Passive sampling of perfluorinated chemicals in water: In-situ calibration

Sarit L. Kaserzon, Darryl W. Hawker, Kees Booij, Dominique S. O’Brien, Karen Kennedy, Etienne L.M. Vermeirssen, Jochen F. Mueller

1. Introduction

Perfluorinated chemicals (PFCs) are a family of anthropogenic pollutants that are the subject of increasing scrutiny and concern due to their widespread distribution in the environment along with their persistence and the toxic properties of some members. Consequently, regulation of some PFCs such as perfluorooctanesulfonate (PFOS) has begun (European Commission, 2012; Houde et al., 2011; Labadie and Chevreuil, 2011; Stockholm Convention, 2010). The requirement for environmental monitoring of these chemicals is expected to increase. Muir and Lohmann (2013) recently observed that with the addition of compounds such as PFOS to the Stockholm Convention, the chemicals addressed no longer comprise just hydrophobic organics. PFOS is an anionic conjugate base of perfluorooctanesulfonic acid, and other perfluorinated sulfonic and carboxylic acids (PFSAs and PFCAs) are also typically present in natural waters as anions with moderate aqueous solubility (Kaserzon et al., 2013).

Passive sampling technologies have been successfully used for the past four decades as useful monitoring tools for a range of environmental pollutants in water (Alvarez et al., 2004b; Booij et al., 2007; Huckins et al., 1999). In more recent years a range of polymeric sorbents, traditionally used for solid phase extraction, have been employed as passive sampling receiving phases due to their high recoveries of analytes, ease of handling and effectiveness with a wider range of polar organic chemical pollutants. In particular, the polar organic chemical integrative sampler (POCIS) configuration involving sorbents including Oasis HLB, Oasis MAX and Oasis MCX has been used for emerging pharmaceuticals and personal care products, pesticides and herbicides (Alvarez et al., 2004a; Li et al., 2011; Mazzella et al., 2008; Vermeirssen et al., 2008). The latter ion exchange sorbents (OASIS MAX and MCX) have been used for chemical species that are ionised in aqueous matrices.

Recently, a modified POCIS comprising a weak anion exchange sorbent phase (StrataTM XAW) was successfully developed for quantifying perfluorinated chemicals in water (Kaserzon et al., 2012, 2013). When the effect of water flow rate (between 0.02 and 0.34 m s\(^{-1}\)) on PFC sampling rates \((R_s)\) was examined, an increase in \(R_s\)'s with increasing water flow rate for PFCs up to...
perfluorononanoate (PFNA, MW = 463) was observed. The effect of in-situ water flow rate should therefore be accounted for when deploying these modified POCIS in the field, to improve water concentration estimates for PFCs (Kaserzon et al., 2013). In this previous study, sampling rates were determined daily using hand held flow meters, but under environmental conditions this may not be practical.

The importance of in-situ calibration methods in order to adjust chemical sampling rates obtained in laboratory calibrations has been widely acknowledged (Huckins et al., 2002a). Without them, derived results of time weighted average aqueous concentrations are at best semi-quantitative (Harman et al., 2012). In the environment, conditions during sampler deployment may affect uptake behaviour of chemicals into passive samplers (PSSs). In particular, deployment-specific water flow rates can affect the sampling behaviour of analytes by influencing mass transfer between the external water and the surface of the sampler (Booij et al., 2007). The effect is particularly pronounced with partition based passive samplers such as the semipermeable membrane device (SPMD) designed for monitoring hydrophobic compounds (Vrana et al., 2005). In the case of PSSs designed for more polar compounds such as the POCIS, limited studies on the effects of water flow rate on sampling behaviour suggest the influence is less pronounced and may be compound-specific (Kaserzon et al., 2013; Li et al., 2010; Vermeersen et al., 2008).

