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極性有機化合物被動採樣器、連續式水體監測主動採樣器、與非連續採樣對量測河水中的個人保健用品濃度之比較

A Comparison of Polar Organic Chemical Integrative Sampler, Continuous Aquatic Monitoring and Discrete Sampling on Determining Personal Care Products in River Water

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本論文係陳妍秀君(R01844020)在國立臺灣大學環境衛生研究所完成之碩士學位論文，於民國 104 年 07 月 16 日承下列考試委員審查通過及口試及格，特此證明

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We, the undersigned, hereby recommend that the thesis entitled, A Comparison of Polar Organic Chemical Integrative Sampler, Continuous Aquatic Monitoring and Discrete Sampling on Determining Personal Care Products in River Water, submitted by Yen-Hsiu Chen, be accepted as fulfilling the thesis requirements for the degree of Master of Science in Public Health.

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中文摘要

此研究中所調查之個人保健用品(Personal care products, PCPs)成份在台灣被大量使用，在非連續採樣的河水樣本中廣泛被量測到；硝化/氧合多環芳香烴(Nitrated and oxygenated polycyclic aromatic hydrocarbon, NPAHs and OPAHs)則具有高致突變性與高致癌性，可經由吸附於大氣中懸浮微粒後沉降至環境水體中。

極性有機化合物被動採樣器(Polar organic chemical integrative sampler, POCIS)是一利用吸附劑累積極性待測物數週至數月的被動採樣器。連續式水體監測主動採樣器(Continuous low-level aquatic monitoring, C.L.A.M.)使用一般固相萃取膜作為採集媒介，可沉在水體中連續抽取水樣超過 24 小時的連續採樣器。相較於傳統非連續式水樣採集的樣品只能提供採樣當下的濃度資料，此二種採樣方式皆可提供時間權重平均濃度(time-weighted average, TWA)，能增加待測物濃度高於實驗室偵測極限的機率。

本研究建立極性有機化合物被動採樣器與連續式水體監測主動採樣器的採樣效率系統，以評估此 13 種個人保健用品與 5 種硝化/氧合多環芳香烴是否適用於此兩種採樣方式。評估結果用於量測基隆河河水各待測物含量，並與非連續採樣結果比較。10 種個人保健用品成分可使用極性有機化合物採樣，並可提供半定量數據；另外 3 種只能提供資料；硝化/氧合多環芳香烴則並不適用於極性有機化合物被動採樣。而幾乎所有的化合物皆可被連續式水體監測主動採樣器(除二苯基酮)與非連續採樣取樣分析。在現場採樣結果中，七天非連續採樣平均結果與極性有機化合物被動採樣二者結果與濃度相似。連續式水體監測主動採樣器，因其較大的採樣體積，可量測到較多種化合物；但經過水量平均後，測得結果較非連續採樣與被動採樣器較小。

連續式水體監測主動採樣器在河水較髒的情況下運作，容易塞住改良模組中過濾濾紙，造成流速不穩定及氣泡產生，進而使採樣片性質改變，造成後續分析困難。此設計需要再經過修正與後續現場測試以適應河川環境變化。

關鍵字：硝化/氧合多環芳香烴、採樣效率、採樣方式、個人保健用品、確效參考化合物

ABSTRACT

Personal care products (PCPs) investigated are utilized in great quantity in Taiwan and had been detected through discrete samples ubiquitously. Nitrated and oxygenated polycyclic aromatic hydrocarbons (NPAHs and OPAHs) possess high mutagenicity and carcinogenicity, and adsorb to air particulates through atmospheric deposition entering the aquatic environment.

Polar organic chemical integrative samplers (POCIS) is an in-situ continuous passive sampling methods that utilize sorbents to accumulate analytes over weeks to months. Continuous low-level aquatic monitoring (C.L.A.M.) sampler is a submersible continuous sampler that draws water through ordinary solid-phase extraction disk over about 24 hours. Compared to discrete sample that provide information only at the sampling instant, both samplers provide time-weighted average (TWA) concentration of analytes in water, and increase the probability of analyte concentrations to be above the laboratory detection limits.

In this study, calibration system was set up to evaluate whether the 13 PCPs and 5 NPAH/OPAHs were suitable for POCIS and C.L.A.M. sampling. These evaluation results were used to determine the concentration of target analytes in Kee-Lung river, and the sampling results were compared with discrete sampling. Ten PCP compounds were suitable for quantitation and three PCPs were only allowed for qualitative results in POCIS; NPAHs/OPAHs were not suitable for POCIS sampling due to their no or low uptake. Almost all analytes can be sampled by C.L.A.M. (except benzophenone) and grab sampling. The results of field deployment showed that the concentrations and detected number of analytes in grabbed were in agreement with those in POCIS samples. C.L.A.M. may detect more analytes due to the larger passed volume of river water, but

the detected concentrations were smaller than other two sampling methods after normalization.

The filters in C.L.A.M. may be clogged in river environment with large particles and high turbidity, causing unstable flow rate. The extracted disk may be dried and wet repeatedly during the sampling period to make further analysis difficult. The modifications of replacement of original disk to a two-stage filter assembly need further improvements and field deployments to accommodate to river environment.

Key words : Nitrated and oxygenated polycyclic aromatic hydrocarbons; sampling rates, sampling method; personal care products; performance reference compounds

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Chapter 1 Introduction

1.1 Personal care products and nitrated/oxygenated polycyclic aromatic hydrocarbons

Personal care products (PCPs) generally refer to product components that are used by individuals for health care or cosmetic reasons, such as non-prescriptive drugs, cosmetics, sun-screen products, preservatives and fragrances. PCPs represent a diverse group of polar organic chemicals that may affect certain physiological functions in certain pathways. These compounds are concerned because they are used in high amount, and also pass through wastewater treatment and enter the aquatic environment. Some PCPs display persistent and bioaccumulative ability; others have short half-life, but their continual discharge into the environment may allow them to exhibit similar exposure potential. Although most of the main biological functions of PCPs are known, there might be unpredicted and non-targeted side effects. The risks may lie on the low-level, chronic exposure and the subtle effects may not be prominent from the beginning [1].

Analgesics, such as acetaminophen, ketoprofen, naproxen and ibuprofen have widely been used due to easy access and have been ubiquitously found in surface water[2-5]. High water solubility and relatively low elimination from wastewater treatment, along with large amount of use cause them to enter freshwater system [2, 6]. The concentrations detected from river water are from few to thousands of ng/L [3, 4].

Caffeine is a common food ingredient for beverages and food products, and is used as a stimulant in medicine for influenza and anti-flammatory drugs. It has prominently been detected in surface water, the concentration detected may be up to 29.1-786 ng/L

in China, 506 ng/L in Taiwan, 224.8 ng/L in USA[7, 8].

Parabens are esters of para-hydroxybenzoic acid, and all compounds share the same skeleton with differed ester groups. They are widely used as preservatives in cosmetics, food products and pharmaceuticals. Parabens are used individually, or in combination, and the most commonly used parabens are methyl paraben and propyl paraben [9]. Parabens cause estrogenic responses in vitro and in fish. The risks were low as the effective concentration was 1000 times more than preliminary measured concentrations in surface water [10, 11].

UV filters, such as benzophenone and oxybenzone, are added to sunscreen products to alleviate the damage of sunlight on skin. They also act as stabilizers in cosmetics, rubber, and plasticizers to prevent polymers from photodegradation [12]. As UV filters are added to the formulation at mg/g level and the relative products need to be applied in large quantities on skin, the main sources of discharge are through washing off the commercial products from the human body directly to the waterbodies during recreational activities and indirectly to the wastewater treatment plant [13]. Oxybenzone has been frequently reported exhibiting endocrine-disrupting effects in vitro and in vivo, and has shown adverse effects on reproduction and development in algae, invertebrates, and fish [14]. The concentrations of UV filters differed a lot by location, season, and frequency of human recreational activities, ranging from no detection to hundreds of ng/L [12, 15].

N,*N*-Diethyl-*meta*-toluamide (DEET) is the most common active ingredient in insect repellents, and have been shown to be slightly neurotoxic to aquatic animals [16]. The concentrations reported by several studies are 0-1,292 ng/L inGermany and 119 ng/L in Taiwan [8, 17].

Nitrated and oxygenated polycyclic aromatic hydrocarbons (NPAHs/OPAHs) are

derivatives of polycyclic aromatic hydrocarbons (PAHs) modified with nitro and carbonyl functional groups. These derivatives are formed by direct photolysis of parent PAHs, and the reaction between PAHs and atmospheric oxidants, such as nitrogen oxides, ozone, and hydroxyl radicals. They may also come from emissions from diesel engines and incomplete combustion of diesel fuels. NPAHs/OPAHs are less volatile and more polar than their parent PAHs, leading to partitioning to particulate matters, and may enter the aquatic environment through atmospheric deposition and surface runoff [18]. A recent study showed that wastewater treatment plant effluent was an important source of OPAHs, contributing 83.5% total mass to the receiving river [19].

NPAHs and OPAHs are more mutagenic and carcinogenic than original PAHs [20, 21]. Moreover, OPAHs may cause allergic diseases through generating reactive oxygen species to cause oxidative stress [22, 23]. Previous studies reported the concentrations of NPAHs and OPAHs at no detection to 4.6 ng/L in river water; the concentrations of OPAHs in WWTP effluents are up to 190 ng/L [18, 19, 24, 25]. According to our previous study, NPAHs and OPAHs were not detected in the water of Kee-Lung river water through traditional discrete sampling [8].

1.2 Polar organic chemical integrative sampler (POCIS)

Passive samplers are sampling devices that requires no moving parts, electricity or fuels, and have the capability to sample a relatively longer period to provide time-weighted average (TWA) concentrations of analytes. The most commonly used passive samplers for pwater sampling are semipermeable membrane device (SPMD) and polar organic chemical integrative sampler (POCIS). The criterion for choosing SPMD or POCIS for sampling is relied on log octanol-water partition coefficient (kow)

of target analytes. Although there is overlapped applicable range between the SPMD and POCIS: the compounds that are neutral and with kow greater than 3 are more suitable for SPMD sampling; the analytes with kow smaller than 3 are more applicable on POCIS. Some compounds that are related to wastewater effluents, such as steroidal hormones, some analgesics, and other chemicals, have Kow greater than 3, but are preferentially sampled by POCIS.

POCIS is composed of solid-phase sorbent sandwiched between two polyethersulfone (PES) membranes (Figure 1(a), p.36). Multiple types of membrane were tested and PES membrane was chosen for a good combination of high uptake rates of analytes, durability of long-term sampling, and resistance of biofouling. As PES membrane is not capable of sealing, the outer two stainless compression rings are used to seal the sorbent between the membranes; screws and bolts are used to fix the membranes and rings [26].

There are two kinds of commercialized standard POCIS: pesticide POCIS and pharmaceutical POCIS. The difference between the two POCIS is the sorbent packed: pesticide POCIS is constituted by Isolute ENV+ (a hyper-crosslinked hydroxylated polystyrene-divinylbenzene copolymer) and Ambersorb 1500 or 572 (a carbonaceous adsorbent) dispersed on neutral porous S-X3 BioBeads; pharmaceutical POCIS uses the universal sorbent Oasis HLB, that is the copolymer of hydrophlilc N-vinylpyrrolidone and lipophilic divinylbenzene. For highly polar organic chemicals, such as pharmaceuticals that have multiple functional groups, their recoveries from pesticide POCIS may be problematic. As a result, pharmaceutical POCIS is more suitable as the accumulative medium [27].

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POCIS provides time-weighted average (TWA) concentrations of organic chemicals during the sampling period through their specific sampling rates (R_s) estimated from the uptake pattern of analytes. The uptake pattern of the chemicals on the POCIS follows the first-order kinetics theoretically, which is typically a linear integrative stage at the beginning, and followed by a pseudolinear stage, then finally reached an equilibrium stage [28], which could be described by equation

$$C_s = C_w \frac{k_u}{k_e}(1 - e^{-k_e t}) \quad (1)$$

C_s is the analyte concentration in the sorbent at time t (μg/g); C_w is the TWA analyte concentration in the water (μg/L); k_u is the uptake rate constant of analyte on the sorbent (L/g/d); k_e is the exchange rate constant of analyte from the sorbent (1/d); t represents time (d)

The TWA concentration is calculated at the linear stage of POCIS uptake, in which the elimination rate of the analyte from the sorbent is negligible compared to uptake rate, that is, $k_u >> k_e$. As k_e is negligible, Equation (1) could be further simplified, and k_u could be replaced by the relationship between sampling rates, time, and sorbent quantity.

$$C_s = \frac{C_w R_s t}{M_s} \quad (2)$$

R_s represents sampling rates (L/d); M_s is the sorbent mass in POCIS (g)

According to the first-order kinetics, the separation point between linear and

pseudolinear regimes is $t_{1/2}$, which is the time needed to reach a half of the equilibrium concentration and the time to reach the end of linear stage. The calculated R_s applied to the time before $t_{1/2}$, that is, the integrative stage of POCIS uptake.

$$t_{1/2} = \frac{ln2}{k_e} \tag{3}$$

The mass transfer of the analytes from bulk water to sorbent includes the movement from bulk water to the water boundary layer (WBL) outside the PES membrane, diffusion through the WBL, diffusion through the PES membrane to reach the sorbent. Each steps have their barrier resistance, and these resistances are assumed to be additive. Any reduction in resistance causes the sampling rates to increase.

As water moves along the surface, the momentum of water is reduced due to the surface friction and a water layer is built. The built water layer then further attenuates the momentum of water at larger distance, building another water layer. This process repeats and results in a viscous layer with increasing thickness along the surface, forming a water boundary layer (WBL). The same mechanism applies for the transfer of analytes from bulk water to the sorbent packed in PES membranes to build a concentration boundary layer. If the distance is large to reach to a steady-state concentration, the concentration is not dependent on distance.

The sampling rates differ for various compounds as their properties and their interaction with PES membrane are different. The sampling rates are also dependent on whether the uptake is controlled by water boundary layer (WBL) outside the PES membrane or the PES membrane.

If a compound is under WBL control, the sampling rates could be represented as:

$$R_s = \left(\frac{D_w}{\delta_w}\right) A \tag{4}$$

D_w is the diffusion coefficient in water; δ_w represents the effective thickness of the WBL, A is the surface area of the sampler.

If a compound is under membrane control, the mass transfer behavior of the analyte in the membrane is more complicated. The behavior may be a biphasic transport with water-filled pores and the membrane matrix. Alvarez et al. suggested that for molecules with molecular weight smaller than 400 Da, the dominant determinant in both kinds of transport is K_{mw}, the equilibrium membrane-water partition coefficient.

