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An analysis of the dissipation of pharmaceuticals under thirteen different soil conditions

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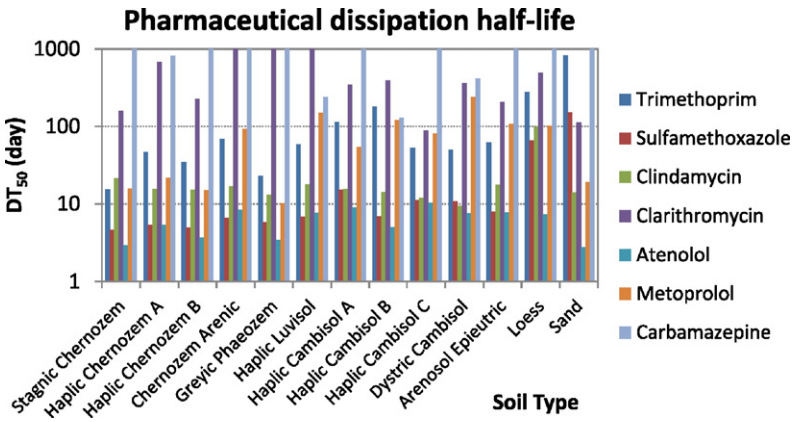
HIGHLIGHTS

- Dissipations of 7 pharmaceuticals in 13 soils were evaluated.
- Half-lives were related to soil properties and their sorption affinity to soils.
- Increasing compound sorption affinity mostly did not decrease its dissipation.
- Compound persistence was mostly dependent on soil type conditions.
- Faster dissipation was observed in soils of high quality.

GRAPHICAL ABSTRACT



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ABSTRACT

The presence of human and veterinary pharmaceuticals in the environment is recognized as a potential threat. Pharmaceuticals have the potential to contaminate soils and consequently surface and groundwater. Knowledge of contaminant behavior (e.g., sorption onto soil particles and degradation) is essential when assessing contaminant migration in the soil and groundwater environment. We evaluated the dissipation half-lives of 7 pharmaceuticals in 13 soils. The data were evaluated relative to the soil properties and the Freundlich sorption coefficients reported in our previous study. Of the tested pharmaceuticals, carbamazepine had the greatest persistence (which was mostly stable), followed by clarithromycin, trimethoprim, metoprolol, clindamycin, sulfamethoxazole and atenolol. Pharmaceutical persistence in soils was mostly dependent on the soil-type conditions. In general, lower average dissipation half-lives and variability (i.e., trimethoprim, sulfamethoxazole, clindamycin, metoprolol and atenolol) were found in soils of better quality (well-developed structure, high nutrition content etc.), and thus, probably better microbial conditions (i.e., Chernozems), than in lower quality soil (Cambisols). The impact of the compound sorption affinity onto soil particles on their dissipation rate was mostly negligible. Although there was a positive correlation between compound dissipation half-life and Freundlich sorption coefficient for clindamycin ($R = 0.604$, $p < 0.05$) and sulfamethoxazole ($R = 0.822$, $p < 0.01$), the half-life of sulfamethoxazole also decreased under better soil-type conditions. Based on the calculated dissipation

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and sorption data, carbamazepine would be expected to have the greatest potential to migrate in the soil water environment, followed by sulfamethoxazole, trimethoprim and metoprolol. The transport of clindamycin, clarithromycin and atenolol through the vadose zone seems less probable.

1. Introduction

Several studies (e.g., Thiele-Bruhn, 2003) have documented how soil can be contaminated by human or veterinary pharmaceuticals. Some pharmaceutical ingredients may be retained in soils, while others can be transported to the surface and groundwater through surface runoff and infiltration. The mobility of contaminants in soils is dependent on many soil and pharmaceutical properties (e.g., sorption affinity onto soil particles and degradation rate). Although the sorption affinity of pharmaceuticals to soil has been extensively reviewed (Tolls, 2001), and the frequency of new experimental studies has recently increased (e.g., Kodešová et al., 2015; Schaffer and Licha, 2015), there have been considerably fewer studies addressing pharmaceutical dissipation in soils.

Pharmaceutical occurrence in the environment depends on their consumption by the population or by animals. The pharmaceuticals that most frequently occur in waste-water (Fedorova et al., 2014a; Golovko et al., 2014a; Golovko et al., 2014b) and surface-water (Fedorova et al., 2014b) in the Czech Republic are trimethoprim, sulfamethoxazole, clindamycin, clarithromycin, atenolol, metoprolol and carbamazepine. The sorption of these pharmaceuticals with respect to different soil properties were studied by Kodešová et al. (2015). In two cases (trimethoprim and carbamazepine), sorption affinity was mainly controlled by organic carbon content, as is often expected for organic compounds. In four cases, sorption depended on two factors: soil pH (affecting compound ionization) and the number of available sites for sorption. Depending on the pKa of the pharmaceutical, either hydrolytic acidity (sulfamethoxazole partially existed in the anionic form) or basic cation saturation (clindamycin, metoprolol, and atenolol molecules mostly existed in cationic form), controlled sorption of pharmaceuticals on selected soils. In the case of clarithromycin (which also mostly existed in cationic form), sorption complex saturation played the most important role. However, the sorption behavior to soils of these mostly ionizable compounds is a complex set of relatively poorly understood processes, and this may strongly hinder clear interpretation of potentially related processes such as dissipation. Carbamazepine is one of the most frequently detected compounds in surface and subsurface waters worldwide, followed by sulfamethoxazole and trimethoprim (e.g., Fram and Belitz, 2011; Godfrey et al., 2007; Huntscha et al., 2012; Li, 2014; Loos et al., 2010; Radovic et al., 2015). This is likely why dissipation of these compounds in soils has been previously evaluated in several studies: carbamazepine (Al-Rajab et al., 2015; Dalkmann et al., 2014; Grossberger et al., 2014; Li et al., 2013; Monteiro and Boxall, 2009; Walters et al., 2010; Yu et al., 2013); sulfamethoxazole (Lin and Gan, 2011; Liu et al., 2010; Srinivasan and Sarmah, 2014; Srinivasan et al., 2014; Wu et al., 2012); and trimethoprim (Lin and Gan, 2011; Liu et al., 2010; Wu et al., 2012). These studies focused on the impacts of different amendments (e.g., biosolids), soil sterilization and incubation conditions (e.g., aerobic vs. anaerobic, and different temperatures) on a particular compound degradation. Compound dissipation mostly due to biodegradation was documented by comparing degradation rates for sterile and nonsterile soils (Li et al., 2013; Lin and Gan, 2011; Liu et al., 2010; Srinivasan and Sarmah, 2014; Wu et al., 2012; Yu et al., 2013) and by decreased dissipation rates for samples incubated under lower temperature (25 and 7.5 C) (Srinivasan and Sarmah, 2014). Dissipation of antibiotic was also controlled by the initial compound concentration in soils. Increasing concentration resulted in decreasing degradation rates (e.g. Srinivasan and Sarmah, 2014) due to inhibition of degrading microorganisms (Caracciolo et al., 2015). However, each study evaluated the pharmaceutical's dissipation half-lives in only a few soils and could not relate compound dissipation half-lives to soil properties or compound sorption in soils. In addition, abundance and activity of microorganisms is controlled by an overall soil quality (i.e. comprehensive influence of basic physical and chemical properties, soil structure, soil-water content and aeration, nutrition content, climate, etc.), which is associated with soil type (FAO, 2006), and land use (e.g. Hao et al., 2014). Thus, dissipation of pharmaceuticals with respect to soil type should be studied as well. To our knowledge, no study has yet reported on the soil dissipation half-lives of clindamycin, clarithromycin, atenolol and metoprolol. Although their occurrence in the water environment has been documented (e.g., de Jongh et al., 2012), only the dissipation of clindamycin in biosolids (Chenxi et al., 2008; Wu et al., 2009), clarithromycin in biosolids (Chenxi et al., 2008) and metoprolol and atenolol in sludge (Ribeiro et al., 2013) have been evaluated.

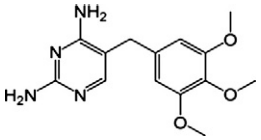
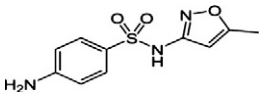
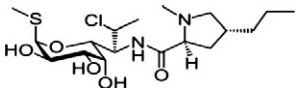
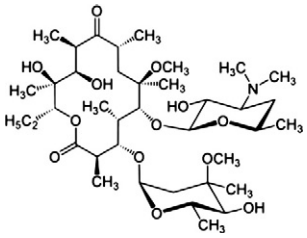
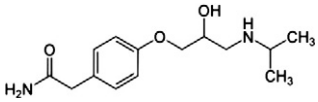
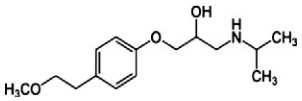
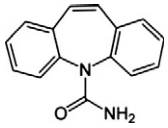
Therefore, the goals of this study were: (1) to evaluate the degradation rates of the same pharmaceuticals in the same soils as addressed by Kodešová et al. (2015); (2) to assess the impact of soil properties and compound sorption affinity to soils on the evaluated pharmaceutical dissipation half-lives; and (3) to estimate the potential for these compounds to migrate into the subsurface water environment and to result in groundwater contamination.