The range of environmental conditions encountered by passive samplers (e.g. water flow rate, temperature, salinity) cannot always be accommodated in laboratory calibration studies due to time and resource restrictions. Methods of in-situ calibration during field deployments such as the co-deployment of data loggers and meters at multiple sites may also be costly and in many cases, multiple visits to sites for manual measurements may be similarly restrictive (O’Brien et al., 2009). An in-situ calibration method for flow rate effects on sampling rates is the co-deployment of passive flow monitors (PFMs) (O’Brien et al., 2009). In this method, the mass loss from a calcium sulphate dihydrate (gypsum) cast is used to estimate water flow rates to which the PFM is exposed. PFMs have been successfully used to calibrate the uptake rates of chemical analytes including phosphate and some polar and nonpolar herbicides and pesticides using PSSs such as the phosphate sampler, Chemcatcher™, PDMS and SPMD (O’Brien et al., 2011a; O’Brien et al., 2009; O’Brien et al., 2012). The calibrations provide chemical-and sampler-specific empirical equations that can be used to account for the change in $R_t$ with water velocity and ionic strength resulting in more accurate estimates of time-weighted average chemical concentrations. While PFMs have been suggested as a possible external correction method for POCIS (Harman et al., 2012), no studies to date have examined this.

Performance reference compounds (PRCs) have been successfully used with partitioning type samplers (e.g. SPMDs) as a means of determining the effect of environmental conditions on sampler behaviour (Booij et al., 2002; Huckins et al., 2002a). In-situ $R_t$ values are derived from the loss rate constants of PRCs that are spiked into samplers before exposure to the aquatic environment. The application of the PRC method to passive samplers whose mode of action is not partition-based is uncertain (Alvarez et al., 2007; Shaw et al., 2009; Vallejo et al., 2013). Recent studies with Oasis HLB and Oasis MAX have suggested that desisopropylatrazine-$d_5$ (Dia-$d_5$) could be a suitable PRC for anionic analytes with POCIS containing an anion-exchange sorbent (Fauvelle et al., 2012; Mazzella et al., 2007, 2010). However, investigation of its applicability as a PRC under varying environmental conditions such as flow rate has been advocated (Mazzella et al., 2010). In addition, other compounds such as isoproturon and metolachlor have shown desorption (<40%) from Oasis HLB under turbulent conditions (Mazzella et al., 2010) and may also be suitable as PRCs. We therefore examine the applicability of these compounds as PRCs with the Strata™ XAW equipped PFC sampler presented here.

Overall in this current study, we evaluate the use of PRCs as well as PFMs as in-situ correction methods for PFCs with the modified POCIS. The effects of flow rate (0.02, 0.06, 0.16, 0.34 m s$^{-1}$) in a flow-through channel system on PFM mass loss rates and the loss kinetics of DIA-$d_5$, isoproturon-$d_6$ and metolachlor-$d_6$ are investigated. PRC kinetic data and PFM mass loss rates are then related to PFC sampling rates so that these methods can be assessed for their ability to provide in-situ flow rate calibration.

2. Materials and methods

2.1. Perfluorinated chemicals

PFAs investigated in this work were perfluorooctanate (PPFoA), perfluorononanoate (PFNa), perfluorooctanoate (PFHxA), perfluorooctane sulfonate (PFOS) and perfluorooctane sulfonate acid (PFSA). The PFASs were perfluorohexanesulfonate (PFHxS) and perfluorooctanesulfonate (PFOS). $R_t$ data for PFCS discussed in this study were taken from Kaserzon et al. (2013).

2.2. Calibration study design

A flow-through channel system at Eawag, Dubendorf, Switzerland (Vermeersen et al., 2008) was used for the modified POCIS calibration study. The configuration of the channel system and methodology used have been described in Kaserzon et al. (2013). Flow velocities in the channel systems were 0.02, 0.06, 0.16 and 0.34 m s$^{-1}$. The water depth in each channel was 0.1 m. Flow rate, temperature and pH were measured daily at several points along each of the four channels (Table 52). Flow rate was measured using a flow meter (MiniAri2, Schlittenknecht, Gossau, Switzerland), and also by measuring the time to fill a bucket of known dimensions (Table 1). PFMs were deployed downstream from POCIS and, similar to the POCIS, parallel to the flow in the middle of each channel from day 0–15. Preparation, extraction and analysis of modified POCIS samplers are described in Kaserzon et al. (2013).

2.3. PRC preparation and analysis

Performance reference compounds (DIA-$d_5$, isoproturon-$d_6$, metolachlor-$d_6$) were purchased from Novachem (Victoria, Australia). The spiking of PRCs was performed according to Mazzella et al. (2010). DIA-$d_5$, isoproturon-$d_6$ and metolachlor-$d_6$ (20 µg each) were dissolved in 25 mL of methanol. The solution was added to Strata™ XAW bulk sorbent and the mixture sonicated for 5 min followed by rotary evaporation. The sorbent was then dried at 60 °C for 1 h. The final bulk sorbent (28 g) contained 0.70 µg g$^{-1}$ of each PRC. This bulk sorbent was used to prepare all deployed POCIS and blanks ($n = 2$) (600 mg per POCIS).

