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# Fate and Partitioning of Selected Pharmaceuticals in Aquatic Environment

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Two nonsteroidal anti-inflammatory drugs (NSAIDs), ibuprofen and acetaminophen, a β-blocker atenolol, and an antidepressant fluoxetine were selected, and their sorption coefficients ($K_{oc}$ values) on the basis of dissolved organic matter (DOM) and model sediments were determined. The highest values were found for fluoxetine for both DOM and sediments, followed by atenolol or ibuprofen. These $K_{oc}$ values were comparable to those of pyrene and 17β-estradiol, a nonpolar four-ring polycyclic aromatic hydrocarbon and a polar natural estrogen, respectively. For these four pharmaceuticals, partition coefficients between synthetic membrane vesicles (liposomes) and water ($K_{lipw}$ values), and removal efficiencies for a simple batch activated sludge treatment were also determined. The highest $K_{lipw}$ values were again found for fluoxetine followed by atenolol. The removal efficiency for a 6-hour batch activated sludge treatment was over 90% for the two NSAIDs whereas that for atenolol was as low as 10%; both agreed with the results obtained in conventional studies, which showed the concentration in the influent and effluent of sewage treatment plants. The removal efficiency for fluoxetine was also over 90%, but it was sorbed by sludge and not biodegraded.

## 1. Introduction

Recent developments in medical science and technology have resulted in with an increase in the average life span as well as an aging society because of the mass production/consumption of various pharmaceuticals primarily in developed countries. In the late 1990s, Daughton and Ternes[1] first recognized that excreted unaltered pharmaceuticals and/or their metabolites included in domestic/industrial wastewater were discharged into the environ-

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ment and possibly had a large impact on aquatic organisms. Since then, several researchers have attempted to monitor these compounds in surface water and have tested their aquatic ecotoxicology. The concentrations of most pharmaceuticals in rivers and effluent of sewage treatment plants are at most on the order of 100 ng $L^{-1}$,[2–8] and at least three orders of magnitude lower than an acute toxicity level such as the effective concentration 50 ($EC_{50}$) of aquatic organisms.[9–15] Thus, the immediate impact of these pharmaceuticals on aquatic organisms may not be significant. Little is known, however, regarding the fate of these pharmaceuticals once discharged into the aquatic environment in terms of chronic, imperceptible, and cumulative effects on aquatic ecosystem.

In this study, the main focus is on the fate and partitioning of selected pharmaceuticals after excretion from dosed users and disposition into the natural aquatic environment or wastewater treatment plants. The fate and partitioning include sorption by dissolved organic matter (DOM), which may increase transport velocity and decrease bioavailability by binding these pharmaceuticals.[16,17] Sorption by model sediments was determined in addition to partitioning between artificial membrane vesicles (liposomes) and water, which simulates the bioconcentration of chemical compounds better than the octanol-water partition coefficient.[18,19] Finally, a simple batch activated sludge treatment was conducted for the selected pharmaceuticals to obtain basic information regarding removal efficiency in sewage treatment plants or septic tanks, both of which may be the first and last steps to prevent these pharmaceuticals from being released into the natural environment.

Four pharmaceuticals were selected on the basis of their production/consumption, concentration in the aquatic environment, the excretion ratio of unchanged parent compounds, and the ease of quick analysis. The pharmaceuticals selected include two nonsteroidal anti-inflammatory drugs (NSAIDs), ibuprofen and acetaminophen, a β-blocker (antihypertensive) atenolol, and an antidepressant (SSRI, selective serotonin reuptake inhibitor) fluoxetine. A four-ring polycylic aromatic hydrocarbon (PAH) pyrene and a natural estrogen 17β-estradiol were used as reference micropollutants for comparison with the four pharmaceuticals selected.

