



# Evaluation of the effect of the intrinsic chemical properties of pharmaceutically active compounds (PhACs) on sorption behaviour in soils and goethite

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## ABSTRACT

The role of the chemical properties of Pharmaceutically Active Compounds (PhACs) in their sorption behaviour and consequently in their fate and mobility is of major environmental interest, but a comprehensive evaluation is still lacking. The sorption of nine PhAC molecules with distinct physico-chemical properties on soils and goethite was described using linear, Freundlich and Langmuir models and the relationship between the chemical structures of the compounds and the parameters of the adsorption was evaluated using redundancy analysis (RDA). The latter showed that the sorption of the pharmaceuticals was determined by the intrinsic chemical characteristics of the molecules, as shown by the 35% value of constrained variability. For the hydrophobic estrogens, E1, E2 and EE2, the logD value and the number of hydrogen bond sites were found to be the main controlling factors for adsorption, indicating that hydrophobic interaction and hydrogen bonding are the dominant sorption mechanisms. The  $\pi$  energy of the molecules also proved a very important parameter, governing the retention of PhACs in soils, especially in the case of carbamazepine, oxazepam and lamotrigine. The main controlling factor for ionic compounds, such as diclofenac sodium, tramadol or lidocaine, is the fraction of PhACs present as charged species, revealing the importance of Coulomb forces. The results of this study will allow semi-quantitative predictions to be made on how the molecular structure governs the sorption of PhACs and which sorption mechanism could be involved.

## 1. Introduction

Nowadays, due to the growing volume of medicines administered to both humans and animals, there is an ever greater need for information about Pharmaceutically Active Compounds (PhACs). This class of compounds includes substances that are widely used in agriculture, medicine and biotechnology, such as antibiotics, drugs and hormones. It is estimated that around 60–80% of PhACs, which amounts to some 100,000 tons or more per year, are not absorbed by animals or humans but are released into the environment (Göbel et al., 2005). These compounds may accumulate in soil and water (Caracciolo et al., 2015; He et al., 2018; Kolpin et al., 2002) and many of them induce serious problems, making them a potential risk to human health and aquatic ecosystems even in low concentrations (Gomes et al., 2017; Klätte et al., 2017). PhACs entry to the environment can be found from improper

disposal of unused drugs, wastewater effluent discharge and even in agricultural runoff (Kusturica et al., 2017; Schlüsener and Bester, 2008). In urban areas, wastewater treatment plants (WWTPs) are considered to be the main contributor to environmental pollution with PhACs, because these WWTPs are not designed to eliminate these molecules (Valdés et al., 2014).

The intrinsic physicochemical properties of PhAC molecules have a great influence on their environmental behaviour, including their sorption characteristics. These molecules are able to interact with the solid phase in several ways (Zhao et al., 2016), so it is crucial to gain more information on how the chemical structure of the sorbent affects the sorption mechanism. Knowledge about this relationship would help to predict the fate and mobility of PhACs in the environment, and would lead to a better understanding of the purification processes taking place in WWTPs. Pharmaceuticals have varied physicochemical behaviour,

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but many exhibit hydrophobicity, which determines their characteristics in the chemical processes taking place on the surface of solid particles (Hari et al., 2005; Kaur et al., 2018). The sorption of hydrophobic, non-ionic molecules, for example, is mainly controlled by hydrophobic (including weak van der Waals) and electron donor-acceptor interactions (Bauerlein et al., 2012; Keiluweit and Kleber, 2009). Hydrogen bonding with hydroxyl groups on the surface has also been found to be an important mechanism for hydrophobic molecules (Tolls, 2001). In the case of hydrophilic, ionic compounds, the main role may be played by interactions with charged surfaces through strong electrostatic mechanisms, for example cation exchange, cation bridging and complex formation (Carmosini and Lee, 2009; Jiang et al., 2015).

However, PhAC molecules contain moieties with very different chemical characteristics (polar, apolar, charged parts), so it is difficult to evaluate the sorption behaviour of these compounds based on a single parameter, such as logP or solubility. The objective of this study was therefore to develop a better understanding of the sorption behaviour of pharmaceuticals on soils and goethite and to clarify which of the interaction mechanisms is dominant and how this is related to pharmaceutical characteristics. This comprehensive study was performed on three types of Hungarian soils and goethite, using nine pharmaceuticals chosen on the basis of their physicochemical properties, which ranged from the hydrophilic, ionic Na-diclofenac to the hydrophobic, neutral estrogens.

The specific objectives of this study were to: (i) to describe the adsorption of nine PhACs on soil and goethite using several adsorption models, (ii) to assess the relationships between the isotherm parameters and the chemical properties of PhACs, and (iii) to gain an insight into the sorption mechanism of PhACs and to evaluate the extent to which chemical properties govern sorption using redundancy analysis (RDA).

## 2. Materials and methods

### 2.1. Chemicals

Nine PhACs, namely carbamazepine (CBZ), diclofenac sodium (DIC), 17 $\alpha$ -ethynylestradiol (EE2), estrone (E1), estradiol (E2), lamotrigine (LAM), lidocaine hydrochloride (LID), oxazepam (OXA) and tramadol hydrochloride (TRA), were selected on the basis of their n-octanol–water partitioning coefficient (logP), water solubility and dissociation constant (pKa). Tramadol HCl and oxazepam were obtained from Lipomed GmbH, while the rest of the PhACs tested were purchased from Sigma-Aldrich. All were of analytical grade purity ( $\geq 99\%$ ). Ultra-pure water with a quality of 0.055  $\mu\text{S}/\text{cm}$  (LaboStar® PRO TWF) was used in all the analytical procedures.

### 2.2. Adsorbents

The soil samples were taken from similar horizons of three soil types (Eutric Arenosol, Mollic Oxygleyic Calcic Gleysol and Eutric Calcic Histosol). The samples were used in a previous study, where they were given the following abbreviations: A\_20 (Arenosol sampled at 20 cm), G\_20 (Gleysol sampled at 20 cm), H\_20 (Histosol, sampled at 20 cm) (Szabó et al., 2020). The commercial goethite standard ( $\alpha$ -FeOOH, Sigma-Aldrich) was also used as a sorbent, because the selected soils contained measurable amounts of this mineral.

The soil samples were air-dried and sieved to  $< 250 \mu\text{m}$  to minimise the effect of particle size variation on sorption. The basic chemical and physical properties were measured using standard laboratory procedures at a constant temperature of 22 °C: soil pH (pH<sub>H2O</sub>, pH<sub>KCl</sub> – Thomas, 1996), CaCO<sub>3</sub> content (Scheibler method, Loeppert and Suarez, 1996), iron content (X-ray fluorescence spectroscopy, Caporale et al., 2018), organic carbon and nitrogen contents (CNS elemental analyser, Thermo Scientific Flashsmart, Matejovic, 1997) and XRD analysis (Flint and Flint, 2002). The specific surface area (SSA) of the sorbents was determined using a BET method (Brunauer-Emmett-Teller equation with

multipoint adsorption isotherms of N<sub>2</sub> at 77 K, Pennell, 2002). The physicochemical properties of the adsorbents are presented in Table S1.

### 2.3. Adsorption experiment and sorption models

The sorption of the nine PhACs was measured in the dark at constant temperature (20 °C) in an incubator (ILW 240 STD) using a batch equilibration method. Preliminary tests showed that sorption reached equilibrium for all nine PhACs within 2 h. The soil/solution ratios were set at 1:12 to obtain enough supernatant for filtration, because at least 10 mL of supernatant is needed to saturate the glass filter. The sediment solutions were equilibrated by shaking in a mechanical round shaker for 2 h. After adsorption, the falcon tubes were centrifuged at 5200 rpm for 15 min (MPW-352RH) and filtered through a 0.45 mm glass filter (Chromafil® GF/PET-45/25).

The following concentrations were used for each PhACs: 100, 500, 1000, 1500, 2000, 2500, 3000, 3500, 4000, 4500 and 5000  $\mu\text{g}/\text{L}$ . Sodium azide was added during the measurements to prevent the degradation of the PhACs. The stock solutions were diluted in methanol, except for diclofenac sodium, which was diluted in ultra-pure water. The limit of detection (LOD) and limit of quantification (LOQ) values are shown in the Supplementary Material (Table S2). The concentrations were measured with HPLC (Shimadzu Prominence LC-20AR), using a PDA and a fluorescence detector. Details of the methods are described in the Supplementary Material (Table S3).

