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RESEARCH ARTICLE

Impact of sludge stabilization processes and sludge origin (urban or hospital) on the mobility of pharmaceutical compounds following sludge landspreading in laboratory soil-column experiments

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Abstract This study aimed to determine the effect of sludge stabilization treatments (liming and anaerobic digestion) on the mobility of different pharmaceutical compounds in soil amended by landspreading of treated sludge from different sources (urban and hospital). The sorption and desorption potential of the following pharmaceutical compounds: carbamazepine (CBZ), ciprofloxacin (CIP), sulfamethoxazole (SMX), salicylic acid (SAL), ibuprofen (IBU), paracetamol (PAR), diclofenac (DIC), ketoprofen (KTP), econazole (ECZ), atenolol (ATN), and their solid–liquid distribution during sludge treatment (from thickening to stabilization) were investigated in the course of batch testing. The different sludge samples were then landspread at laboratory scale and leached with an artificial rain simulating 1 year of precipitation adapted to the surface area of the soil column used. The quality of the resulting leachate was investigated. Results showed that ibuprofen had the highest desorption potential for limed and digested urban and hospital sludge. Ibuprofen, salicylic acid, diclofenac, and paracetamol were the only compounds found in amended soil leachates. Moreover, the leaching potential of these compounds and therefore the risk of groundwater contamination depend mainly on the origin of the sludge because ibuprofen and diclofenac were present in the leachates of soils amended with urban sludge, whereas paracetamol and salicylic acid were found only in the leachates of soils amended with hospital sludge. Although carbamazepine, ciprofloxacin, sulfamethoxazole, ketoprofen, econazole, and atenolol were detected in some sludge, they were not present in any leachate. This reflects either an accumulation and/or (bio)degradation of these compounds (CBZ, CIP, SMX, KTP, ECZ, and ATN), thus resulting in very low mobility in soil. Ecotoxicological risk assessment, evaluated by calculating the risk quotients for each studied pharmaceutical compound, revealed no high risk due to the application on the soil of sludge stabilized by liming or anaerobic digestion.

Keywords Pharmaceutical compounds · Sludge stabilization · Liming · Anaerobic digestion · Desorption · Soil column · Leachate

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Introduction

Wastewater treatment plants (WWTPs), hospital effluents and livestock activities are routes through which organic micropollutants enter the environment and natural resources (Daughton and Ternes 1999). Among these substances, pharmaceutical compounds are of great interest because of their potential adverse effect on health (Ellis 2006). Owing to improvements in analytical techniques for determining concentrations in complex matrices such as sludge and soils, many studies were able to focus on the occurrence and outcome of pharmaceutical compounds in wastewater treatment

(Deblonde et al. 2011) and their potential toxicity on aquatic organisms (Kümmerer 2009; Fent et al. 2006). As most pharmaceutical compounds are consumed and subsequently excreted by the patient, the main environmental exposure route is normally via a wastewater treatment plant. Therefore, the partitioning of pharmaceutical compounds between the aqueous phase and the biosolids (sludge) present in a wastewater treatment plant is important in describing the fate of the micropollutant during this process. WWTPs are not designed to treat micropollutants, although a wide range of emerging contaminants are removed from water. Biological degradation and sorption onto sludge are the main processes responsible for the removal of pharmaceutical compounds in wastewater treatment (Ternes et al. 2004; Jelic et al. 2011). Sorption onto sludge depends on the physical and chemical properties of the compounds and on the characteristics of sewage sludge.

The adsorption coefficient (Kd) between water and solid (e.g., soils, sludge) phases is an important parameter in determining the environmental fate of a chemical. Many Kd values have been published for various pharmaceuticals including antibiotics, anticancers, anti-inflammatories, cardiovasculars, central nervous system drugs and hormones for different sludge types (Pomiès et al. 2013). Two compartments are usually defined for sorption: the aqueous compartment which is the dissolved phase and the solid phase of the sludge or mixed liquor (that can be considered as a biosorbent). When a micropollutant is distributed between these two compartments, sorption and desorption phenomena occur simultaneously.

At low concentration of micropollutants (e.g., $S_{mp}<$ 1 μg/L), the linear model is considered (Limousin et al., 2007). A single sorption coefficient (or partition coefficient) noted Kd is used. For pharmaceutical compounds, Ternes et al. (2004) showed insignificant removal by sorption when Kd value was lower than 0.5 L./kg TS, particularly for diclofenac, ibuprofen, and carbamazepine. Most of the time, the Kd parameter is assumed to be an intrinsic physicochemical property of a micropollutant. Thus, modellers considered Kd values taken from the literature. Nonetheless, it is required to differentiate Kd values according to the type of solid matrix (e.g., activated sludge, particular content of raw/treated wastewater, soil, and sediment) (Pomiès et al. 2013). The sorption potential of hydrophobic contaminants is sometimes described according to their *n*-octanol/water partition coefficient (K_{OW}) where contaminants with log $K_{OW}<2.5$ would have a low sorption potential, those with $2.5<\log K_{OW}<4.0$ would have a medium sorption potential, and those with log $K_{OW}>4.0$ would have a high sorption potential. Indeed other interactions than hydrophobic ones (represented by K_{OW}) play an important role in sorption mechanisms: electrostatic interactions (Ternes et al., 2004), cationic exchanges, cationic bridges, surface complexation, and hydrogen bridges (Tolls, 2001). Sorption mechanism is complex and still remains not sufficiently documented. As explained before, the estimation of Kd value using K_{OW} for organic micropollutants is not sufficiently accurate to be used for prediction. Chemicals with a low Kd value (around log Kd<2.6) will remain mainly in the aqueous effluent, and those with a high Kd value (around log Kd>3.6) will be predominantly adsorbed by the biosolids (sludge) phase. In the latter case, precautions may be required for the disposal of sludge to agricultural land as there are potential risks associated with leaching of desorbed chemicals and the movement of these chemicals into crops destined for livestock or human consumption (Berthod et al. 2014).

Liming and anaerobic digestion are the two main sludge stabilization processes used. These two types of treatment could greatly impact sludge characteristics and thus could greatly influence the concentrations of micropollutants, their mobility, and phase distribution (Barret et al. 2012).

Final disposal of sewage sludge may then lead to a contamination of the environment (soil, water) by these emerging contaminants and lead to serious environmental problems in some cases (Clarke and Smith 2011; Martín et al. 2012, 2015). In Europe and especially in France, agricultural valorization by landspreading represents the main means of sludge disposal (Kelessidis and Stasinakis 2012). Different processes such as chemical or biological stabilization treatments are applied to sewage sludge before using it for landspreading.

To simulate landspreading in laboratory, leaching tests on soil columns are employed but most of column studies on organic micropollutants sorption and biodegradation involve the use of a solution of pharmaceutical compounds as rainfall. Many studies have employed a similar test system using repacked dried and sieved soil to study the mobility of pesticides and veterinary antibiotics in different types of organic fertilizers such as slurry and manure (Worrall et al. 2001; Haberhauer et al. 2002; Oppel et al. 2004; Kay et al. 2005; Salvia et al. 2014).

Highly mobile organic compounds have the potential to leach into groundwater, whereas strongly sorbing pharmaceutical compounds can accumulate in topsoil. Sorption is related to the physicochemical properties of contaminants and soil characteristics such as soil pH, soil type, organic matter content, clay minerals content, and cation exchange capacity (CEC) (Drillia et al. 2005; Topp et al. 2008; Corada-Fernández et al. 2014; Vithanage et al. 2014). Even at trace levels, these organic micropollutants may have adverse effects on the exposed organisms. Moreover, the presence of antibiotics in the environment can lead to bacteria resistance (Stalder et al. 2012). Furthermore, certain crops such as sweetcorn, potatoes, and sweet beans take up pharmaceuticals and hormones, and in some cases, these pollutants can translocate from roots to aerial tissues (Wu et al. 2012). A recent review of Clarke and Cummins (2015) demonstrated that the scientific literature suggests that the application of biosolids to

agricultural soils can be sustainable and economical on the basis of recycling of nutrients and the waste disposal of sewage sludge. However, it also has potential risks with respect to the occurrence of POPs and PPCPs present in the sludge. Throughout the literature, the effects or bioaccumulation of contaminants has mainly focused on aquatic environments and organisms, and there is far less known about the effect of the compounds on terrestrial organisms. Available literature suggests that the risk of direct human exposure to biosolids is very low and realistically may involve only those who work with biosolids such as farmers and sludge workers. The risk of indirect exposure, however, can occur through several pathways (consumption of food crops, animal up-take to meat or milk, or drinking water). Risk assessment has been adopted to assess the environmental fate of contaminants in soils amended by fertilizers (i.e., sewage sludges, manure…), with the determination of risk quotient (RQ) (Li et al., 2015; Martín et al. 2012; Martín et al. 2015 and Wu et al. 2014). Judicious selection of suitable modelling approaches is required to ensure accurate representation of human/environmental risks from emerging contaminants.

The aim of this study was to determine the effect of stabilization treatments (liming and anaerobic digestion) applied to sludge of different origins (urban and hospital) on the mobility of different pharmaceutical compounds following lansdpreading. To achieve this objective, the sorption and desorption potential of pharmaceutical compounds and their phase distribution during sludge treatment (from thickening to stabilization) was investigated in batch tests. In a later phase, the different sludge samples were landspread at laboratory scale, and columns were leached with an artificial rain simulating 1 year of precipitation in Limousin. The quality of the resulting leachate was investigated.