## 2. Materials and Methods

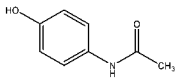
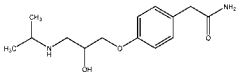
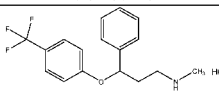
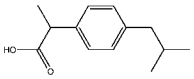
### 2.1 *Materials*

Ibuprofen (98.5%), acetaminophen (97%), and atenolol (98%) were purchased from Wako Pure Chemicals Co. (Osaka, Japan). Fluoxetine hydrochloride was purchased from Sigma Chemical Co. (St. Louis, MO, USA). The chemical structure and physical-chemical characteristics of the selected pharmaceuticals are shown in Table 1. 17β-estradiol (97%) was purchased from Aldrich Chemical Co. (Milwaukee, WI, USA) whereas pyrene was purchased from Wako Pure Chemicals Co. (Osaka, Japan).

Suwannee River natural organic matter (NOM) was purchased from the International Humic Substance Society (IHSS, St. Paul, MN, USA) as model DOM. A reference soil, Elliott Silt Roam Soil (ESRS), was purchased from IHSS and natural river sediment was collected from Tamiya River, a small creek in the western part of Tokushima-city, Japan with a drainage area with no sewage treatment system.

Palmitoyl-oleoyl phosphatidylcholine (POPC), a phospholipid used to make liposomes, was purchased from Nippon Fine Chemical Co. (Osaka, Japan). Activated sludge was

Table 1
Physical-chemical properties of selected pharmaceuticals.

| | acetaminophen | atenolol | fluoxetine | ibuprofen |
|---|---|---|---|---|
| molecular structure |  |  |  |  |
| log $K_{ow}$ | 0.27[a] | -0.03[a] | 4.69[a] (1.57[b]) | 3.79[a] |
| p$K_a$ | 9.5[c] | 9.5[d] | 10.1[b] | 5.7[e] |
| classification | NSAID[f] | β-blocker | Antidepressant (SSRI[g]) | NSAID[f] |
| annual use | 1310 t (2002 Japan)[h] | 19.9 t (2001 Italy)[i] | 22.7 t (2000 USA)[j] | 55.0 t (2002 Japan)[h] |
| excretion ratio of unchanged parent compound | 68%[h] | 90%[i] | 10%[j] | 1-8%[h] |

[a] From ref (20), [b] From ref (13) at *pH* = 7, [c] From ref (21), [d] From ref (22), [e] From ref (23), [f] nonsteroidal anti-inflammatory drug, [g] selective serotonin reuptake inhibitor, [h] From ref (24), [i] From ref (4), [j] From ref (25).

collected from a wastewater treatment plant using the standard activated sludge process with a maximum influent capacity of 22,000 $m^3$.

## 2.2 *Experimental procedure*

The sorption coefficient of the selected pharmaceuticals by DOM ($K_{oc}$ value) was determined using either the fluorescence quenching method[26] or the solubility enhancement method.[27] The detailed experimental procedures for both methods are modified by the authors and described in our previous paper.[28]

The sorption coefficient of sediment samples ($K_d$ value) was determined by measuring the decrease in the aquatic concentration of these pharmaceuticals in 1 h for at least five different solid concentrations (0 to 10,000 mg $L^{-1}$) in 10 mL amber glass centrifuge tubes. The initial concentration of pharmaceuticals was set at 100 μg $L^{-1}$, pH was set at 7 with phosphate buffer, and biocide (0.02 M sodium azide) was added to minimize microbial activity. Because the recoveries of the pharmaceuticals in 1 hour were between 80 and 115%, the effect of biodegradation was neglected. The final aqueous concentrations of the pharmaceuticals were determined by high-performance liquid chromatography (HPLC) equipped with UV-visible and fluorescence detectors. The method is discussed in detail in the next section.

The partition coefficient of the pharamaceuticals between liposomes and water was determined by equilibrium dialysis developed by Escher and Schwarzenbach,[18] later modified by Yamamoto and Liljestrand.[29] A liposome suspension was prepared from POPC by thin film hydration,[30] followed by an extrusion process.[31] Again, the final aqueous concentration of the pharmaceuticals was determined by HPLC.