For WBL controlled compounds, any treatment that reduces the resistance of WBL would cause sampling rates to increase, e.g., increasing flow rate. For membrane controlled chemicals, the sampling rates are not influenced by changing the flow rate.

Sampling rates are usually obtained by exposing the sampler to a well-defined calibration system with known and relatively constant concentrations of spiked compounds. The calculated sampling rates are specific for each compound.

There are four main calibration approaches: static depletion, static renewal, flow-through system and in-situ calibration. Static depletion and static renewal refer to a closed system with chemical standards spiked at the beginning or at constant intervals and have relatively simple operation procedures. Static depletion method is only suitable for analytes that are not degraded or adsorbed during the calibration period, and requires unreasonably high initial spiked concentrations (5~10 μg/L). Estimation of sampling rates of this method is through monitoring decreased concentration of spiked water with time, and application of positive controls was essential to estimate the analytes trapping in PES membranes. Static renewal design is to spike chemical standards over a certain period of time for maintaining the system at relatively constant concentrations of the chemicals, which is relatively labor-intensive. The flow-through system is to pass through spiked water of known chemical concentrations with constant

linear velocity to mimic the real water flow. Lower spiked concentration (usually in ng/L) is enabled to imitate authentic environmental conditions, but the system consumes relatively large volumes of water and chemical standards. The above three designs provide constant laboratory conditions for evaluating the adsorption pattern of POCIS. In-situ calibration features a calibration device on-site for simulating future field sampling, and the analytes have to exist in the water originally. Although this system may provide more accurate sampling rates, it is subject to the fluctuations of chemical concentrations in water and requires intensive measurements to obtain reliable TWA analyte concentrations in water. Sampling rates for static renewal, flow-through system, and in-situ calibration were estimated from measured water concentration and accumulation in sorbent [28, 29].

Many factors were assessed if they were possible to influence the sampling rates during the calibration period. As the sampling rate is proportional to the effective exposure surface area of the device, the optimized surface area and sorbent mass ratio of a standard POCIS is defined as ~229 cm^2/g (sampler diameter: 54 mm, surface area 45.8 cm^2, sorbent mass: 200 mg) [30].

Togola et al. showed that the sampling rates obtained on pharmaceuticals under two different concentrations (0.5 and 5 μg/L) after a 7-day static renewal calibration procedure were not significantly different [31]. Zhang et al. gained similar results on endocrine disruptor chemicals (EDCs) under 7 different concentrations (10-1000 ng/L) from a 10-day flow-through system [30]. Therefore, different spiked analytes concentration does not significantly influence estimated sampling rates.

The combination of calibration conditions, experimental parameters of water and R_s calculation methods could affect R_s widely [28]. The calibration conditions and experimental parameters of water include the agitation conditions, temperature, pH,

salinity, fouling, and exposure matrix.

Agitation conditions are often mimicked by stirring exposure with a stir bar in a static system or using the flow-through design. Several studies reported that the sampling rates were higher (under turbulent conditions than those under quiescent or very low flow rates (0.1 cm/s) conditions. For the effects of different flow rates (> 3-34 cm/s), sampling rates seem to increase by a factor smaller than 2 [32-36]. Most compounds investigated had higher sampling rates with higher flow rates, suggesting these compounds were under water boundary layer control; only a few chemicals had no significant changes on sampling rates, indicating that they are under membrane control[26]. This phenomenon seemed to be compound-specific, as the more hydrophobic compounds exhibited the more pronounced effects.

Togola et al. showed that the sampling rates increased with temperature (15℃, 21 ℃) on pharmaceuticals and EDCs [31]. Li et al. demonstrated that sampling rates of pharmaceuticals and EDCs increased (≤ two-fold) as water temperature (5℃, 15℃, 25 ℃) rises [37]. Ly et al. reported the similar trends on pharmaceuticals on increasing temperature (15℃, 20℃, 25℃)[36]. This nearly two-fold change may be explained by the predicted 50% increase of diffusion coefficients over a temperature range of 20℃ [26, 29].

Although the pH during the calibration period is usually stable, the change in pH may affect sampling rates. Li et al. showed that sample rates are maximized when chemicals are primarily in their neutral form and changes between pH 3-9 were less than 3-fold for pharmaceuticals and EDCs [38]. Consequently, for acidic chemicals, an environment at low pH may increase the sampling rates; whereas for basic compounds, the influence is opposite; as for neutral analytes, the sampling rates are not influenced

by pH [38].

Harman et al. reported that biofouling on the PES membranes could increase the sampling rates of alkylphenols, opposite to the effects on SPMDs [39]. To the best of our knowledge, there were no other groups of chemicals investigated for the effects of biofouling in POCIS.

Deployment of POCIS sampling may be conducted in various aquatic environments, and exposure matrix of calibration system may not be the same with the sampling place. Most sampling made measurements in freshwater systems, such as rivers, lakes, and groundwater, with a few in marine environment and some near or inside the wastewater treatment systems. Sampling rates estimated from Milli-Q water or tap water in laboratory conditions may differ significantly from those obtained from the matrix [36]. Furthermore, specific considerations should be applied on different matrix, for example, salinity effects in marine water.

Estimation methods of sampling rates may also influence Rs values. Macleod et al. adopted a calculation method through only the slope of analyte concentration decrease over the calibration period. This method required suitable static renewal period and mass balance investigation of spiked analytes to gain sampling rates [40]. Morin et al. set up a flow-through system to evaluate if a chemical is suitable for POCIS sampling. The accumulation patterns were evaluated through fitting concentration factor (CF) versus time plot to 1^{st}-order kinetics curve. Concentration factor is defined as the ratio of chemical standards concentration in sorbent and in water at the measured time. The accumulative kinetic patterns of uptake are categorized into 4 types: chemicals with curvilinear or with an inflexion point uptake kinetics are suitable for POCIS sampling; while random or low ($CF < 3.0$) patterns are not [41].

The R_s calculated through laboratory conditions may not represent the real in-situ

R_s in field sampling; therefore, corrections are preferred using appropriate performance reference compounds (PRCs). PRCs are used to adjust R_s for alleviating the influences resulting from chemical fluctuations. The PRC approach is based on the overall uptake kinetics of both analyte and PRC, and the mass transfer are identical in both directions. Although this approach is proved to be suitable on SPMDs, it is not easy to find suitable PRCs for a POCIS system [29, 42].

PRCs are preloaded in the sorbent before packing and sampling, and the dissipation rates of the compounds from the sorbent were evaluated and compared the uptake patterns with corresponding chemicals. Because PCPs represent multi-classes of compounds, it is unlikely to use a single chemical as the PRC. Consequently, isotope-labeled standards would be good candidates as PRCs.

Macleod et al. have shown that the measured concentrations in water with agreement with those using POCIS samples on most detected pharmaceuticals (two sites, total 13 compounds) [40]. A recent study has demonstrated that the use of POCIS detected 21 additional pharmaceuticals (total 141 chemicals) than those using grab samples, while three PCPs were only detected by grab samples [43]. Zenobio et al. found that POCIS detected some relatively hydrophobic ($k$ow > 4) compounds that were not detected in grab samples. These results suggested that POCIS may detect more compounds than grab samples through relatively long-term monitoring.

1.3 Continuous low-level aquatic monitoring (C.L.A.M.)

Continuous Low-level Aquatic Monitoring (C.L.A.M.) is a submersible, active sampler (Figure 3(a), p.37) developed by C.I.Agent Solutions (Louisville, KY, USA). The C.L.A.M. draws water through general solid-phase extraction (SPE) disks at flow

rate < 70 mL/min to continuously capture organic compounds in water. The device weighs about 500 grams, and utilizes four AA-size batteries for continuous sampling about 36 hours. The types of SPE disks are chosen depending on the characteristics of target analytes, and the sampling capacity is limited to about 100 L of water currently. The sampling time is limited by battery life and disk capacity, and usually is shorter than 24 hours. The disks need to be pre-conditioned before sampling and are kept in wet condition after the deployment till analysis. The sampler provides integrated partitioned mass of analytes between the adsorbent and water over the sampling period and the sampled water volume is estimated through measuring the flow rate before and after the deployment [44].

To the best of our knowledge, there is only one peer-reviewed publication involving the utilization of the C.L.A.M. sampler. The study compared POCIS, C.L.A.M., and discrete sampling results at two sites downstream of the outfall of a wastewater treatment plant (WWTP) on a variety of different classes of compounds. Regarding the number of analytes detected, discrete samples were significantly different from other two kinds of samples. This result illustrated that the sampling methods were the most important factor on the detection of a chemical, regardless of the sampling site and detected concentration. On the other hand, the sampling site was the most significant factor on the detected concentration of an analyte, and the second significant factor was the sampling method. Among the three kinds of samples, C.L.A.M. obtained more positive results than POCIS and discrete sampling due to its larger sample volume. Concentrations measured by C.L.A.M. were generally lower than other two sampling approaches, this phenomenon may result from the partial loss of analytes on the HLB disk during the sampling process. The study concluded that all three methods can sample a wide range of compounds and each method had its own advantages and

disadvantages, while C.L.A.M. performed better on the detection of less polar chemicals than the other two methods [45].

1.4 Research objectives

The purpose of this study was to compare the sampling of PCPs and NPAHs/OPAHs in water using POCIS samplers, a modified C.L.A.M. device, and grab sampling. A static renewal calibration system of POCIS with stir-bar agitation was established to evaluate if POCIS is applicable to sample 13 PCPs and 5 NPAH/OPAHs. The uptake pattern of each analyte was plotted through a 14-day calibration experiment to fit the 1^{st}-order kinetics and sampling rates were estimated for analytes with specific uptake patterns to use in field sampling. Because the characteristics of the analytes vary a lot, the PRC evaluation focused on isotope-labeled internal standards rather than a single compound.

The POCIS, C.L.A.M., and grab sampling methods were simultaneously applied to a field sampling spot during the same period. Both POCIS and C.L.A.M. provide TWA concentrations, but are significantly different in sampling time and volumes. Results from the three sampling methods were compared with each other for providing information on further sampling strategies and improvements on PCPs and NPAHs/OPAHs in water.

Chapter 2 Methods

2.1 Reagents, material, and apparatus

HPLC-grade methanol, dichloromethane, acetone for extraction and LC/MS-grade methanol for instrumental analysis were in J.T. Baker brand purchased from Avantor (Center Valley, PA, USA). Milli-Q water was produced from a Milli-Q integral water purification system equipped with a 0.22 μm filter (Merck Millipore, Billerica, MA, USA). Ammonium acetate (≥ 98%, solid), acetic acid (≥ 99.8%, liquid), anisole (> 99.7%, liquid) were purchased from Sigma-Aldrich (St. Louis, MO, USA).

Acetaminophen, caffeine, finasteride, ibuprofen, oxybenzone, methyl paraben, ethyl paraben, propyl paraben, butyl paraben, naproxen, ketoprofen, 1-nitropyrene, 2-nitrofluorene, 9-nitroanthracene, 5,12-naphthacenequinone, ketoprofen-D_3 (> 96%, powder) and DEET (> 96%, liquid) were purchased from Sigma-Aldrich. Benzophenone (99%, powder) was obtained from Alfa Aesar (Heysham, England, UK). 7-nitrobenz[26]anthracene (~100%, powder) was provided by AccuStandard (New Haven, CT, USA). Ibuprofen-$^{13}C_3$, oxybenzone-$^{13}C_6$ (99%, 100 μg/mL in acetonitrile), naproxen-^{13}C-D_3 (98%, 100 μg/mL in acetonitrile), benzophenone-D_{10}, atrazine-D_5 (98%, 100 μg/mL in nonane), caffeine-$^{13}C_3$ (99%, 100 μg/mL in methanol), DEET-D_6 (98%, 100 μg/mL in CD_2Cl_2), 2,4,5-trichorophenoxyacetic-$^{13}C_6$ (99%, 100 μg/mL in dichloromethane), methyl paraben-$^{13}C_6$, n-butyl paraben-$^{13}C_6$ (99%, 1 mg/mL in methanol), 9-nitroanthracene-D_9 (98%, 50 μg/mL in toluene), 1-nitropyrene-D_9 and 2-nitrofluorene-D_9 (98%, 50 μg/mL in toluene-D_8) were purchased from Cambridge Isotope Laboratories (Andover, MA, USA). Acetaminophen-D_4 (100 μg/mL in methanol) was obtained from LGC Standards (Teddington, Middlesex, England, U.K.).

The POCIS sampler and metal holders were purchased from Environmental

Sampling Technologies (St. Joseph, MO, USA). Hydrophilic polyethersulfone (PES) SUPOR 100 membrane filters (diameter: 140 mm, pore size: 0.1μm) were purchased from Pall Corporation (Port Washington, NY, USA). PEP Cleanert adsorbent (80-100 μm, average pore size 70 Å, surface area 600 m^2/g) was obtained with Bonna-Agela Technologies (Wilmington, DE, USA). Empty polypropylene solid-phase extraction (SPE) tubes with polyethylene frits were provided by Supelco (St. Louis, MO, USA).

The C.L.A.M. sampler was purchased from C.I.Agent Storm-Water Solutions (Louisville, KY, USA). Atlantic HLB-M disks were purchased from Horizon Technology (Salem, NH, USA). Two-stage filter assembly that used to replace the original disk assembly developed by C.I.Agent was obtained from Savillex (Eden Prairie, MN, USA).

2.2 POCIS

POCIS is composed of solid-phase sorbent sandwiched between two polyethersulfone (PES) membranes. The outer two compression rings are used to seal the sorbent between the membranes; screws and bolts are used to fix the membranes ((a), p.**錯誤! 尚未定義書籤。**). Triplicates of POCIS are usually set up on the stainless holder for field deployment (**錯誤! 找不到參照來源。**(b), p.**錯誤! 尚未定義書籤。**).

In this study, sorbent PEP Cleanert, polydivinylbenzeone with functionalized vinyl pyrrolidone, from Bonna-Agela Technology was used instead of Oasis HLB.

2.2.1 POCIS laboratory calibration

Two hundred milligrams of pre-cleaned PEP Cleanert sorbent was weighed and sandwiched between two 0.1-μm PES membranes to be held in place by two stainless

compression rings (surface area: 45.8 cm^2). Bolts and screws were used to tighten the compression rings and secure the sorbent. Prepared POCIS were wrapped in aluminum foil pre-cleaned by methanol and stored at 4℃ until deployment. Dissipation of a 24-hour period was done to account for natural decay and adsorption to the glassware to evaluate the concentration changes in calibration system. Dissipation is defined as the analyte loss that is not caused by adsorption to adsorbents, such as adsorption to the tank or hydrolysis and photolysis.