2. Material and methods

2.1. Pharmaceutical and soil properties

This study was performed using the same pharmaceuticals (Table 1) and soils (Table 2) that Kodešová et al. (2015) used to evaluate the impact of soil properties on the different sorption affinities onto soil particles. These soil types had also been previously used by Kodešová et al. (2011) to evaluate the properties that affect the sorption of selected pesticides. Soils were selected to cover a large variety of soil properties that may influence pharmaceutical behavior in soils. The main criteria were: arable soils, variable soil substrate and soil texture, and large range of soil organic carbon contents, active and exchangeable pH, cation exchange capacity, base cation saturation etc. These properties influences compound sorption (Kodešová et al., 2015) and microbial abundance and activity (Jhonson, 2009), which affects organic compound dissipation. Soils represent mayor soil types that occur in the Czech Republic and surrounding countries. Selection was done based on the soil map of the Czech Republic (Kozák et al., 2010), available soil database PUGIS (Kozák et al., 1996) and information about microbial parameters of main taxonomical unites of the Czech Republic (Růžek et al., 2004, 2006, 2009). Samples were taken from the surface (0–25 cm) and subsurface horizons (from depths of 60–80 and 50–70 cm in the cases of loess and sand, respectively). Soils were air-dried, ground and sifted through a 2-mm sieve. The basic chemical and physical properties were obtained using standard laboratory procedures under a constant laboratory temperature of 20 °C, including: soil pH (pH H_2O, pH KCl) (ISO, 10390:1994), organic carbon content (Cox) (Skjemstad and Baldock, 2008), CaCO3 content (Loeppert and Suarez, 1996), cation exchange capacity (CEC) (Bower and Hatcher, 1966), hydrolytic acidity (i.e. sum of H+ cations) (HA) (Klute, 1996), exchangeable acidity (EA) (Hendershot et al., 1993), base cation saturation (BCS, the difference between CEC and HA), sorption complex saturation (SCS, the percentage of BCS in CEC), particle density (ϱ_s) (Flint and Flint, 2002) and particle size distribution (fractions of clay, silt, and sand) (Gee and Or,

Table 1
Selected pharmaceuticals (Kodešová et al., 2015).

Pharmaceutical	CAS number	Use	pKa	log Kow	Mw (g mol^{-1})	Molecular structure
Trimethoprim	738-70-5	Antibiotic (also veterinary use)	7.12[a]	0.91[a]	290.32	
Sulfamethoxazole	723-46-6	Antibiotic (also veterinary use)	$pKa_1 = 1.7$[b] $pKa_2 = 5.6$[b]	0.89[c]	253.28	
Clindamycin	21462-39-5	Antibiotic	7.72[d]	2.16[e]	461.44	
Clarithromycin	81103-11-9	Antibiotic	8.99[e]	3.16[e]	747.95	
Atenolol	29122-68-7	Beta blockers (hypertension)	9.6[f]	0.16[e]	266.34	
Metoprolol	56392-17-7	Beta blockers (hypertension)	9.63[g]	1.95[g]	267.36	
Carbamazepine	298-46-4	Anticonvulsant (brain disorders), analgesic	$pKa_1 = 1.0$[h] $pKa_2 = 13.9$[h]	2.25[i]	236.27	

[a] Thiele-Bruhn et al. (2004).
[b] Lucida et al. (2000).
[c] Figueroa-Diva et al. (2010).
[d] Osol et al. (1975).
[e] http://www.drugbank.ca.
[f] O'Neil (2001).
[g] Caron et al. (1999).
[h] Oppel et al. (2005).
[i] Jones et al. (2002).

2002). The relationships between these soil properties (i.e., the Pearson product moment correlation coefficient and the statistical significance of the estimated correlations assessed as p-value) were documented in Kodešová et al. (2015).

2.2. Pharmaceutical degradation

Pharmaceutical degradation was studied using a standard method (OECD, 2002). Each soil was mixed with solutes of different pharmaceuticals. Fifty grams of air-dried soil was always placed into an incubation polyethylene bottle and solution of one pharmaceutical was added. Time needed for soil sample preparation (i.e. drying, grinding, sieving and dosing) did not exceed 5 days. The amount of added solution and its concentration were calculated individually for each soil to achieve the field water holding capacity (soil water content for pressure head between 100 to 300 cm) and a similar compound load per dry soil unit (2 μg g^{-1}). This compound load was in the same range as has been previously used in similar studies, e.g. 0.05–5.0 μg g^{-1} (Grossberger et al., 2014), 1.0 μg g^{-1} (Monteiro and Boxall, 2009), and 0.5–5.0 μg g^{-1} (Srinivasan and Sarmah, 2014). The following volumes and concentrations were assumed: 12.5 cm^3 and 8.0 μg cm^{-3} (Stagnic Chernozem Siltic, Haplic Chernozem A and B, Haplic Luvisol and Greyic Phaeozem), 12 cm^3 and 8.3 μg cm^{-3} (all Cambisols), 8 cm^3 and 12.5 μg cm^{-3} (Chernozem Arenic), 7 cm^3 and 14.3 μg cm^{-3} (Arenosol Epieutric) and 6 cm^3 and 16.7 μg cm^{-3} (sand). Solutions were applied using VITLAB® Genius (5–50 ml) (VITLAB GmbH). Masses of empty bottles, bottles with soil, and of bottles with soil and solution were recorded to calculate precise solute volume in each bottle. Actual concentrations of studied compounds in solutes were measured using liquid chromatography–tandem mass spectrometry (LC–MS/MS), which is described below. Each bottle was shaken 30 s to achieve uniform distribution of compound in a soil sample. Sixteen bottles were prepared for each soil and pharmaceutical. The soils were incubated in the dark at a constant temperature of 20 °C. The soil moisture content was regularly

controlled at 2- week intervals by weighing of the incubation bottles and water losses were compensated by adding water. Two bottles for each pharmaceutical, soil and at predefined times (1, 2, 5, 12, 23, 40 and 61 days after the pharmaceutical application) were removed from the incubator and stored in a freezer at −20 °C until compound extraction, which was carried out immediately after finishing degradation experiment. Two bottles for each soil with each pharmaceutical were also put in the freezer immediately after applying compound solutes (i.e. time = 0 days). It should be noted that no/negligible sorption of compounds on the inner surface of polyethylene bottles was tested for wide range of compounds including tested ones in waste water (Fedorova et al., 2014a). In addition, the following compound extraction from the soils was carried out directly in these bottles.

2.3. Chemicals

Methanol and acetonitrile (LiChrosolv® Hypergrade) were purchased from Merck (Darmstadt, Germany). Formic acid (LC/MS grade) to acidify the mobile phases was purchased from Labicom (Olomouc, Czech Republic). Ultra-pure water was produced using an Aqua-MAX-Ultra System (Younglin, Kyounggi-do, Korea). All compounds used were of high purity (>98%).

Atenolol, carbamazepine, clarithromycin, clindamycin, metoprolol, sulfamethazine, tramadol and trimethoprim were purchased from Sigma-Aldrich (St. Louis, MO, USA) and sulfamethoxazole was purchased from Riedel-de Haen (Germany). All studied pharmaceuticals were analytical grade or purity higher than 98%.

For internal standards (ISs), carbamazepine (D_{10}) was acquired from CDN Isotopes (Pointe-Claire, Quebec, Canada), atenolol (D_6) and metoprolol hydrochloride (D_7) were purchased from Alsachim (Strasbourg, France), and clarithromycin-N-methyl (D_3) and clindamycin-D3 hydrochloride were acquired from TRC (North York, Canada). Tramadol was used as an internal standard for trimethoprim and sulfamethazine as an internal standard for sulfamethoxazole.

A stock solution of each pharmaceutical was prepared in methanol at a concentration of 1 mg ml^{-1}. A spiking mixture of each was prepared by diluting the stock solution with methanol to a final concentration of 100 μg ml^{-1} for internal standards and 10 μg ml^{-1} for native standards. All stock and spiking solutions were stored at −20 °C.

2.4. Sample preparation and extraction procedure

Extractions in an ultrasonic bath with three solvent mixtures were tested prior to our study to achieve the best extraction efficiency (Golovko et al., 2015 submitted): 1) extraction with acetonitrile and water (1:1 [v/v] with 0.1% formic acid) followed by extraction with acetonitrile, 2-propanol and water (3:3:4 [v/v/v] with 0.1% formic acid); 2) extraction with acetonitrile and water (1:1 [v/v] with 0.1% formic acid); 3) extraction with methanol and water (2:1 [v/v] with 0.1% formic acid). We followed the philosophy “extract and inject”, where no additional steps as evaporation, clean-up etc., which could affect recovery of target analytes and their metabolites, were needed. Possible matrix effects were maintained as it is described below i.e. using matrix matching standards for each soil type studied.