The batch activated sludge treatment test was conducted in a 300-mL Erlenmeyer flask capped with a silicone plug. Activated sludge was sampled from a wastewater treatment plant using the conventional activated sludge process and used for the batch experiments immediately after sampling. The concentration of the mixed liquor suspended solid (MLSS)

was set at 3,000 mg $L^{-1}$, and the initial concentration of each pharmaceutical was 100 μg $L^{-1}$ in a total volume of 200 mL. The flasks were shaken using a reciprocal shaker in the dark at 130 rpm and 25°C for a typical hydraulic retention time of 6 h. No additional food was added during the batch experiments.

For the batch activated sludge treatment, the supernatant was decanted and methanol (for acetaminophen and atenolol) or acetonitrile (fluoxetine and ibuprofen) was added after the centrifugation of the mixed liquor in 10 mL amber glass centrifuge tubes. After 5 min of sonication, the extract was filtered through a glass fiber filter (GF/F, Whatman), and diluted with Milli Q® water ten times before the analysis by HPLC.

### 2.3 *Analysis*

The aqueous concentration of the pharmaceuticals was determined by HPLC after centrifugation. The HPLC system was composed of an LC-10AD VP series (Shimadzu, Kyoto, Japan) equipped with an ODS column (Shimpack VP-ODS, Shimadzu, Kyoto, Japan) and both fluorescence (RF-10A XL, Shimadzu, Kyoto, Japan) and UV/visible absorbance (SPD-10A VP, Shimadzu, Kyoto, Japan) detectors. The UV/visible absorbance detector was used for acetaminophen at a wavelength of 243 nm whereas the fluorescence detector was used for the other pharmaceuticals at 230 nm (excitation) and 302 nm (emission) for atenolol,[32] 230 nm and 293 nm for fluoxetine,[33] and 224 nm and 290 nm for ibuprofen.[34] A mixture of potassium phosphate buffer (0.008 M, $pH$ = 3.0) and methanol (85:15 volume%) was used as an isocratic mobile phase for acetaminophen and atenolol, whereas a mixture of potassium phosphate buffer (0.008 M, $pH$ = 3.0) and acetonitrile (62:38 volume% for fluoxetine and 35:65 volume% for ibuprofen) was used for the other two. The detection limits of the pharmaceuticals were 0.2, 0.1, 0.5, and 0.5 μg $L^{-1}$ for acetaminophen, atenolol, fluoxetine, and ibuprofen, respectively.

The total organic carbon (TOC) concentration of the membrane vesicle suspension was measured using a TOC analyzer (TOC-5000, Shimadzu, Kyoto, Japan). The organic carbon content of the sediment samples was measured using an NC analyzer (Flash EA 1112, Thermo Electron Co., Waltham, MA, USA).

## 3. Results and Discussion

### 3.1 *Sorption by DOM*

The sorption coefficients of selected pharmaceuticals, pyrene, and 17β-estradiol with Suwannee River NOM ($K_{oc}$ values) are shown in Table 2. Fluorescence quenching was not used for acetaminophen due to its relatively weak fluorescence intensity, whereas solubility enhancement was used for neither fluoxetine nor atenolol due to their relatively large aqueous solubility (i.e., larger than 10 g $L^{-1}$).

As can be observed in Table 2, $K_{oc}$ values ranged from $6.0\times10^2$ to $5.2\times10^4$ (L $kg^{-1}$) for the pharmaceuticals. The highest value was found for fluoxetine followed by atenolol and ibuprofen. The $K_{oc}$ values of these three compounds were comparable to pyrene, a highly hydrophobic four-ring PAH and 17β-estradiol, a natural estrogen. For all the compounds tested with both fluorescence quenching and solubility enhancement, $K_{oc}$ values were higher for the fluorescence quenching. The fluorescence quenching technique tends to overesti-

Table 2
Sorption coefficients of selected pharmaceuticals and reference micropollutants by Suwannee River NOM.