The exposure calibration method was a turbulent renewal system, consisted of a cylindrical glass tank (up to 2.5 L) filled up wtih Milli-Q water that was freshly spiked with 3 μg/L (50 μg/mL standard mixture in methanol, spiked 120 μL) concentration of each analyte and the water solution was refreshed every 24 hours. All POCIS were pre-wet in Milli-Q water for 24 hours. Triplicates of POCIS were immersed into the system for 1, 2, 3, 5, 7, 14 days. The water was stirred by a stir bar at 500 rpm to mimic the agitation conditions. One additional POCIS was applied in non-spiked Milli-Q water in order to serve as a negative control; another was applied in air to account for exposure to airborne contaminants during immersion period. During the calibration experiment, physical parameters of the system, such as temperature, pH, conductivity, dissolved oxygen (DO) were measured. Spiked solutions at the end of exposure were analyzed every day.

2.2.2 PRC evalutation

Five grams of the PEP Cleaner sorbent was spiked with 1 μg/g of isotope-labeled internal standards to evaluate if the PRC approach is feasible. The isotope-labeled standards (4 μg/mL in methanol, spiked 1.25 mL) were added into 30 mL of acetone,

and the solution was added into pre-weighted sorbent to achieve the spiked concentration. The sorbent–acetone mixture was stirred and acetone was evaporated in a hood after overnight to dryness. The spiked sorbent was packed as usual POCIS, and the packed POCIS were deployed in non-fortified Milli-Q water as the calibration system described above.

2.3 C.L.A.M.

The original C.L.A.M. extraction assembly was replaced with a two-stage filter assembly with PFA clamp (Savillex, MN, USA) (Figure 3(d), p.29) for repetitive uses. A glass fiber filter was placed on the first stage gripper for filtering, and a disk was positioned at the second stage gripper for trapping analytes (Figure 3(c), p.37). The space between the first stage gripper and the assembly body was filled with small pieces of glass filter to prevent from deposition of large particles in water.

As the characteristics of PCPs and to match with POCIS sorbent, HLB disks were chosen. HLB disks were pre-conditioned with 10 mL of dichloromethane, 10 mL of methanol, and 10 mL of water sequentially. The disks were preserved in wet condition at 4℃ in a refrigerator until deployment.

C.L.A.M. sampler was tested in 21-hour operation in a 70 L tank with Milli-Q water spiked with 30 ng/L analytes (5 μg/mL standard mixture in methanol, spiked 60 μL for 10 L water each time). Flow rates of the sampler were measured before and after deployment for estimating the total passed volume of the spiked water.

2.4 Field sampling

POCIS, C.L.A.M., and grab sampling were applied at Melti Pier in Melti Riverside

Park beside the Kee-Lung River (Figure 4, p.38). The sampling was conducted during June 18th to 25th in 2015. Each of two sets of triplicate of POCIS was set up in a stainless cage at 1.5-m depth under the floating pier during the sampling period (). One POCIS was deployed for each set as a field blank at the same position in air to account for contamination during transport and contaminants during the deployment period. C.L.A.M. was chained on the pier bridge and deployed for 21 hours per day during the sampling period ((b), p.38). As the sampling site is a tidal area, the depth was made certain to be sufficient for C.L.A.M. sampling during the lowest tide. (depth range: 30 cm to 2.2 m). Flow rates of the C.L.A.M. deployment were measured before and after the sampling to estimate the sampling volume. Deployed POCIS and C.L.A.M. disks were packed with aluminum foil pre-rinsed with methanol, placed in an ice bucket during transportation and were stored at -20℃ and 4℃, respectively.

Three liters of river water samples were collected every 24 hours during the same period using a stainless steel sampler and were then transferred to one-liter deactivated amber glass bottles immediately. Water quality parameters, such as pH, conductivity, total dissolved solids, salinity, and dissolved oxygen (DO) were measured on site. The collected river water samples were acidified with acetic acid to pH 3.0 to prevent from biodegradation and were preserved at 4℃ till analysis within 7 days.

2.5 Instrumental analysis and data analysis

2.5.1 Sample preparation

The POCIS samples were left at room temperature for 30 minutes before the analysis, then the surface was rinsed with Milli-Q water. The POCIS were disassembled and the PES membranes were rinsed with 4 mL of milli-Q water for transferring the

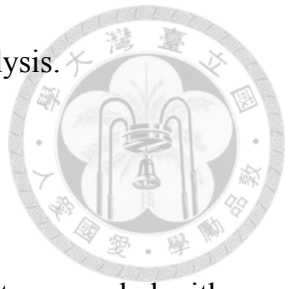
sorbent into pre-weighed empty 6-mL polypropylene SPE cartridges with bottom PTFE frits. The packed sorbent was dried for one hour using a vacuum flow and was spiked with 200 ng of isotope-labeled internal standards (4 μg/mL in methanol, spiked 50 μL). Elution was carried out with 2 × 2 mL of methanol and 2 × 2 mL of methanol/acetone (30:70, v/v). After extraction, the adsorbent was dried with a vacuum flow and was weighed in order to know the exact mass of packed adsorbent in the cartridges.

After drying with nitrogen for 6 hours, the C.L.A.M. disks were added with 200-ng isotope-labeled internal standards. The dried disks were eluted with 2 × 5 mL of methanol and 2 × 5 mL of methanol/dichloromethane (50:50, v/v).

Five hundred milliliters of each discrete water sample was spiked with 200 ng of isotope-labeled internal standards in methanol, and the samples were shaken at 150 rpm for 30 minutes before sequentially filtered through glass filters (pore size 1.0 μm, diameter 90 mm, ChromTech, MN, USA) and a polyvinylidene fluoride (PVDF) membrane (pore size 0.45 μm, diameter 90 mm, ChromTech, MN, USA). The water samples were processed with SPE-DEX 4790 automated solid-phase extraction system (Horizon Technology, Salem, NH, USA). The Bakerbond Speedisk PolarPlus C18 disks (50 mm i.d., J.T. Baker, Center Valley, PA, USA) for extraction were pre-washed with 10-mL methanol/dichloromethane (50:50, v/v), then were sequentially conditioned with 10-mL methanol and 10-mL reagent water. After the water samples passing through and dried for 15 minutes with nitrogen gas, the disks were eluted with 2 × 5 mL of methanol and 2 × 5 mL of methanol/dichloromethane (50:50, v/v).

For all three sampling methods, the eluates were concentrated to barely dry using a Speedvac concentrator (Thermo Savant SPD 1010, Holbrook, NY, USA), and were reconstituted with 500 μL of methanol. The solutions were filtered through a Millex Simplicity filter (hydrophilic PTFE, pore size 0.20 μm) using a Samplicity fitration

system (Merck Millipore, Darmstadt, Germany) for instrumental analysis.



2.5.2 Instrumental analysis and data analysis

All extracts were analyzed with a Waters ACQUITY UPLC system coupled with a Quattro Premier XE triple-quadrupole mass spectrometer fitted with an ESI or APPI interface. Chromatographic separations of 11 compounds, including four NPAHs (2-nitrofluorene, 1-nitropyrene, 7-nitrobenz[a]anthracene, 9-nitroanthracene), one OPAH (5,12-naphthacenequinone) and six basic PCPs (acetaminophen, caffeine, DEET, benzophenone, Finasteride, oxybenzone) were achieved by a Kinetex PFP column (50mm × 2.1 mm, 2.6 μm, Phenomenex, Torrance, CA, USA) (Appendix 1, p.57). Among the 11 analytes, two NPAHs (7-nitrobenz[a]anthracene, 9-nitroanthracene) were detected using APPI (+); while other nine analytes were analyzed with ESI (+) (Appendix 2, p. 58). The rest seven acidic PCPs (Methyl paraben, ethyl paraben, propyl paraben, butyl paraben, ketoprofen, naproxen, ibuprofen) were separated by a Waters CORTECS C18 (30 mm × 2.1 mm, 1.6 μm) (Appendix 1, p. 57), and were detected using ESI(-) mode (Appendix 2, p. 58). Acquisition was performed at multiple reaction monitoring (MRM). The two most abundant ion transitions of each analyte were selected as the quantitative and confirmatory ions, respectively (Appendix 3, p.59). MassLynx 4.1 (Waters) was used for data acquisition and processing, and further data analysis was done by Microsoft Excel 2013 (Microsoft) and OriginPro 8.5.1. (OriginLab, Northampton, MA, USA).

2.6 Method validation

2.6.1 Method validation

Accuracy and precision were determined by pre-spiked Milli-Q water at three pre-spiked levels (100, 300, 600 ng/L). Accuracy was presented as the quantitative bias by comparing the measured concentrations to the spiked concentrations; precision was expressed as the relative standard deviations of the measured differences of the replicates (n = 3).

Extraction efficiency was defined as the slope ratio of regression curves of pre-spiked samples to those of chemical standards. Five levels of analytes (50, 100, 300, 600, 1000 ng/L) were prepared for chemical standards and pre-spiked blank samples for establishing regression curves (n = 3 for each level).

2.6.2 Quality assurance and quality control

For disk-SPE samples, a reagent blank, a sample duplicate and a QC spike sample were analyzed with collected water samples at each batch (20 samples). For C.L.A.M. samples, a reagent blank and two reagent spikes was applied in every batch (10 samples).

Regarding POCIS, a fabrication blank, a laboratory blank, and two reagent spikes and a field blank was also analyzed in every batch (15 samples). A fabrication blank is the blank that sorbent was packed concurrently with the field-deployed POCIS, It was stored at -20℃ till processing with field-deployed samplers. This fabrication blank represented the contaminants through packing process, sample preparation, and instrumental analysis.

Cartridge tubes, frits and aluminum foils were rinsed with methanol before use or

packing. Glassware was soaked with 1.0 M sulfuric acid$_{(aq)}$ for six hours, and was rinsed with Milli-Q water, acetone, and methanol before use.

A standard solution of 200 ng/mL in methanol was analyzed every 20 injections to check the retention time and signal intensity of the instrument.

2.7 Calculation method of sampling rates (R_s)

As TWA concentration is calculated at the linear stage of POCIS uptake with relationship as Equation (2), and the equation could be rearranged to

$$\frac{C_s}{C_w} = CF = \frac{R_s t}{M_s} \quad (5)$$

C_s is the analyte concentration in the sorbent at time t (μg/g); C_w is the TWA analyte concentration in the water (μg/L); CF is the concentration factor (L/g); R_s represents sampling rates (L/d); M_s is the sorbent mass in POCIS (g); t represents time (d)

C_s, C_w, M_s, t were obtained from the calibration experiment (6 points, 1, 2, 3, 5, 7, 14 days), then CF were calculated and used to draw the CF-t plots. The plotted curves were fitted first-order kinetic equation to obtain k_e and to calculate $t_{1/2}$. Through the obtained $t_{1/2}$ values, the CF maximum value could be calculated from the curve. Thus, the sampling rate can be calculated through Equation (5) [41].

Chapter 3 Results and Discussion

3.1 Method validation

For PCPs, the extraction efficiencies of POCIS were 80.7-122% (Table 2, p.47). The lowest was the extraction efficiency for acetaminophen (80.7%) and benzophenone (122%) had the highest value. For NPAHs and OPAHs, the extraction efficiencies were 94.3-110%.

The extraction efficiencies of C.L.A.M. for PCPs were 81.7-103% (Table 2, p.47). Extraction efficiency for acetaminophen (96.6%) and caffeine (99.4%) were better than those of POCIS. The disk used in C.L.A.M. would be conditioned first, and kept in wet condition till extraction, while POCIS sorbent was spiked and dried, and kept in dry condition till extraction, resulting lower extraction efficiencies in POCIS.

Compared to PCPs, extraction efficiencies for NPAHs and OPAHs were much lower. Qiao et al. reported similar extraction efficiencies combining C18 and HLB SPE cartridge process on NPAHs that extraction efficiencies of 2-nitrofluorene, 1-nitropyrene, 7-nitrobenz[a]anthracene, and 9-nitroanthracene were 100.0%, 96.1%, 45.3% and 86.3% [18].

Most of the quantitative biases and relative standard deviations were lower than 25% and 15% in POCIS analysis respectively (Table 3, p.48).

Regarding to C.L.A.M., the extraction efficiency, accuracy, and precision of benzophenone were not calculated because benzophenone was not well-retained by HLB disks (maximum ~30ng/disk), and benzophenone-D_{10} was not retained at all.

For both POCIS and C.L.A.M. extraction, 2-nitrofluorene and 5,12-naphthacenequinone had greater relative standard deviations due to the low recovery or lack of their corresponding isotope standards (Table 3, p.48).

2-nitrofluorene and 5, 12-naphthacenequinone utilized 1-nitropyrene-D_9 as their internal standards, but the structures, ring arrangements and properties were different between these three compounds.

3.2 Characteristics of uptake and sampling rates at POCIS

. The uptake pattern was evaluated through CF-t plot of every analyte. CF was concentration factor, that is the ratio of analyte concentration in sorbent over analyte concentration in water at the measure time. The larger the CF values, the faster the uptake of analytes through water to sorbent.

During the 14-day calibration period, methyl paraben, caffeine, and finasteride exhibited typical curvilinear accumulation (Figure 5(a), p.39; Table 4, p.50). Their sampling rates were calculated through fitting the CF-t plot curve to 1st-order kinetics (Equation 1, p.錯誤! 尚未定義書籤。). DEET, and acidic PCPs, such as ethyl paraben, naproxen, ketoprofen, ibuprofen maintained linear uptake ($R^2 > 0.990$) in 14 days (Figure 5(b), p.39; Table 4, p.50).. The linearity exhibited the uptake was still at the linear stage and allowed the sampling rates of these compounds to be calculated through Equation (2) (p.5). Propyl paraben and butyl paraben had delayed adsorption behavior followed by linear uptake, and their patterns of uptake were neither fitted to 1st-order kinetics or linearity (($R^2 < 0.990$) (Figure 5(c), p.40). The sampling rates for these two parabens were estimated through the CF values at day 5. Random and low ($CF \leq 3$) uptake patterns of benzophenone and oxybenzone were observed, which may result from photodegradation; this can be explained by their properties as UV filters. Regarding NPAHs and OPAHs, 2-nitrofluorene and 5,12-naphthacenequinone exhibited no accumulation on POCIS; 1-nitropyrene, 9-nitroanthrancen, 7-nitrobenz[a]anthracene

showed very low and random accumulation. This phenomenon may be explained by their high k_{ow} values (3.97-5.34).

Among the ten calculated sampling rates, only three compounds exhibited 1st-order kinetics and the time to reach equilibrium ($t_{1/2}$) was calculated. To apply POCIS to field sampling, the time of deployment should be shorter than the calculated $t_{1/2}$ time for presuming the sampler as an infinite sink to extract the analytes. The uptake of five compounds (DEET, ethyl paraben, naproxen, ketoprofen, ibuprofen) showed linear relationship in 14 days, and the calculated $t_{1/2}$ values of three compounds (caffeine, finasteride, and methyl paraben) with a curvilinear uptake pattern were 39, 61, 20 days (Table 5, p.51). For propyl paraben and butyl paraben, as the sampling rates were estimated from CF at day 5, the sampling period between 5 to 14 days would be approapriate. As a result, sampling period between 5 to 14 days would be suitable for sampling these analytes.