Extraction with acetonitrile and water (1:1 [v/v] with 0.1% formic acid) was found to be among the most effective approaches for 24 of 91 tested pharmaceuticals (including the 7 selected in this study), with the recovery of the 7 compounds studied here varying between 61 and 130% (tested for all soils types). The recovery ratio achieved by extraction with acetonitrile and water (1:1 [v/v] with 0.1% formic acid) for each target pharmaceutical was evaluated at a fortification level of 1 μg g^{-1} and was expressed as the ratio between the determined concentration and the nominal concentration. Recoveries and recovery uncertainties are shown in supplementary materials, Table SI 2.

In our study, the pharmaceutical extraction method from the incubated soils was separated into three steps because of the large sample volumes (approximately 50 g of soil into a 100 ml high density polyethylene bottle). We analyzed the whole content of each incubation bottle to reduce the number of sample handling steps (freeze drying and preparation of representative subsamples was skipped). Sixty, 35 and 30 ml of solvent (i.e., 1:1 acetonitrile and water [v/v] with 0.1% formic acid) were subsequently added to the soil and samples were sonicated (DT 255, Bandelin electronic, Sonorex digitec, Berlin, Germany) for 15 min in each step. After soil particle sedimentation, three supernatants from each bottle were mixed and 10 ml aliquots were filtered through a syringe filter (0.45 μm, regenerated cellulose, Labicom, Olomouc, Czech Republic) into 10 ml vials. The possible impact (due to compound sorption) of the syringe filter material on the measured pharmaceuticals' concentrations was tested previously (Lindberg et al., 2014). No noticeable effect on the recovery of the studied compounds was found.

Table 2
Selected soils and their properties: pH_{H2O}, pH_{KCl}, organic carbon content, Cox, $CaCO_3$ content, cation exchange capacity, CEC, soil hydrolytic acidity, HA, exchangeable acidity, EA, basic cation saturation, BCS, sorption complex saturation, SCS, particle density ρ_s, clay, silt and sand contents (%) (Kodešová et al., 2015).

Soil	Location	Parent material	pH_{H2O}	pH_{KCl}	Cox (%)	$CaCO_3$ (%)	CEC ($mmol^+$ kg^{-1})	HA ($mmol^+$ kg^{-1})	EA ($mmol^+$ kg^{-1})	BCS ($mmol^+$ kg^{-1})	SCS (%)	ρ_s (g cm^{-3})	Clay (%)	Silt (%)	Sand (%)
Stagnic Chernozem Siltic	Milčice	Marlite	8.07	7.40	3.2	20.87	315	3.6	0.95	311	98.9	2.50	14.7	45.78	39.53
Haplic Chernozem A	Ivanovice na Hané	Loess	7.17	6.30	1.67	0.07	218	10.8	0.85	207	95.0	2.53	5.91	76.37	17.72
Haplic Chernozem B	Praha Suchdol	Loess	7.97	7.19	2.09	4.07	261	4.5	1.13	256	98.3	2.51	6.52	64.16	29.32
Chernozem Arenic	Velké Chvalovice	Gravely sand	6.36	5.90	1.16	0	92.6	14.4	1.13	78.2	84.4	2.61	2.14	21.40	76.46
Greyic Phaeozem	Čáslav	Loess	7.73	5.31	1.24	0.55	166	4.5	0.76	162	97.3	2.54	4.41	75.39	20.20
Haplic Luvisol	Hněvčeves	Loess	6.51	5.66	0.88	0.18	155	15.3	0.95	140	90.1	2.59	7.41	76.74	15.85
Haplic Cambisol A	Humpolec	Paragneiss	5.37	4.41	1.64	0.18	175	50.5	2.65	125	71.2	2.59	7.46	48.33	44.21
Haplic Cambisol B	Předbořice	Syenite	5.99	5.10	2.58	0.05	155	42.4	1.32	113	72.7	2.55	2.07	43.94	53.99
Haplic Cambisol C	Jince	Quartzite	5.30	4.39	1.71	0.67	132	50.5	2.84	81.6	61.8	2.59	5.27	56.51	38.22
Dystric Cambisol	Vysoké nad Jizerou	Orthogneiss	5.59	4.67	2.17	0.10	186	54.1	1.89	132	70.9	2.59	2.56	58.35	39.09
Arenosol Epieutric	Semice	Sand	5.38	4.30	0.69	0.05	45.5	19.8	2.27	25.7	56.5	2.63	5.02	10.98	84.00
Loess	Praha Suchdol		8.44	7.41	0.36	20.5	191	3.6	1.13	188	98.1	2.60	7.51	48.31	44.18
Sand	Písková lhota		8.71	8.46	0.08	0.1	10.6	0.45	0.66	10.2	95.8	2.65	0	0	100
Average value			6.81	5.88	1.49	3.64	161.8	21.1	1.42	141	83.9	2.58	5.45	48.17	46.36
Standard deviation			1.25	1.37	0.88	7.64	81.7	20.5	0.73	85.8	15.3	0.05	3.65	24.53	26.07
Range			3.41	4.16	3.12	20.9	304	53.6	2.18	301	42.4	0.15	14.7	76.74	84.15

The soil extracts were stored in a freezer at −20 °C until analyzed (max. 24 h). The ISs (at a level corresponding to 100 ng g^{-1}) were added to all soil extracts before analysis.

2.5. LC–MS/MS analysis

A detailed description of MS/MS transitions and LC-MS/MS configuration and set up has been provided elsewhere (Grabic et al., 2012; Khan et al., 2012). A triple stage quadrupole MS/MS TSQ Quantum Ultra mass spectrometer (Thermo Fisher Scientific, San Jose, CA, USA) coupled with an Accela 1250 LC pump (Thermo Fisher Scientific) and an HTS XT-CTC autosampler (CTC Analytics AG, Zwingen, Switzerland) was used for analysis. A Cogent Bidentate C18 column (50 mm × 2.1 mm i.d., 4 μm particle size, from MicroSolv Technology Corporation Eatontown, NJ, USA) was used for chromatographic separation of target compounds. The method parameters are reported in Supplementary Table S3. The IS method was used for quantification of analytes. A five-point calibration curve was prepared by spiking process blanks (solvent extracts of clean bottles) with target compounds at concentrations between 1 ng ml^{-1} and 1000 ng ml^{-1} and ISs at 100 ng ml^{-1}. The matrix effect was assessed for each compound/soil combination. Corrections for ion suppression or enhancement were accomplished using matrix-matched standards, which were prepared as the highest point of calibration curve in blank (non-fortified) soils extracts.

2.6. Pharmaceutical recovery, dissipation rate and half-life

Concentrations of the initially applied solutes and soil extracts, their volumes and soil mass were used to evaluate the amounts of compound initially applied and remaining in the soil (μg) per dry soil unit (g), i.e., compound concentrations in soils (c). First, initial concentrations (c_0) calculated using the applied solute concentration and the soil extract concentrations on the day of solute application (i.e. immediately after the solute application) were used to calculate each compound recovery in each soil (Table 3).

Second, concentrations c_0 and ct (concentration in time) calculated using the soil extract concentrations were used to assess compound dissipation rates. The single first-order kinetic model is a model that has been the most frequently used to express compound dissipation over time. However, there are other models, which can be applied. For instance, Srinivasan et al. (2014) tested also three biphasic kinetic models (bi-exponential decay, first-order double exponential decay, and first-order two-compartment). They showed that dissipation of sulfamethoxazole was successfully predicted by all four models (i.e. single and 3 biphasic). However, the nonlinear biphasic model improved the goodness-of-fit parameters for all datasets. The first-order kinetic expression was used in our study:

$$\frac{c_t}{c_0} = e^{-k_R t} \tag{1}$$

where c_0 (μg g^{-1}) is the initial concentration, c_t (μg g^{-1}) is the concentration in time, t (day) is time, and k_R (day^{-1}) is the first-order rate constant. Compound dissipation half-life DT_{50} (day) was then calculated as follows:

$$DT_{50} = \frac{\ln 2}{k_R} \tag{2}$$

Results are presented in Table 4.

2.7. Sorption isotherms of selected pharmaceuticals

The sorption isotherms evaluated in our previous study (Kodešová et al., 2015 were expressed using the Freundlich equation:

$$s = K_F c^{1/n}, \tag{3}$$

where K_F ($cm^{3/n}$ $\mu g^{1-1/n}$ g^{-1}) and n are empirical coefficients (Table 5). The coefficient n was fixed at the average n coefficient (n_{avg}), which was evaluated for each pharmaceutical, The Freundlich sorption coefficients was calculated for each pharmaceutical and soil.

