to ORW (6586 ng/L), CRW (5670 ng/L), or PVW (5849 ng/L). Some competition between EDC/PPCPs may occur (*64*).

Over the range of PAC dosages commonly applied in WTPs, increasing PAC dosage improved EDC/PPCP percentage removals. Removals for several compounds are summarized in Table 4. For compounds removed well (>90%) at low PAC dosages, higher PAC dosages lead to relatively low additional removal. For these EDC/PPCPs, the maximum *calculated* percentage removal is ultimately limited by the MRL. For other EDC/PPCPs with final concentrations above the MRL, increasing the PAC dosage significantly increases their removal (e.g., diclofenac, ibuprofen, TCEP, atrazine, DEET). The maximum percentage removal that could be calculated on the basis of the MRL is 99% for most EDC/PPCPs. At the low dosage of 1 mg/L, removal of the steroids ranged between 40% and 75%, while significantly lower removals were observed for aliphatic compounds (TCEP, meprobamate). Dosages of 20 mg/L effectively removed >80% of all compounds. Trends in percentage removals were comparable between the two PAC brands (AC800 and WPM) studied (see Supporting Information).

A-priori knowledge of relative removals of EDC/PPCPs by PAC could aid in understanding the fate of other EDC/PPCPs not directly quantified herein. Quantitative structure–activity relationship (QSAR) models have been developed to predict isotherm parameters for herbicides, pesticides, and other low-molecular-weight, neutral compounds on the basis of molar volumes and hydrogen bonding affinity as key predictive parameters (*33*, *65*, *66*). Equilibrium isotherms for distilled water were not developed in our study, so validation of such models is difficult. Furthermore, QSAR models require

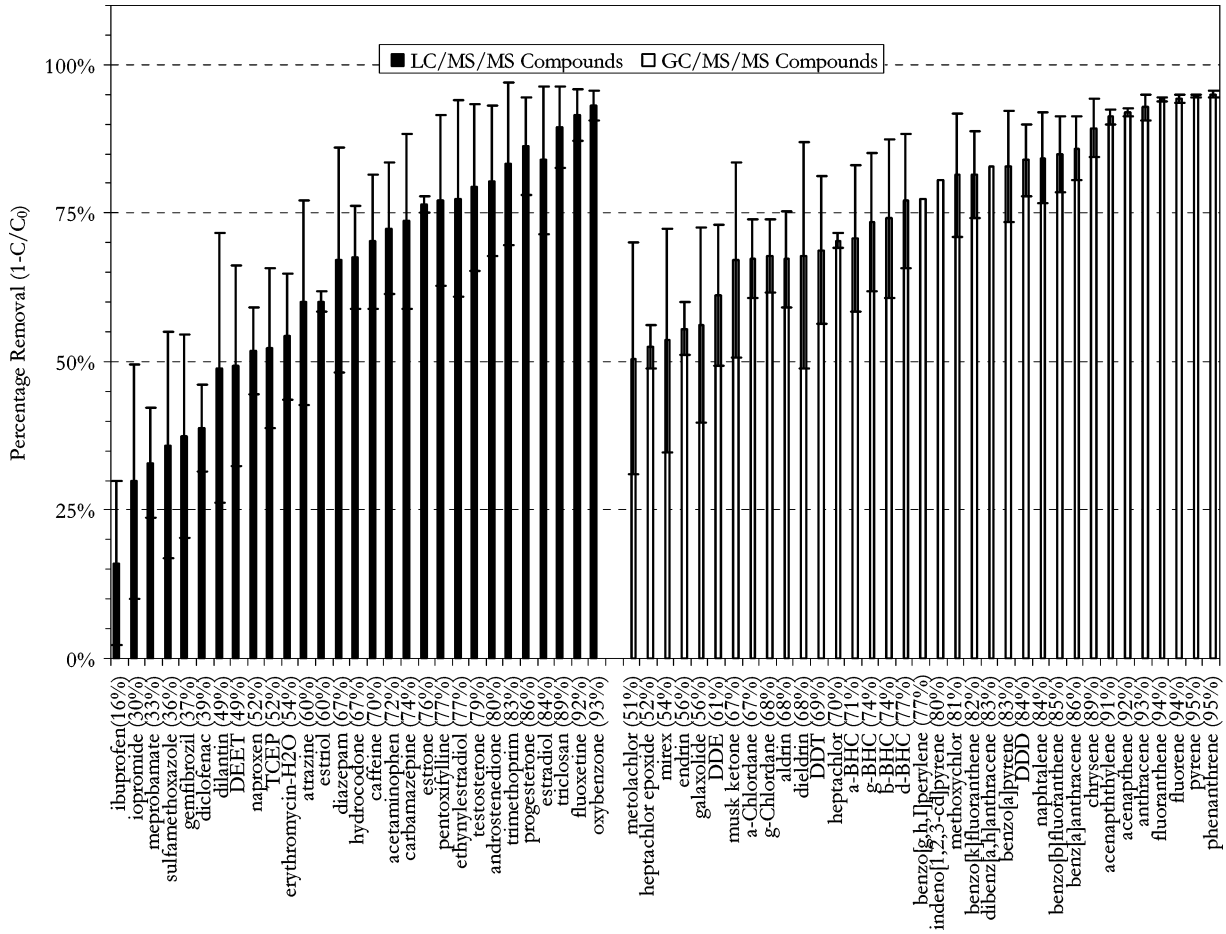

FIGURE 3. Average removal by PAC for SRW, ORW, PVW, and CRW (5 mg/L WPM, 4-h contact time). Values in parentheses are percentage removal after WPM treatment; C_0 is based upon EDC/PPCP spiked concentration in filtered source water. Error bars represent one standard deviation based upon average removal in each of the four water sources.

TABLE 4. Percentage EDC/PPCP Removal in SRW Experiments as a Function of PAC Dose (WPM, 4-h contact time)

compound name	1 mg/L WPM	5 mg/L WPM	20 mg/L WPM
diclofenac	0	44 ± 11	92 ± 1
ibuprofen	2 ± 3	35 ± 15	80 ± 1
meprobamate	8 ± 4	44 ± 4	94 ± 1
TCEP	14 ± 6	69 ± 3	90 ± 1
atrazine	19 ± 8	84 ± 2	99 ± 1
DEET	21 ± 6	73 ± 4	98 ± 1
androstenedione	44 ± 7	96 ± 1	96 ± 1
testosterone	47 ± 5	97 ± 1	99 ± 1
progesterone	47 ± 8	91 ± 1	91 ± 1
ethynylestradiol	55 ± 5	97 ± 1	97 ± 1
estradiol	62 ± 4	97 ± 1	97 ± 1
oxybenzone	75 ± 1	92 ± 1	95 ± 1
fluoxetine	91 ± 3	96 ± 1	96 ± 1

parameters (e.g., hydrogen bonding affinity) that are difficult to obtain for many deprotonated/protonated acid and base compounds, such as many of the LC/MS/MS compounds studied (Table 1). Acknowledging these limitations, observed EDC/PPCP percentage removals in all four waters (5 mg/L WPM; 4-h contact time) were related to octanol−water partition coefficients (log K_{ow}) of the neutral compound (Figure 4). These relationships will be discussed separately for GC/MS/MS and LC/MS/MS compounds.

