Accumulation of PAHs, Me-PAHs, PCBs and total Mercury in sediments and Marine Species in Coastal Areas of Dakar, Senegal: Contamination level and impact

Net, S.[1*], Henry, F.[2], Rabodonirina, S.[1], Diop, M.[2,3], Merhaby, D.[1], Mahfouz, C.[2,4], Amara, R.[2] and Ouddane, B.[1]

[1] Université Lille1, Equipe de Chimie Analytique et Marine, Laboratoire Géosystèmes - UMR CNRS 8217, 59655 Villeneuve d'Ascq Cedex, France

[2] Université du Littoral (ULCO), LOG UMR CNRS 8187, 62930 Wimereux, France

[3] Laboratoire de Toxicologie et hydrologie, Université Cheikh Anta Diop, Dakar, Senegal

[4] CNRS, Centre de Recherches Marines, Batroun, Libya

ABSTRACT: Persistent Organic Pollutants (POPs) have widely aroused public concern due to their ubiquity, environmental persistence, long-range transportability, bioaccumulation capacities and potentially adverse effects on living organisms. Dakar is located in the industrial zone of Senegal (80% of industrial activities) and inhabits 25% of Senegalese population leading to an ideal sink of these persistent organic pollutants. In the present paper, Polychlorinated Biphenyls (PCBs) and polycyclic Aromatic Hydrocarbons (PAHs) were analyzed in sediments and marine organisms. The contamination level of PAHs, Me-PAHs and PCBs in surface sediment and aquatic organisms (one macroalgae, two invertebrate species, four fish species and macroalgae) were determined. The concentration levels in the sediment were detected from 2 to 636 µg/kg dw for Σ_{16}PAHs, from 3 to 31 µg/kg dw for Σ_{18}Me-PAHs and from 4 to 333 µg/kg dw for Σ_{28}PCBs for the selected stations in Dakar costal area. PAHs concentrations determined in edible tissues were lower than sediment samples. Tilapia species present the highest mean levels of PAHs and Me-PAHs at 92 ± 54 and 183 ± 39 µg/kg dw respectively. For PCBs, the highest level was determined in *Perna perna* species (up to 1228 µg/kg dw) and the lowest level was found in *Penaeus kerathurus* species. At the base of the food chain, *Ulvalactula* species displayed low PCB concentrations detected at 7 ± 6 µg/kg dw. The total mercury concentration was also reported in this paper in order to complete the background of pollution degree and to study the potential processes of biomagnification and/or bioaccumulation of contaminants in marine food chain. Mercury concentration were detected in the sediment ranging from 5 to 95 µg/kg dw. For marine species, considering all organisms, the mean concentration of mercury varies between 5 and 442 µg/kg dw. Pyrogenic process was the predominant source of PAHs contamination in our sampling sites. Based on Sediments Quality Guidelines (ERM–ERL/TEL–PEL approaches) rare biological adverse effects of total mercury, PCBs and PAHs on aquatic ecosystems were expected in Dakar coastal area. Finally, in the context of human health, the edible marine species qualities obtained from three stations of Dakar coastal areas have been also evaluated. Based on the European Union legislation, the selected species present good quality for human consumption based on PCBs, PAHs and mercury.

Key words: PAHs, PCBs, mercury, sediment and marine organisms, Dakar

INTRODUCTION

The environmental impact of organic pollutants and metallic trace elements in coastal environment is serious. In order to better understand the potential impact of these pollutants on the ecosystem functioning as well as human health, an increasing attention has been implemented not only on the contamination level of contaminants in water or sediment but also on their accumulation level in aquatic organism (Boon, 1985; Naes *et al.,* 1995; Neff 2002; Borgå *et al.,* 2004; Francioni, 2005; Bastami *et al.,* 2013). Senegalese coast is one of the most productive areas in the world due to the presence of coastal upwelling of deep waters rich in nutrients (Romeo *et al.,* 1999). Even if industrialization did not reach a very high level compared to the developed countries, pollution of coastal aquatic habitats seems to be an inevitable problem mainly in the peninsula of Dakar which

*Corresponding author E-mail: sopheak.net@univ-lille1.fr

concentrates more than 80 % of the industries and inhabits more than 25 % of the population of the country (OIS 2010). Indeed, most of domestic and industrial waste waters are discharged directly into the sea. These permanent anthropogenic discharges can contribute to environmental and ecological degradation of Dakar coastal zones. Consequently, particular attention must be paid to the chemical quality of edible marine organisms consumed by local human populations. Fish and invertebrates consumption remain the major source of protein for the residents in Dakar. However, to date, only few studies were carried out on the trace metals and organic contamination levels in mollusk and fish species sampled from this aquatic ecosystem (Simoudou *et al.,* 2006; Ndiaye *et al.,* 2012).

Among the large variety of organic pollutants, polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) are two classes of pollutants intensively monitored and regulated due to their toxicity, persistence and wide diffusion in the environment. PCBs were widely used in industries until their hazards to the environment and human health became evident. These contaminants have been reported to cause cancer in animals and humans (Bertrand *et al.,* 2010; Kramer *et al.,* 2012; Bräuner *et al.,* 2012; EPA, 2013). PCBs have primarily been used as dielectric fluids of transformers and capacitors; other applications included their use in paints, inks and pesticides; they are extremely stable compounds under environmental conditions (Bennett, 1983; WHO, 1993). Due to their toxicity, persistency and bioaccumulation capacities, these substances have been classified as persistent organic pollutants. Their production was banned by the United States Congress in 1979 and by the Stockholm Convention on Persistent Organic Pollutants in 2001 (Porta and Zumeta, 2002). However, PCBs are still present in water and sediment (Turrio-Baldassarri *et al.,* 2005; Dumoulin *et al.,* 2013; Net *et al.,* 2014a) and continue to affect aquatic organisms from the base of food chain (plankton, algae) to predator organisms (fish, birds, marine mammals) and consequently human health through the diet (Sun *et al.,* 2002). Another class of organic contaminants is represented by the aromatic hydrocarbons group including PAHs and Me-PAHs. These contaminants are widely disseminated in the environment. High level of hydrocarbons represents a serious threat to the ecosystem functioning and human health via food chain and water resources. Their sources can be both natural and anthropogenic (Yunker *et al.,* 2002; Wang *et al.,* 2007; Mostert *et al.,* 2010; Net *et al.,* 2014a,b). Hydrocarbons are highly lipophilic compounds, ubiquitous in coastal, estuarine and river water column, as well as sediments in which they tend to accumulate

(Chiou *et al.,* 1998; Ko and Baker, 1995; Manodori *et al.,* 2006; Cailleaud *et al.,* 2007; Gaspare *et al.,* 2009; Yunker *et al.,* 2012; Net *et al.,* 2014b). Recent studies have reported that marine organisms are prone to bioaccumulate these substances, particularly in lipid-rich tissues (Neff, 2002; Francioni *et al.,* 2005; Dugan *et al.,* 2005). Due to their toxic, carcinogenic and mutagenic effects (Straif *et al.,* 2005; IARC, 2010; US Department of Health and Human Services, 2011), sixteen PAHs have been classified as priority substances by the United States Environmental Protection Agency (US EPA, 2002). Among toxic metals, mercury is an element of special concern because it is known to particularly biomagnify as it moves up the aquatic food chain (Carrasco *et al.,* 2011) and thus to bioaccumulate in higher tropic level consumers (Cossa *et al.,* 1990).

The aim of the present study was to assess the impact of anthropogenic discharges on the ecological ecosystem of Dakar coastal zones. Predominant organic compound (PCBs, PAHs and Me-PAHs) were quantified in one macro algae species (*Ulvalactuca*), two invertebrate species (mussel: *Perna perna* and shrimp: *Penaeus kerathurus*) and four fish species (grey mullets: *Mugilcephalus*, tilapia *Sarotherodon melanotheron*, flatfish *Soleasenegalensis* and round sardinella *Sardinella aurita*). These two invertebrate and four fish species have been selected because they represent a great economic and ecological importance in these coastal zones. In addition, a toxic metal such as mercury has been also considered in this paper in order to complete the background of contamination level and to study their potential biomagnification and/ or bioaccumulation processes in marine food chain.

MATERIALS & METHODS

Dakar is located in the west of Senegal, in the industrial zone (80% of industrial activities) and inhabits 25% of the Senegalese population which is an ideal sink of these persistent organic pollutants. The sampling campaign was conducted in the south of Dakar on February 2013 during dry season in three sampling sites along the South coast of the peninsula of Dakar, noted A, B and C on the Fig. 1. Classical parameter such as pH, temperature, salinity and dissolved oxygen have been also measured simultaneously. The values of pH, temperature, salinity and dissolved oxygen were respectively at 7.43, 21.8°C, 36.9 PSU and 7.42 mg/L for Soumbedioune station, 7.61, 23.4°C, 40.3 PSU and 6.19 mg/L for Yarakh station and 7.64, 23.1°C, 35.8 PSU and 7.02 mg/L for Rufisque station. Surface sediment and marine species have been selected in order to understand the potential impact of micro pollutants from urban and industrial discharges on ecosystem functioning and on human