2.8. Statistical analysis

Simple correlations between measured physical and chemical soil properties, pharmaceutical recoveries (Table 6) and dissipation half-lives (Table 7) were assessed using the Pearson product moment correlation coefficient and p-value, which tests the statistical significance of the estimated correlations. In addition, the Freundlich sorption coefficients, K_F (for a constant average n value), which were evaluated by Kodešová et al. (2015), were also included to relate K_F value and recovery or dissipation half-life for each pharmaceutical agent (Table 8). Analyses were carried out for all soils and then only for topsoils (i.e., excluding two substrates, in which different microbial conditions in comparison to those in topsoils is anticipated). Next, we attempted to evaluate the general behavior of in soil degradable compounds with respect to soil types. For this purpose, basic statistical (i.e., average dissipation half-lives in each topsoil and standard deviations) and principal components analyses were used. Analyses were performed using STAGRAPHICS Centurion XV Version 15.2.06.

Table 3
The recovery (%) of the studied pharmaceuticals in soils.

Soil	Trimethoprim	Sulfamethoxazole	Clindamycin	Clarithromycin	Atenolol	Metoprolol	Carbamazepine
Stagnic Chernozem Siltic	76.2	57.3	87.6	68.6	70.8	58.4	70.2
Haplic Chernozem A	65.8	79.8	59.6	86.0	75.9	99.9	123
Haplic Chernozem B	62.6	85.0	82.4	87.8	77.6	79.2	130
Chernozem Arenic	50.3	112	122	80.9	88.5	90.4	117
Greyic Phaeozem	99.0	111	75.1	93.5	122	92.7	70.7
Haplic Luvisol	60.3	87.0	83.8	68.4	95.3	92.7	102
Haplic Cambisol A	43.3	95.9	87.1	94.0	83.6	93.6	131
Haplic Cambisol B	71.2	120	113	98.0	113	104	123
Haplic Cambisol C	64.8	85.3	63.0	108	85.2	90.9	122
Dystric Cambisol	97.0	71.5	114	93.9	107	106	115
Arenosol Epieutric	84.3	103	119	128	87.3	95.5	124
Loess	39.6	132	73.1	89.6	82.8	73.4	93.0
Sand	50.8	125	109	129	94.8	68.2	74.1
Average value	66.5	97.2	91.5	94.2	91.1	88.0	107
Standard deviation	18.8	22.2	21.6	18.5	15.2	14.2	22.9
Range	59.4	75.1	59.0	60.1	51.6	47.5	60.7

Table 4
The first-order rate constants, k_R (day^{-1}), coefficients of determination, R^2, and dissipation half-lives, DT_{50} (day), of the studied pharmaceuticals in soils.

Soil	Trimethoprim			Sulfamethoxazole			Clindamycin			Clarithromycin			Atenolol			Metoprolol			Carbamazepine		
	k_R	R^2	DT_{50}	k_R	R^2	DT_{50}	k_R	R^2	DT_{50}	k_R	R^2	DT_{50}	k_R	R^2	DT_{50}	k_R	R^2	DT_{50}	k_R	R^2	DT_{50}
Stagnic Chernozem Siltic	0.0452	0.975	15.3	0.1489	0.895	4.66	0.0325	0.987	21.3	0.0044	0.410	159	0.2340	0.962	2.96	0.0441	0.988	15.7			>1000
Haplic Chernozem A	0.0149	0.689	46.5	0.1280	0.966	5.42	0.0447	0.922	15.5	0.0010	0.346	676	0.1280	0.995	5.41	0.0318	0.968	21.8	0.0008	0.498	819.68
Haplic Chernozem B	0.0202	0.981	34.3	0.1388	0.971	4.99	0.0455	0.981	15.3	0.0031	0.858	227	0.1851	0.979	3.74	0.0461	0.674	15.0			>1000
Chernozem Arenic	0.0101	0.920	68.7	0.1059	0.898	6.54	0.0408	0.977	17.0			>1000	0.0825	0.996	8.40	0.0074	0.964	93.2			>1000
Greyic Phaeozem	0.0303	0.962	22.9	0.1188	0.995	5.83	0.0535	0.976	12.9			>1000	0.1997	0.966	3.47	0.0678	0.992	10.2			>1000
Haplic Luvisol	0.0117	0.866	59.1	0.1010	0.980	6.86	0.0387	0.967	17.9			>1000	0.0914	0.997	7.59	0.0046	0.765	151.2	0.0029	0.812	239
Haplic Cambisol A	0.0061	0.890	114.5	0.0459	0.925	15.1	0.0447	0.989	15.5	0.0020	0.512	345	0.0775	0.988	8.95	0.0127	0.942	54.7			>1000
Haplic Cambisol B	0.0039	0.497	178.6	0.1001	0.936	6.92	0.0493	0.992	14.1	0.0018	0.639	392	0.1366	0.999	5.07	0.0058	0.792	120	0.0053	0.812	129
Haplic Cambisol C	0.0130	0.891	53.2	0.0617	0.966	11.2	0.0579	0.995	12.0	0.0078	0.654	88.9	0.0674	0.989	10.3	0.0086	0.927	80.7			>1000
Dystric Cambisol	0.0138	0.547	50.1	0.0646	0.962	10.7	0.0744	0.961	9.32	0.0019	0.544	360	0.0922	0.998	7.51	0.0029	0.637	239	0.0017	0.429	418
Arenosol Epieutric	0.0111	0.899	62.7	0.0869	0.970	7.98	0.0391	0.914	17.7	0.0034	0.349	206	0.0894	0.993	7.75	0.0064	0.960	109			>1000
Loess	0.0025	0.361	278	0.0106	0.933	65.6	0.0070	0.796	99.1	0.0014	0.239	497	0.0942	0.991	7.36	0.0068	0.589	102			>1000
Sand	0.0008	0.335	821	0.0045	0.946	152	0.0236	0.905	29.4	0.0062	0.351	112	0.2469	0.923	2.81	0.0364	0.949	19.0			>1000
All soils																					
Average DT_{50} value			139			23.4			22.8			–			6.26			79.4			–
Standard deviation			218			42.0			23.4			–			2.48			67.4			–
Range			806			148			89.8			–			7.48			229			–
Topsoils																					
Average DT_{50} value			64.2			7.84			15.3			–			6.47			82.8			–
Standard deviation			46.1			3.23			3.25			–			2.45			70.9			–
Range			163.3			10.5			12.0			–			7.33			229			–

Average DT_{50} value, standard deviation and range could not be calculated for clarithromycin and carbamazepine due their persistency in some soils.

3. Results and discussion

3.1. Pharmaceutical recovery

The recoveries of each compound from the tested soils were variable (Table 3). These differences could be attributed to the soil properties variability and measurement uncertainty (e.g., the recoveries of a few analytes were over 100% etc.) (e.g., Wu et al., 2010). Table 3 shows that the extraction efficiencies for sulfamethoxazole, clindamycin, clarithromycin, atenolol, metoprolol and carbamazepine were mostly reasonable. In several cases, compound recovery (Table 6) was negatively related to soil pH (clarithromycin, metoprolol and carbamazepine), Cox (sulfamethoxazole), $CaCO_3$ (sulfamethoxazole and metoprolol), CEC (sulfamethoxazole, clarithromycin and metoprolol), BCS (sulfamethoxazole, clarithromycin and metoprolol), SCS (clarithromycin, metoprolol and carbamazepine), clay (sulfamethoxazole, clarithromycin, atenolol and metoprolol) and silt (clindamycin and clarithromycin) contents and positively related to HA (metoprolol and carbamazepine), EA (clarithromycin and carbamazepine), particle density (clarithromycin) and sand content (clindamycin and clarithromycin). In some cases, the negative correlation coefficients obtained between recoveries and Cox, CEC, BCS or clay content could be explained by the positive correlations of these properties and the Freundlich sorption coefficients, indicating increasing sorption affinity onto soil particles (Kodešová et al., 2015). Despite that, no relationship (except for clarithromycin) was found between the compounds' recoveries and the Freundlich sorption coefficients (Table 8). The negative impact of increasing clarithromycin sorption on its recovery was observed ($R = -0.716$, $p < 0.05$), as could be anticipated for a strongly sorbed organic compound. However, clarithromycin recovery was in acceptable range (68–128%).

Trimethoprim extraction from some soil samples appeared to be less successful. Low recovery (39.6 to 50.8%) was observed in four soils in our experiment. There is no evident grouping factor for these soils. They belong to different types and they show different dissipation half-lives for trimethoprim. However, trimethoprim recovery for the same 13 individual soils obtained from the freeze-dried samples ranged between 89 and 130% (Supplementary Table S3, Appendix A) at a fortification level of 1 μg g^{-1}. Generally, a lower trimethoprim recovery in comparison to other pharmaceuticals has been documented, for example by Mohring et al. (2009). In their study, trimethoprim recovery ($26.6 \pm 4.4\%$) from fermentation substrate was considerably lower than for other compounds (including sulfamethoxazole) (68.4 to 95.0%). Fast biological degradation could be excluded as trimethoprim dissipation half-lives were found between 69 and 821 days in corresponding soils. Abiotic processes were shown to be minor contributing factors compared to biodegradation for trimethoprim dissipation in soils (Liu et al., 2010). Based on above mentioned facts, we assumed that low trimethoprim recovery could be more associated with some unexpected specific interference with analyte or internal standard, which was not corrected with matrix matching standard.