PRC spiked PFMs ($n = 2$) were used to determine the initial spike concentrations of each PRC. PRC analyses were conducted by Queensland Health Forensic and Scientific Services (QHSS) using HPLC–MS/MS. An AB/Sciex API 300 mass spectrometer (Applied Biosystems, Concord, Ontario, Canada) equipped with an electrospray ionization interface was coupled to a Shimadzu SCL-10A VP HPLC system (Shimadzu Corp., Kyoto, Japan). The analytical column used was a 5 µm Aquistar C18 column (150 × 3 mm) (Thermo Electron Corp., Bellefonte, PA). The temperature was 35 °C and flow rate of the mobile phase (A and B mobile phases 10% and 90% methanol)

<table>
<thead>
<tr>
<th>PFM mass loss rate ($R_{PFM}$ g d$^{-1}$), flow rate measured with a flow rate meter and volumetrically in the flow-through channel system together with flow rate predicted from $R_{PFM}$ using Eq. (1).</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>PFM mass</strong> loss rate, $R_{PFM}$</td>
</tr>
<tr>
<td>(g d$^{-1}$)</td>
</tr>
<tr>
<td>0.66$^a$</td>
</tr>
<tr>
<td>1.04</td>
</tr>
<tr>
<td>1.14</td>
</tr>
<tr>
<td>2.99</td>
</tr>
<tr>
<td>4.13</td>
</tr>
</tbody>
</table>

$^a$ Flow rate in the holding tank from which water was distributed into the channels.

$^b$ 3% instrumental measurement error.

$^c$ Calculated from the volumetric flow rate (1 min$^{-1}$) and the cross sectional area of the channels.
water (v/v) respectively both with 5 mM ammonium acetate) was 0.65 mL min\(^{-1}\). The mass spectrometer was operated in positive ion, multiple reaction-monitoring mode using \(\mathrm{N}_2\) as the collision gas. Quantitation criteria included analyte retention time compared to a standard and comparison of the mass ratio of transition ion intensity between the sample and an appropriate standard from the same run. With a minimal signal to noise value of 3, the limits of detection for \(\text{DIA-d}_8\), isoprotron-d\(_8\) and metolachlor-d\(_8\) were 1.3, 0.3 and 1.1 ng mL\(^{-1}\), respectively.

2.4. Preparation of PFMs

PFMs were prepared from gypsum dental plaster (O’Brien et al., 2009) except that the amount of plaster used as well as container height were modified in this study. In brief, plaster was cast into 120 mL specimen containers (Sarstedt) with internal dimensions of 42 mm Ø, 105 mm high. The final exposed surface area of plaster of 13.85 cm\(^2\) was the same as the PFMs used by Brien et al. (2009), but the original size containers were cut to achieve a smaller height (5.5 vs. 10.5 cm). The modification was undertaken due to the size of the channel system used. This did not alter the performance of the PFM (see Results and Discussion below). Once cast and with the contents still moist, the gypsum containers were capped and stored at 4 °C. Prior to deployment, the weight and corresponding identification number of each PFM was recorded. Deployed PFMs were removed, weighed on-site and re-deployed immediately on days 4, 10 and 15. The difference between the mass of each PFM was recorded. Deployed PFMs were removed, weighed on-site and re-deployed immediately on days 4, 10 and 15. The difference between the mass of the PFM on day 15 compared with day 0 was used to calculate total mass loss rate over the deployment period.

2.5. QA/QC

PFM mass loss rates were assessed for reproducibility by placing duplicate PFMs in 10 L of stagnant water (using large buckets) containing (i) tap water and (ii) river water, for 12 days (Table S3). The loss rates (\(\rho_{\text{PFM}, \text{g d}^{-1}}\)) of the PFMs were consistent between replicates and treatments (0.49 ± 0.02 g d\(^{-1}\), \(n = 4\)).