The amount of PhACs adsorbed was calculated using the following equation:

$$q = (C_0 - C_e) \frac{V}{m} \quad (1)$$

where  $q$  is the quantity of PhACs adsorbed ( $\mu\text{g}/\text{g}$ ),  $C_0$  the initial and  $C_e$  the final concentration ( $\mu\text{g}/\text{L}$ ),  $V$  the volume of the initial solution (L), and  $m$  the mass of the soil (g). The adsorption isotherms were analysed with the Freundlich and Langmuir equation models. The non-linear relationship between the sorbed amount and the equilibrium concentration of the solution was defined using the Freundlich equation:

$$q_e = K_f C_e^n \quad (2)$$

where  $q_e$  is the amount of PhACs adsorbed on the soil sample ( $\mu\text{g}/\text{g}$ ),  $K_f$  the Freundlich adsorption coefficient ( $(\mu\text{g}/\text{g})/(\mu\text{g}/\text{L})^n$ ), and  $n$  a dimensionless number which refers to the non-linearity between the adsorbed amount of PhACs and the equilibrium concentration. The Langmuir sorption equation describes interactions between the sorbent and sorbent for a finite number of sorption sites:

$$q_e = Q_{\max} \frac{K_L C_e}{1 + K_L C_e} \quad (3)$$

where  $K_L$  (L/ $\mu\text{g}$ ) and  $Q_{\max}$  ( $\mu\text{g}/\text{g}$ ) are the Langmuir sorption coefficient and the maximum adsorption capacity of the adsorbent, respectively.

### 2.4. Chemical calculations

The octanol–water partition coefficient ( $K_{OW}$ ) can only be used for non-ionisable compounds, where this value is independent of pH. In the present study the distribution coefficient ( $D$ ) was therefore calculated for the nine adsorbents, thus avoiding the over- or underestimation of the hydrophobicity of the PhACs (Jung et al., 2013). The fraction of PhACs present as neutral, negatively or positively charged species ( $\alpha_0$ ,  $\alpha_-$ ,  $\alpha_+$ ), the number of hydrogen bond donor and acceptor sites (nHbd, nHBa),  $\pi$  energy, aromaticity (the percentage of aromatic atoms in the molecule, Ar%) and the van der Waals surface area (vdWSA) were also calculated (Table 1). All the calculations were made with MarvinSketch from ChemAxon (Budapest, Hungary).

**Table 1**  
Chemical properties calculated for the pharmaceuticals.

		logD	$\alpha_0^a$	$\alpha_+^b$	$\alpha^-^c$	nHBd <sup>d</sup>	nHBa <sup>e</sup>	$\pi$ energy	Aromaticity %	Van der Waals surface area
A_20	CBZ	2.77	100	0	0	2	2	29.1	40	312.2
	DIC	0.76	0	0	100	1	6	31.6	40	359.6
	EE2	3.88	97	0	3	1.97	4.03	18.5	13	460.4
	E1	4.30	98	0	2	0.98	4.02	15.6	14	426.8
	E2	3.73	96	0	4	1.96	4.04	16.5	14	436.6
	LAM	1.93	100	0	0	4	5	30.6	52	280.8
	LID	2.80	90	10	0	1.1	2.9	17.6	15	426.0
	OXA	2.92	99	0	1	1.99	5.01	33.6	39	345.8
	TRA	1.93	30	70	0	1.7	4.3	16.5	13	470.5
	G_20	CBZ	2.77	100	0	0	2	2	29.1	40
DIC		0.88	0	0	100	1	6	31.6	40	359.6
EE2		3.90	100	0	0	2	4	18.5	13	460.4
E1		4.31	100	0	0	1	4	15.6	14	426.8
E2		3.74	100	0	0	2	4	16.5	14	436.6
LAM		1.92	99	0	1	4.01	4.99	30.6	52	280.8
LID		2.63	61	39	0	1.39	2.61	14.8	15	430.2
OXA		2.92	100	0	0	2	5	33.6	39	345.8
TRA		1.13	5	95	0	1.95	4.05	16.5	13	470.5
H_20		CBZ	2.77	100	0	0	2	2	29.1	40
	DIC	0.84	0	0	100	1	6	31.6	40	359.6
	EE2	3.89	100	0	0	1.99	4.01	18.5	13	460.4
	E1	4.31	100	0	0	0.99	4.01	15.6	14	426.8
	E2	3.74	100	0	0	1.99	4.01	16.5	14	436.6
	LAM	1.92	100	0	0	4.01	4.99	30.6	52	280.8
	LID	2.69	70	30	0	1.3	2.7	17.6	15	426.0
	OXA	2.92	100	0	0	2	5	33.6	39	345.8
	TRA	1.30	7	93	0	1.93	4.07	16.5	13	470.5
	Goethite	CBZ	2.77	100	0	0	2	2	29.1	40
DIC		4.15	76	0	24	1.77	5.23	33.2	40	360.3
EE2		3.90	100	0	0	2	4	18.5	13	460.4
E1		4.31	100	0	0	1	4	15.6	14	426.8
E2		3.75	100	0	0	2	4	16.5	14	436.6
LAM		-0.43	0	100	0	5	4	31.6	52	285.3
LID		-0.60	0	100	0	2	2	14.8	15	430.2
OXA		2.92	100	0	0	2	5	33.6	39	345.8
TRA		-1.05	0	100	0	2	4	16.5	13	470.5

<sup>a</sup> Fraction of PhAC as neutral form.

<sup>b</sup> Fraction of PhAC as positive form.

<sup>c</sup> Fraction of PhAC as negative form.

<sup>d</sup> The number of hydrogen bond donor sites.

<sup>e</sup> The number of hydrogen bond acceptor sites.

### 2.5. Statistical analysis

Isotherm parameters were estimated by non-linear curve fitting using Origin Pro 2018. The relationship between the adsorption parameters and the intrinsic physico-chemical properties of the PhAC molecules was analysed using redundancy analysis (RDA). The adsorption parameters ( $K_f$ ,  $n$ ,  $K_L$ ,  $Q_{max}$ ) were selected as response variables, while the chemical properties of the PhACs (distribution of neutral, positive and negative species, number of H bonds, logD) were applied as explanatory variables to gain an insight into how the chemical characteristics of the pharmaceuticals influenced adsorption. All the response and exploratory variables were standardised as suggested by Braak and Smilauer (2012). The collinearity of the variables was checked using the variance inflation factor (VIF), and variables having VIF < 20 were used for further analysis.

## 3. Results and discussion

### 3.1. Adsorption of pharmaceuticals on soils and goethite

Batch adsorption experiments were conducted on soil and goethite samples with nine pharmaceuticals (Fig. S1). Considerable differences were found in the extent of adsorption not only between the molecules, but also between the sorbents. A clear trend was noted for the sorption of PhACs on soils, with greater values for hydrophobic compounds (estrogens, CBZ and OXA), while the adsorption of DIC was very limited. In

the case of goethite, the picture was completely different, as DIC was adsorbed to a substantially greater extent than hydrophobic molecules.

The data demonstrated that both the Freundlich and Langmuir models could be used to evaluate the adsorption of the pharmaceuticals over the range of test concentrations (Table 2), as all had  $R^2$  values of over 0.95. The Langmuir model appeared to be more suitable for describing the sorption of PhAC molecules, in contrast to the findings of many authors, e.g. Wu and Bi (2019) and Al-Khazrajy and Boxall (2016), who reported a better fit for the Freundlich adsorption model. Although the Freundlich model generally gives a good description of the sorption equilibrium on heterogeneous surfaces such as soils, the Langmuir isotherm, usually applied in the case of monolayer, homogeneous adsorption, gave a closer fit to the present data, probably due to the relatively low concentrations of PhACs compared to the abundance of adsorption sites on soils (Kumar and Philip, 2006).