Comparisons between sampling rates in literature should be applied carefully because the calibration conditions were varied in each study, and analyte uptake may be significantly differed. Comparisons performed in this study were based on the use of pharmaceutical POCIS with the same configuration (200 mg sorbent, surface area 45.8 cm^2), similar temperature range (15-30℃), and under similar agitation (including both stirred and flow-through systems).

Sampling rates of ketoprofen, naproxen, and ibuprofen were reported by several studies and our values were close to reported values (Table 5, p.51). As for sampling rates for caffeine and acetaminophen, the reported values were much larger than our calculated values. The results may be influenced by the calculation methods and other different conditions of calibration system. As the calibration system varied a lot in different studies, the precise comparison of sampling rates was difficult. Our study

provided sampling rates of six new compounds, including methyl paraben, ethyl paraben, propyl paraben, butyl paraben, finasteride, and DEET. Although the $t_{1/2}$ values of these six compounds were not obtained in the 14-day calibration system, the sampling rates provided good reference for field sampling.

No prominent relationship between estimated sampling rates and log *K*ow for all studies analytes (Figure 6, p.40). For the three acidic analgesics, sampling rates increased with high log *K*ow. For parabens, the sampling rates seemed to decrease with longer alkyl side chain, except for methyl paraben.

3.3 PRC evaluation

The ideal PRCs show the analogue 1st order kinetic pattern as target analytes in release mode, and have the same half-life time compared with the analyte. Other choices may include specific decreasing pattern, such as linear decrease during laboratory calibration experiment.

Compared to native analytes, the isotoped chemicals show different behavior in the 14-day experiment (Figure 7, p.41), and the estimated elimination rates were different from sampling rates for most compounds. Closed sampling rates and elimination rates were shown by ketoprofen and ibuprofen. There were no prominent relationship between the elimination rates and log *K*ow values of the PRC candidates (Figure 8, p.41).

From the elimination pattern showed that the release of methyl paraben-$^{13}C_6$, naproxen-^{13}C-D_3, acetaminophen-D_4, and caffeine-$^{13}C_6$ reached the equilibrium (the lowest quanitity) at day 7. As PRCs were utilized to reflect the fluctuations of environmental conditions, the sampling period of these four PRCs were limited to

shorter than 7 days. For other four candidates, ketoprefen-D_3, butyl paraben-$^{13}C_6$, ibuprofen-$^{13}C_6$, and DEET-D_6, the patterns showed that release lasted at least for 14 days (Figure 7, p.41). As the uptake of the four corresponding native compounds remained linear in 14 days, the releasing mode did not follow the same patterns. Therefore, there was no proper isotope-labeled PRC in this POCIS system, and no corrections would be implemented for further TWA concentration calculation.

3.4 Applications to field sampling

The calculated sampling rates and $t_{1/2}$ suggested that the sampling duration as a 7-day period. POCIS, C.L.A.M., and grab sampling were applied at the Melti Pier using the same period for comparing the results. There were 20 real samples, including 6 POCIS samples applied for a seven-day period, 7 C.L.A.M samples applied 21-hour per day, and 7 river water samples grabbed each day.

3.4.1 Concentrations in grab water samples

The detected compounds were consistent every day. UV filters, analgesics (except for ketoprofen) methyl paraben, propyl paraben, DEET, and caffeine were detected in all grabbed water samples. Caffeine was the most abundant analyte in the river water, ranged from 3,650-5,130 ng/L. For parabens, only methyl paraben and propyl paraben were detected at low levels. NPAHs, OPAH, finasteride, ketoprofen, ethyl paraben, and butyl paraben were not detected in any grabbed sample (Table 6, p.53)

3.4.2 Concentrations in POCIS samples

As for POCIS samples, ketoprofen, naproxen, ibuprofen, propyl paraben, caffeine

and DEET were detected in all 6 samples (Table 6, p.53). Methyl paraben were detected in every POCIS sample, as the concentration is nearly the same as laboratory blank, the measured concentrations were deemed to be not detected. Acetaminophen, benzophenone, and oxybenzone were detected in every POCIS sample. As the accumulation patterns of these three compounds were low and random, the results were not semi-quantitative, but the detection would provide qualitative information for screening. For the 6 detected analytes, their concentrations in water were estimated through the sampling rates gained in the laboratory.

The sampling rates in field may be different from the calculated values in laboratory by several reasons. First, Conditions of river water were fluctuated and were different from the calibration system performed in the laboratory: temperature, pH, conductivity, salinity and TDS were higher (Table 7, p.54; Table 8, p.55). As temperature rises, the sampling rates may increase. Higher pH may lower the sampling rates of acidic compounds, and increase those of basic analytes. Although the sampling site is a tidal area, salinity was almost constant during the sampling period, but the salinity change was minor for sampling rates. Second, the water flow rates may not be the same with the rates applied in the calibration, and the flow rates were not stable through the whole sampling period. Faster flow rate increases the sampling rates. Third, fouling on the PES membrane was not found on the POCIS after deployment, the PES membrane surface was covered with mud, particles, and small debris. These materials may hinder the partition behavior between water and the membrane, causing lower sampling rates. Because there was no suitable PRC to adjust sampling rates with real environmental conditions in POCIS system, the calculated concentrations were considered to be semi-quantitative.

3.4.3 Concentrations in C.L.A.M. samples

The main problem encountered at C.L.A.M. deployment was the clogging of glass filter after deployment. Flow rates may drop to lower than 1 mL/min after the 21-hour deployment, and the problem was more serious after thundershowers. The observed flow after deployment was not continuous and produced bubbled sections, and the extracted disk may be dried and wet by water repeatedly during the deployment period. The results showed that the more hydrophobic compounds may not be retained on the disk as the extracted disk may be dried during the deployment. The internal standards mixture spiked directly spiked on the disk after drying by nitrogen gas was not well retained on the disk, except for the highly water-soluble compounds, acetaminophen and caffeine. Therefore, the C.L.A.M. results needed to be interrupted carefully, and the modified C.L.A.M. filter device needs further improvements.

As the problem stated above, the C.L.A.M. results were carefully evaluated through the performance of internal standards added before analysis. We used only one real sample result to calculate the concentration in water and compare to two other sampling methods (Table 6, p.53). The exact mass extracted in C.L.A.M. for every detected analytes was greater than grab and POCIS samples, but after normalization with passed volume of water, the concentration was lower than other two sampling methods.

3.4.4 Comparison between POCIS, C.L.A.M and grab samples

Regarding the detected number of compounds, POCIS was able to quantify six compounds, and identify three compounds qualitatively. There were also nine compounds detectable in grab samples, which ketoprofen was replaced by the detection

of methyl paraben in water samples. C.L.A.M. was able to detect 10 compounds, with the addition of ethyl paraben compared to grab samples. The concentrations of POCIS and grab samples were in similar trends, as the water volumes extracted were close. For C.L.A.M., the water volume extracted (~35L) was much greater than the other two sampling methods, but the concentrations detected were estimated twenty-fold lower. This result was in agreement with a previous study that detected concentrations of investigated compounds with C.L.A.M. were lower than those with other two methods [45].

Comparing the grabbed samples, POCIS, and C.L.A.M. samples, the representativeness of the samples, sampling time, sampling volume, and concentration type provided are different. Grabbed samples represent only the sampling instant in time, and POCIS and C.L.A.M. represent time-weighted samples. The sampling time of POCIS may last weeks to months; while C.L.A.M. represents about a < 24-hour period. The sampling volumes are quite different: volumes of grabbed samples are usually several liters; while that of C.L.A.M. may be high as to 100 liters. Volumes of POCIS samples are depended on both sampling rates and time, and are usually larger than that of grabbed samples. POCIS and C.L.A.M. provide TWA concentrations, while grabbed sample provide only the instant concentration of analyte in water.

Chapter 4 Conclusions

A calibration system was established to evaluate whether 13 PCPs and 5 NPAHs/OPAHs were suitable for POCIS sampling. Among the 13 PCPs, ten compounds were evaluated as appropriate for POCIS sampling and their sampling rates were calculated and utilized in field sampling to provide semi-quantitative results of river water concentration; other three chemicals were assess as only suitable to provide qualitative results of real samples. NPAHs/OPAHs were not suitable for POCIS sampling due to their no or low and random uptake. The TWA concentrations provided by POCIS were in agreement with grabbed water samples. Grabbed samples may only represent the concentrations at the sampling instant, but they were good indicators for detection of POCIS sampling. C.L.A.M. may detect more chemicals compared to POCIS and grabbed sampling, due to the larger water volume extracted. The detected C.L.A.M. concentrations were lower than other two sampling methods. This result may be attributed to the deposition of analytes on particles and low-retention of analytes on the disks.

Regarding C.L.A.M. sampler, the modifications of extraction disk replaced by a two-stage filter assembly was not applicable under the river environment with high content of particles. The glass filter may be clogged, and the flow rates of C.L.A.M. became unstable, causing the disk to be dried and wet repeatedly. The sorbent conditions were changed and made further analysis difficult. The filter device needs further modifications and more field tests to accommodate to the river environment.

References

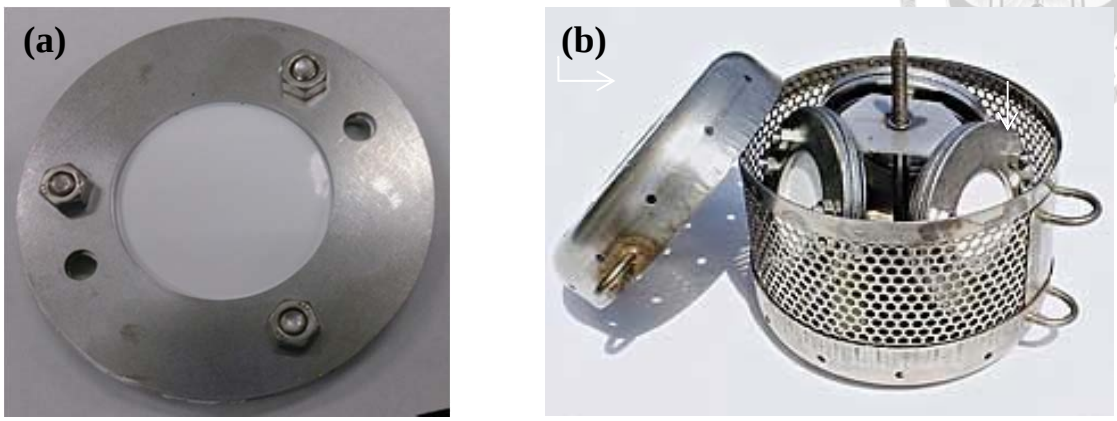
1. Daughton, C.G. and T.A. Ternes, Pharmaceuticals and personal care products in the environment: Agents of subtle change? *Environmental Health Perspectives,* 1999. **107**: p. 907-938.
2. Kasprzyk-Hordern, B., R.M. Dinsdale and A.J. Guwy, The removal of pharmaceuticals, personal care products, endocrine disruptors and illicit drugs during wastewater treatment and its impact on the quality of receiving waters. *Water Research,* 2009. **43**(2): p. 363-380.
3. Peng, X.Z., Y.J. Yu, C.M. Tang, J.H. Tan, Q.X. Huang, and Z.D. Wang, Occurrence of steroid estrogens, endocrine-disrupting phenols, and acid pharmaceutical residues in urban riverine water of the Pearl River Delta, South China. *Science of the Total Environment,* 2008. **397**(1-3): p. 158-166.
4. Vulliet, E., C. Cren-Olive and M.F. Grenier-Loustalot, Occurrence of pharmaceuticals and hormones in drinking water treated from surface waters. *Environmental Chemistry Letters,* 2011. **9**(1): p. 103-114.
5. Wang, C.A., H.L. Shi, C.D. Adams, S. Gamagedara, I. Stayton, T. Timmons, et al., Investigation of pharmaceuticals in Missouri natural and drinking water using high performance liquid chromatography-tandem mass spectrometry. *Water Research,* 2011. **45**(4): p. 1818-1828.
6. Petrovic, M., S. Gonzalez and D. Barcelo, Analysis and removal of emerging contaminants in wastewater and drinking water. *Trac-Trends in Analytical Chemistry,* 2003. **22**(10): p. 685-696.
7. Wu, C.X., X.L. Huang, J.D. Witter, A.L. Spongberg, K.X. Wang, D. Wang, et al., Occurrence of pharmaceuticals and personal care products and associated environmental risks in the central and lower Yangtze river, China. *Ecotoxicology and Environmental Safety,* 2014. **106**: p. 19-26.
8. Tang, T.C., Determination of Nitrated/Oxygenated Polycyclic Aromatic Hydrocarbons and Personal Care Products in River Water, Sediment, Fish Muscle and Liver with Ultra-Performance Liquid Chromatography/tandem Mass Spectrometry, 2015, Academic, Department
9. Soni, M.G., I.G. Carabin and G.A. Burdock, Safety assessment of esters of p-hydroxybenzoic acid (parabens). *Food and Chemical Toxicology,* 2005. **43**(7): p. 985-1015.
10. Haman, C., X. Dauchy, C. Rosin, and J.F. Munoz, Occurrence, fate and behavior of parabens in aquatic environments: A review. *Water Research,* 2015. **68**: p. 1-11.
11. Liao, C.Y., S. Lee, H.B. Moon, N. Yamashita, and K. Kannan, Parabens in Sediment and Sewage Sludge from the United States, Japan, and Korea: Spatial Distribution and Temporal Trends. *Environmental Science & Technology,* 2013. **47**(19): p. 10895-10902.
12. Diaz-Cruz, M.S., M. Llorca and D. Barcelo, Organic UV filters and their photodegradates, metabolites and disinfection by-products in the aquatic environment. *Trac-Trends in Analytical Chemistry,* 2008. **27**(10): p. 873-887.
13. Giokas, D.L., A. Salvador and A. Chisvert, UV filters: From sunscreens to human body and the environment. *Trac-Trends in Analytical Chemistry,* 2007. **26**(5): p. 360-374.
14. Kim, S., D. Jung, Y. Kho, and K. Choi, Effects of benzophenone-3 exposure on

