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Rejection of trace pharmaceutically active compounds present in municipal wastewaters using ceramic fine ultrafiltration membranes: effect of feed solution pH and fouling phenomena

Jorge Garcia-Ivars*[a], Javier Durá-María[b], Carlos Moscardó-Carreño[b], Carlos Carbonell-Alcaina[a], Maria-Isabel Alcaina-Miranda[a,c], Maria-Isabel Iborra-Clar[a,c]

[a]Research Institute for Industrial, Radiophysical and Environmental Safety (ISIRYM), Universitat Politècnica de València, C/Camino de Vera s/n, 46022 Valencia, Spain

[b]School of Industrial Engineering (ETSII), Universitat Politècnica de València, C/Camino de Vera s/n, 46022 Valencia, Spain

[c]Department of Chemical and Nuclear Engineering, Universitat Politècnica de València, C/Camino de Vera s/n, 46022 Valencia, Spain

Tel. +34 963879633

Fax. +34 963877639

Correspondence to: Jorge Garcia-Ivars (E-mail: *jorgariv@posgrado.upv.es*)

ABSTRACT

This study investigates the influence of feed solution pH and fouling on the rejection of ten selected pharmaceutically active compounds (PhACs) with different physicochemical characteristics (molecular weight, water solubility, log K_{OW}, pKa, dipole moment…) by three multichannel ceramic ultrafiltration membranes, ranging from 1 to 8 kDa, in order to improve their removal from water. For this purpose, the comparison between filtration of PhACs in deionised water (Feed I) and in real wastewater effluent (Feed II) was performed, demonstrating that the variation of pH and the formation of a foulant layer altered the separation mechanism and hence the rejection values of each PhAC varied. Higher rejections of most of the PhACs were higher at slightly alkaline pH, especially for anionic compounds in the filtration with

real wastewater. In these conditions, flux decline was more severe. The formed fouling layer onto the hydrophilic membrane surface acted as a secondary barrier for separation with different properties like hydrophobicity and charge. Electrostatic interactions were the main separation mechanism in the filtration of PhACs in deionised water, while the hydrophobic/hydrophilic interactions played a crucial role in the filtration experiments with real wastewater effluent. Thus, the reported results indicated that the rejection of pharmaceutically active compounds was strongly pH-dependent, except for hydrophilic neutral compounds (acetaminophen and caffeine), which showed a pH-independent behaviour with low rejection values.

KEYWORDS Ceramic fine ultrafiltration membranes; rejection efficiency; pharmaceutically active compounds; pH; fouling phenomena.

1. INTRODUCTION

Emerging contaminants have a great interest for institutions, public media and researchers due to the potential health risks associated with their release into the environment and their interactions with the landscape, human beings and wildlife species [1,2]. Several recent studies have demonstrated that emerging contaminants such as endocrine disrupting compounds (EDCs), pharmaceutically active compounds (PhACs), pesticides, disinfection by-products (DBPs), and personal care products (PCPs) are found at trace concentrations in surface waters and the toxicity of many of these compounds can potentially develop hazardous human, animal and ecological problems, depending on their nature and concentration [3-5]. Among the diversity of the emerging contaminants, the increasing use of PhACs leads to a growing occurrence of these organic compounds in wastewater and surface water, which makes them an

important environmental concern. These compounds are originated from veterinary applications and human usage and excretions (without being transformed or as metabolites), including personal hygiene products, hospital waste, therapeutic drugs, and waste from pharmaceutical industry [6]. PhACs have been detected directly or as their metabolites in surface water and effluents. As a direct consequence of their inherent biological activity, PhACs can cause unwanted adverse effects on non-target species after their release into the environment, including human/wildlife reproduction disorders and the appearance of antibiotic resistant bacteria [7,8]. These effects are related to the wide range of physicochemical properties of PhACs (including solubility, biodegradability and polarity), which favour their persistence in the environment, propensity for bioaccumulation in living organisms and capability to be transformed into products after various oxidative treatments [9]. Petrie et al. reported in 2015 that more than 200 different PhACs have been found in river waters worldwide [10].

Conventional wastewater treatment plants (WWTPs) are not specially designed to remove PhACs because they usually receive and treat a large spectrum of pollutants from industrial, domestic and farming wastewater. Due to both the diverse physicochemical properties and low concentrations levels of PhACs (from $ng \cdot L^{-1}$ to $\mu g \cdot L^{-1}$), they are not completely eliminated during treatment processes, obtaining complex outlets which are discharged into rivers [4]. In addition, the concentration of some PhACs has increased during the treatment in WWTPs as a consequence of their transformation into conjugates [11]. Such limitations have led to explore new technological alternatives, such as advanced oxidation processes, activated carbon adsorption or membrane filtration [6,12,13]. Pressure-driven membrane separation processes such as reverse osmosis (RO), nanofiltration (NF), ultrafiltration (UF), and

77 microfiltration (MF) applied at pilot- and full-scale installations are being successfully
78 used either separately or as a combination of membrane techniques in wastewater
79 reclamation to achieve a high quality product by efficiently removing bacteria, viruses,
80 dissolved solids, organic micropollutants, proteins, sugars or inorganic ions [14].
81 Several researchers have investigated the application of membrane technology in the
82 removal of PhACs, especially in synthetic model waters where the target compounds
83 were spiked [11]. NF and RO techniques have been successfully applied to remove
84 PhACs and PCPs from raw wastewaters and natural waters as well, in which the
85 influence of solute interactions between organic matter and PhACs on the membrane
86 performance is a key parameter [12,14,15]. Generally, UF, NF and RO membranes used
87 in PhACs removal are made of polymeric materials and, to a lesser extent, of polymeric
88 membranes modified with inorganic particles [16,17]. Despite their use in
89 pharmaceutical industry, from our knowledge only few studies investigated the
90 performance of ceramic membranes to treat ground and surface waters with PhACs,
91 especially in MF [18]. Thus, it is noteworthy to highlight that the main novelty of this
92 work is the implementation of ceramic fine ultrafiltration membranes to remove PhACs
93 from municipal and industrial wastewaters. Ceramic membranes were selected due to
94 their thermal stability, superior chemical and biological resistance and adaptability for a
95 wide pH range, even though they were more expensive than polymeric membranes
96 [19,20].
97
98 In the present work, the performance of different multichannel ceramic membranes
99 within the fine ultrafiltration range (between 1 and 8 kDa) was studied in terms of
100 permeate flux, membrane fouling and rejection index. A novel aspect of this work is to
101 study the influence of the feed solution pH (ranging from 6 to 8) and the molecular

weight cut-off (MWCO) of the selected membranes on their performances. Experiments were carried out using a cross-flow membrane filtration unit with ten selected PhACs with diverse physicochemical characteristics added in deionised water (Feed I) and in a WWTP secondary effluent (Feed II).

2. EXPERIMENTAL

2.1.WWTP secondary effluent samples

WWTP secondary effluent samples were donated by Carraixet WWTP, located in the region of Valencia (Spain). The characterisation of such samples was performed according to Standard Methods [21]. Their physicochemical characteristics are summarised in Table 1. This effluent is slightly alkaline with a high electrical conductivity, turbidity and a moderate COD value compared with the wastewater used in similar studies by other researchers [22].

2.2.Chemicals and Materials

Ten PhACs (acetaminophen, caffeine, diazepam, diclofenac, erythromycin, ibuprofen, naproxen, sulfamethoxazole, triclosan, and trimethoprim) were examined. All of them were high reagent purity grade (≥ 99 %) and purchased from Sigma-Aldrich (Germany). These PhACs were selected due to their occurrence and persistence in effluents from WWTPs and surface water at the Spanish Mediterranean area of Valencia [23,24]. Their main physicochemical properties are summarised in Table 2. These organic compounds have similar molecular weight (except erythromycin) and distinguishing features including water solubility, molar volume, log K_{OW}, pKa, and dipole moment, which make them interesting to be compared. The pH of feed solutions was adjusted using 0.1

126 M HCl/NaOH solutions before starting filtration experiments and was controlled using a
127 Crison pH meter. Both chemicals (HCl and NaOH) were obtained of reagent grade from
128 Panreac (Spain). Deionised water was used throughout this study.
129
130 Three seven-channel ceramic UF membranes (INSIDE CéRAM^{TM}, supplied by TAMI
131 Industries, France) with a nominal pore size of 1, 5 and 8 kDa were used in order to
132 represent a wide range of nominal MWCO within the fine UF range and to compare
133 their effectiveness in PhACs removal. These membranes consisted of an active layer of
134 TiO_2 with an effective area of 132 cm^2 and their dimensions were 25 cm long with an
135 external diameter of 1 cm.
136
137 *2.3.Experimental procedure*

138 The filtration experiments were conducted in a standard cross-flow ultrafiltration set up
139 that is schematically presented in Fig. 1. This cross-flow system was equipped with a
140 temperature-controlled feed tank with a capacity of 20 L, *a pH meter incorporated in*
141 *the tank*, a pre-filter to protect the pump of undesired pollution, a variable speed
142 volumetric pump to adjust the feed flow (measured by a flow meter), and two
143 manometers (P1 and P2, ranging from 0 to 6 bar) placed at the inlet and outlet streams
144 of the membrane cell to adjust and control the transmembrane pressure. Finally, a scale
145 with an accuracy of ± 0.001 g was used to gravimetrically measure the permeate flux.
146 Before the filtration experiments began, water permeability (*K*) for each membrane
147 using deionised water was calculated. These experiments were performed in the
148 aforementioned standard cross-flow ultrafiltration set up at different transmembrane
149 pressures (ΔP) ranging from 0.5 to 3 bar at a constant flow rate of 300 $L \cdot h^{-1}$. The water

150 permeability (*K*) was 38.2 ± 2.2, 40.4 ± 2.6, and 60.7 ± 3.6 $L \cdot m^{-2} \cdot h^{-1} \cdot bar^{-1}$ for the
151 ceramic ultrafiltration membranes of 1, 5 and 8 kDa, respectively.
152
153 The operating procedure was constituted by three different steps: firstly, the water flux
154 of ceramic membranes was stabilised at a constant transmembrane pressure of 3 bar and
155 300 $L \cdot h^{-1}$ of flow rate for at least 30 min until the difference between the measurements
156 of consecutive permeate fluxes was lower than 2 % [25]. In *the* next stage, the filtration
157 of different feed solutions containing the target PhACs was carried out at 300 ± 5 $L \cdot h^{-1}$,
158 2 bar and 25 ± 1 ºC for 3 h. Two different feed solutions were separately used in this
159 stage: the first one was prepared with an initial concentration of 1000 $ng \cdot L^{-1}$ of
160 acetaminophen, ibuprofen, and sulfamethoxazole, and 300 $ng \cdot L^{-1}$ of caffeine, diazepam,
161 diclofenac, erythromycin, naproxen, triclosan, and trimethoprim spiked in deionised
162 water (Feed I), and the second one consisted of the same PhAC concentrations spiked in
163 a WWTP secondary effluent (Feed II). Filtration experiments were performed in total
164 recirculation mode (permeate samples were returned back to the feed tank to keep
165 constant the feed concentration). In order to qualitatively and quantitatively assess the
166 role of pH on the removal efficiencies, three pH levels (from pH 6 to 8) were tested on
167 the basis of the literature about the removal of PhACs from WWTP secondary effluents
168 and surface waters using membrane technologies [12,14,26,27].
169
170 During filtration experiments, the permeate flux (*Jp*, $L \cdot m^{-2} \cdot h^{-1}$) was measured using the
171 gravimetric method at regular time intervals and was determined by Eq. (1):

172 $$Jp = \frac{m}{\rho \cdot A_m \cdot t} \qquad \text{Eq. (1)}$$

where m is the mass of permeate water (g), ρ is the density of water at the operating temperature ($g \cdot L^{-1}$), A_m is the effective membrane area (m^2), and t is the filtration time (h).