Materials and methods

Sludge origin and stabilization treatments applied to the sludge

A wastewater treatment plant in Bellecombe (France) was designed to treat separately the hospital wastewater (450 beds, 5400 inhabitant equivalents) and the domestic wastewater (10, 600 and 16,000 inhabitant equivalents) with conventional activated sludge systems. Sludge from the urban line and the hospital line of treatment were studied in this work during two campaigns. At pilot scale on the WWTP, activated sludge from the plant was first flocculated with a cationic polymer and thickened through a dewatering grid. At pilot scale, a device was installed on the site to realize a thickening treatment of the activated sludge. For this, 800 L of activated sludge was sampled from the aeration basin of the WWTP and put in a tank in which a cationic polymer was added under agitation to flocculate the sludge. The flocculated sludge was then manually discharged through a flat dewatering grid, allowing the formation of thickened sludge. After this thickening treatment on the activated sludge, one part of the thickened sludge from the two origins (urban and hospital) was stabilized by the chemical process of liming and another part by the biological treatment of anaerobic digestion (Fig. 1). The liming treatment consisted in the addition of lime at a mass corresponding to 30 % of the sludge dry matter. Anaerobic digestion was performed on another part of the thickened sludge. For this biological treatment, biochemical methane potential tests were carried out in glass bottles and maintained at mesophilic temperature (37 °C) for 25 days. The organic mass load was 1 gCOD/gVSS, and a buffer ($NaHCO_3$ and Na_2CO_3 at 3 g/L; Eskicioglu et al., 2006) was used to prevent the pH from decreasing.

The different sludge samples obtained and used for landspreading on soil columns were identified as follows: limed sludge from urban effluents (LS-U), digested sludge from urban effluents (DS-U), limed sludge from hospital effluents (LS-H), and digested sludge from hospital effluents (DS-H). Total solids (TS), total suspended solids (TSS), and volatile suspended solids (VSS) were measured on sludge according to normalized methods (AFNOR, NF T90-105 1997). TS, TSS, and VSS are measured of dry matter. The standard deviation for triplicate samples was below 2 %. Sludge matter distribution is presented in Table 1. The protonic exchange capacity (PEC) corresponding to the number of ionizable functional groups in the sludge was measured according to the protocol described in Laurent et al. (2009).

Determination of pharmaceutical concentrations in solid and liquid matrices

The pharmaceutical compounds studied in this work belong to different therapeutic groups (antiepileptic drugs, antibiotics, anti-inflammatory and anti-fungal drugs, beta-blockers): carbamazepine (CBZ), ciprofloxacin (CIP), sulfamethoxazole (SMX), salicylic acid (SAL), ibuprofen (IBU), paracetamol (PAR), diclofenac (DIC), ketoprofen (KTP), econazole (ECZ), and atenolol (ATN). The chemical properties of these compounds are given in Table 2.

Pharmaceutical compound concentrations were measured in the total sludge, the soluble sludge fraction, and the soil and leachates from the soil columns. The soluble fraction of the sludge was obtained after centrifugation and filtration of the supernatant. Before analysis, the total sludge samples were freeze-dried. The soluble sludge fraction and leachates were identified as the fraction resulting from the filtration of the samples through a cellulose nitrate membrane (0.45 μm pore size). All the pharmaceutical compounds except CIP were extracted using the QuEChERS (Quick, Easy, Cheap, Effective, Rugged and Safe) method and analyzed by liquid

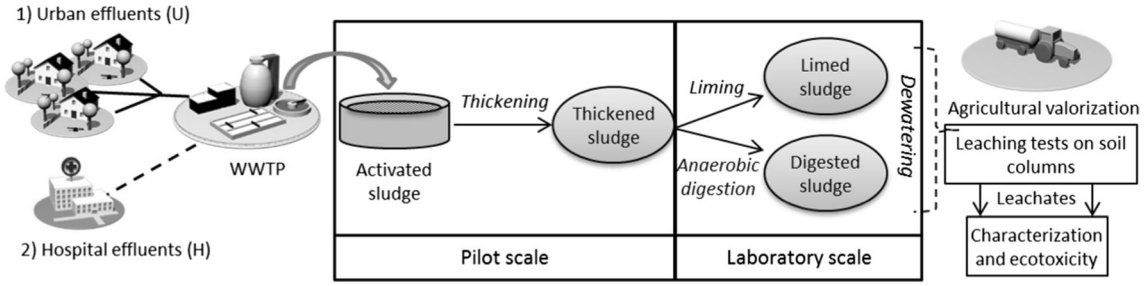


Fig. 1 Scheme of the experimental sludge treatment process

chromatography–time-of-flight-mass spectrometry (LC-TOF-MS) according to the technique described by Peysson and Vulliet (2013). The extraction of ciprofloxacin was performed on an ASE extractor 200 of Dionex mark (extraction solvent MeOH/ACN/0.2 M citric acid 40/40/20, pressure 120 bar, temperature 80 ° C). After extraction, each extract was purified by solid phase extraction. Purification was performed on an automatic extractor AutoTrace SPS Workstation Caliper mark (cartridge: StrataX, 3 mL, 200 mg, conditioning with methanol and then water, sample loading, rinsing the column with water, drying column, eluting with methanol, evaporation to dryness of the eluate; included in the mobile phase). Analyses were performed by the ISA laboratory (CNRS UMR5280).

For the sludge, the concentration of compounds in the particulate fraction (sorbed concentration) was calculated by the difference between the total and the soluble fractions (Eq. 1):

$$C_{\text{sorbed}} = C_{\text{total}} - C_{\text{soluble}} \quad (1)$$

where C_{sorbed} is the concentration of micropollutants in the particulate fraction of the sludge, C_{total} is the concentration in the total sludge sample, and C_{soluble} is the concentration in the soluble fraction of the sludge.

Evaluation of pharmaceutical compounds' mobility

Batch desorption tests on sludge

Following treatment of LS-U, LS-H, DS-U, and DS-H, desorption tests were conducted on centrifugation pellets. Pellets were resuspended with 0.02 mol/L NaCl (Pan et al. 2012), and the bottles were shaken at 180 rpm on a rotary shaker in the obscurity during 24 h. The sludge suspension was then centrifuged and filtered through a cellulose nitrate membrane (0.45 μm pore size). Pharmaceutical compound concentrations were determined in the soluble fraction after the desorption test, and the equilibrium desorption constant ($K_{\text{desorption}}$) was calculated according to Eq. 2:

$$K_{\text{desorption}} = \frac{C_{\text{soluble,24h}}}{C_{\text{sorbed}}} \quad (2)$$

where $C_{\text{soluble,24h}}$ is the concentration of micropollutants in the soluble fraction after the desorption test (expressed in μg/L), and C_{sorbed} is the concentration in the particulate fraction of sludge before the test, calculated by the difference between C_{total} (concentration in the total sludge in μg/g and converted in μg/L with the concentration of dry matter in g/L) and C_{soluble} (expressed in μg/L).

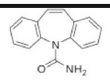
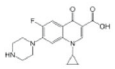
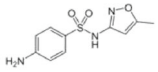
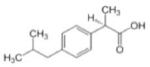
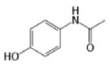
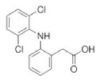
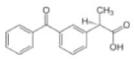
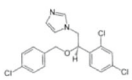
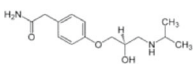
Leaching tests on soil columns

Soil characteristics The topsoil sample of grassland (A horizon; 0–10 cm) was air-dried and sieved to<2 mm. The soil pH was measured in water respecting a soil/solution ratio of 1:5 (NF ISO 10390 Standard). The CEC was determined by the 0.05 N cobalt hexamine method at the soil pH (NF ISO 31–130 Standard). The organic C and N contents were determined by dry combustion with a C/H/N elemental analyzer (Soil Analysis Laboratory of INRA Arras). The particle size distribution of the <2-mm fraction was determined by sieving and the pipette method after destruction of organic matter. The mineralogy of topsoil sample (<2 mm) was determined from crushed powder by X-ray diffraction (XRD) using a PANalytical X'Pert Pro diffractometer equipped with a

Table 1 Various sludge characteristics (TS, TSS and VSS are measured of dry matter)

	LS-U	DS-U	LS-H	DS-H
Total solids (g/L)	24.8	19.2	28.4	17.6
Total suspended solids (g/L)	20.1	11.5	23.6	10.1
Volatile suspended solids (g/L)	11.2	7.1	13.3	6.3
pH	11.9	8.1	12.1	8.3
PEC: protonic exchange capacity (moles of ionizable sites/kg TS of sludge)	6.68	12.75	7.52	19.02

Table 2 Chemical properties of the studied pharmaceutical compounds

Compound	Structure	pKa	Log K_{ow}	Characteristics at pH=7
Carbamazepine		13.9	2.25	Hydrophobic, positive charge
Ciprofloxacin		5.9-8.89	0.4	Zwitterion, neutral
Sulfamethoxazole		1.85-5.6	0.89	Negative charge
Salicylic acid		2.98	2.26	Negative charge
Ibuprofen		4.91	3.97	Negative charge
Paracetamol		9.38	0.46	Neutral
Diclofenac		4.15	4.51	Hydrophobic, negative charge
Ketoprofen		4.45	3.12	Hydrophobic, negative charge
Econazole		6.77	5.61	Neutral
Atenolol		9.6	0.16	Hydrophilic, positive charge

diffracted-beam monochromator. The XRD patterns were interpreted with X'PertHighScore software. Selected soil properties are presented in Table 3.

The studied soil was described as Cambisol (FAO 2006) developed on gneiss. This soil is slightly acidic (pH_{water}=5.6) with an organic matter content of 6 % and clay content of 24 %. The soil texture was characterized as loam. The mineralogical composition of the soil comprised mainly of quartz, phyllosilicates (smectite, chlorite, illite, and kaolinite), muscovite, plagioclase, and iron oxyhydroxydes.

Column leaching experiment All tests were performed according to the OECD guideline "Leaching in Soil Columns" (OECD 2004). The soil was packed in glass columns of 30 cm length and 5 cm diameter. The soil was introduced in small portions under gentle vibration of the columns to obtain uniform packing. The soil columns were then saturated from the bottom with artificial rain (0.01 mol/L $CaCl_2$ to control the ionic strength of the solution, pH=6.8) to their maximum water holding capacity (OECD 2004; Oppel et al. 2004).