endocrine disruption and reproduction of Japanese medaka (Oryzias latipes)-A two generation exposure study. *Aquatic Toxicology,* 2014. **155**: p. 244-252.
15. Poiger, T., H.R. Buser, M.E. Balmer, P.A. Bergqvist, and M.D. Muller, Occurrence of UV filter compounds from sunscreens in surface waters: regional mass balance in two Swiss lakes. *Chemosphere,* 2004. **55**(7): p. 951-963.
16. Swale, D.R., B.N. Sun, F. Tong, and J.R. Bloomquist, Neurotoxicity and Mode of Action of N,N-Diethyl-Meta-Toluamide (DEET). *Plos One,* 2014. **9**(8).
17. Quednow, K. and W. Puttmann, Temporal concentration changes of DEET, TCEP, terbutryn, and nonylphenols in freshwater streams of Hesse, Germany: possible influence of mandatory regulations and voluntary environmental agreements. *Environmental Science and Pollution Research,* 2009. **16**(6): p. 630-640.
18. Qiao, M., W.X. Qi, H.J. Liu, and J.H. Qu, Simultaneous determination of typical substituted and parent polycyclic aromatic hydrocarbons in water and solid matrix by gas chromatography-mass spectrometry. *Journal of Chromatography A,* 2013. **1291**: p. 129-136.
19. Qiao, M., W. Qi, H. Liu, and J. Qu, Oxygenated, nitrated, methyl and parent polycyclic aromatic hydrocarbons in rivers of Haihe River System, China: occurrence, possible formation, and source and fate in a water-shortage area. *Sci Total Environ,* 2014. **481**: p. 178-85.
20. Umbuzeiro, G.A., A. Franco, M.H. Martins, F. Kummrow, L. Carvalho, H.H. Schmeiser, et al., Mutagenicity and DNA adduct formation of PAH, nitro-PAH, and oxy-PAH fractions of atmospheric particulate matter from Sao Paulo, Brazil. *Mutation Research-Genetic Toxicology and Environmental Mutagenesis,* 2008. **652**(1): p. 72-80.
21. Durant, J.L., W.F. Busby, A.L. Lafleur, B.W. Penman, and C.L. Crespi, Human cell mutagenicity of oxygenated, nitrated and unsubstituted polycyclic aromatic hydrocarbons associated with urban aerosols. *Mutation Research-Genetic Toxicology,* 1996. **371**(3-4): p. 123-157.
22. Sklorz, M., J.J. Briede, J. Schnelle-Kreis, Y. Liu, J. Cyrys, T.M. de Kok, et al., Concentration of oxygenated polycyclic aromatic hydrocarbons and oxygen free radical formation from urban particulate matter. *Journal of Toxicology and Environmental Health-Part a-Current Issues,* 2007. **70**(21): p. 1866-1869.
23. Chung, S.W., H.Y. Chung, A. Toriba, T. Kameda, N. Tang, R. Kizu, et al., An environmental quinoid polycyclic aromatic hydrocarbon, acenaphthenequinone, modulates cyclooxygenase-2 expression through reactive oxygen species generation and nuclear factor kappa B activation in A549 cells. *Toxicological Sciences,* 2007. **95**(2): p. 348-355.
24. Kurihara, R., F. Shiraishi, N. Tanaka, and S. Hashimoto, Presence and estrogenicity of anthracene derivatives in coastal Japanese waters. *Environmental Toxicology and Chemistry,* 2005. **24**(8): p. 1984-1993.
25. Mekiki, D., N. Kalogerakis and E. Psillakis, Application of solid-phase microextraction for the analysis of nitropolycyclic aromatic hydrocarbons in water. *Chromatographia,* 2006. **63**(1-2): p. 85-89.
26. Alvarez, D.A., J.D. Petty, J.N. Huckins, T.L. Jones-Lepp, D.T. Getting, J.P. Goddard, et al., Development of a passive, in situ, integrative sampler for hydrophilic organic contaminants in aquatic environments. *Environmental Toxicology and Chemistry,* 2004. **23**(7): p. 1640-1648.
27. Alvarez, D.A., Guidelines for the Use of the Semipermeable Membrane Device

(SPMD) and the Polar Organic Chemical Integrative Sampler (POCIS) in Environmental Monitoring Studies, 2010, U.S.G. Survey ^Reston, Virginia
28. Morin, N., C. Miege, J. Randon, and M. Coquery, Chemical calibration, performance, validation and applications of the polar organic chemical integrative sampler (POCIS) in aquatic environments. *Trac-Trends in Analytical Chemistry,* 2012. **36**: p. 144-175.
29. Harman, C., I.J. Allan and E.L.M. Vermeirssen, Calibration and use of the polar organic chemical integrative sampler-a critical review. *Environmental Toxicology and Chemistry,* 2012. **31**(12): p. 2724-2738.
30. Zhang, Z.L., A. Hibberd and J.L. Zhou, Analysis of emerging contaminants in sewage effluent and river water: Comparison between spot and passive sampling. *Analytica Chimica Acta,* 2008. **607**(1): p. 37-44.
31. Togola, A. and H. Budzinski, Development of polar organic integrative samplers for analysis of pharmaceuticals in aquatic systems. *Analytical Chemistry,* 2007. **79**(17): p. 6734-6741.
32. Bayen, S., X.Z. Yi, E. Segovia, Z. Zhou, and B.C. Kelly, Analysis of selected antibiotics in surface freshwater and seawater using direct injection in liquid chromatography electrospray ionization tandem mass spectrometry. *Journal of Chromatography A,* 2014. **1338**: p. 38-43.
33. Di Carro, M., L. Bono and E. Magi, A simple recirculating flow system for the calibration of polar organic chemical integrative samplers (POCIS): Effect of flow rate on different water pollutants. *Talanta,* 2014. **120**: p. 30-33.
34. Kaserzon, S.L., E.L.M. Vermeirssen, D.W. Hawker, K. Kennedy, C. Bentley, J. Thompson, et al., Passive sampling of perfluorinated chemicals in water: Flow rate effects on chemical uptake. *Environmental Pollution,* 2013. **177**: p. 58-63.
35. Li, H.X., E.L.M. Vermeirssen, P.A. Helm, and C.D. Metcalfe, Controlled Field Evaluation of Water Flow Rate Effects on Sampling Polar Organic Compounds Using Polar Organic Chemical Integrative Samplers. *Environmental Toxicology and Chemistry,* 2010. **29**(11): p. 2461-2469.
36. Ly, E.B., Y. Levi and S. Karolak, Calibration and field evaluation of polar organic chemical integrative sampler (POCIS) for monitoring pharmaceuticals in hospital wastewater. *Environmental Pollution,* 2013. **174**: p. 100-105.
37. Li, H.X., P.A. Helm and C.D. Metcalfe, Sampling in the Great Lakes for Pharmaceuticals, Personal Care Products, and Endocrine-Disrupting Substances Using the Passive Polar Organic Chemical Integrative Sampler. *Environmental Toxicology and Chemistry,* 2010. **29**(4): p. 751-762.
38. Li, H.X., P.A. Helm, G. Paterson, and C.D. Metcalfe, The effects of dissolved organic matter and pH on sampling rates for polar organic chemical integrative samplers (POCIS). *Chemosphere,* 2011. **83**(3): p. 271-280.
39. Harman, C., K.E. Tollefsen, O. Boyum, K. Thomas, and M. Grung, Uptake rates of alkylphenols, PAHs and carbazoles in semipermeable membrane devices (SPMDs) and polar organic chemical integrative samplers (POCIS). *Chemosphere,* 2008. **72**(10): p. 1510-1516.
40. Macleod, S.L., E.L. Mcclure and C.S. Wong, Laboratory calibration and field deployment of the polar organic chemical integrative sampler for pharmaceuticals and personal care products in wastewater and surface water. *Environmental Toxicology and Chemistry,* 2007. **26**(12): p. 2517-2529.
41. Morin, N., J. Camilleri, C. Cren-Olive, M. Coquery, and C. Miege, Determination of uptake kinetics and sampling rates for 56 organic

micropollutants using "pharmaceutical" POCIS. *Talanta,* 2013. **109**: p. 61-73.
42. Harman, C., I.J. Allan and P.S. Bauerlein, The Challenge of Exposure Correction for Polar Passive Samplers-The PRC and the POCIS. *Environmental Science & Technology,* 2011. **45**(21): p. 9120-9121.
43. Zenobio, J.E., B.C. Sanchez, J.K. Leet, L.C. Archuleta, and M.S. Sepulveda, Presence and effects of pharmaceutical and personal care products on the Baca National Wildlife Refuge, Colorado. *Chemosphere,* 2015. **120**: p. 750-755.
44. *C.I.Agent® C.L.A.M. Product Sheet*. [cited 2013 August 16]; Available from: http://www.ciagent-stormwater.com/documents/watermonitoring/CLAM_ProductSheet.pdf.
45. Coes, A.L., N.V. Paretti, W.T. Foreman, J.L. Iverson, and D.A. Alvarez, Sampling trace organic compounds in water: A comparison of a continuous active sampler to continuous passive and discrete sampling methods. *Science of the Total Environment,* 2014. **473**: p. 731-741.
46. Bayen, S., E. Segovia, L.L. Loh, D.F. Burger, H.S. Eikaas, and B.C. Kelly, Application of Polar Organic Chemical Integrative Sampler (POCIS) to monitor emerging contaminants in tropical waters. *Science of the Total Environment,* 2014. **482**: p. 15-22.
47. Miege, C., H. Budzinski, R. Jacquet, C. Soulier, T. Pelte, and M. Coquery, Polar organic chemical integrative sampler (POCIS): application for monitoring organic micropollutants in wastewater effluent and surface water. *Journal of Environmental Monitoring,* 2012. **14**(2): p. 626-635.
48. Bartelt-Hunt, S.L., D.D. Snow, T. Damon-Powell, D.L. Brown, G. Prasai, M. Schwarz, et al., Quantitative Evaluation of Laboratory Uptake Rates for Pesticides, Pharmaceuticals, and Steroid Hormones Using Pocis. *Environmental Toxicology and Chemistry,* 2011. **30**(6): p. 1412-1420.
49. Shabeer, A., T.P.K. Banerjee, M. Jadhav, R. Girame, S. Utture, S. Hingmire, et al., Residue dissipation and processing factor for dimethomorph, famoxadone and cymoxanil during raisin preparation. *Food Chemistry,* 2015. **170**: p. 180-185.
50. Amdany, R., L. Chimuka and E. Cukrowska, Determination of naproxen, ibuprofen and triclosan in wastewater using the polar organic chemical integrative sampler (POCIS): A laboratory calibration and field application. *Water Sa,* 2014. **40**(3): p. 407-414.

Figures



(a) POCIS (b) the POCIS cage assembly

Figure 1 POCIS assembly

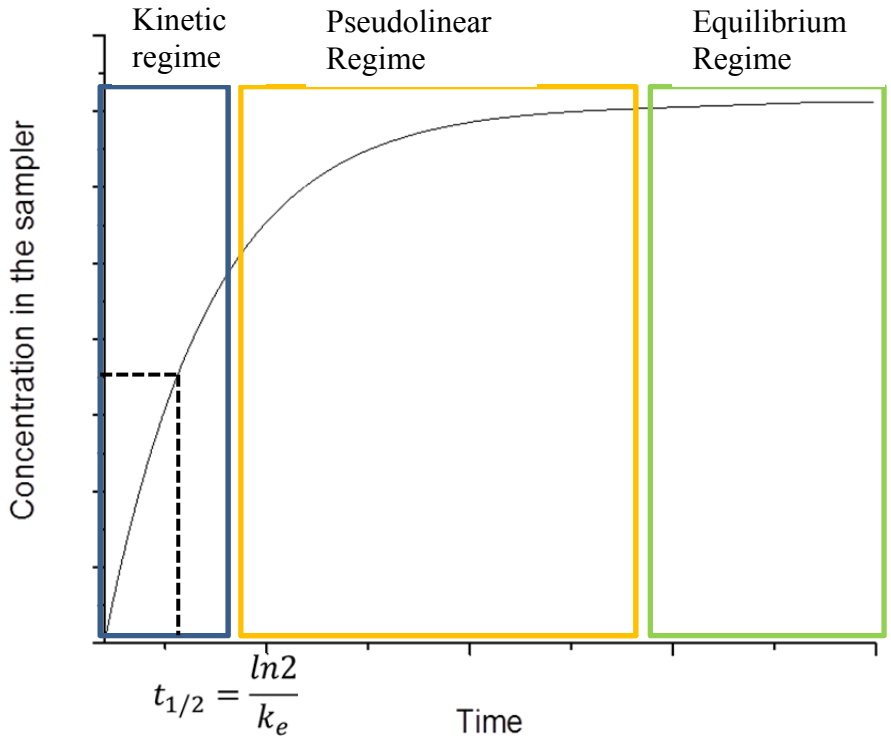
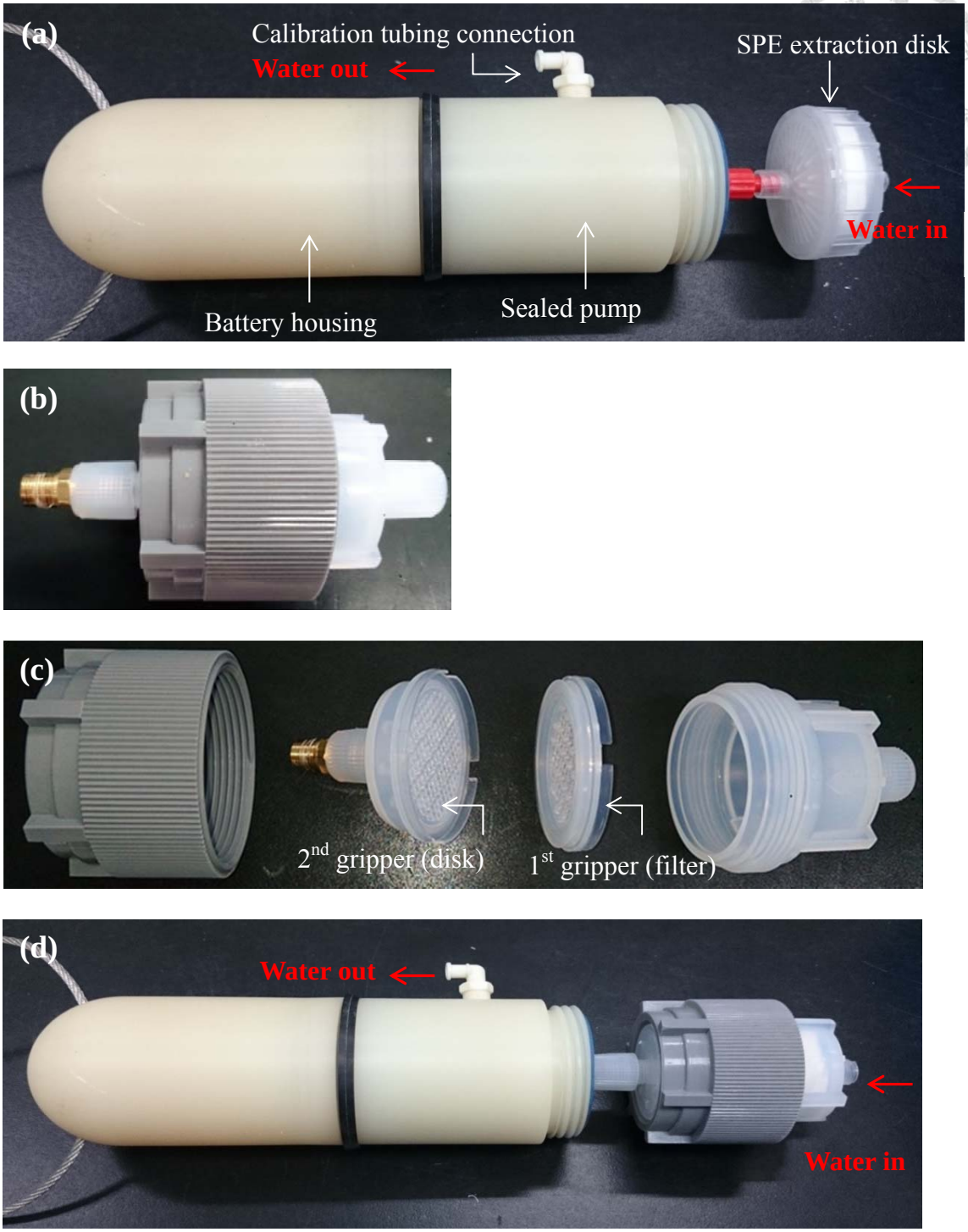