health. Seven marine species were considered in this work starting from macroalgae, bivalve, crustacean and four species of fish. The choice was based on the frequent consumption of these species by the population inhabiting the Senegalese coast. The main characteristics of marine organisms selected in this study are briefly presented as following: (i) Macroalgae: *Ulvalactuca* provide qualitative information about the contamination level and environmental quality in an ecosystem due to their lifestyle sedentary and abundance in coastal seawater (Rainbow and Phillips, 1993). (ii) Mussel: *Perna perna* is the only mussel of this genus in the Western coast of Africa (Sidoumou *et al.,* 2006). Mussels are sedentary filtering organisms, which have been widely used as environmental sentinel for the contamination. (iii) Crustacean: *Penaeus kerathurus* is commercially one of the most important shrimp species in fishery in Senegal. It is also a target species for fishermen using trammel nets in Dakar bays. (iv) Fish species: *Sardinella aurita* is a small pelagic fish feeding on plankton. These fat fish are the more often consumed species in Senegal. In this study, *Sardinella aurita* presents from 228 to 337 cm length with mean average of 309±37 cm and from 110 to 396 g with mean average of 304±91 g wet weight. *Flathead mullet Mugilcephalus* is cosmopolitan and occupies a wide variety of marine, estuarine and freshwater environment in tropical, subtropical and temperate coastal waters. This benthopelagic species is omnivorous which diet consists mainly of zooplankton, benthic organisms and detritus for larger juveniles and adult stages. In this word, *Mugilcephalus* collected were from 318 to 362 cm length with mean average of 343±19 cm and from 282 to 426 g wet weight with mean average of 334±57 g wet weight. *Sarotheroron melanotheron:* As Mugilcephalus, Sarotheroron

melanotheron are tolerant to a broad range of environmental conditions and natural populations were found in many different habitats from freshwater to hypersaline waters (Panfili *et al.,* 2003). This species has an omnivorous diet and can change its diet in function of the environment. Tilapia is a fast-growing fish which has been an essential source of protein food. Therefore, this fish species show an increasing demand in many developed countries. *Sarotheroron melanotheron* collected were from 141 to 253 cm length with mean average of 196±58 cm and from 85 to 342 g with mean average of 206 ± 129 g wet weight. *Soleasenegalensis* is one of the most abundant and representative species of the Atlantic coasts. This flat benthic fish with a practically sedentary life lives in sandy or muddy bottoms in coastal areas and feeds on benthic invertebrates such as larvae of polychaets, bivalve mollusks and small crustaceans. This species is well adapted to warm climates and have been used in field and laboratory toxicity assays because of its sensitive character to pollutants (Costa *et al.,* 2009). *Soleasenegalensis* collected were from 269 to 371 cm length with mean average of 311±40 cm and from 178 to 445 g with mean average of 279 ± 97 g wet weightThese three sampling sites were chosen due to their locations near the waste discharge channels into the sea. Zone A (Soumbedioune) is dominated by discharges of domestic wastewater, hospital discharges and road traffics. Zones B (Yarakh) and C (Rufisque) are located in the Hann Bay: Yarakh is surrounded with industrial activities with the predominance of food industries while Rufisque is close to the Refining African society and cement factory. For each station, five superficial sediment samples were collected at low tide. Similarly, macroalgae and mussels were handpicked from substratum of intertidal zone. After
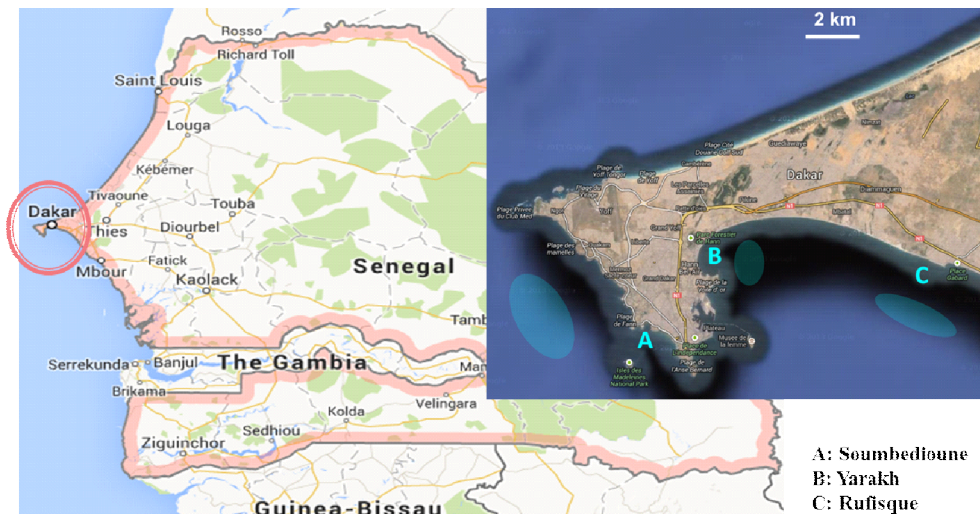

A: Soumbedioune
B: Yarakh
C: Rufisque

Fig. 1. Location of sampling sites from Dakar coastal zone, in Senegal

collection, samples were transported to the laboratory in icebox and biota samples were rinsed with pure water. The whole soft body of mussels was collected for chemical analysis. Samples of fish and shrimp were caught by fishermen's nets and the fishing zones were indicated approximately in the Fig. 1. Species were purchased from the local fishermen in the same day of capture and brought to laboratory on ice immediately. Before collecting muscle tissues, fish length and weight were measured. All samples were dried in an oven at 40°C to constant weight and were stored in individual aluminum foils at -20°C until further treatments and analysis. Dried sediment and biota samples were ground mechanically with an agate mortar and manually with a ceramic mortar and pestle, respectively. Mixed standard solutions of PAHs and Me-PAHs were purchased from Restek Corp (Bellefonte, PA, USA). PCBs standard solution was obtained from Accustandard, Inc. (New Haven, CT, USA). Tetrachloronaphtalene (TCN), 2,3,3',5,6-tetrachlorobiphenyl (PCB112) and octachloro-naphtalene (OCN), used as internal standard for PCBs quantification, were purchased from Dr Ehrenstorfer (Augsburg, Germany). Deuterated internal standards for PAHs and Me-PAHs quantification were acenaphthene-d10 (A-d10), naphtalene-d8 (N-d8), perylene-d12 (Per-d12), phenanthrene-d10 (Phe-d10) and pyrene-d10 (Pyr-d10) and they were provided by LGC-Promochem (Middlesex, UK). HPLC-grade solvents (hexane, dichloromethane, methanol and acetone) were purchased from Dislab (France). Ultrapure water (Milli-Q) was produced by a Millipore apparatus with 18.2 MΩ/cm resistivity. Merck silica gel 60 (70-230 mesh ASTM) activated at 450 °C was stored at 120°C for 12h prior to use. Glassware was systematically washed with detergent (Decon, East Sussex, UK), rinsed with ultrapure water and acetone and finally dried at 120 °C prior to use.

In this work, 16 PAHs, 18 Me-PAHs and 28 PCBs including 12 dioxin-like PCBs (dl-PCBs) and 7 PCB indicators (PCBi) were analyzed as follow:

PAHs (16 PAHs) : naphthalene (N), acenaphtylene (Acy), acenaphtene (Acn), fluorene (F), anthracene (An), fluoranthene (Fl), benzo*[a]*anthracene (BaA), chrysene (Chr), benzo*[a]*pyrene (BaP), phenanthrene (Pn), benzo*[b]*fluoranthene (BbF), benzo*[k]*-fluoranthene (BkF), benzo*[ghi]*perylene (Bghi), dibenzo*[a,h]*anthracene (DhA), indeno*[1,2,3-cd]*pyrene (IP), pyrene (Py).

Me-PAHs (18 Me-PAHs): 1-methylnaphthalene (1M-Na), 2-methylnaphthalene (2M-Na), 1,2-dimethylnaphthalene (1,2DM-Na), 1,6-dimethylnaphthalene (1,6DM-Na), 2,6-dimethylnaphthalene (2,6DM-Na), 1-methylphenanthrene (1M-Pn), 2-methylphenanthrene (2M-Pn), 3-

methylphenanthrene (3M-Pn), 9-methylphenanthrene (9M-Pn), 2-methylanthracene (2M-An), 1,7-dimethylphenanthrene (1,7DM-Pn), retene, 1-methylfluoranthene (1M-Fl), 3-methylfluoranthene (3M-Fl), 1-methylpyrene (1M-Py), 4-methylpyrene (4M-Py), 3-methylchrysene (3M-Ch), 6-methylchrysene (6M-Ch).

PCBs No. (28 PCBs): 8, 18, 28, 44, 52, 66, 77, 81, 101, 105, 114, 118, 123, 126, 128, 138, 153, 156, 157, 167, 169, 170, 180, 187, 189, 195, 206 and 209.

Mercury: Total mercury.

Sieved powder samples were spiked with deuterated internal standards A-d10, N-d10, Per-d12, Phe-d10 and Pyr-d10 for PAHs and Me-PAHs analyses and with TCN, PCB112 and OCN for PCBs analysis. After a delay of equilibration, samples were then extracted using an accelerated solvent extraction (ASE 200, Dionex Corp., USA). The extraction conditions were: heat 5 min, temperature 100°C, static solvent extraction time 2 min with 5 static cycles, pressure 138 bars, purge 3 min and 35 % flush according to the method developed by Tronczynski et al. (2005). High purity nitrogen was employed as the purge gas.

Molecular sulfur was removed from the extract by addition of activated metallic copper (Blumer, 1997) to the extracts. The extracts were concentrated, solvent-exchanged to hexane, and then purified and fractioned by liquid chromatography on a silica column to eliminate organic interferences (Jeanneau, 2007). PCBs were recovered by elution with hexane (Fraction 1), and aromatic hydrocarbons (PAHs and Me-PAHs) were recovered with hexane/dichloromethane mixtures (3/1 *v/v*) followed by hexane/dichloromethane mixture (1/1 *v/v*) (Fraction 2). Each fraction was concentrated using a rotary evaporator followed by a slight stream of nitrogen to a final volume of 200μL.

