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1
2
3 **Diffusion and sorption of organic micropollutants**
4 **in biofilms with varying thicknesses**
5

6 Elena Torresi^{1,2†**}, Fabio Polese^{1†***}, Kai Bester³, Magnus Christensson², Barth F. Smets¹,
7 Stefan Trapp¹, Henrik R. Andersen¹, Benedek Gy. Plósz^{1,4*}

8
9 ¹DTU Environment, Technical University of Denmark, Bygningstorvet B115, 2800 Kongens Lyngby, Denmark

10 ²Veolia Water Technologies AB, AnoxKaldnes, Klosterängsvägen 11A, SE-226 47 Lund, Sweden

11 ³Department of Environmental Science, Århus University, Frederiksborgvej 399, 4000 Roskilde, Denmark

12 ⁴Department of Chemical Engineering, University of Bath, Claverton Down, Bath BA2 7AY, UK

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15 † The authors equally contributed to this manuscript.

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17
18 *Benedek Gy. Plósz: b.g.plosz@bath.ac.uk

19 **Elena Torresi: elto@env.dtu.dk

20 ***Fabio Polese: fabp@enf.dtu.dk

23 Abstract

24 Solid-liquid partitioning is one of the main fate processes determining the removal of
25 micropollutants in wastewater. Little is known on the sorption of micropollutants in biofilms,
26 where molecular diffusion may significantly influence partitioning kinetics. In this study, the
27 diffusion and the sorption of 23 micropollutants were investigated in novel moving bed biofilm
28 reactor (MBBR) carriers with controlled biofilm thickness (50, 200 and 500 μm) using targeted
29 batch experiments (initial concentration = $1 \mu\text{g L}^{-1}$, for X-ray contrast media $15 \mu\text{g L}^{-1}$) and
30 mathematical modelling. We assessed the influence of biofilm thickness and density on the
31 dimensionless effective diffusivity coefficient f (-, equal to the biofilm-to-aqueous diffusivity
32 ratio) and the distribution coefficient $K_{d,eq}$ (L g^{-1}). Sorption was significant only for eight
33 positively charged micropollutants (atenolol, metoprolol, propranolol, citalopram, venlafaxine,
34 erythromycin, clarithromycin and roxithromycin), revealing the importance of electrostatic
35 interactions with solids. Sorption equilibria were likely not reached within the duration of
36 batch experiments (4 h), particularly for the thickest biofilm, requiring the calculation of the
37 distribution coefficient $K_{d,eq}$ based on the approximation of the asymptotic equilibrium
38 concentration ($t > 4 \text{ h}$). $K_{d,eq}$ values increased with increasing biofilm thickness for all sorptive
39 micropollutants (except atenolol), possibly due to higher porosity and accessible surface area
40 in the thickest biofilm. Positive correlations between $K_{d,eq}$ and micropollutant properties
41 (polarity and molecular size descriptors) were identified but not for all biofilm thicknesses,
42 thus confirming the challenge of improving predictive sorption models for positively charged
43 compounds. A diffusion-sorption model was developed and calibrated against experimental
44 data, and estimated f values also increased with increasing biofilm thickness. This indicates
45 that diffusion in thin biofilms may be strongly limited ($f \ll 0.1$) by the high biomass density
46 (reduced porosity).

47

48 **Keywords:** Pharmaceuticals, wastewater, moving bed biofilm reactor, partitioning, biofilm

49 density, ionizable chemicals

50

51 1. Introduction

52 In wastewater treatment systems, partitioning of organic micropollutants to solid matrices is one of
53 the mechanisms leading to their removal from the aqueous phase. The extent of partitioning is
54 typically compound-dependent, and is governed by its affinity for organic phase (i.e., hydrophobic
55 partitioning) and/or by electrostatic and other similar interactions between ionized molecules and
56 charged solid surfaces (i.e., non-hydrophobic partitioning) (Franco and Trapp, 2008; Hyland et
57 al., 2012; Ternes et al., 2004; Mackay and Vasudevan, 2012; Polesel et al., 2015).

58 Partitioning describes the distribution of molecules between the aqueous and the solid phase.
59 At equilibrium, sorption and desorption rates are equal, and the ratio of sorbed and dissolved
60 concentrations—normalized to the concentration of solids—is defined as the (linear) solid-
61 liquid partition coefficient K_d (expressed in units of L kg^{-1} or, alternatively, L g^{-1}) (Joss et al.,
62 2006; Ternes et al., 2004). Non-linear expressions (Freundlich and Langmuir isotherms) have
63 been also used to describe partitioning equilibria to account for saturation of solid surfaces or
64 synergistic effects (Delle Site, 2001).

65 Solid-liquid partitioning has been characterized for activated sludge biomass for a high number
66 of pharmaceuticals. Considerably less evidence is available for wastewater treatment biofilms,
67 being limited to antibiotics (sulfamethoxazole, erythromycin, ciprofloxacin, tetracycline) and
68 psycho-active drugs (fluoxetine) in biofilters (Wunder et al., 2011) and granules (Alvarino et
69 al., 2015; Shi et al., 2011). Additionally, partitioning kinetics of other organic contaminants
70 (polycyclic aromatic hydrocarbons, estrogens, nonylphenols, biocides) have been assessed for
71 pure culture biofilms (Wicke et al., 2008, 2007) and river biofilms (Headley et al., 1998;
72 Writer et al., 2011).

73 Although considered a fast process, partitioning is influenced by mass transfer limitation
74 through diffusive boundary layers and inside the solid matrices, which likely determines the

75 time needed to achieve equilibrium between aqueous and sorbed concentrations (Joss et al.,
76 2004, 2006). While for activated sludge the equilibrium time is sufficiently fast to prevent an
77 empirical evaluation of mass transfer limitation (Joss et al., 2004; Plósz et al., 2010; Barret et
78 al., 2011), molecular diffusion may have a major role in determining partitioning kinetics in
79 biofilms. Biofilm characteristics such as biomass density and porosity have been found to
80 influence intra-biofilm diffusion of a number of organic and inorganic chemical compounds.
81 This effect has been described by introducing a coefficient f , defined as the ratio of effective
82 diffusivity in biofilms and in free aqueous media, thus defining diffusivity reduction in
83 biofilms (Fan et al., 1990; Guimerà et al., 2016; Horn and Morgenroth, 2006; Trapp and
84 Matthies, 1998; Zhang and Bishop, 1994a). While f was determined for a number of organic and
85 inorganic chemical compounds, no conclusive evidence currently exists for organic micropollutant
86 diffusion in biofilms, which was therefore investigated in this study.

87 In our previous work (Torresi et al., 2016), we investigated the biological transformation of
88 pharmaceuticals in nitrifying moving bed biofilm reactors (MBBRs) using novel MBBR
89 carriers (AnoxKaldnes Z-carriers), allowing the control of the biofilm thickness.

90 In this study, the main objective set was to assess how the diffusion and partitioning of 23
91 selected pharmaceuticals vary at different biofilm thicknesses (50, 200 and 500 μm) and to
92 quantify corresponding single point K_d values at environmentally relevant concentration levels.

93 By developing and calibrating a model that describes diffusive transport and partitioning in
94 biofilms, we aimed at elucidating the influence of biofilm thickness on (i) the molecular
95 diffusion of micropollutants within biofilm matrix, described by the dimensionless effective
96 diffusivity coefficient f ; (ii) the extent of partitioning, described by coefficient K_d .
97 Additionally, using experimental and modelling results, the influence of biofilm characteristics
98 (porosity, density) and molecular properties (e.g., hydrophobicity, ionization) on the mass

99 transfer limitation and sorption of micropollutants in biofilms were assessed.

100 2. Model development

101

102 2.1 Conceptual approach for diffusion and sorption in biofilms and model implementation

103 Considering molecular diffusion of dissolved micropollutants from the bulk aqueous phase into
104 biofilms as the dominant mechanism (Zhang and Bishop, 1994a), the partitioning of organic
105 micropollutants consists of three consecutive steps (Joss et al., 2004): (1) diffusion of
106 dissolved micropollutant from bulk aqueous phase, through a boundary layer, into the biofilm
107 matrix; (2) diffusion of dissolved micropollutant through the biofilm matrix via its pores; (3)
108 sorption to the solid phase of the biofilm matrix (Fig. 1a). The diffusivity of organic chemicals
109 in a free aqueous medium ($D_{W,i}$, $\text{m}^2 \text{d}^{-1}$) can be predicted from properties of the chemical (e.g.,
110 molar volume) and of the medium. In this study, $D_{W,i}$ values for each chemical were calculated
111 according to Hayduk and Laudie (1974), although alternative approaches were also tested
112 (Table S1 in Supplementary Information).