The  $Q_{max}$  parameter from the Langmuir equation, which expresses the maximum monomolecular coverage of the surface of the adsorbent, showed great variability for the sorbents with values ranging from 4.5  $\mu\text{g/g}$  to 413  $\mu\text{g/g}$  (Table 2). PhAC molecules bind to the surfaces of soils/sediments and minerals, such as goethite, using several adsorptive interactions. These include Coulomb attraction between the ionic moieties, hydrogen bonding, electron donor-acceptor mechanisms, van der Waals forces ( $\pi$ - $\pi$  interaction), ligand exchange and hydrophobic interactions (Keiluweit and Kleber, 2009). The maximum adsorption capacity is thus equal to the number of sites on the adsorbent to which the given organic compound is able to bind. The  $Q_{max}$  values for the most

**Table 2**  
Sorption parameters of the PhACs on three soils and goethite.

	A <sub>20</sub>			G <sub>20</sub>			Freundlich isotherm			H <sub>20</sub>			Goethite		
	K <sub>f</sub> ((μg/g)/(μg/L) <sup>n</sup> )	n	R <sup>2</sup>	K <sub>f</sub> ((μg/g)/(μg/L) <sup>n</sup> )	n	R <sup>2</sup>	K <sub>f</sub> ((μg/g)/(μg/L) <sup>n</sup> )	n	R <sup>2</sup>	K <sub>f</sub> ((μg/g)/(μg/L) <sup>n</sup> )	n	R <sup>2</sup>	K <sub>f</sub> ((μg/g)/(μg/L) <sup>n</sup> )	n	R <sup>2</sup>
CBZ	0.07 ± 0.00	0.75 ± 0.00	0.998	0.07 ± 0.00	0.75 ± 0.01	0.998	0.08 ± 0.00	0.73 ± 0.00	0.999	0.01 ± 0.000	0.93 ± 0.04	0.980	0.01 ± 0.000	0.93 ± 0.04	0.980
DIC	0.01 ± 0.00	0.66 ± 0.03	0.979	0.06 ± 0.01	0.62 ± 0.01	0.981	0.06 ± 0.00	0.59 ± 0.00	0.979	0.296 ± 0.015	0.46 ± 0.01	0.953	0.296 ± 0.015	0.46 ± 0.01	0.953
EE2	0.31 ± 0.02	0.58 ± 0.01	0.992	0.92 ± 0.03	0.62 ± 0.00	0.998	1.09 ± 0.03	0.59 ± 0.01	0.995	0.028 ± 0.005	0.66 ± 0.02	0.962	0.028 ± 0.005	0.66 ± 0.02	0.962
E2	0.73 ± 0.05	0.29 ± 0.01	0.973	0.72 ± 0.04	0.75 ± 0.01	0.998	2.08 ± 0.17	0.40 ± 0.01	0.963	0.005 ± 0.001	0.86 ± 0.02	0.996	0.005 ± 0.001	0.86 ± 0.02	0.996
E1	0.80 ± 0.05	0.43 ± 0.01	0.970	0.80 ± 0.05	0.43 ± 0.01	0.970	1.99 ± 0.01	0.41 ± 0.00	0.979	0.044 ± 0.004	0.52 ± 0.01	0.963	0.044 ± 0.004	0.52 ± 0.01	0.963
LAM	0.07 ± 0.02	0.56 ± 0.03	0.977	0.14 ± 0.01	0.63 ± 0.01	0.987	0.26 ± 0.04	0.56 ± 0.02	0.988	0.002 ± 0.001	0.90 ± 0.05	0.977	0.002 ± 0.001	0.90 ± 0.05	0.977
LID	0.04 ± 0.02	0.62 ± 0.06	0.969	0.02 ± 0.00	0.85 ± 0.01	0.957	0.41 ± 0.03	0.03 ± 0.01	0.985	0.002 ± 0.001	0.81 ± 0.03	0.952	0.002 ± 0.001	0.81 ± 0.03	0.952
OXA	0.08 ± 0.01	0.68 ± 0.02	0.989	0.40 ± 0.03	0.62 ± 0.01	0.996	0.33 ± 0.01	0.65 ± 0.00	0.997	0.026 ± 0.010	0.67 ± 0.05	0.982	0.026 ± 0.010	0.67 ± 0.05	0.982
TRA	0.46 ± 0.03	0.73 ± 0.02	0.993	0.12 ± 0.00	0.90 ± 0.00	0.999	0.18 ± 0.01	0.73 ± 0.00	0.999	0.003 ± 0.001	0.78 ± 0.03	0.966	0.003 ± 0.001	0.78 ± 0.03	0.966

	Langmuir isotherm			Goethite		
	Q <sub>max</sub> (μg/g)	K <sub>L</sub> (10 <sup>-3</sup> ) (L/μg)	R <sup>2</sup>	Q <sub>max</sub> (μg/g)	K <sub>L</sub> (10 <sup>-3</sup> ) (L/μg)	R <sup>2</sup>
CBZ	63.3 ± 0.9	0.25 ± 0.00	0.999	61.6 ± 0.6	0.29 ± 0.01	0.999
DIC	2.7 ± 0.3	0.58 ± 0.10	0.979	7.7 ± 0.1	0.95 ± 0.01	0.996
EE2	45.8 ± 1.4	0.66 ± 0.03	0.994	78.3 ± 1.6	2.52 ± 0.08	0.998
E1	8.7 ± 0.6	2.49 ± 0.46	0.987	45.8 ± 1.0	3.15 ± 0.17	0.998
E2	28.7 ± 0.5	1.42 ± 0.08	0.987	45.6 ± 0.1	3.25 ± 0.02	0.995
LAM	8.4 ± 0.5	0.64 ± 0.10	0.984	35.4 ± 0.0	0.59 ± 0.06	0.989
LID	12.3 ± 2.6	0.34 ± 0.09	0.960	20.0 ± 0.8	0.94 ± 0.07	0.979
OXA	40.5 ± 2.4	0.29 ± 0.02	0.991	73.4 ± 1.3	0.70 ± 0.01	0.995
TRA	119.9 ± 6.6	1.08 ± 0.05	0.991	96.1 ± 2.2	0.43 ± 0.02	0.994

hydrophobic compounds (EE2, E2, E1, LAM and OXA) were found to be higher (44–160 μg/g) for soils with high organic matter content (Gleysol and Histosol) than for Arenosol or goethite (4.6–45.8 μg/g), indicating that the quantity of organic matter is a crucial factor for adsorbent capacity. Greater organic matter content implies a higher number of adsorption sites, including charged sites with deprotonated –COO<sup>-</sup> and –O<sup>-</sup> groups, C=O groups and aromatic moieties capable of forming π-π interactions, and protonated COOH and –OH groups able to form H bonds. Surprisingly, in the case of CBZ, no differences were found between the soils. This can be attributed to the fact that CBZ has very limited ability to form H bonds, having only 4 H-bonding sites (Table 1).

For fully or partly ionised compounds, such as TRA, DIC, LID or LAM, the number of charged sites controls the maximum adsorption capacity. Very high Q<sub>max</sub> values were estimated for TRA (96.1–347.0 μg/g) in soils with high organic matter content, indicating the high capacity of organic matter to retain positively charged molecules via their –COO<sup>-</sup> groups. At the pH of all the soils, which was above 7, most of the –COOH groups in SOM is dissociated, creating negative charges such as –COO<sup>-</sup> because the pK<sub>a</sub> value of the carboxyl group in the surface soil is about 2.5–5 (Paul et al., 2015). In contrast, the maximum adsorption of DIC on the tested soils was rather low due to its negative charge (Table 2). This is consistent with other studies; Martínez-Hernández et al. (2014), for example, investigated the adsorption of ionisable pharmaceuticals on a sediment with 1.4% organic matter and found that the sorption of cationic species was higher than that of anionic species. Nevertheless, DIC had the greatest Q<sub>max</sub> value on goethite, because most of the adsorption sites on goethite are positively charged due to its PZC value of about 8.3 (Antelo et al., 2005), providing sites that attract negatively charged species such as DIC.