3.2. Dissipation of pharmaceuticals in soils and their potential to contaminate the subsurface water environment

For each tested pharmaceutical agent, the determined first-order rate constants, coefficients of determination (R^2) and dissipation half-lives are shown in Table 4. The R^2 values indicate that the first-order kinetic expressions are mostly well-fitted to the experimental data, with the exceptions of trimethoprim dissipation in two soil substrates, and clarithromycin and carbamazepine dissipation in almost all soils. The lower R^2 values in these cases are likely due to very low, or no compound dissipation, as well as the relatively larger variability in the evaluated concentrations of remaining compound in the soil and over time.

Trimethoprim dissipation half-lives in topsoils varied between 15.3 and 179 days and noticeably differed from dissipation half-lives in

Table 5
The Freundlich adsorption coefficients, K_F, ($cm^3\ \mu g^{1-1/n}\ g^{-1}$) for the studied pharmaceuticals and soils for average n values, n_{avg} (Kodešová et al., 2015).

Average n value for pharmaceutical/soil type	The Freundlich adsorption coefficients, K_F, for different soils and pharmaceuticals						
	Trimethoprim	Sulfamethoxazole	Clindamycin	Clarithromycin	Atenolol	Metoprolol	Carbamazepine
n_{avg} (−)	1.27	1.65	1.13	1.00	1.17	1.20	1.13
Stagnic Chernozem Siltic	33.24	0.64	10.71	601.6	13.9	17.22	4.66
Haplic Chernozem A	40.39	1.19	8.28	490.1	10.11	15.32	3.84
Haplic Chernozem B	28.97	0.88	6.73	286.0	16.24	11.57	3.86
Chernozem Arenic	24.33	1.41	4.13	159.9	3.22	6.90	2.13
Greyic Phaeozem	21.13	0.57	6.32	290.8	7.54	10.13	2.23
Haplic Luvisol	28.39	1.28	7.48	436.3	6.04	10.07	1.88
Haplic Cambisol A	23.15	4.01	3.16	54.4	5.36	8.26	2.97
Haplic Cambisol B	23.00	3.43	2.84	36.6	4.69	6.59	3.83
Haplic Cambisol C	20.77	4.84	2.57	36.1	4.28	7.36	3.28
Dystric Cambisol	27.74	4.73	2.93	64.1	5.95	8.77	4.31
Arenosol Epieutric	12.21	1.39	2.89	82.8	2.11	4.00	0.71
Loess	6.95	0.07	8.86	930.4	5.08	8.80	0.67
Sand	1.67	0.11	3.67	203.4	0.67	1.71	0.26
Average K_F value	22.46	1.89	5.43	282.5	6.55	8.98	2.66
Standard deviation	10.52	1.73	2.77	269.3	4.47	4.17	1.47
Range	38.72	4.77	8.14	894.2	15.57	15.51	4.4

substrates (278 and 821 days) (Table 4). Larger trimethoprim dissipation half-lives were found in two Cambisols and in soils on sand and gravely-sand substrates. Trimethoprim dissipation half-lives positively correlated with pH_{KCl} and negatively correlated with Cox, when evaluated for all soils (Table 7). No similar relationships were found when evaluated for topsoils only. The negative relationship between trimethoprim dissipation half-lives and Cox is likely a result of the negative correlation between Cox and K_F (Kodešová et al., 2015), as well as the negative correlation between dissipation half-lives and K_F values (Table 8, Fig. 1a). This is in contrast to the usually anticipated organic compound behavior (i.e., greater persistence of greatly sorbed compounds). The most probable explanation is that better conditions for compound degradation were present in Chernozems, in which favorable microbial conditions would be anticipated due to better soil quality, i.e. well-developed soil structure and associated soil chemical and physical condition (Altermann et al., 2005; FAO, 2006; Hao et al., 2014; Jhonson, 2009; Růžek et al., 2004, 2006, 2009), than in other soils (particularly in two Cambisols and sand substrate) (FAO, 2006; Hao et al., 2014; Růžek et al., 2004, 2006, 2009). In general, soils of higher base cation saturation (Table 2) (which indicates amount of positively-charged nutrient ions) provide better condition for bacteria. Acidity (i.e. low pH, Table 2) reduces bacterial activity due to depleting soil nutrient reserves via displacing nutrient cations by H^+ ions. Microbial abundance also associates with the soil specific surface, which is positively related to fraction of silt and colloid (clay, organic matter) particles in Table 2. In fact, no relationship between DT_{50} and K_F was found when assuming topsoils only. It could also be hypothesized that compound dissipation may associate with the dissociation degree of the molecule. Under decreasing soil pH conditions, the compound's cationic fraction increases and its neutral form fraction decreases. The cationic form is strongly sorbed on negatively charged soil, however, compound sorption due to ionic strength does not protect the molecule against dissipation, as would occur if the neutral form was sorbed on organic matter due to the hydrogen bonding or van der Waals forces (i.e., negative but not significant correlation between DT_{50} and soil pH when evaluated for topsoils).

The trimethoprim dissipation half-lives reported here were mostly higher in value than that (26.1 days) reported by Lin and Gan (2011) for 1 of 2 two soils. Whereas, in the second soil used by Lin and Gan (2011), the compound appeared to be recalcitrant to microbial degradation. These differences might be explained by the higher initial concentration applied to a larger amount of soil (concentration of 2 $\mu g\ g^{-1}$ in 50 g of soil) in our study compared to those (concentration of 0.04 $\mu g\ g^{-1}$ in 5 g of soil) used by Lin and Gan (2011). Similar dissipation half-lives (62.4–161 days) to those reported here were reported by Wu et al. (2012) in 3 soils under aerobic conditions. Wu et al. (2012) also found that compound dissipation half-life moderately decreased after adding manure and considerably declined under anaerobic conditions

Table 6
The correlation coefficients describing relationships between recovery of the studied pharmaceuticals and soil properties: pH_{H2O}, pH_{KCl}, organic carbon content, Cox (%), $CaCO_3$ content (%), cation exchange capacity, CEC ($mmol^+\ kg^{-1}$), soil hydrolytic acidity, HA ($mmol^+\ kg^{-1}$), exchangeable acidity, EA ($mmol^+\ kg^{-1}$), basic cation saturation, BCS ($mmol^+\ kg^{-1}$), sorption complex saturation, SCS (%), particle density ρ_s ($g\ cm^{-3}$), clay, silt and sand contents (%).

Soil property	Trimethoprim		Sulfamethoxazole		Clindamycin		Clarithromycin		Atenolol		Metoprolol		Carbamazepine	
	All soils	Topsoils	All soils	Topsoils	All soils	Topsoils	All soils	Topsoils	All soils	Topsoils	All soils	Topsoils	All soils	Topsoils
pH_{H2O}	−0.204	0.154	0.219	−0.277	−0.259	−0.370	−0.150	−0.601	−0.161	−0.162	−0.750**	−0.627*	−0.694**	−0.587
pH_{KCl}	−0.395	−0.126	0.166	−0.426	−0.114	−0.240	−0.125	−0.708*	−0.351	−0.460	−0.802	−0.704*	−0.558	−0.353
Cox	0.376	0.095	−0.676*	−0.470	−0.063	−0.060	−0.465	−0.330	−0.094	−0.169	0.068	−0.436	0.166	−0.208
$CaCO_3$	−0.205	0.088	−0.082	−0.629*	−0.263	−0.106	−0.380	−0.460	−0.457	−0.467	−0.728**	−0.899***	−0.468	−0.589
CEC	0.125	0.019	−0.617*	−0.685*	−0.507	−0.454	−0.747**	−0.633*	−0.340	−0.355	−0.209	−0.606*	−0.033	−0.362
HA	0.142	−0.080	−0.209	0.069	0.151	0.187	0.147	0.409	0.236	0.237	0.644*	0.548	0.588*	0.494
EA	0.006	−0.173	−0.182	0.014	0.032	0.086	0.347	0.664*	−0.118	−0.129	0.376	0.246	0.602*	0.529
BCS	0.085	0.037	−0.538	−0.632*	−0.519	−0.453	−0.747**	−0.669*	−0.381	−0.378	−0.353	−0.679*	−0.172	−0.446
SCS	−0.205	−0.014	0.050	−0.265	−0.366	−0.407	−0.484	−0.789**	−0.151	−0.132	−0.588*	−0.499	−0.602*	−0.537
ρ_s	−0.304	−0.122	0.539	0.384	0.470	0.496	0.617*	0.496	0.155	0.182	0.105	0.537	0.041	0.408
Clay	−0.043	−0.127	−0.571*	−0.686*	−0.472	−0.396	−0.589*	−0.457	−0.593*	−0.620*	−0.482	−0.843**	−0.199	−0.474
Silt	0.252	0.119	−0.443	−0.315	−0.712**	−0.760**	−0.638*	−0.463	0.095	0.164	0.307	0.062	0.090	−0.254
Sand	−0.231	−0.095	0.497	0.413	0.736**	0.796**	0.683*	0.519	−0.006	−0.061	−0.221	0.073	−0.056	0.320

* $p < 0.05$.
** $p < 0.01$.
*** $p < 0.001$.