3. Results and discussion

3.1. Prediction of water flow rate from PFM mass loss rate (\(\rho_{\text{PFM}}\))

PFM mass loss rates (\(\rho_{\text{PFM}, \text{g d}^{-1}}\)) in the channel system were consistent with those of O’Brien et al. (2009), and hence the effect of shorter PFMs is negligible (Fig. 1). Combining PFM mass loss rates (\(\rho_{\text{PFM}, \text{g d}^{-1}}\)) in the channel system determined in this study with data from O’Brien et al. (2009) (Fig. 1) resulted in a significant linear relationship (Eq. (1)). This relationship means an estimation of water flow rate (\(v, \text{m s}^{-1}\)) from PFM mass loss rate (\(\rho_{\text{PFM}, \text{g d}^{-1}}\)) was possible (Table 1). The rationale for the use of PFMs as an in-situ calibration method is that dissolution rate and hence mass loss from gypsum is limited by transport through the water boundary layer (WBL). Increasing external water flow velocity decreases the WBL and increases the mass loss rate (O’Brien et al., 2009). Good agreement was observed between PFM based flow velocities and measurements with a flow meter and velocities that were calculated from the volume rate of flow and the cross sectional area of the channels (Table 1).

\[
v(\text{m s}^{-1}) = \frac{\rho_{\text{PFM}} - 0.61}{12}
\]

The solubility of gypsum and its mass loss rate has previously been observed to vary with salinity and temperature of the ambient water as well as background \(\text{Ca}^{2+}\) and \(\text{SO}_4^{2-}\) levels. These factors have been incorporated into relationships between water flow rate and PFM mass loss rate as a function of ionic strength (O’Brien et al., 2011b). This means co-deployed PFMs can be used as in-situ calibration devices for this sampler in most environmental waters, extending its utility.

3.2. Relationships between sampling rates and water flow velocity

Sampling rates for the PFCs of interest in the flow through channel system have been recently determined and shown to depend on analyte and flow rate (Kaserzon et al., 2013). \(R_s\)'s ranged from 0.09 to 0.29 L d\(^{-1}\) over the flow rate range of 0.02–0.34 m s\(^{-1}\). Expressed on an area-normalised basis, sampling rates ranged from 0.6 to 1.8 L dm\(^{-2}\) d\(^{-1}\). The relationships between \(R_s\) and water flow rate were examined here indirectly, using \(\rho_{\text{PFM}}\). An increase of \(R_s\) (by factors of 1.2–19) with \(\rho_{\text{PFM}}\) was observed for PFPeA, PFHxA, PFHpA, PFOA, PFNA and PFHxS (Fig. 2). For PFHpA and PFOA, however, the effect of flow on \(R_s\) appeared to plateau above \(\rho_{\text{PFM}}\) of 3 g d\(^{-1}\) (corresponding to a water flow rate of 0.20 m s\(^{-1}\)). The \(R_s\) of the larger molecular weight PFSAs and PFCAs (PFDA, PFUnDA and PFOS) did not exhibit any clear relationship with increasing \(\rho_{\text{PFM}}\) (Fig. S1). Therefore, these PFCs were excluded from further consideration in this current study.

Similar behaviour was observed for PFMs with the herbicides atrazine and prometryn, using a Chemcatcher™ passive sampler (O’Brien et al., 2011a) where the sampling rates did not increase with \(\rho_{\text{PFM}}\) above \(\rho_{\text{PFM}}\) and water flow rate thresholds almost exactly the same as observed here. This behaviour was ascribed to a shift in the limiting resistance for chemical uptake from the WBL to transmembrane diffusion (O’Brien et al., 2011a; Vermeirssen et al., 2009).

A nonlinear exponential empirical model (Eq. (2)) best fitted the relationship between \(R_s\) and \(\rho_{\text{PFM}}\) for PFHpA and PFOA (\(R^2\) values 0.95 and 0.97, respectively) (Fig. 2, Table 2) and may be used to correct for in-situ \(R_s\).

![Fig. 1. PFM loss rate (\(\rho_{\text{PFM}}\)) vs measured water flow rate (m s\(^{-1}\)) in this study and O’Brien et al. (2009).](image1)