For all the molecules investigated, the n parameter was lower than 1, ranging from 0.29 to 0.93 (Table 2). The n value shows the non-linearity of the curve, which is correlated with the adsorption energy, so the lower the value of n the higher the energy of adsorption (Dada et al., 2012; Dai et al., 2016). Sorption isotherm non-linearity has also been reported for hydrophobic compounds. Sangsupan et al. (2006), for example, found the n value to be 0.88–0.92 for estradiol in agricultural soils, while Li et al. (2013) measured 0.36 for E2 and 0.77 for EE2. In a study where CBZ was adsorbed onto soils, values of 0.87 and 0.89 were recorded (Fenet et al., 2012). The study of Stein et al. (2008) clearly demonstrated that the organic matter content of the adsorbents had a great influence on the n value; one sediment with high organic matter content had an n value of 0.77 for OXA, while this value was 0.98 for another sediment with much lower organic matter content. Surprisingly, in the case of hydrophobic compounds, high adsorption energy was found for goethite, with n values of 0.5–0.6, indicating that the lack of Coulomb interactions on goethite is compensated by van der Waals/hydrophobic, π-π interactions and H-bonding. This is due to interactions with surface hydroxyls, possibly by means of hydrogen bonding, suggesting that hydrophobic PhACs bind primarily to uncharged surface hydroxyl groups (Van Emmerik et al., 2003).

In the case of fully or partly ionic compounds, such as DIC, LID or TRA, the main question is whether the molecule is negatively or positively charged. Unexpectedly, in spite of its negative charge, DIC has a rather low n value, indicating relatively high adsorption affinity to soils (Table 2). This could be attributed to two reasons: on the one hand the –NH group of the DIC molecule is a very strong hydrogen bond donor (Taylor, 2017), which is able to form hydrogen bonds with the H bond acceptor sites of SOM, and on the other hand the surface of goethite in soils (Table 1) could in part be positively charged, resulting in attractive Coulomb forces. Though relatively high adsorption capacity was calculated for TRA on soils, the n values suggested low adsorption energy (n = 0.73–0.90; Table 2). This may reflect concentration-independent electrostatic interactions between the cationic TRA species and the negatively charged sites on SOM or on the charged surfaces of 2:1 clay minerals (Sumpter and Johnson, 2005).

### 3.2. Comprehensive effect of the chemical properties of PhACs on their sorption on soils and goethite

Based on VIF values, only six of the nine explanatory variables (logD, the number of H bond acceptor and donor sites, the fraction of negatively and positively charged species and the  $\pi$ -energy) were used in the redundancy analysis (RDA) because of the collinearity between the variables. The first two axes on the RDA ordination plot explained 34.5% of the total variance, implying that the intrinsic chemical properties of the molecules had a great effect on sorption. The unconstrained variability was more than 60%, indicating that besides the chemical properties of the compounds the environmental parameters of the adsorbents, such as pH, organic matter content, mineral composition, etc., were also important factors that could control sorption characteristics.

To provide a comprehensive evaluation of the role of the chemical characteristics of PhACs in sorption processes and to reveal the relationship between them, a correlation biplot was constructed (Fig. 1). The positive association with axis 1 was strong for logD and slight for the number of H bond acceptor sites, while the number of H bond donor sites, the pi energy and the fraction of positive species ( $\alpha_+$ ) was negatively associated with axis 1. The second axis was positively correlated with the fraction of negative species ( $\alpha_-$ ).

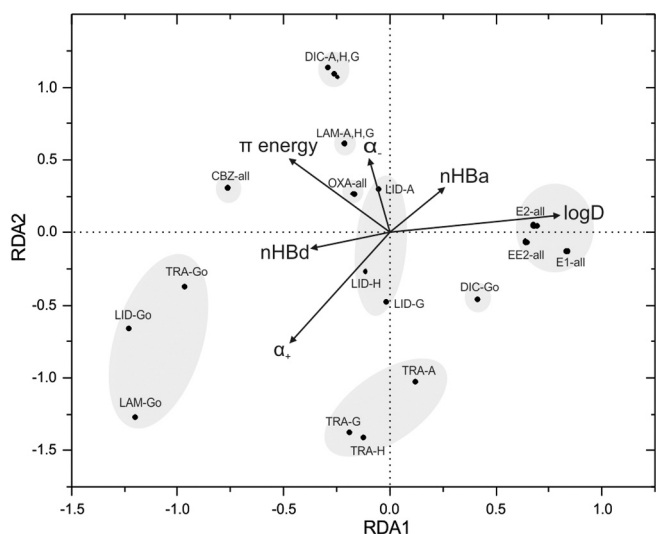
The hydrophobicity of the steroid hormones investigated here (E1, E2, EE2; Fig. 1) had a significant effect on adsorption; a very strong correlation was found between the logD values and the site constraints (the coordinates of the sites in the space of explanatory variables) for the adsorption of E1, E2 and EE2 on the soils and on goethite. It should also be noted that there was no difference between the soils and goethite for the given steroid hormone in the ordination space: the site points representing the adsorption of E1, E2 and EE2 on soils and goethite were in almost the same place.

The hydrophobic interaction is of entropic origin and influences the sorption of hydrophobic compounds or moieties via the formation of aggregates on the solid-liquid interface, thus reducing contact between the hydrophobic molecules and the neighbouring water molecules (Meyer et al., 2006). A number of studies on the sorption of hydrophobic compounds (HOC) on soils and sediments have shown that the

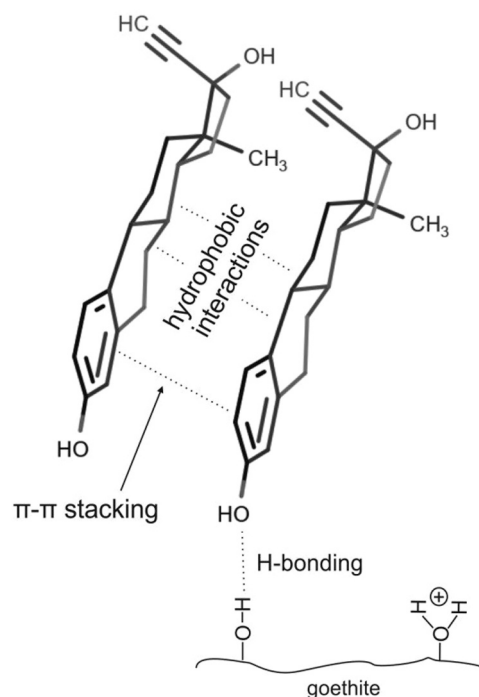
hydrophobic interaction is dominant (Franco et al., 2009; Higgins and Luthy, 2007; Rybacka and Andersson, 2016). However, Yamamoto et al. (2003) reported that the hydrophobic interaction made a relatively weak contribution to the adsorption of PhACs on DOM surrogates. For soils with high organic carbon content, organic matter may act as a hydrophobic sorbent, and aromatic structures may be adsorbed on this organic phase via van der Waals forces (Keiluweit and Kleber, 2009). Murphy et al. (1990) highlighted the fact that mineral surfaces also contribute to HOC sorption. The siloxane and aluminol surfaces of clay minerals contribute to the adsorption of hydrophobic compounds (Van Emmerik et al., 2003). This was confirmed by the fact that logD, which represents hydrophobic interactions, not only controls soil-steroid hormone relationships but also the goethite-estrogens system. The sorption mechanism appears to proceed as follows: the estrogen molecules bind to goethite via hydrogen bonds, after which further estrogen molecules are adsorbed onto the first layer, forming aggregates that are stabilised by hydrophobic interactions and  $\pi$ - $\pi$  stacking (Fig. 2), as demonstrated in the study of Lima et al. (2011) for EE2. Similar results were found for phenolic compounds (Nthunya et al., 2019).

Another mechanism that could contribute to the sorption process of estrogens is hydrogen bonding. It can be clearly seen in Fig. 1 that the number of H-bond acceptor sites in the molecules correlates well with the points representing the sorption of steroids on both soil and goethite. All the estrogen molecules investigated have four acceptor sites (two O atoms each with two non-bonding electron pairs), which are able to form H bonds with the -OH groups of soils and goethite. This mechanism was confirmed by other studies for both soils (Neale et al., 2009; Qi and Zhang, 2016; Tolls, 2001) and goethite (Van Emmerik et al., 2003). In spite of the great differences in adsorption capacity and affinity values (Table 2), the sorption mechanism of steroid hormones could be similar on both soils and goethite (Fig. 3).

