



# Investigation of the sorption of 17 $\alpha$ -ethynylestradiol (EE2) on soils formed under aerobic and anaerobic conditions

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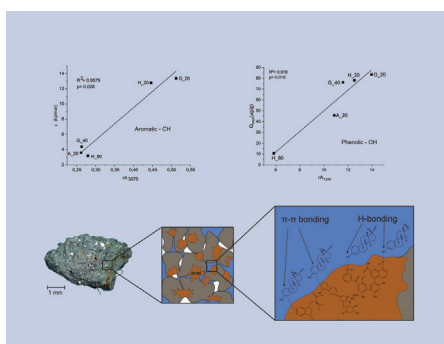
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## HIGHLIGHTS

- The adsorption of EE2 was connected with the redox potential of the soils.
- Aromatic and phenolic compounds were found to be abundant in anaerobic layers.
- Hydromorphic soils with organic matter accumulation had high adsorption capacity.
- $\pi$ - $\pi$  interaction and H-bonding were found the dominant sorption mechanisms.

## GRAPHICAL ABSTRACT



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## ABSTRACT

A study was conducted on the sorption of 17 $\alpha$ -ethynylestradiol (EE2) on five soils formed under different redox conditions: an Arenosol (A\_20) with fully aerobic conditions, two Gleysol samples (G\_20 and G\_40) with suboxic and anoxic conditions and two Histosols (H\_20 and H\_80) with mostly anoxic conditions. The soils were characterized on the basis of total organic carbon (TOC), specific surface area (SSA) and the Fourier transform infrared spectra of the humic acid and humin fractions (the soil remaining after alkali extraction) of the soil. The maximum adsorption capacity of the soils ( $Q_{max}$ ) ranged from 10.7 to 83.6 mg/g in the order G\_20 > H\_20 > G\_40 > A\_20 > H\_80, which reflected the organic matter content of the soils. The sorption isotherms were found to be nonlinear for all the soil samples, with Freundlich  $n$  values of 0.45–0.68. The strong nonlinearity found in the adsorption of the H\_80 samples could be attributed to their high hard carbon content, which was confirmed by the high aromaticity of the humin fraction. The maximum sorption capacity ( $Q_{max}$ ) of the soils did not increase indefinitely as the organic carbon content of the soils rose. There could be two reasons for this: (i) the large amount of organic matter may reduce the number of binding sites on the surface, and (ii) the decrease in SSA with increasing soil OC content may limit the ability to adsorb EE2 molecules. In anaerobic soil samples, where organic matter accumulation is pronounced, the amount of aromatic and phenolic compounds was higher than in better aerated soil profiles. Strong correlations were found between the amount of aromatic and phenolic

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compounds in the organic matter and the adsorption of EE2 molecules, indicating that  $\pi$ - $\pi$  interaction and H-bonding are the dominant sorption mechanisms.

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## 1. Introduction

The fate and transport of endocrine-disrupting chemicals (EDCs), such as estrogens, in the soil-water system are a growing concern, because these bioactive compounds have been widely detected in wastewater, reclaimed water and rivers (Khanal et al., 2006; Citulski and Farahbakhsh, 2010; Song et al., 2018). In the study of Koplin et al. (2002), 139 streams were sampled in the United States and hormones were found in approximately 40% of them. In a study on Spanish groundwater samples (Jurado et al., 2019) for the emerging organic contaminants (EOCs), including several EDCs molecules, showed that the most of these substances are usually detected at low ng/L concentration range in the groundwater bodies. However, they are reported at concentrations >100 ng/L, consequently, it is required to set up threshold quality values, because groundwater is a valuable water resource worldwide. The animal manure applied to agricultural soils is considered to be a major source of estrogens in rural environments (Hanselman et al., 2003; Scherr et al., 2008; Adeel et al., 2017), while in urban areas, the wastewater effluents are the major source of hormones (Schlüsener and Bester, 2008). Moreover, the EDCs can interfere with the endocrine system in humans and aquatic life (Hanselman et al., 2003; Windsor et al., 2018). However, the current, three-step cleaning technologies are not able to fully eliminate these hormones from the wastewater (Kim et al., 2015).

Ethinyl estradiol (EE2) is a synthetic analogue of estradiol (E2), and is a highly potent estrogen receptor agonist (Laurenson et al., 2014). EE2 is a component in many frequently used contraceptives (Robles, 2010). The growing use of EE2 in human medicine and livestock farming has led to an increase in their occurrence in the environment, that is why EE2 was added to the European Watch List in 2018 (EC, 2018). Research has shown that EE2 is extremely resistant to the oxidation in the environment owing to the ethynyl-group in the 17-position due to the high energy of C $\equiv$ C bonding (Li et al., 2013). Furthermore, EE2 showed the highest estrogenic potency in tests on fish, being 11–30 times more potent than E2 (Colman et al., 2009). Depending on the physical and chemical properties of the various soil phases, e.g. its pH, organic matter content and texture, EE2 can either be strongly bound to the soil and accumulate in the top soil layer or be moved down to deeper soil horizons and the groundwater (Tong et al., 2019).

A key process controlling the concentration, mobility, toxicity and fate of EDCs, including EE2, in the environment is their sorption

on soils (Yamamoto et al., 2016). The organic matter (OM) of soils and sediments was found to be the principal factor controlling the sorption of organic compounds (Lambert, 1968). Moreover, researchers have found that the adsorption of organic compounds by soils or sediments is affected not only by the quantity of soil organic matter (SOM), but also by the chemical properties of the OM (Cornelissen et al., 2005; Jakab et al., 2018; Ping and Lou, 2019). The composition, quality and binding features of organic matter are, in turn, influenced by several environmental factors. For example, a change in the redox state of a soil can cause an alteration in microbial activity, leading to the leaching of dissolved organic carbon into surface water and causing a depletion of the labile organic carbon fraction, while more recalcitrant compounds are left behind (Kalbitz, 2001). Furthermore, in anaerobic conditions the decomposition of plant residues is slower and less complete than under aerobic conditions, leading to the accumulation of slowly decomposing organic compounds such as lipids in the residue (Sahrawat, 2003; Loffredo et al., 2016).

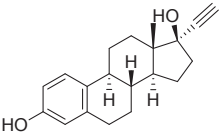
Although the sorption of EE2 has been widely studied in soils and sediments during the last decade (Bonin and Simpson, 2007; Sun et al., 2012; Mashtare et al., 2011; Li et al., 2013; Oliveira et al., 2019), the potential impact of this hormone on soils is still not fully understood, especially in environments where anaerobic conditions occur. Therefore, a series of experiments was conducted on soils having different oxidation stages with the following objectives: (i) to quantify the sorption of EE2 on the soils investigated, (ii) to reveal the intrinsic features of organic matter responsible for the differences in sorption by characterizing soil organic matter quality using FT-IR spectroscopy and (iii) to evaluate the parameters of isotherms used (Freundlich, Langmuir and Dubinin-Radushkevich) with respect to SOM quality.

## 2. Materials and methods

### 2.1. Chemicals

Standard (99%) 17 $\alpha$ -ethynylestradiol (EE2) and fluorescence grade acetonitrile and methanol were purchased from Sigma-Aldrich. Table 1 summarizes the physico-chemical properties of EE2. The ultra-pure water was produced using ELGA labwater (resistance 18.2  $\Omega$ ). The stock solutions were diluted in methanol and prepared in amber-stained borosilicate beakers. Aqueous solutions of EE2 were measured at different concentrations (ranging

**Table 1**  
Physiochemical properties of EE2.

Chemical structure	Molecular weight (g/mol)	Water solubility (mg/l)	Octanol/Water partition coefficient (log $k_{ow}$ )	Dissociation constants (pK $_a$ )
	296.41	9.20 $\pm$ 0.09 <sup>a</sup>	3.6–3.8 <sup>b</sup>	10.4 <sup>c</sup>

<sup>a</sup> Shareef et al. (2006).

<sup>b</sup> Zhang et al. (2010).

<sup>c</sup> Hurwitz and Liu (1977).

from 100 µg/L to 5000 µg/L) before instrumental analysis.

The sensitivity of a method is defined by the values of limit of detection (LOD) and limit of quantification (LOQ). The detection limit is the lowest concentration of drug in a sample which can be detected (LOD), while LOQ is the minimum quantifiable amount of analyte by the suggested method. The LOD and LOQ values were found to be 2.3 and 6.9 ng/L.