113 Transport from the bulk liquid to the biofilm is controlled by the diffusion rate through a
114 boundary layer, for which the diffusivity was assumed equal to $D_{W,i}$ (**Assumption I**, Fig 1b).
115 The thickness of the boundary layer, L_L (μm), was assumed to be equal to 10 μm for all the Z-
116 carriers (Brockmann et al., 2008, see section 1 in SI). In biofilms, molecular diffusivity is
117 reduced compared to free aqueous media (Wanner and Reichert, 1996). This has been
118 attributed to the “tortuosity” of the transport path in biofilms, i.e. the increased (non-linear)
119 path length needed for diffusive transport as compared to free aqueous media (Zhang and
120 Bishop, 1994b). Molecular diffusivity reduction is described by the dimensionless coefficient
121 f , resulting in Eq. 1:

$$122 D_{bf,i} = f \cdot D_{W,i} \quad (\text{Eq. 1})$$

123 where $D_{bf,i}$ ($\text{m}^2 \text{d}^{-1}$) is the effective diffusivity of micropollutants within biofilms and f (-) is
124 always lower than 1. While f values of 0.5–0.8 have been assigned for micropollutant diffusion
125 (Ort and Gujer, 2008; Vasiliadou et al., 2014), this parameter is likely to vary significantly
126 depending on the biofilm structure and properties (biofilm thickness, density, porosity and
127 tortuosity).

128 It has previously been shown that biofilm porosity and density can vary over the biofilm depth
129 (Zhang and Bishop, 1994a). In the model, we assume the biofilm as a homogenous porous
130 medium (**Assumption II**), although we accept that biofilms with different depth can have
131 different average porosities and densities. As a consequence, only one f value was used to
132 describe diffusion reduction into a biofilm with a certain thickness.

133 Sorption/desorption kinetics were described using first-order rate equations (see matrix in Fig.
134 1c). Sorption was considered as an equilibrium process (**Assumption III**), by attributing an
135 arbitrarily high value to the desorption rate k_{des} , thereby making diffusion from the bulk
136 aqueous phase and within the biofilm the rate-limiting steps for solid-liquid partitioning. At
137 micropollutant concentration levels targeted in this study (ng L^{-1} to $\mu\text{g L}^{-1}$), sorption can be
138 considered linear and better described by the distribution coefficient K_d (**Assumption IV**).

139 Based on the presented conceptual approach, a diffusion-sorption model was implemented as
140 one-dimensional biofilm model in Aquasim 2.1 (Reichert, 1994). Design and measured biofilm
141 properties (biofilm thickness, surface area, biomass density, porosity) were used as input to the
142 model (see Table 1). Each biofilm was spatially discretized in 20 completely mixed layers.
143 This allowed solving the generic mass balance equation for dissolved micropollutant
144 concentration C_L (ng L^{-1}) in biofilm (Eq. 2):

$$145 \quad \frac{\partial C_L}{\partial t} = D_{bf,i} \frac{\partial^2 C_L}{\partial z^2} - k_{des} K_d C_L X + k_{des} C_S \quad (\text{Eq. 2})$$

146 (where X is the biomass concentration in biofilm, g L^{-1} ; C_S is the sorbed micropollutant
 147 concentration, ng L^{-1} ; C_L varies with time t and depth z) as a set of ordinary differential
 148 equations by using the method of lines (Wanner and Reichert, 1996). According to the
 149 diffusion-sorption model, micropollutants undergo equilibrium microscale partitioning as they
 150 diffuse through biofilm, in analogy to the approach proposed by Wu and Gschwend (1986).
 151 Further details on the conceptual biofilm model, on microscopic mass balances and on the
 152 initial conditions are given in the Supplementary Information (section S1 and Figure S1).

153 < Figure 1 >

154

155 **2.2 Calculation of sorption coefficients**

156 At equilibrium, the micropollutant concentration sorbed onto biomass ($C_{S,eq}$, $\mu\text{g L}^{-1}$) is
 157 proportional to the dissolved concentration ($C_{L,eq}$, $\mu\text{g L}^{-1}$), and their ratio, normalized by the
 158 concentration of solids ($X_{biomass}$, g L^{-1}), is used to calculate the sorption coefficient K_d (L g^{-1}).
 159 With negligible transformation, it is commonly assumed (e.g., in activated sludge) that the
 160 sorbed concentration is equivalent to the decrease in dissolved concentration ($C_{L,0} - C_{L,eq}$)
 161 between the beginning and the end of batch sorption experiments.

162 When considering biofilm systems, transport in biofilm pores, along with sorption, can also
 163 determine a decrease of micropollutant concentrations in the bulk phase. Hence, the coefficient
 164 $K_{d,eq}$ (L g^{-1}) was defined to describe sorption in Z-carrier biofilms based on mass balance
 165 considerations (Eq. 3):

$$166 \quad K_{d,eq} = \frac{\left[\frac{C_{L,0} V_{bulk}}{V_{bulk} + V_{bf,wet}} - \frac{C_{L,eq} (V_{bulk} + V_{PW})}{V_{bulk} + V_{bf,wet}} \right]}{C_{L,eq} X_{biomass}} \quad (\text{Eq. 3})$$

167 where V_{bulk} (L) denotes the volume of the bulk liquid, $V_{bf,wet}$ (L) the volume of wet biofilm
 168 (equal to the total surface area of Z-carriers times the defined biofilm thickness) and V_{PW} (L)

169 the volume of the pore water in the biofilm matrix, not accounting for cellular water content
 170 (see 3.4). The procedure used to derive Eq. 3 is presented in detail in the Supplementary
 171 Information (section S3).

172 The ‘asymptotic’ concentration $C_{L,eq}$, defining true sorption equilibrium, was estimated by
 173 fitting measured concentration profiles in batch sorption experiments with a first-order decay
 174 equation (Eq. 4)

$$175 \quad C_L(t) = (C_{L,0} - C_{L,eq})e^{-kt} + C_{L,eq} \quad (\text{Eq. 4})$$

176 In activated sludge, it has been widely accepted that sorption equilibrium can be reached
 177 within 0.5–1 h (Ternes et al., 2004; Andersen et al., 2005; Yi and Harper, 2007, Hörsing et al.,
 178 2011). To verify whether sorption equilibrium was achieved relatively fast (i.e., within the 4-
 179 hour duration of sorption experiments) also in Z-carrier biofilms, the sorption coefficient $K_{d,4h}$
 180 (L g^{-1}) was calculated (Eq. 5):

$$181 \quad K_{d,4h} = \frac{\left[\frac{C_{L,0}V_{bulk}}{V_{bulk} + V_{bf,wet}} - \frac{C_{L,4h}(V_{bulk} + V_{PW})}{V_{bulk} + V_{bf,wet}} \right]}{C_{L,4h}X_{biomass}} \quad (\text{Eq. 5})$$

182 where $C_{L,4h}$ is the measured dissolved concentration in bulk aqueous phase at $t=4$ h (the last
 183 measurement in sorption experiments), replacing $C_{L,eq}$ in Eq. 3. Specifically, the 4-hour
 184 equilibrium assumption was verified by comparing $K_{d,4h}$ and $K_{d,eq}$ and assessing the relative
 185 deviation between the two coefficients.

186 As mentioned above, the decrease of bulk micropollutant concentration during sorption
 187 experiments with biofilms results from transport in biofilm pores (besides sorption in
 188 biofilms). To verify the impact of neglecting mass transfer to biofilm pores on sorption
 189 coefficient determination, the sorption coefficient $K_{d,susp}$ was calculated (Eq. 6):

$$190 \quad K_{d,susp} = \frac{C_{L,0} - C_{L,eq}}{C_{L,eq}X_{biomass}} \quad (\text{Eq. 6})$$

191 where $C_{L,eq}$ was calculated using Eq. 4. Notably, Eq. 6 is commonly used to describe sorption
192 onto suspended activated sludge, where the effect of porosity is neglected. The comparison
193 between $K_{d,eq}$ and $K_{d,susp}$ (together with relative deviation the two coefficients) was used to
194 quantify the contribution of transport to biofilm pores, hence the impact of porosity, on the
195 estimated sorption coefficient.

196

197 ***2.3 Parameter estimation approach***

198 The assessment of diffusion and sorption of micropollutants in biofilms consisted of two main
199 consecutive steps performed for each micropollutant and at different biofilm thicknesses: (i)
200 calculation of the coefficient $K_{d,eq}$ (section 2.2); (ii) calibration of the diffusion-sorption model
201 (section 2.1) against experimental data and estimation of the coefficient f , which was the only
202 parameters fitted in the model. Estimation of f was performed using the secant model
203 calibration algorithm embedded in Aquasim 2.1.

204 **3. Materials and methods**

205 **3.1. System description and operation**

206 Nitrifying MBBRs used in this study have been described elsewhere (Torresi et al., 2016).
207 Briefly, two laboratory-scale nitrifying MBBRs were operated in parallel under continuous-
208 flow conditions for approximately 300 days. Z-carriers (AnoxKaldnes AB, Lund, Sweden)
209 were used to obtain biofilm of different thicknesses. Z-carriers have a saddle shaped grid
210 covered surface allowing for biofilm growth only up to the height of the grid wall (Torresi et
211 al., 2016). Three different Z-carriers (named Z50, Z200, and Z500) were used in this study,
212 with the numbers indicating the grid wall height in μm (hence the maximum controlled biofilm
213 thickness). Biofilms were enriched by feeding the MBBRs with effluent wastewater from a
214 local municipal treatment plant (Källby, Lund, Sweden), spiked with ammonium (50 mg L^{-1} of
215 $\text{NH}_4\text{-N}$ as NH_4Cl) and phosphate (0.5 mg L^{-1} of $\text{PO}_4\text{-P}$ as KH_2PO_4). The MBBRs were operated
216 under similar conditions, i.e. hydraulic residence time of 2 h, dissolved oxygen concentration
217 of $4.5 \pm 0.5 \text{ mg L}^{-1}$, pH of 7.5 ± 0.5 and temperature of 20°C (achieved using a thermostat).