![Fig. 2. PFC Sampling rates (\(R_s\)) versus PFM mass loss rate (\(\rho_{\text{PFM}}\)). Lines represent predicted \(R_s\) values for PFCs of interest based on a linear equation, Eq. (2) and data in Tables 2 and 3.](image2)
The result of molecular size restricting transport in the polymer and water sampling rate for the larger anionic PFCs (PFHpA, PFOA) at higher assumed with POCIS, however a recent review concluded that very PFNa mass loss rate. WBL control of analyte mass transfer is usually evidenced by no change in sampling rate with external water filled pores. Plots relating to WBL but also by mass transfer through the microporous polyethersulfone membranes due to transport through water-filled pores and the polymer matrix itself (Alvarez et al., 2007; Shaw et al., 2009). Membrane control would be evidenced by no change in sampling rate with external water flow rate or consequently, PFM mass loss rate. WBL control of analyte mass transfer is usually assumed with POCIS, however a recent review concluded that very little was known of the relative magnitude of these resistances (Harman et al., 2012; Vermeirssen et al., 2012). The plateauing of sampling rate for the larger anionic PFCs (PFHpA, PFOA) at higher water flow rates suggests a transition to membrane control (Kaserzon et al., 2013). Increased resistance in this phase may be the result of molecular size restricting transport in the polymer and in the water-filled pores. Plots relating to PFM (Fig. 2.) are broadly in agreement with this.

3.3. Relationship between sampling rates and PRC elimination rate constant ($k_e$)

The use of PRCs with the modified POCIS for PFCs was unsuccessful. Following 15 days of exposure in the channel system, relatively small losses (i.e. < 10%) were apparent for isoproturon-$d_5$ and metolachlor-$d_6$ while a loss of up to 45% of the initial concentration was observed for DIA-$d_5$ (Fig. S2). However, no discrimination with water flow rate was observed for elimination rate constants of spiked DIA-$d_5$ (Fig. S2). Initial concentrations in PRC spiked POCIS ($n = 2$) of DIA-$d_5$, isoproturon-$d_6$ and metolachlor-$d_6$ were 145 ± 5, 225 ± 6 and 567 ± 36 ng g$^{-1}$ respectively. The low desorption losses for isoproturon-$d_6$ and metolachlor-$d_6$ were outside the acceptable loss range for PRCs, i.e. 20–80% (Huckins et al., 2002b). This indicates that these compounds cannot be used as PRCs with the modified POCIS. The loss rate of DIA-$d_5$ in this study is comparable with results from previous studies of other groups, where losses of approximately 60% were observed after 15 days of deployment (Fauvelle et al., 2012; Mazzella et al., 2007). The effects of flow rate on this loss have not previously been investigated however.

Elimination rate constants ($k_e$) calculated under the four water flow rate regimes in this study were almost identical at 0.06 ± 0.01, 0.06 ± 0.01, 0.06 ± 0.01 and 0.06 ± 0.02 d$^{-1}$ (for water flow rates of 0.02, 0.06, 0.16 and 0.34 m s$^{-1}$, respectively) (Fig. 3). These data (Fig. S2) compare well with those from previous studies by others (0.057 ± 0.02 and 0.104 ± 0.009 d$^{-1}$, at water velocities of 0.02–0.03 (i.e. effectively quiescent flow) and 0.05–0.08 m s$^{-1}$ respectively (Fauvelle et al., 2012). However, within the larger flow rate (0.02–0.34 m s$^{-1}$) and PFM mass loss rate range examined in this study we show there is no discrimination between elimination rates. In order for PRCs to be employed as a valid in-situ calibration method, the uptake and release of a compound must follow isotropic first order kinetics, i.e. the same mass transport and rate limiting diffusion steps should apply equally to each. Some previous studies have reported pseudo first order kinetic desorption and isotropic exchange of DIA-$d_5$ with both Oasis HLB and Oasis MAX sorbent phases and therefore concluded it has potential application as a PRC for organic acids (Fauvelle et al., 2012). It was considered that the kinetic exchange of the PRC chemicals was similar to that of analytes (i.e. under aqueous boundary layer control). Results from this current study indicate that this may not be the case for PFSAs and PFCA. If there is a discrepancy between the factors controlling the uptake of analytes and release of the PRC (DIA-$d_5$) with POCIS type samples, the application of these compounds as PRCs may not be appropriate (Harman et al., 2012). Understanding the surface chemistry of sorbents is still an area in which significant future work is needed, especially where PRCs are employed and sorption/desorption is not isotropic. For anionic sorbates and anion-exchange sorbents such as Oasis WAX, coulombic attractions are particularly important and it has been suggested that adsorption dominates absorption (Stephens, 2012; Bauerlein et al., 2012). This is in contrast to PSs for hydrophobic compounds such as the SPMDs, which are characterised by bulk hydrophobic absorption. In addition, binding of non-dissociated analytes and ions occur at different sorption sites than dissociated analytes, which complicates understanding of the situation further (Bauerlein et al., 2012). Alternative external correction methods are required for quantitative assessment of PFCs with this modified POCIS. One additional PRC approach also suggested for polar organic compounds is to co-