Table 3 Physicochemical properties of soil

pH	Organic matter (g/kg)	CEC (cmol+/kg)	Sand (%)	Silt (%)	Clay (%)	Texture	Mineralogical composition
5.6	60.1	17	47	29	24	Loam	Clay minerals (smectite, chlorite, kaolinite, illite), plagioclase, quartz, muscovite, iron oxyhydroxydes

Each limed and digested sludge sample was applied on a column at a concentration corresponding to 3 tTS/ha, which is the recommended concentration for land application in France. Sludge was applied by manually mixing it with the first 4 cm of soil to simulate incorporation into soil. Artificial rain was added at an amount equal to 393 mL over 48 h. This was repeated six times to simulate 1 year of rainfall in the Limousin region. In fact, the annual rainfall is approximately 1020 mm, which corresponds to an amount of 2358 mL of rain for the surface area of the soil column used. Glass fiber disks on top of the columns ensured even distribution of the artificial rain. All tests were performed in obscurity at ambient temperature (Fig. 2). Moreover, a test was performed as a control with the soil, excluding sludge application.

Soil leachate analysis The different leachate samples obtained were identified as follows: Lix-T-soil, Lix-LS-U, Lix-DS-U, Lix-LS-H, and Lix-DS-H, corresponding to leaching of the soil amended with the different sludge samples (urban and hospital limed and digested sludge) in different columns.

During the test, leachates were collected and analyzed each 48 h to follow the evolution of physicochemical characteristics. During all the experiments, the columns were kept at ambient temperature, the volume of leachates collected was measured, and they were kept in the fridge at 4 °C and frozen for micropollutant analyses. The measured parameters were pH, conductivity, turbidity, TSS, UV absorbance at 254 nm, dissolved organic carbon (DOC), and SUVA index. The UV absorbance at 254 nm was measured by using a PharmaSpec 1700 spectrophotometer (Shimadzu, Japan; precision ± 0.005 cm^{-1}) with 1-cm-long quartz cells. DOC analyses were performed by a TOC-L analyzer (Shimadzu, Japan; precision ±50 μgC/L) according to the Non-Purgeable Organic Carbon measurement procedure. The specific UV absorbance (SUVA index) is defined as the ratio of the UV absorbance to the DOC values and was expressed in L cm/gC. The percentage of aromaticity was determined according to Eq. 3, with SUVA expressed in L m/mgC (Weishaar et al., 2003).

$$\text{Percentage of aromaticity} = 6.53 \times \text{SUVA} + 3.63 \quad (3)$$

Fluorescence excitation–emission matrices (EEM) of the sludge soluble fractions and the leachates were also measured using a Shimadzu RF-5301 PC spectrofluorophotometer (Shimadzu Corp., Kyoto, Japan) with a 1-cm-path quartz cuvette. Fluorescence EEM were constructed from a series of emission spectra up to 550 nm by varying the excitation wavelength from 220 to 450 nm in 5-nm increments. The stability of the fluorometer was checked by monitoring the Raman water peak. The relative fluorescence intensity was calculated with the maximal fluorescence intensity of each significant area divided by the dissolved organic carbon concentration of the analyzed sample.

At the end of the leaching test (30 days), for each column (corresponding to each kind of landspreaded sludge), a composite sample leachate was created with all volumes of leachates collected. Then, the composite sample leachate was characterized. The pharmaceutical compound concentrations in the leachates were also determined. The method used to determine the concentration levels of pharmaceutical compounds in the leachate is the one described in "Determination of pharmaceutical concentrations in solid and liquid matrices".

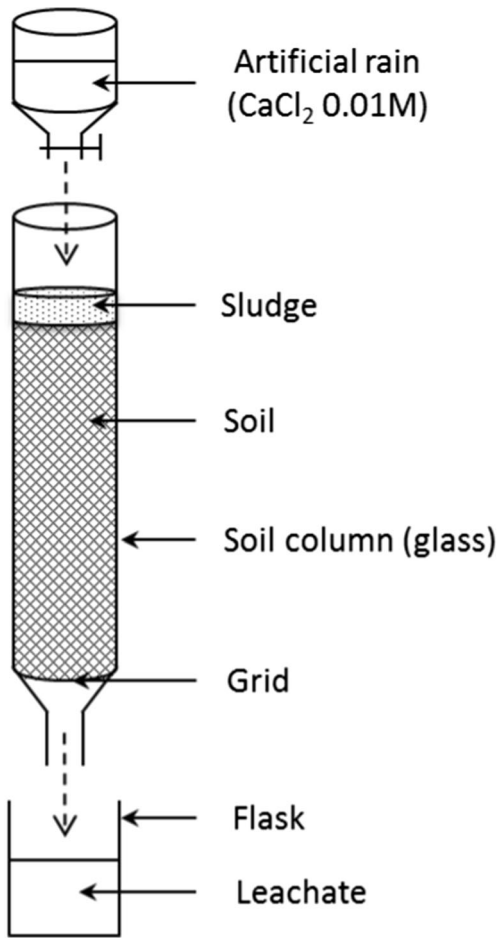


Fig. 2 Scheme of soil column device

Ecotoxicity tests and ecotoxicological risk assessment

Environmental risk assessment was estimated using risk quotient values of acute and chronic effects calculated for each pharmaceutical compound. Risk assessment studies establish different risk levels: low risk (RQs values from 0.01 to 0.1), medium risk (RQs values between 0.1 and 1), and high risk (RQs values higher than 1).

The RQs was defined as the ratio between the predicted environmental concentration in soil (PEC_{soil}) and the predicted no-effect concentration ($PNEC_{soil}$). The $PNEC_{soil}$ values used for calculations were obtained from literature (supplementary material 2 of Martín et al. (2015)). The PEC_{soil} values

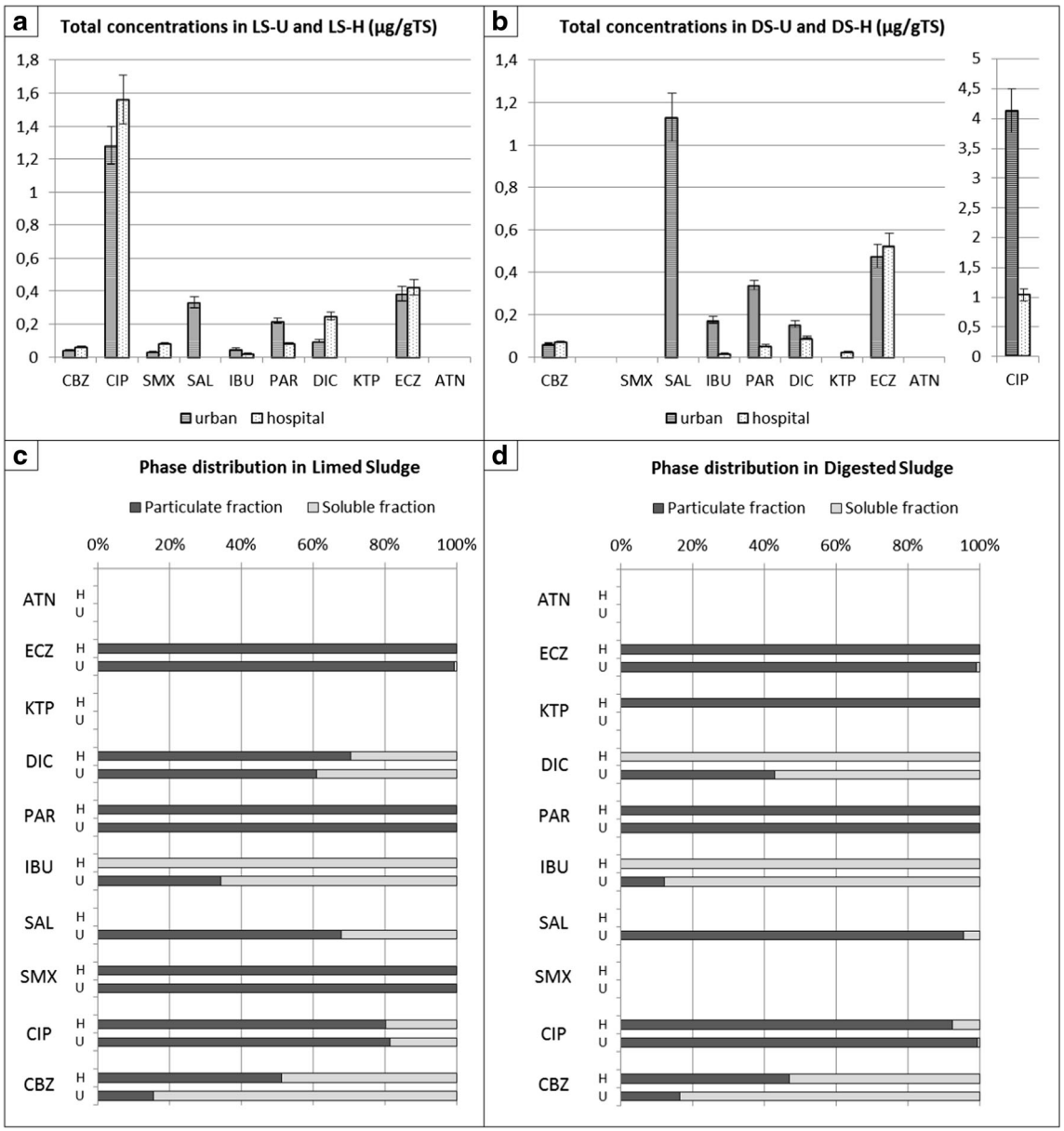


Fig. 3 Pharmaceutical compound concentrations in limed (**a**) and digested (**b**) sludge of urban and hospital origins and their phase distribution in limed (**c**) and digested (**d**) sludge of urban (*U*) and hospital (*H*) origins

were an estimation of the expected concentrations of the pharmaceutical compound in topsoil to 1 year after one sludge dose application. As recommended by the European Commission in the Technical Guidance Document on Risk Assessment EUR 20418 EN/2 (EC-TGD 2003), the PEC_{soil} values were calculated according to the following equation:

$$PEC_{soil} = C_{sludge} \times APPL_{sludge}/DEPTH_{soil} \times RHO_{soil}$$

where C_{sludge} is the concentration of the pharmaceutical compound in different sludge samples (urban and hospital limed and digested sludge) expressed as µg/kg dm, $APPL_{sludge}$ is the dry sludge application rate (0.5 kg/m^2 year for agricultural soils), $DEPTH_{soil}$ is the mixing depth of soil (0.20 m for agricultural soils), and RHO_{soil} is the bulk density of wet soil (1700 kg/m^3 for agricultural soils). PEC_{soil} values calculated are shown in the "Electronic supplementary material" (Table S2).