| compound | $K_{oc}$ (L kg C$^{-1}$) [a] | | log $K_{oc}$ [b] | sorbed fraction [b,c] |
|---|---|---|---|---|
| | solubility enhancement | fluorescence quenching | | |
| acetaminophen | $6.0(\pm1.7)\times10^2$ | — | 2.81 | 0.2% |
| atenolol | — | $2.4(\pm0.1)\times10^4$ | 4.38 | 6.4% |
| fluoxetine | — | $5.2(\pm0.3)\times10^4$ | 4.72 | 13.5% |
| ibuprofen | $4.9(\pm0.6)\times10^3$ | $1.6(\pm0.1)\times10^4$ | 4.20 | 4.7% |
| pyrene | $1.4(\pm0.3)\times10^4$ | $4.0(\pm0.2)\times10^4$ | 4.61 | 10.8% |
| 17β-estradiol | $3.7(\pm0.2)\times10^3$ | $9.4(\pm1.4)\times10^3$ | 3.97 | 2.7% |

[a]mean(±standard deviation), [b]based on $K_{oc}$ values obtained using fluorescence quenching except for acetaminophen, [c]Estimated fraction sorbed by DOM in a typical aquatic environment with TOC = 3.0 mg L$^{-1}$

mate sorption coefficient and the trend obtained in this study agrees with the results of conventional studies.[28,35,36]

Furthermore, the log $K_{oc}$ values of the pharmaceuticals showed a poor linear correlation with log $K_{ow}$ values, which contradicted the linear correlation between log $K_{oc}$ and log $K_{ow}$ values for nonpolar hydrophobic compounds such as PAHs shown by several researchers[26,27] but agreed with the results found for polar endocrine disrupting chemicals with diverse molecular structures shown previously by the author.[28] The lack of a linear relationship suggests the relatively small contribution of hydrophobic interaction between DOM and the pharmaceuticals, and the relatively large contribution of other sorption mechanisms such as the interaction between π-electrons, ionic interaction, and hydrogen bonding. Both the selected pharmaceuticals and DOM possess functional groups such as amines, amides, carboxyls, and hydroxyls, which may be ionic under the conditions of a neutral pH (*pH* = 7) used in this study. Further investigations with various compounds, at different pHs with DOM are necessary to specify and reveal the predominant mechanism of sorption.

The fraction sorbed by DOM was also estimated using the equation shown below at the typical TOC concentration (= 3.0 mg C L$^{-1}$) in the aquatic environment.

$$(\text{Fraction sorbed by DOM}) = K_{oc}\,[\text{DOM}] / (1 + K_{oc}\,[\text{DOM}]) \quad (1)$$

where [DOM] is the concentration of DOM (kg C L$^{-1}$). For acetaminophen, the sorbed fraction may be as low as 0.2% and the sorption may not significantly affect the fate of the compound in the aquatic environment. For the other compounds, as high as 13% was sorbed by DOM, which may not significantly affect the fate of the compound in a typical aquatic environment such as river or lake, but may significantly affect the fate of the compound in water with a high TOC concentration such as marshes and wastewater.

### 3.2 *Sorption by sediment*

The sorption coefficients of pharmaceuticals in the sediment samples are shown in Table 3. Coefficients were calculated on the basis of both solid concentration ($K_d$ value) and organic carbon content ($K_{oc}$ value). The relationship between log $K_{ow}$ and log $K_{oc}$ is plotted in Fig. 1

with the conventional correlation obtained for nonpolar organic contaminants such as PAHs.[37] The organic carbon content of Elliott Silt Roam Soil (ESRS) was 2.7%[38] whereas that of the river sediment was 0.79%.

As can be seen in Table 3, $K_d$ values ranged from $5.5\times10^{-1}$ to $2.2\times10^{3}$ (L kg$^{-1}$) for ESRS and $9.1\times10^{-1}$ to $1.3\times10^{2}$ (L kgC$^{-1}$) for the river sediment. For both soil samples, the highest $K_d$ values were found for the SSRI fluoxetine, followed by atenolol, acetaminophen, and ibuprofen. The $K_d$ value of fluoxetine was similar to that of pyrene and that of atenolol was similar to that of 17β-estradiol, whereas $K_d$ values for the other two compounds were one to two orders of magnitude lower than these values. Comparing the two model sediments, $K_{oc}$ values were similar for all six compounds.