Targeted compounds were analyzed using a Varian 3900 gas chromatograph (GC) equipped with a deactivated fused-silica guard column (5 m, 0.25 mm i.d.) and a fused-silica low polarity si-arylene ZB-XLB capillary column (60 m length, 0.25 mm i.d., 0.25 μm film thickness, Phenomenex) and coupled with a Varian Ion Trap Saturn 2000 Mass Spectrometer (MS). The carrier gas was helium held at a constant flow rate of 1 mL/min. Each group of organic compounds was analyzed separately. Temperature of the GC oven was programmed as follow: from 70 °C (1 min) to 170 °C at 10 °C/min, then to 230 °C at 4 °C/min, and then to 300 °C at 3 °C/min (13 min) for HAPs and Me-HAPs and from 80 °C (1 min) to 170 °C at 10 °C/min, then to 230 °C at 4 °C/min, and then to 300 °C at 3 °C/min (19 min) for PCBs. Samples were injected in the splitless mode at 280 °C and the injector was purged with helium after 1

min. The transfer line and the ion trap were held at 280 °C and 220 °C, respectively. Identification of each compound was done on the basis of the retention time and the mass spectrum from chromatograph of standard solutions acquired in full scan mode. Quantification was then performed in the single ion storage (SIS) mode for better selectivity. Response factors were determined relative to the internal standards previously chosen to better fit to the properties of each compounds. No significant amount of analytes was detected in procedural blanks. Thus, the data did not need the blank corrected. The recovery rates of the analytical procedure for extraction of PAHs and PCBs have been previously studied and validated by Tronczynski *et al.* (2005). The procedure have been slightly modified for fractionation on a silica column step. The recovery rates have been studied and validated in the laboratory by spiking the targeted compounds into the natural sediment. The recovery rates obtained were more than 79% for PAHs, more than 73% for Me-PAHs and more than 75% for PCBs in algae and sediment samples. The limit of quantification of individual PCBs, PAHs and Me-PAHs were 0.8 µg/kg dw (except for PCB180 which was 2 µg/kg dw), 0.1-2 µg/kg dw and 0.2-0.4 µg/kg dw, respectively. Total mercury analysis was carried out with an Advanced Mercury Analyser (Altec, AMA 254) on powder samples without treatment at least three replicates for each sample. For a known amount of dried sample (10-40 mg), the metal was evaporated by progressive heating up to 800 °C under oxygen atmosphere and finally amalgamated on a gold-sand trap. Afterwards, the amalgamator was heated to liberate the collected mercury, which was finally measured by atomic absorption spectrophotometry. Certified reference materials (MESS-

3 and DORM-3, National Research Council Canada) were used to assess the accuracy and precision of the analyses. Mean recovery for total mercury was more than 86 % and the limit of quantification was 5 µg/kg dried weight (dw).

RESULTS & DISCUSSION

The mean contaminant levels of total mercury, total PAHs (Σ_{16}PAHs), total Me-PAHs (Σ_{18}Me-PAHs) and total PCBs (Σ_{28}PCBs) measured in surface sediment were at 22±22 µg/kg dw with the range of 5-95 (n=15), 197±240 with the range of 2-636 (n=15), 11±8 with the rage of 3-31 (n=15) and 58±81 µg/kg dw with the range of 4-333 µg/kg dw (n=15) respectively. Mean ± S.D (standard deviation) of individual concentration of targeted compounds were presented in Table 1. Individual concentration of PAH and Me-PAH detected in Soumbedioune station were one order of magnitude lower compared to previous study reported by Ndiaye *et al.* (2012) for the same station. Individual concentration of PCB were also detected at lower concentration. However, there is no data on organic contamination reported for two other sites selected in this study. Among these three sampling sites, Yarakh was the less contaminated by PAHs and PCBs. Our results showed that the distribution of PAH, Me-PAH, PCB and mercury was not homogenous. The concentration varied strongly form one sampling point or site to another (Table 1). Despite a large range between the minimum and maximum for contaminant levels, Σ_{16}PAHs showed a relatively high contamination level, around 9 times higher than mercury concentration, 3 times higher than Σ_{28}PCBs concentration and 18 times higher than Σ_{18}Me-PAHs concentration.

Table 1. Means ± S.D of individual concentration of targeted compounds (16PAHs, 18Me-PAHs, 28PCBs and total mercury) analyzed in sediment and marine organisms collected from three sampling stations in the Dakar coastal area: invertebrate species (soft body) and fish species (muscle). nd = level not detected (<LOQ) and na = not analysis. (n ≥3 replicate)

Compounds	Maximum admissible content (µg/kg wet weight)			
	Mussel	Fish	Crustacean	Reference
BaP, BaA, BbF, Chr	30	30[*]	30[*]	UE N° 835/2011
	3.4-20 (8)	0.2-4.3 (1.1)[1] 8 – 34(19)[2] 3.2-18 (10.1)[3] 1.8-3.2 (2.7)[4]	8-12 (10)	This work
PCB: 28, 52, 101, 138, 153, 180	75	75	75	UE N° 1259/2011
	0.7-12 (6)	n.d-12 (5)[1] n.d[2] 7-10 (8.5)[3] n.d-27 (6)[4]	nd	This work
Total mercury	100-500	500-1000	100-500	UE N° 466/2001/2006
	9-17 (12)	3-101 (34)[1] 11-38 (21)[2] 9-30 (22)[3] 4-18 (7)[4]	7-12 (9)	This work

The PAHs ranged between 2 and 636 µg/kg dw and the highest PAHs concentrations were obtained in sediments collected at Rufisque station and sampled from a zone close to the Refining African society and cement factory. The mean concentration of Σ_{16} PAHs was 396 ± 326 µg/kg dw (Fig. 2). Our results of PAHs was at least an order of mag nitude lower, compared to concentrations measured in superficial sediment collected in the Soumbedioune station (Hann Bay) during the dry season (Ndiaye *et al.,* 2012). Abdolahpur Monikh et al. (2014) have performed the study on PAHs contamination in Persian Gulf. The authors have reported the concentrations of total PAHs in sediment varied from 310.76 µg/kg dw at Boshehr province to 1106 µg/kg dw at Tangestan estuary; thus two order of magnitude higher than our results obtained from Dakar coastal. Ndiaye *et al.* (2012) reported a concentration level of 19 µg/kg dw for PCBs indicator in a sediment sample from Hann Bay. The levels of PCBs indicator (Σ_7 PCBs) were similar in Rufisque station and lower than literature data for the two other stations (Fig. 2). Similarly, surface sediments from two Senegalese stations (100 – 150 km South from Dakar) exhibited Σ_7 PCB concentrations ranging between 0.3 and 19.1 µg/kg dw (Bodin *et al.,* 2011). Sediments from Yarakh and Soumbedioune displayed levels of POPs lower than those from Rufisque station (Fig. 2). Yarakh station seems to be the less contaminated compared to other ones. The location of Rufisque station near the Refining African society and cement factory could explain the highest level of organic pollutants found at this station and more particularly the PAH compounds. Mercury concentrations ranged between 5 and 95 µg/kg dw and sediments from Rufisque station were the less enriched in this element. To our knowledge, no data for mercury concentration in sediments from this area or along the Senegalese coast were reported.

In general, the contamination level of pollutions are strongly conditioned by their origin. Numerous methods could be used to identify the origin of PAHs contaminations (De Lucas 2005; Gogou *et al.,* 1996; Simo *et al.,* 1997; Yunker *et al.,* 2002; Yunker and Macdonald, 1995; Dickhut *et al.,* 2000; Zhang *et al.,* 2005). However, to our knowledge, there is no specific method to identify the origin of PCBs and mercury. In this work, PAHs origin for the sediment samples was characterized by using the ratio of low molecular weight and high molecular weight (LMW/HMW, the sum of (2-3) / (4-6) aromatic rings). This ratios allow to distinguish the petrogenic (LMW/HMW > 1) from pyrolytic origins (LMW/HMW < 1) (De Lucas, 2005). Our results on LMW/HMW ratios revealed values less than 1, thus suggesting that combustion should be the dominant source of PAHs in the studied areas. In addition, ratios of molecular masses 178 and 228 are commonly used to distinguish combustion from petroleum sources. For the mass 178, Ant/(Ant + Phe) <0.10 suggests pollution of petroleum origin, while a ratio >0.10 indicates a dominance of combustion (Budzinski *et al.,* 1997; Yunker *et al.,* 2002; Liu *et al.,* 2008). For the mass 228, BaA/(BaA + Ch) suggests that a ratio <0.20 indicates petroleum inputs, a ratio between 0.20 and 0.35 indicates a mixed sources (either petroleum or combustion), and a ratio >0.35 indicates combustion sources (Yunker *et al.,* 2002). Our results obtained from isomers ratios of molecular masses 178 and 228 reinforce the combustion process as the major source of PAHs contamination in these three sampling sites. Moreover, such high Ant/(Ant+Phe) and BaA/(BaA+Ch) ratios involve high proportions of anthracene and benz*[a]*anthracene that are both among the most photoreactive PAHs (Gogou *et al.,* 1996). Considering a combustion/transport/ deposition process, it can be admitted that no significant
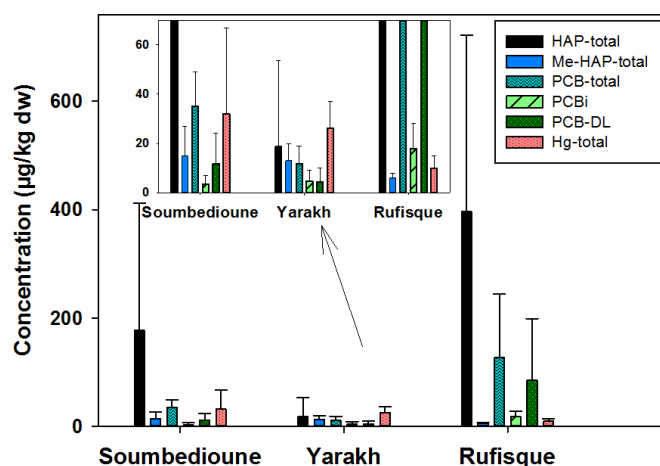
Fig. 2. Mean (±S.D.) of total PAH, Me-PAH, PCB and mercury concentrations (µg/kg dw) obtained in surface sediments collected at Soumbedioune, Yarakh and Rufisque stations. Levels of PCB indicators (PCBi) and dioxin-like PCBs (PCB-DL) are also indicated. n=five replicates

photolysis degradation occurred in this case study. These ratios can also imply close contamination sources that can be attributed to the nearby industrial activities or vehicle emissions of Dakar city.