218

219 **3.2. Sorption batch experiments**

220 Sorption batch experiments were performed after reaching stable nitrogen removal (Torresi et
221 al., 2016), roughly, around day 300. Prior to batch experiments, the two MBBRs were
222 disconnected and three types of Z-carriers (Z50, Z200, Z500) were manually separated.
223 Subsequently, Z-carriers were left overnight at 4°C in a beaker with tap water to allow for
224 desorption of micropollutants possibly sorbed during continuous-flow operation.

225 Sorption batch experiments were carried out in three 200-mL glass beakers using filtered (0.2
226 μm Munktell MG/A glass fiber filters) effluent wastewater from Källby treatment plant.
227 Ammonium and nitrate in the feed were at concentration of $<0.5 \text{ mgN L}^{-1}$ and 6 mgN L^{-1} ,

228 respectively, while organic carbon concentration was lower than 35 mgCOD L⁻¹, mostly in
229 inert form.

230 The biomass concentration in the three glass beakers was adjusted to 0.8 g L⁻¹ based on
231 attached biomass concentration measurements for the different carriers and adjusting the
232 number of carriers accordingly (56 carriers for Z50, 32 for Z200 and 16 for Z500), resulting in
233 a total biofilm surface area of 0.06, 0.04, 0.02 m² for the batch containing Z50, Z200 and Z500
234 carriers, respectively. Other abiotic removal processes, such as volatilization, sorption of
235 micropollutants on plastic carriers and glass wall, had been previously assessed and found
236 negligible in MBBRs (Torresi et al., 2016).

237 Twenty-three micropollutants were spiked in all the beakers with an initial concentration of 1
238 µg L⁻¹ except for X-ray contrast media (15 µg L⁻¹), as they are usually found at higher
239 concentrations in effluent wastewater (Margot et al., 2015). A stock solution, containing
240 micropollutants dissolved in methanol (40 mg L⁻¹), was first spiked into empty glass beakers
241 and the methanol was allowed to evaporate in the fumehood for approximately 1 hour.
242 Subsequently, the solution was resuspended in filtered effluent for approximately 30 min to
243 dissolve the spiked micropollutants. Biomass inactivation was achieved by: (i) addition of
244 allylthiourea (ATU, 10 mg L⁻¹, Tran et al., 2009; Khunjar and Love, 2011) and nitrogen
245 sparging (Hamon et al., 2014) to inhibit nitrifying bacteria; and (ii) addition of sodium azide
246 (0.5 g L⁻¹; Rattier et al., 2014) to inhibit the activity of heterotrophic bacteria.

247 The experiment duration was set to 4 hours. Homogenous aqueous samples were collected at
248 regular intervals from the bulk phase in each beaker at 0, 5, 10, 30, 90 and 240 min. The batch
249 experiments were performed at ambient temperature and initial pH was measured to be 7.5 ±
250 0.5. Since only one spiking concentration was tested, results from sorption experiments were
251 used to determine single point K_d values.

252

253 **3.3. Chemicals**

254 Twenty-three environmentally relevant micropollutants were selected for this study. The
255 targeted pharmaceuticals were grouped in six categories according to their use: (i) four beta-
256 blocker pharmaceuticals (atenolol, metoprolol, propranolol and sotalol); (ii) five X-ray contrast
257 media (diatrizoic acid, iohexol, iopamidol, iopromide, iomeprol); (iii) three sulfonamide
258 antibiotics (sulfadiazine, sulfamethizole and sulfamethoxazole), one metabolite (acetyl-
259 sulfadiazine) and one combination product (trimethoprim); (iv) three non-steroidal anti-
260 inflammatory pharmaceuticals (phenazone, diclofenac, ibuprofen); (v) three psycho-active
261 drugs (carbamazepine, venlafaxine and citalopram); (vi) three macrolide antibiotics
262 (erythromycin, clarithromycin and roxithromycin). Further information regarding chemical
263 structure and properties, CAS numbers and chemical suppliers can be found in Table S2–S3
264 and in Escolà Casas et al. (2015).

265

266 **3.4. Analytical methods**

267 Samples for micropollutant analysis were collected (4 mL) and analysed via direct injection
268 using internal standards (injected volume of 100 µL). Details regarding sample preparation,
269 internal standards, HPLC and mass spectrometry conditions, limits of detection and
270 quantification are shown in Escolà Casas et al. (2015). Biomass concentration on Z-carriers was
271 measured in two ways: (i) as attached biomass concentration (expressed as total attached
272 solids, TAS), calculated from the difference in weight of three dried carriers (105°C for > 24 h)
273 before and after biofilm removal (using 2M H₂SO₄ with subsequent brushing) (see also Escolà
274 Casas et al., 2015; Falås et al., 2013; Torresi et al., 2016); and (ii) by scraping and suspending
275 the biofilm in tap water and measuring total suspended solids (TSS) and volatile suspended

276 solids (VSS) according to APHA standard methods (Clesceri, 1989). Biofilm properties such
277 as biofilm dry density ρ_d (g cm^{-3}), biomass density in wet biofilm ρ (kg m^{-3}) and porosity ε (%)
278 were calculated according to Tchobanoglous et al. (2003) and Hu et al. (2013) using measured
279 biofilm properties (e.g., solids content), as detailed in the Supplementary Information (section
280 S2). Porosity is defined as the fraction of the biofilm volume occupied only by water outside
281 the cells and not inside the cells (Hu et al., 2013). Furthermore, ρ_d denotes the dry mass of
282 biofilm per volume of dry biofilm (i.e., defines a *true* density) while ρ denotes the dry mass of
283 biomass per volume of wet biofilm (i.e., defines a concentration of biomass within the
284 biofilm). Further discussion on the calculation methodology used and on the biofilm properties
285 can be found in section S2.

286

287 **3.5. Statistical analysis and influence of chemical properties**

288 Pearson's and Spearman's correlations between $K_{d,eq}$ and chemical properties (expressed in
289 logarithmic base) were assessed at different biofilm thicknesses. A significance level of 0.05
290 was used for all statistical tests in this study. The investigated physico-chemical properties
291 include: the molecular volume MV ($\text{cm}^3 \text{mol}^{-1}$); the dissociation constant(s) pK_a ; the number of
292 rotatable bonds (nRB); the van der Waals area ($vdWA$, $\text{m}^2 \text{kmol}^{-1}$) (Sathyamoorthy and
293 Ramsburg, 2013); McGowan's approximation of the molecular volume (V_X , $\text{cm}^3 \text{mol}^{-1}$) (Droge
294 and Goss, 2013a); and the topological polar surface area ($TPSA$, \AA^2) (Mannhold, 2008).
295 Chemical properties and $K_{d,eq}$ were log transformed (Vasudevan et al., 2009) with exception of
296 nRB (Sathyamoorthy and Ramsburg, 2013) and molecular size descriptors MV and V_X .
297 Chemical properties for each compound were retrieved using ACD/Labs predictions and the
298 database Mol-inisticts (for $\log vdWA$) or calculated based on previously defined equations (for
299 V_X : Abraham and McGowan, 1987). Pearson's and Spearman's correlations and their

300 significance were assessed using GraphPad Prism 5.0. Furthermore, possible correlations
301 between f and the abovementioned properties were also investigated. Significant differences
302 between estimated f values for each chemical at different biofilm thickness were determined by
303 examining the overlap between standard deviations of the estimate (Cumming et al., 2007)

304 4. Results and discussion

305 4.1. Biofilm properties

306 Measured and calculated values for a number of biofilm properties are reported in Table 1. Dry
307 biofilm mass per surface area of carrier (gTAS m^{-2} , Table 1) increased with biofilm thickness,
308 being approximately four times higher in Z500 compared to Z50. Biofilm thickness in Z-
309 carriers was recently measured using optical coherence tomography (OCT), revealing good
310 agreement between measured and nominal thickness based on carrier design (Piculell et al.,
311 2016). Conversely, biofilm density in wet biofilm ρ (section 1 in SI) in Z50 was up to 3-fold higher
312 as compared to Z200 and Z500. This suggests a change in biofilm porosity as a function of
313 biofilm thickness. Biofilm porosity ε (Eq. S12), ranged from 75% (Z50) to 93% (Z500) (Table
314 1). An approximate porosity of 80% is commonly assumed in one-dimensional biofilm models
315 (Wanner and Reichert, 1996; Brockmann et al., 2008) and similar values have been previously
316 determined using modelling approximations (Zacarias et al., 2005; Zhang and Bishop, 1994b).
317 The observed increasing ε with biofilm thickness is in agreement with previous findings for Z-
318 carrier biofilms (Piculell et al., 2016), although lower porosities (approximately of 10 and 30%
319 for Z50 and Z400) were estimated using OCT. Values of biofilm dry density ρ_d (Table 1) for
320 the three biofilms were comparable to that shown in literature (Hu et al., 2013), indicating a
321 higher content of fixed solids in Z500.