No linear correlation was found for the four polar pharmaceuticals as shown in Fig. 1. This lack of correlation contradicts the strong linear relationship found for PAHs and other

Table 3
Sorption coefficients of pharmaceuticals, namely, pyrene and 17β-estradiol in reference soil and river sediment.

| Compound | $K_d$ (L kg$^{-1}$ )/$K_{oc}$ (L kg C$^{-1}$) (ESRS) | $K_d$ (L kg$^{-1}$ )/$K_{oc}$ (L kg C$^{-1}$) (river sediment) | log $K_{oc}$ (ESRS) | log $K_{oc}$ (river sediment) |
|---|---|---|---|---|
| acetaminophen | $7.7(\pm0.9)/2.7(\pm0.0)\times10^{2}$ | $2.6(\pm0.7)/3.3(\pm0.9)\times10^{2}$ | 2.43 | 2.52 |
| atenolol | $1.1(\pm0.3)\times10^{2}/3.7(\pm0.1)\times10^{3}$ | $8.1(\pm0.6)/1.0(\pm0.1)\times10^{3}$ | 3.57 | 3.00 |
| fluoxetine | $2.2(\pm0.2)\times10^{3}/7.4(\pm0.8)\times10^{4}$ | $1.3(\pm0.2)\times10^{2}/1.7(\pm0.2)\times10^{4}$ | 4.87 | 4.23 |
| ibuprofen | $5.5(\pm1.3)\times10^{-1}/1.9(\pm0.5)\times10$ | $9.1(\pm2.9)\times10^{-1}/1.2(\pm0.4)\times10^{2}$ | 1.28 | 2.08 |
| pyrene | $1.7(\pm0.0)\times10^{3}/5.7(\pm0.1)\times10^{4}$ | $7.5(\pm0.4)\times10^{2}/9.5(\pm0.5)\times10^{4}$ | 4.76 | 4.98 |
| 17β-estradiol | $7.1(\pm0.5)\times10/2.4(\pm0.2)\times10^{3}$ | $1.6(\pm0.1)\times10/2.0(\pm0.2)\times10^{3}$ | 3.39 | 3.30 |

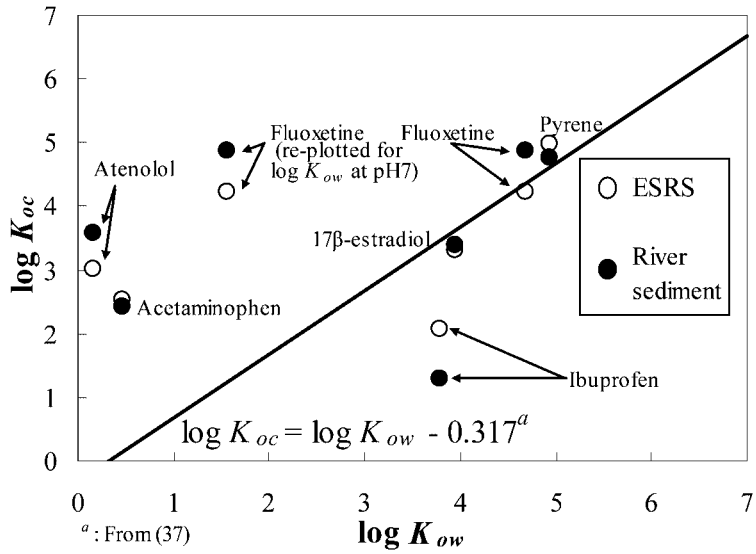



Fig. 1. Relationship between log $K_{ow}$ and log $K_{oc}$.