However, there is no natural source of PCBs. Atmospheric depositions, runoff from the land, wood burning and food chain transport have been regarded as the major sources of PCBs in aquatic environment (Totten *et al.,* 2006). PCBs are mainly produced by pyrogenic process due to the combustion of organic matter (Pereira *et al.,* 1980; Kjeller and Rappe, 1995; Rose and Rippey, 2002; Rose *et al.,* 2004; Pandelova *et al.,* 2009). PCBs detected in Dakar coastal area could be originated from unintentionally atmospheric deposition, released from equipment, generators, ships, vehicles and trucks exhaust in the surrounding zone but the their origin could not be determined. Similarly, the specific origin of mercury could not identified. It is a global pollutant that is ubiquitous in the environment from both natural and anthropogenic sources reaching the ocean through river inputs and atmospheric deposition (Fitzgerald *et al.,* 2007). However, given their relatively low concentration of mercury detected, the domestic and industrial wastes discharged directly into Dakar Bay have not yet produced environmental mercury contamination in this coastal zone. Contrarily, two other toxic metals such as cadmium and lead present at high concentrations level in sediment of Dakar coastal (Diop *et al.,* 2012). For the ecotoxicological risk assessment, concentrations of total PCBs or Σ_{28}PCBs, sum of 12 PAHs or Σ_{12}PAHs (Aen, Ayl, An, F, N, Pn, BaA, BaP, Ch, DhA, Fl, Py) and total mercury were calculated. The mean concentrations of Σ_{12}PAHs were 279 (6 - 507), 119 (2 - 298) and 18 (1.2 - 80) µg/kg dw for Rufisque, Soumbedioune and Yarakh respectively. While Σ_{28}PCBs mean concentrations detected in the sediments from

Rufisque, Soumbedioune and Yarakh were 127 (57 - 332), 34 (16 - 47) and 12 (4 - 22) µg/kg dw respectively. Total mercury concentrations did not exceed 95 µg/kg dw. Total mercury, Σ_{12}PAHs and Σ_{28}PCBs concentrations were compared to Sediment Quality Guidelines (SQGs) which provide a valuable tool to evaluate the potential biological adverse effect on aquatic organism caused by contaminated sediments (Long et al. 1995). The ERL/ERM (Effect range low/Effect range median) developed by Long and Morgan (1990) and the TEL/PEL (threshold effect level/Probable effect level) developed by Macdonald *et al.,* (1996) are two based approaches which can be used to assess the ecological toxicity of total PAHs and PCBs concentrations (Σ_{12}PAHs and Σ_{28}PCBs) in sediments collected from Dakar coastal in Senegal. According to SQGs, the classification of biological adverse effects are rarely expected (<ERL/TEL) with minimal-effects range, occasionally (≥ERL/TEL and <ERM/PEL) with possible-effects range, and frequently (≥ERM/PEL) with a probable-effects range (Long *et al.,* 1995; Cardellicchio *et al.,* 2007). TEL-PEL values were 655-6676 and 22-189 µg/kg dw for the ΣPAHs and ΣPCB, respectively (Macdonald *et al.,* 1996). While ERL-ERM were 4022-44792, 23-180 and 150-710 µg/kg dw for the ΣPAHs, ΣPCBs and mercury, respectively (Long and Morgan, 1990). Our results showed that Dakar coastal sediments have rarely to occasionally biological adverse effects for mercury, PCBs and PAHs in Dakar aquatic ecosystems.

Overall, even if there is a strong influence of urban and industrial activities in this coastal zone, sediment contamination by PAHs, Me-PAHs, PCBs, and mercury were detected at moderate levels. This is probably due to the tide which is responsible to the rapid renewal of sea water, therefore sedimentation processes are not preferred. Indeed, the coastline around Dakar is called
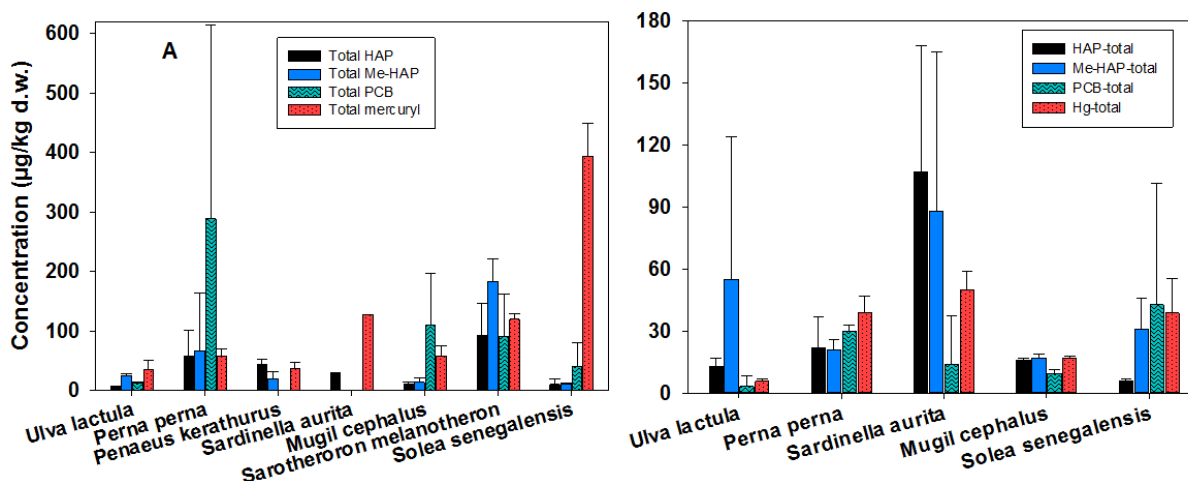

Fig. 3. Mean (±S.D.) concentrations of total PCBs, PAHs, Me-PAHs and Hg in biota samples: A) Soumbedioune station and B) Rufisque station

microtidal, the tidal range which varies between 0.5 m (neap period) and 1.6 m (during spring tides) (Ruffman *et al.,* 1977). This suggests that urban and industrial waste discharged into Dakar coastal zones could be diluted or transported rapidly seaward. Due to their high toxicity, persistency and bioaccumulation capacities, the concentration of these contaminants have been also studied in marine organism in order to evaluate the quality of edible species. Mean ±S.D of individual concentration of PAH, Me-PAH and PCB and mercury determined in biota samples from all sampling stations of the Dakar coastal area are shown in Table 1. Comparisons of targeted compounds were significantly different between the stations and between species. Considering all organisms, the mean concentration of mercury varies between 5 and 442 µg/kg dw. Generally, muscle tissues of fishes exhibited the highest concentrations of Hg compared to macroalgae and invertebrate species. The highest levels of mercury were detected in *Soleasenegalensis* from Soumbedioune station (Fig. 3). Mercury levels measured in *Sardinella aurita* were in the same order of magnitude compared to the mean concentration values reported for the same species collected along the coast of Mauritania in a previous study at 90 ± 80 µg/kg dw (Romeo *et al.,* 1999). For PCBs concentrations, the highest level was determined in *Perna perna* species (up to 1228 µg/kg dw) and the lowest level was found in *Penaeus kerathurus* species. At the base of the food chain, *Ulvalactula* species displayed low PCB concentrations (7 ± 6 µg/kg dw) while fish species exhibited mean values ranging between 10 ± 20 µg/kg dw for *Sardinella aurita* and 95 ± 32 µg/kg dw for *Sarotheroron melanotheron*. Except for *Perna perna*, *Penaeus kerathurus* and *Sardinella aurita*, PCB concentrations were in the same range between organisms and sediments while mercury concentrations were higher in biota than sediment samples. Whereas, PAH concentrations determined in edible tissues were lower than sediment samples. This may be due to a rapid transformation of PAHs into more hydrophilic metabolites. Therefore, marine organisms exposed to these compounds indicate only trace quantities of PAHs in their tissues (Vuorinen *et al.,* 2006). The concentration level in marine organisms varied from species to species (Table 1 and Fig. 3). This may be translated by behavioral patterns of organisms such as feeding habit, the rate of movement and reproduction status (Vuorinen *et al.,* 2006). The highest mean levels of PAHs and Me-PAHs (92 ± 54 and 183 ± 39 µg/kg dw respectively) were detected in the tilapia species, therefore the present study supports the idea to use this species as a sentinel in order to monitor chemical pollutants (Ndiaye *et al.,* 2012; Harrison and Whitfield, 2006). Our results showed moderate accumulations of PAHs and Me-PAHs in *Mugilcephalus* and *Soleasenegalensis*. For these two families of organic compounds, similar concentrations were determined in