322 < Table 1 >

323

324 4.2. Sorption coefficients in biofilms

325 Sorption was considered significant when a relative concentration drop $(C_{L,0} - C_{L,4h})/C_{L,0}$
326 higher than 10% was observed (Hörsing et al., 2011), thus accounting for analytical

327 uncertainty. Profiles of aqueous concentration of the sorptive micropollutants measured during
328 batch experiments are shown in Fig. 3 (duplicate measurement) and in Fig. S2.

329 Out of the 23 targeted compounds, sorption was significant only for eight micropollutants,
330 namely atenolol, metoprolol, propranolol, citalopram, venlafaxine, erythromycin,
331 clarithromycin and roxithromycin. The presence of chemicals not exhibiting sorption (e.g.,
332 diclofenac and the targeted sulfonamides) suggests that biomass was successfully inhibited
333 during batch experiments, as most targeted compounds were significantly biodegraded in the
334 same MBBRs without biomass inhibition (Torresi et al. 2016). Interestingly, micropollutants
335 that were positively charged (>90% cationic fraction) at the experimental pH of 7.5 presented
336 significant sorption, with exception of sotalol and trimethoprim. Higher sorption potential of
337 positively charged compounds compared to negatively charged or neutral compounds was
338 previously observed for activated sludge biosolids (Stevens-Garmon et al., 2011; Polesel et al.,
339 2015) and soil (Franco and Trapp, 2008).

340

341 *4.2.1. Sorption coefficients $K_{d,eq}$ and comparison with activated sludge*

342 Sorption coefficients $K_{d,eq}$ in Z50, Z200 and Z500 biofilms were calculated for the above listed
343 cationic micropollutants (Table 2). $K_{d,eq}$ values were compared with previously found sorption
344 coefficients in activated sludge, for which the large majority of micropollutant sorption data
345 are available.

346

< Table 2 >

347 Values of $K_{d,eq}$ for atenolol at all the three biofilm thickness were up to 2-fold higher than
348 literature values (Radjenović et al., 2009; Stevens-Garmon et al., 2011), while values in Z50
349 and Z200 were comparable with findings for secondary sludge (Hörsing et al., 2011). As
350 atenolol presents similar molecular properties to other beta-blockers (e.g., molecular weight,

351 pK_a), the reasons behind this high sorption potential are unclear. As to metoprolol, $K_{d,eq}$ values
352 in Z50, Z200 and Z500 were comparable to previously measured coefficients in activated
353 sludge biomass (Maurer et al., 2007; Sathyamoorthy et al., 2013). Similarly to previous
354 studies, propranolol exhibited the highest sorption potential of all selected beta-blockers
355 (Maurer et al., 2007; Radjenović et al., 2009). Notably, a fourth targeted beta blocker sotalol
356 did not show any significant sorption, in agreement with previous findings in activated sludge
357 (Maurer et al., 2007; Sathyamoorthy et al., 2013).

358 Values of $K_{d,eq}$ for Z50 and Z200 were comparable with previous studies on conventional
359 activated sludge and membrane bioreactor (MBR) sludge for clarithromycin (Abegglen et al.,
360 2009; Göbel et al., 2005), erythromycin (Radjenović et al., 2009; Xue et al., 2010) and
361 roxithromycin (Abegglen et al., 2009; Hörsing et al., 2011). On the contrary, $K_{d,eq}$ for Z500
362 differed by one order of magnitude from previously reported values. Nevertheless, 50–80% of
363 dissolved clarithromycin and roxithromycin sorbed on MBR sludge (Abegglen et al., 2009),
364 similarly to clarithromycin and erythromycin in this study (~80%). Furthermore, highly
365 variable macrolide sorption was shown in soil and onto humic acids (Sibley and Pedersen,
366 2008; Uhrich et al., 2014), with estimated K_d values also higher than 8 L g⁻¹ or 20 L g⁻¹,
367 respectively. Macrolides exhibited the highest $K_{d,eq}$ of all sorptive compounds in Z500 but not
368 at lower biofilm thickness (Table 2). This might be related to the low porosity of the biofilms
369 Z50 and Z200. According to Lipinski's rule of five (Lipinski et al., 1997), macrolides are
370 expected to poorly permeate across cell membranes and thus to move only in the intracellular
371 space (depending on the porosity) due to their high molecular weight (>500 g mol⁻¹).
372 Furthermore, macrolides are mainly excreted in feces (Göbel et al., 2005) and due to
373 protonation of the tertiary amino group, strong ionic interaction of macrolides with the
374 negatively charged surface of the biomass could be expected.

375 Few studies investigated the sorption of the antidepressant venlafaxine and the antiepileptic
376 citalopram. While sorption coefficients for Z50 and Z200 for both compounds are in agreement
377 with existing literature on activated sludge (Hörsing et al., 2011), higher values were found in
378 Z500 for citalopram.

379 In general, sorption coefficients of all the compounds at the three biofilm thicknesses were
380 comparable or higher than values observed with activated sludge biomass. Studies comparing
381 sorption onto MBR sludge and conventional activated sludge biomass (Joss et al., 2006;
382 Abegglen et al., 2009; Reif et al., 2011; Yi and Harper, 2007) revealed a sorption enhancement
383 in the former case. Increased sorption was associated to the smaller size of MBR sludge flocs
384 (assumed to be around 80–300 μm in diameter), thus resulting in higher accessible surface area
385 (Tchobanoglous et al., 2003). In analogy with MBR sludge, it can be postulated that the high
386 accessible surface area in Z-carrier biofilms (related to the biofilm porous structure) may
387 explain the increased sorption capacity of most of the compounds compared to conventional
388 activated sludge biomass.

389

390 *4.2.2. Comparison between $K_{d,eq}$ and $K_{d,4h}$*

391 Sorption coefficients $K_{d,eq}$ were compared with $K_{d,4h}$ values for each chemical and relative
392 deviations Δ (%) between these two coefficients were calculated at different biofilm
393 thicknesses (Table 2) to verify the equilibrium assumption within the experiment duration (4
394 hours). For most compounds, relative deviations for Z50 and Z200 were on average around
395 10%, with the exception of atenolol (>50%). Conversely, Δ values in Z500 were – for most of
396 the eight compounds – higher than 30% (up to 80% for atenolol).

397 Overall, while the assumption of equilibrium reached within 4 h seems justified for Z50 and
398 Z200, diffusive mass transfer can significantly influence observations at higher biofilm

399 thickness. Atenolol was the main exception, for which the 4-h equilibrium assumption seems
400 not valid at any biofilm thickness. On the contrary, propranolol appeared to reach partitioning
401 equilibrium within 4-h in Z50, Z200 and Z500, and similar considerations could be made for
402 citalopram and venlafaxine. Therefore, to reduce uncertainties in sorption experiments,
403 parameter estimation can benefit from calculating the asymptotic aqueous concentration value
404 using e.g., simplified first-order decay equations (Eq. 4).

405

406 *4.2.3. Comparison between $K_{d,eq}$ and $K_{d,susp}$ and trends with biofilm thickness*

407 To assess the impact of biofilm porosity and mass transfer in pores on sorption coefficient
408 estimation, the sorption coefficients $K_{d,eq}$ and $K_{d,susp}$ were compared (Table S4). In Fig. 2, this
409 comparison is presented for two key chemicals (a: metoprolol, b: roxithromycin). For all
410 micropollutants, neglecting the transport from bulk aqueous phase to biofilm pores resulted in
411 an overestimation of sorption coefficients ($K_{d,susp}$ always greater than $K_{d,eq}$). The relative
412 deviation between $K_{d,susp}$ and $K_{d,eq}$ was on average $\leq 10\%$ for most compounds and 30% for less
413 sorptive compounds (metoprolol and venlafaxine).

414 We further observed that both $K_{d,eq}$ and $K_{d,susp}$ generally increased with increasing biofilm
415 thickness (Fig. 2). Specifically, $K_{d,eq}$ values in Z500 were from 4-fold (most of the compounds)
416 up to 30-fold higher (macrolides antibiotics) than in Z50 (Table 2). It should be highlighted
417 that batch experiments were carried out at the same biomass concentration in the reactors (0.8
418 g L⁻¹). Consequently, the observed $K_{d,eq}$ increase with biofilm thickness likely derives from
419 differences in biofilm composition and/or in its physical properties. Two possible explanations
420 of this observation were proposed:

421 (i) Biomass composition, such as the relative fraction of autotrophic and heterotrophic bacteria
422 and/or the content of extracellular polymeric substances (EPS), can influence sorption

423 properties. EPS protein content was previously positively correlated with K_d for aromatic
424 chemicals in untreated and treated sewage sludge and colloids (Barret et al., 2010) and for the
425 estrogen EE2 and trimethoprim in nitrifying and heterotrophic biomass (Khunjar and Love,
426 2011). Bassin et al., (2012) further observed higher concentration of proteins and
427 polysaccharides (that mainly compose EPS) in heterotrophic MBBRs than in nitrifying
428 MBBRs. Higher fractions of heterotrophic bacteria (determined using quantitative PCR of 16S
429 rRNA) were measured in Z200 and Z500 compared to Z50 (Torresi et al., 2016), possibly
430 justifying the increased sorption capacity in thicker biofilms (Z200, Z500). Further
431 investigation on the EPS content in the different biofilms is thus required to support this
432 hypothesis, given the key role of EPS in the sorption of neutral and ionizable organic
433 chemicals (Späth et al., 1998; Barret et al., 2010; Khunjar and Love, 2011).