The rejection index (R, %) was calculated as follows:

$$R(\%) = \frac{C_f - C_p}{C_f} \cdot 100 \quad \text{Eq. (2)}$$

where C_p is the concentration of each PhAC in the permeate stream ($ng \cdot L^{-1}$), and C_f is the concentration of the same PhAC in the feed solution ($ng \cdot L^{-1}$).

Once the filtration with the corresponding feed solution was finished, ceramic membranes were rinsed in cross-flow mode with deionised water to remove the reversible fouling from the membrane (R_{rev}, m^{-1}). The duration of this step was 30 minutes at 300 $L \cdot h^{-1}$, 1 bar and 25 ºC. In order to evaluate the influence of fouling phenomena on the flux decline and on the separation of PhACs during the filtration experiments, total hydraulic resistance (R_T, m^{-1}) can be determined from the Darcy's law, which correlates such a resistance with J_p and the transmembrane pressure (ΔP, bar):

$$R_T = \frac{\Delta P}{\mu \cdot J_p} \quad \text{Eq. (3)}$$

where μ is the viscosity of the feed solution (Pa·s).

Therefore, this total hydraulic resistance comprises the different resistances that take place during the filtration process and can be defined as the sum of the membrane intrinsic resistance (obtained from the water permeability, K) and the hydraulic resistance of the membrane after each step (fouling and rinsing) as follows:

197 $$R_m = \frac{1}{\mu \cdot K}$$ Eq. (4)

198 $$R_{irr} = \frac{\Delta P}{\mu \cdot J_r} - R_m$$ Eq. (5)

199 $$R_{rev} = \frac{\Delta P}{\mu \cdot J_f} - R_m - R_{irr}$$ Eq. (6)

200 $$R_T = R_m + R_{rev} + R_{irr}$$ Eq. (7)

201 where R_m is the membrane intrinsic resistance (m^{-1}), J_r is the permeate flux during the
202 rinsing process ($L \cdot m^{-2} \cdot h^{-1}$), R_{irr} is the irreversible resistance due to fouling and can be
203 defined as the permeate flux loss that can be recovered by chemical cleaning or even
204 cannot be recovered (m^{-1}), J_f is the permeate flux at the end of the filtration experiment
205 with each feed solution ($L \cdot m^{-2} \cdot h^{-1}$), and R_{rev} is the reversible resistance caused by
206 concentration-polarisation phenomenon and the filtration cake and can be defined as the
207 permeate flux loss that can be recovered by physical cleaning (m^{-1}) [28,29].

208

209 In order to quantitatively assess the adsorption of PhACs during filtration of both feed
210 solutions, mass balances based on the concentration of each PhAC in the feed, permeate
211 and retentate streams were calculated using Eq. (8). Adsorbed mass (m_{ads}, $ng \cdot m^{-2}$) can
212 be defined as the amount of PhAC adsorbed per unit area onto the membrane surface
213 and into the pores:

214 $$m_{ads} = \frac{C_f \cdot V_f - C_p \cdot V_p - C_r \cdot V_r}{A_m}$$ *Eq. (8)*

215 where C_r is the concentration of each PhAC in the retentate stream ($ng \cdot L^{-1}$), and V_f, V_p,
216 and V_r *(L)* are the volume of the same PhAC in the feed, permeate, and retentate stream,
217 respectively.

218

The adsorption percentage (M_{ads}, *%*) can be defined as follows:

$$M_{ads} = \frac{C_f \cdot V_f - C_p \cdot V_p - C_r \cdot V_r}{C_f \cdot V_f} \qquad \textit{Eq. (9)}$$

2.4.Analytical methods

Concentrations of PhACs in permeate, retentate and feed samples were determined by High-Performance Liquid Chromatography tandem-mass spectrometry (HPLC-MS/MS) method. An Agilent Technologies 1260 Infinity Ultra High-performance Liquid Chromatograph coupled to an Agilent Technologies 6410 Triple Quadrupole Mass Spectrometer with an electrospray Turbo V ionisation source and a C18 column (Kinetex, 1.7 μm, 100 Å, 50 x 2.10 mm) from Phenomenex (France) were used. The different PhACs concentrations were determined in both positive and negative ionisation modes, depending on the PhAC measured. Quantified and qualified transitions were optimised for each PhACs by selected reaction monitoring (SRM), which were previously described [13,23,30].

The HPLC-MS/MS method was validated by determining seven-point calibration curves using standard solutions (which were injected in triplicate), where concentrations varied from the limit of quantification (*LOQ*) of each PhAC to 30 $\mu g \cdot L^{-1}$. The method's integrity was evaluated by assessing the linearity, *LOQ* and limits of detection (*LOD*). The linearity of the method was evaluated by the linear correlation coefficient (R^2), which was higher than 95 % for all the PhACs tested. The *LOQ* was calculated as the lowest amount of analyte added to the water sample that produced a peak signal of 10 times the background noise in the chromatograph, while the *LOD* was expressed by the

242 equation *LOD = LOQ/3*. The values of *LOD* and *LOQ* for each PhAC are displayed in
243 Table 3.
244
245 ## 3. RESULTS AND DISCUSSION
246 ### *3.1. Filtration of PhACs in deionised water (Feed I)*
247 The feed solution pH is an important parameter in the rejection of PhACs during the
248 membrane separation process, regardless of whether model solutions with deionised
249 water or real wastewater effluent were used. Like other emerging contaminants, the
250 rejection indexes of PhACs vary with respect to their physicochemical properties, such
251 as structure, molecular weight and dissociation constant (pKa). This latter property is
252 linked to the strength of its inherent bonds and determines its ionic state. Due to the fact
253 that MWCO of all the membranes tested is higher than the molecular weight of these
254 PhACs, the electric charge property of each PhAC is an important factor that may affect
255 the performance of the separation process. A change in feed solution pH can
256 significantly vary the behaviour of a PhAC. One PhAC will be negatively charged at
257 higher pH values than its pKa value; otherwise this PhAC will be neutral or positively
258 charged or even a mixture of both. So, the rejection of PhACs is strongly dependent on
259 the feed solution pH [12]. Fig. 2 shows the evolution of $Jp(t)/J_0$ over filtration time for
260 each ceramic membrane using Feed I. Despite the pore size of the membrane, the
261 permeate flux slightly decreased with increasing pH, but this decline was lower than 10
262 % of the initial permeate flux (J_0). For this reason, the observed flux decline could be
263 considered as insignificant; indicating that the effect of the adsorption and deposition of
264 PhACs on the surface had no effect. This could be explained by the very low PhACs
265 concentration used in this study (from 0.3 to 1 $\mu g \cdot L^{-1}$), which are too low to be
266 influential. Comparing Fig. 2a, 2b, and 2c, permeate flux was higher for ceramic

membranes with larger MWCO at the same pH conditions. This is caused by the lower resistance offered by the membrane with larger pores for the solution to pass through it.

The rejection index of each selected PhACs during the filtration experiments of Feed I with the three ceramic membranes at different pH conditions is depicted in Fig. 3. It can be seen that higher retention values were obtained using membranes with smaller nominal pore size, which indicates that membranes with low MWCO (close to NF range) are more selective in PhACs separation. In the same way, low retention values were achieved for almost all the PhACs tested, except for erythromycin, diclofenac, ibuprofen, naproxen, and sulfamethoxazole (with rejection indexes higher than 50 % using ceramic 1 kDa and 5 kDa membranes). The separation mechanism that prevails in UF process is generally based on the size exclusion or sieving effect, where solutes are solely separated according to their dimensions. This point of view is widely shared by several researchers but could be considered as insufficient, especially in this case, where the molecular weight of the PhACs is much smaller than the MWCO of an UF membrane. For this reason, the separation mechanism is not a simple sieve effect and cannot be considered as a simple filtration process, because the existing solute-solute and solute-membrane surface interactions (hydrophobic and electrostatic interactions) can play an important role in the retention of PhACs from different waters [12,31]. As was explained before, speciation of PhACs depends on their characteristic pKa value and the feed solution pH and then, it has a significant impact on their rejection. Several researchers demonstrated in their studies that the isoelectric point of the selected multichannel ceramic membranes was 6.2 ± 0.1, resulting in membranes with slightly positive charge at pH 6 and negative at pH 7 and 8 [32-34]. At pH 6, membranes showed higher rejections for erythromycin and trimethoprim compared to the other

PhACs for different reasons. Erythromycin existed as a neutral species at these conditions, while trimethoprim was positively charged. Despite being a non-ionic PhAC at the pH conditions tested (with a pKa value of 8.9), the high dipole moment of erythromycin (above 3 D) could be sufficient to induce an electrostatic attraction between the membrane surface and the polar centers of the molecule. These electrostatic interactions combined with the size exclusion due to the similarity of the molecular size of erythromycin with the nominal pore size of 1 kDa membrane could favour its retention [35,36]. In the case of 5 and 8 kDa membranes, the high rejection obtained for erythromycin could be explained taking into account that both membranes were hydrophilic in nature and changed their surface charge from positive to negative at pHs 7 and 8, while erythromycin was hydrophobic ($log\ K_{OW} > 2$) and remained neutral during all the experiments. The neutral charge of erythromycin could favour its adsorption on the negatively charged membrane surface because of electrostatic attraction, despite the different hydrophilicity between this compound and ceramic 5 and 8 kDa membranes. Other authors have noted that the increasing rejection of erythromycin at basic pHs (< pKa) might be caused by the limited solubility of erythromycin in basic aqueous solutions, in which molecules may precipitate out of the aqueous feed solution and be adsorbed on the membrane surface [37]. For trimethoprim, its rejection value is higher than the other compounds at pH 6 mainly due to a weak but important electrostatic repulsion between the ceramic surface and trimethoprim, at which both the membrane and PhAC were positively charged. The charge of the ceramic membranes changed with increasing pH value from positive to negative, whereas trimethoprim was neutral at pH 7 and was negative at pH 8. At pH 7, such changes resulted in a significant decrease in its rejection index because both electrostatic and hydrophobic interactions (since trimethoprim was neutral and

hydrophilic at these conditions) were hindered. Due to the electrostatic repulsion between the molecules of trimethoprim and the ceramic surface, an increase in the rejection of trimethoprim was observed at pH 8, at which both had negative charge. Therefore, the rejection of trimethoprim had a similar behaviour at pH 6 and 8 when these multichannel ceramic membranes were used.