biota samples collected from Soumbedioune and Rufisque stations (Fig. 3). Rose *et al.* (2012) have reported the concentration of ΣPAHs in Tilapia guineensis obtained from Lagos Lagoon of Nigeria at the same order level of 62.24 µg/kg dw. This result were the same order of magnitude as the concentration of ΣPAHs in Tilapia obtained in our study. Compared to other species of fish, Mullus barbatus and Serranus cabrilla collected from Tarragona of Mediterranean accumulated ΣPAHs at the levels of 164.9 and 62.9 µg/kg dw respectively (Escartin and Porte, 1999). Amodio-Cocchieri *et al.,* (1993) reported the concentration of ΣPAHs in Engralis enchrasicholus collected from the Bay of Naples of Italy at the level of 965 µg/kg dw. The highest concentration in fish have reported by McGill et al. (1987) in Limanda limanda collected from the British North Sea at the level of 2345 µg/kg dw. This concentration was much higher than ΣPAHs concentration detected in four species selected in our study. More recently, Abdolahpur Monikh et al. (2014) have also reported the concentration of PAHs in benthic, benthopelagic and pelagic fish species from the Persian Gulf much higher than our results. Σ_{16}PAHs concentrations detected in targeted species were lower compared to concentrations in sediment. However, Σ_{16}PAHs concentrations were detected in all marine species selected in this work. ΣMe-PAHs was present at low levels in the sediment samples (Fig. 2). However, their concentrations were detected at higher levels in most of marine species (Table 1; Fig. 3). The results clearly showed that all species does not have the same affinity to each type of contaminant. Generally, we observed the bioaccumulation of these contaminants in marine species. However, the bioaccumulation level varies strongly from one species to another (Table 1; Fig. 3). Each species accumulates at least two types of pollutants. Among targeted species, both *Sarotheroron melanotheron* and *Perna perna* accumulated all type of contaminants (PAHs, Me-PAHs, PCBs and mercury). The contamination level of PCBs in *Sarinella aurita* and *Penaeus Kerathurus* were under the detection limits (Fig. 3). For Yarakh station, all targeted compounds were quantified in the sediments. However, only mercury was quantified in marine species. For marine species collected in Yarakh site, high levels of mercury were detected in *Sardinella aurita* (71.6±23.3 µg/kg dw) and *Sarotheroron melanotheron* (71.3±28.0 µg/kg dw). *Ulvalactula, Mugilcephalus, Perna perna, Penaeus Kerathurus* and *Solea senegalensis* species accumulated levels of total mercury at 40.7±5.6, 21.7±6.0, 49.0, 31.0, 15.7±3.0 µg/kg dw respectively. To estimate the potential public health risks, the European Union legislation has established maximum allowed levels in aquatic products for different toxic compounds including some PAHs, PCBs and mercury (Table 2). PAHs, PCBs and mercury concentrations obtained in dry weight (µg/kg dw) were converted into µg/kg wet

weight (w.w.) in order to compare with the guideline values. According to the weighed mass before and after drying, the following factors were applied to obtain the concentrations in wet weight: 0.23 for muscle of *Mugilcephalus*, *Sarotherodon melanotheron*, *Solea senegalensis* and edible tissue of *Perna perna*, 0.30 for muscle of *Sardinella aurita* and 0.26 for *Penaeus kerathurus*. Two limit values are listed for mercury depending on the type of mussel and fish (Table 2). For all species, mercury and PCBs levels do not show

Table 2. Σ_4PAHs, Σ_6PCBs and total mercury detected in this work (range and mean values into brackets) compared to maximum admissible content expressed in µg/kg wet weight. n.d. = not detectable (<LOQ)

Compound	Sediment	*Ulvalactula* (green algae)	*Perna perna* (mussel, mollusks)	*Penaeus kerathurus* (shrimp, crustaceans)	*Sardinella aurita* (sardine, pelagicfish)	*Mugilcephalus* (mullet, bentho-pelagic fish)	*Sarotheroron melanotheron* (tilapia, bentho-pelagic fish)	*Soleasenegalensis* (sole, benthic fish)
Soumbedioune								
PAHs (µg/kg dw)								
N	3.4±6.3	nd	nd	nd	nd	nd	nd	nd
Acy	6.7±8.9	0.5±0.0	1.9±1.4	1.9±0.3	1.3	0.5±0.3	6.5±0.9	0.8±0.1
Acn	nd		nd	nd	nd	nd	nd	nd
F	4.2±5.7	0.4±0.2	1.5±2.0	0.3±0.2	0.9	nd	1.1±0.3	nd
An	25.7±35.1		1.4±2.8	0.2±0.4	nd	nd	1.5±1.8	nd
Fl	3.2±7.1	0.5±0.7	3.7±5.7	nd	nd	nd	8.9±8.4	nd
BaA	63.3±86.8	0.3±0.0	22.7±9.2	35.2±6.1	24.2	9.6±3.5	4.7±2.2	6.3±9.6
Chr	6.4±12.9	0.3±0.0	5.4±4.7	4.3±0.7	2.9	1.2±0.4	6.9±3.1	1.3±0.9
BaP	nd	nd	nd	nd	nd	nd	nd	nd
Pn	1.6±3.6	1.8±0.6	3.1±1.7	0.2±0.2	nd	nd	5.4±5.1	nd
BbF	33.7±35.5	0.3±0.0	11.6±16.7	nd	nd	nd	32.2±28.1	0.3±0.2
BkF	24.5±33.4	0.3±0.0	1.5±3.3	nd	nd	nd	11.8±15.0	0.3±0.2
Bghi	nd	nd	nd	nd	nd	nd	nd	nd
DhA	nd	nd	nd	nd	nd	nd	nd	nd
IP	nd	nd	nd	nd	nd	nd	nd	nd
Py	4.5±5.5	0.9±0.6	5.6±7.4	1.6±0.2	0.7	0.3±0.1	13.1±4.2	0.6±0.3
Me-PAHs (µg/kg dw)								
1M-Na	2.9±2.6	3.9±3.6	6.9±6.3	17.6±9.1	nd	4.0±5.7	28.2±31.9	nd
2M-Na	1.0±1.1	2.8±3.0	3.1±5.2	4.1±7.7	nd	0.1±0.1	38.4±30.7	nd
1,2-DM-Na	2.4±1.7	4.4±4.0	9.6±3.5	17.6±8.7	nd	6.2±1.9	27.5±31.0	9.5±1.2
1,6-DM-Na	2.7±2.0	5.8±6.8	7.1±11.4	8.4±17.3	nd	0.7±0.3	84.7±69.4	0.6±0.3
2,6DM-Na	0.9±1.0	1.2±0.0	1.6±1.1	0.6±1.0	nd	0.4±0.5	4.7±2.2	0.7±0.4
1M-Pn	nd	0.7±0.2	nd	nd	nd	nd	nd	nd
2M-Pn	1.2±2.7	1.8±0.5	nd	nd	nd	nd	nd	nd
3M-Pn	nd	2.1±0.4	nd	nd	nd	nd	nd	nd
9M-Pn+2Me-An	1.4±3.1	1.3±0.6	nd	nd	nd	nd	nd	nd
1,7DM-Pn	1.7±3.8	0.9±1.3	nd	nd	nd	nd	nd	nd
Retene	0.8±1.8	nd	11.9±22.7	1.6±1.0	nd	nd	nd	nd
1M-Fl	nd	nd	1.2±2.7	nd	nd	nd	nd	nd
3M-Fl	nd	nd	5.5±12.3	nd	nd	nd	nd	nd
1M-Py	nd	nd	6.6±14.7	nd	nd	nd	nd	nd
4M-Py	nd	nd	3.6±8.2	nd	nd	nd	nd	nd
3M-Ch	nd	nd	3.9±8.6	nd	nd	nd	nd	nd
6M-Ch	nd	nd	6.1±13.5	nd	nd	nd	nd	nd
PCBs n° (µg/kg dw)								
8	8.1±5.3	4.1±1.2	nd	nd	nd	nd	nd	nd
18	3.4±5.7	nd	nd	nd	nd	nd	nd	nd
28	0.8±1.2	0.9±0.2	nd	nd	nd	nd	5.5±5.5	nd
44	nd	nd	0.9±1.1	nd	nd	nd	1.6±2.7	0.9±1.0
52	nd	nd	7.4±7.6	nd	nd	nd	2.0±3.5	nd
66	6.4±14.1	nd	4.1±6.1	nd	nd	nd	nd	2.3±4.0
77	nd	nd	42.4±94.8	nd	nd	nd	nd	15.0±18.2
81	nd	nd	nd	nd	nd	nd	nd	nd
101	nd	nd	14.4±21.9	nd	nd	4.7±6.7	12.5±12.4	0.8±1.4
105	2.7±3.5	nd	nd	nd	nd	nd	nd	nd
114	nd	nd	88.3±197.4	nd	nd	13.3±5.4	44.3±76.7	nd
118	1.2±2.4	nd	5.6±8.0	nd	nd	nd	nd	nd
123	2.5±5.3	nd	nd	nd	nd	nd	nd	nd
126	nd	nd	nd	nd	nd	nd	nd	nd
128	nd	nd	nd	nd	nd	nd	nd	nd
138	0.8±1.5	4.0±1.1	54.4±116.8	nd	nd	35.3±42.8	nd	8.8±7.6
153	1.4±0.6	nd	42.1±61.1	nd	nd	27.0±19.3	7.7±13.4	1.0±8.5
156	nd	nd	nd	nd	nd	nd	nd	nd
157	nd	nd	nd	nd	nd	nd	nd	nd
167	nd	nd	nd	nd	nd	nd	nd	nd
169	nd	nd	nd	nd	nd	nd	nd	nd
170	nd	nd	23.7±37.5	nd	nd	2.4±3.3	nd	4.5±4.1
180	nd	nd	5.7±5.5	nd	nd	24.3±19.3	nd	nd
187	nd	nd	nd	nd	nd	nd	nd	nd
189	2.7±6.0	nd	nd	nd	nd	nd	nd	nd
195	2.3±4.2	4.1±0.7	nd	nd	nd	2.9±4.1	17.0±29.5	3.2±5.4
206	nd	nd	nd	nd	nd	nd	nd	nd
209	nd	nd	nd	nd	nd	nd	nd	nd
Total mercury (µg/kg dw)								
Mecury	32.0±35.0	36.0±15.0	58.8±11.3	37.6±9.2	128.0±0.0	58.7±17.1	119.7±9.3	393.7±55.5

concentrations exceeding limits fixed by the European Union. In addition, the levels of these compounds in the investigated area were low compared with these threshold values. Concerning PAHs, the proposed limit values are for smoked products of fish and crustacean species because recently the European legislation (Official Journal of the European Union, 2011) abrogated the limit of PAHs in these fresh seafood because it has been shown that PAHs are quickly metabolized in these species (Storelli *et al.*, 2013). In our study, limit level (30 µg/kg wet weight) was exceeded only in a *Sardinella aurita* sample with a muscle tissue concentration of 34 µg/kg wet weight. Based on European legislation, the selected species present good quality for food based on PCBs, PAHs and mercury.