Figure 2 Uptake of POCIS



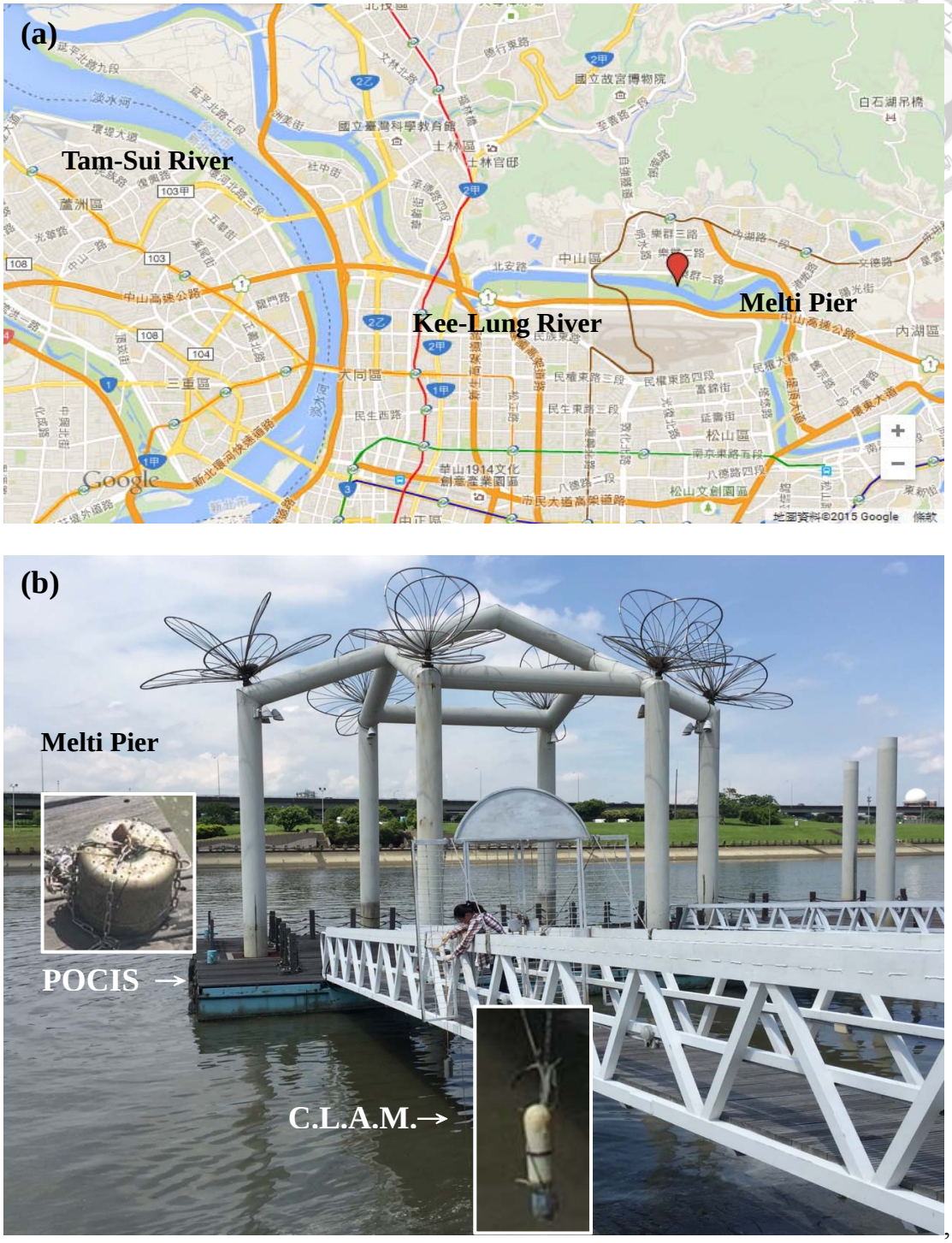
(a) Original C.L.A.M. from C.I.Agent Solutions®

(b) Two-stage filter assembly from Savillex (the whole assembly)

(c) Two-stage filter assembly from Savillex (separated parts)

(d) The modified C.L.A.M. with two-stage filter assembly

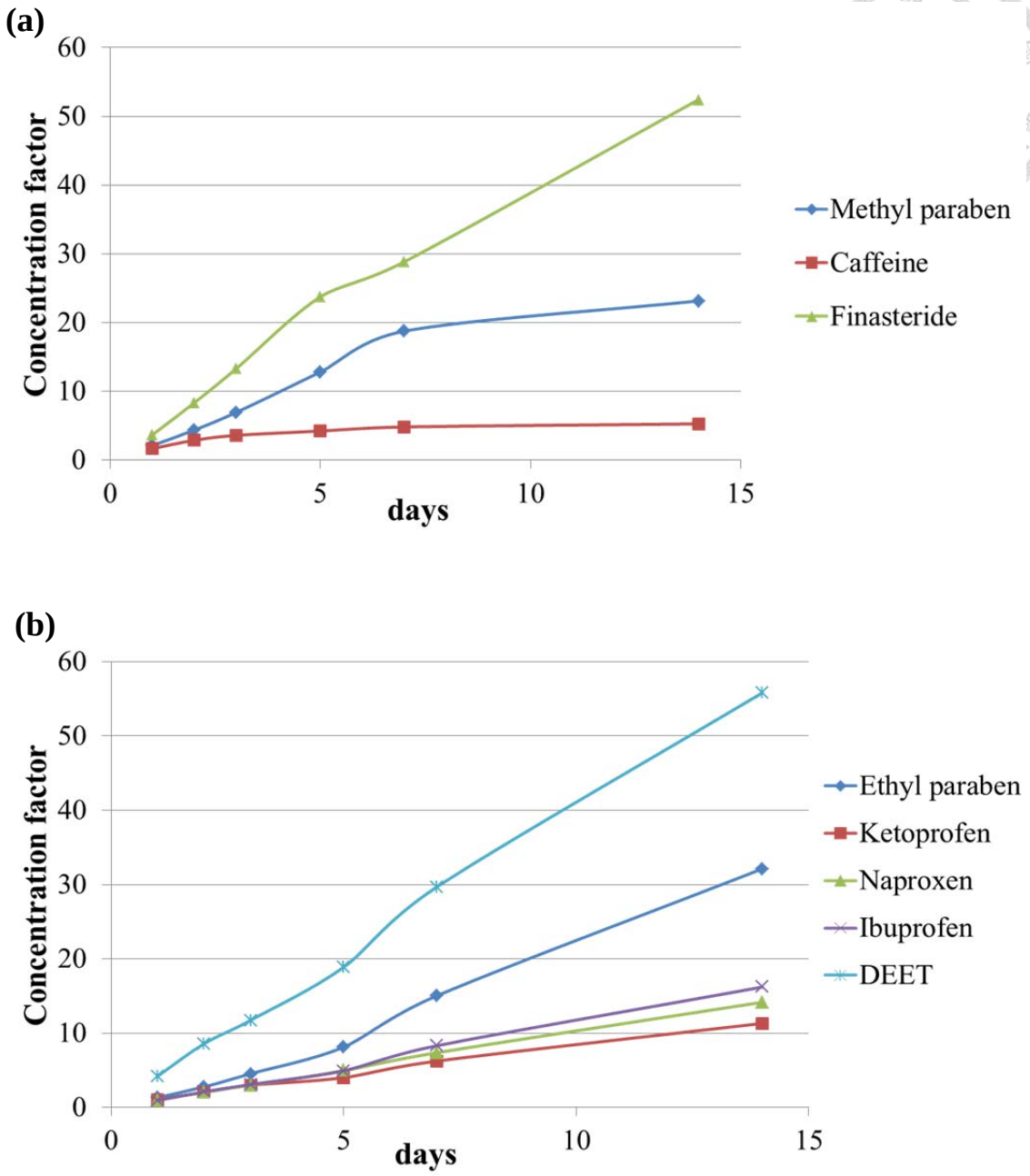
Figure 3 C.L.A.M. assembly and Savillex filter modification



(a) Sampling location (wide area)

(b) Sampling position (Melti Pier)

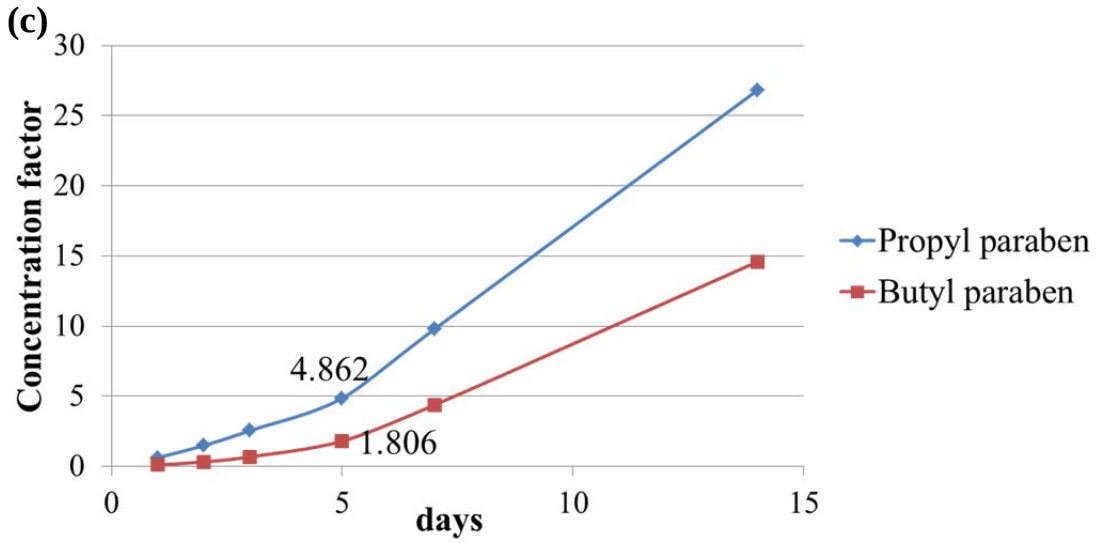
Figure 4 Sampling location



(a) Curvilinear uptake pattern during the 14-day calibration period (caffeine, methyl paraben, finasteride)

(b) Linear uptake pattern during 14-day calibration period (DEET, ethyl paraben, ibuprofen, ketoprofen, naproxen)

Figure 5 Uptake patterns and fitting curves of analytes for POCIS (n=3)



(c) Uptake pattern with a lag phase (propyl paraben, butyl paraben)

Figure 5 Uptake patterns and fitting curves of analytes for POCIS (n=3)