434 (ii) Porosity can influence the available surface area inside the biofilm. Sorption has been
435 previously positively impacted by reduced particle size, i.e., greater surface area, in suspended
436 biomass (Khunjar and Love, 2011) and biomass floc suspension derived from MBRs (Yi and
437 Harper, 2007). Thicker biofilms, having lower biomass density and substantially higher
438 porosity than thin biofilms, could accordingly provide for higher available surface (and thus
439 more accessible sites) for solid-liquid partitioning.

440 Finally, $K_{d,eq}$ values were normalized to the highest value of $K_{d,eq}$ (i.e., for Z500, $K_{d,eqZ500}$). The
441 obtained profiles followed two distinct trends as a function of biofilm thickness (Fig. 2c–d): (i)
442 beta-blockers and venlafaxine, exhibiting a logarithmic-like increase between Z50 and Z500;
443 and (ii) macrolides and citalopram, presenting significantly higher values for Z500, thus an
444 exponential-like increase of $K_{d,eq}$ with thickness. The question arises as to the influence of the
445 specific chemical properties of micropollutants on partitioning in biofilms, which was further
446 assessed using correlation analysis (see 4.5.2).

447 < Figure 2 >

448

449 **4.3. Modelling diffusion and sorption in biofilm**

450 Based on the considerations above, calculated $K_{d,eq}$ were used to calibrate the diffusion-
451 sorption model against experimental data for the estimation of the dimensionless effective
452 diffusivity coefficient f (the only parameter estimated with the model). Simulated aqueous
453 concentrations (continuous lines, Fig. 3) predicted reasonably well the measured
454 concentrations in bulk liquid (circles, Fig. 3) for most of the targeted compounds (i.e., for
455 propranolol, clarithromycin, erythromycin, roxithromycin, citalopram, venlafaxine $R^2 > 0.9$;
456 Table S5). For atenolol, measured concentrations were less well predicted for Z50 and Z500
457 (R^2 equal to 0.8).

458 The simulated micropollutant concentrations in the bulk liquid and in the biofilm pores liquid
459 (dashed lines, Fig. 3) should converge when partitioning equilibrium is reached. This
460 equilibrium condition was satisfied for most compounds in Z50 and Z200 within 4 h
461 experimental time, with an average 10% relative deviation between simulated concentrations in
462 bulk and in biofilm pores. For the thickest biofilm (Z500), however, model predictions for
463 most of targeted chemicals suggested that equilibrium was not reached within 4 h (60%
464 average discrepancy with the last measurement). It is likely that, due to the greater thickness,
465 increased time to diffuse in deeper biofilm and thus to achieve sorption equilibrium is required
466 in Z500. The exception was propranolol, for which equilibrium seemed to be reached in all the
467 three biofilms, thus supporting results (relative deviation between $K_{d,4h}$ and $K_{d,eq}$) presented in
468 Table 2. For macrolide antibiotics, this discrepancy was significant and simulation results
469 suggested a time for partitioning equilibrium of approximately 10 days—in good agreement
470 with equilibrium times (days, months and years) in other environmental matrices (Delle Site,

471 2001). Furthermore, the large molecular volume and weight of macrolides (2- to 3-fold higher
472 than the other targeted compounds, Table S2), as well as their high sorption potential in Z500,
473 suggest slower diffusive transport inside the biofilm, as previously observed for hydrophobic
474 organic molecules in sediments and soil (Wu and Gschwend, 1986).

475 There is a large variability concerning the time to reach partitioning equilibrium for organic
476 chemicals in biofilms (Alvarino et al., 2015; Headley et al., 1998; Shi et al., 2011; Wicke et
477 al., 2008; Writer et al., 2011), with values ranging from, e.g., 4 to 80 h for biofilm of 0.1 mm
478 thickness (Wicke et al., 2008). In conclusion, our observations conflict with the widely held
479 assumption of significantly shorter period of time (i.e. minutes to 1–2 hours) necessary to
480 reach equilibrium in activated sludge (e.g., Hörsing et al., 2011; Pomiès et al., 2013). This may
481 be explained by differences in pore-scale (hydro)dynamic conditions in MBBRs and activated
482 sludge reactors, resulting in more pronounced mass transfer limitation in MBBRs.

483

< **Figure 3** >

484

485 **4.4. Influence of biofilm and chemical properties on f and $K_{d,eq}$**

486 **4.4.1. Estimation of f and proposed empirical correlation**

487 Values of the dimensionless effective diffusivity coefficient f estimated for the three biofilm
488 thicknesses and the eight sorptive compounds are reported in Fig. 4. For most of the
489 compounds, with the exception of roxithromycin, f decreased with biofilm density and thus
490 increased with biofilm thickness and porosity (with f in Z500 significantly higher than in Z200
491 and Z50 for all the compounds, and f in Z200 significantly higher than Z50 for six
492 compounds). In thinner biofilms ($\leq 50 \mu\text{m}$), the transport of micropollutants could thus be
493 limited by the high biomass density and the reduced porosity. A number of regressions to
494 estimate f of solutes in biofilms as a function of biofilm density or porosity have been

495 previously developed (Fan et al., 1990; Guimerà et al., 2016; Horn and Morgenroth, 2006;
496 Zhang and Bishop, 1994a), suggesting a negative correlation between f and density. Selected
497 regression profiles (i.e., Guimerà et al., 2016; Horn and Morgenroth, 2006; Zhang and Bishop,
498 1994a; see Table S6) are reported in Fig. S3 for comparison with our f estimations. In
499 particular, Guimerà et al. (2016) observed strong mass transfer limitation ($f < 0.1$) for oxygen
500 in biofilm with density greater than 50 gVSS L⁻¹, in close agreement with findings (specifically
501 for Z50) presented in this study.

502 **< Figure 4 >**

503 In general, estimated f were lower than values calculated from proposed regressions (Guimerà
504 et al., 2016; Horn and Morgenroth, 2006; Zhang and Bishop, 1994a) (Fig. S3). While these
505 regressions were identified for solutes with lower molecular weight (< 100 g mol⁻¹) and high
506 solubility (e.g., O₂, sodium chloride, sodium nitrate), lower values of f (~0.2) were reported for
507 most organic solutes with larger molecular weight (e.g., sugars and fatty acids; Stewart, 2003,
508 1998).

509 Given the possible influence of chemical properties on micropollutant diffusivity, we evaluated
510 the relationship between f and several physico-chemical descriptors (section 3.5). No specific
511 correlation was observed between f and molecular volume and other descriptors (Fig. S4). We
512 observed a positive correlation only between f and $\log K_{OW}$ of the targeted compounds (Fig.
513 S5), while negative dependence was reported in literature for organic compounds (Headley et
514 al., 1998; Wicke et al., 2007; Wu and Gschwend, 1986). Notably, in this study the correlation
515 was found for less hydrophobic ($0.1 < \log K_{OW} < 3.7$) and positively charged compounds
516 (differently from previous studies), for which electrostatic interactions may also have
517 influenced transport and partitioning. Thus, an empirical correlation between f , biofilm density
518 ρ (as function of biofilm thickness) and $\log K_{OW}$ is proposed (Eq. 7):

$$f = \frac{1}{488 \cdot e^{-0.0072L_F}} \ln \left(\frac{-127 - \log K_{OW,max}}{\log K_{ow} - \log K_{OW,max}} \right) \quad (\text{Eq. 7})$$

where L_F is the biofilm thickness (μm) and $\log K_{OW,max}$ is the asymptotic $\log K_{OW}$, approximating the highest value for the compounds selected. Profiles of f deriving from Eq. 7 were then depicted in Fig. 5, along with the estimated f values for the three biofilm thickness (symbols, see also Fig. 4). Further details on the formulation of Eq. 7 are given in the SI (section S4). We note that the size of the available data set may not be sufficiently large to validate the correlation, and additional experimental evidence (higher biofilm thickness, wider range of $\log K_{OW}$) may be required for further confirmation.

< Figure 5 >

4.4.2. Predictors of micropollutant $K_{d,eq}$ in biofilms

Correlation analyses were performed between $K_{d,eq}$ and a number of physico-chemical micropollutant descriptors.

First, the octanol-water partitioning coefficient of the neutral species ($\log K_{OW}$) and the species-dependent octanol-water distribution coefficient ($\log D$) were assessed, exhibiting insignificant correlation with $K_{d,eq}$ ($-0.27 < \text{Pearson's } r < 0.15$ for the three biofilms). This finding confirms the limited reliability of $\log K_{OW}$ and $\log D$ as sorption predictors for organic cations, as previously shown in soil (Tolls, 2001; Franco and Trapp, 2008; Droge and Gross, 2013a).

Following this preliminary assessment, correlations with physico-chemical descriptors for ionizable compounds (i.e., $\log pK_a$, nrB , MV , $\log TPSA$, $\log vdWA$, V_X) were investigated (Fig. 6 and S6). Correlations for biofilm Z50 was performed only considering six compounds ($K_{d,eq} = 0$ for venlafaxine and roxithromycin).