### Table 2

<table>
<thead>
<tr>
<th>Compound</th>
<th>$R_s$ (max) ($\text{L d}^{-1}$)</th>
<th>$k_{PFM}$ (d g$^{-1}$)</th>
<th>Std. error</th>
<th>$R^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>PFHpA</td>
<td>0.26 ± 0.02</td>
<td>7.3 ± 4.4</td>
<td>0.02</td>
<td>0.95</td>
</tr>
<tr>
<td>PFOA</td>
<td>0.19 ± 0.03</td>
<td>5.8 ± 3.3</td>
<td>0.007</td>
<td>0.97</td>
</tr>
</tbody>
</table>

$$R_s = R_s(\text{max})(1 - \exp(-k_{PFM}PPM))$$

Here $R_s(\text{max})$ is the maximum sampling rate at infinite flow and $k_{PFM}$ (d g$^{-1}$) is an empirical fitting parameter.

For the remaining PFCs under investigation (PFPeA, PFHxA, PFNA and PFHxS), a linear empirical relationship ($R^2 = 0.74, 0.91, 0.99$ and 0.92, respectively) best described the relationship between $R_s$ and $k_{PFM}$ (Fig. 2, Table 3) suggesting WBL control for these PFCs over the flow rate range investigated. For these PFCs, the slopes and intercepts presented in Table 3 may be used to correct in-situ $R_s$’s. The precision of these predictive relationships is about 0.01 L d$^{-1}$ (Tables 2 and 3).

### Table 3

<table>
<thead>
<tr>
<th>Compound</th>
<th>Intercept ($\text{L d}^{-1}$)</th>
<th>Slope ($\text{g}^{-1}$)</th>
<th>Std. Error ($\text{L d}^{-1}$)</th>
<th>$R^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>PFPeA</td>
<td>0.078 ± 0.02</td>
<td>0.017 ± 0.007</td>
<td>0.02</td>
<td>0.74</td>
</tr>
<tr>
<td>PFHxA</td>
<td>0.118 ± 0.01</td>
<td>0.023 ± 0.005</td>
<td>0.01</td>
<td>0.91</td>
</tr>
<tr>
<td>PFNA</td>
<td>0.165 ± 0.004</td>
<td>0.018 ± 0.002</td>
<td>0.004</td>
<td>0.99</td>
</tr>
<tr>
<td>PFHxS</td>
<td>0.182 ± 0.01</td>
<td>0.016 ± 0.004</td>
<td>0.01</td>
<td>0.92</td>
</tr>
</tbody>
</table>

Fig. 3. PRC (DIA-$d_5$) elimination rate constant ($k_e$) versus water flow rate illustrating the effect of flow rate with this calibration technique.
deploy PRC loaded SPMDs (as the PRC method for these samplers has been verified) to account for differences in exchange kinetics between sampling deployments (Alvarez et al., 2007; Shaw et al., 2009).

4. Conclusions

This study has demonstrated that PFM s can be a useful tool when co-deployed with modified PO CIs for estimating in-situ water flow rates. Together with the empirical models derived in this study, PFMs can be used to correct PFC specific sampling rates and achieve more reliable estimates of environmental concentrations. The low cost and simplicity of use of this device means that it can be easily accessible to users. A PRC approach for the modified PO CIs employed here may not be possible for PFCs or a wider range of analytes.

Acknowledgement

This research was funded by the Australian Research Council’s Linkage Projects scheme (LP0883675), with industry partner support of SEQWater, NMI, DW WA, EPA Victoria, EPA South Australia, Office of Environment and Heritage NSW, Brisbane City Council, GBRMPA, DERM, Queensland Health Forensic and Scientific Services (QHFSS), Griffith University and The University of Queensland. Sarit Kaserzon is also supported by a top-up scholarship from Water Research Australia (WaterRA). The authors thank Juliane Hollender (Eawag) for support, and Christie Bentley (Entox), Jack Thompson (Entox) and Chris Paxman (Entox) for laboratory support. The National Research Centre for Environmental Toxicology (Entox) is a joint venture of the University of Queensland and QHFSS.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.envpol.2013.11.030.

References