Continue Table 2

Compound	Sediment	*Ulvalactula* (green algae)	Perna perna (mussel, mollusks)	*Penaeus kerathurus* (shrimp, crustaceans)	*Sardinella aurita* (sardine, pelagicfish)	*Mugilcephalus* (mullet, bentho-pelagic fish)	*Sarotheroron melanotheron* (tilapia, bentho-pelagic fish)	*Soleasenegalensis* (sole, benthic fish)
Rufisque								
PAHs (µg/kg dw)								
N	2.8±1.7	nd	nd	na	nd	nd	na	nd
Acy	10.9±8.8	0.4±0.1	0.7±0.1	na	3.7±1.6	0.4±0.4	na	1.3±0.0
Acn	nd	nd	nd	na	nd	nd	na	nd
F	7.6±6.1	0.3±0.1	0.3±0.2	na	1.9±2.2	0.3±0.1	na	0.1±0.0
An	76.2±76.3	0.1±0.2	2.1±0.1	na	9.6±6.2	0.5±0.7	na	0.7±1.1
Fl	1.4±1.9	0.4±0.7	0.9±0.6	na	3.1±5.3	nd	na	nd
BaA	117.0±95.0	7.2±2.2	13.3±1.3	na	68.0±30.2	11.3±1.2	na	0.8±0.2
Chr	0.1±0.2	0.9±0.3	1.6±0.4	na	8.3±3.7	1.4±0.1	na	0.8±0.2
BaP	nd	nd	nd	na	nd	nd	na	nd
Pn	83.3±186.3	2.2±0.6	2.9±1.5	na	7.6±7.6	1.5±1.5	na	nd
BbF	68.9±46.3	nd	nd	na	nd	nd	na	0.7±0.0
BkF	47.9±32.2	1.0±1.8	nd	na	nd	nd	na	0.7±0.0
Bghi	nd	nd	nd	na	nd	nd	na	nd
DhA	nd	nd	nd	na	nd	nd	na	nd
IP	nd	nd	nd	na	nd	nd	na	nd
Py	3.6±3.1	1.0±0.7	0.4±0.1	na	5.1±5.7	0.6±0.2	na	1.1±0.2
Me-PAHs (µg/kg dw)								
1M-Na	2.2±1.3	7.2±7.5	8.6±0.8	na	35.9±43.3	7.5±0.4	na	10.8±6.9
2M-Na	0.1±0.1	2.9±2.5	0.4±0.2	na	12.9±12.7	0.4±0.1	na	4.3±6.6
1,2-DM-Na	2.4±1.4	35.4±5.5	8.0±0.9	na	33.8±40.3	7.2±0.5	na	5.8±7.0
1,6-DM-Na	0.2±0.2	6.8±5.6	2.1±0.5	na	26.6±32.5	0.5±0.1	na	9.2±15.3
2,6DM-Na	nd	2.5±2.2	1.4±0.3	na	5.4±0.3	0.8±0.3	na	0.6±0.7
1M-Pn	nd	nd	nd	na	3.6±5.0	nd	na	nd
2M-Pn	nd	nd	nd	na	5.0±7.1	nd	na	nd
3M-Pn	nd	nd	nd	na	6.8±9.6	nd	na	nd
9M-Pn+2Me-An	nd	nd	nd	na	nd	nd	na	nd
Retene	nd	nd	nd	na	nd	nd	na	nd
1,7DM-Pn	0.6±1.4	0.2±0.4	nd	na	nd	nd	na	nd
1M-Fl	0.1±0.1	nd	nd	na	2.4±3.4	nd	na	nd
3M-Fl	nd	nd	nd	na	nd	nd	na	nd
1M-Py	nd	nd	nd	na	nd	nd	na	nd
4M-Py	0.1±0.1	nd	nd	na	nd	nd	na	nd
3M-Ch	nd	nd	nd	na	nd	0.7±1.2	na	nd
6M-Ch	nd	nd	nd	na	nd	0.4±0.6	na	nd
PCBs N° (µg/kg dw)								
8	15.8±9.3	nd	nd	na	nd	nd	na	nd
18	1.5±2.9	nd	nd	na	nd	nd	na	nd
28	2.0±2.3	nd	nd	na	nd	nd	na	0.8±1.1
44	nd	nd	nd	na	nd	nd	na	nd
52	nd	nd	nd	na	nd	0.9±0.9	na	nd
66	nd	nd	nd	na	nd	nd	na	nd
77	nd	4.4±6.2	nd	na	nd	nd	na	nd
81	nd	nd	nd	na	nd	nd	na	nd
101	5.3±6.0	nd	nd	na	nd	1.0±0.9	na	1.5±1.9
105	5.5±9.2	nd	nd	na	nd	1.1±1.9	na	nd
114	3.4±5.5	0.8±0.6	6.8±9.6	na	nd	nd	na	nd
118	3.2±3.0	nd	nd	na	nd	nd	na	nd
123	2.0±1.3	nd	nd	na	nd	nd	na	nd
126	4.9±8.2	nd	6.7±9.5	na	nd	nd	na	nd
128	1.0±2.1	nd	nd	na	nd	nd	na	nd
138	1.0±2.2	nd	nd	na	nd	3.4±0.7	na	10.8±16.3
153	0.2±0.3	nd	1.5±2.1	na	nd	3.4±1.2	na	3.8±5.4
156	nd	nd	nd	na	nd	nd	na	nd
157	2.4±5.4	nd	nd	na	nd	nd	na	nd
167	31.0±69.3	nd	nd	na	nd	nd	na	nd
169	nd	nd	nd	na	nd	nd	na	nd
170	1.0±1.4	nd	nd	na	nd	nd	na	1.7±2.6
180	6.3±11.6	nd	nd	na	nd	0.9±0.8	na	12.1±24.1
187	nd	nd	nd	na	nd	nd	na	nd
189	32.8±42.4	nd	nd	na	nd	nd	na	nd
195	7.6±11.0	nd	nd	na	13.7±23.8	nd	na	0.8±1.6
206	nd	nd	nd	na	nd	nd	na	nd
209	nd	nd	nd	na	nd	nd	na	nd
Total mercury (µg/kg dw)								
Mercury	10.0±5.0	5.6±1.4	39.0±8.0	na	49.7±9.3	17.0±1.0	na	38.7±16.8

CONCLUSIONS

Our study provide a data base about the organic contamination levels of PCBs, PAHs, Me-PAHs and total mercury in sediments and marine organisms obtained from Dakar coastal zone. The distributions of PAHs and Me-PAHs suggested that their sources were mainly originated from pyrolytic origin of nearby activities. The evaluation of ecotoxicological risk based on SQGs suggested that the biological adverse effects of PAHs, PCBs and total mercury are expected from rarely to occasionally for marine organisms. However, even if the impact of this type of pollution is moderate, it is now urgent to adopt adequate pollution control strategies into this coastal area before the problem become irreversible given the high density

Continue Table 2

Compound	Sediment	*Ulvalactula* (green algae)	**Perna perna** (mussel, mollusks)	*Penaeus kerathurus* (shrimp, crustaceans)	*Sardinella aurita* (sardine, pelagicfish)	*Mugilcephalus* (mullet, bentho-pelagic fish)	*Sarotheroron melanotheron* (tilapia, bentho-pelagic fish)	*Soleasenegalensis* (sole, benthic fish)
Yarakh								
PAHs (µg/kg dw)								
N	nd	na	na	na	na	na	na	na
Acy	0.2±0.0	na	na	na	na	na	na	na
Acn	1.2±1.8	na	na	na	na	na	na	na
F	nd	na	na	na	na	na	na	na
An	nd	na	na	na	na	na	na	na
Fl	5.38±11.7	na	na	na	na	na	na	na
BaA	1.3±2.6	na	na	na	na	na	na	na
Chr	1.7±2.5	na	na	na	na	na	na	na
BaP	nd	na	na	na	na	na	na	na
Pn	0.9±0.8	na	na	na	na	na	na	na
BbF	0.7±1.0	na	na	na	na	na	na	na
BkF	nd	na	na	na	na	na	na	na
Bghi	nd	na	na	na	na	na	na	na
DhA	nd	na	na	na	na	na	na	na
IP	nd	na	na	na	na	na	na	na
Py	6.8±14.2	na	na	na	na	na	na	na
Me-PAHs (µg/kg dw)								
1M-Na	1.7±1.4	na	na	na	na	na	na	na
2M-Na	0.9±1.4	na	na	na	na	na	na	na
1,2-DM-Na	1.7±1.1	na	na	na	na	na	na	na
1,6-DM-Na	3.1±2.7	na	na	na	na	na	na	na
2,6DM-Na	0.7±0.9	na	na	na	na	na	na	na
1M-Pn	0.6±1.2	na	na	na	na	na	na	na
2M-Pn	0.2±0.2	na	na	na	na	na	na	na
3M-Pn	0.4±0.6	na	na	na	na	na	na	na
9M-Pn+2Me-An	1.7±1.6	na	na	na	na	na	na	na
Retene	nd	na	na	na	na	na	na	na
1,7DM-Pn	nd	na	na	na	na	na	na	na
1M-Fl	nd	na	na	na	na	na	na	na
3M-Fl	nd	na	na	na	na	na	na	na
1M-Py	1.4±3.1	na	na	na	na	na	na	na
4M-Py	0.3±1.0	na	na	na	na	na	na	na
3M-Ch	nd	na	na	na	na	na	na	na
6M-Ch	nd	na	na	na	na	na	na	na
PCBs n° (µg/kg dw)								
8	2.4±5.4	na	na	na	na	na	na	na
18	0.8±0.5	na	na	na	na	na	na	na
28	nd	na	na	na	na	na	na	na
44	nd	na	na	na	na	na	na	na
52	nd	na	na	na	na	na	na	na
66	nd	na	na	na	na	na	na	na
77	3.2±3.6	na	na	na	na	na	na	na
81	nd	na	na	na	na	na	na	na
101	1.6±1.8	na	na	na	na	na	na	na
105	nd	na	na	na	na	na	na	na
114	nd	na	na	na	na	na	na	na
118	1.0±2.2	na	na	na	na	na	na	na
123	nd	na	na	na	na	na	na	na
126	nd	na	na	na	na	na	na	na
128	nd	na	na	na	na	na	na	na
138	0.9±0.9	na	na	na	na	na	na	na
153	1.5±1.3	na	na	na	na	na	na	na
156	nd	na	na	na	na	na	na	na
157	nd	na	na	na	na	na	na	na
167	nd	na	na	na	na	na	na	na
169	nd	na	na	na	na	na	na	na
170	0.8±0.7	na	na	na	na	na	na	na
180	nd	na	na	na	na	na	na	na
187	nd	na	na	na	na	na	na	na
189	nd	na	na	na	na	na	na	na
195	nd	na	na	na	na	na	na	na
206	nd	na	na	na	na	na	na	na
209	nd	na	na	na	na	na	na	na
Total mercury (µg/kg dw)								
Mercury	26.0±11.0	40.7±5.6	49.0±0.0	31.0±0.0	71.6±23.3	21.7±6.0	71.3±28.0	15.7±3.0