Average removals for GC/MS/MS compounds were high (>50%) in the four spiked source waters (Figure 3). Under these conditions, a significant correlation did not exist between observed percentage removals and log K_{ow} (Figure 4). However, data from a lower PAC dose (1 mg/L WPM; 4-h contact time) were also evaluated; the percentage removals were lower and not as influenced by MRLs. A weak statistical trend exists ($R^2 = 0.22$) with higher percentage removals at higher log K_{ow} values (see Supporting Information).

The log K_{ow} value predicts reasonably well trends in percentage removals of LC/MS/MS after contact by PAC (5 mg/L WPM; 4-h contact time) (Figure 4); LC/MS/MS compounds not following the general trend are labeled. Excluding these seven outlier compounds, a linear regression analysis yields the following relationship: [percentage removal] = 15 × [log K_{ow}] + 27% ($n = 22$; $R^2 = 0.88$). The two aliphatic compounds (TCEP and meprobamate) follow the trendline. Protonated acids locate below this trendline, whereas protonated bases locate above it. The log K_{ow} value is estimated only for the neutral molecular form. Some heterocyclic/aromatic nitrogen compounds (caffeine, pentoxifylline) also are located above the trendline, while other heterocyclic or aromatic nitrogen compounds do not (e.g., atrazine). Compounds (carbamazepine, diazepam, dilantin, sulfamethoxazole) containing both heterocyclic/aromatic nitrogen and aromatic moieties tend to locate near the predicted trendline. Predictions of log K_{ow} have recently been criticized (*67*, *68*), and the inability to accurately estimate the log K_{ow} of some heterocyclic or aromatic nitrogen-containing compounds (e.g., caffeine, pentoxifylline) may be partially responsible for their behavior not agreeing with the trendline observed for the other LC/MS/MS-analyzed compounds. Overall, protonated bases appear to be well removed by PAC; compounds with low log K_{ow} or deprotonated acid functional groups seem the most difficult to

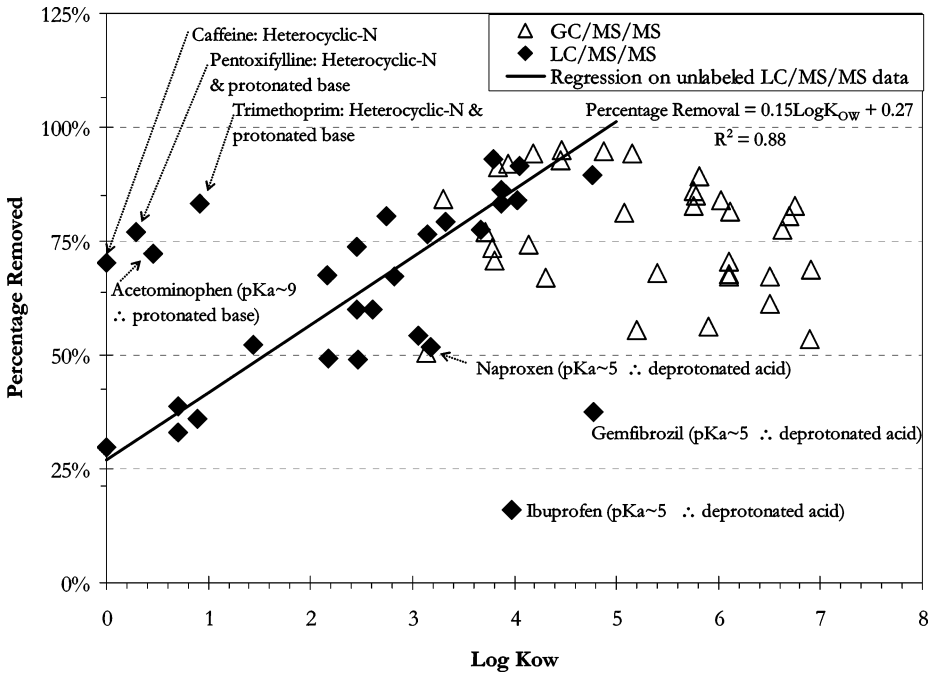

FIGURE 4. Relationship of percentage EDC/PPCP removal as a function of predicted log K_{ow} based upon average removal data from all four source water experiments (5 mg/L WPM; 4-h contact time). Selected LC/MS/MS compounds are identified.

remove with PAC. Future research should conduct single-solute isotherms for a limited number of more polar compounds. Existing QSAR adsorption models require calibration for less volatile neutral compounds and protonated or deprotated acids and bases to predict the performance of PAC in removing many of the newly emerging contaminants.

Oxidation Processes. *Oxidant Dose Selection.* Oxidant dosages and residuals targeted conditions commonly practiced in potable water treatment; these dosages are based upon satisfying microbial disinfection requirements (*25*, *69*). The target free chlorine residual after 24 h was 1 ± 0.1 mg of Cl_2/L for SRW, PVW, and CRW; ORW targeted a lower residual (0.5 mgCl_2/L). The target ozone residual was 0.2−0.3 mg of O_3/L after 3 min of contact time and zero residual within 10 min. Preliminary dose−response experiments (see Supporting Information) were used to select dosages (Table 2) for EDC/PPCP experiments. Higher DOC and higher specific ultraviolet absorbance at 254 nm (SUVA) led to increased oxidant demand to meet target oxidant residuals.

Chlorine Oxidation. For comparison among source waters, one set of experiments was conducted at pH 5.5. Data for PVW and CRW experiments are presented in Figure 5, where ascorbic acid was added to quench residual chlorine after 24-h contact time. Some residual EDC/PPCP concentrations were below detection, indicating a high degree of reactivity with chlorine (e.g., acetaminophen, diclofenac, estradiol, estriol, estrone, ethynylestradiol, naproxen, oxybenzone, sulfamethoxazole, triclosan, and several PAHs). Concentrations of other compounds changed very little after chlorine exposure (e.g., DEET, meprobamate, TCEP, BHC, fluorine, or heptachlor epoxide). Most other compounds exhibited residual concentrations ranging from 25 to 75% of the initial concentrations. Comparable removals were achieved in PVW and CRW experiments. The observed "removals" of parent EDC/PPCP compounds were not artifacts of compound recovery changes brought about by reaction of the oxidants with water matrix constituents (e.g., DOC) because consistent recoveries were observed for stable isotopically labeled surrogates spiked prior to solid-phase extraction.

Ascorbic acid was added to quench chlorine residuals in PVW and CRW experiments but not in ORW or SRW experiments. Thus, for ORW and SRW experiments, EDC/PPCPs were in contact with chlorine until the residual dissipated, samples were acidified, or samples were extracted (less than 7-days total contact time). A few EDC/PPCPs did react more (higher percentage removals) in ORW compared to percentage removals in CRW or PVW experiments. For example, percentage removals without chlorine quenching in ORW were more than 25% higher than removals in PVW or CRW experiments (see removals in Figure 5 for PVW and CRW, and values in the following parentheses are removals in ORW): androstenedione (83%), caffeine (99%), pentoxifylline (98%), progesterone (91%), testosterone (94%), methoxychlor (77%), and phenanthrene (92%). This illustrates two important points. First, some EDC/PPCPs will continue to react within a distribution system in the presence of chlorine after water leaves the WTP as it experiences longer contact times. Second, future field sampling at WTPs or locations where oxidant residuals occur should quench chlorine residuals or take place with the understanding that failure to quench the samples can lead to higher percentage removals than actually occurred at the sampling point.