The retention of PhACs is significantly greater at pH 8, especially for diclofenac, erythromycin, ibuprofen, naproxen, sulfamethoxazole, and trimethoprim. Despite their low molecular weight, rejection indexes of diclofenac, ibuprofen, naproxen, and sulfamethoxazole were higher with increasing pH values, due to the fact that these anionic compounds (pKa < 6) were electrostatically repulsed by the negatively charged surface. This increase in their rejection values indicated that the electrostatic repulsion between these anionic PhACs and the membrane surface was the predominant separation mechanism. These strong repulsive forces prevented their adsorption and/or diffusion through the membrane [38]. This statement was corroborated by other researchers, which demonstrated that the membrane adsorption of uncharged PhACs is higher than the same PhACs but with negative charge (such as ibuprofen) [39,40]. Moreover, this increasing trend was observed in the retention of diazepam, obtaining lower rejection values than the aforementioned organic compounds although diazepam had similar characteristics to them. Similar observation was also found in the retention of triclosan, where the rejection index was higher with increasing pH until a highest value obtained at pH 8, once the feed solution pH exceeded its characteristic pKa value [41]. Other PhACs such as acetaminophen and caffeine showed a pH-independent behaviour, presenting similar rejections at different pHs using the same ceramic membrane. Both compounds were neutral and hydrophilic during the filtration

experiments, indicating that the electrostatic interactions as well as the hydrophilic/hydrophobic affinity were not the predominant separation mechanisms. Other researchers observed similar pH-independent behaviour of carbamazepine and acetaminophen using different membrane technologies [39,42].

3.2. Filtration of PhACs in a WWTP secondary effluent (Feed II)

In this section, the membrane performance during the filtration of Feed II (the selected PhACs added in a WTTP secondary effluent) is discussed in terms of flux decline, retention of PhACs, and effect of fouling phenomena. Fig. 4 shows the evolution of $J_p(t)/J_0$ at different feed solution pHs as a function of time for each ceramic membrane. It can be observed that the flux decline was higher at pH 8 (where $Jp(t)/J_0$ is 42.8, 34.2 and 33.1 % for ceramic 8 kDa membrane at pH 6, 7 and 8, respectively), principally due to the adsorption and deposition of effluent organic matter (EfOM) onto the membrane surface. J_p rapidly decreased at low time scales in which the fast accumulation of retained solute particles from the WWTP secondary effluent occurred on the surface and within the membrane pores. After that period, a gradual but slow flux decline took place, reaching an almost constant value of J_p when the equilibrium between the attachment and detachment of foulants on the membrane was achieved [43]. Based on the effluent water quality and the results obtained in the previous section, the observed flux decline could be attributed to fouling phenomena by EfOM. The ceramic UF membranes used in this study are hydrophilic in nature, contrarily to the mostly hydrophobic EfOM presented in Feed II [44]. In addition, the inherent rougher surface of these membranes could favour the entrapment of solute molecules. Several researchers have demonstrated that ceramic membranes have rougher surfaces than polymeric membranes in the same MWCO range [28]. During the filtration

experiments, a foulant layer could be formed by adsorbed organic and inorganic compounds from Feed II onto the ceramic UF membrane and might act as a second barrier for separation. This fouling layer formed on the membrane surface is a hydrophobic and negatively charged layer, which reduces both the porosity and pore size of the ceramic membrane principally because both complete and intermediate pore blocking occurred during the first stages of the filtration [41,45]. As a result, the rejection values of some PhACs could be improved compared to those obtained for clean membranes [46]. Mousaab et al. confirmed the modification of the removal efficiencies of different PhACs (such as diclofenac, naproxen, ketoprofen and codeine) in the presence of biomass and organic solutes in an ultrafiltration membrane system coupled with biofilm biological reactor [47].

Due to the complexity of the different fouling mechanisms, the flux decline during filtration experiments using Feed II was investigated by calculating R_m, R_{rev}, R_{irr}, and R_T. Their results are shown in Fig. 5, where the membrane with the highest water permeability showed the lowest R_m value (which was the ceramic 8 kDa membrane) and vice-versa. All the R_T values are much higher for the membranes at pH 8 than at the other conditions tested, which indicates that ceramic membranes suffered more severe fouling at pH 8. The highest values of R_{irr} were also obtained for all the ceramic membranes at that pH, at which the strongest attachment of organic matter occurs. This fact is confirmed by the flux decline displayed in Fig. 4, where the highest irreversible fouling resistance and flux decline are remarkably for ceramic 8 kDa membrane (see Fig. 4c and 5c). Therefore, the effect of membrane fouling on PhAC rejection can be considered as relevant and thereby, the nominal pore size of membranes could affect the

391 extent of fouling, where more severe fouling (irreversible fouling) was observed in
392 membranes with larger pore sizes [39,41].
393
394 The rejection indexes of selected PhACs for each pH tested during the filtration of Feed
395 II are illustrated in Fig. 3. Higher PhACs rejections were obtained using real wastewater
396 effluents (Feed II) than those obtained when PhACs were spiked in deionised water
397 (Feed I). This fact may be mainly because of the presence of EfOM (mostly
398 hydrophobic) in the membrane structure, especially adsorbed on the surface due to
399 fouling phenomena (as was explained before). The hydrophobic/adsorptive separation
400 mechanism became more important in the rejection of PhACs due to the presence of
401 organic matter in the feed solution and the hydrophobic nature of the fouling layer
402 formed on the membrane surface. As a result, the degree of PhACs rejection depended
403 on both the pH and ionic strength of the aqueous feed solution, and also on the presence
404 of organic matter. Some PhACs became charged at different pH conditions (as was
405 indicated in the previous section) and could be adsorbed onto colloids, trapped by
406 organic matter or associated with cations in the feed solution [12]. Such rejections could
407 be explained by two mechanisms: EfOM fouling and solute-solute interactions. The
408 accumulation of organic matter on the membrane surface during filtration (caused by
409 EfOM fouling) might act as an additional secondary barrier that could modify the
410 separation mechanism of PhACs. This supplementary filtration layer (or foulant layer)
411 was generally hydrophobic and negatively charged, which contrasted with the
412 hydrophilic ceramic membrane. In such conditions, the rejection of some PhACs could
413 increase by the repulsion between the negative charge of the additional foulant layer and
414 the negatively charged PhACs and also by the hydrophobic interactions between the
415 foulant layer and PhACs. In the same way, solute-solute interactions in effluent matrix

had a relevant role on PhACs rejections. The association of PhACs with organic macromolecules in the effluent led to form EfOM-compounds complexes that could be the result of hydrogen bonding and electrostatic attraction between the polar moieties of PhAC molecules and the phenolic or carboxylic groups of the humic-like substances. These complexes could be rejected by sieving effect or charge repulsion between them and the membrane surface [48-50]. This new scenario can be seen in Fig. 3. When the rejection values in Feed I are compared with those obtained during filtration of Feed II, the presence of organic matter and the EfOM-compound complexation improve the rejection indexes of most of the PhACs, especially in the rejection of neutral and negative compounds. Nevertheless, sulfamethoxazole showed the opposite behaviour, presenting lower rejection values in the presence of Feed II. This interesting fact was observed by other researchers using nanofiltration membranes, who demonstrated that the reduction in the rejection value could be provoked by the inherent high dipole moment of sulfamethoxazole (7.366 D), which electrostatically attracted the molecule to the membrane pores to facilitate its diffusion and permeate in an oriented way [36,41]. This proves that polarity of PhACs could have a more important influence on rejection in real wastewaters or effluents than in model solutions with pure water [50].

With respect to PhACs separation during filtration of Feed II, PhACs rejections are higher for membranes with smaller nominal pore size (retention values were higher for ceramic 1 kDa membrane than for both membranes with MWCO of 5 kDa and 8 kDa). The influence of feed solution pH on PhACs rejection is clearly visible, obtaining higher rejection values with increasing pH conditions, except for trimethoprim, acetaminophen and caffeine. This could be related to the hydrophilic character of these compounds at the studied pH range. The hydrophobicity of PhACs is typically

441 evaluated using the octanol-water partition coefficient of a compound (K_{OW}) or its
442 logarithm ($log\ K_{OW}$) that can be used as a descriptor of the sorption potential and
443 bioaccumulation of a compound in the aquatic environment. $Log\ K_{OW}$ is generally used
444 for uncharged (neutral) compounds [51]. In this study, acetaminophen, caffeine and
445 erythromycin existed as neutral species during all the experiments. By contrast,
446 hydrophobicity and solubility of a compound changes as a function of pH, especially in
447 a pH range around the pKa value. In these cases, solute hydrophobicity is predicted
448 using $log\ D_{OW}$ (a pH-corrected value of $log\ K_{OW}$) that considers the ratio between the
449 ionised and unionised form of the compound at a specific pH value. $Log\ K_{OW}$ and log
450 D_{OW} are the same for non-ionisable compounds. A compound can be considered as
451 hydrophobic when its characteristic $log\ D_{OW}$ (or $log\ K_{OW}$) is higher than 2, whereas the
452 same compound is hydrophilic when it has a $log\ D_{OW}$ (or $log\ K_{OW}$) value below 2 [26].
453 In order to properly analyse the rejection of PhACs that were in ionic state within the
454 studied pH range, Table 4 represents the $log\ D_{OW}$ values of the PhACs that are not
455 neutral (pKa < feed solution pH) at the tested pH conditions. At pH 6, trimethoprim was
456 positively charged and highly hydrophilic ($log\ D_{OW}$: 0.27). At these conditions, the
457 effect of fouling on the ceramic membranes was less relevant compared to the fouling
458 resistances obtained at higher pH conditions and two opposite effects affected the
459 rejection efficiencies of this compound: the electrostatic attraction between the formed
460 foulant layer on the membrane surface (which is negatively charged in general) and the
461 cationic compound, and the difference between the hydrophilic compound (with low
462 sorption potential, $log\ D_{OW} < 2$, see Table 4) and the hydrophobic foulant layer. In this
463 case, the electrostatic attraction became the main separation mechanism and could lead
464 to an accumulation of molecules of trimethoprim at the vicinities of the formed foulant
465 layer, being adsorbed and increasing its rejection. At pH 7, such electrostatic attraction

forces did not exist because trimethoprim was a neutral species and hence, hydrophilic non-ionic trimethoprim was not adsorbed on the foulant layer. Also, uncharged trimethoprim had a smaller molecular weight than the nominal pore size of the ceramic membranes (size exclusion was not the major rejection mechanism), passing through the membrane matrix [52]. Although trimethoprim was less hydrophilic at pH 8 (its log D_{OW} are higher with increasing pH), charge repulsion was the dominant mechanism to reject this compound because both trimethoprim and foulant layer are negatively charged. For acetaminophen and caffeine, their rejection values slightly increased in Feed II compared to Feed I. However, the stable rejection profile of both compounds was relatively pH-independent due to their neutral form and high hydrophilicity. Both acetaminophen and caffeine had a *log* K_{OW} value lower than 0.5, indicating their high hydrophilic character. Therefore, the formation of a foulant layer on the membrane surface barely altered the rejection values of such compounds regardless of the membrane used. Sheng et al. confirmed that acetaminophen and caffeine showed the same behaviour in ultrafiltration experiments with real wastewater effluent [53]; whereas Mahlangu and colleagues demonstrated that the presence of a foulant layer (colloidal and inorganic molecules as foulants) on the membrane surface did not alter the rejection of caffeine using a polyamide NF-270 membrane [54].

