

## Meta-Analysis of Mass Balances Examining Chemical Fate during Wastewater Treatment

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Mass balances are an instructive means for investigating the fate of chemicals during wastewater treatment. In addition to the aqueous-phase removal efficiency ( $\Phi$ ), they can inform on chemical partitioning, transformation, and persistence, as well as on the chemical loading to streams and soils receiving, respectively, treated effluent and digested sewage sludge (biosolids). Release rates computed on a per-capita basis can serve to extrapolate findings to a larger scale. This review examines over a dozen mass balances conducted for various organic wastewater contaminants, including prescription drugs, estrogens, fragrances, antimicrobials, and surfactants of differing sorption potential (hydrophobicity), here expressed as the 1-octanol–water partition coefficient ( $K_{OW}$ ) and the organic carbon normalized sorption coefficient ( $K_{OC}$ ). Major challenges to mass balances are the collection of representative samples and accurate quantification of chemicals in sludge. A meta-analysis of peer-reviewed data identified sorption potential as the principal determinant governing chemical persistence in biosolids. Occurrence data for organic wastewater compounds detected in digested sludge followed a simple nonlinear model that required only  $K_{OW}$  or  $K_{OC}$  as the input and yielded a correlation coefficient of 0.9 in both instances. The model predicted persistence in biosolids for the majority (>50%) of the input load of organic wastewater compounds featuring a  $\log_{10} K_{OW}$  value of greater than 5.2 ( $\log_{10} K_{OC} > 4.4$ ). In contrast, hydrophobicity had no or only limited value for estimating, respectively,  $\Phi$  and the overall persistence of a chemical during conventional wastewater treatment.

### Introduction

Effluent of municipal sewage treatment plants is a well established and constantly scrutinized source of organic wastewater compounds (OWCs) detectable in the environment (1–4). In contrast, the solid end-product of wastewater treatment, i.e., digested municipal sludge or biosolids, is less well understood and continues to draw attention among environmental analytical chemists and risk assessors (5). Gaining additional knowledge concerning the chemical

composition of biosolids is important because use of digested sewage sludge as fertilizer in agriculture, forestry, and landscaping has become common practice following the 1988 ban on ocean dumping. Recent studies concerning the fate of OWCs during sewage treatment show a trend toward the analysis of mass flows in aqueous and solid process streams. In addition to the traditionally reported aqueous-phase removal efficiency ( $\Phi$ ), the resultant mass balances can reveal important information on a given chemical's (i) mass loading, (ii) partitioning behavior, (iii) degradability, and (iv) overall persistence under real-world conditions, as well as (v) mass loading to surface water via effluent discharge and (vi) mass loading to terrestrial environments via land application of biosolids as fertilizer, soil conditioner, or simply for inexpensive disposal of these abundant materials.

Much of the work focusing on the fate of OWCs through wastewater treatment and their corresponding  $\Phi$  values has been performed by monitoring of only the aqueous phase, i.e., chemical concentrations of plant influent and effluent (6, 7). Although such studies are quite valuable, additional information on chemical fate during sewage treatment can be obtained when also considering the volume and composition of sludges. Aiding in these efforts is the increased availability of liquid chromatography tandem mass spectrometry instrumentation featuring improved sensitivity and selectivity, as well as the use of stable isotope-labeled analogues for quantification of target compounds. These tools have significantly enhanced the reliability of qualitative and quantitative information on OWCs in very complex sample matrixes such as sewage sludge. As the number of studies exploring the chemical composition of sludge increases (5, 8), more and more investigators opt to integrate this information into mass balances of OWCs during wastewater treatment.

In this review, over a dozen studies are examined in terms of their approaches to sample acquisition and processing, methods used for chemical detection and quantification, as well as mathematical approaches for closing of mass balances. Study-specific details presented online (Supporting Information) served as the basis for the ensuing critical review. Furthermore, a meta-analysis was conducted of chemical concentrations reported for aqueous and solid process flows. Meta-analysis is a data examination strategy commonly employed to increase the power of statistical analyses by combining and jointly analyzing the results of multiple individual studies. In this review, the term is used more broadly to describe the process of extracting data from multiple studies, combining these data in scatter plots, and

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**TABLE 1. Chemical Abstract Service (CAS) Registry Number and Logarithmic 1-Octanol–Water Partitioning and Organic Carbon Normalized Sorption Coefficients ( $K_{OW}$  and  $K_{OC}$ , respectively) of Organic Wastewater Compounds Examined in This Review**

compound	CAS no.	log $K_{OW}$	log $K_{OC}$
Estrogens			
estrone	53-16-7	3.13 (31)	3.59 (29)
17 $\beta$ -estradiol	50-28-2	4.01 (31)	3.41 (29)
17 $\alpha$ -ethinylestradiol	57-63-6	3.67 (31)	3.53 (29)
Antimicrobials			
triclosan	3380-34-5	4.8 (4)	4.1 <sup>a</sup>
triclocarban	101-20-2	4.9 (4)	4.5 <sup>a</sup>
sulfamethoxazole	723-46-6	0.5 <sup>b</sup>	2.77 (29)
trimethoprim	738-70-5	0.91 (31)	2.7 <sup>c</sup>
clarithromycin	81103-11-9	3.16 (33)	2.8 <sup>c</sup>
ciprofloxacin	85721-33-1	−0.001 <sup>b</sup>	4.23 (29)
norfloxacin	70458-96-7	−0.3 <sup>b</sup>	4.6 <sup>d</sup>
Prescription Drugs			
carbamazepine	298-46-4	2.45 (30)	2.87 (29)
Fragrances			
galaxolide (HHCB)	1222-05-5	5.9 (34)	5.22 (29)
tonalide (AHTN)	21145-77-7	5.7 (34)	5.36 (29)
Surfactants and Industrial Chemicals			
nonylphenol	104-40-5	5.76 (35)	4.52 (32)
perfluorooctanesulfonate (PFOS)	1763-23-1 <sup>e</sup>	6.3 <sup>b</sup>	2.6 (28)
perfluorodecanesulfonate (PFDS)	335-77-3 <sup>e</sup>	8.2 <sup>b</sup>	3.5 (28)
perfluorooctanoate (PFOA)	335-67-1 <sup>e</sup>	6.3 <sup>b</sup>	2.1 (28)
perfluorononanoate (PFNA)	375-95-1 <sup>e</sup>	7.3 <sup>b</sup>	2.4 (28)
perfluorodecanoate (PFDA)	335-76-2 <sup>e</sup>	8.2 <sup>b</sup>	2.8 (28)