## 2.2. Soil sampling

Five soils were collected from different soil depths at three different locations in Ceglédbercel, Hungary (Fig. 1). The soil types of the samples were Eutric Arenosol (calic, humic), Mollic Oxygleyic Calcic Gleysol (loamic, hyperhumic) and Eutric Calcic Histosol (haplic), respectively, according to the World Reference Base (WRB) soil classification (IUSS Working Group WRB, 2006). The soil samples were denoted as: A\_20 (Arenosol sampled at 20 cm), G\_20 (Gleysol sampled at 20 cm), G\_40 (Gleysol sampled at 40 cm), H\_20 (Histosol sampled at 20 cm) and H\_80 (Histosol sampled at 80 cm). The redox potential measurements were performed using a combined Pt electrode with a reference. The electrodes were installed at depth of the soil samples by using a Pürckhauer hand auger. Eh, and pH values were recorded every 1 h for 9 months till the first frosts. Redox potential measurements were used to characterize the soils based on the duration of anoxic ( $E_h < 0$  mV) conditions: A\_20 (0% anoxic/year), G\_20 (65% anoxic/year), G\_40 (30% anoxic/year), H\_20 (85% anoxic/year) and H\_80 (100% anoxic/year).

## 2.3. Soil analysis

The samples were dried to constant weight at room temperature. To minimize the effect of the particle size variation on the adsorption of EE2, the dried soil samples were passed through a 250 µm metal sieve.

The soil pH was measured in 1:2.5 soil:water and soil:1 M KCl suspensions 12 h after mixing (MSZ-08-0206/2, 1978). The organic carbon and nitrogen contents were determined using a CNS elemental analyzer (Thermo Scientific Flashsmart) (Matejovic,

1997). The  $\text{CaCO}_3$  content was measured with a calcimeter using the Scheibler method (Loeppert and Suarez, 1996). The total iron content of the soils was determined by X-ray fluorescence spectroscopy (XRF) (Voglar and Lestan, 2013). The specific surface area (SSA) was calculated using the Brunauer–Emmett–Teller (BET) equation with multipoint adsorption isotherms of  $\text{N}_2$  at 77 K (Petersen et al., 1996). These physicochemical properties are listed in Table 2.

## 2.4. Adsorption experiment and sorption models

The sorption of EE2 was measured at room temperature ( $22 \pm 1^\circ\text{C}$ ) using a batch equilibration method. The sorption experiment was carried out at a 1:12 soil:solution ratio for 2 h. The soil:solution ratio was set to obtain enough amount of supernatant for the filtration, because as shown in preliminary studies at least 10 mL of supernatant is needed to saturate the filter. According to the results of preliminary kinetic experiments, 2 h was sufficient to reach the equilibrium. Prior to measuring the EE2 concentration the suspensions were centrifuged at 5200 rpm for 10 min (MPW-352RH) and filtered through a 0.45 µm glass filter (Chromafil® GF/PET-45/25). Before every experiments the stock solution of EE2 was prepared in methanol with concentration of 10 mg/L. The EE2 concentrations applied were as follows: 100, 500, 1000, 1500, 2000, 2500, 3000, 3500, 4000, 4500 and 5000 µg/L. To prevent the degradation of the EE2 during the batch experiment sodium-azide was added to each centrifuge tube. The measurements were done in dark under controlled temperature using 3 replicates. The 17 $\alpha$ -ethynylestradiol concentrations were measured by HPLC (Shimadzu Prominence LC-20AR), using a fluorescence detector with an excitation wavelength of 280 nm and an emission wavelength of 310 nm. The mobile phase was a 50-50% mixture of ultra-pure water (acidified with 10 mM  $\text{H}_3\text{PO}_4$ ) acetonitrile. The flow rate through the column (SunShell C18, 2.6 µm) was set at 0.5 mL/min at 40 °C. The injection volume of standards and samples was 10 µL. The amount of EE2 adsorbed on the soils was calculated using the following equation:

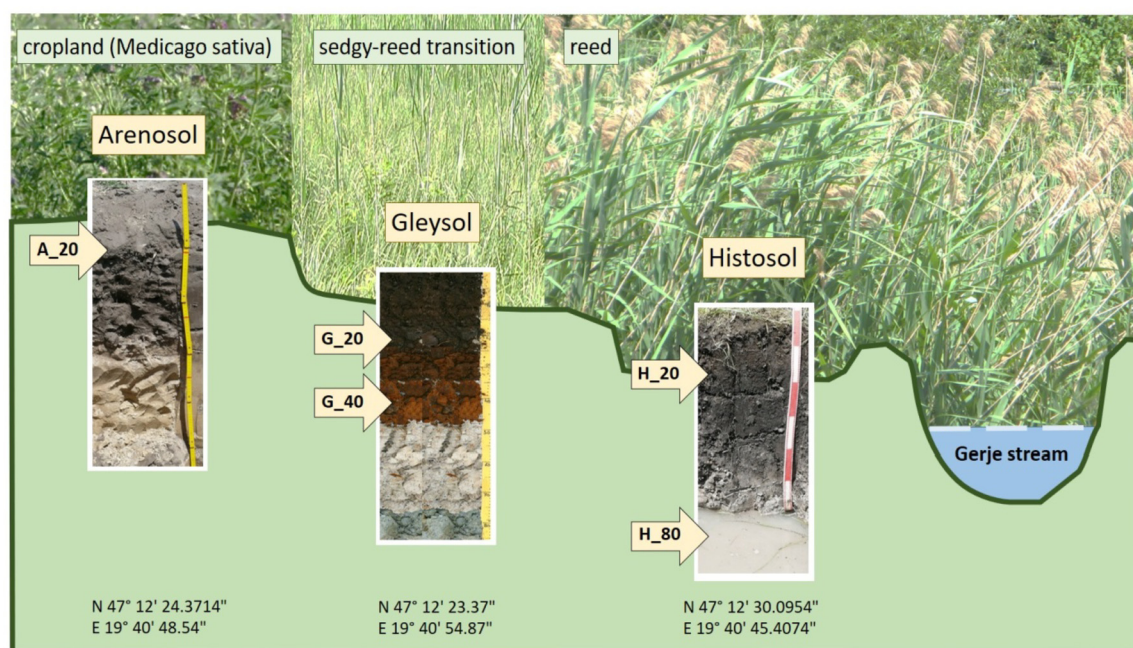


Fig. 1. The profiles and locations of the soil samples.

**Table 2**  
Basic parameters of the soils.

Sorbent	pH (H <sub>2</sub> O)	pH (KCl)	SOC (%)	C/N	CaCO <sub>3</sub> (%)	Fe (%)	SSA-N <sub>2</sub> (m <sup>2</sup> /g)
G_40	7.5	7.3	2.9	10.3	7.2	37.2	93.6
H_80	8.4	8.1	0.25	nd	7.5	1.4	34.7
G_20	7.3	7.2	14.6	8.9	36.0	3.0	12.4
A_20	7.9	7.8	1.8	9.3	11.3	0.9	4.9
H_20	7.5	7.2	24.4	8.0	55.4	1.2	5.6

nd: not determined as the N content of the sample was below the detection limit.

G\_40: Gleysol collected from 40 cm; G\_20: Gleysol collected from 20 cm; H\_80: Histosol collected from 80 cm; H\_20: Histosol collected from 20 cm; A\_20: Arenosol collected from 20 cm.

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

where  $q$  is the amount of EE2 adsorbed on the soil (mg/g),  $C_0$  and  $C_t$  are the initial and final concentrations (mg/L), respectively,  $V$  is the volume of the initial solution (L), and  $m$  is the mass of the soil (g).

The Freundlich, Langmuir and Dubinin-Radushkevich models were used to describe the adsorption of EE2 on selected soils.

The Freundlich model:

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

where  $q_e$  is the amount of EE2 adsorbed on the soil sample (μg/g),  $K_F$  is the Freundlich adsorption coefficient ((μg/g)/(μg/L)<sup>1/n</sup>),  $C_e$  is the equilibrium concentration of the aqueous phase (μg/L), and  $n$  is a dimensionless number which refers to the nonlinearity between the equilibrium concentration and the amount of EE2 adsorbed.

The Langmuir model:

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

where  $Q_{\max}$  is the maximum adsorption capacity (g/g) and  $K_L$  is the Langmuir fitting parameter (L/μg).