The formation of electron donor-acceptor complexes (EDA interactions) between PhACs and soil organic matter is very important for the sorption of these molecules. Due to their high  $\pi$  energy (Table 1), OXA, CBZ and LAM are able to interact with the aromatic moieties of SOM through  $\pi$ - $\pi$  interactions, forming weak non-covalent bonds between aromatic rings (Keiluweit and Kleber, 2009; Senesi, 1992; Zhao et al., 2017). RDA showed that the adsorption of OXA, CBZ and LAM is



**Fig. 1.** RDA biplot showing the relationship between the explanatory variables (solid arrows) and the sites (indicating the sorption characteristics of the given PhACs on the given adsorbents). A, H, G and Go denotes Arenosol, Histosol, Gleysol and goethite, while term “all” denotes all the adsorbents. nHBa, nHBd,  $\alpha_+$  and  $\alpha_-$  are the number of H bond acceptor sites, the number of H bond donor sites, fraction of PhAC as negative form and fraction of PhAC as positive form, respectively.



**Fig. 2.** Theoretical mechanism of the sorption of EE2 on goethite.

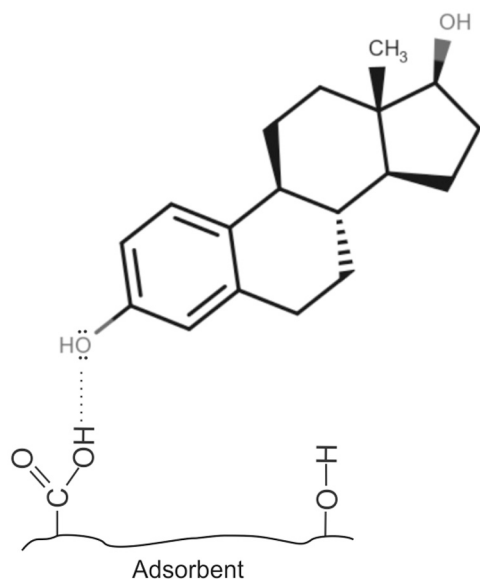


Fig. 3. Mechanism of the sorption of estrogens on soil/goethite via hydrogen bonding.

controlled to a major extent by their  $\pi$  energy (Fig. 1 and Table 1), implying that several EDA interactions may occur:  $\pi$ - $\pi$ , n- $\pi$ , OH $\cdots\pi$  and C-H $\cdots\pi$  interactions. The electron-rich arene moieties of these pharmaceuticals, as electron donor structures, are able to interact with the electron-acceptor moieties of organic matter, including quinones (Sensi, 1992; Zhu et al., 2004). These  $\pi$ - $\pi$  EDA complexes can be depicted as a parallel-planar arrangement that enables the two complementary quadrupoles to interact via electrostatic forces (Hunter and Sanders, 1990).

Although the C-H $\cdots\pi$  interaction is considered to be very weak (0.5–1 kcal/mol) (Kumar and Balaji, 2014), it may represent another way of binding aromatic compounds to soil organic matter or the mineral phase (Reid et al., 2013). This could involve interactions between aliphatic C-H donors and aromatic  $\pi$ -acceptors and between aromatic C-H donors and aromatic  $\pi$ -acceptors. It is also possible that the aromatic  $\pi$ -orbitals of PhAC molecules interact with the surface hydroxyl groups of soils via OH $\cdots\pi$  interactions (Cusumano and Low, 1971). Therefore, OXA, CBZ and LAM could combine with the silanol and aluminol groups to form OH $\cdots\pi$  (Jabraoui et al., 2019).

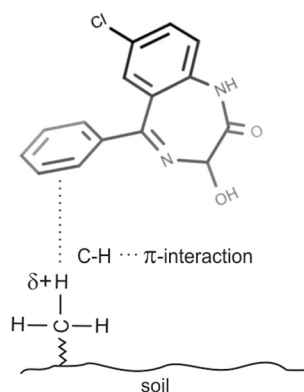
These molecules are also capable of forming n- $\pi$  interactions, since the non-bonding electrons of the  $-\text{COO}^-$  groups in SOM, which were abundant in the soils used in this study, act as strong n-donors in n- $\pi$  EDA

interactions with  $\pi$ -accepting PhACs (Qu et al., 2008). Due to the presence of Cl as an electron-withdrawing group, the  $\pi$  structures of LAM and OXA could be excellent  $\pi$ -acceptors. This was confirmed in other studies, such as that of Haderlein and Schwarzenbach (1993) on the adsorption of nitrobenzenes and nitrophenols on mineral surfaces. The primary sorption sites for electron-poor  $\pi$  systems are basal siloxane surfaces, which exhibit significant n-donor capacity (Keiluweit and Kleber, 2009). Some examples are shown for EDA interactions in Fig. 4.

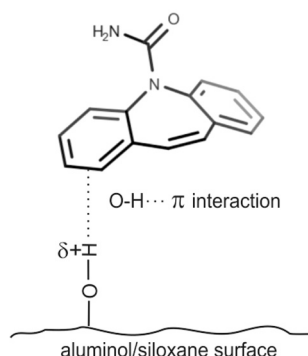
The sorption of LAM, LID and TRA on goethite is controlled mainly by their positive charge and ability to form hydrogen bonds (Fig. 1). The fraction of the positively charged form of the molecules was 100% for all three compounds (Table 1), and the charge of the goethite surface was also dominantly positive. It can therefore be assumed that strong repulsive forces exist between the positively charged protonated  $-\text{OH}_2^+$  groups and the protonated PhAC molecules. However, strong hydrogen donor sites are also available in the LAM, LID and TRA molecules, which possess  $-\text{NH}$ ,  $-\text{NH}_2$  and  $-\text{OH}$  groups. In spite of the repulsive forces, these compounds have quite high  $Q_{\text{max}}$  values and high adsorption affinity, especially in the case of LAM (Table 2), probably due to the large number of hydrogen bond donor sites (Table 1), which make hydrogen bonds possible between amines and surface hydroxyl groups (Guo et al., 2016) (Fig. 5).

Positive charges were dominant in Tramadol under the chemical conditions found in the tested soils, the fraction of positively charged species ranging from 70% to 95% in Arenosol, Gleysol and Histosol (Table 1). The influence of the positive charge on the sorption of the molecule is clearly shown in Fig. 1, where a correlation can be observed between the positive charge of the molecule and the sites. At such a high soil pH, the cationic form of TRA becomes dominant, protonating the  $\text{R}_2\text{-N}$ : group to  $\text{R}_2\text{-NH}^+$  (for structure, see Supplementary Material, Table S4). At this range of pH, most of the variable charge sites are negative ( $-\text{COO}^-$ ,  $-\text{O}^-$ ), in addition to which the mineral soil particles also bear negative charges due to isomorphous substitution, which results in strong Coulomb forces between the positively charged TRA and negatively charged soil particles. This was demonstrated by the very high  $Q_{\text{max}}$  value of TRA (Table 2). However, there is no clear trend in the  $Q_{\text{max}}$  values; the OM content in itself does not seem to be the only factor that governs this process. The organic matter quality and the composition of the mineral phases could also be crucial in the control of sorption processes (Chefetz et al., 2008; Droge and Goss, 2013; Jung et al., 2013). FT-IR analysis on the humic acids (HA) of the soil samples highlighted considerable differences between the HA quality of the samples (Table S5). For example, the relative absorbance of the  $1709\text{ cm}^{-1}$  peak, which is characteristic of  $\text{C}=\text{O}$  stretching in carboxylic acids, was lower in the samples with the highest OC content (G\_20 and H\_20: 9.7% and 11.4%, respectively) than in the Arenosol soil (15.9%). This is likely to be due to the tendency for organic matter to retain water, leading to

a.) OXA



b.) CBZ



c.) LAM

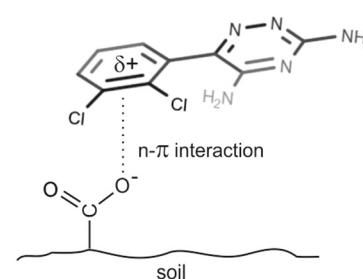


Fig. 4. Examples of EDA interactions between PhACs and soil/goethite.