At the end of the leaching experiment, two kinds of tests were conducted to measure the potential acute toxicity of each leachate sample: a test with *Daphnia magna* from a kit "Daphtoxkit F^{TM} *Magna*" (OECD 2004a, 2004b; ISO 6341–1996) and a Microtox® test with marine bacteria *Vibrio fischeri* (NF EN ISO 11348–3 2008). The toxic effluent concentration was expressed as the percentage effect corresponding to the number of immobilized *Daphnia* and determined with the graphical interpolation sheet provided in the Daphtoxkit F^{TM} *Magna*.

Table 4 Desorption constant values for stabilized sludge

$K_{desorption}$	Limed sludge		Digested sludge	
	LS-U	LS-H	DS-U	DS-H
Carbamazepine	$742.6.10^{-3}$	0	0	0
Ciprofloxacin	$76.7.10^{-3}$	$341.6.10^{-3}$	$1.59.10^{-3}$	$12.7.10^{-3}$
Sulfamethoxazole	0	0	N.d.	N.d.
Salicylic acid	$516.6.10^{-3}$	N.d.	$7.32.10^{-3}$	N.d.
Ibuprofen	$240.8.10^{-3}$	$193.3.10^{-3}$	$158.5.10^{-3}$	107.10^{-3}
Paracetamol	0	0	0	0
Diclofenac	$48.2.10^{-3}$	$11.5.10^{-3}$	$83.9.10^{-3}$	$78.3.10^{-3}$
Ketoprofen	N.d.	N.d.	N.d.	0
Econazole	$5.28.10^{-3}$	$4.14.10^{-3}$	0	0
Atenolol	N.d.	N.d.	N.d.	N.d.

N.d. not detected in the total sludge

Results and discussion

Pharmaceutical concentration and phase distribution in the different sludge samples

Pharmaceutical concentration in the different sludge samples

The total concentrations of ten pharmaceuticals in digested and limed sludge were measured for the two sludge samples: sludge obtained from urban effluent and sludge obtained from hospital effluent (Fig. 3a, b). The results showed discrepancies in both the concentration and distribution of micropollutants with respect to the stabilization process used and the origin of the sludge. For most of the pharmaceutical compounds analyzed, the concentrations were very low (<0.6 μg/gTS) and comparable to those in the literature review (Lindberg et al. 2006; Soulier et al. 2011). The antibiotic ciprofloxacin exhibited the highest concentration in sludge, whatever the treatment and origin of the sludge. In limed and digested sludges, the highest concentration was obtained for ciprofloxacin: 1.6 μg/gTS for limed sludge and 4 μg/gTS for digested sludge. This result is in accordance with previous results confirming that this fluoroquinolone has a high affinity for sludge (Golet et al. 2003; Jia et al. 2012; Chen et al. 2013). As anaerobic digestion is a biological process, some biodegradation of pharmaceutical compounds could be expected. Regarding the results obtained on Fig. 3b, SMX was the only compound completely removed. This result is in accordance with other experiments on anaerobic digestion conducted at lab, pilot, or full scale (Stasinakis 2012; Narumiya et al. 2013). Nevertheless, the removal could be due to complete mineralization or to partial biodegradation. Except SMX, some other compounds like SAL and CIP seemed to concentrate in the digested sludge. This result could be explained by the absence of biodegradation coupled with TS decrease during anaerobic digestion, resulting in an increase of some pharmaceutical compounds' concentration at the end of anaerobic digestion.

The concentrations of the studied pharmaceutical compounds were not radically different in the urban or hospital sludge samples, except for salicylic acid and, to a lesser degree, paracetamol. Salicylic acid was not detected in the hospital sludge, whereas all the other compounds were detected in both the urban and hospital sludge. Paracetamol concentrations were always higher in urban sludge compared with hospital sludge.

Phase distribution (solid–liquid repartition) of pharmaceutical compounds in the different sludge samples

The phase distribution of the different compounds is presented in Fig. 3c, d. In the context of sludge valorization by landspreading, the soluble fraction is of interest as it could be assimilated to the phase containing the more available

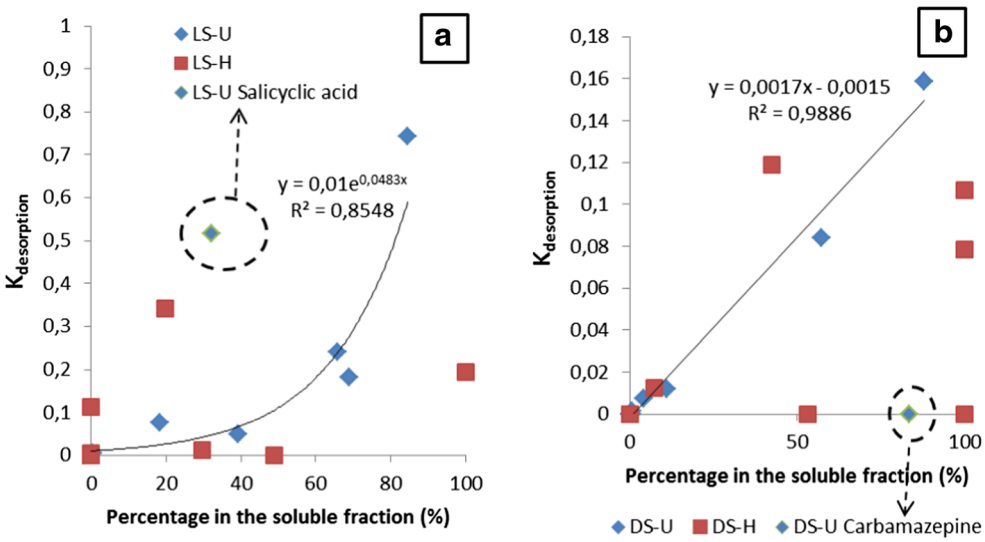


Fig. 4 Relation between $K_{desorption}$ and the percentage in the soluble phase for the different pharmaceutical compounds in limed sludge (LS, **a**) and digested sludge (DS, **b**)

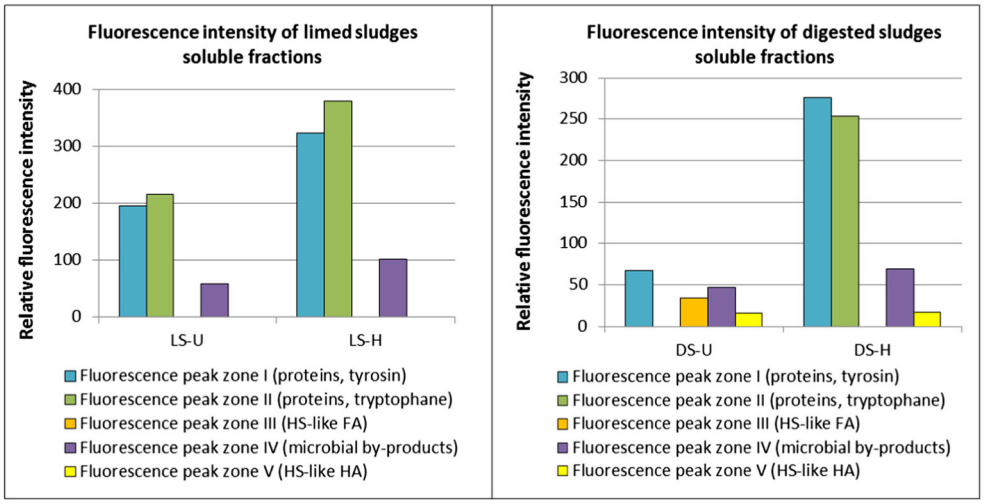


Fig. 5 Fluorescence of the soluble phases of limed and digested sludge. Comparison between urban and hospital origins

micropollutant fraction. Results of phase distribution of pharmaceutical compounds showed that distribution differed from one compound to another and varied according to the type of treatment. Liming and anaerobic digestion had no impact on the distribution of the following compounds: carbamazepine (mainly present in the soluble fraction), ciprofloxacin (mainly present in the particulate fraction), ibuprofen (mainly present in the soluble fraction), paracetamol (mainly present in the particulate fraction), and econazole (mainly present in the particulate fraction). Despite the stabilization treatment, salicylic acid was present mainly in the particulate fraction of the urban sludge but was not detected in the hospital sludge. Ketoprofen and atenolol were not detected in limed and digested sludge samples nor were they present in thickened sludge before treatment. Martín et al. (2015) also investigated the fate of pharmaceutical compounds in different sludge samples, among them the anaerobically digested sludge sample: ketoprofen and atenolol were not detected in the digested sludge. Sulfamethoxazole was detected in the particulate fraction of limed sludge but not in the digested sludge (which does not mean biodegradation). In digested sludge, non-extractable residues could be involved in the apparent removal due to a decrease in extraction efficiency.

Determination of pharmaceutical compound mobility before landspreading

$K_{desorption}$ values for the different pharmaceutical compounds in urban and hospital digested and limed sludge

Limed and digested sludge were submitted to desorption batch tests as described in "Batch desorption tests on sludge". The desorption constant values are given in Table 4. The results of this study confirmed that the estimation of pharmaceutical compounds' desorption ability based on either preliminary experimental sludge–water partition coefficients, octanol–water coefficients (K_{OW}), or acid–base constants (pKa) was not reliable. It was not possible to determine this constant for atenolol as this compound was not detected in the total sludge (probably due to a decrease of extraction efficiency). For some other compounds, the value was 0: like for CBZ in LS-H, DS-U, and DS-H, SUL in LS-U and LS-H, PAR in all the sludge samples, and ECZ in DS-U and DS-H, which means that these compounds were susceptible to remain in the particulate phase (as demonstrated by Fig. 3c, d). When $K_{desorption}$ was measurable, there was an important discrepancy from one compound to another from 1.59 to 742.6 $\times 10^{-3}$.