A few LC/MS/MS compounds (atrazine, DEET, dilantin, fluoxetine, ibuprofen, iopromide, and meprobamate) with removals of <60% exhibited higher removals in PVW than CRW, ORW, and SRW. Figure 5 compares CRW and PVW data. No explanation is readily apparent for the higher PVW removals. Among all four source waters, the following compounds were poorly removed during chlorination: atrazine, DEET, fluoxetine, iopromide, meprobamate, and TCEP.

Experiments were conducted at both a fixed pH 5.5 and ambient pH for all waters. At pH 5.5, hypochlorous acid (HOCl) accounts for nearly 99% of the free chlorine ($pK_{a\ HOCl/OCl^-}$ = 7.5); HOCl is a more powerful oxidant than OCl^- (*70*). Ionized functional groups in the EDC/PPCPs also have significant effects on chlorine reactivity; deprotonated groups generally have second-order rate constants several orders of magnitude greater than those of protonated groups (*40*). In the pH range of our experiments (pH 5.5−8.2), only weak acids would become protonated (pK_a values summarized in Table 1). The percentage EDC/PPCP removals after hypochlorite addition at pH 5.5 were always equal to or higher than those at ambient pH. In most cases, the difference in percentage reacted between ambient pH and pH 5.5 was

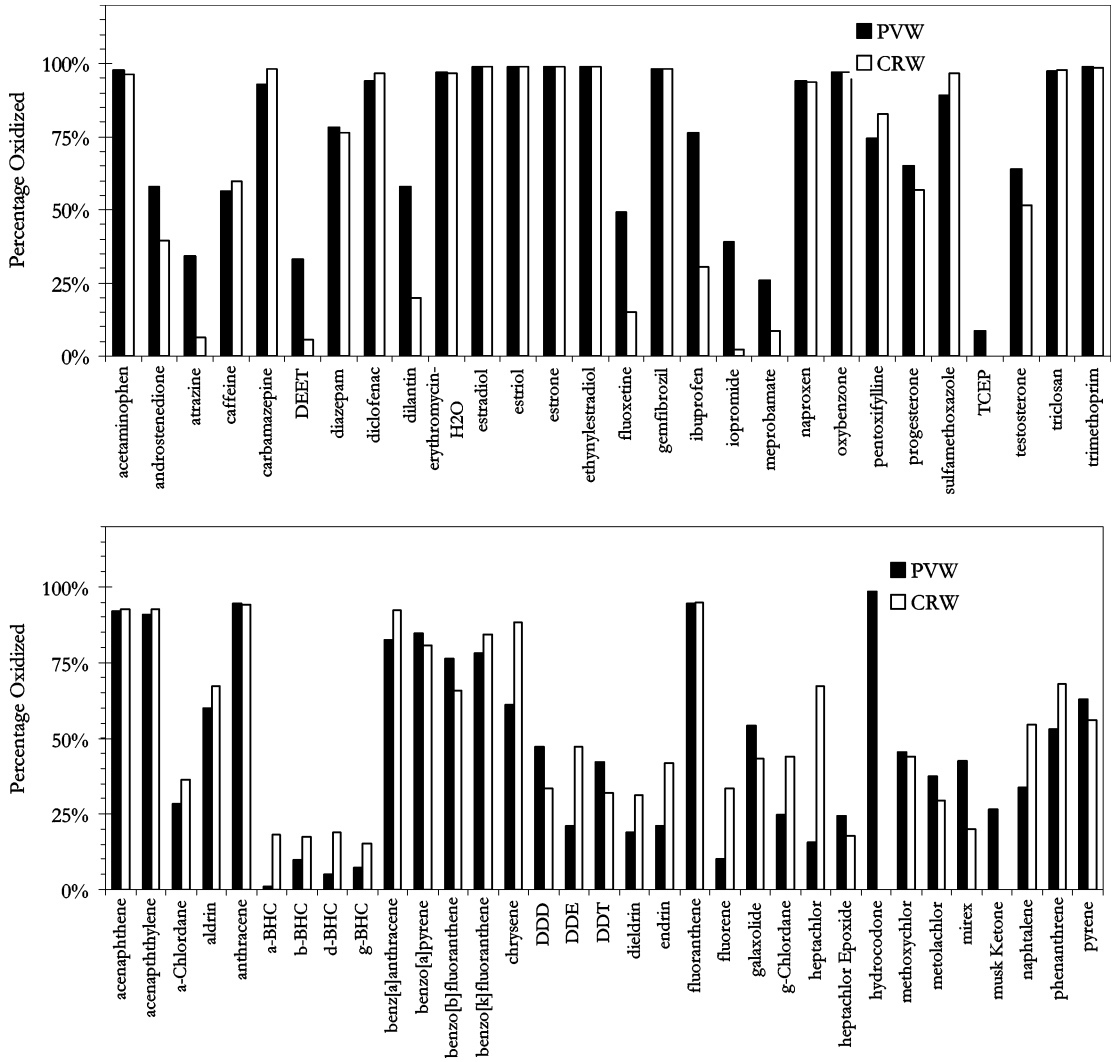

FIGURE 5. Percentage reacted of LC/MS/MS (upper) and GC/MS/MS (lower) compounds in PVW (3.8 mg of Cl_2/L) and CRW (3.5 mg of Cl_2/L) spiked with EDC/PPCPs after hypochlorite addition (pH 5.5). Residuals quenched after 24 h of contact by adding 25 mg/L ascorbic acid. C_0 is based upon EDC/PPCP spiked concentration in filtered source water.

small, less than 20% of the initial concentration. Only the following compounds exhibited greater than 20% additional oxidation at pH 5.5 compared to ambient pH: carbamazepine, caffeine, diazepam, gemfibrozil, methoxychlor, and pentoxifylline. Musk ketone had a much higher percentage oxidized at ambient pH (86% ± 9%) than at pH 5.5 (36% ± 15%). Several well-oxidized compounds (>90% removal) have pK_a values between 5.5 and 8.5 (trimethoprim, sulfamethoxazole, dilantin, triclosan, and erythromycin), but due to the experimental conditions implemented to study a water treatment plant scenario (24-h contact time), differences in their reactivity could not be distinguished. Use of shorter chlorine contact times (e.g., seconds to minutes) would be necessary to observe such differences for these highly reactive compounds.