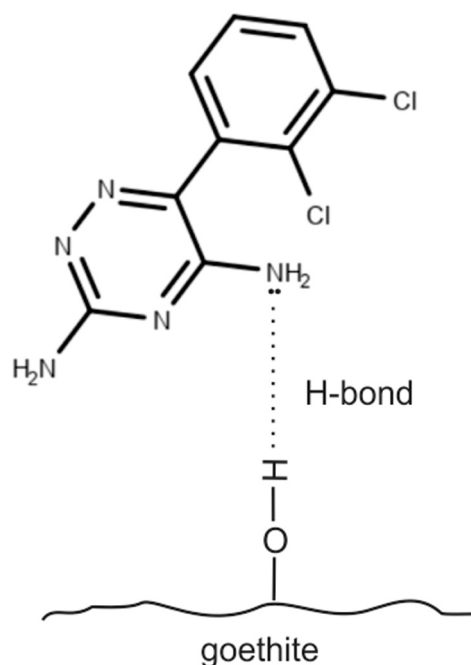


Fig. 5. Sorption mechanism of lamotrigine on goethite via hydrogen bonding.

anaerobic conditions where oxidation is limited. By contrast, Arenosol may contain more oxidised organic compounds, resulting in more carboxylic groups and greater ability to retain positively charged molecules. Great differences were also detected in the composition of the mineral phase: Gleysol contained much more smectite and mica, which have enormous surface areas with negative charges.

Under neutral or alkali conditions Na-diclofenac is present in anionic form (Table 1), so, not surprisingly, the fraction of negatively charged species was the molecular attribute that governed sorption on the calcareous soils investigated, with strong repulsive forces between the negative charges of the soil particles and the negatively charged DIC. In addition, the deprotonated hydroxyl groups ( $-O^-$ ) in soils are able to interact with the electron-poor arene moieties of the DIC molecule as  $\pi$ -acceptors, leading to  $n$ - $\pi$  interactions (Chen et al., 2008). This interaction may be responsible for the fact that the sorption of DIC on soils is controlled by the  $\pi$  energy (Fig. 1). However, the sorption of DIC on goethite described by the isotherm parameters is very different from the sorption on soils, visualised in the ordination space (Fig. 1), suggesting a completely different sorption mechanism. When the pH decreases the  $-O^-$  group of DIC starts to protonate, reaching a 76% proportion in the neutral form with a logD value of 4.15 at pH 3.8. Under such conditions DIC was the most hydrophobic molecule, so the most plausible mechanism for the neutral form of diclofenac is hydrophobic interaction, based on the data visualised in Fig. 1. The sorption could be similar to that of the estrogens, with a first layer binding to the surface via hydrogen bonds, followed by the formation of aggregates, which are stabilised by hydrophobic interactions (Fig. 2). Furthermore, DIC has great ability to form H bonds, having six acceptor sites ( $-OH$ ,  $=O$ ,  $-NH$ ), taking hydrogen atoms from  $-OH$  and  $-OH_2$  groups.

#### 4. Conclusions

This study provided data on the importance of the effect of chemical structure on sorption processes, as indicated by the fact that the chemical properties of the PhACs explained one third of the total variance. It highlighted the priority of environmental factors for the retention of PhACs in soils and sediments. Furthermore, the sorption of PhACs is controlled mainly by environmental factors, such as pH, organic matter content and quality, as was clearly demonstrated by the extensive

adsorption of hydrophobic molecules (E1, E2 and EE2) on soils with high organic matter content and by the relatively high adsorption of TRA on the A\_20 sample, which has lower OM content but highly oxidised organic matter.

However, the intrinsic chemical properties of PhACs cannot be separated from the environmental components of sorption: dividing the factors that control the sorption process into intrinsic and environmental ones is rather theoretical. For example, any change in pH may cause an alteration in the hydrophobicity (logD) of PhACs, as seen in the case of LAM, LID, OXA and TRA (Table 1). Even small changes in pH ( $< 0.9$ ) can result in changes in the size of the fraction of positively charged molecules, as found for TRA (from 70% to 95% in the tested soils). So each environmental factor that controls the chemical properties of the compounds acts through an alteration in the molecular structure.

This study also revealed that for PhACs whose structure is independent of these factors (pH, etc.), the adsorption mechanisms are identical for both soils and goethite: sorption is determined purely by the number of adsorption sites, as proved by the adsorption of estrogens and the neutral compounds CBZ and OXA, where the site points representing the adsorption of these molecules on soils and goethite were in almost the same place. Our study could be useful for understanding of PhACs fate in soils, and for application in transport and risk assessment models.

#### CRedit authorship contribution statement

**Tibor Filep:** Conceptualization, Writing - original draft. **Lili Szabó:** Writing - reviewing & editing, Data curation. **Attila Csaba Kondor:** Visualization, Data curation. **Gergely Jakab:** Data curation. **Zoltán Szalai:** Supervision.

#### Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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#### Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.ecoenv.2021.112120](https://doi.org/10.1016/j.ecoenv.2021.112120).

#### References

- Al-Khazrajy, O.S.A., Boxall, A.B.A., 2016. Impacts of compound properties and sediment characteristics on the sorption behaviour of pharmaceuticals in aquatic systems. *J. Hazard. Mater.* 317, 198–209. <https://doi.org/10.1016/j.jhazmat.2016.05.065>.
- Antelo, J., Avena, M., Fiol, S., López, R., Arce, F., 2005. Effects of pH and ionic strength on the adsorption of phosphate and arsenate at the goethite-water interface. *J. Colloid Interface Sci.* 285, 476–486. <https://doi.org/10.1016/j.jcis.2004.12.032>.
- Bäuerlein, P.S., Mansell, J.E., Ter Laak, T.L., De Voogt, P., 2012. Sorption behavior of charged and neutral polar organic compounds on solid phase extraction materials: which functional group governs sorption? *Environ. Sci. Technol.* 46, 954–961. <https://doi.org/10.1021/es203404x>.
- Braak, C.J.F. ter, Smilauer, P., 2012. Canoco reference manual and user's guide: software for ordination, version 5.0.
- Caporale, A.G., Adamo, P., Capozzi, F., Langella, G., Terribile, F., Vingiani, S., 2018. Monitoring metal pollution in soils using portable-XRF and conventional laboratory-based techniques: evaluation of the performance and limitations according to metal properties and sources. *Sci. Total Environ.* 643, 516–526. <https://doi.org/10.1016/j.scitotenv.2018.06.178>.
- Caracciolo, B.A., Topp, E., Grenni, P., 2015. Pharmaceuticals in the environment: biodegradation and effects on natural microbial communities. A review. *J. Pharm. Biomed. Anal.* 106, 25–36. <https://doi.org/10.1016/j.jpba.2014.11.040>.
- Carmosini, N., Lee, L.S., 2009. Ciprofloxacin sorption by dissolved organic carbon from reference and bio-waste materials. *Chemosphere* 77, 813–820. <https://doi.org/10.1016/j.chemosphere.2009.08.003>.