<sup>a</sup> Estimated using Advanced Chemistry Development (ACD/Laboratories) Software V8.14 for Solaris for standard conditions of 25 °C and pH 7. <sup>b</sup> Estimated using KOWWIN v1.67. <sup>c</sup> Calculated from ref 11 assuming 40%<sub>(w/w)</sub> organic carbon in activated sludge. <sup>d</sup> Calculated from ref 13 assuming 40%<sub>(w/w)</sub> organic carbon in activated sludge. <sup>e</sup> CAS no. represents the protonated molecule.

performing regression analyses and data fitting to appropriate models. The goal was to identify principal parameters and to define empirical formulas suitable for describing and predicting the fate of OWCs during conventional wastewater treatment. In the following, we examine previous work, present conclusions reached concerning best practices of study design and implementation, and provide empirical relationships helpful in predicting the persistence of hydrophobic organic compounds (HOCs) during municipal wastewater treatment and the extent of sequestration of these chemicals in biosolids.

### Chemicals Monitored in Mass Balances

Organic wastewater contaminants for which mass balances have been conducted can be divided into five groups (Table 1). First are natural and synthetic steroids, including estrone, 17 $\beta$ -estradiol, and the synthetic contraceptive 17 $\alpha$ -ethinylestradiol, whose various fates were tracked during passage through conventional wastewater treatment unit operations (9, 10). Second are antimicrobial compounds including clinical antibiotics and sanitizing agents. Among the antibiotics, mass balances were conducted for the sulfonamide drug sulfamethoxazole (11), for the macrolide compound clarithromycin, (11) and for the dihydrofolate reductase inhibitor trimethoprim (11, 12). Two other studies investigated the mass flow of the fluoroquinolone antibiotics ciprofloxacin and norfloxacin (12, 13). Mass balances also were performed for the sanitizing compounds triclosan (14, 15) and triclocarban (16), two active ingredients of widely used antimicrobial and antibacterial personal care products. The third group is represented by a single prescription drug, the antiepileptic compound carbamazepine, that was monitored along with its major metabolites as part of a mass balance assessment (17). The fourth group consists of two fragrances, namely the synthetic musks galaxolide (HHCB)

and tonalide (AHTN) (18, 19). The fifth and last group comprises a number of surfactants and industrial chemicals. Within this group, one study considered the fate of the degradate nonylphenol to assess the mass flow of alkylphenol polyethoxylates, which represent nonionic surfactants used in many industrial and domestic commodities (20). Other surfactants investigated included various perfluorochemicals belonging to the structural groupings of perfluoroalkyl sulfonates, perfluoroalkyl carboxylates, fluoroalkyl sulfonamides, and fluorotelomer sulfonate (21).

### Occurrence and Fate of Organic Wastewater Compounds Monitored

**Concentrations Found in Influent.** Among the compounds considered in this review, nonyl phenol (20) (up to 119 000 ng/L) and the antimicrobials triclocarban (16) and triclosan (14, 15) (6100 and 4700 ng/L, respectively) showed the highest influent concentrations, whereas natural and synthetic hormones (9, 10) (<5 to 65.7 ng/L) and perfluorinated surfactants (21) (1.1 to 15.7 ng/L) occurred at much lower levels (Table 2). Multiplication of influent concentrations with the corresponding sewage flow rate at the time of sampling yielded the total mass loading for a given compound of interest.

**Effluent Concentrations and Aqueous Removal Efficiencies ( $\Phi$ ).** The concentrations of OWCs in treated effluent are a function of both the initial mass loading and the aqueous phase removal efficiency ( $\Phi$ ) of each individual compound (Table 2). Nonylphenol was the most abundant OWC in effluent (~1400 to 5700 ng/L) and its  $\Phi$  values determined in three different plants ranged from 49 to 95% (20). The next most abundant OWC in effluent was carbamazepine (and its metabolites), which passed through the treatment works unattenuated and exited at a combined level of 1524 ng/L (17). In contrast to their substantial influent concentrations, levels of triclocarban (97%) and triclosan (96–98%) in effluent

**TABLE 2. Concentrations of Compounds Reported in Wastewater Influent, Effluent, and Digested Sludge as Well as Their Corresponding Aqueous-Phase Removal Efficiency ( $\Phi$ )<sup>a</sup>**