High rejection results (> 70 %) were observed for diclofenac (75.9%), diazepam (72.6 %), erythromycin (85.4 %), and triclosan (72.9 %) during filtration experiments using ceramic 1 kDa membrane at pH 8. As explained in the previous section, the high rejection of erythromycin could be inferred as a combination of adsorption and electrostatic attraction between the foulant layer and this compound. This was probably due to its higher molecular weight (733.94 $g \cdot mol^{-1}$), hydrophobicity (*log* K_{OW}: 3.06),

491 neutral charge, and high dipole moment (3.99 D). In spite of not being the main
492 separation mechanism for erythromycin, size exclusion became more important under
493 these conditions due to the formation of EfOM-erythromycin complexes, which led to
494 an increase in the size of such complexes and thus, the highest rejection of all the
495 PhACs was achieved. In the case of the rejection of triclosan, its pKa value was
496 reported to be 7.8, indicating that this compound would exist in both neutral and ionised
497 forms on the operating pH conditions of the most WWTPs, which are within the range
498 of pH 8 approximately. The formation of the negative species of the triclosan at pH 8
499 could result in an increase in the rejection values of this compound due to the
500 electrostatic repulsion between this ionic triclosan and the negatively charged
501 membrane surface (as explained in the previous section). However, the intrinsic
502 hydrophobic nature of triclosan (with $log\ D_{OW} > 2$, see Table 4) could significantly
503 increase its rejection values during the filtration experiments with real wastewater
504 effluents, regardless of its neutral or ionic state. According to several authors, PhACs
505 with high $log\ D_{OW}$ (> 4.5) have a high sorption potential and could be easily adsorbed
506 on hydrophobic surfaces, such as hydrophobic polymeric membranes or even the
507 biofilm and foulant layer formed onto the membrane surface caused by fouling
508 phenomena [55]. This could explain the high retention of triclosan at all the tested pH
509 conditions, especially at pH 8, once its characteristic pKa value (7.8) was exceeded
510 [41,56]. Similar observation was also found for diclofenac, where its high rejection
511 values may be related to its high characteristic $log\ K_{OW}$ and $log\ D_{OW}$ values (4.64 and
512 4.28, respectively), which this organic compound could be adsorbed on the hydrophobic
513 foulant layer formed onto the ceramic surface. Diazepam showed quite hydrophobicity
514 and negative charge (pKa < feed solution pH) at these pH conditions, where the
515 electrostatic repulsion between the foulant layer and this anionic compound could

favour its rejection. In the same way, Lopez-Fernandez and her colleagues demonstrated that diazepam was partially adsorbed by the submerged hydrophobic UF membrane due to the hydrophobic solute-membrane interactions [57].

Moreover, rejection values higher than 50 % were obtained at pH 8 for ibuprofen, naproxen, and sulfamethoxazole, increasing from 44.3, 44.7, and 26.4 % at pH 6 to 62.4, 61.3, and 57.5 % at pH 8, respectively. The main separation mechanism for such PhACs was electrostatic repulsion with the negatively charged surface of both foulant layer and ceramic membrane (as was observed in the previous section). However, the feed solution pH had a considerable effect on the hydrophobicity and solubility of such compounds, which caused an improvement in their rejection. These PhACs were negatively charged at the entire pH range tested, but they presented different hydrophilicity: ibuprofen and naproxen were hydrophobic while sulfamethoxazole was hydrophilic. The pH-dependence of their hydrophobicity can be observed in Table 4 and Fig. 6, where the values of *log* D_{OW} of ibuprofen, naproxen, and sulfamethoxazole decreased when pH increased. This indicates that such compounds became more hydrophilic at higher pH values, especially for naproxen and sulfamethoxazole with *log* $D_{OW} < 0$ (no sorption potential). Nghiem and Hawkes demonstrated that the solubility of ibuprofen and sulfamethoxazole significantly increased at neutral and basic conditions, resulting in a decrease in the hydrophobicity of the anionic ibuprofen [39]. This change in hydrophobicity for ibuprofen and naproxen at pH 8 was also observed by Jin et al. [58]. Therefore, the decrease in hydrophobicity at higher pHs together with the electrostatic repulsion between the negative foulant layer (which acts as a supplementary hydrophobic membrane onto the ceramic hydrophilic membrane) and

such negative molecules result in a remarkable improvement of their removal (see Fig. 3).

3.3. PhAC adsorption

In order to understand the retention mechanisms for the selected PhACs during filtration experiments, PhACs adsorption was quantitatively assessed by applying mass balances based on the concentration of each PhAC in the feed, permeate and retentate. The percentages of the adsorbed mass or adsorption percentage (M_{ads}) of each PhAC and membrane for Feed I and Feed 2 are listed in Table 5 and 6, respectively. It must be remarked that the results of M_{ads} reflect not only the adsorption onto the membrane surface and pore walls but also the amount adsorbed on the foulant layer (during filtration of Feed II). Very low adsorption percentages (< 0.1 %) of PhACs were obtained during filtration of Feed I (see Table 5). As expected, the adsorption of PhACs on ceramic ultrafiltration membranes was very limited. Hydrophobic PhAC molecules could not be adsorbed onto the hydrophilic ceramic surface, but can be entrapped due to its roughness, as was explained before in fouling analysis. Similar percentages can be observed for each PhAC and membrane with very small differences (which cannot be considered as significant) associated with the PhACs properties (such as charge and hydrophilicity). However, higher adsorption percentages were found for Feed II (see Table 6), especially for hydrophobic compounds (diazepam, diclofenac, erythromycin, ibuprofen, naproxen, and triclosan). This fact may be because the higher hydrophobic character (high sorption potential) of a compound results in a higher adsorption on the hydrophobic separation layer, and it can be even increased when this compound is neutral (such as erythromycin within the studied pH range and triclosan at pH 6 and 7). The adsorbed mass of both acetaminophen and caffeine (hydrophilic and neutral

PhACs) is almost constant at each pH, which confirms their pH-independent behaviour. In the same way, a decline in the adsorbed mass of ibuprofen, naproxen and sulfamethoxazole on the hydrophobic foulant layer formed onto the membranes can be seen in Table 6. As discussed previously, an increasing pH enhances the solubility of ibuprofen, naproxen and sulfamethoxazole in an aqueous solution and hence, may reduce the hydrophobic character of these PhACs. This improvement in their apparent hydrophilic character leads to lower adsorptions during filtration. The highest adsorptions observed were for diclofenac and triclosan (M_{ads} between 8 and 12 % at pH 8). Due to their high hydrophobic character ($log\ K_{OW} > 4.5$ and $log\ D_{OW} > 4.2$), diclofenac and triclosan were strongly adsorbed by hydrophobic interactions with the foulant layer [57], as was explained before. Finally, the behaviour of trimethoprim is also corroborated with the results shown in Table 6, where the highest adsorbed mass of this compound was found at pH 6 due to the charge attraction between its positive molecules and the negatively charged foulant layer.

4. CONCLUSIONS

In order to elucidate the influence of feed solution pH and fouling phenomena on the removal of emerging contaminants, the rejection of ten selected PhACs with different physicochemical characteristics (such as molecular weight, water solubility, log K_{OW}, pKa, dipole moment, and charge) using ceramic ultrafiltration membranes was investigated. As expected, ceramic membranes with smaller nominal pore size showed higher rejection values than the larger ceramic membranes. Variations in the pH conditions demonstrated the key role of pKa and log K_{OW} on the rejection of pharmaceutically active compounds, obtaining higher removal efficiencies at basic pHs, especially for anionic compounds. The study of the rejection of anionic compounds

590 demonstrated that electrostatic repulsion was the predominant mechanism in the
591 rejection of ionic compounds, especially when feed solutions formed by PhACs spiked
592 in deionised water were used. However, hydrophobic/hydrophilic interactions and the
593 adsorptive mechanism gained importance when real wastewater effluent was used. The
594 hydrophilic character and solubility of the anionic compounds improved with increasing
595 pH up to 8 (slightly alkaline), at which anionic compounds were effectively rejected at
596 basic pHs. Non-ionic erythromycin was the only compound that was significantly
597 affected by the sieving effect due to its similarities between the molecular weight of this
598 compound and the nominal pore size of the smallest ceramic membrane tested (ceramic
599 1 kDa membrane). Therefore, the experimental results highlighted that the geometry,
600 dipole moment, charge and hydrophobicity of the compound plays an active role in the
601 membrane rejection, even more than its molecular weight. Membrane fouling was also
602 influenced by the pH variations of the feed solution, observing higher irreversible
603 fouling at slightly alkaline pHs. At these conditions, PhACs rejection was higher. Thus,
604 the resulting foulant layer formed onto the membrane surface improved the adsorption
605 of some compounds and the charge repulsions between anionic compounds and the
606 negatively charged membrane surface. In the same way, the formation of EfOM-PhACs
607 complexes as a result of the association of PhACs with organic macromolecules
608 significantly improved the rejection of neutral compounds such as erythromycin. The
609 reported results indicated that the rejection of PhACs was strongly pH-dependent,
610 except for hydrophilic neutral compounds (acetaminophen and caffeine), which showed
611 a pH-independent behaviour with low rejection values.
612

613 ## 5. ACKNOWLEDGEMENTS

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796
797 ## 7. LIST OF SYMBOLS
798 **Variables**

799	A_m	Effective area of the membrane (m^2)
800	C_f	Concentration of each pharmaceutically active compound in the feed
801		stream ($ng \cdot L^{-1}$)
802	C_p	Concentration of each pharmaceutically active compound in the permeate
803		stream ($ng \cdot L^{-1}$)
804	C_r	Concentration of each pharmaceutically active compound in the retentate
805		stream ($ng \cdot L^{-1}$)

J_0	Initial permeate flux ($L \cdot m^{-2} \cdot h^{-1}$)
J_f	Permeate flux at the end of filtration experiments ($L \cdot m^{-2} \cdot h^{-1}$)
Jp	Permeate flux ($L \cdot m^{-2} \cdot h^{-1}$)
J_r	Permeate flux during the rinsing process ($L \cdot m^{-2} \cdot h^{-1}$)
K	Water permeability ($L \cdot m^{-2} \cdot h^{-1} \cdot bar^{-1}$)
LOD	Limit of detection of a compound ($ng \cdot L^{-1}$)
$log\ D_{OW}$	pH-corrected value of the logarithm of the octanol-water partition coefficient (dimensionless)
$log\ K_{OW}$	Logarithm of the octanol-water partition coefficient (dimensionless)
LOQ	Limit of quantification of a compound ($ng \cdot L^{-1}$)
m	Mass of permeate water (g)
m_{ads}	Adsorbed mass of a compound ($ng \cdot m^{-2}$)
M_{ads}	Adsorption percentage (%)
$MWCO$	Molecular weight cut-off (Da)
pKa	Dissociation constant (dimensionless)
R	Solute rejection index (%)
R_{irr}	Membrane irreversible resistance (m^{-1})
R_m	Membrane intrinsic resistance (m^{-1})
R_{rev}	Membrane reversible resistance (m^{-1})
R_T	Membrane total resistance (m^{-1})
t	Filtration time (h)
T	Temperature (°C)
V	Total volume permeated during an experimental time interval (L)
V_f	Volume of each PhAC in the feed stream (L)
V_p	Volume of each PhAC in the permeate stream (L)

831	V_r	Volume of each PhAC in the retentate stream (L)
832	ΔP	Transmembrane pressure (bar)
833	ρ	Density of water at the operating temperature ($g \cdot L^{-1}$)
834		

835 **Abbreviations**

836	APIs	Active pharmaceutical ingredients
837	DBPs	Disinfection by-products
838	EDCs	Endocrine disrupting compounds
839	EfOM	Effluent organic matter
840	HPLC	High-Performance liquid chromatography
841	LC-MS/MS	Liquid chromatography tandem-mass spectrometry
842	MF	Microfiltration
843	NF	Nanofiltration
844	NOM	Natural organic matter
845	PCPs	Personal care products
846	PhACs	Pharmaceutically active compounds
847	RO	Reverse osmosis
848	SRM	Selected reaction monitoring
849	UF	Ultrafiltration
850	WWTPs	Wastewater treatment plants

851

Table 1. Characteristics of the secondary effluents from a local wastewater treatment plant at 25 ºC.