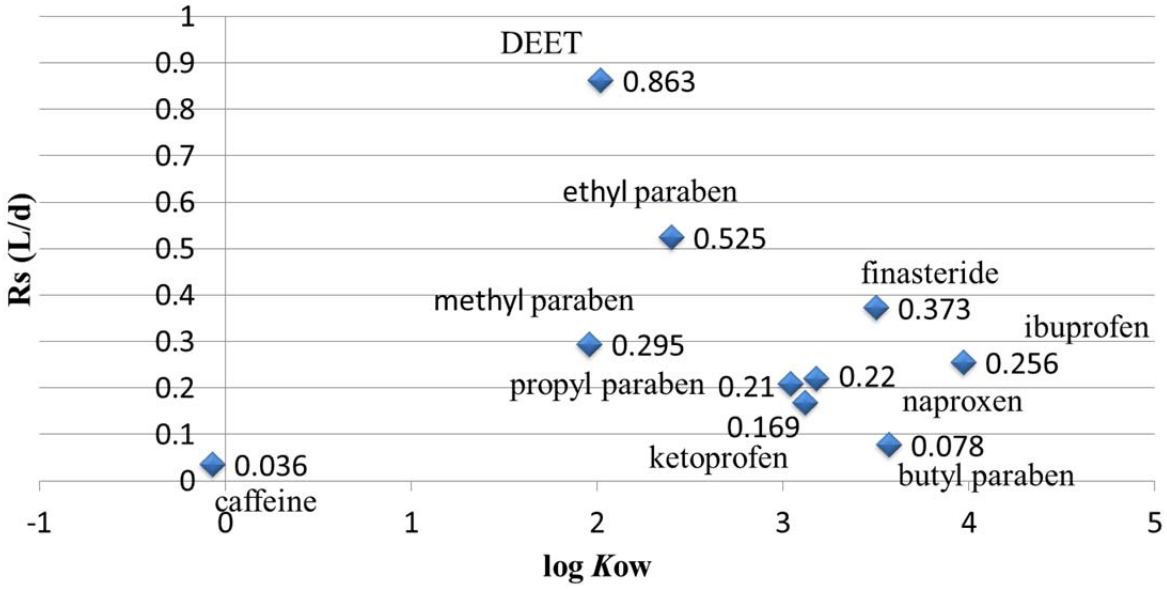


Figure 6 Sampling rates in POCIS versus log Kow for all studied chemicals

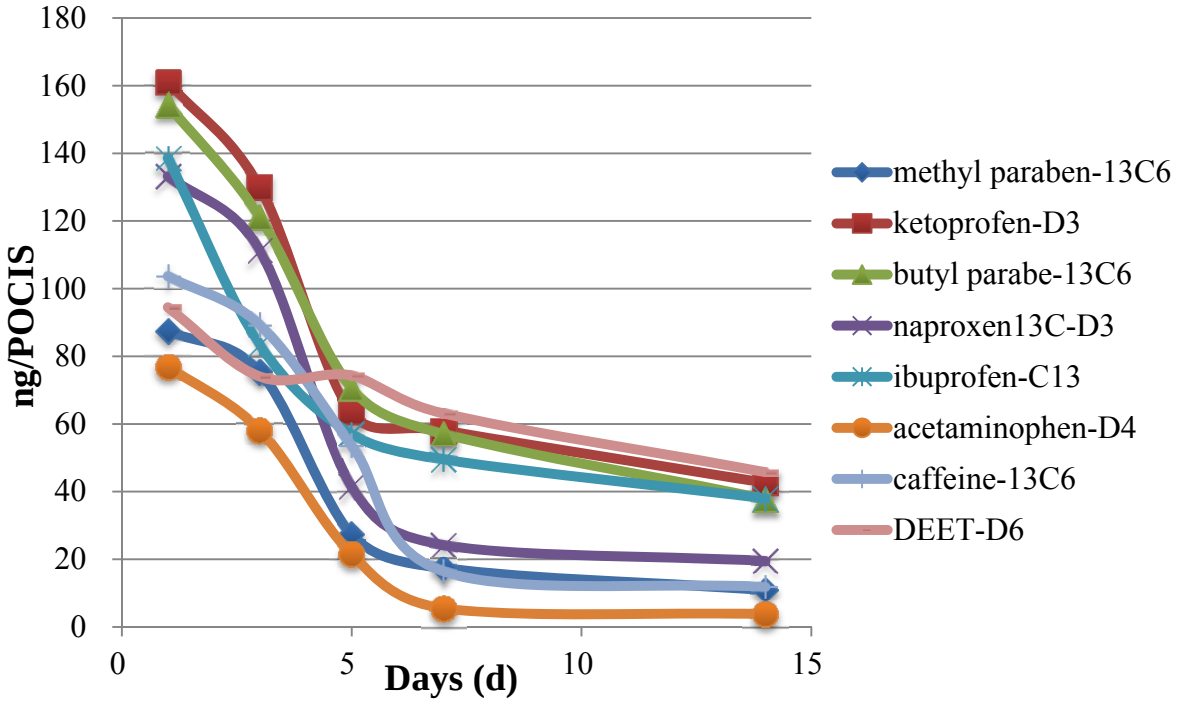


Figure 7 Elimination trends of isotope-labeled PRC candidates

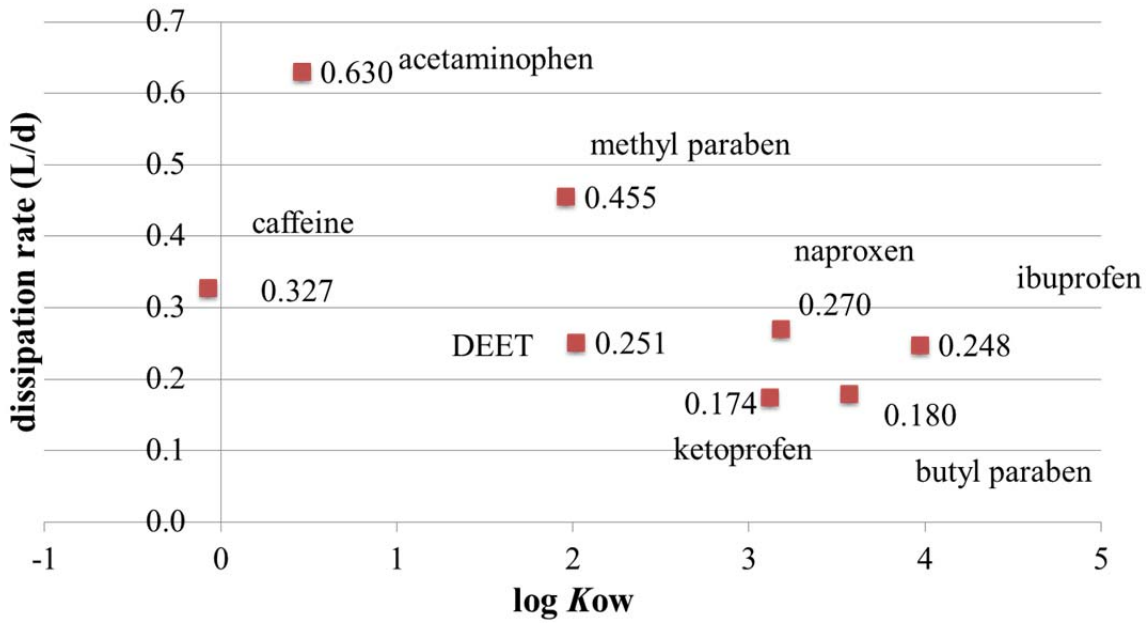
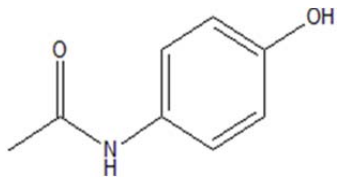
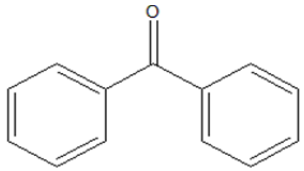
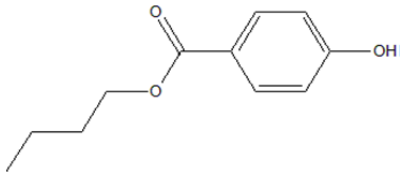
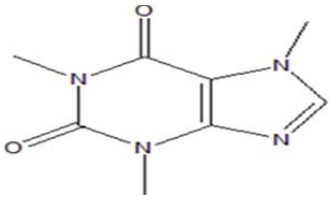
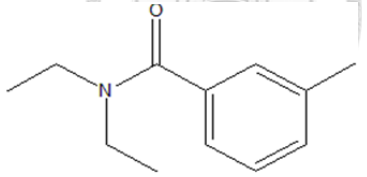
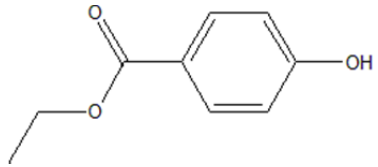
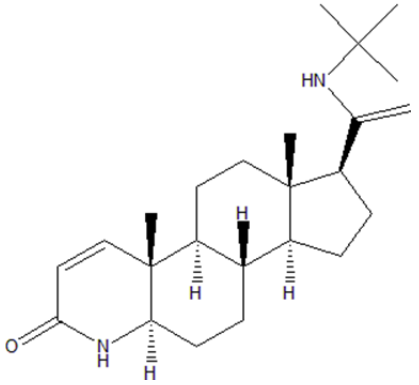
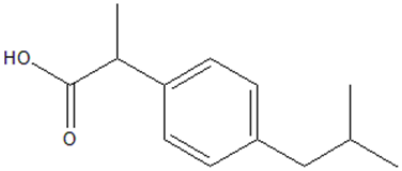


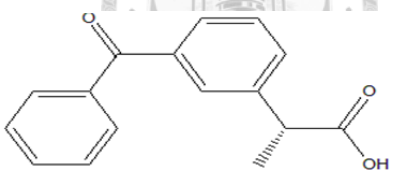
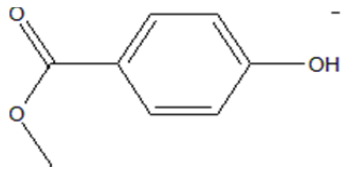
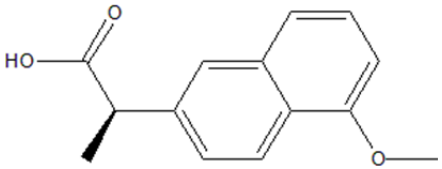
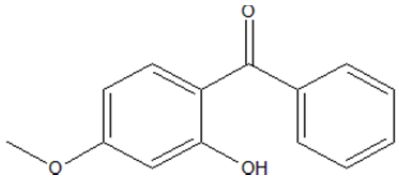
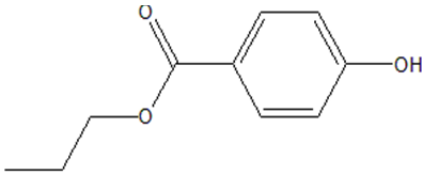
Figure 8 Dissipation rates versus log Kow for all isotope-labeled PRC candidates

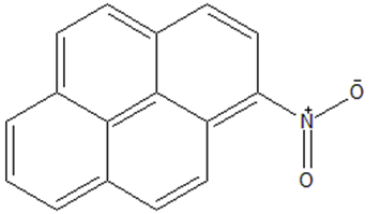
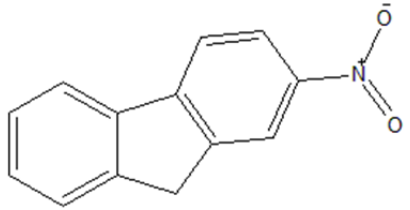
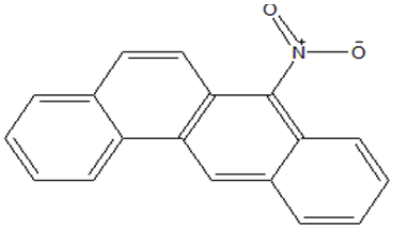
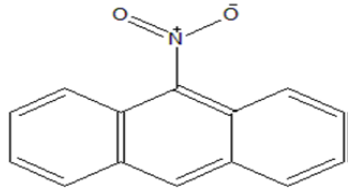
Tables

Table 1　Chemical structures and characteristics of analytes

Compounds	CAS number	molecular weight	log k_{ow}	pKa	water solubility (@25℃, mg/L)	structure
Personal care products						
Acetaminophen	103-90-2	151.16	0.46		soluble	
Benzophenone	119-61-9	182.22	3.18		40.1	
Butyl paraben	94-26-8	194.23	3.57		54.1	
Caffeine	58-08-2	194.19	-0.07		1.6×10^4	

Compounds	CAS number	molecular weight	log k_{ow}	pKa	water solubility (@25℃, mg/L)	structure
DEET	134-62-3	191.27	2.02	-	$> 10^3$	
Ethyl paraben	120-47-8	166.17	2.40		348	
Finasteride	98319-26-7	372.54	3.50		1.98	
Ibuprofen	15687-27-1	206.28	3.97		21	

Compounds	CAS number	molecular weight	log k_{ow}	pKa	water solubility (@25℃, mg/L)	structure
Ketoprofen	22071-15-4	254.28	3.12		21.3	
Methyl paraben	99-76-3	152.15	1.96		1.05×10^3	
Naproxen	22204-53-1	230.26	3.18		15.9	
Oxybenzone	131-57-7	228.24	3.79		128	
Propyl paraben	94-13-3	180.20	3.04		121	

Compounds	CAS number	molecular weight	log k_{ow}	pKa	water solubility (@25℃, mg/L)	structure
Nitrated polycyclic aromatic hydrocarbons						
1-nitropyrene	5522-43-0	247.25	5.06		1.7×10^{-2}	
2-nitrofluorene	607-57-8	211.22	3.97		0.216	
7-nitrobenz[a]anthracene	20268-51-3	273.29	5.34		-	
9-nitroanthracene	602-60-8	223.23	4.16		-	

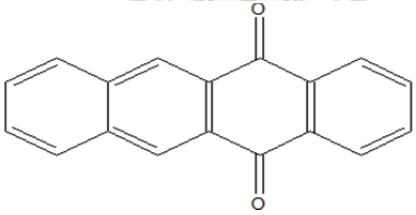
Compounds	CAS number	molecular weight	log k_{ow}	pKa	water solubility (@25℃, mg/L)	structure
Oxygenated polycyclic aromatic hydrocarbons						
5,12-naphthacenequinone	1090-13-7	258.27	4.52		-	

Table 2 Extraction efficiencies (%) of POCIS and C.L.A.M. (n = 3)

Compounds	POCIS (%RSD)	C.L.A.M. (%RSD)
Acetaminophen	80.7 (3.48 %)	96.6 (1.52 %)
Benzophenone	122 (6.24%)	-
Butyl paraben	96.0 (5.04 %)	81.7 (16.0 %)
Caffeine	93.8 (4.51 %)	99.4 (3.48 %)
DEET	102 (4.79 %)	98.6 (4.21 %)
Ethyl paraben	96.2 (4.09 %)	83.4 (4.05 %)
Finasteride	97.9 (2.74 %)	101 (4.93 %)
Ibuprofen	94.8 (3.45 %)	93.2 (9.98 %)
Ketoprofen	92.5 (10.2 %)	96.1 (17.6 %)
Methyl paraben	97.3 (4.92 %)	96.5 (1.60 %)
Naproxen	101 (5.53 %)	103 (1.80 %)
Oxybenzone	99.0 (4.48 %)	97.5 (4.25 %)
Propyl paraben	89.3 (2.77 %)	81.8 (2.78 %)
1-nitropyrene	94.3 (11.8 %)	92.7 (2.34 %)
2-nitrofluorene	110 (23.8 %)	101 (21.8 %)
7-nitrobenz[a]anthracene	104 (2.63 %)	86.5 (11.4 %)
9-nitroanthracene	101 (3.67 %)	76.6 (4.43 %)
5,12-naphthacenequinone	110 (8.45 %)	50.3 (15.0 %)

Table 3 Accuracy and precision of the spiked samples (n = 3)

Compounds	Spiked level	POCIS		C.L.A.M.	
		% RSD	Bias (%)	% RSD	Bias (%)
Acetaminophen	Low	5.22%	-2.85%	4.60%	6.77%
	Medium	2.95%	-22.1%	3.35%	-6.07%
	High	3.24%	-16.6%	8.10%	-1.67%
Benzophenone	Low	11.1%	6.57%	29.2%	-27.7%
	Medium	2.54%	4.04%	23.4%	-79.0%
	High	6.16%	5.87%	11.6%	-90.1%
Butyl paraben	Low	5.08%	12.7%	5.40%	-3.40%
	Medium	5.00%	-11.3%	5.45%	-11.5%
	High	4.78%	-2.47%	9.49%	-6.35%
Caffeine	Low	3.80%	10.6%	3.29%	-4.63%
	Medium	3.69%	-11.0%	1.48%	-3.00%
	High	3.18%	-6.85%	6.64%	0.43%
DEET	Low	3.16%	19.3%	1.14%	3.63%
	Medium	3.18%	-3.32%	3.64%	-0.68%
	High	3.75%	1.81%	8.27%	0.43%
Ethyl paraben	Low	5.76%	13.4%	6.26%	-9.53%
	Medium	5.25%	-13.9%	7.03%	-18.2%
	High	2.78%	-4.11%	7.55%	-16.4%
Finasteride	Low	2.31%	-3.70%	3.43%	-12.4%
	Medium	1.60%	-17.6%	1.82%	-9.11%
	High	3.47%	-4.97%	7.11%	-10.5%
Ibuprofen	Low	4.66%	11.2%	13.2%	-4.57%
	Medium	3.93%	-12.6%	14.6%	-12.2%
	High	2.01%	-7.95%	10.4%	-3.96%
Ketoprofen	Low	4.34%	8.77%	19.7%	26.6%
	Medium	3.20%	-13.7%	6.02%	-1.81%
	High	5.89%	-6.63%	3.59%	0.64%
Methyl paraben	Low	4.69%	-1.40%	20.2%	21.4%
	Medium	4.33%	-2.97%	14.4%	0.18%
	High	1.59%	-1.33%	4.05%	0.12%