Table 7
The correlation coefficients describing relationships between dissipation half-life of the studied pharmaceuticals and soil properties: pH_{H2O}, pH_{KCl}, organic carbon content, Cox (%), $CaCO_3$ content (%), cation exchange capacity, CEC ($mmol^+ kg^{-1}$), soil hydrolytic acidity, HA ($mmol^+ kg^{-1}$), exchangeable acidity, EA ($mmol^+ kg^{-1}$), basic cation saturation, BCS ($mmol^+ kg^{-1}$), sorption complex saturation, SCS (%), particle density ρ_s ($g\ cm^{-3}$), clay, silt and sand contents (%).

Soil property	Trimethoprim		Sulfamethoxazole		Clindamycin		Atenolol		Metoprolol	
	All soils	Topsoils	All soils	Topsoils	All soils	Topsoils	All soils	Topsoils	All soils	Topsoils
pH_{H2O}	0.551	−0.545	0.5357	−0.794**	0.415	0.3855	0.179	0.196	−0.571*	−0.642*
pH_{KCl}	0.594*	−0.442	0.627*	−0.761**	0.372	0.535	0.038	0.056	−0.487	−0.520
Cox	−0.613*	0.084	−0.611*	−0.1400	−0.371	0.025	−0.083	−0.075	−0.006	−0.137
$CaCO_3$	0.403	−0.404	0.100	−0.3762	0.715**	0.599	0.151	−0.092	−0.186	−0.384
CEC	−0.312	−0.345	−0.508	−0.3186	0.136	0.192	0.044	−0.105	−0.174	−0.412
HA	−0.336	0.567	−0.332	0.855***	−0.321	−0.622*	−0.203	−0.211	0.583*	0.599
EA	−0.266	0.277	−0.287	0.866***	−0.154	−0.335	−0.142	−0.183	0.328	0.283
BCS	−0.217	−0.449	−0.404	−0.497	0.207	0.326	0.091	−0.042	−0.305	−0.518
SCS	0.318	−0.474	0.283	−0.693*	0.313	0.345	0.219	0.205	−0.518	−0.540
ρ_s	0.479	0.273	0.563*	0.566	0.133	−0.178	−0.170	−0.168	0.384	0.614*
Clay	−0.207	−0.438	−0.375	−0.178	0.251	0.685*	−0.012	−0.166	−0.270	−0.473
Silt	−0.413	−0.291	−0.570*	−0.138	−0.021	−0.310	0.137	0.0478	0.031	−0.160
Sand	0.418	0.350	0.589*	0.162	−0.016	0.191	−0.128	−0.0199	0.009	0.229

The correlation coefficients could not be calculated for clarithromycin and carbamazepine due their persistency in some soils.
* $p < 0.05$.
** $p < 0.01$.
*** $p < 0.001$.

(similar to the findings of Mohring et al., 2009). Wu et al. (2012) also demonstrated the relatively large contribution of biotic compound transformation by comparing compound dissipation half-lives in sterile and non-sterile soils.

The degree of degradability of studied compounds in soils can be assessed base on FAO (2000) criteria: readily degradable (<20), fairly degradable (20–60), slightly degradable (60–180) and very slightly degradable (>180), which indicate compound instability, low stability, stability and high stability, respectively. The compound mobility is usually evaluated according FAO (2000) using the logarithm of organic carbon partition coefficient (Log K_{oc}): highly mobile (<1), mobile (1–2), moderately mobile (3–4), slightly mobile (3–4), hardly mobile (4–5) and immobile (>5). Assuming that the K_F value can be approximated as the K_{oc} value multiplied by the fraction of the soil organic carbon content (set at 2%), criteria could be modified for the K_F parameter as follows: highly mobile (<0.2), mobile (0.2–2), moderately mobile (2–20), slightly mobile (20–200), hardly mobile (200–2000) and immobile (>2000), due to very low, low, moderate, high, very high and extremely high sorption affinity to soils, respectively. Trimethoprim persistence in some of our soils and substrates was high but its sorption affinity (except for on soil substrates) (Fig. 1a, Kodešová et al., 2015) was also high. Thus, due to its slight mobility in topsoils, it could be assumed that trimethoprim should not greatly migrate in the subsurface water environment when applied (e.g., via irrigation, manure or sediments) on soils. However, if this compound reaches (due to preferential flow or from other sources such as river banks) layers of lower trimethoprim sorption affinity and slower compound dissipation, it may contaminate ground and drinking water. This postulation is supported by trimethoprim occurrence in groundwaters (e.g., Fram and Belitz, 2011; Godfrey et al., 2007; Li, 2014). However, trimethoprim was not found in groundwater samples analyzed by Radovic et al. (2015).

Sulfamethoxazole dissipation half-lives in topsoils varied between 4.7 and 15.1 days and again significantly differed from the dissipation half-lives in substrates (65.6 and 152 days) (Table 4). Larger values were found in three Cambisols and in soils on sand and gravely-sand substrates. Dissipation half-lives positively correlated with HA and EA

Table 8
The correlation coefficients describing relationships between the Freundlich adsorption coefficient, K_F ($cm^3\ \mu g^{1-1/n}\ g^{-1}$), recovery (%) and dissipation half-life, DT_{50} (day).

Pharmaceutical		All soils			Topsoils		
		K_F	RE	DT_{50}	K_F	RE	DT_{50}
Trimethoprim	K_F	1			1		
	RE	0.280	1		−0.170	1	
	DT_{50}	−0.765**	−0.549	1	−0.280	−0.343	1
Sulfamethoxazole	K_F	1			1		
	RE	−0.298	1		−0.012	1	
	DT_{50}	−0.365	0.542	1	0.822**	0.039	1
Clindamycin	K_F	1			1		
	RE	−0.528	1		−0.461	1	
	DT_{50}	0.442	−0.250	1	0.604*	0.059	1
Clarithromycin	K_F	1			1		
	RE	−0.480	1		−0.716*	1	
	DT_{50}	–	–	–	–	–	–
Atenolol	K_F	1			1		
	RE	−0.390	1		−0.427	1	
	DT_{50}	−0.045	−0.003	1	−0.098	0.062	1
Metoprolol	K_F	1			1		
	RE	−0.178	1		−0.586	1	
	DT_{50}	−0.270	0.545	1	−0.499	0.570	1
Carbamazepine	K_F	1			1		
	RE	0.269	1		−0.072	1	
	DT_{50}	–	–	–	–	–	–

* $p < 0.05$.
** $p < 0.01$.