Parameter	Feed solution[a]
pH	7.98 ± 0.13
m-Alkalinity (mg $CaCO_3 \cdot L^{-1}$)	340.12 ± 13.55
Electrical conductivity ($\mu S \cdot cm^{-1}$)	1574.50 ± 36.81
TSS (ppm)	157.00 ± 53.92
Turbidity (NTU)	19.43 ± 1.96
COD (mg $O_2 \cdot L^{-1}$)	86.02 ± 12.59
UV_{254}	0.504 ± 0.002
Total Nitrogen (mg $N \cdot L^{-1}$)	73.30 ± 16.10
Proteins ($mg \cdot L^{-1}$)	65.25 ± 10.03

[a]Average ± standard deviation.

Table 2. Physicochemical properties of the selected PhACs studied.

Pharmaceutical active compound	CAS no.	Formula	Molecular weight ($g \cdot mol^{-1}$)	Log K_{ow}[a]	pK_a[a]	Charge (pH= 7)	Hydrophobic / Hydrophilic[b]	Dipole moment (D)[c]	Molar volume ($cm^3 \cdot mol^{-1}$)[c]
Acetaminophen	103-90-2	$C_8H_9NO_2$	151.166	0.494	9.86	0	Hydrophilic	3.850	121.0
Caffeine	58-08-2	$C_8H_{10}N_4O_2$	194.194	-0.040	10.4	0	Hydrophilic	3.401	133.9
Diazepam	439-14-5	$C_{16}H_{13}ClN_2O$	284.746	2.820	3.4	-1	Hydrophobic	2.173	226.0
Diclofenac	15307-79-6	$C_{14}H_{10}Cl_2NNaO_2$	318.136	4.640	4.08	-1	Hydrophobic	2.508	207.0
Erythromycin	114-07-8	$C_{37}H_{67}NO_{13}$	733.942	3.060	8.9	0	Hydrophobic	3.988	611.6
Ibuprofen	15687-27-1	$C_{13}H_{18}O_2$	206.286	3.679	4.40	-1	Hydrophobic	1.223	200.5
Naproxen	22204-53-1	$C_{14}H_{14}O_3$	230.265	2.816	4.15	-1	Hydrophobic	2.838	192.4
Sulfamethoxazole	723-46-6	$C_{10}H_{11}N_3O_3S$	253.283	0.564	5.7	-1	Hydrophilic	7.366	173.2
Triclosan	3380-34-5	$C_{12}H_7Cl_3O_2$	289.546	5.529	7.8	0	Hydrophobic	2.450	194.3
Trimethoprim	738-70-5	$C_{14}H_{18}N_4O_3$	290.32	0.981	6.6-7.1	+1	Hydrophilic	2.535	231.9

[a]SciFinder Scholar, data calculated at 20ºC and 760 torr using Advanced Chemistry Development (ACD/Labs) Software V11.02 (©1994-2016 ACD/Labs).
[b]Hydrophobic when log $K_{OW} > 2$.
[c]Chem3D Ultra 8.0.

Table 3. Selecting validation parameters for the HPLC-MS/MS method, where LOD is the limit of detection and LOQ is the limit of quantification (LOQ) for each compound tested.

Compound	LOD ($ng \cdot L^{-1}$)	LOQ ($ng \cdot L^{-1}$)
Acetaminophen	0.9	2.7
Caffeine	0.6	1.8
Diazepam	1.0	3.0
Diclofenac	0.3	1.0
Erythromycin	6.0	18.0
Ibuprofen	5.0	15.0
Naproxen	0.5	1.5
Sulfamethoxazole	0.9	2.7
Triclosan	0.3	1.0
Trimethoprim	0.9	2.7

Table 4. Log D_{OW} values calculated for PhACs that are not neutral (pKa < feed solution pH) at the tested pH conditions.

Pharmaceutical active compound	Calculated Log D_{OW}[a]		
	pH 6	pH 7	pH 8
Diazepam	3.08	3.08	3.08
Diclofenac	4.28	4.28	4.28
Ibuprofen	2.67	1.71	0.85
Naproxen	1.18	0.25	-0.36
Sulfamethoxazole	0.60	0.14	-0.11
Triclosan	4.97	4.90	4.50
Trimethoprim	0.27	0.92	1.23

[a]Software Calculator Plugins was used to calculate Log D_{OW} at each pH.

Table 5. Adsorption percentage of each PhAC for different ceramic membranes using Feed I within the studied pH range calculated by mass balances.

Compound	**M_{ads} (%) for Feed I**								
	pH 6			**pH 7**			**pH 8**		
	1 kDa	**5 kDa**	**8 kDa**	**1 kDa**	**5 kDa**	**8 kDa**	**1 kDa**	**5 kDa**	**8 kDa**
Acetaminophen	0.066	0.041	0.026	0.069	0.042	0.032	0.069	0.044	0.034
Caffeine	0.044	0.030	0.018	0.043	0.031	0.021	0.046	0.031	0.024
Diazepam	0.046	0.044	0.038	0.049	0.048	0.042	0.056	0.053	0.049
Diclofenac	0.071	0.049	0.029	0.076	0.057	0.049	0.076	0.068	0.056
Erythromycin	0.076	0.073	0.063	0.074	0.076	0.070	0.064	0.076	0.074
Ibuprofen	0.072	0.043	0.036	0.076	0.057	0.048	0.076	0.070	0.053
Naproxen	0.057	0.043	0.021	0.072	0.064	0.036	0.077	0.070	0.044
Sulfamethoxazole	0.073	0.071	0.036	0.076	0.074	0.050	0.067	0.077	0.055
Triclosan	0.058	0.041	0.030	0.074	0.054	0.049	0.074	0.063	0.045
Trimethoprim	0.076	0.066	0.046	0.056	0.050	0.021	0.076	0.070	0.045

Table 6. Adsorption percentage of each PhAC for different ceramic membranes using Feed II within the studied pH range calculated by mass balances.

Compound	**M_{ads} (%) for Feed II**								
	pH 6			**pH 7**			**pH 8**		
	1 kDa	**5 kDa**	**8 kDa**	**1 kDa**	**5 kDa**	**8 kDa**	**1 kDa**	**5 kDa**	**8 kDa**
Acetaminophen	6.142	6.096	5.324	6.044	6.191	5.716	6.149	6.116	5.757
Caffeine	5.290	4.459	2.094	5.084	4.746	2.019	4.660	4.730	2.257
Diazepam	7.324	6.734	5.369	8.428	7.002	5.582	8.488	8.097	6.559
Diclofenac	7.392	6.267	7.093	8.985	7.711	8.488	10.121	8.317	9.636
Erythromycin	6.214	6.977	6.908	7.791	8.369	7.469	8.981	9.549	7.941
Ibuprofen	8.452	7.799	7.393	7.583	7.545	5.853	7.025	6.342	5.640
Naproxen	8.274	8.131	6.368	7.975	7.016	5.811	7.599	5.854	5.611
Sulfamethoxazole	7.608	7.121	5.773	6.975	5.982	5.101	5.946	5.302	4.165
Triclosan	7.308	4.934	4.913	8.110	8.101	8.159	9.963	11.168	11.464
Trimethoprim	7.190	6.909	5.026	5.791	4.746	3.619	6.200	5.132	3.993

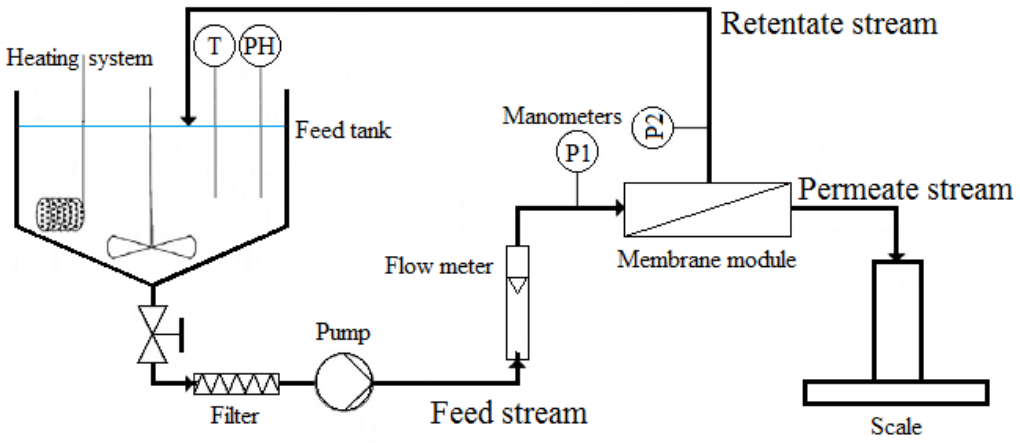
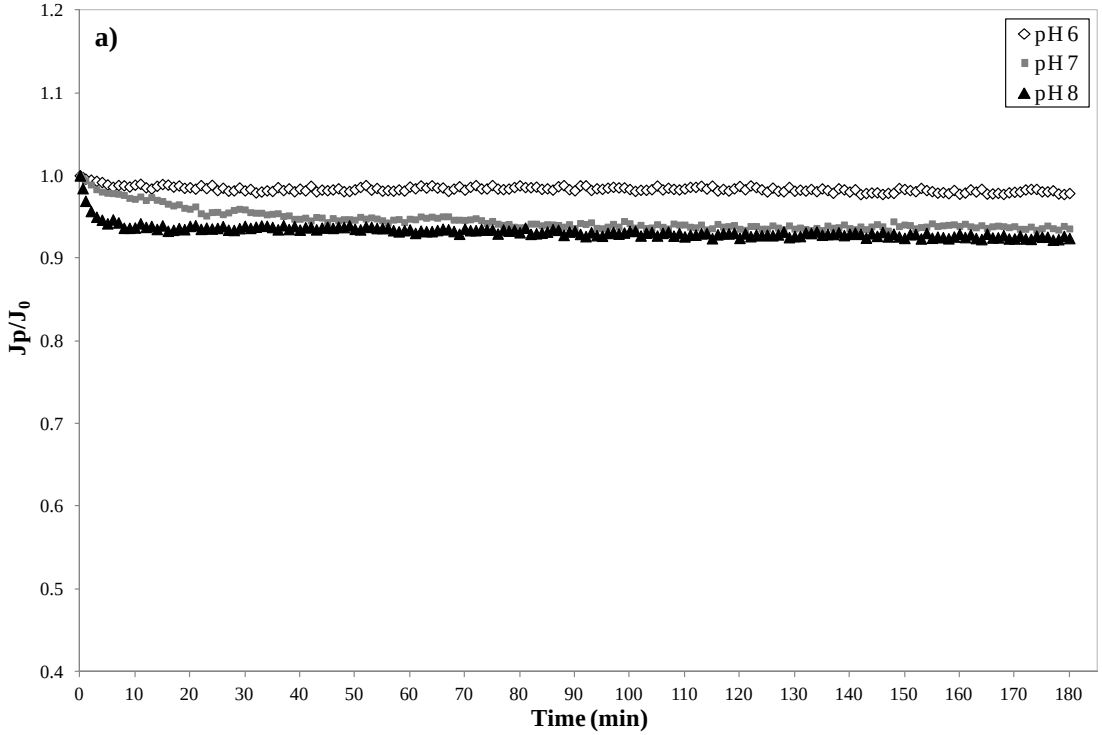
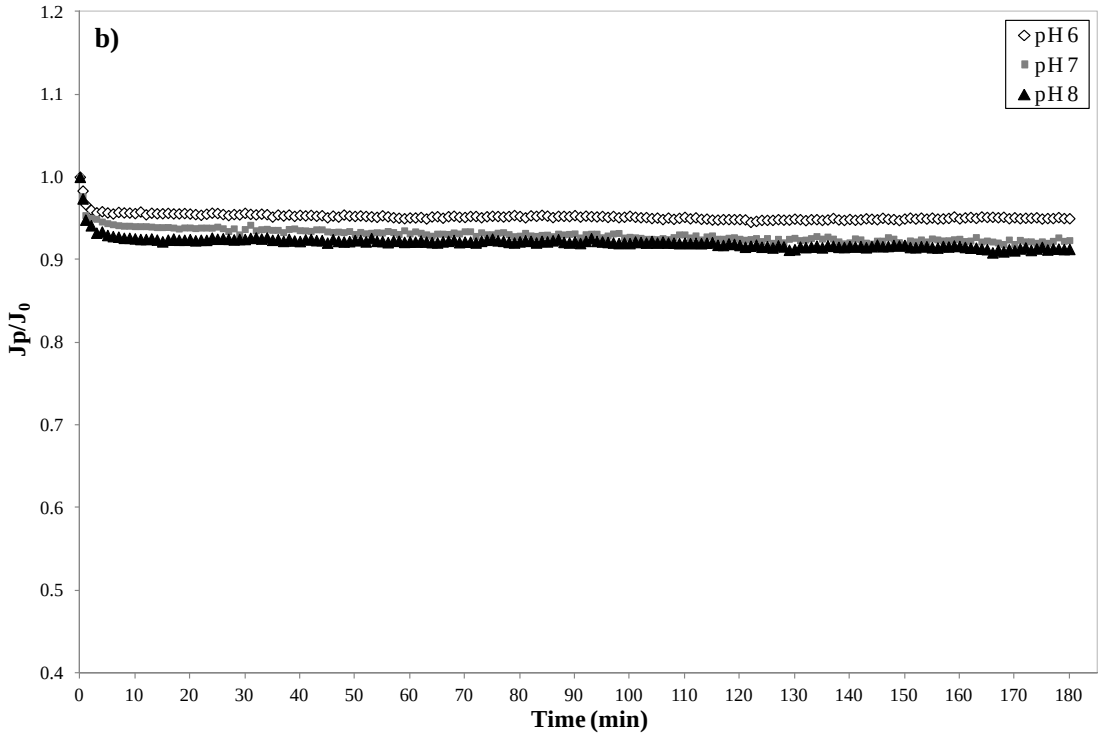