While PAHs and organochlorine pesticides are EDCs, these compounds were also selected to provide reference points between our work and existing studies. Other studies report half-lives for several PAHs exposed to 2 mg of Cl_2/L (pH 7): 15−30 min (benzo[*a*]pyrene, benzo[*a*]anthracene), 60 min for anthracene, 120 min for pyrene, 400 min for naphthalene (*71*). Our results are consistent with observed reactivity trends for PAHs, indicating that aromatic rings react with chlorine. The high reactivity of chlorine with gemfibrozil may be due to ring activation by the oxy group; our results are consistent with recently published work on the reactivity of chlorine with several aromatic pharmaceuticals (*24*). Hydrocodone is also very reactive (Figure 5), and oxidation may occur at the conjugated carbon bond or amino group. Compounds with primary or secondary amines (diclofenac, sulfamethoxazole, trimethoprim) were also very reactive with chlorine, except for dilantin, in which the amine is part of a heterocyclic ring structure. Consequently, dilantin was only partially oxidized by chlorine (Figure 5). Carbamazepine contains a urea group and aromatic rings, and chlorine oxidized it well. Other very reactive compounds contained phenols (e.g., estradiol, estrone, ethynylestradiol, acetaminophen) or substituted phenols (oxybenzone, triclosan). Bisphenol-a (a plasticizer), another phenolic compound, was very reactive (>90% oxidized) with chlorine (*17*). Triclosan, a chlorine-substituted phenol, is quite reactive with chlorine (*40*). However, chlorine-substituted aromatic rings (e.g., 2,2-bis(4-chloropheny)-1,1-dichloroethane (DDD), 2,2-bis(4-chloropheny)-1,1-dichloroethylene (DDE), 2,2-bis(4-chloropheny)-2,2,2-trichloroethane (DDT), diazepam) were much less reactive with oxidants because the chlorine atom is electron-withdrawing. The least reactive compounds (atrazine, BHC, DEET, fluoxetine, iopromide, meprobamate, and TCEP) either have electron-withdrawing functional groups or lack conjugated carbon bonds entirely.

Ozone Oxidation. Similar trends were observed in the four source waters during ozonation of EDC/PPCPs. Ozon-

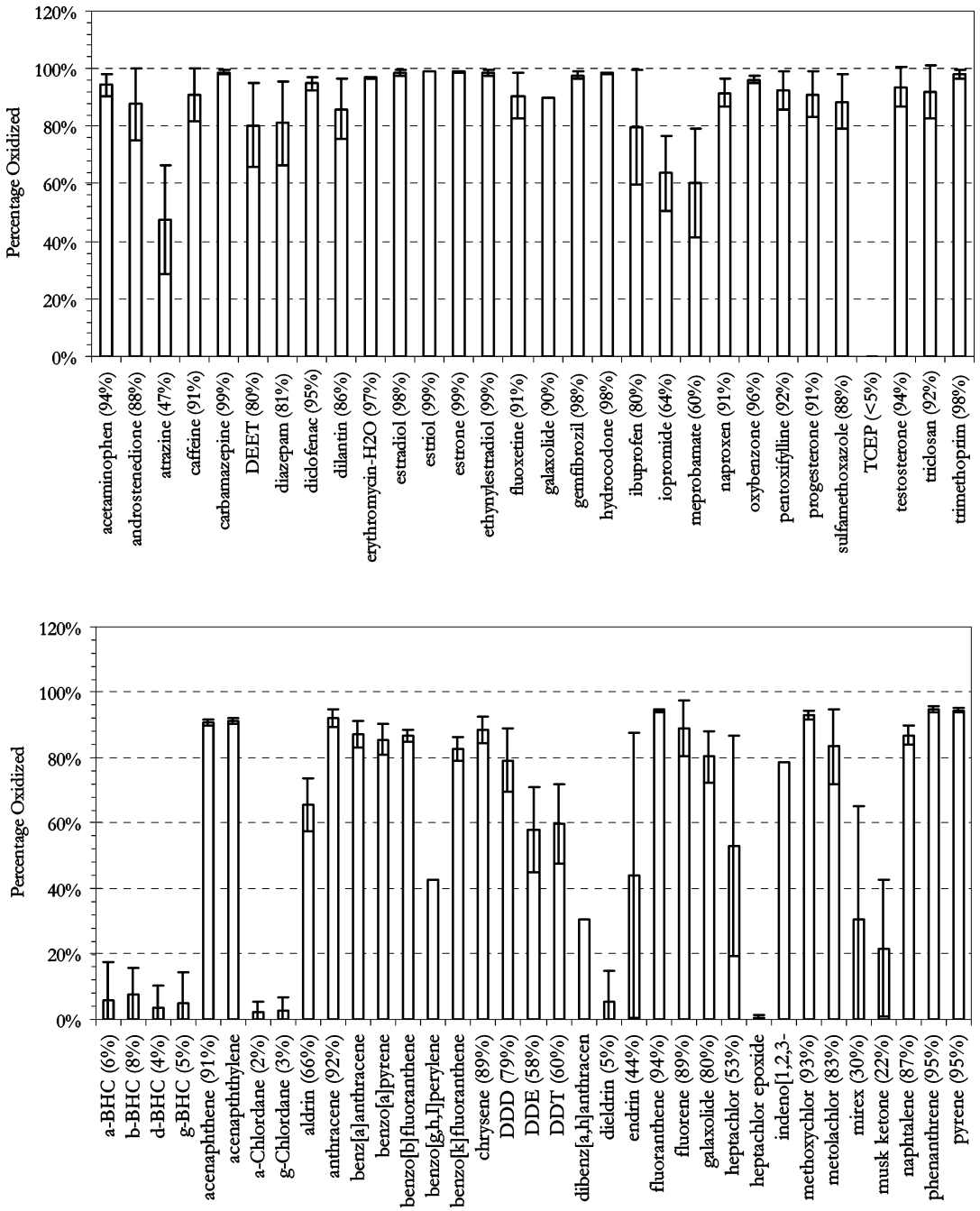

FIGURE 6. Average percentage reacted of LC/MS/MS (upper) and GC/MS/MS (lower) compounds during ozone experiments with PVW, ORW, SRW, and CRW spiked with EDC/PPCPs (ambient pH). Error bars represent one standard deviation in percentage removal. C_0 is based upon EDC/PPCP spiked concentration in filtered source water.

ation experiments were conducted in duplicate, and variation in post-ozonated samples was less than 10%. The percentage removal of trace-level pollutants is often independent of initial pollutant concentration (*17*, *18*, *25*). Figure 6 summarizes the average removals across the four source waters for each compound. Ozone oxidized most LC/MS/MS compounds by >80% except for atrazine, meprobamate, and TCEP, which do not contain aromatic moieties, and ibuprofen which has an electron-withdrawing functional group on an aromatic ring. Several GC/MS/MS compounds exhibited minimal oxidation during ozonation (BHC, chlordane, dieldrin, heptachlor epoxide, mirex, or musk ketone). Ozone oxidized steroids containing phenolic moieties (estradiol, ethynylestradiol, or estrone) more efficiently than those without phenolic moieties (androstenedione, progesterone, and testosterone) (Figure 7). Hydroxyl functional groups

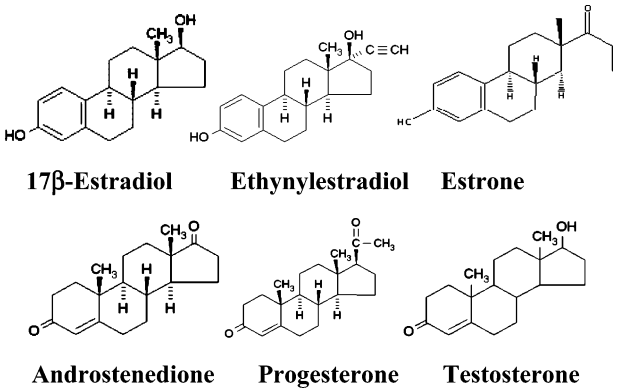

FIGURE 7. Structures of selected hormones.