compound	reference	influent (ng/L)	effluent (ng/L)	digested sludge ( $\mu\text{g/kg}$ )	$\Phi$ (%)	per-capita mass input ( $\mu\text{g/person/day}$ )
<b>Estrogens</b>						
estrone	9	65.7	<1 <sup>c</sup>	25.2	>99	15
	10	54.8	<0.1 <sup>b,c</sup>	14.3 <sup>d</sup>	100	33
17 $\beta$ -estradiol	9	15.8	<1 <sup>c</sup>	5.1	>94	4
	10	22.0	<0.1 <sup>b,c</sup>	0.57 <sup>d</sup>	100	14
17 $\alpha$ -ethinylestradiol	9	8.2	<1 <sup>c</sup>	<1.5 <sup>c</sup>	>88	2
	10	<5.0 <sup>c</sup>	<0.1 <sup>b,c</sup>	0.61 <sup>d</sup>	>98	NA
<b>Antimicrobials</b>						
triclosan	14	1200	51	1200	96	620
	15	4.700	70	30000	98	2490
triclocarban	16	6100	170	51000	97	2870
sulfamethoxazole	11	1700 <sup>e,h</sup> (1400 <sup>g</sup> )	400 <sup>e,h</sup> (10 <sup>g</sup> )	ND	77	450
trimethoprim	11	290 <sup>e</sup>	70 <sup>e</sup>	<0.1 <sup>c</sup>	76	100
trimethoprim	12	1373 <sup>f</sup>	1424 <sup>f</sup>	ND	−4	500
clarithromycin	11	380 <sup>e</sup>	240 <sup>e</sup>	0.7	37	180
ciprofloxacin	12	220 <sup>f</sup>	48 <sup>f</sup>	5970 <sup>f</sup>	78	480
ciprofloxacin	13	427	71	3100	83	340
norfloxacin	12	293 <sup>f</sup>	58 <sup>f</sup>	6970 <sup>f</sup>	80	610
norfloxacin	13	431	51	2,900 <sup>f</sup>	88	350
<b>Prescription Drugs</b>						
carbamazepine	17	1533 <sup>h</sup> (1001 <sup>i</sup> )	1523 <sup>h</sup> (1081 <sup>i</sup> )	281 <sup>h</sup> (15 <sup>i</sup> )	<1	1050
<b>Fragrances</b>						
galaxolide (HHCB)	(18)	1941	695	3068	64	1030
	(19)	390	173	6788	56	530
tonalide (AHTN)	(18)	583	212	1525	64	310
	(19)	86	42	1349	51	110
<b>Surfactants and Industrial Chemicals</b>						
nonylphenol	20	6800 <sup>i</sup> 8500 <sup>j</sup> ~119000 <sup>i,k</sup>	~3400 <sup>i,k</sup> ~1400 <sup>j,k</sup> 5700 <sup>i</sup>	~24000 <sup>k</sup> ~13000 <sup>k</sup> 898000	49 83 95	NA NA NA
perfluorooctanesulfonate (PFOS)	21	15.7 <sup>i</sup>	24	100	−53	8
perfluorodecanesulfonate (PFDS)	21	6.3 <sup>i</sup>	8.2	91	−30	3
perfluorooctanoate (PFOA)	21	15.0	11	<3	27	8
perfluorononanoate (PFNA)	21	1.1 <sup>i</sup>	3.4	9.9	−209	0.6
perfluorodecanoate (PFDA)	21	5.6	2.3	5.9	59	3

<sup>a</sup> Italics indicate calculated data. <sup>b</sup> After chlorination. <sup>c</sup> Below limit of quantification. <sup>d</sup> From sequential batch reactors (SBRs). <sup>e</sup> Median concentration of two plants. <sup>f</sup> Average concentration calculated from multiple sampling days. <sup>g</sup> *N*-acetylsulfamethoxazole metabolite. <sup>h</sup> Sum of parent compound and metabolites. <sup>i</sup> 10,11-Dihydro-10,11-dihydroxycarbamazepine (CBZ-DiOH) metabolite. <sup>j</sup> Concentrations were averaged over four sampling events. <sup>k</sup> Concentrations were estimated from plots. <sup>l</sup> Calculated by dividing the daily mass loading by the wastewater flow.

were at 170 and  $\leq 70$  ng/L only, due to efficient removal from the aqueous phase during treatment (14, 16). The lowest effluent concentrations were observed for natural and synthetic hormones, whose concentrations dropped below the limit of detection during sewage treatment (<1 ng/L); corresponding  $\Phi$  values ranged from >88 to 100% (9, 10). Effluent concentrations of fluorochemicals ranged from 1.2 to 24 ng/L (21), which was a reflection of low input levels rather than significant removal. Corresponding  $\Phi$  values showed a wide range ( $\leq 59$  to  $-209\%$ ) (21); the phenomenon of negative removal (in-plant production) is explored later in the text.

**Concentrations in Digested Sludge.** The concentration of OWCs present in municipal sludge following digestion is a function of the mass loading to the plant, the compound's partitioning behavior, and the extent of loss by processes including biodegradation. Accumulation of OWCs in the parts-per-million range has been reported for nonylphenol (13 000–898 000  $\mu\text{g/kg}$ ) (20) and for the antimicrobials triclocarban (51 000  $\mu\text{g/kg}$ ) (16), triclosan (1200 and 30 000  $\mu\text{g/kg}$ ) (14, 15), ciprofloxacin (3100 to 5970  $\mu\text{g/kg}$ ), (12, 13) and norfloxacin (2900 to 6967  $\mu\text{g/kg}$ ) (12, 13), as well as for the fragrances galaxolide (3068–108000  $\mu\text{g/kg}$ ), and tonalide

(809–13 500  $\mu\text{g/kg}$ , respectively) (18, 19). In contrast, concentrations of hormones ( $\leq 25.2$   $\mu\text{g/kg}$ ) (9) and perfluorochemicals ( $\leq 100$   $\mu\text{g/kg}$ ) (21) were in the parts-per-billion range only (Table 2).

**Per-Capita Mass Estimates.** Normalizing release rates to a per-capita basis yields a useful metric for extrapolating plant specific observations to the regional or national scale. Per-capita rates were calculated by dividing the daily mass input by the size of the population in the sewershed of the plant, i.e., the geographic area serviced by the WWTP. Calculated loading rates ranged from a high of 2870  $\mu\text{g/person/day}$  for triclocarban (16) to a low of 2  $\mu\text{g/person/day}$  for 17 $\beta$ -estradiol (9) (Table 2).