The Polanyi-based Dubinin-Radushkevich equation was applied to characterize the free energy of the adsorption:

$$q_e = q_m e^{(-\beta \epsilon^2)} \quad (4)$$

where  $q_m$  is the theoretical saturation capacity (μg/g),  $\beta$  is the isotherm constant (mol<sup>2</sup>kJ<sup>-2</sup>) and  $\epsilon$  is the Polanyi potential, which can be expressed by the formula below:

$$\epsilon = RT \ln \left( 1 + \frac{1}{C_e} \right) \quad (5)$$

where  $R$  is the gas constant (8.314 kJ/mol) and  $T$  is the absolute temperature (K).

The adsorption energy ( $E$ ) can be calculated from the following relationship:

$$E = \frac{1}{\sqrt{-2\beta}} \quad (6)$$

## 2.5. Preparation of humic acids

Humic acids (HA) were extracted from the soil samples with 0.1 M NaOH and purified using standard methods of the International Humic Substances Society (Swift, 1996). Briefly, 5 g of soil was shaken with 0.1 M HCl to remove the CaCO<sub>3</sub>, followed by the extraction of humic substances with 0.1 M NaOH for 24 h, after

which the solution was centrifuged, the supernatant was adjusted to a pH of 1 and 2, the solution was left to settle for a further 24 h. The precipitated HAs were then collected, washed and purified by dialysis. The remaining part of the soil sample (extracted with 0.1 M NaOH) was washed with water and dried at 60 °C; this sample contains the humin fraction (Hu) of the soil.

## 2.6. FTIR spectroscopy

The IR spectra of the HAs in the soils were obtained using a Bruker Vertex 70 with an ATR attachment containing a diamond crystal (Bruker Optics Ltd., Coventry, UK). The freeze-dried, powdered HA samples were measured with 64 scans and a resolution of 4 cm<sup>-1</sup> between 4000 and 400 cm<sup>-1</sup>. The IR spectra of the Hu (samples extracted with alkali) were measured with a Bruker Vertex 70 fitted with a diffuse reflectance attachment, again with 64 scans and a resolution of 4 cm<sup>-1</sup> between 4000 and 400 cm<sup>-1</sup>.

Relative absorbances (rA) were calculated by dividing the corrected peak height of a distinct peak by the sum of the heights of all peaks at 3070, 2920, 2850, 1730, 1630, 1510, 1450, 1370, 1270 and 1050 cm<sup>-1</sup> and multiplying it by 100 (rA = % of the sum of all peak heights from 2920 to 1050 cm<sup>-1</sup>).

The aromaticity index based on Chefetz et al. (1996) was calculated by dividing the intensity of absorption around 1620 cm<sup>-1</sup> ( $I_{1620}$ ) by the intensity of absorption at 2920 cm<sup>-1</sup> ( $I_{2920}$ ) for both HA (Ari\_HA) and NaOH-treated humin samples (Ari\_Hu).

## 2.7. Statistical analysis

Regression analysis was performed to quantify how the relative absorbance values (rA) of different samples influenced the parameters of the isotherms using Origin Pro 2018 software. To describe the relationship between two variables while controlling or adjusting the effect of one or more additional variables (e.g. SOM) a partial correlation analysis was made using SPSS 22.0 software.

## 3. Results and discussion

### 3.1. Characterization of soils and their organic fractions

The basic soil properties of the soils used in this experiment are shown in Table 2. The organic carbon content of the samples was in the order: H\_20 > G\_20 > G\_40 > A\_20 > H\_80. The organic matter accumulation in the upper part of the H and G profiles is likely due to the anoxic conditions during most of the year (85 and 65%). Although the H\_80 sample was formed under absolutely anoxic conditions, its organic matter content was very low (0.25%). This could be explained by the fact that this layer was formed under the root zone, thus excluding organic matter accumulation.

The specific surface area of the sorbents was in the order: G\_40 > H\_80 > G\_20 > H\_20 > A\_20. Although SSA can be



considered as one of the factors controlling the sorption of organic pollutants on soils (Kaiser et al., 1996; Nambu and Yonebayashi, 2000), the BET-N<sub>2</sub> SSA of soils is inversely related to their OC content (e.g. Kaiser et al., 1996; Rasmussen et al., 2018). Consequently, the soil sample with the highest OC content had one of the lowest SSA values (H\_20: 5.6 m<sup>2</sup>/g). It can be concluded from the Fe contents of the samples that substantial amounts of Fe-oxides may be formed in the lower part of the G profile, which may help to explain the very high SSA value of this sample (Heister, 2016).

FT-IR analysis was carried out on the HAs and Hu in adsorbent samples from different soil layers to characterize the organic matter in terms of the spectral data. The most characteristic bands in the IR spectra are presented in Table 3.

In general, the spectra of the HA samples were very similar and showed the same spectral patterns (Fig. 2). The evolution of the FTIR absorbance of humic substances was evaluated by establishing the ratios between the main absorbance peaks (Table 4).

As shown in Table 4, the band at around 3070 cm<sup>-1</sup>, which could be assigned as aromatic C–H stretching vibration, differed greatly among the HA samples; those formed at lower depths, below the root zone, and under more anoxic conditions had lower relative absorbance values than those from the top soil. The bands at around 2920 cm<sup>-1</sup> and 2850 cm<sup>-1</sup> represent the aliphatic C–H vibrations of aliphatic methyl and methylene groups. In HA samples from H\_80 and G\_40, these peaks had lower intensity than in those from the upper part of the soils, probably because roots release a wide range of organic compounds, including aliphatic organic acids (Nardi et al., 2000). The 1720 cm<sup>-1</sup> peak, which is characteristic of the aromatic carboxylic acids in HA samples, was low in samples with the highest OC content (G\_20 and H\_20) presumably due to the tendency for organic matter to retain water, leading to anaerobic conditions where oxidation is limited. A doublet in the spectrum at around 1650 cm<sup>-1</sup> indicative of the aromatic C=C and carboxylate ions (Niemeyer et al., 1992) was slightly greater for the HA of aerobic samples than for hydromorphic ones, suggesting that aerated soils contain more carboxylic groups in HA molecules. This was especially true for the A\_20 sample, which is under aerobic conditions throughout year.

There were no significant differences in the band at 1420 cm<sup>-1</sup>, which is assigned to aromatic ring stretch COO<sup>-</sup> of humic acids (Inbar et al., 1990; Dhillon et al., 2017). In samples G\_20 and H\_20 the relative absorbance of the band at around 1220 cm<sup>-1</sup>, which can be assigned to phenolic-OH, was the highest, indicating organic matter accumulation. Substantial differences were found in the spectra of HA samples at around 1035 cm<sup>-1</sup> (C–O vibrations of polysaccharides, carbohydrates). A significant difference was observed for the band at 1050 cm<sup>-1</sup> between waterlogged and aerated samples; the HA fraction of the aerobic soil samples contained more O-alkyl structures than the hydromorphic HAs, which is in agreement with the findings of Artz et al. (2008) that the concentration of polysaccharides decreases with depths.

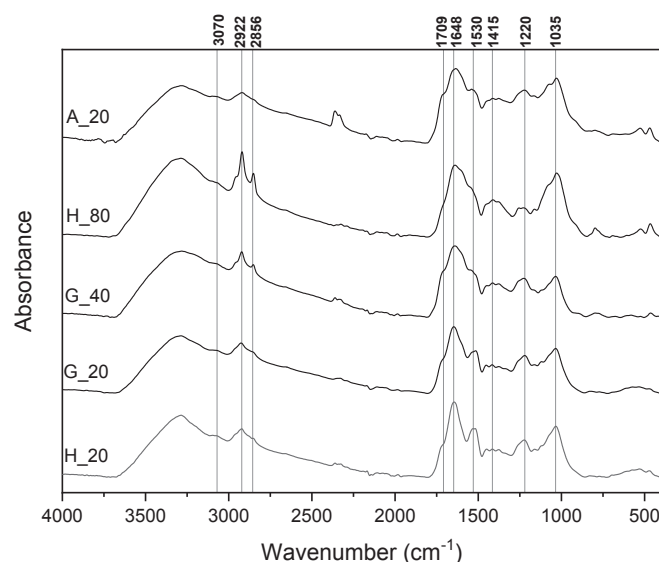


Fig. 2. FT-IR spectra of HA samples.