Fig. 1. Schematic diagram of the standard cross-flow ultrafiltration set up.


a)
pH 6
pH 7
pH 8
Jp/J0
Time (min)


b)
pH 6
pH 7
pH 8
Jp/J0
Time (min)

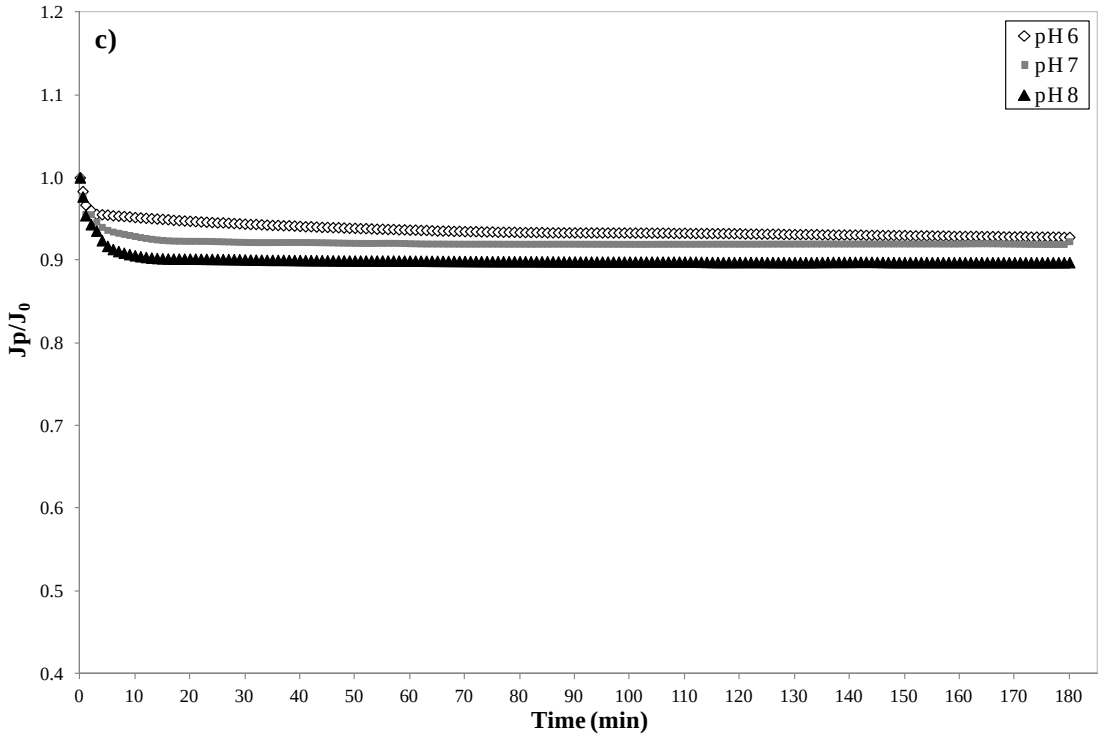
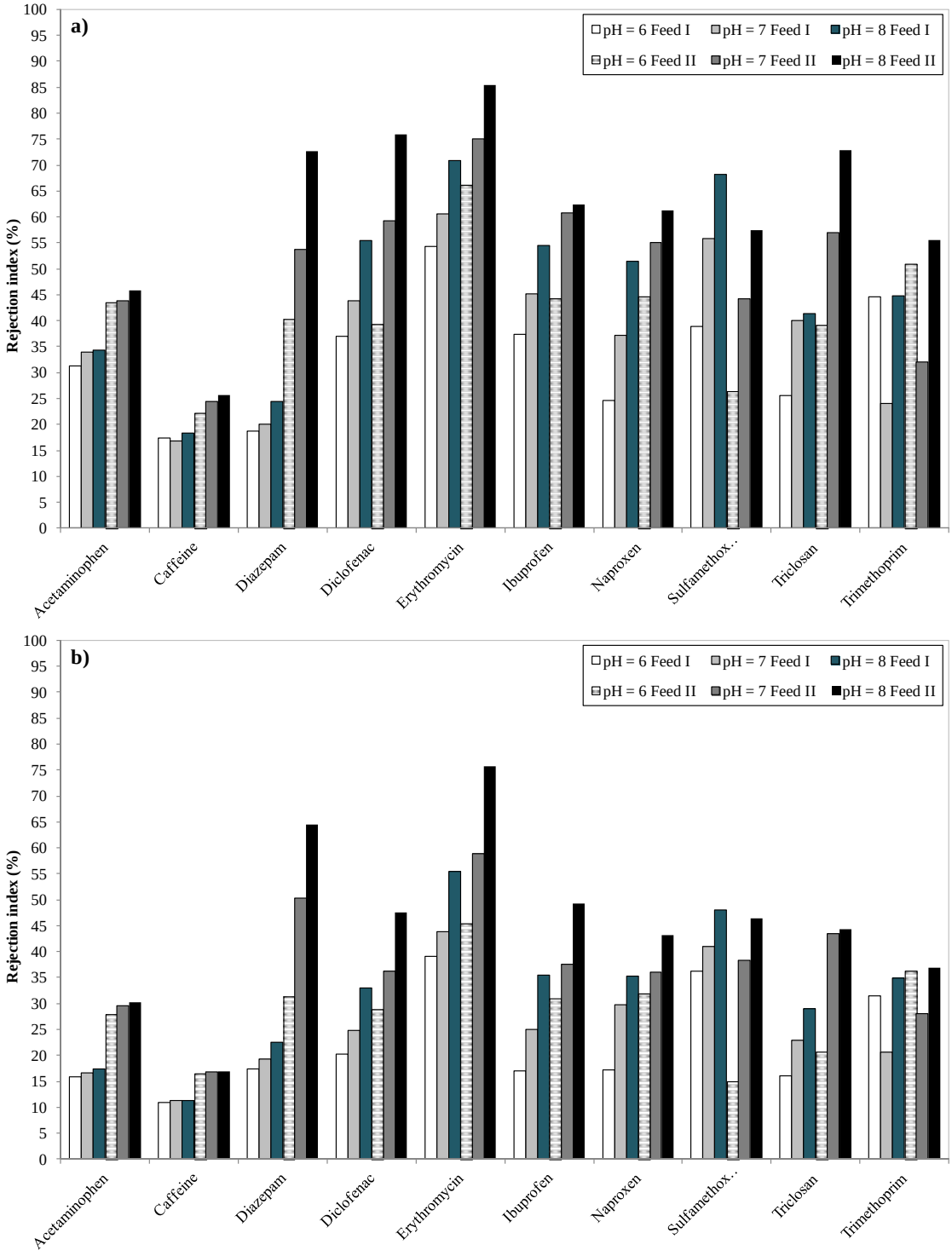


Fig. 2. Evolution of $Jp(t)/J_0$ over time (3 h) during filtration experiments of Feed I at different pHs for: a) ceramic 1 kDa membrane, b) ceramic 5 kDa membrane, and c) ceramic 8kDa membrane. Experimental conditions: 2 bar, 300 L h^{-1}, and 25 ± 2 ºC.


a)
pH = 6 Feed I
pH = 7 Feed I
pH = 8 Feed I
pH = 6 Feed II
pH = 7 Feed II
pH = 8 Feed II
Rejection index (%)
100
95
90
85
80
75
70
65
60
55
50
45
40
35
30
25
20
15
10
5
0
Acetaminophen
Caffeine
Diazepam
Diclofenac
Erythromycin
Ibuprofen
Naproxen
Sulfamethox..
Triclosan
Trimethoprim
b)
pH = 6 Feed I
pH = 7 Feed I
pH = 8 Feed I
pH = 6 Feed II
pH = 7 Feed II
pH = 8 Feed II
Rejection index (%)
100
95
90
85
80
75
70
65
60
55
50
45
40
35
30
25
20
15
10
5
0
Acetaminophen
Caffeine
Diazepam
Diclofenac
Erythromycin
Ibuprofen
Naproxen
Sulfamethox..
Triclosan
Trimethoprim