Table 5 Various final leachates characteristics

Parameters	Units	Lix-T-soil	Lix-LS-U	Lix-DS-U	Lix-LS-H	Lix-DS-H
pH		7.14±0.01	7.27±0.01	6.99±0.01	7.45±0.01	7.11±0.01
χ	mS/cm	2.28±0.01	2.63±0.01	2.59±0.01	2.35±0.01	2.38±0.01
DOC	mgC/L	65.1±0.13	234.2±0.04	206.5±0.02	79.9±0.12	143.7±0.01
Abs 254 nm		2.17±0.002	2.84±0.003	4.25±0.001	2.33±0.002	3.04±0.002
SUVA index	L cm/gC	37.7±3.1	12.1±0.02	20.6±0.01	29.1±0.01	21.2±0.01
Percentage of aromaticity		28±1.6	12±0.33	17±0.06	23±0.26	18±0.37

Based on the previous part, we determined that some compounds were found predominantly in the soluble fraction of the sludge, whereas others were found in the particulate fraction. A greater desorption capacity could thus be expected for compounds found in the soluble part of the sludge. To demonstrate this, $K_{desorption}$ values as a function of the percentage of pharmaceutical compound in the soluble phase were plotted on Fig. 4. Limed and digested sludge were considered separately.

Effects of treatment process (liming versus digestion) and sludge origin (urban versus hospital)

Urban sludge For limed sludge, $K_{desorption}$ can be exponentially correlated to the percentage of pharmaceutical compounds in the soluble phase of the corresponding sludge, except for salicylic acid. This compound exhibited a relatively high $K_{desorption}$ value, whereas its percentage in the soluble sludge phase was very low. Despite the quite high log K_{OW} value (2.26), the pKa of this molecule was the lowest of all the compounds considered in this study. Therefore, salicylic acid is negatively charged at the limed sludge pH (around 12), which could explain its greater ability to be desorbed from the sludge.

For digested sludge, $K_{desorption}$ can be linearly correlated to the soluble percentage of the corresponding compounds in the soluble phase, except for one molecule: carbamazepine. Carbamazepine was found mainly in the soluble phase (Fig. 3), but it was not desorbed ($K_{desorption}$ of CBZ was equal to 0 supposing that CBZ was not desorbable at all). It meant that if CBZ was associated with dissolved organic matter, and this association could kept intact after filtration at 0.45 μm. This observation could be explained by the very strong bonds between carbamazepine and the compounds in the soluble sludge phase. Carbamazepine exhibited a log K_{OW} of 2.25 which was supposed to lead to low affinity with the aqueous phase. Nevertheless, the occurrence of this molecule in water is clearly demonstrated in the literature. In the environment, carbamazepine should be present in its neutral form; the aqueous solubility and relatively low log K_{OW} are indicative that it is not expected to be extensively sorbed onto particulate matrix. (Calisto and Esteves, 2012).

The main discrepancy between limed and digested urban sludge soluble-phase biochemical composition lied on the fact that digested sludge was mainly composed of humic-like substances while limed sludge was equally composed of proteins and humic-like substances (see "Electronic supplementary material" (Table S3)). One explanation could lie on the fact that carbamazepine has a high pKa value, being thus positively charged at basic pH values, which was the case for digested sludge. The main probable hypothesis could be that the positive charge of the molecule strongly interacts with the negatively charged extracellular polymeric substances in the sludge and especially humic-like substances.**Hospital sludge** For either the limed or digested sludge from the hospital effluent, it was not possible to mathematically correlate $K_{desorption}$ and the percentage of the corresponding pharmaceutical compound in the soluble phase. One might hypothesize that the different results observed between treatment and origin of the sludge could be due to possible effects of the dissolved organic matter in the proportion between soluble/solid fractions. This aspect was investigated hereafter.

Fig. 6 Identification of specific fluorescence zones (**a**) for the different leachates: Lix-T-soil (**b**), Lix-LS-U (**c**), Lix-DS-U (**d**), Lix-LS-H (**e**) and Lix-DS-H (**f**) and corresponding relative fluorescence intensity (**g**)

Investigations into sludge composition were conducted in an attempt to explain the significant differences observed between urban and hospital sludge behavior to pharmaceutical compounds. Protonic exchange capacity, pKa determination and biochemical composition (proteins, polysaccharides, humic-like substances) were measured both in urban and hospital sludge and compared for the same type of treatment (liming, digestion) (data not shown).

Conventional EPS characterization (i.e., by colorimetric techniques) did not reveal any differences in the results between urban and hospital sludge for the same type of sludge stabilization treatment. Thus, 3D spectroscopic fluorescence was used to try to identify possible differences in the biochemical sludge composition (Fig. 5).

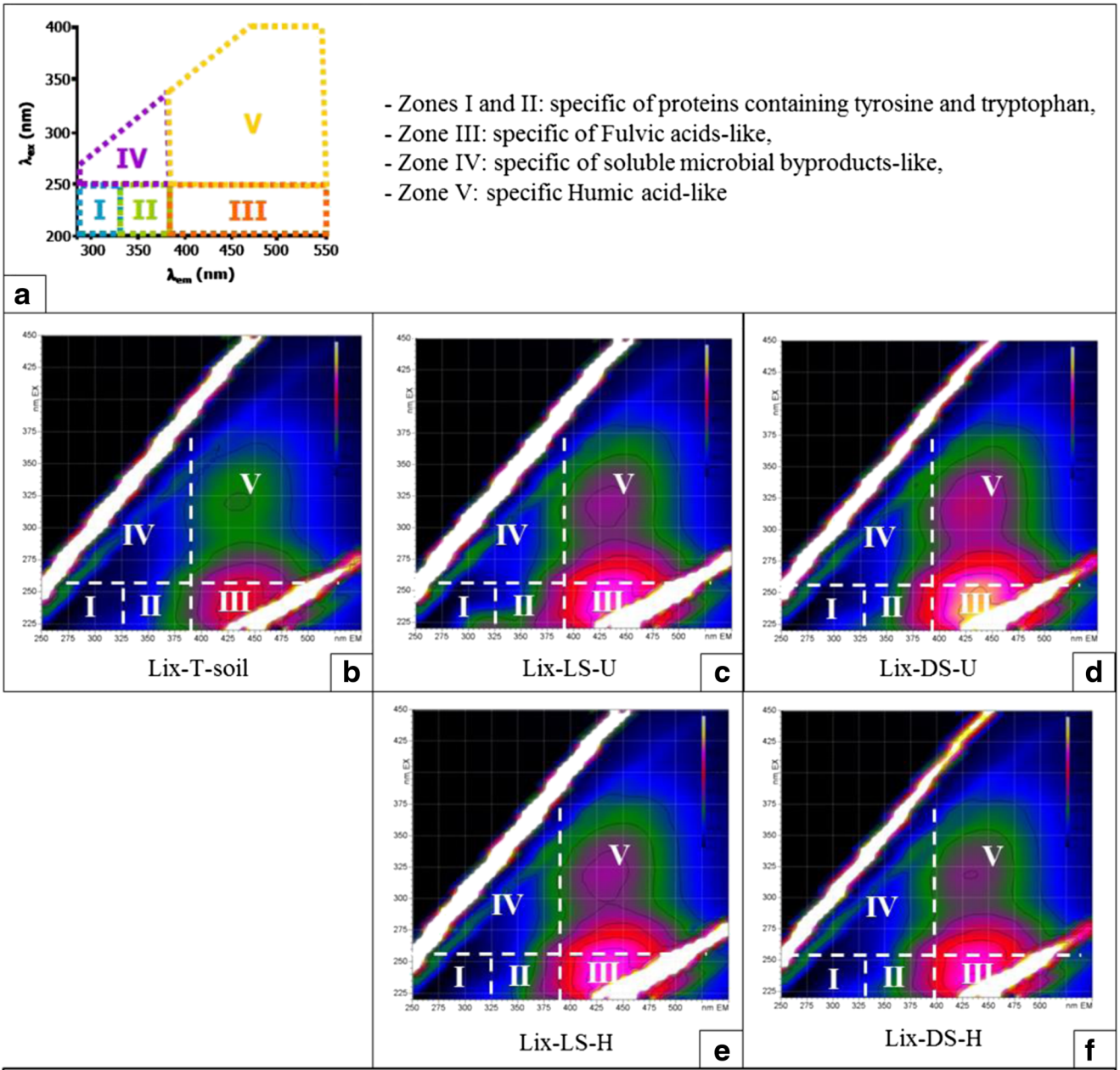
3D spectroscopic fluorescence clearly demonstrated that relative fluorescence intensity in the protein tyrosine and protein tryptophan zones could explain the observed differences in the interaction with pharmaceutical compounds. In fact, the nature of the influent (domestic or hospital origin) impacted the organic matter composition. For example, the increase of fluorescence intensity in zones I and II of hospital effluents could be linked to the presence of hemoglobin (Louvet et al. 2013; Wu et al. 2014).

Soil columns

Leachate characteristics

The physicochemical properties of the leachate samples were measured periodically, and at the end of the experiment, the leachates obtained were mixed. The measurements obtained on the final mixture are presented in Table 5.