The aromaticity index ( $I_{1620}/I_{2920}$ , Ari), calculated from the FTIR data ranged from 5.2 to 11.3 for the extracted humic acids, while there was enormous variability for the humin fraction (Table 4). Dick et al. (2006) found a strong relationship between the aromaticity index obtained from FTIR and the ratio of aryl and alkyl C calculated from the NMR data, so Ari could be a useful tool for characterizing organic matter. Based on this evaluation it can be stated that the A\_20 sample had the most aromatic HA. In the case of H\_80 and A\_20, the aromaticity of the humin fraction was found to be extremely high, this value was very low for G\_40 and G\_20. This confirmed findings suggesting that the organic carbon associated with mineral surfaces in aquifers has high aromatic carbon (Murphy et al., 1990; Fu et al., 2018).

### 3.2. Adsorption of EE2 on soils

Batch adsorption experiments were conducted with EE2 on soil samples with organic matter in different stages of decomposition. The adsorption isotherms for EE2 on soils collected from field sites, shown in Fig. 3, were found to be nonlinear for all the soil samples, which is consistent with other reports on the adsorption of EE2 (Bonin and Simpson, 2007; Sun et al., 2012). The sorption of EE2 increased in the order H\_80 < A\_20 < G\_40 < H\_20 < G\_20.

The Freundlich, Langmuir and Dubinin-Radushkevich equations were used to fit the isotherm data (Table 5). Although the Langmuir model fitted the data best (Table 5), the Freundlich and DR parameters were also calculated for sake of comparison.

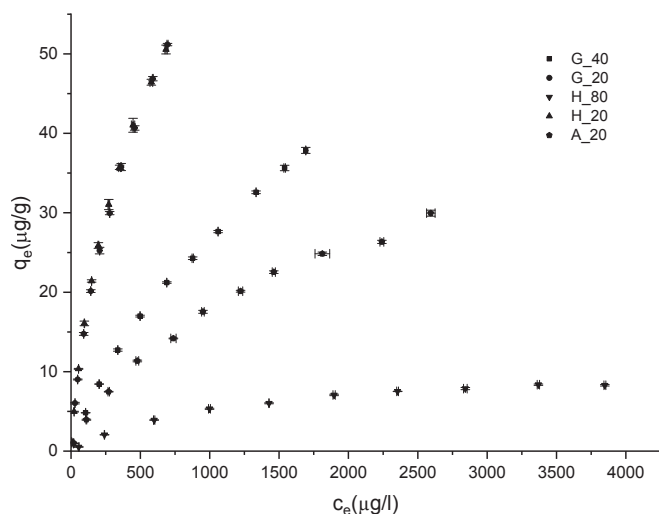
**Table 3**  
Assignment of the principal IR absorption bands.

Absorption band (cm <sup>-1</sup> )	Assignment	Reference
~3070 <sup>a</sup> (3030)	Aromatic C–H stretching	Niemeyer et al. (1992)
~2922 (2920)	Aliphatic asymmetric C–H stretching	Haberhauer et al. (1998)
~2856 (2840)	Aliphatic symmetric C–H stretching	Niemeyer et al. (1992)
~1709 (1720)	–C=O stretching in –COOH	Niemeyer et al. (1992)
~1648 (1632)	Aromatic C=C vibrations and C=O vibrations of carboxylic acid anions	Tivet et al. (2013)
~1530 (1510)	Aromatic C=C stretching	Haberhauer et al. (1998)
~1415 (1420)	O–H deformation and C–O stretching of phenolic OH	Wu et al. (2016)
~1220 (1215)	–C–O, phenolic	Niemeyer et al. (1992)
~1035 (1050)	Combination of C–O stretching and O–H deformation	Grube et al. (2006)

<sup>a</sup> Values measured in this study, band values found in the given reference are in parenthesis.

**Table 4**  
Relative absorbance (%) of the sum of all selected peak heights in the FTIR spectra.

Sample	1035	1220	1415	1540	1648	1709	2858	2922	3070	Ari_HA	Ari_Hu
G_40	21.9	11.6	5.6	12.7	25.1	15.6	2.5	4.8	0.26	5.2	1.1
H_80	23.3	5.8	5.9	13.6	26.7	16.7	2.7	5.1	0.28	5.2	84.1
G_20	26.0	13.9	6.4	12.7	27.9	11.4	1.7	3.4	0.51	7.2	1.9
A_20	30.0	10.9	4.4	11.4	29.1	15.9	1.0	2.1	0.26	11.3	161.2
H_20	27.6	12.5	5.5	14.5	27.1	9.7	1.4	2.9	0.45	8.7	2.6



**Fig. 3.** Adsorption isotherms of EE2 on soils. G\_40: Gleysol collected from 40 cm; G\_20: Gleysol collected from 20 cm; H\_80: Histosol collected from 80 cm; H\_20: Histosol collected from 20 cm; A\_20: Arenosol collected from 20 cm.

In the Langmuir equation  $Q_{\max}$  shows the maximum adsorption capacity for EE2, while in the Freundlich model the constant  $K_F$  is an approximate indicator of the relative adsorption capacity (Dada et al., 2012). The samples showed considerable variability for  $Q_{\max}$  (Table 5).

Soil organic matter is believed to be the most important component in the adsorption of hydrophobic organic pollutants (Chiou et al., 1979; Bielská et al., 2018), which explain why in this experiment the  $Q_{\max}$  values ranged from 10.7 to 83.6 mg/g in the order  $G_{20} > H_{20} > G_{40} > A_{20} > H_{80}$ , reflecting the organic matter content of the soils. The calculated  $K_F$  values showed the same pattern as  $Q_{\max}$ , indicating that the soils exhibited different sorption capacity for EE2, probably due to the different redox states, which resulted in different SOM quantity and quality. For example, the H\_20 sample had the highest  $K_F$  value of 1.09, while the H\_80 had the lowest, with a value of 0.22. Apart from the two soils with very high organic matter content,  $K_F$  values were comparable to those reported in the literature. Hildebrand et al. (2006) found  $K_F$  values ranging from 0.028 to 0.121, Sun et al. (2012) values of

0.064–0.144, while in the study of Ma et al. (2016) the  $K_F$  values are ranging from 0.181 to 0.316. The high  $K_F$ , and consequently the high adsorption capacity, of H\_20 and G\_20 was probably caused by the higher organic matter content compared to that in the studies mentioned above.

In order to compare samples with different organic matter content, it was proposed to normalize solid-water distribution coefficients to the organic carbon content of the sorbents (Chiou et al., 1979; Karickhoff, 1980). The Freundlich  $K_F$  and Langmuir  $Q_{\max}$  values were organic carbon-normalised to cancel the effect of the abundance of OM ( $K_{OC}$  and  $Q_{OC}$ ; Table 5). The higher the value of these normalised parameters, the higher the contribution of minerals to the adsorption of EE2 on the soils.  $Q_{OC}$  ranged from 320 to 4280 mg/g, in the order  $H_{80} > G_{40} > A_{20} > G_{20} > H_{20}$ , and a similar pattern was found in the case of  $K_{OC}$  (Table 5). This can be attributed to the large contribution of the mineral phase to the sorption of EE2 on H\_80, G\_40 and A\_20. It could be expected that these samples would have greater SSA, but in fact the A\_20 sample had the lowest SSA value (Table 2). This may indicate that QOC only has limited applicability for complex systems such as soils, which contain minerals and organic matter with varying quality.