hydrophobic organic contaminants[38,39] but agrees with the results for steroidal estrogens.[40,41] Furthermore, this lack of linear relationship suggests relatively small contributions from hydrophobic interactions between the organic contents of sediment/soil and pharmaceuticals. In contrast, the log $K_{oc}$ values of both reference compounds, pyrene and 17β-estradiol, were close to the regression line, which suggests relatively large contributions of hydrophobic interactions between the organic contents of sediment/soil and the compounds, although further investigation is necessary for a wide variety of PAHs and steroidal estrogens for the two sediment samples used in this study. Among the four selected pharmaceuticals, only fluoxetine was close to the linear regression line, but the data were plotted for the log $K_{ow}$ value (= 4.69) of the nonionic form, which is mostly protonated, and the log $K_{ow}$ value (= 1.57) is approximately three orders of magnitude lower at pH 7.[13] Thus, the data were re-plotted in Fig. 1. These log $K_{oc}$ values are far above the regression line. The log $K_{oc}$ values of ibuprofen are located far below the regression line. Most protons are dissociated from ibuprofen molecules at pH 7 ($> pK_a = 5.7$), and the log $K_{ow}$ value is likely to be two to three orders of magnitude lower than that of the nonionic form (i.e., 3.67). This prediction results in all four pharmaceuticals being located above the regression line, which suggests a relatively large contribution of other sorption mechanisms in addition to the hydrophobic interaction between the organic contents of sediment/soil and the pharmaceuticals. Other possible mechanisms include ionic bonds between clay minerals and the functional groups of the pharmaceuticals, although further investigation is necessary to determine the sorption coefficient of the selected pharmaceuticals in clay minerals and other components of soil/sediment.

### 3.3 *Partitioning between liposomes and water*

The partitioning coefficients of the selected pharmaceuticals between liposomes and water ($K_{lipw}$ values) and the logarithms of these values were determined and are shown in Table 4. As shown in Table 4, the highest $K_{lipw}$ value was found for the antidepressant fluoxetine, and this value was similar to that of 17β-estradiol followed by ibuprofen. Fluoxetine was detected as high as 2 ng $g^{-1}$ in the brain and liver of fish caught in northern Texas, USA, by Brooks and co-workers,[9] although the concentration detected in American sewage effluent is at most on the order of 0.01 mg $L^{-1}$.[3] This relatively high accumulation suggests a relatively high bioconcentration factor (BCF), although this phenomenon should be confirmed by BCF measurements using aquatic organisms such as fish. The log $K_{lipw}$ values showed a moderate linear relationship with log $K_{ow}$ values for the selected pharmaceuticals but are considered to be better estimates of the BCF of these compounds due to the

Table 4
Liposome-water partition coefficients of pharmaceuticals.

| Compound | $K_{lipw}$ (L $kg^{-1}$) | log $K_{lipw}$ |
|---|---|---|
| acetaminophen | 6.0(±2.8) | 0.78 |
| atenolol | 3.2(±2.8)×10 | 1.50 |
| fluoxetine | 6.2(±0.3)×$10^3$ | 3.79 |
| ibuprofen | 1.4(±0.3)×$10^2$ | 2.15 |
| 17β-estradiol[a] | 6.2(±0.9)×$10^3$ | 3.79 |

[a]: From (29)

use of membrane vesicles made of phospholipid rather than the organic solvent 1-octanol, particularly for those chemicals with ionized functional groups.[18,29]

### 3.4 *Efficiency of removal by activated sludge*

The efficiency of removal, the fraction that remained in the aqueous phase and the fraction sorbed by activated sludge at 6 h are shown in Table 5 with literature values obtained for the concentrations of the influent and effluent of wastewater treatment plants.[2,42] The "unknown" fraction can be considered either as biological or abiotic degradation, because the recovery of the selected pharmaceuticals at 0 h was between 101 and 119%. Assuming no biodegradation and the liposomes as a model of suspended bacteria (i.e., activated sludge), the efficiency of the removal of the pharmaceuticals simply by sorption onto activated sludge was estimated from $K_{lipw}$ values using the following equation:

$$\text{(Estimated removal efficiency)} = K_{lipw}\,[\text{MLSS}] / (1 + K_{lipw}\,[\text{MLSS}]), \qquad (2)$$

where [MLSS] is the mixed liquor suspended solid concentration (= 3000 mg $L^{-1}$) in the batch test. These values are shown in Table 5.