- Chefetz, B., Mualem, T., Ben-Ari, J., 2008. Sorption and mobility of pharmaceutical compounds in soil irrigated with reclaimed wastewater. *Chemosphere* 73, 1335–1343. <https://doi.org/10.1016/j.chemosphere.2008.06.070>.
- Chen, W., Duan, L., Wang, L., Zhu, D., 2008. Adsorption of hydroxyl- and amino-substituted aromatics to carbon nanotubes. *Environ. Sci. Technol.* 42, 6862–6868. <https://doi.org/10.1021/es8013612>.
- Cusumano, J.A., Low, M.J.D., 1971. Interaction between surface hydroxyl groups and adsorbed molecules. III. The nature of the adsorbate-hydroxyl interaction. *J. Catal.* 23, 214–227. [https://doi.org/10.1016/0021-9517\(71\)90043-1](https://doi.org/10.1016/0021-9517(71)90043-1).
- Dada, A.O., Olalekan, A.P., Olatunya, A.M., Dada, O., 2012. Langmuir, Freundlich, Temkin and Dubinin–Radushkevich isotherms studies of equilibrium sorption of Zn 2+ onto phosphoric acid modified rice husk. *IOSR J. Appl. Chem.* 3, 38–45. <https://doi.org/10.9790/5736-0313845>.
- Dai, Y.N., Dan, A., Yang, Y., Tam, N.F.Y., Tai, Y.P., Tang, X.Y., 2016. Factors affecting behavior of phenolic endocrine disruptors, estrone and estradiol, in constructed wetlands for domestic sewage treatment. *Environ. Sci. Technol.* 50, 11844–11852. <https://doi.org/10.1021/acs.est.6b02026>.
- Droge, S.T.J., Goss, K.U., 2013. Development and evaluation of a new sorption model for organic cations in soil: contributions from organic matter and clay minerals. *Environ. Sci. Technol.* 47, 14233–14241. <https://doi.org/10.1021/es4031886>.
- Fenet, H., Mathieu, O., Mahjoub, O., Li, Z., Hillaire-Buys, D., Casellas, C., Gomez, E., 2012. Carbamazepine, carbamazepine epoxide and dihydroxycarbamazepine sorption to soil and occurrence in a wastewater reuse site in Tunisia. *Chemosphere* 88, 49–54. <https://doi.org/10.1016/j.chemosphere.2012.02.050>.
- Flint, A.L., Flint, L.E., 2002. Particle density. In: Dane, J.H., Topp, G.C. (Eds.), *Methods of Soil Science*. Soil Science Society of America, Madison, pp. 229–240.
- Franco, A., Fu, W., Trapp, S., 2009. Influence of soil pH on the sorption of ionizable chemicals: modeling advances. *Environ. Toxicol. Chem.* 28, 458–464. <https://doi.org/10.1897/1552-8618-28.9.2018>.
- Göbel, A., Thomsen, A., McARDell, C.S., Joss, A., Giger, W., 2005. Occurrence and sorption behavior of sulfonamides, macrolides, and trimethoprim in activated sludge treatment. *Environ. Sci. Technol.* 39, 3981–3989. <https://doi.org/10.1021/es048550a>.
- Gomes, J., Costa, R., Quinta-Ferreira, R.M., Martins, R.C., 2017. Application of ozonation for pharmaceuticals and personal care products removal from water. *Sci. Total Environ.* 586, 265–283. <https://doi.org/10.1016/j.scitotenv.2017.01.216>.
- Guo, X., Yin, Y., Yang, C., Zhang, Q., 2016. Remove mechanisms of sulfamethazine by goethite: the contributions of pH and ionic strength. *Res. Chem. Intermed.* 42, 6423–6435. <https://doi.org/10.1007/s11164-016-2472-4>.
- Haderlein, S.B., Schwarzenbach, R.P., 1993. Adsorption of substituted nitrobenzenes and nitrophenols to mineral surfaces. *Environ. Sci. Technol.* 27, 316–326. <https://doi.org/10.1021/es00039a012>.
- Hari, A.C., Paruchuri, R.A., Sabatini, D.A., Kibbey, T.C.G., 2005. Effects of pH and cationic and nonionic surfactants on the adsorption of pharmaceutical to a natural aquifer material. *Environ. Sci. Technol.* 39, 2592–2598. <https://doi.org/10.1021/es048992m>.
- He, Y., Sutton, N.B., Lei, Y., Rijnaarts, H.H.M., Langenhoff, A.A.M., 2018. Fate and distribution of pharmaceutically active compounds in mesocosm constructed wetlands. *J. Hazard. Mater.* 357, 198–206. <https://doi.org/10.1016/j.jhazmat.2018.05.035>.
- Higgins, C.P., Luthy, R.G., 2007. Modeling sorption of anionic surfactants onto sediment materials: an a priori approach for perfluoroalkyl surfactants and linear alkylbenzene sulfonates. *Environ. Sci. Technol.* 41, 3254–3261. <https://doi.org/10.1021/es062449j>.
- Hunter, C.A., Sanders, J.K.M., 1990. The nature of  $\pi$ - $\pi$  Interactions. *J. Am. Chem. Soc.* 112, 5525–5534. <https://doi.org/10.1021/ja00170a016>.
- Jabraoui, H., Hessou, E.P., Chibani, S., Cantrel, L., Lebègue, S., Badawi, M., 2019. Adsorption of volatile organic and iodine compounds over silver-exchanged mordenites: a comparative periodic DFT study for several silver loadings. *Appl. Surf. Sci.* 485, 56–63. <https://doi.org/10.1016/j.apsusc.2019.03.282>.
- Jiang, M., Yang, W., Zhang, Z., Yang, Z., Wang, Y., 2015. Adsorption of three pharmaceuticals on two magnetic ion-exchange resins. *J. Environ. Sci.* 31, 226–234. <https://doi.org/10.1016/j.jes.2014.09.035>.
- Jung, C., Park, J., Lim, K.H., Park, S., Heo, J., Her, N., Oh, J., Yun, S., Yoon, Y., 2013. Adsorption of selected endocrine disrupting compounds and pharmaceuticals on activated biochars. *J. Hazard. Mater.* 263, 702–710. <https://doi.org/10.1016/j.jhazmat.2013.10.033>.
- Kaur, H., Bansiwala, A., Hippargi, G., Pophali, G.R., 2018. Effect of hydrophobicity of pharmaceuticals and personal care products for adsorption on activated carbon: adsorption isotherms, kinetics and mechanism. *Environ. Sci. Pollut. Res.* 25, 20473–20485. <https://doi.org/10.1007/s11356-017-0054-7>.
- Keilweil, M., Kleber, M., 2009. Molecular-level interactions in soils and sediments: the role of aromatic  $\pi$ -systems. *Environ. Sci. Technol.* 43, 3421–3429. <https://doi.org/10.1021/es8033044>.
- Klatte, S., Schaefer, H.C., Hempel, M., 2017. Pharmaceuticals in the environment – a short review on options to minimize the exposure of humans, animals and ecosystems. *Sustain. Chem. Pharm.* 5, 61–66. <https://doi.org/10.1016/j.scp.2016.07.001>.
- Kolpin, D.W., Furlong, E.T., Meyer, M.T., Thurman, E.M., Zaugg, S.D., Barber, L.B., Buxton, H.T., 2002. Pharmaceuticals, hormones, and other organic wastewater contaminants in U.S. streams, 1999–2000: a national reconnaissance. *Environ. Sci. Technol.* 36, 1202–1211. <https://doi.org/10.1021/es011055j>.
- Kumar, M., Balaji, P.V., 2014. C-H $\pi$  interactions in proteins: prevalence, pattern of occurrence, residue propensities, location, and contribution to protein stability. *J. Mol. Model.* 20, 1–14. <https://doi.org/10.1007/s00894-014-2136-5>.
- Kumar, M., Philip, L., 2006. Adsorption and desorption characteristics of hydrophobic pesticide endosulfan in four Indian soils. *Chemosphere* 62, 1064–1077. <https://doi.org/10.1016/j.chemosphere.2005.05.009>.
- Kusturica, M., Tomas, A., Sabo, A., 2017. Disposal of unused drugs: knowledge and behavior among people around the world. In: *Reviews of Environmental Contamination and Toxicology*. Springer, New York LLC, pp. 71–104. <https://doi.org/10.1007/978-1-4939-2016-3>.
- Li, J., Jiang, L., Liu, X., Lv, J., 2013. Adsorption and aerobic biodegradation of four selected endocrine disrupting chemicals in soil-water system. *Int. Biodeterior. Biodegrad.* 76, 3–7. <https://doi.org/10.1016/j.ibiod.2012.06.004>.
- Lima, D.L.D., Calisto, V., Esteves, V.I., 2011. Adsorption behavior of 17 $\alpha$ -ethynylestradiol onto soils followed by fluorescence spectral deconvolution. *Chemosphere* 84, 1072–1078. <https://doi.org/10.1016/j.chemosphere.2011.04.060>.
- Loeppert, Richard H., Suarez, D.L., 1996. Carbonate and gypsum. In: Sparks, D.L., Page, A.L., Helmke, P.A., Loeppert, R.H., Soltanpour, P.N., Tabatabai, M.A., Johnston, C.T., Sumner, M.E. (Eds.), *Methods of Soil Analysis: Part 3 Chemical Methods*. John Wiley & Sons, Ltd, pp. 437–474. <https://doi.org/10.2136/sssabookser5.3.c15>.
- Martínez-Hernández, V., Meffe, R., Herrera, S., Arranz, E., de Bustamante, I., 2014. Sorption/desorption of non-hydrophobic and ionisable pharmaceutical and personal care products from reclaimed water onto/from a natural sediment. *Sci. Total Environ.* 472, 273–281. <https://doi.org/10.1016/j.scitotenv.2013.11.036>.
- Matejovic, I., 1997. Determination of carbon and nitrogen in samples of various soils by the dry combustion. *Commun. Soil Sci. Plant Anal.* 28, 1499–1511. <https://doi.org/10.1080/00103629709369892>.
- Meyer, E.E., Rosenberg, K.J., Israelachvili, J., 2006. Recent progress in understanding hydrophobic interactions. *Proc. Natl. Acad. Sci. USA* 103, 15739–15746. <https://doi.org/10.1073/pnas.0606422103>.
- Murphy, E.M., Zachara, J.M., Smith, S.C., 1990. Influence of mineral-bound humic substances on the sorption of hydrophobic organic compounds. *Environ. Sci. Technol.* 24, 1507–1516. <https://doi.org/10.1021/es00080a009>.
- Neale, P.A., Escher, B.I., Schäfer, A.I., 2009. pH dependence of steroid hormone-organic matter interactions at environmental concentrations. *Sci. Total Environ.* 407, 1164–1173. <https://doi.org/10.1016/j.scitotenv.2008.09.035>.
- Nthunya, L.N., Gutierrez, L., Dereze, S., Mamba, B.B., Verliefdé, A.R., Mhlanga, S.D., 2019. Adsorption of phenolic compounds by polyacrylonitrile nanofibre membranes: a pretreatment for the removal of hydrophobic bearing compounds from water. *J. Environ. Chem. Eng.* 7. <https://doi.org/10.1016/j.jece.2019.103254>.
- Paul, S., Sharma, T., Saikia, D., Saikia, P., Borah, D., Baruah, M., 2015. Evaluation of pKa values of soil humic acids and their complexation properties. *Int. J. Plant Soil Sci.* 6, 218–228. <https://doi.org/10.9734/ijps/2015/16033>.
- Pennell, K.D., 2002. 2.5 Specific surface area. In: Dane, J.H., Topp, G.C. (Eds.), *Methods of Soil Analysis: Part 4 Physical Methods*. John Wiley & Sons, Ltd, pp. 295–315. <https://doi.org/10.2136/sssabookser5.4.c13>.
- Qi, Y., Zhang, T.C., 2016. Sorption and desorption of testosterone at environmentally relevant levels: effects of aquatic conditions and soil particle size fractions. *J. Environ. Eng.* 142, 04015045. [https://doi.org/10.1061/\(ASCE\)EE.1943-7870.0001001](https://doi.org/10.1061/(ASCE)EE.1943-7870.0001001).
- Qu, X., Xiao, L., Zhu, D., 2008. Site-specific adsorption of 1,3-dinitrobenzene to bacterial surfaces: a mechanism of  $n$ - $\pi$  electron-donor-acceptor interactions. *J. Environ. Qual.* 37, 824–829. <https://doi.org/10.2134/jeq2007.0236>.
- Reid, S.A., Nyambo, S., Muzangwa, L., Uhler, B., 2013.  $\pi$ -Stacking, C-H/ $\pi$ , and halogen bonding interactions in bromobenzene and mixed bromobenzene-benzene clusters. *J. Phys. Chem. A* 117, 13556–13563. <https://doi.org/10.1021/jp407544c>.
- Rybacka, A., Andersson, P.L., 2016. Considering ionic state in modeling sorption of pharmaceuticals to sewage sludge. *Chemosphere* 165, 284–293. <https://doi.org/10.1016/j.chemosphere.2016.09.014>.
- Sangsupan, H.A., Radcliffe, D.E., Hartel, P.G., Jenkins, M.B., Vencill, W.K., Cabrera, M.L., 2006. Sorption and transport of 17 $\beta$ -estradiol and testosterone in undisturbed soil columns. *J. Environ. Qual.* 35, 2261–2272. <https://doi.org/10.2134/jeq2005.0401>.
- Schlüsener, M.P., Bester, K., 2008. Behavior of steroid hormones and conjugates during wastewater treatment – a comparison of three sewage treatment plants. *CLEAN Soil Air Water* 36, 25–33. <https://doi.org/10.1002/clen.200700096>.
- Senesi, N., 1992. Binding mechanisms of pesticides to soil humic substances. *Sci. Total Environ.* 123–124, 63–76. [https://doi.org/10.1016/0048-9697\(92\)90133-D](https://doi.org/10.1016/0048-9697(92)90133-D).
- Stein, K., Ramil, M., Fink, G., Sander, M., Ternes, T.A., 2008. Analysis and sorption of psychoactive drugs onto sediment. *Environ. Sci. Technol.* 42, 6415–6423. <https://doi.org/10.1021/es702959a>.
- Sumpter, J.P., Johnson, A.C., 2005. Lessons from endocrine disruption and their application to other issues concerning trace organics in the aquatic environment. *Environ. Sci. Technol.* <https://doi.org/10.1021/es048504a>.
- Szabó, L., Vancsik, A., Király, C., Ringer, M., Kondor, A., Jakab, G., Szalai, Z., Filep, T., 2020. Investigation of the sorption of 17 $\alpha$ -ethynylestradiol (EE2) on soils formed under aerobic and anaerobic conditions. *Chemosphere* 240. <https://doi.org/10.1016/j.chemosphere.2019.124817>.
- Taylor, R., 2017. The hydrogen bond between N-H or O-H and organic fluorine: favourable yes, competitive no. *Acta Crystallogr. Sect. B Struct. Sci. Cryst. Eng. Mater.* 73, 474–488. <https://doi.org/10.1107/S2052520617005923>.
- Thomas, G.W., 1996. Soil pH and soil acidity. In: Sparks, D.L., Page, A.L., Helmke, P.A., Loeppert, R.H., Soltanpour, P.N., Tabatabai, M.A., Johnston, C.T., Sumner, M.E. (Eds.), *Methods of Soil Analysis: Part 3 Chemical Methods*. John Wiley & Sons, Ltd, pp. 475–490. <https://doi.org/10.2136/sssabookser5.3.c16>.
- Tolls, J., 2001. Sorption of veterinary pharmaceuticals in soils: a review. *Environ. Sci. Technol.* <https://doi.org/10.1021/es0003021>.
- Valdés, M.E., Amé, M.V., Bistoni, M. de los A., Wunderlin, D.A., 2014. Occurrence and bioaccumulation of pharmaceuticals in a fish species inhabiting the Suquia River