The sorption isotherms were observed to be nonlinear for all the soil samples with, Freundlich  $n$  values ranging from 0.45 to 0.68 (Table 5). Sorption isotherm nonlinearity has also been reported for estrogen compounds by other researchers (Yu et al., 2004; Lai et al., 2000; Sun et al., 2012; Ma et al., 2016). For example, Sun et al. (2012) reported  $n$  values in the range of 0.73–0.90 for EE2. In the present study, the lowest  $n$  value was recorded for the H\_80 sample, indicating the lowest energy of adsorption for EE2, while the  $n$  value of the G\_40 sample was much closer to unity, suggesting that the sorption of EE2 by the soil increased more linearly when increasing concentrations of EE2 were added to the soil. It also showed that the adsorption of EE2 molecules on G\_40 was much stronger. This was confirmed by the Langmuir  $K_L$  values and the adsorption energy calculated from the Dubinin-Radushkevich model (Table 5). Based on the dual-mode model (Pignatello, 1998), in which SOM is classified into two domains, namely, the hard and soft carbon domains, another interpretation of the Freundlich  $n$  parameter is possible. The nonlinearity of EE2 sorption on soils was attributed to the specific sorption between the functional sorbate group and the specific sorption sites of hard carbon (Li et al., 2013), because the degree of curvature of the isotherm

**Table 5**  
Parameters of the isotherms.

Sample	Freundlich				Langmuir				Dubinin-Radushkevich	
	$K_F ((\mu\text{g/g})/(\mu\text{g/L})^{1/n})$	$K_{OC}^a$	$n$	$R^2$	$Q_{\max} (\text{g/g})$	$Q_{OC}^b$	$K_L (10^{-3}) (\text{L}/\mu\text{g})$	$R^2$	$E (\text{kJ/mol})$	$R^2$
A_20	0.31	17.2	0.58	0.994	45.9	2550	0.65	0.996	3.6	0.883
G_20	0.94	6.4	0.61	0.998	83.6	573	2.14	0.996	13.4	0.842
H_20	1.09	4.5	0.59	0.996	78.2	320	2.52	0.998	12.8	0.820
G_40	0.23	7.9	0.68	0.999	76.3	2631	0.56	0.998	4.4	0.891
H_80	0.22	88.0	0.45	0.969	10.7	4280	0.97	0.998	3.2	0.906

<sup>a</sup>  $K_{OC}$  organic carbon-normalised  $K_F$ .

<sup>b</sup>  $Q_{OC}$  organic carbon-normalised  $Q_{\max}$ .

expressed by the value of  $n$  represents the contribution of the hard carbon domain: the smaller the  $n$  value, the greater the hard carbon contribution (Pignatello, 1998). In the present study, the smaller  $n$  value of the H\_80 sample can be explained by the extremely high aromaticity of the humin fraction (Table 4), which may be correspond to the amount of the hard carbon phase.

### 3.3. Control of EE2 sorption mechanisms by the organic matter quality

The organic matter content was found to be the most important property of the soils with regard to the capacity and strength of sorption. However, since soils consist of many inorganic and organic compounds with various structures and properties, several mechanisms can be assumed to function during the sorption process.

The  $Q_{\max}$  of the soils in this study did not always reflect the increasing organic carbon content of the soils (Fig. 4). The accessibility of binding sites in the soils to organic pollutants may be influenced by the OM of the sorbents (Chen et al., 2005; Ren et al., 2018). For example, several studies have demonstrated that the removal of soil lipid resulted in an increase in organic compound sorption, suggesting that soil lipid components may block high affinity sorption sites (Kohl and Rice, 1999; Ahangar et al., 2009). It was also reported by Mitchell and Simpson (2012) that the O-alkyl components (carbohydrates and peptides) of the organic matter block high affinity sorption sites in soil OM and decrease the sorption of organic contaminants.

Besides the blocking effect of molecules in the soil organic matter, numerous investigations have revealed that the removal of organic matter increases the external specific surface area of the soils (Burford et al., 1964; Kaiser and Guggenberger, 2003; Rabot et al., 2017). Correspondingly, organic coatings reduce the surface of the soil particles and mineral surfaces (Kaiser and Zech, 1998; Rabot et al., 2017). Thus, there may be two reasons for relative decrease in sorption capacity in the two samples with high organic matter content: (i) the large amount of organic matter may reduce EE2 binding on the surface by blocking high affinity sites, and (ii)

the decrease in SSA with increasing OC content may lower the ability to adsorb EE2 molecules. In the present study, the relative content of O-alkyl compounds assessed by FTIR spectra at  $1035\text{ cm}^{-1}$  was high for G\_20 and H\_20 (26.0 and 27.6, respectively), so the hypothesis of Mitchell and Simpson (2012) was confirmed.

Our own hypothesis in this research was that the binding of hydrophobic molecules such as ethinyl estradiol could be significantly affected by not only by the amount of organic matter in different degradation stages but also by its quality. Several mechanisms may be responsible for the binding of such molecules: hydrophobic interaction,  $\pi$ - $\pi$  interaction and H-bonding. In simple system of sediments the hydrophobic interaction was found to be the main adsorption mechanism (Sun and Zhou, 2015), but in the study of Yamamoto et al. (2003) the hydrophobic interaction did not prove to be the predominant sorption mechanism. In the present study, no significant relationship was found between the quantity of alkyl compounds and any of the adsorption parameters (e.g. E, Q,  $K_F$ ) suggesting that the hydrophobic interaction was not the dominant mechanism in the system investigated.

According to a previous study (Grathwohl, 1990) the sorption of EE2 will decrease with an increase in the degree of weathering of the organic matter. As the O-alkyl-C abundance can serve as a sensitive indicator of the decomposition and oxidation organic matter (Tfaily et al., 2014), the relative absorbance of humic acids at  $1035\text{ cm}^{-1}$  (Table 3) as a proxy for the decomposition and oxidation of the soil organic matter. The data showed that the upper part of the soils had high relative amounts of O-alkyl compounds due to the enhanced microbial activity. These results contradicted Grathwohl's hypothesis, since the decomposition and oxidation of the soil organic matter increased the adsorption of EE2, especially for A\_20, which had very high adsorption capacity for organic matter.

The aromatic structures of the organic matter in soils and sediments are believed to be important binding sites (Lima et al., 2012) by creating  $\pi$ - $\pi$  bonds between the aromatic moieties of organic matter and the EE2 molecule. In the upper and well aerated region the organic matter was found to have higher aromaticity, expressed by the aromaticity index and the relative absorbance at  $3070\text{ cm}^{-1}$  (Table 4), likely due to the abundance of plant residues with high lignin content in these profiles. The interaction between the aromatic compounds of the organic matter and EE2 molecules was demonstrated by the strong relationship between  $rA_{3070}$  and the adsorption energy of EE2 (Fig. 5). A partial correlation analysis was made to check the effect of the organic matter content of the samples, and significant correlation ( $r = 0.927$ ) was detected. It can thus be concluded that a strong relationship exists between  $rA_{3070}$  and E.

As shown in Fig. 6,  $Q_{\max}$  correlated with the amount of phenolic groups in SOM for EE2. The highest  $Q_{\max}$  values were found for the G\_20, H\_20 and G\_40 samples. This high affinity may be the result of interactions between the abundant phenolic groups in the organic matter of these soils and the phenolic group in the steroid EE2. Phenolic hydroxyl moieties are known to be active proton donors, and these can bond to EE2 molecules via intermolecular hydrogen bonds (Han et al., 2012). The strong relationship between adsorption capacity and the relative amount of phenolic groups confirms the findings of Yamamoto et al. (2003), who reported that the sorption of estrogen compounds on dissolved organic matter correlated well with the amount of phenolic groups.

In Fig. 7, a strong correlation was evident between the  $rA_{1709}$  values, which show the amount of carboxylic groups, and the energy of adsorption for EE2. Most carboxylic groups are negatively charged at the pH of the soils investigated here, and can thus significantly contribute to the sorption of cations. However, the selected endocrine disruptors with  $pK_a$ s above 10 (Table 1) are