As can be observed in Table 5, the efficiency of the removal of acetaminophen and ibuprofen was as high as 99.5%, which agrees with conventional studies.[2] Although as much as 93% fluoxetine was removed from the aqueous phase, this fraction was not degraded but only sorbed by the activated sludge. As little as 10% was removed from the aqueous phase for a b-blocker atenolol, which agrees with the literature value.[42]

The efficiency of the removal of atenolol and fluoxetine was successfully estimated using the simple assumption of liposomes as suspended bacteria. The estimated fraction sorbed by activated sludge was slightly underestimated for acetaminophen and ibuprofen. This underestimation may be attributed to the effects of an unknown fraction as well as the difference in the physical-chemical properties of activated sludge and liposomes. In this study, the unknown fraction was not identified and mass balance was not verified at 6 h. Further investigation using radiolabeled compounds and discovering the fate of metabolites is absolutely necessary to accurately evaluate the environmental risk of the selected pharmaceuticals.

Table 5
Efficiency of removal of pharmaceuticals by activated sludge at 6 h.
removal efficiency (%).

| Compound | aqueous phase (%) | sorbed fraction (%) | unknown (%) | removal efficiency (%) | | |
|---|---|---|---|---|---|---|
| | | | | experimental[b] | literature | estimated (sorbed)[c] |
| acetaminophen | ND[d] | 2.9 | 96.9< | 99.8< | 98 [e] | 1.8 |
| atenolol | 89.6 | 8.3 | 2.1 | 10.4 | <10 [f] | 8.8 |
| fluoxetine | 6.9 | 93.1 | 0.0 | 93.1 | — | 95 |
| ibuprofen | ND[d] | 37.9 | 61.6< | 99.5< | 90 [e] | 30 |

[a] calculated by subtracting (aqueous phase) and (sorbed fraction) from 100, [b] equals to (sorbed fraction) + (unknown), [c] determined using Equation (2), [d] not detected (under detection limit of 0.2 mg $L^{-1}$ for acetaminophen and 0.5 mg $L^{-1}$ for ibuprofen), [e] From ref (2), [f] From ref (42)

### 3.5 *Possible fate of selected pharmaceuticals*

Acetaminophen was only slightly bioaccumulative as shown by the relatively low $K_{lipw}$ value but probably highly biodegradable as shown by the nearly 100% removal in the batch activated sludge treatment. Thus, most compounds discharged into wastewater are removed by the activated sludge process. In a watershed area with neither wastewater treatment systems nor septic tanks, acetaminophen that is directly discharged into the aquatic environment is probably quickly biodegraded. The accumulation of this compound in either DOM or sediment is unlikely.

Ibuprofen is also highly biodegradable as suggested from the batch activated sludge test but nearly 40% accumulates in activated sludge. This accumulation in activated sludge may cause desorption into the aqueous phase and the efficiency of the removal obtained in this study might be overestimated. Most of this compound discharged into the aqueous environment is probably removed by the activated sludge process if either wastewater treatment systems or septic tanks are available. The accumulation of ibuprofen in sediment is unlikely, but its transport in the aquatic environment is slightly enhanced by DOM.

Fluoxetine has a relatively high bioaccumulation factor and was only sorbed by the activated sludge in the wastewater treatment plant. Because the supernatant of the digested sludge concentrate is recycled into the activated sludge tank, fluoxetine might be accumulated in the wastewater treatment system. Furthermore, as presented herein, fluoxetine has a relatively high sorption coefficient for DOM. The efficiency of its removal in the activated sludge process may be significantly affected by sorption[43–45] and further investigation is necessary. Once discharged in the aquatic environment, fluoxetine is easily accumulated in sediment.

Atenolol was resistant to biodegradation. Ninety percent of this compounds discharged into wastewater is possibly released into the aquatic environment, although the effects of additional treatment such as chlorination are unknown and should be examined. DOM could bind some fractions and enhance their transport in the environment, whereas sediment moderately accumulates this compound. Both the predicated environmental concentration (PEC) and the predicated no-effect concentration (PNEC) of atenolol need to be carefully reevaluated.

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