As previously demonstrated, sewage sludge amendments weakly modified the physicochemical properties of the leachates (Usman et al. 2004; Hattab et al. 2014). A slight increase in pH is observed for the leachate resulting from limed sludge landspreading. A slight increase in conductivity was highlighted for the leachates of the soil amended with urban sludge. The landspreading of sludge led to an increase of DOC concentrations in amended soils. This increase was 1.3 to 3.5 times greater compared to the unamended soil. The fluorescence spectra of the different leachate samples and the relative fluorescence intensity of the specific zones are presented in


a
- Zones I and II: specific of proteins containing tyrosine and tryptophan,
- Zone III: specific of Fulvic acids-like,
- Zone IV: specific of soluble microbial byproducts-like,
- Zone V: specific Humic acid-like
Lix-T-soil
b
Lix-LS-U
c
Lix-DS-U
d
Lix-LS-H
e
Lix-DS-H
f

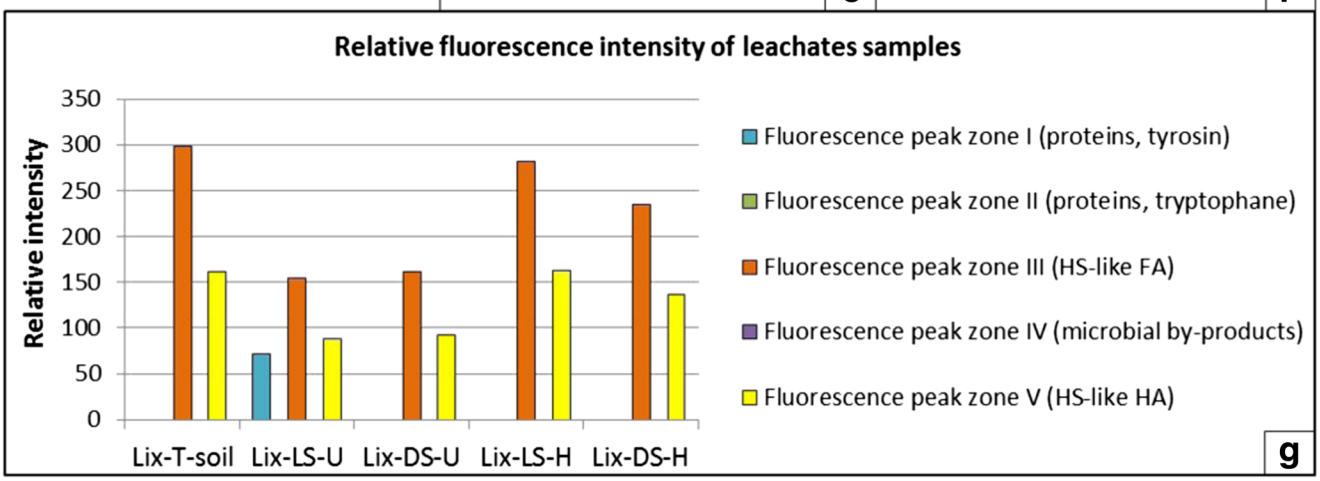

Relative fluorescence intensity of leachates samples
Relative intensity
Lix-T-soil
Lix-LS-U
Lix-DS-U
Lix-LS-H
Lix-DS-H
Fluorescence peak zone I (proteins, tyrosin)
Fluorescence peak zone II (proteins, tryptophane)
Fluorescence peak zone III (HS-like FA)
Fluorescence peak zone IV (microbial by-products)
Fluorescence peak zone V (HS-like HA)
g

Fig. 6. For all leachates, fluorescence peaks are detected in zone III (HS-like FA) and zone V (HS-like HA), but no peaks appeared in the protein zones (II and III), except for the leachate obtained from the limed hospital sludge. The SUVA index can be used to characterize the aromaticity of DOM (Fellman et al. 2008). The value of SUVA index ranges from 12.1 to 37.7 L/gC/cm, corresponding to an aromaticity percentage between 12 and 28 %. For amended soils, the decrease in the aromaticity percentage of the leachates may highlight an input of low aromatic molecules like polysaccharides, amino acids, and low molecular weight organic acids to the soil. These results could be explained by the biochemical composition of landspread sludge (data shown in the "Electronic supplementary material" (Table S3). Indeed landspread sludge mainly consisted in "humic-like substances" and polysaccharides.

Therefore, the amendments of urban and hospital sludge modified not only the quantity but also the nature of dissolved organic matter in soil (Usman et al. 2004). This may play a role in the fate of some pharmaceutical compounds in the soil. In fact, hydrophobic and hydrophilic properties of natural organic matter facilitate intermolecular interactions, such as H-bonding or ion exchange reactions, with antibiotics, for example (Aristilde and Sposito 2013).

Pharmaceutical compound concentrations in leachates after soil leaching tests

Pharmaceutical compound concentrations in leachates are given in Table 6. The pharmaceutical compounds found in amended soil leachates were IBU>SAL- DIC >>>PAR. Moreover, this distribution depends on the origin and the treatment of the influent. Ibuprofen and diclofenac were present at quantifiable levels (the limits of quantification of the methods are given in Table S1) only in the soil leachates amended by urban sludge (Lix-LS-U and Lix-DS-U), whereas paracetamol and salicylic acid were found only in the soil leachate amended by hospital sludge (LS-H and DS-H). The comparison with the desorption constant obtained from the sludge only (Table 4) demonstrated that $K_{desorption}$ was equal to zero for paracetamol in all sludge samples. This discrepancy showed that not only sludge composition but also soil composition greatly impacted pharmaceutical compound mobility during landspreading.

Ibuprofen was detected in Lix-LS-U and Lix-DS-U as well as in the control sample. In both cases, the mass balance indicated that ibuprofen was totally leached from the soil. This means that ibuprofen showed high mobility in soil, which was also linked with its high desorption coefficient value (Table 4). Several studies have demonstrated the low sorption affinity of ibuprofen in soils (Xu et al. 2009; Lin and Gan 2011; Estevez et al. 2014). These authors found that ibuprofen had the potential to move downward with percolating water, thereby resulting in negligible adsorption onto soil for this compound. This fact suggests high mobility in soil and a potential risk of groundwater contamination. The same environmental fate has been observed by different authors (Haguenoer 2010; Martín et al. 2010; Grossberger et al. 2014) for other anti-inflammatory drugs such as diclofenac and salicylic acid. In our case, up to 20 % of diclofenac and salicylic acid was leached during the leaching test.

Although CBZ, CIP, SMX, KTP, ECZ, and ATN were detected in some sludge samples before landspreading, they were not present in any leachate. This reflects either a retention by and/or transformation in the soil, thus resulting in very low mobility. Salvia et al. (2014) showed that most pharmaceutical compounds are trapped in the surface layers of the soil column, except for sulfonamide which could migrate deeply. In fact, Tamtam et al. (2011) showed that sulfamethoxazole could be associated to clay–iron complexes in the soil, suggesting immobilization mechanisms by electrostatic adsorption. Ciprofloxacin was not detected in leachates obtained from the leaching of the different sludge samples in the soil

Table 6 Concentrations of pharmaceutical compounds in the composite leachate samples after the simulated rainfall (1020 mm for a year corresponding to 2358 mL during 30 days)

Compound	Concentration (μg/L)			
	LS-U	LS-H	DS-U	DS-H
CBZ	n.d.	n.d.	n.d.	n.d.
CIP	n.d.	n.d.	n.d.	n.d.
SMX	n.d.	n.d.	n.d.	n.d.
SAL	n.d.	0.045±0.009	n.d.	0.104±0.014
IBU	0.055±0.0045	n.d.	0.515±0.004	n.d.
PAR	n.d.	$2.5.10^{-4}$±0.0006	n.d.	n.d.
DIC	n.d.	n.d.	0.102±0.008	n.d.
KTP	n.d.	n.d.	n.d.	n.d.
ECZ	n.d.	n.d.	n.d.	n.d.
ATN	n.d.	n.d.	n.d.	n.d.

n.d. not detected in the total sludge

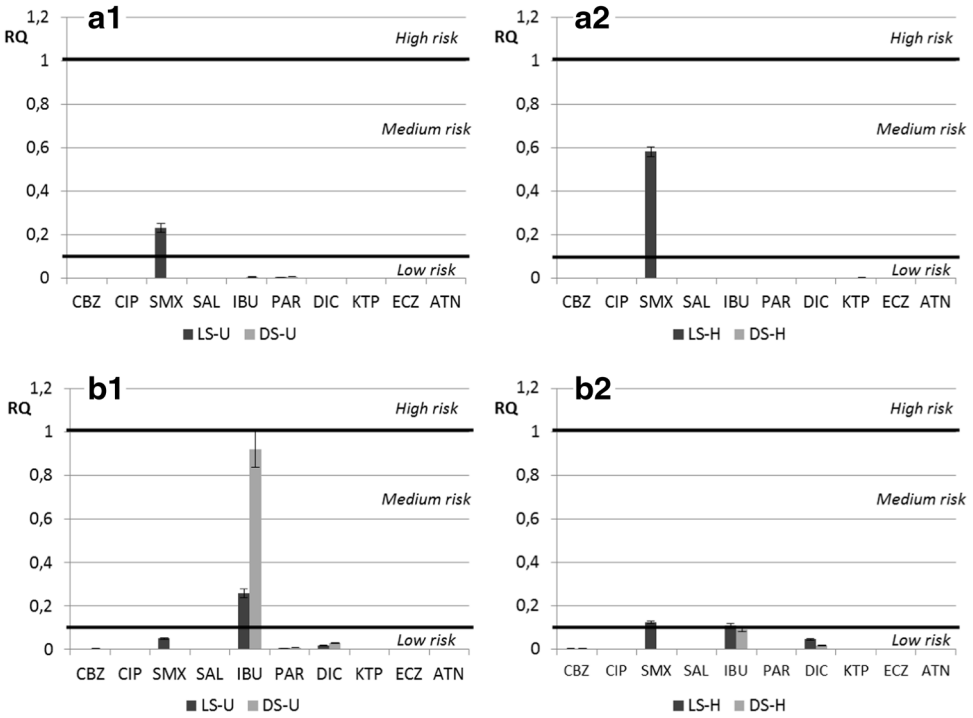


Fig. 7 Risk quotients (RQ) calculated for acute (**a**) and chronic (**b**) effects of studied pharmaceutical compounds in soils amended with limed (LS) and digested (DS) sludge from urban (*1*) and hospital (*2*) origins

columns. Golet et al. (2003) conducted field experiments on sludge application to agricultural land. They confirmed the long-term persistence of trace amounts of fluoroquinolones in sludge-treated soils and indicated the limited mobility of these compounds in the subsoil. In fact, fluoroquinolones are considered to be strongly adsorbed, which means that they possess limited mobility in soils. Leal et al. (2013) showed that the most important sorption mechanism of fluoroquinolones in soil is cationic exchange. According to this mechanism, they explain that the sorption of ciprofloxacin is related to the presence of clay minerals such as kaolinite and/or smectite. Moreover, Aristilde and Sposito (2013) showed that natural organic matter (especially humic substances) is also involved in the retention of antibiotics by particles in soils. Consequently, the environmental fate of ciprofloxacin is directly affected by soil CEC (CEC being linked to soil clay and organic matter contents). In our case, the literature revealed that the antibiotics are readily and strongly sorbed to soils with acidic pH and high clay content (Awad et al. 2014). Oppel et al. (2004) also investigated the leaching behavior in a soil-testing-system with laboratory columns; they did not detect carbamazepine in the leachates. This compound was recalcitrant and was not significantly degraded during sewage treatment, suggesting that it may accumulate in soil after landspreading (Grossberger et al. 2014).