541 No significant correlations were found with the stereochemistry parameter nRB and $\log pK_a$
542 (Fig.S6). While previous studies positively correlated the sorption of cationic compounds with
543 pK_a ($r^2=0.5$) (Franco and Trapp, 2008), the narrow range of pK_a values covered in this study
544 prevented us from concluding on the significance of this indicator.

545 Interestingly, our analysis revealed a significant positive correlation only for Z500 between
546 $\log K_{d,eq}$ and $\log TPSA$, $\log vdWA$, McGowan's V_X (Fig. 6) and MV (Fig. S6a). The parameter
547 $TPSA$ was previously identified as sorption predictor only for neutral and negatively charged
548 compounds, although with a negative correlation (Sathyamoorthy and Ramsburg, 2013). $TPSA$
549 reflects the polarity of the organic chemical by accounting for the oxygen and nitrogen atoms,
550 and the significance of the correlation thus suggests (at least for thicker biofilm) the strong
551 contribution of the polar fraction on the sorption of positively charged compounds.

552 On the other hand, the positive correlation of $\log K_{d,eq}$ with $\log vdWA$, MV and V_X still suggests
553 a contribution of hydrophobicity in sorption of positively charged compounds in Z500 biofilm.
554 This finding is in line with previously established regressions for the prediction of distribution
555 coefficients based on van der Waals volume (Kamlet et al., 1998) or V_X (Abraham, 1993;
556 Abraham and Acree, 2010; Droge and Goss, 2013a,b,c) for neutral and ionized molecules.
557 Notably, McGowan's volume positively correlates with van der Waals volume (Zhao et al.,
558 2003), which is itself correlated to $vdWA$. Hence, both $vdWA$ and V_X provide an indication of
559 the influence of the molecular size in the cavity formation mechanism, through which solute
560 molecules can distribute to an organic phase at the expenses of (i.e., by replacing) water
561 molecules.

562 Considering the relevance of the correlation between $\log K_{d,eq}$ and V_X for Z500, an empirical
563 regression model (Eq. 8) was tested based on the equation previously proposed by Droge and
564 Goss (2013a,c) for sorption prediction of organic cations to soil organic matter:

565 $\log K_{d,eq} = a \cdot V_x / 100 + b \cdot NA_i + c$ (Eq. 8)

566 where $K_{d,eq}$ is expressed in $L\ kg^{-1}$ and NA_i indicates the number of hydrogen atoms bound to the
567 charged nitrogen moiety. The coefficients a , b and c were estimated by fitting Eq. 8 to
568 measured sorption coefficients. The comparison between predicted and measured $\log K_{d,eq}$ for
569 Z500 is shown in Fig. 6d ($a=0.35$; $b=0.44$; $c=1.49$). The regression ($r^2 = 0.59$) could only
570 partly describe sorption of cationic micropollutants in Z500 biofilms, yielding rather good $K_{d,eq}$
571 predictions (within factor 1.5 from measurements) for propranolol, clarithromycin,
572 erythromycin and roxythromycin. Potential improvement of sorption predictions may be
573 expected from the identification of correction factors for polar functional groups—an area
574 beyond the scope of this study due to the limited number of substances.

575 Overall, results from this assessment confirm the challenges in the identification of unique and
576 reliable sorption predictors for positively charged micropollutants in biofilm, as previously
577 recognized for other matrices (Kah and Brown, 2007; Franco and Trapp, 2008; Franco et al.,
578 2009; Sathyamoorthy and Ramsburg, 2013; Droge and Goss, 2013a,c; Bittermann et al., 2016).
579 Nevertheless, it should be highlighted that in this study sorption was consistently observed
580 only for positively charged compounds, indicating that electrostatic interaction with negatively
581 charged biomass surfaces play a major role for sorption in biofilms.

582 < **Figure 6** >

583

584 **5. Conclusions**

585 This study investigated the sorption and the diffusion of selected micropollutants in nitrifying
586 MBBR biofilms (thickness=50, 200, 500 μm) by means of targeted experiments and process
587 modelling, leading to the following conclusions:

- 588 • Sorption in biofilm occurred only for eight positively charged micropollutants (i.e.,

589 three macrolides, three beta-blockers and two psycho-active pharmaceuticals) out of 23
590 targeted substances. Electrostatic interaction with the negatively charged biomass
591 surfaces appears to play a major role in the sorption to biofilms.

592 • Values of the partitioning coefficient $K_{d,eq}$ increased with increasing biofilm thickness
593 for most of the sorbed compounds, being related to the increasing biofilm porosity and
594 thus the higher surface area accessible for sorption. Sorption equilibria were reached
595 within the duration of sorption experiments (4 h) for a number of compounds in 50 and
596 200 μm thick biofilms, but not in the thickest biofilm. Slower equilibrium in thick
597 biofilms ($\geq 500 \mu\text{m}$) is likely determined by the longer time required to diffuse in deeper
598 biofilm.

599 • Dimensionless effective diffusivity coefficients f for micropollutants (estimated for the
600 first time in wastewater treatment biofilms) were negatively correlated with biofilm
601 density, while showing an increase with increasing porosity. This indicates that
602 diffusive transport may be strongly limited by the higher biomass density (and the
603 lower porosity) of thinner biofilms.

604 • Significant positive correlations were observed between $\log K_{d,eq}$ and a limited number
605 of chemical properties of micropollutants (topological polar surface area, van der Waals
606 area and McGowan's volume) but not for all biofilm thicknesses, confirming the
607 challenges in the prediction of sorption in biofilms and other matrices for positively
608 charged compounds.

609

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839 ions and correlation with van der Waals volumes. *J. Chem. Inf. Comput. Sci.* 43, 1848–
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- 841

842 Tables

843 **Table 1.** Biofilm characteristics and input parameters used in the sorption and diffusion model
 844 in this study.

Parameter	Z50	Z200	Z500	Reference
Boundary layer thickness L_L (μm)	10	10	10	(Brockmann et al., 2008)
Dry biofilm mass per carrier (gTS m^{-2})	2.33	4.46	9.00	Measured
Biomass dry density in biofilm D_d (g cm^{-3})	1.17	1.05	1.05	Calculated
Porosity ε (%)	76	91	93	Calculated
Biofilm wet density ρ (kg m^{-3})	51	19	16	Calculated

845

846 **Table 1.** Sorption coefficients calculated according to final concentration at equilibrium ($K_{d,eq}$)
 847 and according to last measured aqueous concentration after 4 h of batch experiment ($K_{d,4h}$) for
 848 eight of the 22 spiked chemical compounds, exhibiting significant.

849

	Z50			Z200			Z500			Literature K_d (L g^{-1})
	$K_{d,eq}$ (L g^{-1})	$K_{d,4h}$ (L g^{-1})	Δ (%)	$K_{d,eq}$ (L g^{-1})	$K_{d,4h}$ (L g^{-1})	Δ (%)	$K_{d,eq}$ (L g^{-1})	$K_{d,4h}$ (L g^{-1})	Δ (%)	
Atenolol	1.12±2.21	0.26	77	1.10±0.34	0.68	38	4.72±0.73	0.95	80	(0.006) ¹ -1.9 ²
Metoprolol	0.08±0.01	0.08	3	0.19±0.06	0.16	15	0.27±0.02	0.15	44	<0.01 ³ -0.23 ⁴
Propranolol	0.50±0.04	0.54	-9	1.69±0.03	1.67	1	1.90±0.06	1.92	-1	0.2 ¹ -0.32 ³
Clarithromycin	0.42±0.11	0.34	20	0.41±0.02	0.39	4	10.90±0.25	5.63	48	0.26 ⁵ -1.2 ⁶
Erythromycin	0.33±0.07	0.34	-3	0.19±0.01	0.20	-4	10.68±2.09	6.13	43	0.31 ¹ -1 ⁷
Roxithromycin	0.00	0.00	/	0.85±0.13	1.05	-24	10.80±0.32	3.92	64	<0.1 ⁹ -0.5 ⁶
Citalopram	0.47±0.08	0.46	1	0.66±0.08	0.61	8	2.46±0.15	2.06	16	0.54 ²
Venlafaxine	0.00	0.00	/	0.12±0.05	0.09	25	0.14±0.06	0.12	16	<0.1 ²