of population in this zone and continuous discharges of domestic and industrial effluents for most case without any treatment. Indeed, most of domestic and industrial waste waters are rejected directly into the sea and these permanent anthropogenic discharges can produce environmental and ecological degradation in the coastal zones. Moreover, the tide is quiet important in this coastal zone leading to a significant transport process of pollutants out to the sea. Particular attention must be paid to the chemical levels in edible marine organisms consumed by local population. Fish and invertebrates consumption remain the major source of protein for the residents in Dakar area. The result on the contamination level of PCBs, PAHs, Me-PAHs and total mercury in marine comestible species showed rarely to occasionally impact on aquatic organisms and no significant impact on human health by consuming targeted species namely mussel, crustacean, sole, sardine, tilapia, and mullet sampled from these coastal areas of Dakar. However, given the only few data available on the degree of contamination levels in these marine species and before making any final conclusion, it is strongly recommended to carry out complementary studies on these comestible species and others.

ACKNOWLEDGEMENTS

REFERENCES

Abdolahpur Monikh, F., Hosseini, M., Kazemzadeh Khoei, J., and Ghasemi, A. F. (2014). Polycyclic Aromatic Hydrocarbons Levels in Sediment, Benthic, Benthopelagic and Pelagic Fish Species from the Persian Gulf. Interntaional Journal of Environmental Research, **8(3)**, 839-848.

Amodio-Cocchieri, R., Del Prete, U., Arnese, A., Giuliano, M. and Roncioni, A. (1993). Heavy metals and polycyclic aromatic hydrocarbons (PAHs) in marine organisms from the Ionian Sea (Italy). Bull. Environ. Contam. Toxicol., **50,** 618-625.

Bastami, K. D., Afkhami, M., Ehsanpour, M., Kazaali, A., Mohammadizadeh, M., Haghparast, S., Soltani, F., Zanjani, S. A., Ghorghani, N. F., Pourzare, R. (2013). Polycyclic aromatic hydrocarbons in the coastal water, surface sediment and mullet Liza klunzingeri from northern part of Hormuz strait (Persian Gulf). Marine Pollution Bulletin, **76,** 411-416.

Bennett, B. G. (1983). (Staff member of the United Nations Environment Programme). Science Total Environment, **29,** 101.

Bertrand, K. A., Spiegelman, D., Aster, J. C., Altshul, L. M., Korrick, S. A., Rodig, S. J., Zhang, S. M., Kurth, T., Laden, F. (2010). Plasma Organochlorine Levels and Risk of Non-Hodgkin Lymphoma in a Cohort of Men. Epidemiology, **21(2),** 172–80.

Blumer M (1957) Removal of elemental sulfur from hydrocarbon fractions. Analytical Chemistry, **29,** 1039-1041.

Bodin, N., N'GomKa, R., Le Loc'h, F., Raffray, J., Budzinski, H., Peluhet, L., Tito de Morais, L. (2011). Are exploited mangrove molluscs exposed to Persistent Organic Pollutant contamination in Senegal, West Africa? Chemosphere, **84(3),** 318-327.

Boon, J. P. (1985). Processes determining the kinetics of PCB congeners in marine organisms: A comparison between laboratory and environmental studies. Marine Environmental Research, **17,** 301-305.

Borgå, K., Fisk, A. T., Hoekstra, P. F., Muir, D. C. G. (2004) Biological and chemical factors of importance in the bioaccumulation and trophic transfer of persistent organochlorine contaminants in Arctic marine food webs. Environmental Toxicology and Chemistry, **23,** 2367–238.

Bräuner, E. V., Sørensen, M., Gaudreau, E., LeBlanc, A., Eriksen, K. T., Tjønneland, A., Overvad, K., Raaschou-Nielsen, O. (2012). A prospective study of organochlorines in adipose tissue and risk of non-Hodgkin lymphoma. Environmental Health Perspectives, **120(1),** 105–111.

Cailleaud, K., Forget-Leray, J., Souissi, S., Hilde, D., Le Menach, K., Budzinski, H. (2007). Seasonal variations of hydrophobic organic contaminant concentrations in the water-column of the Seine Estuary and their transfer to a planktonic species Eurytemora affinis (Calanoïda, copepoda). Part 1: PCBs and PAHs. Chemosphere, **70,** 270-280.

Cardellicchio, N., Buccolieri, A., Giandomenico, S., Lopez, L., Pizzulli, F., Spada, L. (2007). Organic pollutants (PAHs, PCBs) in sediments from the Mar Piccolo in Taranto (Ionian Sea, Southern Italy). Marine Pollution Bulletin, **55,** 451–458.

Chiou, C. T., McGroddy, S. E., Kile, D. E. (1998). Partition characteristics of polycyclic aromatic hydrocarbons on soils and sediment. Environmental Science and Technology, **32,** 264-269.

Costa, P. M., Diniz, M. S., Caeiro, S., Lobo, J., Martins, M., Ferreira, A. M., Caetano, M., Vale, C., Del Valls, T. A., Costa, M. H. (2009). Histological biomarkers in liver and gills of juvenile Soleasenegalensis exposed to contaminated estuarine sediments: a weighted indices approach. Aquatic Toxicology, **92(3),** 202–212.

De Luca, G., Furesi, A., Micera, G., Panzanelli, A., Piu, P. C., Pilo, M. I., Spano, N., Sanna, G. (2005). Nature distribution and origin of polycyclic aromatic hydrocarbons (PAHs) in the sediments of Olbia Harbor (Northern Sardinia, Italy). Marine Pollution Bulletin, **50,** 1223–1232.

Dickhut, R. M., Canuel, E. A., Gustafson, K. E., Liu, K., Arzayus, K. M., Walker, S. E., Edgecombe, G., Gaylor, M. O., MacDonald, E. H. (2000). Automotive sources of carcinogenic polycyclic aromatic hydrocarbons associated with particulate matter in the Chesapeake Bay region. Environmental Science and Technology, **34,** 4635-4640.

Diop, C., Dewaele, D., Toure, A., Cabral, M., Cazier, F., Fall, M., Ouddane, B., Diouf, A. (2012). Étude de la contamination par les éléments traces métalliques des sédiments côtiers au niveau des points d'évacuation des eaux usées à Dakar (Sénégal). Revue des sciences de l'eau / Journal of Water Science, **25(3),** 277-285.

Dugan, J. E., Ichikawa, G., Stephenson, M., Crane, D.B., McCall, J., Regalado, K. (2005). Final report prepared for Central Coast Regional Water Quality Control Board, 38p.

Dumoulin, D., Net, S., Ouddane, B. (2013). Etude de la contamination en PCB de la Somme rivière-Campagne 2012. Repportpublished by Agence de l'Eau Artois-Picardie, (http://www.eauartois-picardie.fr/Etude-de-la-contamination-en-PCB.html).

Escartin, E. and Porte, C. (1999). Assessment of PAH Pollution in Coastal Areas from the NW Mediterranean through the Analysis of Fish Bile. Mar. Pollut. Bull., **38,** 1200-1206.

Fitzgerald, W. F., Lamborg, C. H., Hammerschmidt, C. R. (2007). Marine Biogeochemical Cycling of Mercury. Chemical Reviews, **107(2),** 641-662.

Francioni, E. F., Wagener, A. L. R., Scofield, A. L., Cavalier, B. (2005). Biomonitoring of polycyclic aromatic hydrocarbon in Perna perna from Guanabara Bay, Brazil. Environmental Forensics, **6,** 361–370.

Gaspare, L., Machiw,a J. F., Mdachi, S. J. M., Streck, G., Brack, W. (2009). Polycyclic aromatic hydrocarbon (PAH) contamination of surface sediments and oysters from the intertidal areas of Dar es Salaam, Tanzania. Environmental Pollution, **157(1),** 24-34.

Gogou, A., Stratigakis, N., Kanakidou, M., Stephanou, E.G. (1996). Organic aerosols in eastern Mediterranean: Components source reconciliation by using molecular markers and atmospheric back trajectories. Organic Geochemistry, 25, 79–96.

Harrison, T. D., Whitfield, A. K. (2006). Temperature and salinity as primary determinants influencing the biogeography of fishes in South African estuaries. Estuarine, Coastal and Shelf Science, **66,** 335–345

Health Effects of PCBs. U. S. Environmental Protection Agency. January 31, 2013.