Risk assessment and ecotoxicity measurement

The harmful ecological impacts of the pharmaceutical compounds on soil amended need to be determined. Risk assessment was estimated with RQs of acute and chronic effects calculated for each pharmaceutical compound in soil amended with the different sludge samples (Fig. 7). The RQs values of acute and chronic effects were lower than 1 for each studied pharmaceutical compound and for each sludge sample. Regarding acute toxicity results, sulfamethoxazole showed medium risk with RQs values of 0.23 and 0.58 in soil amended with limed sludge from urban and hospital origins, respectively. Regarding chronic toxicity results, ibuprofen showed medium risk in soil amended with urban limed and digested sludge (RQs values of 0.26 and 0.92, respectively). These results suggest that only ibuprofen and sulfamethoxazole may induce a

Table 7 Results of ecotoxicological test (*D. magna*) on the different leachates

	Percentage effect of toxicity				
	Lix-T-soil	Lix-LS-U	Lix-DS-U	Lix-LS-H	Lix-DS-H
% Eeffect (24 h)	No effect	No effect	No effect	No effect	65 %
% effect (48 h)	No effect	No effect	No effect	No effect	32 %

potential ecotoxicological risk for soil organisms and for groundwater when sludge is applied to soils. Moreover, as mentioned by Li et al. (2015) and Chen and Zou (2014), this approach did not take into account the mixed toxicity of several pharmaceutical compounds and/or other coexisting pollutants, which can be more significant than individual effects. Consequently, the ecotoxicity of each leachate sample was measured by two types of tests: Microtox® and *D. magna*. Regarding the Microtox® test, no extinction of *Vibrio fisheri* bioluminescence was measured after exposure to each non-diluted leachate sample, so no toxic effect could be determined. As for the test with *D. magna*, the results are presented in Table 7. After 24 and 48 h of exposure, no toxic effect was measured for Lix-T-soil, Lix-LS-U, Lix-DS-U, and Lix-LS-H. However, a toxic effect was detected in soil leachate after digested hospital sludge landspreading (Lix-DS-H). As this leachate contained none of the molecules identified as potentially toxic (SMX or IBU), it could be suggested that other compounds (not investigated in this study) such as surfactants or disinfectants (Orias and Perrodin 2013), or compounds created or not degraded during anaerobic digestion, might be detrimental to the quality of the leachate.

Conclusion

The objective of this study was to evaluate the environmental fate of several pharmaceutical compounds (antiepileptic drugs, antibiotics, anti-inflammatory and anti-fungal drugs, beta-blockers) in sludge landspreading. The screening of these compounds in sludge, soil, and leachates (soil solution) demonstrated the leaching potential according to (1) the origin of influent (urban or hospital), (2) the sludge stabilization process, and (3) the pharmaceutical compound. In all cases, only four anti-inflammatory drugs (ibuprofen, salicylic acid, diclofenac, and paracetamol) were leached (from to 20 up 100 % of total content) from all amended soils and may pose an environmental risk through groundwater contamination.

Regarding the desorption potential of pharmaceutical compounds in the different stabilized sludge samples, ibuprofen showed the highest desorption coefficient despite the type of sludge. For urban limed and digested sludge, the desorption potential was correlated to the percentage of pharmaceutical compounds in the soluble fraction, except for salicylic acid in limed sludge and carbamazepine in digested sludge. Their behavior could be explained by their physicochemical properties and the evolution of sludge composition during the treatment. Indeed despite the high log K_{OW} value of salicylic acid, its pKa was the lowest of all the compounds studied. Consequently, salicylic acid is negatively charged at the pH of limed sludge (around pH 12), which could explain its higher ability to be desorbed from the sludge. For carbamazepine, which was mainly in the soluble fraction and was only slightly desorbed, the main probable hypothesis is that the positively charged molecule strongly interacts with the negatively charged extracellular polymeric substances in the sludge.

Ecotoxicological tests performed on leachates obtained after sludge landspreading showed no effect, except for the digested sludge from hospital effluents (DS-H). The environmental risk assessment revealed that following the application of stabilized sludge (limed and digested) onto soil, no acute or chronic ecotoxicological risk on leachates could be expected. Nevertheless, sulfamethoxazole and ibuprofen were the only two compounds with a medium risk. Sulfamethoxazole was only present in limed sludge because it was dissipated during anaerobic digestion, and ibuprofen was the compound with the highest desorption capacity for each type of sludge sample studied. Therefore, the presence of ibuprofen should be monitored in sludge prior to land application.

To the best of our knowledge, previous experiments on the fate of pharmaceutical compounds in the soil column were achieved by direct spiking of micropollutants in the soil column. In this study, we clearly demonstrated that the fate of micropollutants is dependent on soil characteristics but is also obviously dependent on the interactions of the soil with the landspread sludge, which itself closely depends on the sludge stabilization process used before landspreading.

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References

AFNOR, NF T90-105 (1997) NF T90-105 - Qualité de l'eau - Dosage des matières en suspension - Méthode par centrifugation. Association Française de Normalisation, Paris. http://www.boutique.afnor.org/norme/nf-t90-105-2/qualite-de-l-eau-dosage-des-matieres-en-suspension-methode-par-centrifugation/article/639936/fa041874. Accessed 11 Jun 2014

Aristilde L, Sposito G (2013) Complexes of the antimicrobial ciprofloxacin with soil, peat, and aquatic humic substances. Environ Toxicol Chem 32:1467–1478. doi:10.1002/etc.2214

Awad YM, Kim S-C, El-Azeem SAMA et al (2014) Veterinary antibiotics contamination in water, sediment, and soil near a swine manure composting facility. Environ Earth Sci 71:1433–1440. doi:10.1007/s12665-013-2548-z

Barret M, DELGADILLO-MIRQUEZ L, TRABLY E et al (2012) Anaerobic removal of trace organic contaminants in sewage sludge: 15 years of experience. Pedosphere 22:508–517

Berthod L, Roberts G, Withley DC et al (2014) A solid-phase extraction method for rapidly determining the adsorption coefficient of pharmaceuticals in sewage sludge. Water Res 67:292–298

Calisto V, Esteves VI (2012) Adsorption of the antiepileptic carbamazepine onto agricultural soils. J Environ Monit 14(6):1597–1603

Chen K, Zhou JL (2014) Occurence and behavior of antibiotics in water and sediments from the Huangpu River, Shanghai, China. Chemosphere 95:604–612

Chen Y, Yu G, Cao Q et al (2013) Occurrence and environmental implications of pharmaceuticals in Chinese municipal sewage sludge. Chemosphere 93:1765–1772. doi:10.1016/j.chemosphere.2013.06.007

Clarke BO, Smith SR (2011) Review of "emerging" organic contaminants in biosolids and assessment of international research priorities for the agricultural use of biosolids. Environ Int 37:226–247. doi:10.1016/j.envint.2010.06.004

Corada-Fernández C, Jiménez-Martínez J, Candela L et al (2014) Occurrence and spatial distribution of emerging contaminants in the unsaturated zone. Case study: Guadalete river basin (Cadiz, Spain). Chemosphere. doi:10.1016/j.chemosphere.2014.04.098

Daughton CG, Ternes TA (1999) Pharmaceuticals and personal care products in the environment: agents of subtle change? Environ Health Perspect 107:907–938

Deblonde T, Cossu-Leguille C, Hartemann P (2011) Emerging pollutants in wastewater: a review of the literature. Int J Hyg Environ Health 214:442–448. doi:10.1016/j.ijheh.2011.08.002

Drillia P, Dokianakis SN, Fountoulakis MS et al (2005) On the occasional biodegradation of pharmaceuticals in the activated sludge process: the example of the antibiotic sulfamethoxazole. J Hazard Mater 122:259–265. doi:10.1016/j.jhazmat.2005.03.009

EC-TGD (2003) Technical guidance document on risk assessment, part II. EUR 20418 EN/2, European Commision, Joint Research Centre

Ellis JB (2006) Pharmaceutical and personal care products (PPCPs) in urban receiving waters. Environ Pollut 144:184–189. doi:10.1016/j.envpol.2005.12.018

Eskicioglu C, Kennedy KJ, Droste RL (2006) Characterization of soluble organic matter of waste activated sludge before and after thermal pretreatment. Water Res 40:3725–3736. doi:10.1016/j.watres.2006.08.017

Estevez E, Hernandez-Moreno JM, Fernandez-Vera JR, Palacios-Diaz MP (2014) Ibuprofen adsorption in four agricultural volcanic soils. Sci Total Environ 468–469:406–414. doi:10.1016/j.scitotenv.2013.07.068

FAO (2006) World reference base for soil researches. World Soil Resour. Rep. 103:

Fellman JB, D'Amore DV, Hood E, Boone RD (2008) Fluorescence characteristics and biodegradability of dissolved organic matter in forest and wetland soils from coastal temperate watersheds in southeast Alaska. Biogeochemistry 88:169–184. doi:10.1007/s10533-008-9203-x4

Fent K, Weston AA, Caminada D (2006) Ecotoxicology of human pharmaceuticals. Aquat Toxicol 76:122–159. doi:10.1016/j.aquatox.2005.09.009

Golet EM, Xifra I, Siegrist H et al (2003) Environmental exposure assessment of fluoroquinolone antibacterial agents from sewage to soil. Environ Sci Technol 37:3243–3249

Grossberger A, Hadar Y, Borch T, Chefetz B (2014) Biodegradability of pharmaceutical compounds in agricultural soils irrigated with treated wastewater. Environ Pollut 185:168–177. doi:10.1016/j.envpol.2013.10.038