**Mass Balance Results.** An analysis of mass flow into and out of the WWTPs yields important information on the ultimate fate of OWCs during treatment. Mass flow is determined by taking into account analyte concentrations in influent, effluent, and sludge as well as volumetric flow rates (Supporting Information). Loss of chemicals due to volatilization ideally also should be assessed, at least for volatile compounds. The mass fractions of chemical loading (Table 3) reported to be present in effluent and sludge either

**TABLE 3. Summary of Mass Balance Studies Indicating the Mass Fraction of Individual Compounds Found in Effluent or Digested Sludge or Lost from the System, Relative to the Total Loading (100%) Entering the Plant in Influent**

compound	ref	mass in effluent (%)	mass in processed sludge (%)	mass lost (%)	label in Figure 3
Estrogens					
estrone + 17 $\beta$ -estradiol	9	<2	11	87	1 + 2
	10	12	4	84	3 + 4
17 $\alpha$ -ethinylestradiol	12	<13	<6	>81	5
Antimicrobials					
ciprofloxacin	12	4	77	19	6
	13	12	83	5	7
clarithromycin	12	79	<1	>21	8
norfloxacin	12	3	72	25	9
	13	8	75	17	10
sulfamethoxazole	11	38	<0.2	>62	11
triclocarban	16	3	76	21	12
triclosan	14	4	31	65	13
	15	2	50	48	14
trimethoprim	11	36	<0.2	>64	15
	12	104	NA	-4	
Prescription Drugs					
carbamazepine	17	116 (126 <sup>a</sup> )	0.1 (<0.02 <sup>a</sup> )	-16 (>-26 <sup>a</sup> )	16
Fragrances					
galaxolide (HHCB)	18	36	48	16	17
	19	26 <sup>b</sup>	72	2	18
tonalide (AHTN)	18	37	80	-17	19
	19	30 <sup>b</sup>	67	3	20
Surfactants and Industrial Chemicals					
nonylphenol	20	51.5	43.5	5	21
		17.6	57.4	25	22
		4.8	93.5	1.7	23
perfluorodecanesulfonate (PFDS)	21	129	118	-147	NA
perfluorodecanoate (PFDA)	21	40	9	51	24
perfluorononanoate (PFNA)	21	317	76	-293	NA
perfluorooctanesulfonate (PFOS)	21	143	55	-98	NA
perfluorooctanoate (PFOA)	21	75	0.3	25	25

<sup>a</sup> 10,11-Dihydro-10,11-dihydroxycarbamazepine (CBZ-DiOH) metabolite. <sup>b</sup> Sum of compound mass entering the plant in aqueous and suspended solid phase.

were taken from the literature (10–13, 15, 16, 20) or calculated based on published data (9, 14, 17–19, 21).

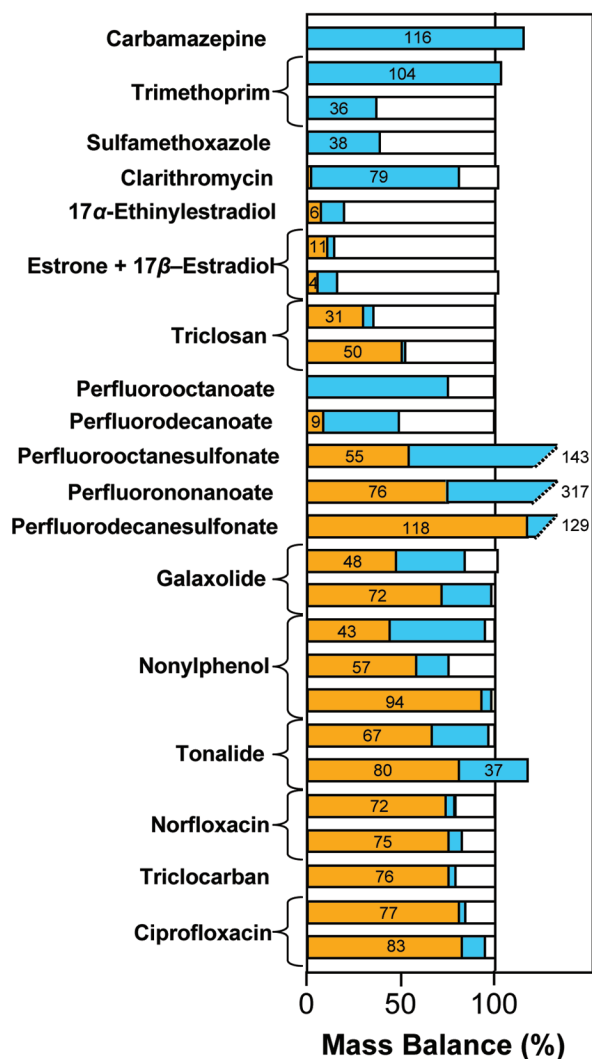
Two studies investigated the fate of estrogens during treatment by combining the naturally occurring hormones estrone and 17 $\beta$ -estradiol in their mass balances (Figure 1; Table 3). Both studies showed a similar reduction in chemical mass by 84 and 87% during treatment (9, 10). Mass fractions found in effluent (<2 vs 12%) and sludge (11 vs 4%) also were similar (9, 10). The only study to track the mass of 17 $\alpha$ -ethinylestradiol during sewage treatment reported >81% degradation or loss, whereas <13% was found in effluent, and only 6% sorbed to sludge (9).

Two studies conducted for triclosan showed consistent results with only 2–4% of the chemical loading passing through the plant in effluent, 30–50% becoming sequestered in sludge and 48–65% being lost by biotransformation and other removal mechanisms (14, 15). Persistence of triclocarban was similar but more pronounced, as 76% of the mass loading was found in sludge (16). Another 3% was found in effluent, and only 21% of the compound mass was lost, presumably because of removal by microorganisms (16). The antibiotics sulfamethoxazole, trimethoprim, and clarithromycin showed similar behavior during wastewater treatment. All three compounds were found in sludge only at trace levels (<1% of the plant loading) (11, 12). While most of the loading

(~63%) of sulfamethoxazole and trimethoprim was degraded or lost (11), only 21% of the clarithromycin input was degraded and a large fraction (79%) of the chemical mass was found in effluent (11). A second study conducted for trimethoprim showed complete lack of removal or degradation, as 104% of the mass input was found in effluent and none in sludge (12).