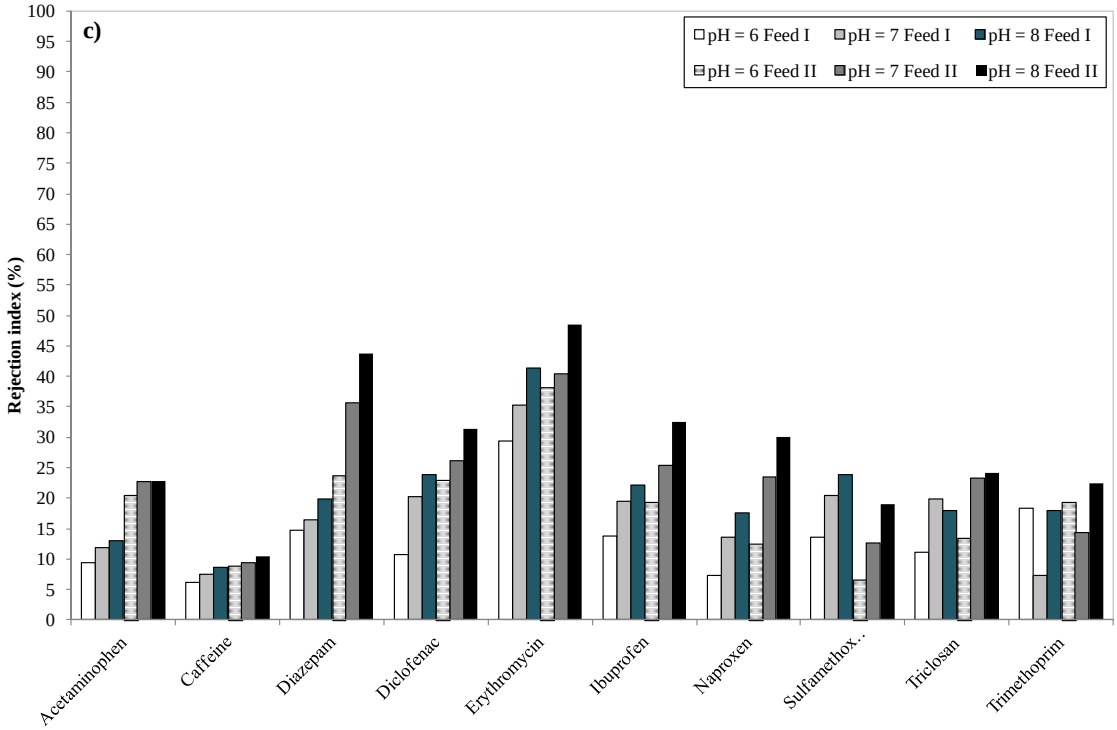
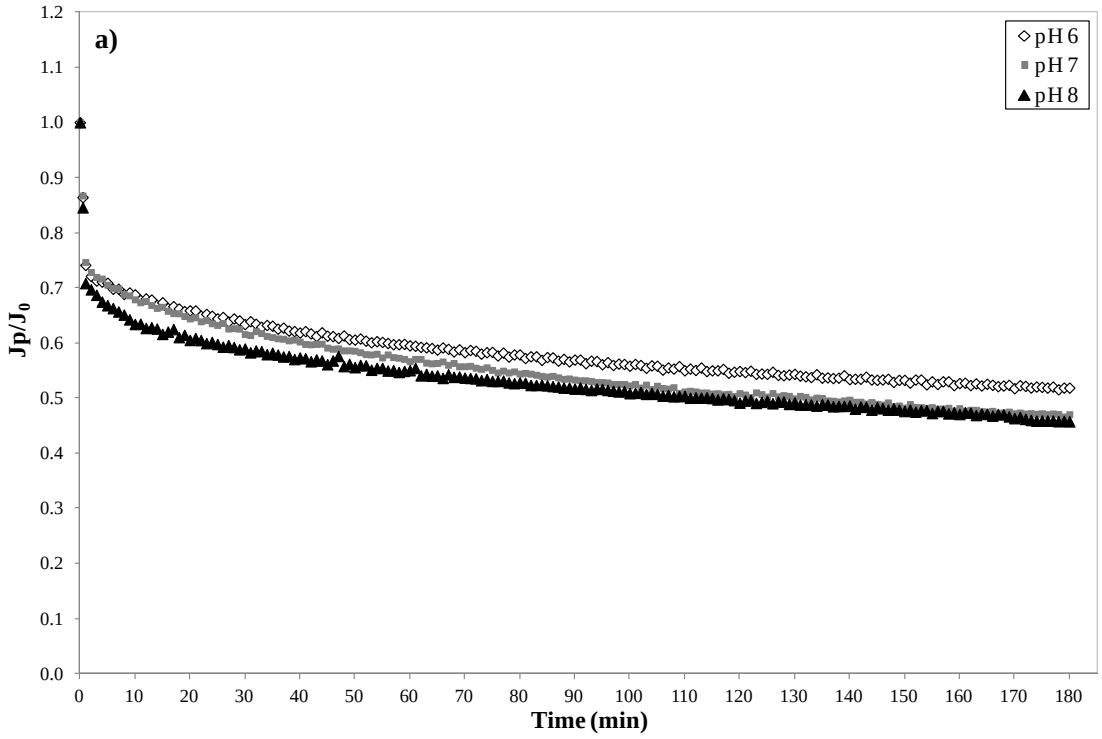
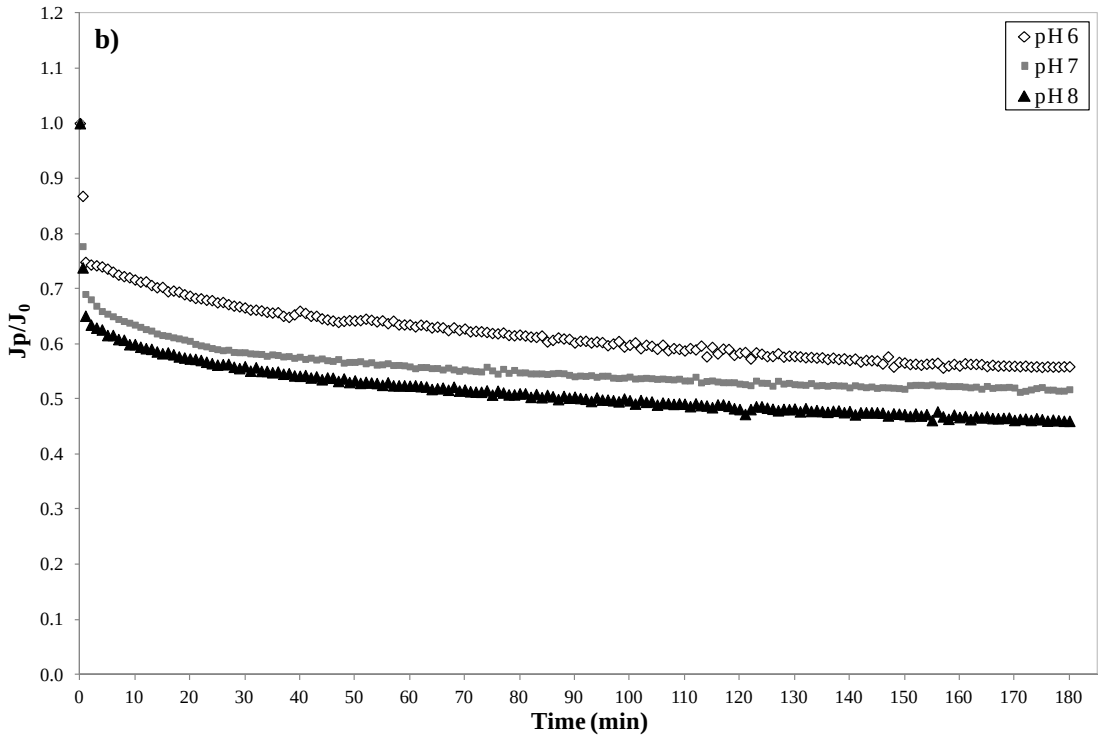


Fig. 3. Rejection values of PhACs during filtration experiments of Feed I and Feed II at different pHs for: a) ceramic 1 kDa membrane, b) ceramic 5 kDa membrane, and c) ceramic 8kDa membrane. Experimental conditions: 2 bar, 300 L h^{-1}, and 25 ± 2 ºC.


a)
pH 6
pH 7
pH 8
Jp/J0
Time (min)


b)
pH 6
pH 7
pH 8
Jp/J0
Time (min)