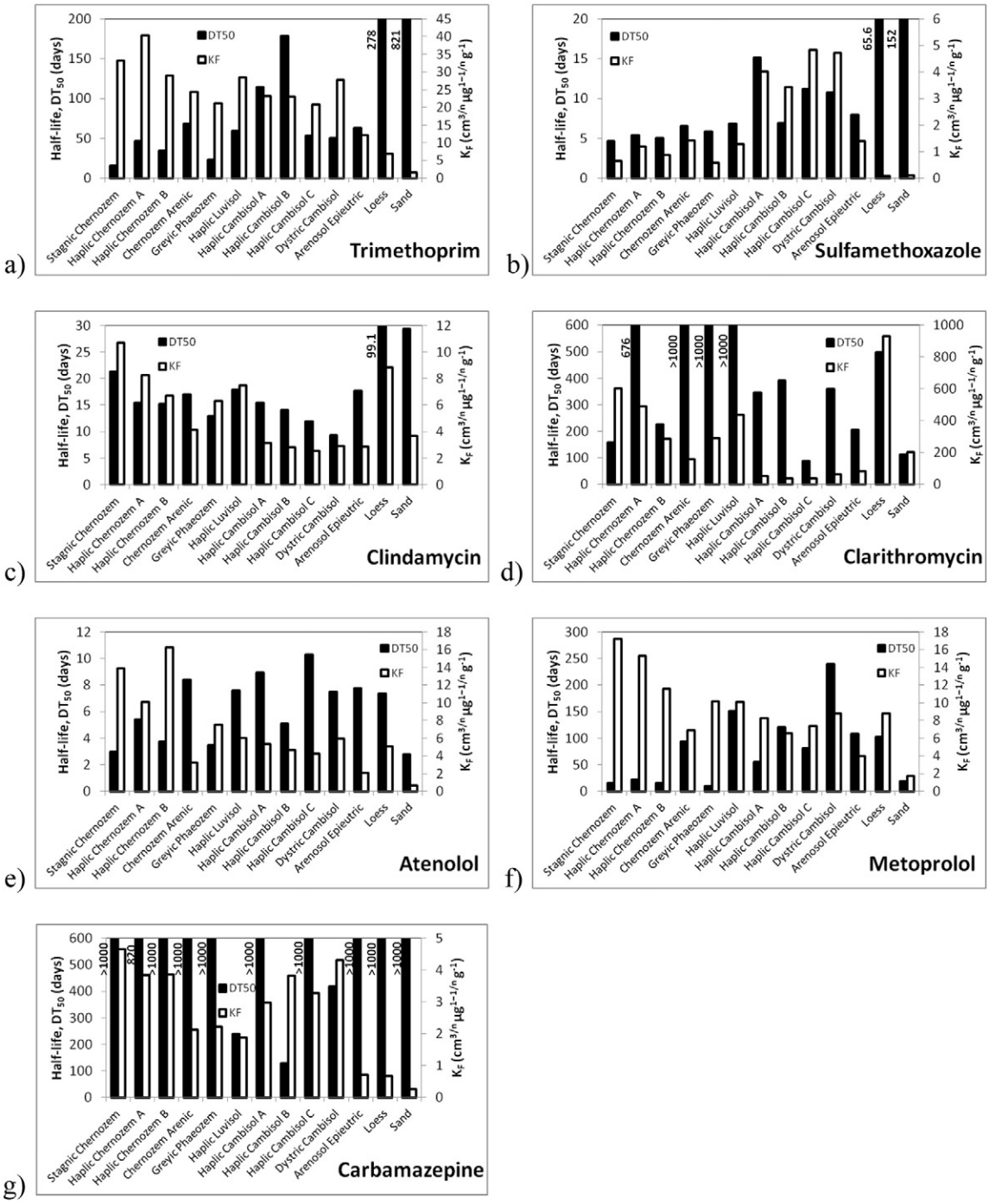


Fig. 1. The Freundlich adsorption coefficients, K_F ($cm^{3/n}\ \mu g^{1-1/n}\ g^{-1}$), for average n values and dissipation half-lives, DT_{50} (days), evaluated for each pharmaceutical and soil. Single values of K_F and DT_{50} were obtained via fitting all experimental data points using the corresponding equation.

and negatively with soil pH, when evaluated for topsoils (Table 7). The negative correlation between sulfamethoxazole dissipation half-lives and soil pH can be accounted for by the negative correlation of soil pH with the K_F (Kodešová et al., 2015) and the negative correlation of dissipation half-lives and K_F values (Table 8, Fig. 1b). This means that, with decreasing soil pH the negatively charged fraction of a molecule decreases and the fraction of neutral form increases, which is more strongly sorbed onto soils. Anions adsorb less strongly to soils containing organic carbon and clay than their neutral counterparts mainly because of electrostatic repulsion between the negatively charged compounds and negatively charged soil particle surfaces. Thus, compounds are less bioavailable at lower pH values and, therefore, degrade more slowly than under higher pH conditions. However, it should once again be noted that dissipation half-lives may be associated with the microbial conditions in different soils, as discussed above for trimethoprim. The similar DT_{50} patterns for sulfamethoxazole and trimethoprim (i.e. low DT_{50} values in Chernozems and high DT_{50} values in Cambisols and substrates) are shown in Fig. 1a and b. In addition, a positive correlation was found between the values obtained for both compounds when evaluated for all soils ($R = 0.846$, $p < 0.001$), but not when evaluated for topsoils ($R = 0.368$).

The sulfamethoxazole dissipation half-lives in topsoils reported here corresponded to the values summarized and experimentally evaluated for soils by Srinivasan and Sarmah (2014) (4–13 days), as well as the

values published by Lin and Gan (2011) (9–11 days). However, the dissipation half-lives obtained for substrates were significantly higher (66 and 152 days) than the values (12–14 days) presented for subsoils by Srinivasan and Sarmah (2014). The higher DT_{50} values in our study could be due to the considerably lower organic carbon contents and the associated lower microbial activities in comparison to those in the study by Srinivasan and Sarmah (2014). Our values for subsoils were also higher than the dissipation half-lives (41.5–57.3 days) evaluated for lake sediments by Zhang et al. (2013). The dissipation half-lives (38.5–55.0 days) published for three topsoils by Wu et al. (2012) were higher than our topsoil values. This might be due to the fact that they applied a considerably higher initial concentration (10 μg g^{-1}) to a smaller amount of soil (5 g) and incubated samples under different conditions (temperature of 25 ± 3 °C, water content corresponding to 60% of the maximum hording capacity). The biotic sulfamethoxazole transformation appeared to again be the major factor affecting compound dissipation (Lin and Gan, 2011; Srinivasan and Sarmah, 2014; Wu et al., 2012).

As was documented in several studies, including that by Kodešová et al. (2015) (Fig. 1b), sulfamethoxazole sorption affinity to soils is low. Thus, its transport through topsoils (where this contaminant may relatively quickly dissipate) could be rapid under wet conditions (i.e., under intensive precipitation or irrigation). Subsequently, due to its low sorption affinity and high subsoil dissipation half-life, sulfamethoxazole easily migrates in the subsurface water environment. This assumption is supported by findings published by Fram and Belitz (2011); Godfrey et al. (2007); Li (2014) and Loos et al. (2010), among others, who documented frequent sulfamethoxazole occurrence in groundwaters.

The clindamycin dissipation half-life varied between 12 and 21.3 days in topsoils and differed from the dissipation half-lives found in substrates (29.4 and 99.1 days) (Table 4). The topsoil dissipation half-lives did not significantly vary and there was no similar trend with respect to the soil type, as was discussed above for trimethoprim and sulfamethoxazole. The clindamycin dissipation half-lives positively correlated with clay content (Table 7) and with K_F value (Table 8, Fig. 1c), which is in agreement with the anticipated increasing compound persistency with its increasing sorption. Our results could only be compared with published compound dissipation half-lives in biosolids (Wu et al., 2009, and Chenxi et al., 2008). In both studies, fast degradation in the first few days and stabilization thereafter was documented, which was probably associated with the more favorable conditions for compound microbial degradation in these biosolids (i.e. different abundance and activity of degradable microorganisms in mostly organic material) compared to that in our soils. Assuming evaluated sorption coefficients and dissipation half-lives, clindamycin should not widely spread in the subsurface water environment. This is supported by the low clindamycin mobility in ground water documented by de Jongh et al. (2012), who found this compound in the surface waters but not in the river bank filtrates.

The clarithromycin dissipation half-lives did not differ between topsoils and substrates (Table 4). The lowest value was obtained in the Haplic Cambisol on quartzite (88.9 days) and for sand (112 days) and almost no dissipation was observed in three of the topsoils. Regression analysis (relating dissipation half-lives to other soil properties and K_F value) was not carried out due to the compound's persistency in these soils. Fig. 1d shows no evident trend of compound dissipation half-life with respect to K_F value and also no evident trend with respect to soil type. In general, clarithromycin is very persistent, and also strongly sorbs onto soil materials. Thus, this pharmaceutical should not migrate in the environment unless sorbed onto mobile colloids or eroded soil particles, etc. It should be mentioned that very fast degradation of clarithromycin was observed in biosolids by Chenxi et al. (2008), which was probably associated with the favorable microbial conditions (i.e. high nutrition content and specific surface area) in these organic materials.

The dissipation half-lives of atenolol did not differ between topsoils and substrates (Table 4). Values varied between 2.8 and 10.3 days. Larger values were found in Cambisols and in soils on sand and gravely-sand substrates. Fig. 1e indicates that higher dissipation half-lives were obtained in soils of lower compound sorption. However, no close relationship was found between dissipation half-life and K_F values (Table 8) or soil properties (Table 7). Atenolol dissipation in soils was considerably faster than dissipation in the activated sludge (DT_{50} = 17.8 days) and similar to that in the activated sludge with sodium acetate (DT_{50} = 6.2 days) that was reported by Ribeiro et al. (2013). Assuming evaluated sorption coefficients (Fig. 1e, Kodešová et al., 2015) and dissipation half-lives (Fig. 1e, Table 4), groundwater contamination by atenolol should not be considered a likely environmental threat. Low atenolol mobility in groundwater was again documented by de Jongh et al. (2012), who found this compound in the surface waters but not in the river bank filtrates.