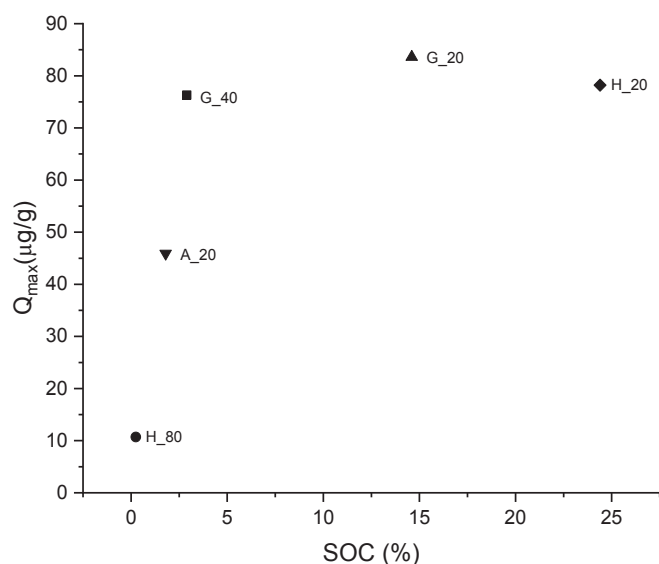
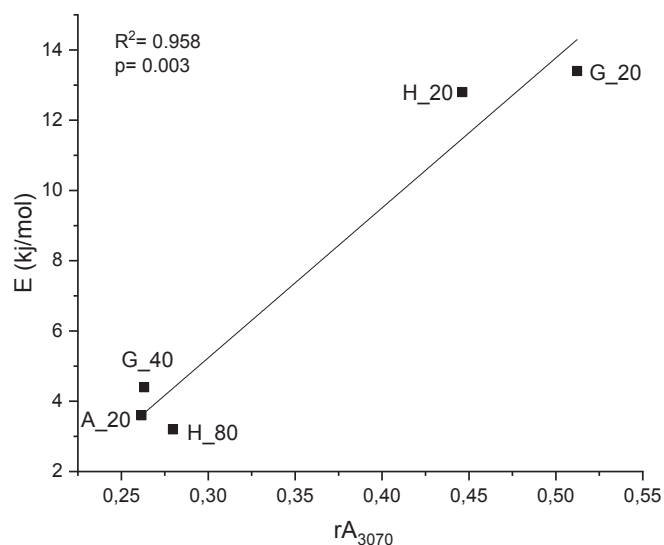
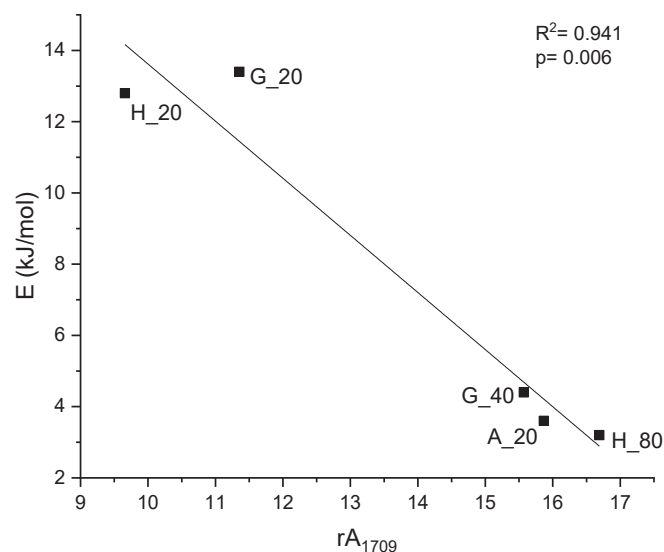


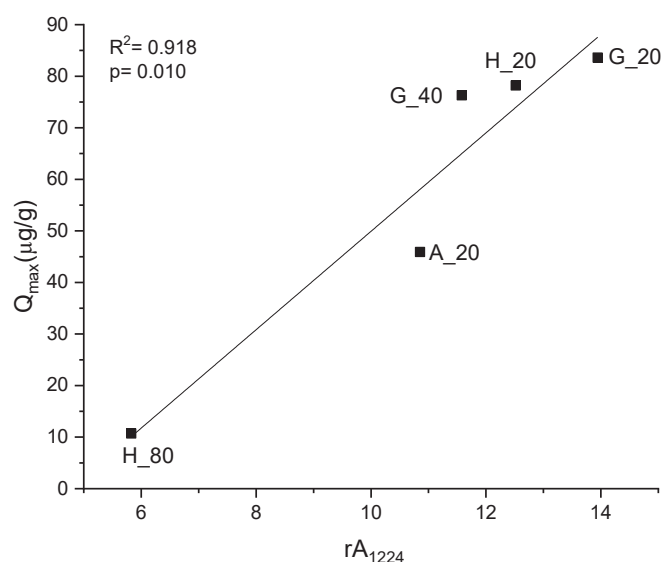
Fig. 4. Relationship between the organic carbon content of the soil (SOC) and the maximum adsorption capacity ( $Q_{\max}$ ). G\_40: Gleysol collected from 40 cm; G\_20: Gleysol collected from 20 cm; H\_80: Histosol collected from 80 cm; H\_20: Histosol collected from 20 cm; A\_20: Arenosol collected from 20 cm.



**Fig. 5.** Relationship between the relative absorbance of the humic acids in the samples at 3070  $\text{cm}^{-1}$  ( $rA_{3070}$ ) and the adsorption energy (E). G\_40: Gleysol collected from 40 cm; G\_20: Gleysol collected from 20 cm; H\_80: Histosol collected from 80 cm; H\_20: Histosol collected from 20 cm; A\_20: Arenosol collected from 20 cm.



**Fig. 7.** Relationship between the relative absorbance of the humic acids of the samples at 1709  $\text{cm}^{-1}$  ( $rA_{1709}$ ) and the energy of adsorption (E). G\_40: Gleysol collected from 40 cm; G\_20: Gleysol collected from 20 cm; H\_80: Histosol collected from 80 cm; H\_20: Histosol collected from 20 cm; A\_20: Arenosol collected from 20 cm.



**Fig. 6.** Relationship between the relative absorbance of the humic acids of the samples at 1224  $\text{cm}^{-1}$  ( $rA_{1224}$ ) and maximum adsorption capacity ( $Q_{\text{max}}$ ). G\_40: Gleysol collected from 40 cm; G\_20: Gleysol collected from 20 cm; H\_80: Histosol collected from 80 cm; H\_20: Histosol collected from 20 cm; A\_20: Arenosol collected from 20 cm.

predominantly non-ionic at these pH values, and their charge contributions to overall sorption are small. The lowest carboxylic group concentration was found in the G\_20 and H\_20 samples probably due to their very high organic matter content, which has a substantial ability to retain water.

#### 4. Conclusions

This study investigated the adsorption of EE2 on soils with organic matter in different decomposition stages and with different quality based on redox status. The oxidation state of the soils, which may influence the decomposition of organic matter, was

found to be the reason for differences in the organic matter quality of the soils, which can be expressed in terms of decomposition rate and litter quality. More aromatic and phenolic compounds were found in reduced, anaerobic layers where substantial organic matter accumulation occurred. These two types of compounds resulted from the limited decomposition of lignin-rich plant residues due to the limited oxygen supply to the microbes and were found to be sites with the highest affinity for the adsorption of EE2. As a result, hydromorphic soils, in which organic matter accumulation occurred, were found to have high adsorption capacity for EE2, not only due to their high organic matter content, but also as a consequence of the chemical structures of the OM. These sorption data could be useful for predicting the ecological risks of EE2 through an understanding of its fate in soils forming under different oxidation conditions, and for application in transport and risk assessment models.

#### Conflicts of interest

There is no conflict of interest between any of the authors.

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#### References

- Adeel, M., Song, X., Wang, Y., Francis, D., Yang, Y., 2017. Environmental impact of estrogens on human, animal and plant life: a critical review. *Environ. Int.* 99, 107–119.
- Ahangar, A.G., Smernik, R.J., Kookana, R.S., Chittleborough, D.J., 2009. The effect of lipids on the sorption of diuron and phenanthrene in soils. *Chemosphere* 74, 1062–1068.
- Artz, R.R.E., Chapman, S.J., Robertson, A.H.J., Potts, J.M., Laggoun-Defarge, F., Gogo, S., Comont, L., Disnar, J.R., Francez, A.J., 2008. FTIR spectroscopy can be used as a screening tool for organic matter quality in regenerating cutover peatlands. *Soil Biol. Biochem.* 40, 515–527.
- Bielská, L., Škulcová, L., Neuwirthová, N., Cornelissen, G., Hale, S.E., 2018. Sorption, bioavailability and ecotoxic effects of hydrophobic organic compounds in biochar amended soils. *Sci. Total Environ.* 624, 78–86.
- Bonin, J.L., Simpson, M.J., 2007. Sorption of steroid estrogens to soil and soil constituents in single- and multi-sorbate systems. *Environ. Toxicol. Chem.* 26,