Mass balances conducted for ciprofloxacin and norfloxacin yielded comparable results. The fractions found in wastewater effluent or lost, respectively, ranged from 3–12% and from 5–25% (12, 13). However, with 72–83%, about three-quarters of the initial mass of both compounds persisted in sludge (12, 13). An investigation of the fate of carbamazepine resulted in a negative mass balance, i.e., a net increase in chemical mass during passage through the plant. With sorption to sludge not occurring to any substantial degree, carbamazepine and its metabolites completely passed through the plant, and 116% of the initial loading was determined to exit the treatment facility in effluent (17). Fractions of the musk fragrances HHCB and AHTN found in effluent were very similar in both studies conducted, with values ranging from 26–37% (18, 19). However, the fraction of chemical mass sorbed to sludge was more variable, with a range of 48–72% and 67–80% for HHCB and AHTN, respectively (18, 19). The fraction of nonylphenol measured





**FIGURE 1.** Compilation of select mass balances for organic wastewater compounds published in the peer-reviewed literature. Shown for each compound are the mass fractions emitted by the plant in effluent (blue), lost to degradation or otherwise unaccounted for (white), and persisting in sludge after digestion of wastewater solids (orange). Compounds are grouped based on structural similarities and sorted according to ascending sequestration and accumulation in biosolids. Cumulative values of >100% indicate a net increase of compound mass during treatment caused by measurement errors, compound formation, or a combination of the two.

in effluent of three plants ranged from 5 to 52% (20), whereas the fraction detected in sludge ranged from 44 to 94%. Loss of this compound during treatment ranged from 2 to 25% (20). One potential explanation for the observed wide range in removal efficiency may be the in-plant formation of nonylphenol from nonylphenol polyethoxylates as an in-plant source term whose magnitude may vary. Many mass balances conducted for fluorochemicals showed a net increase of mass flow during wastewater treatment, thereby resulting in overall positive mass balances. Loss was only observed for PFOA (25%) and PFDA (51%) (21). Mass fractions of fluorochemicals in effluent ranged from 40 to 317% of the influent mass, whereas the fraction found in sludge was generally lower, ranging from zero to 118% (21).

### Study Design Considerations

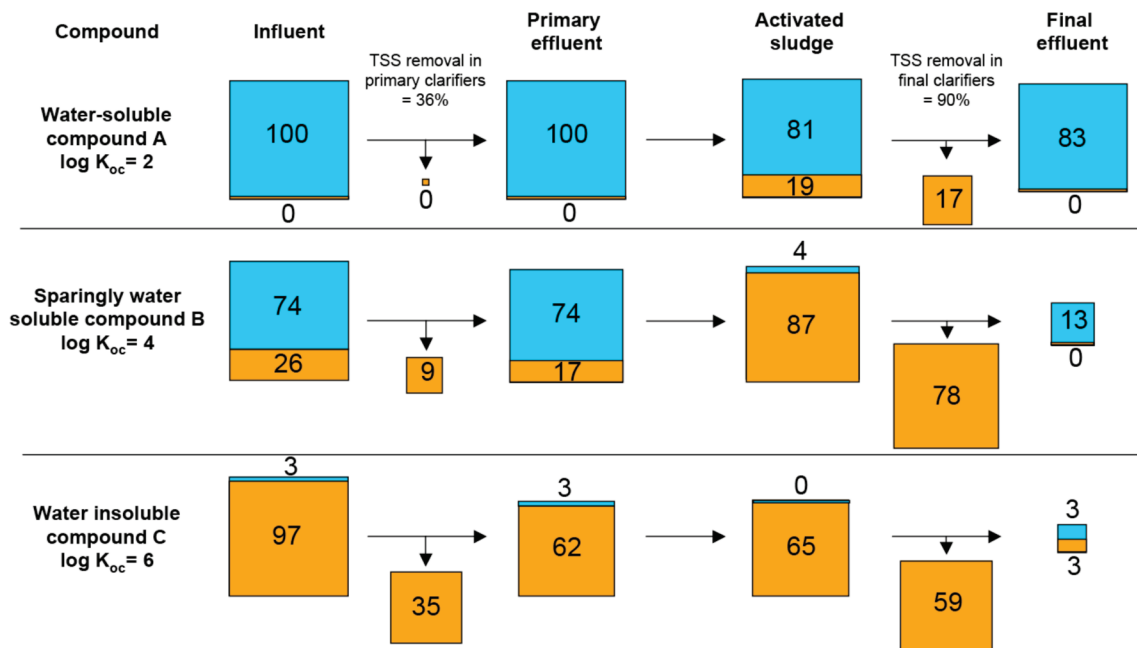
**The Role of Sorption in Wastewater Treatment.** The fates of hydrophobic and hydrophilic compounds during sewage treatment are remarkably distinct. Upon joint entry into a

treatment facility in a parcel of wastewater, both types of compounds quickly are separated and treated by different techniques and to different extents (12). This may best be visualized by considering a conventional activated sludge treatment plant in which degradative processes and compound loss due to various removal mechanisms such as volatilization are absent or insignificant. In this purely hypothetical scenario, sorption of organic wastewater compounds represents the only mechanism influencing the path and removal of OWCs from the aqueous phase during advective transport of chemicals through the plant. It is further assumed that the plant features typical concentrations of total suspended solids (TSS) in influent (119 mg/L), primary effluent (78 mg/L), final effluent (3 mg/L), primary sludge (2000 mg/L), and activated sludge (8000 mg/L), and that 30% of the TSS mass consists of organic carbon. In this situation, nonsorptive, hydrophilic compounds having an organic carbon partition coefficient ( $K_{OC}$ ) of  $\leq 0.33$  pass through the hypothetical plant without any observable change in concentration ( $<0.01\%$ ) because of only insignificant sorption (12). This implies that hydrophilic chemicals effectively bypass the solids treatment train, i.e., sludge digestion and thickeners, and that their average residence time in the plant is identical to the facility's hydraulic residence time.