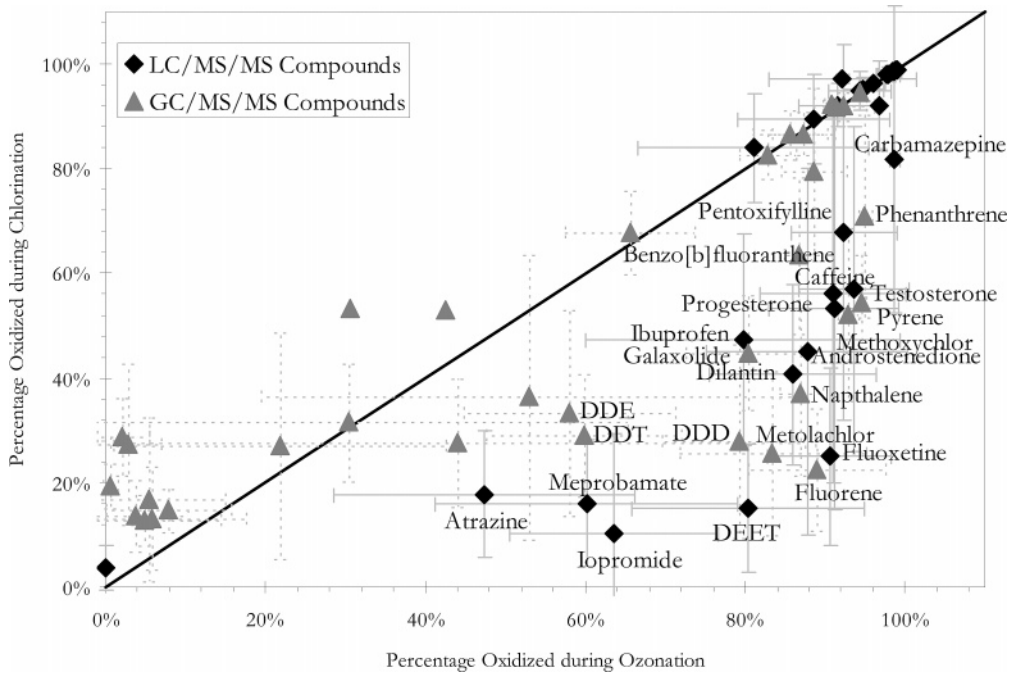

FIGURE 8. Summary of average percentage removal of LC/MS/MS (◆) and GC/MS/MS (▲) compounds by ozone and chlorine across four waters spiked with EDC/PPCPs (PVW, ORW, SRW, CRW). Solid line represents 1:1 removal between ozonation and chlorination experiments. Error bars represent one standard deviation in percentage removal based on experiments in the four waters.

donate electrons to the aromatic rings, resulting in compounds that are more reactive with ozone compared to nonaromatic ring structures or conjugated bonds with carboxyl or ketone functional groups. For a subset of EDC/PPCPs, molecular ozone second-order rate constants are available (see Supporting Information and ref *3*). These vary between <1 and 3×10^6 M^{-1} s^{-1}, while hydroxyl radical rate constants differ by less than 1 order of magnitude. Observed removal trends (Figure 6) agree well with order of magnitude molecular ozone second-order rate constant literature values for the subset of PPCPs and pesticides for which data are available (*21*, *49*). Compounds with higher molecular ozone second-order rate constants in the literature also have higher percentage removals in our study.

Addition of small amounts of H_2O_2 prior to ozonation generally improved by 5−15% the extent of EDC/PPCP oxidation as compared to ozone alone. Four compounds (androstenedione, atrazine, musk ketone, testosterone) exhibited >20% higher oxidation in the presence of H_2O_2. H_2O_2 addition increases the rate of molecular ozone decay (i.e., lower molecular ozone concentrations) but also increases $HO^•$ concentrations. Thus, $HO^•$ probably constitutes a major oxidation pathway for EDC/PPCPs during ozonation. This is consistent with previous work on 8 compounds that are a subset of the ~60 EDC/PPCPs examined herein. That study demonstrated the importance of $HO^•$ reactions and suggested that advanced oxidation is a promising system for efficient removal of pharmaceuticals (*21*).

Comparison of Chlorine and Ozone Oxidation. Figure 8 compares the percentage of initial EDC/PPCP oxidized by chlorine versus ozone. While the oxidant type and dosages differ, the dosages reflect levels that could be applied during potable water treatment. EDC/PPCP removals in Figure 8 are divided into three general categories. The first group contains compounds easily oxidized (>80% reacted) by chlorine and oxidized at least as efficiently by ozone. Such compounds locate on the 1:1 line in Figure 8 at the higher percentage oxidized values. Common structural properties exist for the most highly reacted compounds (>95% reacted), generally including activated aromatic ring structures (i.e., hydroxyl or amine functionalities) and low pK_a values. Deprotonated species react faster with electrophilic ozone because they are stronger nucleophiles. Additional compounds in this group include other substituted (chlorine, methyl, aldehydes) aromatics and PAHs.

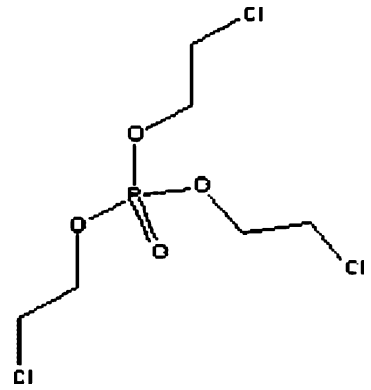

FIGURE 9. Structure of tri(2-chloroethyl) phosphate (TCEP).

A second group includes compounds poorly oxidized (<20% reacted) by either chlorine or ozone. TCEP, the only LC/MS/MS compound in this group, is an example of an aliphatic compound with polar (chlorine) functional groups (Figure 9). The GC/MS/MS compounds in this second group include nonaromatic chlorine-substituted pesticides (BHC, chlordane, dieldrin, and heptachlor epoxide). Non-oxidative treatment processes (e.g., membranes) may be necessary to achieve any measurable removal of these types of compounds.

A third group consists of compounds that react preferentially with ozone rather than chlorine. These compounds are located farther below the 1:1 line in Figure 8. Compound names for this group are labeled in Figure 8. Generally, nucleophilic sites react with both chlorine and ozone. However, for these sites, ozone is still a better oxidant than hypochlorous acid. An additional explanation for higher removal during ozonation is the role of hydroxyl radicals,

which are powerful oxidants that react nonselectively with most organic compounds. Thus, during ozonation, both selective molecular ozone and nonselective HO$^{\bullet}$ oxidation of EDC/PPCPs occurs.