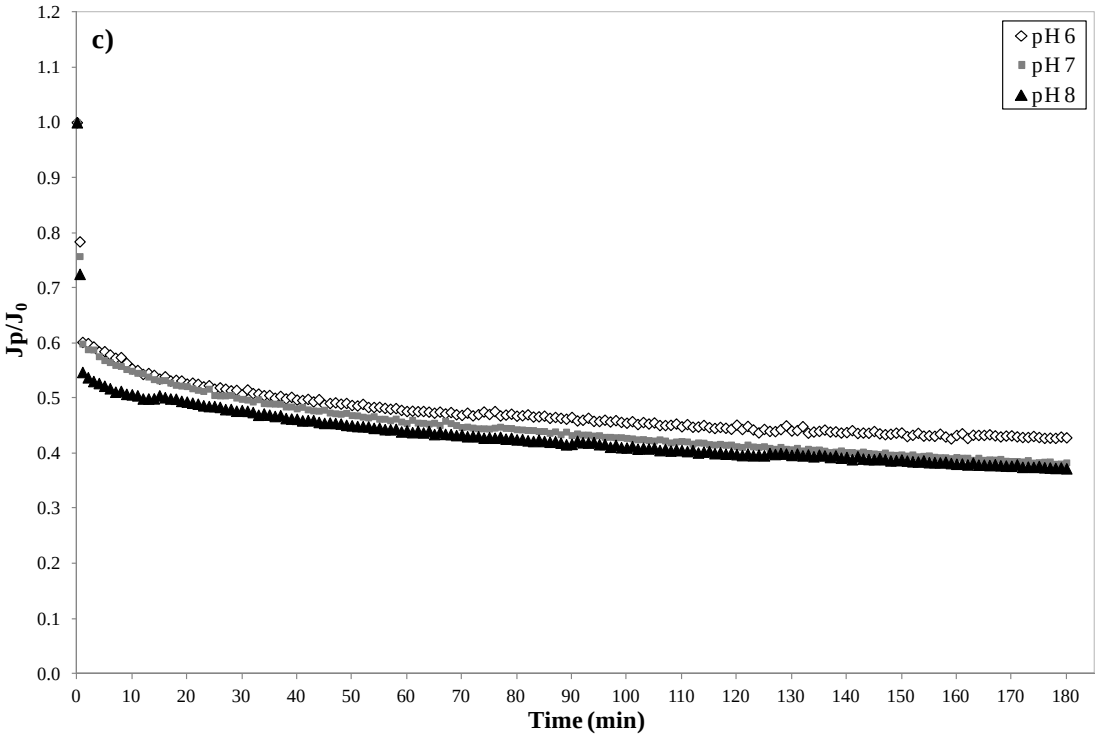
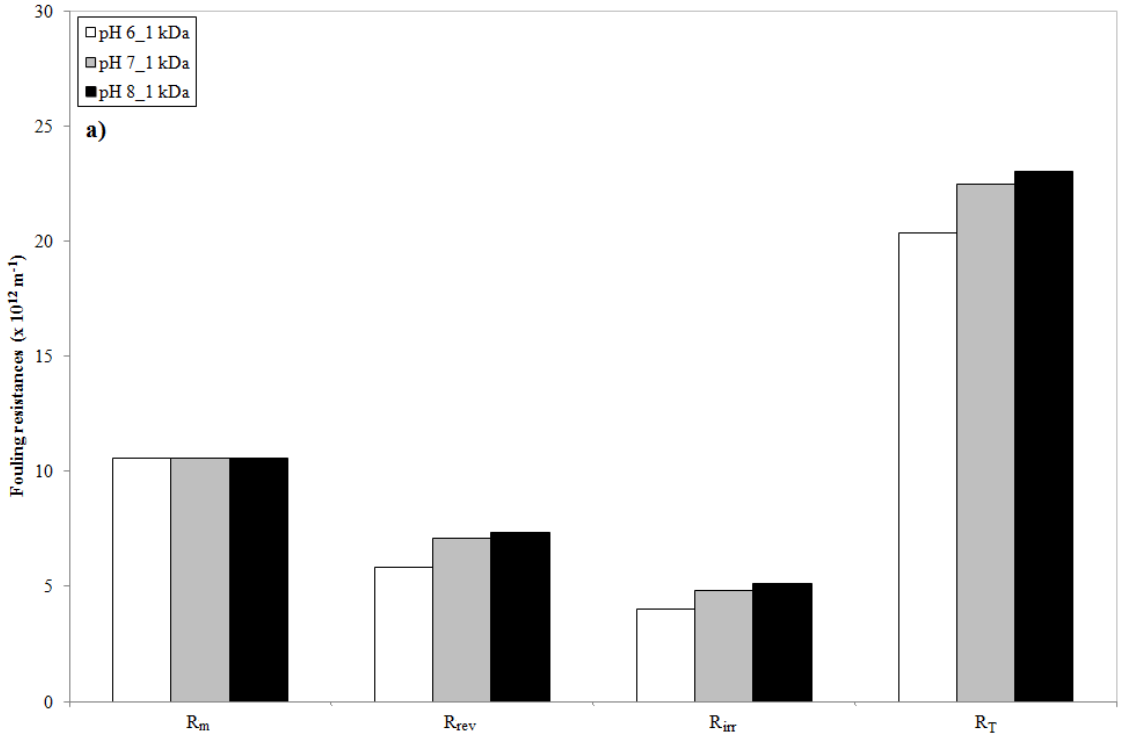
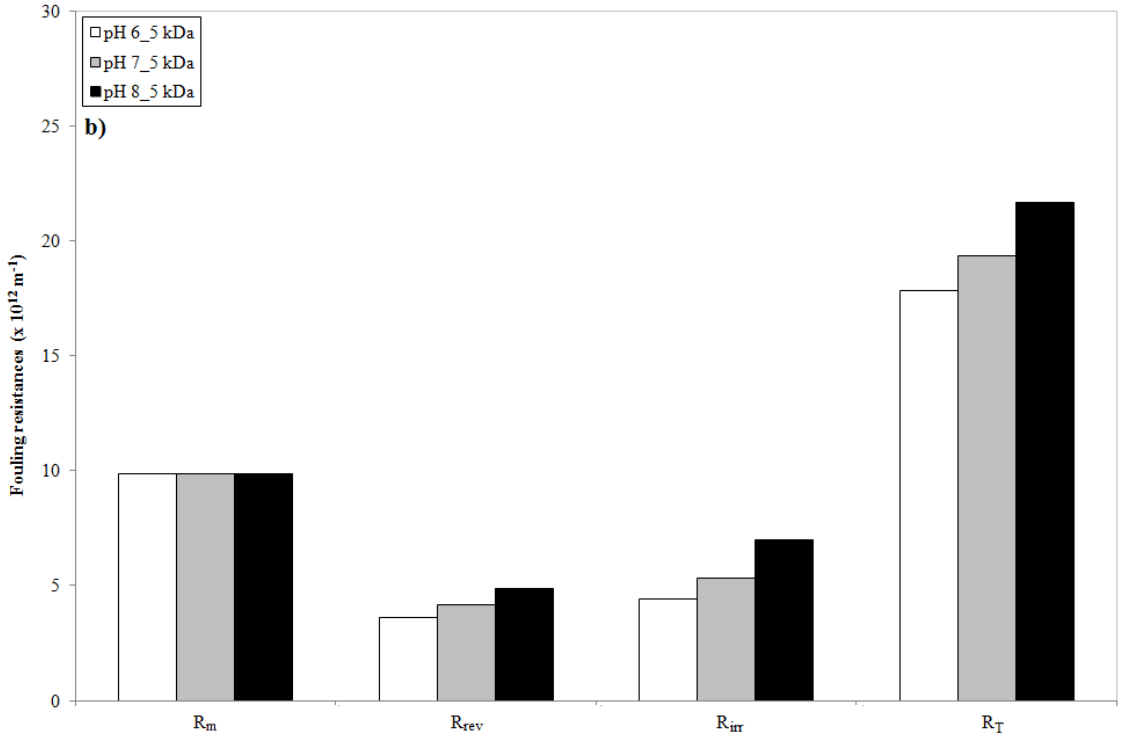


Fig. 4. Evolution of $Jp(t)/J_0$ over time (3 h) during filtration experiments of Feed II at different pHs for: a) ceramic 1 kDa membrane, b) ceramic 5 kDa membrane, and c) ceramic 8kDa membrane. Experimental conditions: 2 bar, 300 L h^{-1}, and 25 ± 2 ºC.


pH 6_1 kDa
pH 7_1 kDa
pH 8_1 kDa
a)
Fouling resistances (x 10^12 m^-1)
30
25
20
15
10
5
0
Rm
Rrev
Rirr
RT


pH 6_5 kDa
pH 7_5 kDa
pH 8_5 kDa
b)
Fouling resistances (x 10^12 m^-1)
30
25
20
15
10
5
0
Rm
Rrev
Rirr
RT

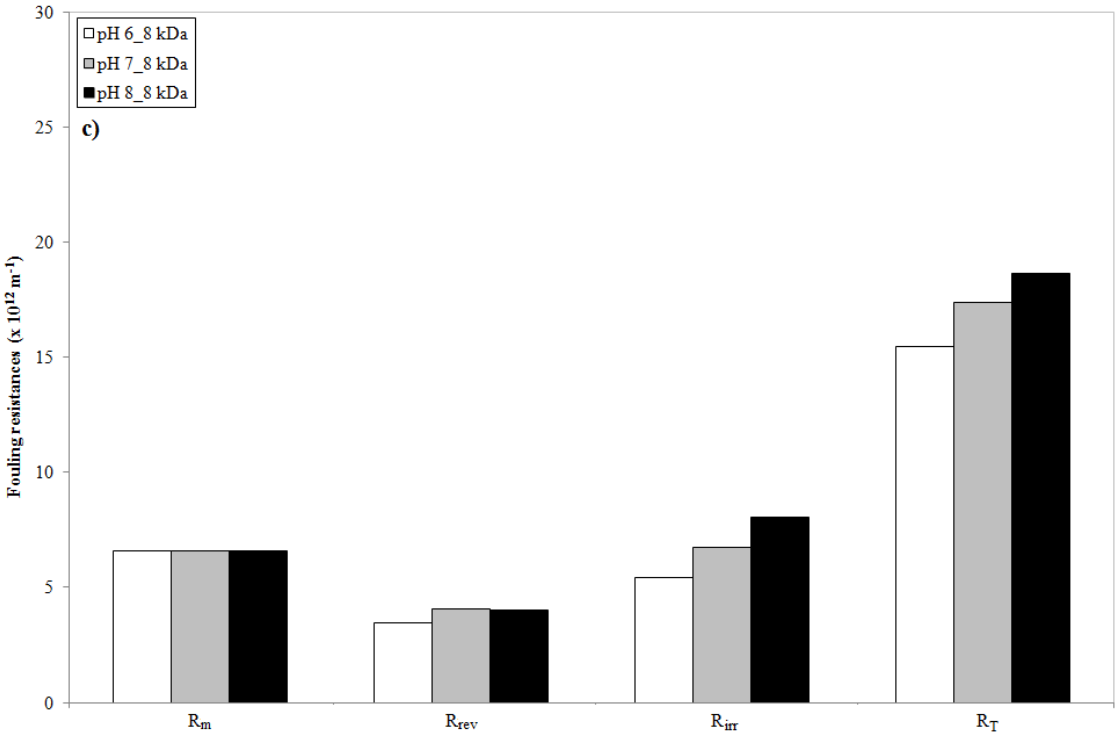


Fig. 5. Intrinsic membrane resistance (R_m), reversible fouling resistance (R_{rev}), irreversible fouling resistance (R_{irr}), and total fouling resistance (R_T) of each ceramic ultrafiltration membrane determined from filtration of PhACs in Feed II: a) ceramic 1 kDa membrane, b) ceramic 5 kDa membrane, and c) ceramic 8kDa membrane.

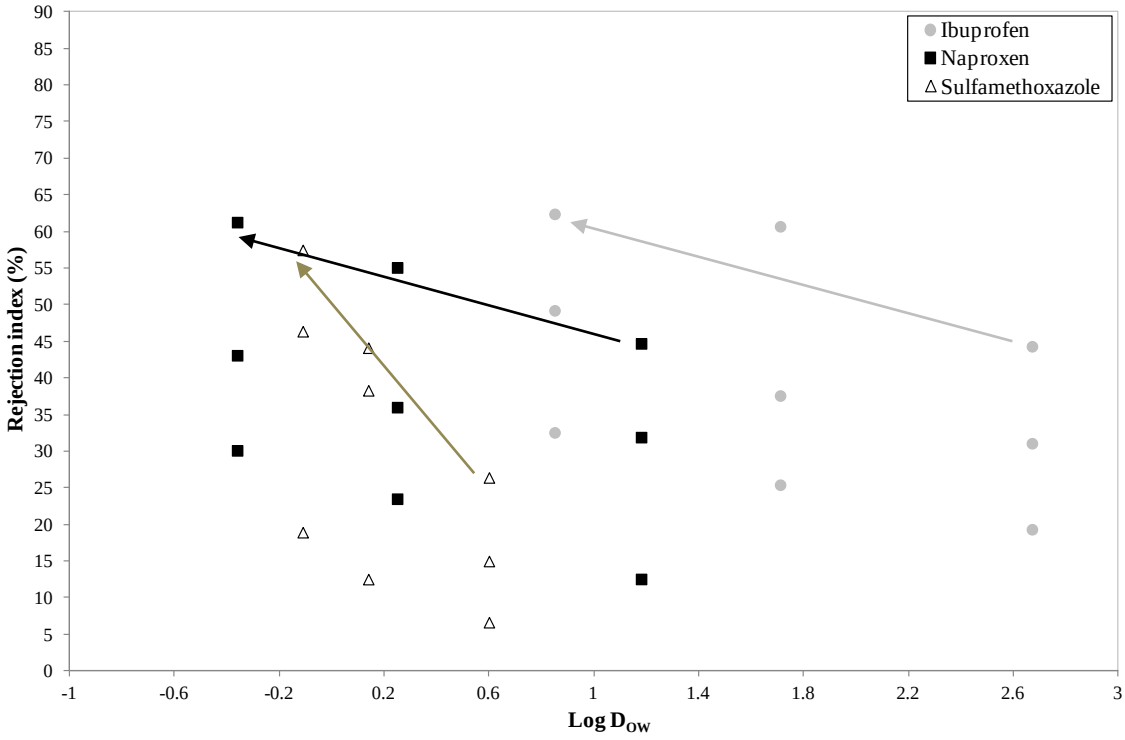


Fig. 6. Rejection values of ibuprofen (●), naproxen (■), and sulfamethoxazole (Δ) during filtration experiments of Feed II as a function of the pH-corrected octanol-water partition coefficient (Log D_{OW}). Coloured arrows indicate the evolution of feed solution pH from 6 to 8. Experimental conditions: 2 bar, 300 L h^{-1}, and 25 ± 2 ºC.