The schematic in Figure 2 illustrates the influence of sorption due to hydrophobic interactions for three different types of compounds featuring low, intermediate, and high log  $K_{OC}$  values of 2, 4, and 6, respectively. A very hydrophobic organic compound (vHOC; here defined as having a log  $K_{OC}$  of  $\geq 6$ ) entering the plant at a concentration of 10  $\mu\text{g/L}$  is reduced in mass by at least one-third of its initial loading, because of sorption to primary sludge in the primary clarifiers (Figure 2). Thus, a considerable fraction ( $\geq 35\%$ ) of the vHOC loading to the plant never reaches the aerobic treatment step of activated sludge processing and instead is subjected to (typically anaerobic) sludge digestion only (12, 13). For the remaining vHOC mass passing through the clarifier in primary effluent, this process of sequestration into particulates and removal with sludge is repeated during secondary treatment, which typically consists of aerobic degradation followed by sedimentation in secondary clarifiers. In contrast to primary treatment, effective residence times of vHOCs in the aerobic sludge process exceed those of hydrophilic compounds, because of the extensive recycling of solids (Figure 2). Removal of particulates during secondary clarification further depletes the vHOC mass, because of preferential partitioning of the compounds into activated sludge (typical solids content of 0.8–1.2%) (22). This sequential partitioning process effectively renders vHOC concentrations in final effluent at very low levels; even in the absence of any biological, chemical, or physical transformation, the aqueous-phase removal efficiency will be at least 97%. In contrast, the aqueous-phase removal efficiency for moderately hydrophobic (e.g., triclosan, estrogens) (9, 14) and more hydrophilic OWCs (e.g., sulfamethoxazole, trimethoprim, and carbamazepine) (17, 23) featuring log  $K_{OC}$  values of 4 and 2 is on the order of 87 and 17%, respectively, if sorption represents the only removal mechanism (Figure 2).

In summary, hydrophobic and hydrophilic OWCs differ in their susceptibility to physical removal by sorption and sedimentation, which leads to divergences in (i) the path they take during sewage treatment, (ii) the type of treatment they experience, (iii) and their residence times in each unit operation. This implies that, as discussed in the following, different sampling strategies may be beneficial for effectively tracking hydrophobic and hydrophilic contaminants during sewage treatment.

**Selection of Sampling Strategies.** In general, a long sampling duration and a high sampling frequency will increase the representativeness and accuracy of mass balance



**FIGURE 2.** Schematic illustrating the role of sorption in the fate of organic wastewater compounds during their hypothetical passage through a conventional activated sludge wastewater treatment plant assuming a lack of both transformation and loss processes. The partitioning of compounds between the dissolved phase (blue) and wastewater solids (orange) is shown for three organic wastewater compounds featuring logarithmic organic carbon normalized sorption coefficients ( $\log K_{oc}$ ) of 2, 4, and 6 (top, middle, and bottom panels, respectively).

results obtained. For example, a longer sampling duration spanning several months may allow for observing seasonal differences and long-term variability (11). If a sampling campaign is done over a very short period of time, the representativeness of the obtained information potentially is low. Similarly, a low sampling frequency compromises the recognition of interday variation and, if only a few samples are taken over a long period of time, will make it impossible to tell whether seasonal differences observed are real or only a function of normal variation. In contrast, a high sampling frequency, for example several consecutive days, should allow for the detection of mass flow variations occurring between different days of the week (12, 14).

Day-to-day variation is most pronounced when comparing workdays and weekends (22). Seasonal variations can result from snowmelt, drought, and similar weather events (22). For nonsorbing, hydrophilic compounds, sampling of influent and effluent ideally should be performed in sequence, temporally separated by the hydraulic retention time of the treatment facility. For highly sorptive compounds that readily partition between the dissolved and sorbed state, such temporal spacing of sampling events will not provide much benefit, in contrast. In this case, the average retention time of solids in wastewater is of comparatively greater importance. The average residence time of sludge in the digester probably does not figure prominently into sampling design because most digesters are operated as continuously stirred tank systems. This implies that concentrations of chemicals sorbed to sludge will represent an integrated measure of mass flux (and treatment efficiency) over a long period of time (several days to weeks). Thus, short-term deviations in average influent concentrations are unlikely to affect concentrations detectable in digested biosolids.

Finally, the complexity and redundancy of unit operations of modern treatment plants contributes to the challenge of obtaining representative samples. Especially large plants are composed of multiple parallel treatment trains, a design which allows for continued operation during system maintenance. Although system redundancy is not important for

collecting influent and effluent, it does need to be considered when collecting samples at different stages during treatment.

**Considerations for Chemical Analysis.** A second major challenge of mass balances is the difficulty of detecting and quantifying target analytes in difficult matrixes including raw sewage and, more importantly, municipal sludge. Analyte recovery rates can be as low as 6% (see Supporting Information of reference 5) due to extensive and sometimes irreversible sorption. Indeed, suboptimal recovery rates for OWCs in sludge are a pervasive but underreported phenomenon reflective of the superb analytical challenge at hand. The multitude of chemicals and interferences contained in raw sewage pose additional challenges. Poor chromatographic separation and coelution of compounds is rampant and ion suppression and other analytical limitations can yield quantitative data of questionable quality. However, analytical methods continue to evolve and improve over time. For example, the use of isotope dilution techniques in conjunction with highly selective and sensitive instrumentation has greatly enhanced low-level detection capabilities in complex matrixes.

Isotope dilution quantification techniques use stable isotope labeled analogues of analytic targets, typically by relying on the use of carbon-13 ( $^{13}\text{C}$ ) and deuterium ( $^2\text{H}$ ) (10, 11, 15, 16, 21). Isotope-labeled analogues behave essentially identically to the analyte of interest during sample preparation and detection. Therefore, any loss during cleanup can be accounted for by adjusting measured concentrations for the recovery rates of individual standards spiked into the samples prior to sample extraction and processing.

Another notable improvement stems from the use of tandem mass spectrometers capable of detecting analytes using selective reaction monitoring (SRM). This involves the detection and quantification of analytes based on peak areas of compound-specific product ions that serve as reference ions and are generated during fragmentation of precursor masses (typically the intact target compound). This technique helps to achieve practical quantitation limits for organic wastewater contaminants in the parts-per-trillion range even

in complex matrixes, such as raw sewage and sludge (11, 12, 17, 21).

However, because of the high cost of tandem mass spectrometers and limited commercial availability of isotope-labeled standards for OWCs, alternatives such as structural analogues or related compounds see continued use (12, 20). These alternative approaches are rarely suitable for adjusting (normalizing) for analyte losses during sample preparation and detection, because of potentially very different chemical and physical behaviors of the target and its structurally distinct surrogate, however. Other methods of quantification are external calibration (19) and standard addition (17). External calibration does not correct for losses resulting from either sample cleanup or ion-suppression during detection. Standard addition is superior to the latter by accounting for ion suppression but also fails to account for analyte loss occurring during sample cleanup. Therefore, results may represent underestimates of the true concentrations, despite recoveries in excess of 80% (24).