Hammett-based correlations have previous related organic compound structures to their reactivity with common drinking water disinfectants, but such correlations have been limited to single aromatic-ring analogues (*40*). Recently, reactive moieties or functional groups were used to categorize and estimate rate constants when ozone reacts with estrogens and several PPCPs (*21*). Our work extends this concept to include a much broader array of EDC/PPCPs as well as reactivities with both chlorine and ozone. As a consequence, one can identify not only compounds with a high probability of reaction (e.g., amines, substituted phenols, aromatic rings) but also several classes that are unlikely or slow to react, for example, classes of aliphatics (e.g., TCEP, meprobamate). Future work could focus on identifying other compounds with characteristics of those that react slowly. Then, along with understanding the usage of such compounds, one could develop appropriate sampling and analytical schemes to search for them in water systems (*45*).

More sophisticated reactivity−structure linkages use molecular orbital energy modeling to predict tendencies for neutral pesticide oxidation by ozone (*71*). The basic premise is that oxidants react at sites of high electron density, computed as the energy of the highest occupied molecular orbital (ϵ_{HOMO}). However, many models capable of predicting orbital energies or redox states are calibrated for vacuums, not aqueous systems, and they produce questionable results when modeling ionized functional groups. Recent advances in molecular modeling within the pharmaceutical community could greatly enhance the ability to predict the fate of EDC/PPCPs with oxidants. Developing a predictive molecular-level modeling tool will greatly enhance our understanding of the fate of new compounds in the environment and potable water systems.

Oxidation of EDC/PPCPs produces byproducts. Identified oxidation byproducts of atrazine, for example, pose as serious a health risk as the parent compound (*28*, *29*, *70*). In contrast, some byproducts pose less risk than the parent compound. Chlorination and ozonation byproducts of 17β-estradiol and a few other estrogenic compounds were identified and tested for estrogenic response (*17*, *22*). Removal of the parent compound corresponded with reductions in estrogenic activity on the basis of several bio-assays.

Most of our understanding of oxidation-rate determination, byproduct identification, and potential health effects of parent compounds and byproducts comes from empirical laboratory studies. This approach for discovering the behavior of contaminants is time-consuming, expensive, relies upon advanced measurement techniques, and often requires synthesis of byproducts that are not commercially available. With thousands of new chemicals produced each year, perhaps a new paradigm should be embraced that utilizes advanced computing techniques capable of predicting potential oxidation products (e.g., Euler graph theory-based models), with identified products then screened for toxicity using existing toxicity models (*55*, *56*). Studies such as the one included here could be used as data sets to validate the ability of such models to a-priori predict the relative oxidation potential for EDC/PPCPs.

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Supporting Information Available

Figures comparing EDC/PPCP removal using two different PAC brands, relationship between log K_{OW} and GC/MS/MS compounds removed by PAC, and preliminary chlorine and ozone demand curves. Tables summarize select studies on the ability of chlorine and ozone or HO radicals to oxidize organic compounds including many EDC/PPCPs, and molecular weight, common use, and chemical structure for the EDC/PPCPs used in this study. This material is available free of charge via the Internet at http://pubs.acs.org.

Literature Cited

(1) Kolpin, D. W.; Furlong, E. T.; Meyer, M. T.; Thurman, E. M.; Zaugg, S. D.; Barber, L. B.; Buxton, H. T. Pharmaceuticals, hormones, and other organic wastewater contaminants in U.S. streams, 1999−2000: A national reconnaissance. *Environ. Sci. Technol.* **2002**, *36*, 1202−1211.

(2) Ternes, T. A. Occurrence of drugs in German sewage treatment plants and rivers. *Water Res.* **1998**, *32*, 3245−3260.

(3) Snyder, S. A.; Westerhoff, P.; Yoon, Y.; Sedlak, D. L. Pharmaceuticals, personal care products, and endocrine disruptors in water: Implications for water treatment. *Environ. Eng. Sci.* **2003**, *20*, 449−469.

(4) Daughton, C. H.; Ternes, T. A. Special Report: Pharmaceuticals and personal care products in the environment: Agents of subtle change? (Vol. 107, p 907, 1999), *Environ. Health Perspect.* **2000**, *108*, 598−598.

(5) Ongerth, J. E.; Khan, S. Drug Residuals: How xenobiotics can affect water supply sources. *J.−Am. Water Works Assoc.* **2004**, *96*, 94−101.

(6) Weyer, P.; Riley, D. *Endocrine Disruptors and Pharmaceuticals in Drinking Water*; Awwa Research Foundation: Denver, CO, 2001.

(7) Warren, R.; Walker, B.; Nathan, V. R. Environmental factors influencing public health and medicine: Policy implications. *J. Natl. Med. Assoc.* **2002**, *94*, 185−193.

(8) Foster, P. M. D.; McIntyre, B. S. Endocrine active agents: Implications of adverse and nonadverse changes. *Toxicol. Pathol.* **2002**, *30*, 59−65.

(9) Mantovani, A.; Macri, A. Endocrine effects in the hazard assessment of drugs used in animal production. *J. Exp. Clin. Cancer Res.* **2002**, *21*, 445−456.

(10) Ankley, G.; Mihaich, E.; Stahl, R.; Tillitt, D.; Colborn, T.; McMaster, S.; Miller, R.; Bantle, J.; Campbell, P.; Denslow, N.; Dickerson, R.; Folmar, L.; Fry, M.; Giesy, J.; Gray, L. E.; Guiney, P.; Hutchinson, T.; Kennedy, S.; Kramer, V.; LeBlanc, G.; Mayes, M.; Nimrod, A.; Patino, R.; Peterson, R.; Purdy, R.; Ringer, R.; Thomas, P.; Touart, L.; Van der Kraak, G.; Zacharewski, T. Overview of a workshop on screening methods for detecting potential (anti-) estrogenic/androgenic chemicals in wildlife. *Environ. Toxicol. Chem.* **1998**, *17*, 68−87.

(11) Daston, G. P.; Cook, J. C.; Kavlock, R. J. Uncertainties for endocrine disrupters: Our view on progress. *Toxicol. Sci.* **2003**, *74*, 245−252.

(12) Snyder, S. A.; Vanderford, B. J.; Pearson, R. A.; Quinones, O.; Yoon, Y. Analytical methods used to measure endocrine disrupting compounds in water. *Pract. Periodical Hazard., Toxic, Radioact. Waste Manage.* **2003**, *7*, 224−234.

(13) Zhang, T. C.; Emary, S. C. Jar tests for evaluation of atrazine removal at drinking water treatment plants. *Environ. Eng. Sci.* **1999**, *16*, 417−432.

(14) Adams, C.; Wang, Y.; Loftin, K.; Meyer, M. Removal of antibiotics from surface and distilled water in conventional water treatment processes. *J. Environ. Eng.−ASCE* **2002**, *128*, 253−260.