The metoprolol dissipation half-lives did not differ between topsoils and substrates (Table 4). Values varied between 15 and 239 days. Larger values were found in Cambisols and in soils on sand and gravely-sand substrates. Fig. 1f shows that higher dissipation half-lives were obtained in soil with lower compound sorption, but this relationship was not statistically significant (Table 8). A negative correlation was found between the dissipation half-lives and soil pH (Table 7). This can be accounted for in a similar way as was discussed above for trimethoprim (i.e., better conditions for compound microbial degradation in Chernozems compared to those in other soil and a negligible impact of compound sorption on metoprolol dissipation rates). Metoprolol persistence in soils has been reported to be rather higher than in activated sludge (DT_{50} = 15.5 days) and activated sludge with sodium acetate (DT_{50} = 12.8 days) (Ribeiro et al., 2013). While atenolol and metoprolol dissipation half-lives in sludge were similar, both pharmaceutical dissipation half-lives considerably differed in our soils. Metoprolol moderately sorbed in our studied soils (except sand substrate) (Fig. 1f). Thus, despite its great persistency in soils, metoprolol should not migrate significantly in subsurface water. For instance, de Jongh et al. (2012) documented metoprolol's occurrence in the surface waters but not in the river bank filtrates. On the other hand, metoprolol behaves similarly to trimethoprim (i.e. similar compound dissipation rates and impact of soil pH on compound sorption affinity), which has been reported in groundwater by several authors. Thus, we assume that metoprolol could potentially contaminate groundwater and, therefore, its analysis should be included in groundwater quality monitoring studies.

Carbamazepine, which is considered to be one of the most persistent pharmaceutical in the environment due to its stable heterocyclic structure, for the most part, did not noticeably degrade during our experiments. Thus, no relationships between dissipation half-lives and other

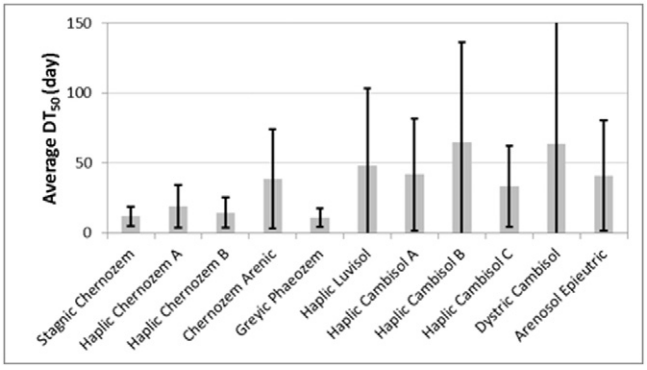


Fig. 2. Average dissipation half-lives, DT_{50} (days), and standard deviations (error bars) calculated from trimethoprim, sulfamethoxazole, clindamycin, atenolol, and metoprolol half-lives in topsoils.

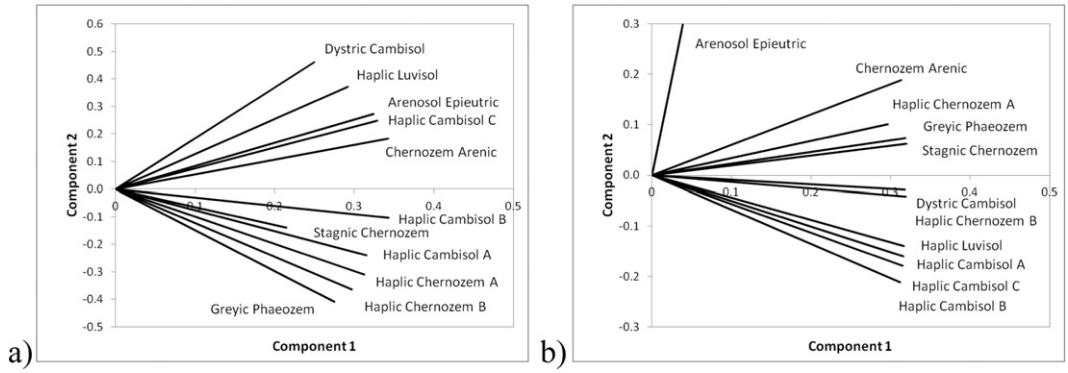


Fig. 3. Plot of component weights evaluated from trimethoprim, sulfamethoxazole, clindamycin, atenolol, and metoprolol dissipation half-lives (a) and Freundlich sorption coefficients (b) in topsoils.

soil properties could be assessed. Carbamazepine degradation has been previously evaluated in many studies. Our results were similar to the dissipation half-lives (355–1624 days) published by Dalkmann et al. (2014) and were moderately higher than the dissipation half-lives evaluated in outdoor mesocosms by Walters et al. (2010) (462–533 days) and Grossberger et al. (2014) (147 and >200 days). Considerably lower values, obtained under laboratory conditions, were reported by Monteiro and Boxall (2009) (60 days), Yu et al. (2013) (28.0–39.1 days) and Li et al. (2013) (46.2–173.3 days). Lower values were also obtained under field conditions (97.6 and 74.5 days for surface and subsurface soils, respectively) by Al-Rajab et al. (2015). It is widely accepted that the organic matter fraction substantially reduces carbamazepine bioavailability (e.g., Yu et al., 2013; Al-Rajab et al., 2015). Applications of different nitrifying enrichments had variable impacts on carbamazepine degradation; e.g., moderately positive (Dawas-Massalha et al., 2014) or none (Kruglova et al., 2014). The capability of the fungi *Trametes versicolor* to efficiently degrade carbamazepine has been reported by Jelic et al. (2012). Carbamazepine, due to its great persistency and low sorption affinity in soils, is a highly mobile compound in the subsurface water environment. Thus, this compound frequently occurs in groundwater (e.g., Fram and Belitz, 2011; Godfrey et al., 2007; Li, 2014; Huntscha et al., 2012; Loos et al., 2010; Radovic et al., 2015).

3.3. Dissipation of pharmaceuticals with respect to soil types

Fig. 2 shows the average values of the tested pharmaceuticals' dissipation half-lives in topsoils calculated from data obtained for trimethoprim, sulfamethoxazole, clindamycin, atenolol and metoprolol. It is evident that noticeably lower variability of dissipation half-lives (and also lower average values) was obtained for soils on loess and marlite substrates. Considerably larger variability was observed in all Cambisols and soils on sand or gravely-sand substrates. This may be attributed to the favorable microbial conditions (with respect to biodegradation of all studied pharmaceuticals) in soils on loess and marlite substrates compared to those in the other soils. The large variability of dissipation half-lives in Cambisols may indicate a selective ability of the microbial community to decompose particular compounds. Another reason could be that some antibiotics killed or inhibited growth of microorganisms in the soils of less favorable microbial conditions (Caracciolo et al., 2015).

Principal component analysis (Fig. 3a), again assuming only trimethoprim, sulfamethoxazole, clindamycin, atenolol and metoprolol dissipation half-lives in topsoils, indicated similar compound behaviors (considering their dissipation rates) in similar soil types and substrates (e.g., Group 1: Haplic Chernozem A — loess, Haplic Chernozem B — loess, Greyic Phaeozem — loess; Group 2: Arenosol Epieutric — quartz sand, Haplic Cambisol C — quartzite, Chernozem Arenic — gravely sand), which again may be associated with the similar microbial conditions. Apparently, different results were obtained from principal component analysis when assuming Freundlich sorption coefficients (Fig. 3b), which associated with their sorption mechanisms (Kodešová et al., 2015). While pharmaceutical dissipation half-lives were controlled by overall soil quality (e.g., soil particles aggregation, water content and aeration, nutrition content and biological conditions), their sorption were mainly controlled by soil chemical properties (which is in detail discussed our previous study).

4. Conclusions

The dissipation half-lives of 7 pharmaceuticals in 13 representative soils of the Czech Republic were evaluated. The greatest persistence in soils was measured for carbamazepine, followed by clarithromycin, trimethoprim, metoprolol, clindamycin, sulfamethoxazole and atenolol. Dissipation half-lives (at least partly) reflected the sorption of the studied pharmaceuticals onto soil particles and increased with increasing sorption (sulfamethoxazole and clindamycin), which is usually presumed. However, in 3 cases (atenolol, metoprolol and trimethoprim) the dissipation half-lives decreased with increasing sorption and carbamazepine and clarithromycin mostly did not considerably degrade during our experiments. Based on these findings, we propose that pharmaceutical persistence in soils is mostly depended on soil type. In general, for compounds that were degradable in the studied soils, lower average dissipation half-lives and variability were calculated for soils of better quality (soils with well-developed structure, high nutrition content and associated biological conditions as Chernozems) in comparison to those of lower quality (Cambisols). Regarding the dissipation rates of the studied compounds, and sorption affinities, which were evaluated in our previous study, the highest potential to migrate in the soil water environment is expected for carbamazepine, followed by sulfamethoxazole, trimethoprim and metoprolol. Based on our findings, extended transport of clindamycin, clarithromycin and atenolol through the vadose zone seems improbable.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2015.11.085.

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