Haberhauer G, Temmel B, Gerzabek MH (2002) Influence of dissolved humic substances on the leaching of MCPA in a soil column experiment. Chemosphere 46:495–499. doi:10.1016/S0045-6535(01)00194-1

Haguenoer J-M (2010) Do pharmaceutical waste and drug residue pose a risk to public health? Sante Publique (Bucur) 22:325–342

Hattab N, Soubrand M, Guégan R et al (2014) Effect of organic amendments on the mobility of trace elements in phytoremediated technosoils: role of the humic substances. Environ Sci Pollut Res 21:10470–10480

ISO 6341:1996 : Qualité de l'eau - Détermination de l'inhibition de la mobilité de *Daphnia magna* Straus (Cladocera, Crustacea) - Essai de toxicité aiguë

Jelic A, Gros M, Ginebreda A et al (2011) Occurrence, partition and removal of pharmaceuticals in sewage water and sludge during wastewater treatment. Water Res 45:1165–1176. doi:10.1016/j.watres.2010.11.010

Jia A, Wan Y, Xiao Y, Hu J (2012) Occurrence and fate of quinolone and fluoroquinolone antibiotics in a municipal sewage treatment plant. Water Res 46:387–394. doi:10.1016/j.watres.2011.10.055

Kay P, Blackwell PA, Boxall ABA (2005) Column studies to investigate the fate of veterinary antibiotics in clay soils following slurry application to agricultural land. Chemosphere 60:497–507. doi:10.1016/j.chemosphere.2005.01.028

Kelessidis A, Stasinakis AS (2012) Comparative study of the methods used for treatment and final disposal of sewage sludge in European countries. Waste Manag 32:1186–1195. doi:10.1016/j.wasman.2012.01.012

Kümmerer K (2009) The presence of pharmaceuticals in the environment due to human use—present knowledge and future challenges. J Environ Manag 90:2354–2366. doi:10.1016/j.jenvman.2009.01.023

Laurent J, Casellas M, Pons MN, Dagot C (2009) Flocs surface functionality assessment of sonicated activated sludge in relation with physico-chemical properties. Ultrason Sonochem 16:488–494. doi:10.1016/j.ultsonch.2008.12.012

Leal RMP, Alleoni LRF, Tornisielo VL, Regitano JB (2013) Sorption of fluoroquinolones and sulfonamides in 13 Brazilian soils. Chemosphere 92:979–985. doi:10.1016/j.chemosphere.2013.03.018

Li C, Chen J, Wang J et al (2015) Occurrence of antibiotics in soils and manures from greenhouse vegetable production bases of Beijing, China and an associated risk assessment. Sci Total Environ 521–522:101–107. doi:10.1016/j.scitotenv.2015.03.070

Limousin G, Gaudet JP, Charlet L et al (2007) Sorption isotherms: a review on physical bases, modeling and measurement. Appl Geochem 22(2):249–275

Lin K, Gan J (2011) Sorption and degradation of wastewater-associated non-steroidal anti-inflammatory drugs and antibiotics in soils. Chemosphere 83:240–246. doi:10.1016/j.chemosphere.2010.12.083

Lindberg RH, Olofsson U, Rendahl P et al (2006) Behavior of fluoroquinolones and trimethoprim during mechanical, chemical, and active sludge treatment of sewage water and digestion of sludge. Environ Sci Technol 40:1042–1048. doi:10.1021/es0516211

Louvet JN, Homeky B, Casellas M et al (2013) Monitoring of slaughterhouse wastewater biodegradation in a SBR using fluorescence and UV–visible absorbance. Chemosphere 91:648–655. doi:10.1016/j.chemosphere.2013.01.011

Martín J, Santos JL, Aparicio I, Alonso E (2010) Multi-residue method for the analysis of pharmaceutical compounds in sewage sludge, compost and sediments by sonication-assisted extraction and LC determination. J Sep Sci 33:1760–1766. doi:10.1002/jssc.200900873

Martín J, Camacho-Muñoz D, Santos JL et al (2012) Occurrence of pharmaceutical compounds in wastewater and sludge from wastewater treatment plants: removal and ecotoxicological impact of wastewater discharges and sludge disposal. J Hazard Mater 239–240:40–47. doi:10.1016/j.jhazmat.2012.04.068

Martín J, Santos JL, Aparicio I, Alonso E (2015) Pharmaceutically active compounds in sludge stabilization treatments: anaerobic and aerobic digestion, wastewater stabilization ponds and composting. Sci Total Environ 503–504:97–104. doi:10.1016/j.scitotenv.2014.05.089

Narumiya M, Nakada N, Yamashita N, Tanaka H (2013) Phase distribution and removal of pharmaceuticals and personal care products during anaerobic sludge digestion. J Hazard Mater 260:305–312. doi:10.1016/j.jhazmat.2013.05.032

NF EN ISO 11348–3 (2008) Qualité de l'eau - Détermination de l'effet inhibiteur d'échantillons d'eau sur la luminescence de Vibrio fischeri

(Essai de bactéries luminescentes) - Partie 3 : méthode utilisant des bactéries lyophilisées

OECD (2004a) Essai n° 202: Daphnia sp., essai d'immobilisation immédiate, Lignes directrices de l'OCDE pour les essais de produits chimiques, Section 2

OECD (2004b) Essai no. 312: Lixiviation sur des colonnes de sol, Lignes directrices de l'OCDE pour les essais de produits chimiques, Section 3

Oppel J, Broll G, Löffler D et al (2004) Leaching behaviour of pharmaceuticals in soil-testing-systems: a part of an environmental risk assessment for groundwater protection. Sci Total Environ 328: 265–273. doi:10.1016/j.scitotenv.2004.02.004

Orias F, Perrodin Y (2013) Characterisation of the ecotoxicity of hospital effluents: a review. Sci Total Environ 454–455:250–276

Pan B, Huang P, Wu M et al (2012) Physicochemical and sorption properties of thermally-treated sediments with high organic matter content. Bioresour Technol 103:367–373. doi:10.1016/j.biortech.2011.09.054

Peysson W, Vulliet E (2013) Determination of 136 pharmaceuticals and hormones in sewage sludge using quick, easy, cheap, effective, rugged and safe extraction followed by analysis with liquid chromatography–time-of-flight-mass spectrometry. J Chromatogr A 1290:46–61. doi:10.1016/j.chroma.2013.03.057

Pomiès M, Choubert JM, Wisniewski C et al (2013) Modelling of micropollutant removal in biological wastewater treatments: a review. Sci Total Environ 443:733–748

Salvia M-V, Experton J, Geandel C et al (2014) Fate of pharmaceutical compounds and steroid hormones in soil: study of transfer and degradation in soil columns. Environ Sci Pollut Res 21:10525–10535. doi:10.1007/s11356-014-3038-x

Soulier C, Gabet V, Lardy S et al (2011) Zoom on pharmaceutical substances: presence, distribution and behaviour in wastewater treatment plants. Technol Sci Methods 63:77

Stalder T, Barraud O, Casellas M et al (2012) Integron involvement in environmental spread of antibiotic resistance. Antimicrob Resist Chemother 3:119. doi:10.3389/fmicb.2012.00119

Stasinakis AS (2012) Review on the fate of emerging contaminants during sludge anaerobic digestion. Bioresour Technol 121:432–440. doi:10.1016/j.biortech.2012.06.074

Tamtam F, van Oort F, Le Bot B et al (2011) Assessing the fate of antibiotic contaminants in metal contaminated soils four years after cessation of long-term waste water irrigation. Sci Total Environ 409: 540–547. doi:10.1016/j.scitotenv.2010.10.033

Ternes TA, Herrmann N, Bonerz M et al (2004) A rapid method to measure the solid–water distribution coefficient (Kd) for pharmaceuticals and musk fragrances in sewage sludge. Water Res 38: 4075–4084. doi:10.1016/j.watres.2004.07.015

Tolls J (2001) Sorption of veterinary pharmaceuticals in soils: a review. Environ Sci Technol 35:3397–3406

Topp E, Monteiro SC, Beck A et al (2008) Runoff of pharmaceuticals and personal care products following application of biosolids to an agricultural field. Sci Total Environ 396:52–59. doi:10.1016/j.scitotenv.2008.02.011

Usman ARA, Kuzyakov Y, Stahr K (2004) Dynamics of organic C mineralization and the mobile fraction of heavy metals in a calcareous soil incubated with organic wastes. Water Air Soil Pollut 158:401–418. doi:10.1023/B:WATE.0000044864.07418.8f

Vithanage M, Rajapaksha AU, Tang X et al (2014) Sorption and transport of sulfamethazine in agricultural soils amended with invasive-plant-derived biochar. J Environ Manag 141:95–103. doi:10.1016/j.jenvman.2014.02.030

Weishaar JL, Aiken GR, Bergamaschi BA et al (2003) Evaluation of specific ultraviolet absorbance as an indicator of the chemical composition and reactivity of dissolved organic carbon. Environ Sci Technol 37:4702–4708. doi:10.1021/es030360x

Worrall F, Fernandez-Perez M, Johson AC, Flores-Cesperedes F, Gonzalez-Pradas E (2001) Limitations on the role of incorpored organic matter in reducing pesticide leaching. J Contam Hydrol 49:241–262

Wu C, Spongberg AL, Witter JD, Sridhar BBM (2012) Transfer of wastewater associated pharmaceuticals and personal care products to crop plants from biosolids treated soil. Ecotoxicol Environ Saf 85:104–109. doi:10.1016/j.ecoenv.2012.08.007

Wu XL, Xiang L, Yan QY et al (2014) Distribution and risk assessment of quinolone antibiotics in the soils from organic vegetable farms of a subtropical city, southern China. Sci Total Environ 487:399–406. doi:10.1016/j.scitotenv.2014.04.015

Xu J, Wu L, Chang AC (2009) Degradation and adsorption of selected pharmaceuticals and personal care products (PPCPs) in agricultural soils. Chemosphere 77:1299–1305 doi:10.1016/j.chemosphere.2009.09.063