(15) Petrovic, M.; Eljarrat, E.; de Alda, M. J. L.; Barcelo, D. Analysis and environmental levels of endocrine-disrupting compounds in freshwater sediments. *Trac-Trends Anal. Chem.* **2001**, *20*, 637−648.

(16) Ternes, T. A.; Meisenheimer, M.; McDowell, D.; Sacher, F.; Brauch, H. J.; Gulde, B. H.; Preuss, G.; Wilme, U.; Seibert, N. Z. Removal of pharmaceuticals during drinking water treatment. *Environ. Sci. Technol.* **2002**, *36*, 3855−3863.

(17) Alum, A.; Yoon, Y.; Westerhoff, P.; Abbaszadegan, M. Oxidation of bisphenol A, 17B-estradiol, and 17a-ethynyl estradiol and byproduct estrogenicity. *Environ. Toxicol.* **2004**, *19*, 257−264.

(18) Hu, J. Y.; Aizawa, T.; Ookubo, S. Products of aqueous chlorination of bisphenol A and their estrogenic activity. *Environ. Sci. Technol.* **2002**, *36*, 1980−1987.

(19) Hu, J. Y.; Cheng, S. J.; Aizawa, T.; Terao, Y.; Kunikane, S. Products of aqueous chlorination of 17 beta-estradiol and their estrogenic activities. *Environ. Sci. Technol.* **2003**, *37*, 5665−5670.
(20) Huber, M.; Canonica, S.; von Gunten, U. Oxidative treatment of pharmaceuticals in drinking waters, AWWA EDC Workshop, Cincinnati, OH, 2002.
(21) Huber, M. M.; Canonica, S.; Park, G. Y.; Von Gunten, U. Oxidation of pharmaceuticals during ozonation and advanced oxidation processes. *Environ. Sci. Technol.* **2003**, *37*, 1016−1024.
(22) Lee, B. C.; Kamata, M.; Akatsuka, Y.; Takeda, M.; Ohno, K.; Kamei, T.; Magara, Y. Effects of chlorine on the decrease of estrogenic chemicals. *Water Res.* **2004**, *38*, 733−739.
(23) Lopez, A.; Mascolo, G.; Tiravanti, G.; Passino, R. Degradation of herbicides (ametryn and isoproturon) during water disinfection by means of two oxidants (hypochlorite and chlorine dioxide). *Water Sci. Technol.* **1997**, *35*, 129−136.
(24) Pinkston, K. E.; Sedlak, D. L. Transformation of aromatic ether and amine containing pharmaceuticals during chlorine disinfection. *Environ. Sci. Technol.* **2004**, *38*, 4019−4025.
(25) von Gunten, U. Ozonation of drinking water: Part I. Oxidation kinetics and product formation. *Water Res.* **2003**, *37*, 1443−1467.
(26) Zwiener, C.; Frimmel, F. H. Oxidative treatment of pharmaceuticals in water. *Water Res.* **2000**, *34*, 1881−1885.
(27) Huber, M. M.; Ternes, T. A.; von Gunten, U. Removal of estrogenic activity and formation of oxidation products during ozonation of 17 alpha-ethinylestradiol. *Environ. Sci. Technol.* **2004**, *38*, 5177−5186.
(28) Beltran, F. J.; Garcia-Araya, J. F.; Acedo, B. Advanced oxidation of atrazine in water. 1. Ozonation. *Water Res.* **1994**, *28*, 2153−2164.
(29) Beltran, F. J.; Garcia-Araya, J. F.; Rivas, J.; Alvarez, P. M.; Rodriguez, E. Kinetics of simazine advanced oxidation in water. *J. Environ. Sci. Health, Part B* **2000**, *35*, 439−454.
(30) Gould, J. P.; Richards, J. T. The kinetics and products of the chlorination of caffeine in aqueous-solution. *Water Res.* **1984**, *18*, 1001−1009.
(31) Eldib, M. A.; Aly, O. A. Removal of phenylamide pesticides from drinking waters. 1. Effect of chemical coagulation and oxidants. *Water Res.* **1977**, *11*, 611−616.
(32) Montiel, A.; Welte, B. Alternative options for atrazine—Incidence on water-treatment device. *Water Sci. Technol.* **1992**, *25*, 103−110.
(33) Crittenden, J. C.; Sanongraj, S.; Bulloch, J. L.; Hand, D. W.; Rogers, T. N.; Speth, T. F.; Ulmer, M. Correlation of aqueous-phase adsorption isotherms. *Environ. Sci. Technol.* **1999**, *33*, 2926−2933.
(34) Bruce, D.; Westerhoff, P.; Brawley-Chesworth, A. Removal of 2-methylisoborneol and geosmin in surface water treatment plants in Arizona. *J. Water Supply Res. Technol.—AQUA* **2002**, *51*, 183−197.
(35) Dosoretz, C. G.; Boddeker, K. W. Removal of trace organics from water using a pumped bed-membrane bioreactor with powdered activated carbon. *J. Membr. Sci.* **2004**, *239*, 81−90.
(36) Herzberg, M.; Dosoretz, C. G.; Tarre, S.; Michael, B.; Dror, M.; Green, M. Simultaneous removal of atrazine and nitrate using a biological granulated activated carbon (BGAC) reactor. *J. Chem. Technol. Biotechnol.* **2004**, *79*, 626−631.
(37) Qi, S. Y.; Adham, S. S.; Snoeyink, V. L.; Lykins, B. W. Prediction and verification of atrazine adsorption by PAC. *J. Environ. Eng.—ASCE* **1994**, *120*, 202−218.
(38) Boyd, G. R.; Reemtsma, H.; Grimm, D. A.; Mitra, S. Pharmaceuticals and personal care products (PPCPs) in surface and treated waters of Louisiana, USA, and Ontario, Canada. *Sci. Total Environ.* **2003**, *311*, 135−149.
(39) von Gunten, U. Ozonation of drinking water: Part II. Disinfection and byproduct formation in the presence of bromide, iodide or chlorine. *Water Res.* **2003**, *37*, 1469−1487.
(40) Gallard, H.; Von Gunten, U. Chlorination of phenols: Kinetics and formation of chloroform. *Environ. Sci. Technol.* **2002**, *36*, 884−890.
(41) Faust, B. C.; Hoigne, J. Sensitized photooxidation of phenols by fulvic-acid and in natural-waters. *Environ. Sci. Technol.* **1987**, *21*, 957−964.
(42) Acero, J. L.; Von Gunten, U. Characterization of oxidation processes: Ozonation and the AOP O-3/H2O2. *J.—Am. Water Works Assoc.* **2001**, *93*, 90−100.
(43) Huang, C. H.; Sedlak, D. Analysis of estrogenic hormones in wastewater and surface water using ELISA and GC/MS/MS. *Abstr. Pap. Am. Chem. Soc.* **2000**, *219*, 5-ENVR.
(44) Carlile, P.; Fielding, M.; Harding, L.; Hart, J.; Hutchison, J.; Kanda, R. Effect of water treatment processes on oestrogenic chemicals, UK WIR Report 96/DW/05/01, 1996.
(45) Ternes, T. A.; Stuber, J.; Herrmann, N.; McDowell, D.; Ried, A.; Kampmann, M.; Teiser, B. Ozonation: A tool for removal of pharmaceuticals, contrast media and musk fragrances from wastewater? *Water Res.* **2003**, *37*, 1976−1982.
(46) Sacher, F.; Haist-Gulde, B.; Brauch, H.; Preub, G.; Wilme, U.; Zullei-Seibert, N.; Meisenheimer, M.; Welsch, H.; Ternes, T. Behavior of selected pharmaceuticals during drinking water treatment. American Chemical Society Meeting, San Francisco, March, 2000.
(47) Carlson, K. Presented at the Endocrine Disruptors and Pharmaceutically Active Chemicals in Drinking Water Workshop, Chicago, IL, 2000.
(48) Haag, W. R.; Yao, C. C. D. Rate constants for reaction of hydroxyl radicals with several drinking-water contaminants. *Environ. Sci. Technol.* **1992**, *26*, 1005−1013.
(49) Buxton, G. V.; Greenstock, C. L.; Helman, W. P.; Ross, A. B. Critical review of rate constants for reactions of hydrated electrons, hydrogen atoms and hydroxyl radicals (•OH/•O-) in aqueous solution. *J. Phys. Ref. Data* **1988**, *17*, 513−851.
(50) Acero, J. L.; Stemmler, K.; Von Gunten, U. Degradation kinetics of atrazine and its degradation products with ozone and OH radicals: A predictive tool for drinking water treatment. *Environ. Sci. Technol.* **2000**, *34*, 591−597.
(51) Vanderford, B. J.; Pearson, R. A.; Rexing, D. J.; Snyder, S. A. Analysis of endocrine disruptors, pharmaceuticals, and personal care products in water using liquid chromatography/tandem mass spectrometry. *Anal. Chem.* **2003**, *75*, 6265−6274.
(52) Ahrer, W.; Scherwenk, E.; Buchberger, W. Determination of drug residues in water by the combination of liquid chromatography or capillary electrophoresis with electrospray mass spectrometry. *J. Chromatogr., A* **2001**, *910*, 69−78.
(53) USEPA, Enhanced Coagulation and Enhanced Precipitative Softening Guidance Manual (EPA 815-R-99-012), Washington, DC, 1999.
(54) USEPA, Interim Enhanced Surface Water Treatment Rule (EPA 815-F-98-009), Washington, DC, 1998.
(55) Chao, P.; Westerhoff, P. Assessment and optimization of chemical and physicochemical softening processes. *J.—Am. Water Works Assoc.* **2002**, *94*, 109−119.
(56) Yoon, Y.; Westerhoff, P.; Snyder, S.; Esparza, M. HPLC-fluorescence detection and adsorption of bisphenol A, 17b-estradiol, and 17a-ethynyl estradiol on powdered activated carbon. *Water Res.* **2003**, *37*, 3530−3537.
(57) Trenholm, R.; Snyder, S.; Vanderford, B. Gas chromatographic analysis of endocrine disrupting compounds. *Anal. Chem.*, submitted.
(58) Westerhoff, P.; Song, R. G.; Amy, G.; Minear, R. NOM's role in bromine and bromate formation during ozonation. *J.—Am. Water Works Assoc.* **1998**, *90*, 82−94.
(59) Westerhoff, P.; Aiken, G.; Amy, G.; Debroux, J. Relationships between the structure of natural organic matter and its reactivity towards molecular ozone and hydroxyl radicals. *Water Res.* **1999**, *33*, 2265−2276.
(60) Werner, S. L.; Burkhardt, M. R.; DeRusseau, S. N. Methods of Analysis by the U.S. Geological Survey National Water Quality Laboratory: Determination of Pesticides in a Water by CarboPak-B Solid-Phase Extraction and High Performance Liquid Chromatography; U.S. Geological Survey: Reston, VA, 1996.
(61) Snyder, S. A.; Leising, J.; Westerhoff, P.; Yoon, Y.; Mash, H.; Vanderford, B. Biological attenuation of EDCs and PPCPs: Implications for water reuse. *Groundwater Monitor. Remed.* **2004**, *24*, 108−118.
(62) Knappe, D. R. U.; Matsui, Y.; Snoeyink, V. L.; Roche, P.; Prados, M. J.; Bourbigot, M. M. Predicting the capacity of powdered activated carbon for trace organic compounds in natural waters. *Environ. Sci. Technol.* **1998**, *32*, 1694−1698.
(63) Graham, M. R.; Summers, R. S.; Simpson, M. R.; MacLeod, B. W. Modeling equilibrium adsorption of 2-methylisoborneol and geosmin in natural waters. *Water Res.* **2000**, *34*, 2291−2300.
(64) Najm, I. N.; Snoeyink, V. L.; Richard, Y. Effect of initial concentration of a soc in natural-water on its adsorption by activated carbon. *J.—Am. Water Works Assoc.* **1991**, *83*, 57−63.
(65) Blum, D. J. W.; Suffet, I. H.; Duguet, J. P. Quantitative structure−activity relationship using molecular connectivity for the activated carbon adsorption of organic chemicals in water. *Water Res.* **1994**, *28*, 687−699.
(66) Luehrs, D. C.; Hickey, J. P.; Nilsen, P. E.; Godbole, K. A.; Rogers, T. N. Linear solvation energy relationship of the limiting partition coefficient of organic solutes between water and activated carbon. *Environ. Sci. Technol.* **1996**, *30*, 143−152.
(67) Pontolillo, J.; Eganhouse, R. The search for reliable aqueous solubility (Sw) and octanol−water partition coefficient (K_{ow})