**Considerations for Sample Processing.** During sample preparation of aqueous samples, a frequently applied cleanup step is filtration, which removes any analyte sorbed to particulates. Especially when targeting hydrophobic compounds, this practice can have a significant impact on results for samples featuring an elevated level of total suspended solids. This phenomenon is conceptualized in eq 1,

$$f_{iw} = V / (V + K_D C_{ss}) \quad (1)$$

where  $f_{iw}$  is the fraction of chemical  $i$  dissolved in the aqueous phase;  $V$  is the sample volume (L);  $K_D$  is the partition coefficient (L/kg); and  $C_{ss}$  is the concentration of total suspended solids (kg/L). This equation can serve to illustrate the substantial bias a filtration step may introduce. Let us consider the placement of equal amounts of a compound  $X$  in 1 L of wastewater influent and in 1 L of treated effluent each containing 100 mg/L of  $C_{ss}$ . Assuming a  $K_D$  value of 10 000 L/kg at pH 7 for compound  $X$ , its mass fraction in the aqueous phase following equilibration will be equal to 98% for effluent but only 50% for influent. If the latter volume is subjected to filtration, 50% of the mass of compound  $X$  contained in influent will be removed along with the suspended particulates prior to analysis. This will cause a severe underestimation of both the true concentration of compound  $X$  and the actual mass loading entering the plant, typically calculated as the product of concentration and volumetric flow. To account for this loss, strategies used were to (i) add a surrogate standard prior to filtration (10), (ii) collect separate wastewater samples specifically for particulate analysis (12), and (iii) estimate the fraction sorbed using  $K_D$  (11). Still better is the separation of particulates from the aqueous phase, a separate analysis of solid and liquid fractions, and reporting of the sum of both measurements as the final results (9, 15, 16, 21). The latter approach does not only lead to more reliable estimates of contaminant mass but also informs on the physical phase in which the analytes of interest are arriving at the plant (dissolved vs sorbed). However, one common limitation to consider in this context is the difficulty of reproducibly sampling wastewater slurries that consist of two distinct phases (aqueous and solid) having potentially very different chemical concentrations.

Mass balances in which the amount of analyte leaving the plant exceeds that of the initial loading are not uncommon (Figure 1) (17, 21, 25, 26). Analytical errors aside, possible explanations for this behavior are the formation of a target compound through degradation of precursors, or the de-conjugation of conjugated metabolites during treatment (11, 21). Similarly, a detected net loss in a mass balance could be attributed to the formation of metabolites during treatment, such as the reported formation of methyltriclosan from

triclosan (27). Therefore, the overall accuracy of mass balances can be enhanced by considering potential metabolites in chemical analysis and mass calculations (17). However, in practice this is rarely done because the various transformation products of target compounds frequently require distinctive analytical methods (e.g., gas chromatography as opposed to liquid chromatography or vice versa) and isotope-labeled surrogates of degradates often are not commercially available.

## Meta-Analysis

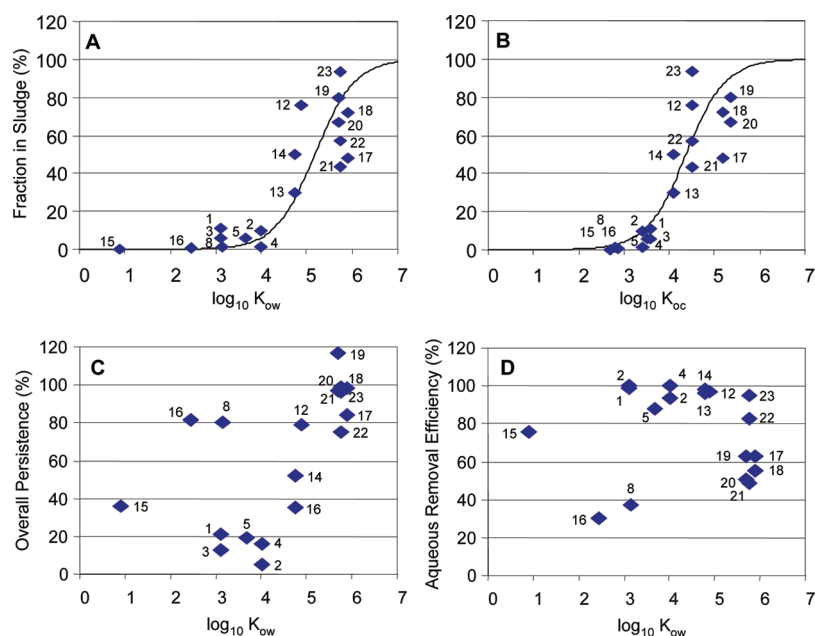
To examine the importance of partitioning of a given compound for its accumulation and persistence in sludge during wastewater treatment, empirical persistence data from available studies were plotted against the 1-octanol–water partition coefficient ( $K_{OW}$ ) of the corresponding analyte and against the organic carbon normalized sorption coefficient ( $K_{OC}$ ) (Figure 3; panels A and B). Values for  $K_{OW}$  and  $K_{OC}$  were taken from the literature or, when unavailable, estimated using EPI Suite KOWWIN v1.67 (see Table 1). Both plots reveal a positive association between hydrophobicity and the accumulation of chemicals in digested sludge. To explore this relationship further, both empirical data sets were fit to a simple model that produced an S-shaped curve (see Supporting Information for additional information). The model, derived as shown in the SI from an equation describing the dissolved and sorbed fractions of a compound in a system (36), took the form