- 2604–2610.
- Burford, J.R., Deshpande, T.L., Greenland, D.J., Quirk, J.P., 1964. Influence of organic materials on the determination of the specific surface areas of soils. *J. Soil Sci.* 15, 192–201.
- Chefetz, B., Hatcher, P., Hadar, Y., Chen, Y., 1996. Chemical and biological characterization of organic matter during composting of municipal solid waste. *J. Environ. Qual.* 25, 776–785.
- Chen, B., Johnson, E.J., Chefetz, B., Zhu, L., Xing, B., 2005. Sorption of polar and nonpolar aromatic organic contaminants by plant cuticular materials: role of polarity and accessibility. *Environ. Sci. Technol.* 39, 6138–6146.
- Chiou, C.T., Peters, L.J., Freed, V.H., 1979. A physical concept of soil-water equilibria for nonionic compounds. *Science* 206, 831.
- Citlowski, J.A., Farahbakhsh, K., 2010. Fate of endocrine-active compounds during municipal biosolids treatment: a review. *Environ. Sci. Technol.* 44 (22), 8367–8376.
- Colman, J.R., Baldwin, D., Johnson, L.L., Scholz, N.L., 2009. Effects of the synthetic estrogen, 17 $\alpha$ -ethinylestradiol, on aggression and courtship behaviour in male zebrafish (*Danio rerio*). *Aquat. Toxicol.* 91, 346–354.
- Cornelissen, G., Gustafsson, O., Bucheli, T.D., Jonker, M.T.O., Koelmans, A.A., Van Noort, P.C.M., 2005. Extensive sorption of organic compounds to black carbon, coal, and kerogen in sediments and soils: mechanisms and consequences for distribution, bioaccumulation, and biodegradation. *Environ. Sci. Technol.* 39, 6881–6895.
- 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<sup>2+</sup> onto phosphoric acid modified rice husk. *IOSR J. Appl. Chem.* 3 (1), 38–45.
- Dhillon, G.S., Gillespie, A., Peak, D., Van Rees, K.C.J., 2017. Spectroscopic investigation of soil organic matter composition for shelterbelt agroforestry systems. *Geoderma* 298, 1–13.
- Dick, D.P., Knicker, H., Ávila, L.G., Inda Jr., A.V., Giasson, E., Bissani, C.A., 2006. Organic matter in constructed soils from a coal mining area in southern Brazil. *Org. Geochem.* 37, 1537–1545.
- EC, 2018. Commission Implementing Decision (EU) 2018/840 of 5 June 2018 Establishing a Watch List of Substances for Union-wide Monitoring in the Field of Water Policy Pursuant to Directive 2008/105/EC of the European Parliament and of the Council and Repealing Commission Implementing Decision (EU) 2015/495.
- Fu, H., Wei, C., Qu, X., Li, H., Zhu, D., 2018. Strong binding of apolar hydrophobic organic contaminants by dissolved black carbon released from biochar: a mechanism of pseudomolecular partition and environmental implications. *Environ. Pollut.* 232, 402–410.
- Grathwohl, P., 1990. Influence of organic matter from soils and sediments from various origins on sorption of some chlorinated aliphatic hydrocarbons: implications on Koc correlations. *Environ. Sci. Technol.* 24, 1687–1693.
- Grube, M., Lin, J.G., Lee, P.H., Kokorevicha, S., 2006. Evaluation of sewage sludge-based compost by FT-IR spectroscopy. *Geoderma* 130 (3–4), 324–333.
- Haberhauer, G., Rafferty, B., Strebl, F., Gerzabek, M.H., 1998. Comparison of the composition of forest soil litter derived from three different sites at various decomposition stages using FTIR spectroscopy. *Geoderma* 83 (3–4), 331–342.
- Han, J., Wei, Q., Suwan, M., Wei, G., 2012. Removal of ethinylestradiol (EE2) from water via adsorption on aliphatic polyamides. *Water Res.* 46 (17), 5715–5724.
- Hanselman, T.A., Graetz, D.A., Wilkie, A.C., 2003. Manure-borne estrogens as potential environmental contaminants: a review. *Environ. Sci. Technol.* 37, 5471–5478.
- Heister, K., 2016. How accessible is the specific surface area of minerals? A comparative study with Al-containing minerals as model substances. *Geoderma* 263, 8–15.
- Hildebrand, C., Londry, K.L., Farenhorst, A., 2006. Sorption and desorption of three endocrine disruptors in soils. *Environmental Science and Health, Part B* 41 (6), 907–921.
- Hurwitz, A.R., Liu, S.T., 1977. Determination of aqueous solubility and pKa values of estrogens. *J. Pharm. Sci.* 66, 624–627.
- Inbar, Y., Chen, Y., Hadar, Y., 1990. Humic substances formed during the composting of organic matter. *Soil Sci. Soc. Am. J.* 54, 1316–1323.
- IUSS Working Group, WRB, 2006. World Reference Base for Soil Resources. World Soil Resources Report No. 103. FAO, Rome.
- Jakab, G., Rieder, A., Vancsik, A.V., Szalai, Z., 2018. Soil organic matter characterisation by photometric indices or photon correlation spectroscopy: are they comparable? *Hungarian Geographical Bulletin* 67/2, 109–120.
- Jurado, A., Walther, M., Díaz-Cruz, S., 2019. Occurrence, fate and environmental risk assessment of the organic microcontaminants included in the watch lists set by EU decisions 2015/495 and 2018/840 in the groundwater of Spain. *Sci. Total Environ.* 663, 285–296.
- Kaiser, K., Guggenberger, G., 2003. Mineral surfaces and soil organic matter. *Eur. J. Soil Sci.* 54, 219–236.
- Kaiser, K., Zech, W., 1998. Soil dissolved organic matter sorption as influenced by organic and sesquioxide coatings and sorbed sulfate. *Soil Sci. Soc. Am. J.* 62, 129–136.
- Kaiser, K., Guggenberger, G., Zech, W., 1996. Sorption of DOM and DOM fractions to forest soils. *Geoderma* 74, 281–303.
- Kalbitz, K., 2001. Properties of organic matter in soil solution in a German fen area as dependent on land use and depth. *Geoderma* 104, 203–214.
- Karickhoff, S.W., 1980. Sorption kinetics of HOCs in natural sediments. In: Baker, R.A. (Ed.), *Contaminants and Sediments*, vol. 2. Ann Arbor Press, Ann Arbor, MI, p. 193.
- Khanal, S.K., Xie, B., Thompson, M.L., Sung, S., Ong, S.K., van Leeuwen, J.H., 2006. Fate, transport, and biodegradation of natural estrogens in the environment and engineered systems. *Environ. Sci. Technol.* 40, 6537–6546.
- Kim, S.H., Tian, Q., Fang, J.S., Sung, S.W., 2015. Removal of 17- $\beta$ -estradiol in water by sonolysis. *Int. Biodeterior. Biodegrad.* 102, 11–14.
- Kohl, S.D., Rice, J.A., 1999. Contribution of lipids to the nonlinear sorption of polycyclic aromatic hydrocarbons to soil organic matter. *Org. Geochem.* 30, 929–936.
- Koplin, 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, 1990–2000: a national reconnaissance. *Environ. Sci. Technol.* 36, 1202–1211.
- Lai, K.M., Johnson, K.L., Scrimshaw, M.D., Lester, J.N., 2000. Binding of waterborne steroid estrogens to solid phases in river and estuarine systems. *Environ. Sci. Technol.* 34, 3890–3894.
- Lambert, S.M., 1968. Omega, a useful index of soil sorption equilibria. *J. Agric. Food Chem.* 16, 340.
- Laurenson, J.P., Bloom, R.A., Page, S., Sadrieh, N., 2014. Ethinyl estradiol and other human pharmaceutical estrogens in the aquatic environment: a review of recent risk assessment data. *AAPS J.* 16, 299–310.
- 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.
- Lima, D., Schneider, R.J., Esteves, V.I., 2012. Sorption behavior of EE2 on soils subjected to different long-term organic amendments. *Sci. Total Environ.* 423, 120–124.
- Loeppert, R.H., Suarez, D.L., 1996. Carbonate and gypsum. In: Bigham, J.M., et al. (Eds.), *Methods of Soil Analysis. Part 3. — Chemical Methods. SSSA and ASA, Madison, Wisconsin, USA*, pp. 437–474.
- Loffredo, I., Castellana, G., Taskin, E., 2016. A two-step approach to eliminate pesticides and estrogens from a wastewater and reduce its phytotoxicity: adsorption onto plant-derived materials and fungal degradation. *Water, Air, Soil Pollut.* 227, 1–12.
- Ma, W., Nie, C., Gao, X., Qu, D., Lun, X., Chen, B., 2016. Sorption characteristics and factors affecting the adsorption behavior of bisphenol A and 17 $\beta$ -estradiol/ethinylestradiol in river- and farmland-based artificial groundwater recharge with reclaimed water. *Desalin. Water Treat.* 57, 8015–8025.
- Mashtare, M.L., Khan, B., Lee, L.S., 2011. Evaluating stereoselective sorption by soils of 17 $\alpha$ -estradiol and 17 $\beta$ -estradiol. *Chemosphere* 82, 847–852.
- Matejovic, I., 1997. Determination of carbon and nitrogen in samples of various soils by dry combustion. *Commun. Soil Sci. Plant Anal.* 28, 1499–1511.
- Mitchell, P.J., Simpson, M.J., 2012. High affinity sorption domains in soil are blocked by polar soil organic matter components. *Environ. Sci. Technol.* 47, 412–419.
- MSZ-08-0206/2, 1978. Evaluation of Some Chemical Properties of the Soil. Laboratory Tests. (pH Value, Phenolphthaleine Alkalinity Expressed in Soda, All Water Soluble Salts, Hydrolite (Y1-value) and Exchanging Acidity (Y2-value)). Hungarian Standard Association, Budapest (In Hungarian).
- 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.
- Nambu, K., Yonebayashi, K., 2000. Quantitative relationship between soil properties and adsorption of dissolved organic matter onto volcanic ash and non-volcanic ash soils. *Soil Sci. Plant Nutr.* 46, 559–569.
- Nardi, S., Concheri, G., Pizzeghello, D., Sturaro, A., Rella, R., Parvoli, G., 2000. Soil organic matter mobilization by root exudates. *Chemosphere* 41, 653–658.
- Niemeyer, J., Chen, Y., Bollag, J.M., 1992. Characterization of humic acids, compost, and peat by diffuse reflectance Fourier-transform infrared-spectroscopy. *Soil Sci. Soc. Am. J.* 56, 135–140.
- Oliveira, R.A. de, Tardelli, E.R., Jozala, A.F., Grotto, D., 2019. Evaluation of the 17- $\alpha$ -ethinyl estradiol sorption capacity in soil. *Water, Air and Soil Pollution* 230, 85.
- Petersen, L.W., Moldrup, P., Jacobsen, O.H., Rolston, D.E., 1996. Relations between specific surface area and soil physical and chemical properties. *Soil Sci.* 161, 9–21.
- Ping, L., Lou, Y.M., 2019. Phenanthrene adsorption on soils from the Yangtze River Delta region under different pH and temperature conditions. *Environ. Geochem. Health* 41, 267–274.
- Pignatello, J.J., 1998. Soil organic matter as a nanoporous sorbent of organic pollutants. *Adv. Colloid Interface Sci.* 76–77, 445–467.
- Rabot, E., Wiesmeier, M., Schlüter, S., Vogel, H.J., 2017. Soil structure as an indicator of soil functions: a review. *Geoderma* 314, 122–137.
- Rasmussen, C., Heckman, K., Wieder, W.R., Keiluweit, M., Lawrence, C.R., Berhe, A.A., Blankinship, J.C., Crow, S.E., Druhan, J.L., Hicks Pries, C.E., Marin-Spiotta, E., Plante, A.F., Schädel, C., Schimel, J.P., Sierra, C.A., Thompson, A., Wagai, R., 2018. Beyond clay: towards an improved set of variables for predicting soil organic matter content. *Biogeochemistry* 137, 297–306.
- Ren, X.Y., Zeng, G.M., Tang, L., Wang, J.J., Wan, J., Liu, Y., Yu, J.F., Yi, H., Ye, S.J., Deng, R.C., 2018. Sorption, transport and biodegradation - an insight into bioavailability of persistent organic pollutants in soil. *Sci. Total Environ.* 610–611, 1154–1163.
- Robles, T.A., 2010. Use of Oral Contraceptives. *Decision Making in Medicine*, third ed., pp. 616–619.
- Sahrawat, K.L., 2003. Organic matter accumulation in submerged soils. *Adv. Agron.* 81, 169–201.
- Scherr, F.F., Sarmah, A.K., Di, H.J., Cameron, K.C., 2008. Modelling degradation and