- basin (Córdoba, Argentina). *Sci. Total Environ.* 472, 389–396. <https://doi.org/10.1016/j.scitotenv.2013.10.124>.
- Van Emmerik, T., Angove, M.J., Johnson, B.B., Wells, J.D., Fernandes, M.B., 2003. Sorption of 17 $\beta$ -estradiol onto selected soil minerals. *J. Colloid Interface Sci.* 266, 33–39. [https://doi.org/10.1016/S0021-9797\(03\)00597-6](https://doi.org/10.1016/S0021-9797(03)00597-6).
- Wu, L., Bi, E., 2019. Sorption of ionic and neutral species of pharmaceuticals to loessial soil amended with biochars. *Environ. Sci. Pollut. Res.* 26, 35871–35881. <https://doi.org/10.1007/s11356-019-06721-7>.
- Yamamoto, H., Liljestrand, H.M., Shimizu, Y., Morita, M., 2003. Effects of physical-chemical characteristics on the sorption of selected endocrine disruptors by dissolved organic matter surrogates. *Environ. Sci. Technol.* 37, 2646–2657. <https://doi.org/10.1021/es026405w>.
- Zhao, H., Liu, X., Cao, Z., Zhan, Y., Shi, X., Yang, Y., Zhou, J., Xu, J., 2016. Adsorption behavior and mechanism of chloramphenicols, sulfonamides, and non-antibiotic pharmaceuticals on multi-walled carbon nanotubes. *J. Hazard. Mater.* 310, 235–245. <https://doi.org/10.1016/j.jhazmat.2016.02.045>.
- Zhao, Q., Zhang, S., Zhang, X., Lei, L., Ma, W., Ma, C., Song, L., Chen, J., Pan, B., Xing, B., 2017. Cation- $\pi$  interaction: a key force for sorption of fluoroquinolone antibiotics on pyrogenic carbonaceous materials. *Environ. Sci. Technol.* 51, 13659–13667. <https://doi.org/10.1021/acs.est.7b02317>.
- Zhu, D., Hyun, S., Pignatello, J.J., Lee, L.S., 2004. Evidence for  $\pi$ - $\pi$  electron donor-acceptor interactions between  $\pi$ -donor aromatic compounds and  $\pi$ -acceptor sites in soil organic matter through pH effects on sorption. *Environ. Sci. Technol.* 38, 4361–4368. <https://doi.org/10.1021/es035379e>.