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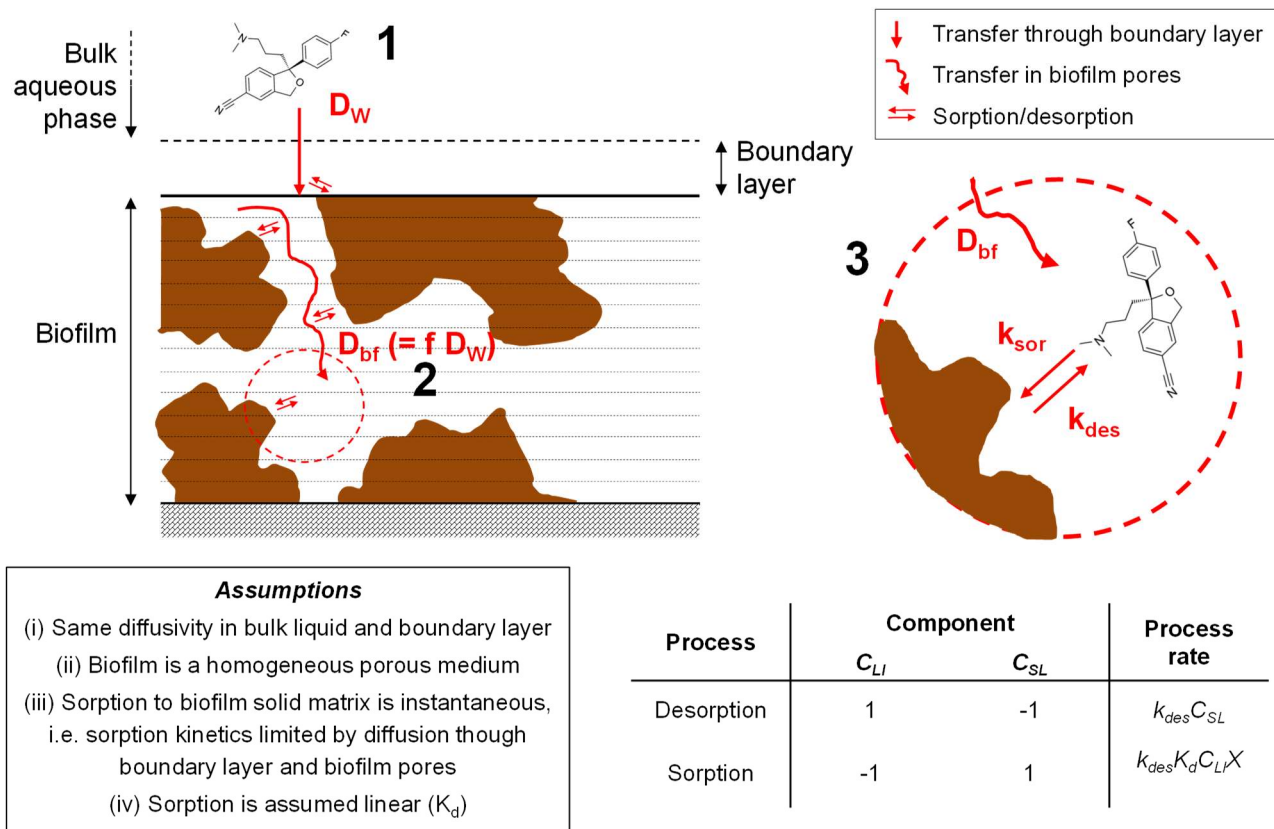
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¹ (Radjenović et al., 2009) for atenolol the lowest value was found in FS MBR; ² (Hörsing et al., 2011) for atenolol in secondary sludge; ³ (Maurer et al., 2007); ⁴(Sathyamoorthy et al., 2013); ⁵ (Göbel et al., 2005); ⁶ (Abegglen et al., 2009) in MBR; ⁷ (Xue et al., 2010) in activated sludge and MBR; ⁹ (Fernandez-Fontaina et al., 2012).

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856 Figures



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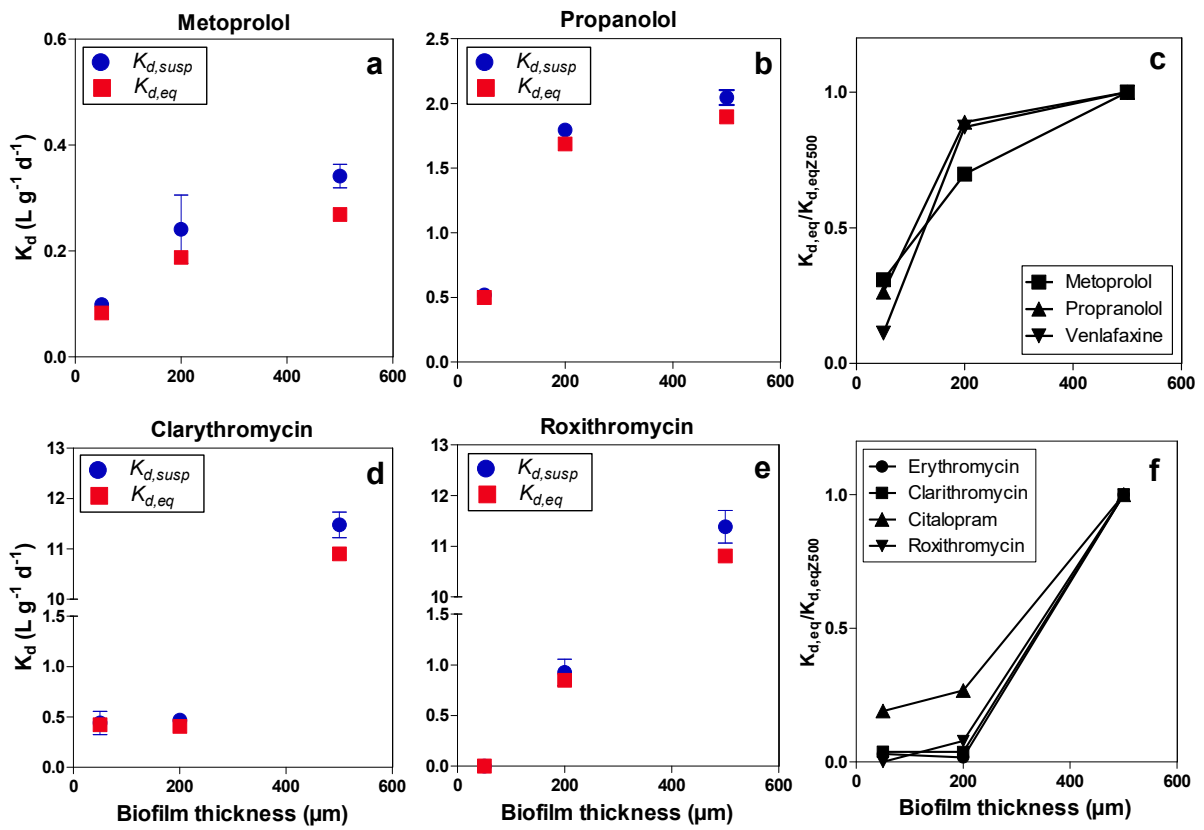
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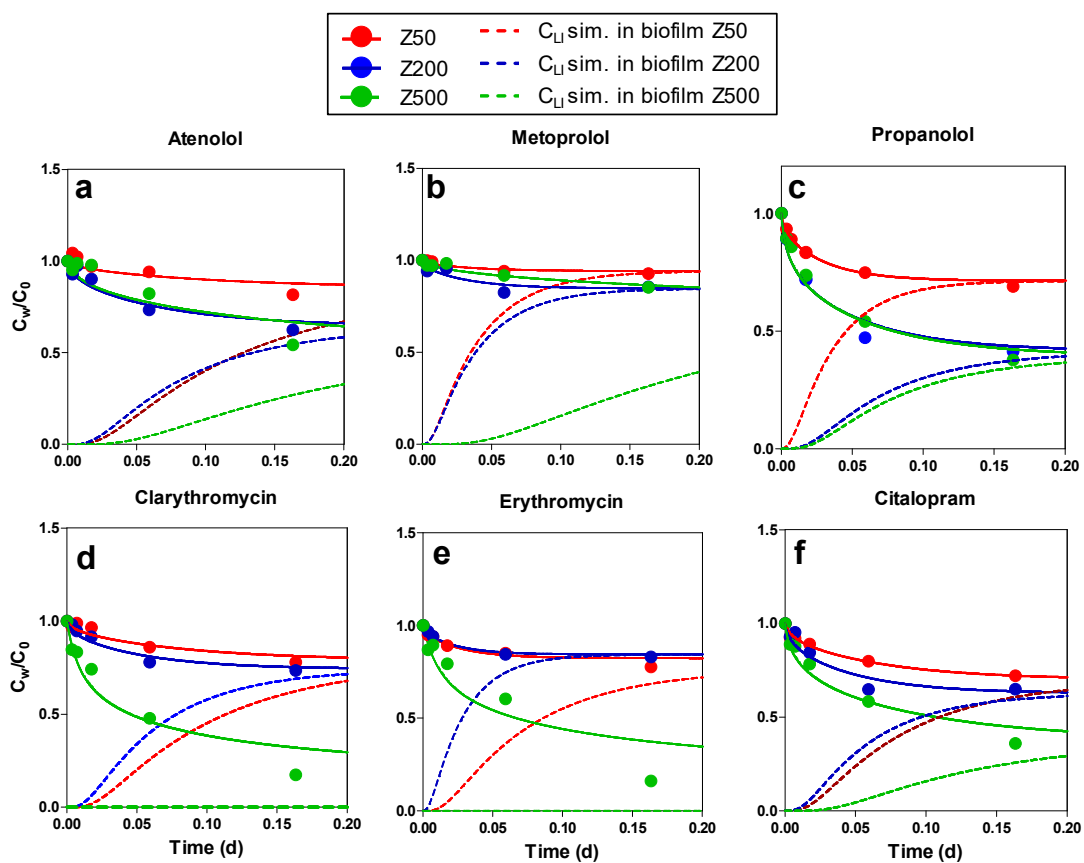
Figure 1. Conceptual model for diffusion and sorption of micropollutants into biofilms, including the consecutive steps required for partitioning onto biofilm solids (processes 1–3, see text), the assumptions considered in the model and the process matrix describing sorption and desorption kinetics ($X=X_{biomass}$).