Compounds	Spiked level	POCIS		C.L.A.M.	
		% RSD	Bias (%)	% RSD	Bias (%)
Naproxen	Low	3.07%	8.40%	31.4%	-24.9%
	Medium	2.94%	-10.4%	9.44%	-5.91%
	High	5.19%	-4.52%	3.52%	-8.59%
Oxybenzone	Low	2.91%	16.9%	3.32%	2.30%
	Medium	1.88%	-6.85%	3.82%	-2.94%
	High	3.47%	-0.96%	9.32%	-1.72%
Propyl paraben	Low	4.80%	4.80%	6.13%	-16.5%
	Medium	4.43%	-15.1%	6.32%	-20.7%
	High	2.01%	-9.02%	5.10%	-19.7%
1-nitropyrene	Low	4.73%	9.33%	9.60%	-8.30%
	Medium	14.3%	-9.87%	3.63%	-5.22%
	High	5.44%	-8.51%	11.6%	1.82%
2-nitrofluorene	Low	3.78%	24.6%	11.4%	51.6%
	Medium	14.0%	1.90%	6.17%	-13.9%
	High	2.67%	4.44%	19.4%	-7.12%
7-nitrobenz[a]anthracene	Low	4.78%	14.3%	31.0%	103%
	Medium	6.11%	1.38%	23.2%	79.0%
	High	4.93%	4.10%	10.8%	32.5%
9-nitroanthracene	Low	1.00%	18.1%	6.99%	3.20%
	Medium	3.18%	-2.62%	3.97%	0.81%
	High	6.17%	-1.06%	4.33%	-10.4%
5,12-naphthacenequinone	Low	6.05%	-13.3%	8.42%	-0.40%
	Medium	13.4%	-5.01%	6.17%	-37.0%
	High	7.90%	7.56%	19.5%	-33.5%

Spiked levels: low (100 ng), medium (300 ng), high (600 ng)

Table 4 Equation and R-squared values of fitting curves of calibration system

Compounds	Uptake pattern (14 days)	Rs (L/d)	Equation of fitting curve	R^2 of fitting curve
Caffeine	curvilinear	0.036 ± 0.022	$y = 5.19\ (1-e^{-0.382x})$	
Finasteride	curvilinear	0.373 ± 0.148	$y = 130\ (1-e^{-0.037x})$	-
Methyl paraben	curvilinear	0.295 ± 0.153	$y = 63.6\ (1-e^{-0.044x})$	-
DEET	linear	0.863 ± 0.010	y = 3.995x + 0.1701	0.9976
Ethyl paraben	linear	0.525 ± 0.021	y = 2.4281x -2.3034	0.9927
Ibuprofen	linear	0.256 ± 0.004	y = 1.1853x -0.3931	0.9965
Ketoprofen	linear	0.169 ± 0.017	y = 0.7828x + 0.4324	0.9959
Naproxen	linear	0.220 ± 0.034	y = 1.0171x -0.0090	0.9992

Table 5 Sampling rates and comparison with literature values

Compounds	log Kow	uptake pattern (14 days)	$t_{1/2}$ (days)	sampling rates Rs (L/d)	Rs (L/d) from literature
Caffeine	-0.07	curvilinear	31	0.036 ± 0.022	0.550 ± 0.108 [46] 0.168 ± 0.047 [47] 0.044 ± 0.005 [48] 0.127 ± 0.021 [49] NA [40]
Finasteride	3.50	curvilinear	69	0.373 ± 0.148	-
Methyl paraben	1.96	curvilinear	15	0.295 ± 0.153	-
DEET	2.02	linear	linear ≤ 14 days	0.863 ± 0.010	0.19
Ethyl paraben	2.40	linear	linear ≤ 14 days	0.525 ± 0.021	-
Ibuprofen	3.97	linear	linear ≤ 14 days	0.256 ± 0.004	0.182 ± 0.037 [50] 0.118 ± 0.006 [41] 0.279 ± 0.056 [47] 0.400 ±0.081 [48] 0.348 ±0.052 [37] NA [40]
Ketoprofen	3.12	linear	linear ≤ 14 days	0.169 ± 0.017	0.118 ± 0.007 [41] 0.206 ± 0.001 [36] 0.139 ± 0.042 [47] 0.135 ±0.035 [40]

Naproxen	3.18	linear	linear ≤ 14 days	0.220 ± 0.034	0.125 ± 0.048 [50] 0.324 ± 0.087 [46] 0.084 ± 0.011 [41] 0.144 ± 0.044 [47] 0.392 ± 0.024 [37] 0.116 ± 0.053 [40]
Propyl paraben	3.04	with a lag phase	-	0.210 ± 0.013	-
Butyl paraben	3.57	with a lag phase	-	0.078 ± 0.006	-
Benzophenone	3.18	low, random	-	-	-
Acetaminophen	0.46	low, random	-	-	0.048 ± 0.135 [46] 0.048 ± 0.011 [48] 0.145 ± 0.033 [37] NA [40]
Oxybenzone	3.79	low, random	-	-	-
1-nitropyrene	5.06	low, random	-		-
7-nitrobenz[a]anthracene	5.34	low, random	-		--
9-nitroanthracene	4.16	low, random	-		--
2-nitrofluorene	3.97	no accumulation	-		-
5,12-naphthacenequinone	4.52	no accumulation	-		--

Table 6 Concentrations of real samples

Compounds	n	POCIS (ng/L) mean ± SD	n	water (ng/L) mean ± SD	n	C.L.A.M (ng/L) mean ± SD
Acetaminophen	7	-	6	208 ± 61	1	0.81
Benzophenone	7	-	6	36.9 ± 24.1	1	13.3
Butyl paraben	7	ND	6	ND	1	ND
Caffeine	7	3510 ± 230	6	4370 ± 620	1	252
DEET	7	19.5 ± 2.0	6	120 ± 16	1	6.24
Ethyl paraben	7	ND	6	ND	1	0.059
Finasteride	7	ND	6	ND	1	ND
Ibuprofen	7	94.3 ± 8.4	6	236 ± 40	1	12.7
Ketoprofen	7	20.4 ± 3.4	6	ND	1	ND
Methyl paraben	7	ND	6	14.6 ± 6.98	1	0.597
Naproxen	7	35.3 ± 7.3	6	70.4 ±16.7	1	4.73
Oxybenzone	7	-	6	23.9 ± 9.13	1	1.89
Propyl paraben	7	23.5 ± 5.5	6	36.0 ± 16.1	1	1.18
1-nitropyrene	7	-	6	ND	1	ND
2-nitrofluorene	7	-	6	ND	1	ND
7-nitrobenz[a]anthracene	7	-	6	ND	1	ND
9-nitroanthracene	7	-	6	ND	1	ND
5,12-naphthacenequinone	7	-	6	ND	1	ND

ND: Not Detected -: No semi-quantitative result

Table 7　Conditions of grab water samples

date	temperature (℃)	pH	DO (mg/L)	Conductivity (μS/cm)	SAL (ppt)	TDS (mg/L)
6/18-1	36.3	7.27	5.40	296	0.16	162.3
6/18-2	36.3	7.19	5.55	291	0.16	162.8
6/18-3	36.3	7.19	4.74	295	0.16	162.5
6/19-1	34.7	7.28	5.61	322	0.16	161.0
6/19-2	34.7	7.29	5.25	324	0.16	161.5
6/19-3	34.7	7.30	5.15	322	0.16	160.8
6/20-1	32.7	7.28	5.16	333	0.16	166.4
6/20-2	32.7	7.30	5.47	332	0.16	153.3
6/20-3	32.7	7.30	5.62	331	0.16	165.6
6/21-1	32.7	7.49	3.61	336	0.16	168.2
6/21-2	32.7	7.49	3.68	336	0.16	168.4
6/21-3	32.7	7.49	3.72	337	0.16	168.7
6/22-1	33.0	7.32	2.46	293	0.14	146.1
6/22-2	33.0	7.27	2.47	293	0.14	145.9
6/22-3	33.0	7.27	2.47	291	0.14	145.7
6/23-1	35.4	7.28	2.64	306	0.15	152.6
6/23-2	35.4	7.24	2.54	304	0.15	151.9
6/23-3	35.4	7.24	2.55	305	0.15	152.0
6/24-1	32.5	7.28	2.58	301	0.14	150.3
6/24-2	32.5	7.28	2.72	302	0.14	150.4
6/24-3	32.5	7.25	2.60	301	0.14	149.7

SAL: Salinity; TDS: Total dissolved solids

Table 8 Environmental parameters during calibration system and field deployment of POCIS

	temperature (℃)	pH	DO (mg/L)	Conductivity (μS/cm)	SAL (ppt)	TDS (mg/L)
Calibration system	22.3-26.9	5.78-6.02	3.25-29.7	-	--	4.82-6.68
Field sampling	32.5-36.3	7.19-7.49	291-337	0.14-0.16	145.7-168.7	2.47-5.62

Appendices

Appendix 1　LC gradients

Column		Phenomenex Kinetex PFP
Oven temperature (℃)		40
Flow rate (mL/min)		0.50
Mobile phase / Gradient (min)	Acetonitrile	10 mM ammonium acetate/ 0.25% acetic acid pH = 4.08
Initial	10	90
0.5	10	90
4.5	90	10
5.5	90	10
6.0	10	90
8.0	10	90

Column		Waters CORTECS UPLC C18
Oven temperature (℃)		30
Flow rate (mL/min)		0.50
Mobile phase / Gradient (min)	Methanol	0.04% acetic acid pH = 3.50
Initial	15	85
0.5	15	85
3.0	100	0
3.5	100	0
4.0	15	85
6.0	15	85

Appendix 2　The LC conditions and the parameters of ionization sources

Compound group	Group I	Group II	Group III
Ion source	ESI +	ESI -	APPI +
Column	Kinetex PFP 50 × 2.1 mm, 2.6 μm	Waters Cortecs UPLC C18 30 × 2.1 mm, 1.6 μm	Kinetex PFP 50 × 2.1 mm, 2.6 μm
Mobile phase	MeOH / 10-mM ammonium acetate with 0.25% acetic acid	MeOH 0.04% acetic acid	MeOH/ 10-mM ammonium acetate with 0.25% acetic acid
Flow rate (mL/min)	0.50	0.50	0.50
Dopant	-	-	Anisole
Dopant flow rate (mL/min)	-	-	0.050
MS/MS parameters			
Capillary voltage (kV)	3.5	2.0	-
Repeller voltage (kV)	-	-	1.5
Extractor voltage (V)	3	3	3
Desolvation gas flow (L/hr)	900	900	400
Cone gas flow (L/hr)	150	150	50
Source temp (℃)	120	130	110
Desolvation temp (℃)	450	500	-
APPI probe temp (℃)	-	-	500

Appendix 3 The MS/MS parameters for native and stable isotope-labeled analytes

Compounds	Cone voltages (V)	Ion transitions (m/z) (Collision energy)	Corresponding IS	Cone voltages (V)	Ion transitions (m/z) (Collision energy)
Group I					
Acetaminophen	25	(+) 151.0 > 110.0 (14), 92.9 (22)	Acetaminophen $^{2}D_{4}$	25	(+) 155.0 > 114.0 (14)
Caffeine	38	(+) 194.9 > 137.9 (20), 110.0 (20)	Caffeine $^{13}C_{3}$	38	(+) 197.9 > 139.9 (20)
DEET	22	(+) 192.1 > 118.9 (18), 91.0 (30)	DEET $^{2}D_{6}$	22	(+) 198.1 > 118.9 (18)
Finasteride	55	(+) 373.1 > 304.9 (28), 57.0 (43)			
Oxybenzone	30	(+) 229.1 > 150.9 (18), 105.0 (30)	Oxybenzone $^{13}C_{6}$	30	(+) 192.9 > 109.8 (18)
Benzophenone	18	(+) 182.9 > 104.8 (18), 76.9 (31)			
1-nitropyrene	24	(+) 247.8 > 230.7 (12), 200.8 (25)	1-nitropyrene $^{2}D_{9}$	24	(+) 256.8 > 239.7 (12)
2-nitrofluorene	24	(+) 211.9 > 194.7 (14), 164.8 (24)			
5,12-naphthacenequinone	40	(+) 258.9 > 201.8 (30), 230.9 (16)			
Group II					
Methyl paraben	27	(-) 150.8 > 91.8 (18), 135.7 (15)	Methyl paraben $^{13}C_{6}$	27	(-) 156.8 > 97.8 (18)
Ethyl paraben	30	(-) 164.7 > 91.9 (23), 97.8 (15)	Butyl paraben $^{13}C_{6}$	27	(-) 198.9 > 97.8 (28)
Propyl paraben	29	(-) 178.8 > 91.9 (28), 135.9 (18)			
Butyl paraben	35	(-) 192.9 > 91.8 (28), 136.0 (20)			

Compounds	Cone voltages (V)	Ion transitions (m/z) (Collision energy)	Corresponding IS	Cone voltages (V)	Ion transitions (m/z) (Collision energy)
Ibuprofen	15	(-) 205.0 > 160.9 (8)	Ibuprofen $^{13}C_3$	15	(-) 208.0 > 163.9 (8)
Ketoprofen	14	(-) 253.1 > 209.0 (10)	Ketoprofen $^{13}C_3$	14	(-) 256.0 > 212.0 (7)
Naproxen	15	(-) 229.0 > 168.9 (30), 170.0 (30)	Naproxen-$^{13}C_1{}^2D_3$	15	(-) 233.0 > 172.9 (30)
Group III					
9-nitroanthracene	20	(+) 222.7 > 192.7 (18), 166.8 (26)	9-nitroanthracene$^2D_9$	15	(+) 231.7 > 201.7 (18)
7-nitrobenz[a]anthracene	20	(+) 272.9 > 214.9 (25), 227.9 (14)			