- metabolite formation kinetics of estrone-3-sulfate in agricultural soils. *Environ. Sci. Technol.* 42, 8388–8394.
- 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.
- Shareef, A., Angove, M.J., Wells, J.D., Johnson, B.B., 2006. Sorption of bisphenol A, 17-ethynylestradiol and estrone to mineral surfaces. *J. Colloid Interface Sci.* 297, 62–69.
- Song, X., Wen, Y., Wang, Y., Adeel, M., Yang, Y., 2018. Environmental risk assessment of the emerging EDCs contaminants from rural soil and aqueous sources: analytical and modeling approaches. *Chemosphere* 198, 546–555.
- Sun, K., Jin, J., Gao, B., Zhang, Z., Wang, Z., Pan, Z., Xu, D., Zhao, Y., 2012. Sorption of 17a-ethinyl estradiol, bisphenol A and phenanthrene to different size fractions of soil and sediment. *Chemosphere* 88, 577–583.
- Sun, W., Zhou, K., 2015. Adsorption of three selected endocrine disrupting chemicals by aquatic colloids and sediments in single and binary systems. *J. Soils Sediments* 15, 456–466.
- Swift, R.S., 1996. Organic matter characterization (chap 35). In: Sparks, D.L., et al. (Eds.), *Methods of Soil Analysis. Part 3. Chemical Methods*, vol. 5. Soil Science Society of America Book Series, pp. 1018–1020.
- Tfaily, M.M., Cooper, W.T., Kostka, J.E., Chanton, P.R., Schadt, C.W., Hanson, P.J., Iversen, C.M., Chanton, J.P., 2014. Organic matter transformation in the peat column at Marcell Experimental Forest: humification and vertical stratification. *J. Geophys. Res.: Biogeosciences* 119, 661–675.
- Tivet, F., de Moraes Sa, J.C., Lal, R., Milori, D., Briedis, C., Letourmy, P., Pinheiro, L., Borszowski, P.R., Hatman, D., 2013. Assessing humification and organic C compounds by laser-induced fluorescence and FTIR spectroscopies under conventional and no-till management in Brazilian Oxisols. *Geoderma* 207 – 208, 71–81.
- Tong, X., Li, Y., Zhang, F., Chen, X., Zhao, Y., Hu, B., Zhang, Xuelian, 2019. Adsorption of 17β-estradiol onto humic-mineral complexes and effects of temperature, pH, and bisphenol A on the adsorption process. *Environ. Pollut.* 254 (part A).
- Voglar, D., Lestan, D., 2013. Pilot-scale washing of Pb, Zn and Cd contaminated soil using EDTA and process water recycling. *Chemosphere* 91, 76–82.
- Windsor, F.M., Ormerod, S.J., Tyler, C.R., 2018. Endocrine disruption in aquatic systems: up-scaling research to address ecological consequences. *Biol. Rev.* 93, 626–641.
- Wu, Z., Xu, E., Long, J., Pan, X., Xu, X., Jin, Z., Jiao, A., 2016. Comparison between ATR-IR, Raman, concatenated ATR-IR and Raman spectroscopy for the determination of total antioxidant capacity and total phenolic content of Chinese rice wine. *Food Chem.* 194, 671–679.
- Yamamoto, H., Liljestrand, H.M., Shimizu, Y., Morita, M., 2003. Effects of physical-chemical characteristics of selected endocrine disruptors by dissolved organic matter surrogates. *Environ. Sci. Technol.* 37, 2646–2657.
- Yamamoto, H., Takemoto, K., Tamura, I., Shin-oka, N., Nakano, T., Nishida, M., Honda, Y., Moriguchi, S., Nakamura, Y., 2016. Contribution of inorganic and organic components to sorption of neutral and ionizable pharmaceuticals by sediment/soil. *Environ. Sci. Pollut. Control Ser.* 25, 1–12.
- Yu, Z., Xiao, B., Huang, W., Peng, P., 2004. Sorption of steroid estrogens to soils and sediments. *Environ. Toxicol. Chem.* 23, 531–539.
- Zhang, Z.H., Feng, Y.J., Liu, Y., Sun, Q.F., Sun, P., Gao, N.Q., 2010. Kinetic degradation model and estrogenicity changes of EE2 (17a-ethinylestradiol) in aqueous solution by UV and UV/H<sub>2</sub>O<sub>2</sub> technology. *J. Hazard Mater.* 181, 1127–1133.