data for hydrophobic organic compounds: DDT and DDE as a case study, U.S. Geological Survey, Water-Resources Investigations Report 01-4201, 2001.
(68) Renner, R. The K_{ow} controversy. *Environ. Sci. Technol.* **2002**, *411*, A-413 A.
(69) Summers, R. S.; Hooper, S. M.; Shukairy, H. M.; Solarik, G.; Owen, D. Assessing the DBP yield: Uniform formation conditions. *J.—Am. Water Works Assoc.* **1996**, *88*, 80−93.
(70) Westerhoff, P.; Chao, P. F.; Mash, H. Reactivity of natural organic matter with aqueous chlorine and bromine. *Water Res.* **2004**, *38*, 1502−1513.
(71) Ravacha, C.; Blits, R. The different reaction-mechanisms by which chlorine and chlorine dioxide react with polycyclic aromatic-hydrocarbons (PAH) in water. *Water Res.* **1985**, *19*, 1273−1281.
(72) *SPARC On-Line Calculator* (*http://ibmlc2.chem.uga.edu/sparc/*), 2005.
(73) USEPA *Handbook for Non-Cancer Health Effects Valuation by EPA Science Policy CouncilNon-Cancer: Health Effects Valuation Subcommittee of the EPA Social Science Discussion Group*; USEPA: Washington, DC, 2000.
(74) Snyder, S. A.; Villeneuve, D. L.; Snyder, E. M.; Giesy, J. P. Identification and quantification of estrogen receptor agonists in wastewater effluents. *Environ. Sci. Technol.* **2001**, *35*, 3620−3625.

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