862

863 **Fig. 2.** Comparison between sorption coefficient accounting for diffusion ($K_{d_Suspended}$) and
 864 not accounting for diffusion.

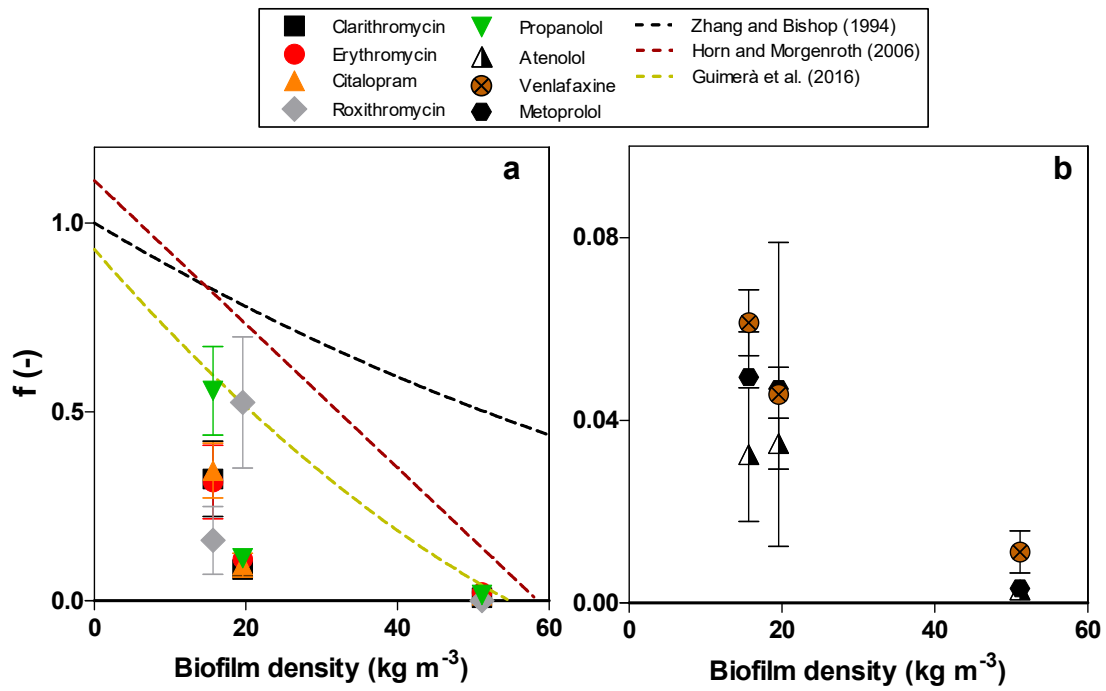
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867 **Fig. 3.** Measured (circles) and simulated (continuous line) aqueous concentrations in bulk
 868 aqueous phase and simulated concentrations in biofilm pores (dashed lines) of six selected
 869 chemicals compounds during batch experiments with Z50 (red), Z200 (blue) and Z500 (green)
 870 biofilms.

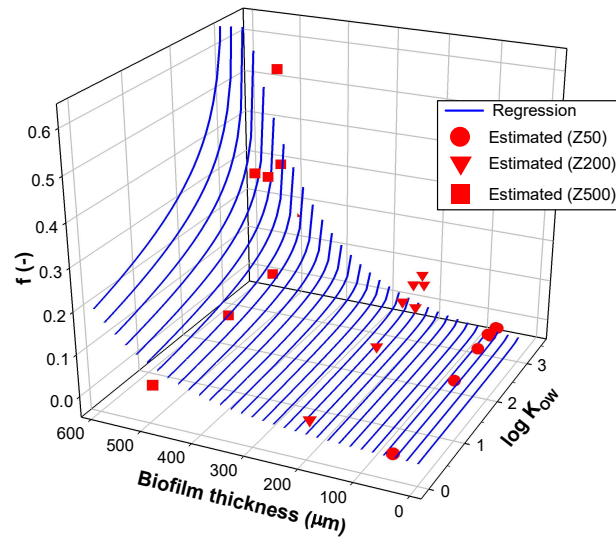
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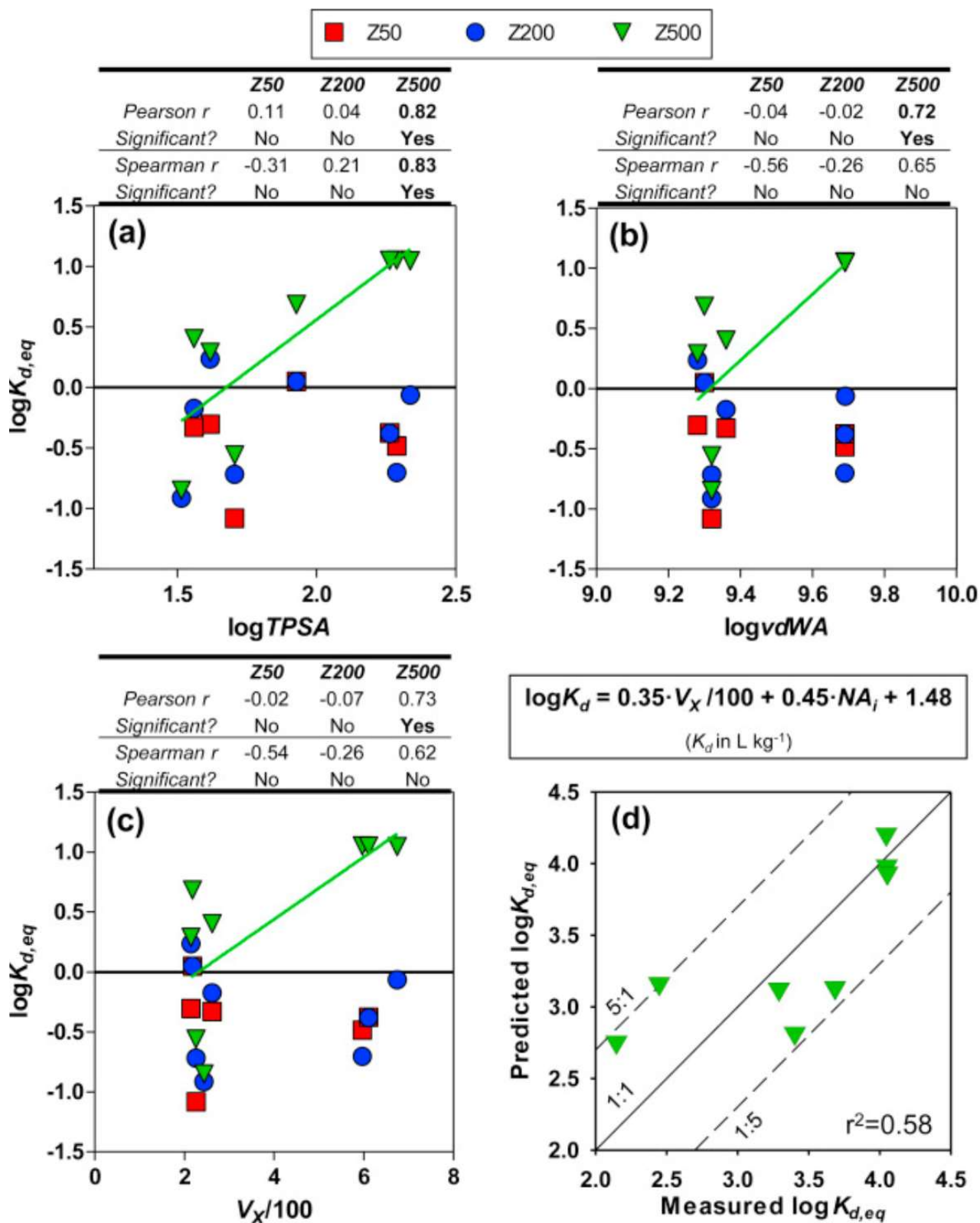
873 **Figure 4.** Estimated values of effective diffusivity for the three biofilm thickness and the

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875

876 **Fig. 5.** Plots of the empirical equation describing f as a function of biofilm thickness and
 877 $\log K_{ow}$, together with estimated f values (red symbols) in Z50, Z200 and Z500.



878

879 Fig. 6. Correlation analysis between $\log K_{d,eq}$ of the targeted micropollutants for the
 880 three biofilms (Z50, Z200, Z500) and physico-chemical descriptors: (a) $\log TPSA$;

881 (b) log vdWA; (c) McGowan's volume VX (divided by a factor of 100). Linear
882 regression lines were reported only for significant correlations. Based on the
883 correlation with VX/100, an empirical regression (Eq. (8)) was tested according to
884 Droge and Goss (2013a, c). The comparison between measurements and predictions
885 using Eq. (8) (in both cases, with $K_{d,eq}$ in $L\ kg^{-1}$) is presented in (d).