$$f_{\text{digested sludge}} = p_{OC} K / (1 + p_{OC} K) \quad (2)$$

where  $f_{\text{digested sludge}}$  is the mass fraction of a given compound that was found to persist in digested sludge relative to the total mass loading (1.0) arriving at the plant,  $K$  is either the 1-octanol–water partition coefficient ( $K_{OW}$ ) (Figure 3A) or the organic carbon normalized sorption coefficient ( $K_{OC}$ ) (Figure 3B), and  $p_{OC}$  is a fitting parameter (see Supporting Information for further information). The model tracked the empirical data closely (correlation coefficients of 0.9 were obtained in both cases), thereby indicating that compounds with high  $K_{OW}$  and thus a high  $K_{OC}$  are more frequently encountered in sludge than hydrophilic ones featuring lower values. Fluorochemicals and fluoroquinolone antibiotics were excluded from this analysis because of their amphiphilic and zwitter-ionic structure, respectively. The two relationships derived from the fitted data shown in Figures 3A and 3B suggest that >50% of the mass loading of hydrophobic organic compounds featuring a log  $K_{OW}$  value of >5.2 or a log  $K_{OC}$  value of >4.4 will persist during wastewater treatment and accumulate in biosolids. The corresponding values for the dimensionless fitting parameter ( $p_{OC}$ ) were  $6.51 \times 10^{-6}$  and  $4.21 \times 10^{-5}$ , respectively.

It is important to note that the data presented in Figures 3A and 3B do not simply represent sorption behavior: in contrast to the hypothetical situation shown in Figure 2, the data summarized in Figure 3 provide a composite measure of analyte partitioning, accumulation, and persistence in sludge under real-world conditions, including the combined effects of all transformation and removal processes such as biotic and abiotic degradation. Thus, it is concluded that sorptive behavior (parametrized here using either  $K_{OW}$  or  $K_{OC}$ ) is the master variable governing not only partitioning into sludge but also accumulation and persistence of organic pollutants in this matrix during wastewater treatment and solids digestion.

The overall persistence of a compound during wastewater treatment may be defined as the sum of the chemical mass found in sludge and the mass exiting the plant in effluent after treatment, divided by the initial mass loading to the plant. As can be seen from Figure 3C, overall persistence of



**FIGURE 3.** Analysis of mass balances conducted for various organic wastewater compounds (OWCs). The fraction of the mass loading that persisted in digested sludge ( $\diamond$ ) is plotted against the logarithmically transformed 1-octanol–water partition coefficient ( $\log_{10} K_{OW}$ ; panel A) and the organic carbon normalized sorption coefficient ( $\log_{10} K_{OC}$ ; panel B); empirical data were fit to a nonlinear model (S-shaped curve; see text and Supporting Information for details). Panels C shows the relationship between  $K_{OW}$  and the OWC's overall persistence, here defined as the chemical mass contained in both sludge and effluent after treatment divided by the initial loading. Panel D shows the aqueous removal efficiency of various OWCs as a function of their respective  $K_{OW}$  values. Data points are annotated with numbers that link the OWCs to original references listed in Table 3.

a chemical and  $K_{OW}$  are not closely related ( $R$  of 0.59). Overall persistence is expected to be greater for hydrophobic compounds (data in the right half of Figure 3C) that are sequestered into biosolids where biodegradation may be limited because of reduced bioavailability. In contrast, the fate of more hydrophilic compounds is primarily a function of their biodegradability, which can vary widely and presumably is independent of  $K_{OW}$  (scattered data in the left half of Figure 3C).

A plot of the aqueous removal efficiency ( $\Phi$ ) versus mass lost or transformed shows that the disappearance of a substance from the aqueous phase during passage through a wastewater treatment plant cannot inform on whether it actually is degraded or merely becomes sequestered in biosolids (Figure 3D). On the basis of previous discussions, the observed lack of an association between  $K_{OW}$  and  $\Phi$  was expected.

Although the trends identified in Figures 3A and 3B are consistent with the theoretical considerations presented in Figure 2, their quantitative informational value is limited by the scarcity of mass balance data on the fate of OWCs during municipal wastewater treatment. An important caveat of the present analysis is that only a few compounds have been studied to date, and these may not necessarily be representative of the full spectrum of OWCs occurring in wastewater. In other words, observed associations may change somewhat in both trend and significance as the knowledge base increases concerning the behavior of OWCs during municipal sewage treatment. Still, the directionality of the observed associations likely will stand the test of time because of the underlying principles of chemical behavior discussed here and illustrated in Figure 2.

## Concluding Remarks

Mass balances are valuable tools for investigating the fate of chemicals during wastewater treatment. Their effective use requires sound strategies for sampling and sample preparation that take into consideration the chemical and physical

properties of the analytes at hand. Before conducting a mass balance, compounds of interest should be evaluated in terms of their partitioning behavior (using  $K_{OW}$  or  $K_{OC}$  as surrogate parameters) to guide the selection of sampling locations within the liquid and solid treatment train and of the sampling frequency. To ensure sound and scientifically defensible results, filtration of aqueous samples should be avoided if an analysis of the filtrate is not part of the analysis strategy; this is especially important when targeting hydrophobic compounds. If separation of solids is required, a viable option is a centrifugation step followed by extraction of harvested solids in parallel with processing of the dissolved analyte mass for closing of the mass balance. Quantification of compounds in very complex matrixes, such as sludge, ideally should be performed using isotope-dilution techniques in order to obtain the most accurate results possible. Also, the potential in-plant formation of a compound by deconjugation of derivatives should be taken into consideration when conducting mass balances. Finally, results of mass balances ideally should be communicated along with statistics informing on the confidence of the analyses performed.

The compounds included in this study represent only an entry point for future screening efforts that need to be undertaken to comprehensively assess the full spectrum and behavior of OWCs released into the environment by wastewater treatment plants via effluent discharge and biosolids recycling. Future work should include an environmental impact assessment for chemicals that accumulate to parts-per-million concentrations (mg/kg) in digested sludge and for those that show persistence during municipal sewage treatment in both aerobic and anaerobic unit operations. If degradation is observed to be limited or absent, a replacement of these persistent OWCs with more biodegradable and sustainable alternatives should be considered as a precautionary step to prevent potential adverse environmental and human health impacts.



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## Supporting Information Available

Experimental details. This material is available free of charge via the Internet at <http://pubs.acs.org>.

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