IARC, (2010). Monographs on the evaluation of carcinogenic risks to humans: some non-heterocyclic polycyclic aromatic hydrocarbons and some related exposures, International Agency for Research on Cancer, France.

Jeanneau, L. (2007). Approche moléculaire quantitative appliquée à l'étude du transfert de micropolluants organiques à la confluence entre la Fensch et la Moselle (France). Thèse de Doctorat, Institut National Polytechnique de Lorraine, Nancy, France.

Kjeller, L.O. and Rappe, C. (1995). Time trends in levels, patterns and profiles for polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in a sediment core from the Baltic proper. Environmental Science and Technology, **29,** 346-355.

Ko, F. C., Baker, J. E. (1995). Partitioning of hydrophobic organic contaminants to resuspended sediments and plankton in the mesohaline Chesapeake Bay. Marine Chemistry, **49,** 171-188.

Kramer, S., Hikel, S. M., Adams, K., Hinds, D., Moon, K. (2012). Current Status of the Epidemiologic Evidence Linking Polychlorinated Biphenyls and Non-Hodgkin Lymphoma, and the Role of Immune Dysregulation. Environmental Health Perspectives, **120(8),** 1067–1075.

Long, E. R., Macdonald, D. D., Smith, S. L., Calder, F. D. (1995). Incidence of Adverse Biological Effects With in Ranges of Chemical Concentrations in Marine and Estuarine Sediments. Environmental Management, **19(1),** 81-97.

Long, E. R., Morgan, L. G. (1990). The potential for biological effects of sediment-sorbet contaminants tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52, 220.

Macdonald, D. D., Carr, R. S., Calder, F. D., Long, E. R., Ingersoll, C. G. (1996). Development and evaluation of sediment quality guidelines for Florida coastal waters. Ecotoxicology, **5,** 253–278.

Manodori, L., Gambaro, A., Piazza, R., Ferrari, S., Stortini, A. M., Moret, I., Capodaglio, G. (2006). PCBs and PAHs in sea-surface microlayer and sub-surface water samples of the Venice Lagoon (Italy). Marine Pollution Bulletin, **52,** 184-192.

McGill, A. S., Mackie, P. R., Howgate, P. and Mar, M. J. G. (1987). The flavour and chemical assessment of dabs (Limanda limanda) caught in the vicinity of the Beatrice oil platform. Pollut. Bull., **18,** 186-189.

Mostert, M. M. R., Ayoko, G. A., Kokot, S. (2010). Application of chemometrics to analysis of soil pollutants. Trends Analytical Chemistry, **29,** 430-435.

Naes, K., Knutzen, J., Berglind, L. (1995). Occurrence of PAH in marine organisms and sediments from smelter discharge in Norway. Science of Total Environment, **24,** 93-106.

Ndiaye, A., Sanchez, W., Durand, J. D., Budzinski, H., Palluel, O., Diouf, K., Ndiaye, P., Panfili, J. (2012). Multiparametric approach for assessing environmental quality variations in West African aquatic ecosystems using the black-chinned tilapia (Sarotherodon melanotheron) as a sentinel species. Environmental Science and Pollution Research, **19,** 4133–4147.

Neff, J. M. (2002). Bioaccumulation in marine organisms - effect of contaminants from oil well produced water, Elsevier, The Netherlands, 452.

Net, S., Dumoulin, D., El-Osmani, R., Rabodonirina, S., Ouddane, B. (2014a). Case study of PAHs, Me-PAHs, PCBs, phthalates and pesticides contamination in the Somme River water, France. Accepted for publication in International Journal of Environmental Research.

Net, S., Dumoulin, D., El-Osmani, R., Delcourt, V., Bigan, M., Ouddane, B. (2014b). Experimental design approach to the optimization of hydrocarbons extraction from the sediment: Method development and application. Applied Geochemistry, **40,** 126-134.

OIS (Observatoire des Industries du Sénégal) (2010) Répartition régionale des industries en 2000. (http:/www.obs-industrie.sn).

Pandelova, M., Stanev, I., Henkelmann, B., Lenoir, D., Schramm, K. W. (2009). Correlation of PCDD/F and PCB at combustion experiments using wood and hospital waste.

Influence of (NH4)2SO4 as additive on PCDD/F and PCB emissions. Chemosphere, **75,** 685–691.

Panfili, J., Mbow, A., Durand, J.D., Diop, K., Diouf, K., Thior, D., Ndiaye, P., Laë, R. (2004). Influence of salinity on the life-history traits of the West African black-chinned tilapia (Sarotherodon melanotheron): Comparison between the Gambia and Saloum estuaries. Aquatic Living Resources, **17(01),** 65-74.

Pereira, W. E., Rostad, C. E., Taylor, H. E. (1980). Mount St. Helens, Washington, 1980 volcanic eruption: characterization of organic compounds in ash samples. Geophysical Research Letter, **11,** 953–954.

Porta, M., Zumeta, E. (2002). Implementing the Stockholm Treaty on Persistent Organic Pollutants. Occupational and Environ Medicine, **10(59),** 651-2.

Rainbow, P.S., Phillips, D. J. H. (1993). Cosmopolitan biomonitors of trace metals. Marine Pollution Bulletin, **26(11),** 593-601.

Roméo, M., Siau, Y., Sidoumou, Z., Gnassia-Barelli, M. (1999). Heavy metal distribution in different fish species from the Mauritania coast. Science of Total Environment, **232(3),** 169-175.

Rose, A., Ken, D., Kehinde, O. and Babajide, A. (2012). Bioaccumulation of Polycyclic Aromatic Hydrocarbons in Fish and Invertebrates of Lagos Lagoon, Nigeria. J. Emerging Trends Engineering Applied Sci., **3,** 287-296.

Rose, N. L., Rippey, B. (2002). The historical record of PAH, PCB, trace metal and fly-ash particle deposition at a remote lake in north-west Scotland. Environmental Pollution, **117,** 121-132.

Rose, N. L., Rose, C. L., Boyle, J. F., Appleby, P. G. (2004). Lake sediment evidence for local and remote sources of atmospherically deposited pollutants on Svalbard. Journal of Paleolimnology, **31,** 499–513.

Ruffman, A., Meagher, L. J., Stewart, J. M. G. (1977). Bathymétrie du talus et du plateau continental du Sénégal et de la Gambie, Afrique de l'Ouest, in Le Baffin. Levé au large. Sénégal et la Gambie, vol. 1, Ministère des pêches et de l'environnement, Ottawa, pp. 23-90.

Sidoumou, Z., Gnassia-Barelli, M., Siau, Y., Morton, V., Roméo, M. (2005). Distribution and Concentration of Trace Metals in Tissues of Different Fish Species from the Atlantic Coast of Western Africa. Bulletin of Environmental Contamination and Toxicology, **74(5),** 988-995.

Sidoumou, Z., Gnassia-Barelli, M., Siau, Y., Morton, V., Roméo, M. (2006). Heavy metal concentrations in molluscs from the Senegal coast. Environmental International, **32(3),** 384-387.

Simo, R., Grimalt, J. O., Albaigés, J. (1997). Loss of unburned-fuel hydrocarbons from combustion aerosols during atmospheric transport. Environmental Science and Technology, **31,** 2697-2700.

Storelli, M. M., Barone, G., Perrone, V. G., Storelli, A. (2103). Risk characterization for polycyclic aromatic hydrocarbons

and toxic metals associated with fish consumption. Journal of Food Composition and Analysis, **31,** 115–119

Straif, K., Baan, R., Grosse, Y., Secretan, B., El Ghissassi, F., Cogliano, V. (2005). On behalf of the WHO International Agency for Research on Cancer Monograph Working Group. Carcinogenicity of polycyclic aromatic hydrocarbons. Lancet Oncology, **6,** 931-932.

Totten, L. A., Panangadan, M., Eisenreich, S. J., Cavallo, G. J., Fikslin, T. J. (2006). Direct and indirect atmospheric deposition of PCBs to the Delaware River Watershed. Environmetnal Science and Technology, **40,** 2171-2176.

Tronczynski, J., Munschy, C., Héas-Moisan, K., Guiot, N., Truquet, I. (2005). Analyse de contaminants organiques (PCB, OCP, HAP) dans les sédiments marins, Ed. Ifremer. Méthodes d'analyse en milieu marin, 44p.

Turrio-Baldassarri, L., Abballe, A., Casella, M., di Domenico, A., Iacovella, N., La Rocca, C. (2005). Analysis of 60 PCB congeners in drinkable water samples at 10–50 pg/L level. Microchemical Journal, **79,** 193-199.

US EPA (Environmental Protection Agency) (2002) Priority Pollutants. <http://www.epa.gov/NE/npdes/permits/generic/prioritypollutants.pdf> (accessed 01.06.12).

Vuorinen, P. J., Keinanen, M., Vuontisjarvi, H., Barsien, J., Broeg, K., Forlin, L., Gercken, J., Kopecka, J., Kohler, A., Parkkonen, J., Pempkowiak, J. and Schiedek, D. (2006). Use of biliary PAH metabolites as a biomarker of pollution in fish from the Baltic Sea. Mar. Pollut. Bull., 53, 479-487.

Wang, Z., Chen, J., Yang, P., Qiao, X., Tian, F. (2007). Polycyclic aromatic hydrocarbons in Dalian soils: distribution and toxicity assessment. Journal of Environmental Monitoring, 9(2), 199-204.

WHO, World Health Organization, Environmental Health Criteria 140: Polychlorinated Biphenyls and Terphenyls, Second edition, International Program on Chemical Safety (IPCS), Geneva, 1993, p. 21.

Yunker, B., Perreault, A., Lowe, C. J. (2012). Source apportionment of elevated PAH concentrations in sediments near deep marine outfalls in Esquimalt and Victoria, BC, Canada: is coal from an 1891 shipwreck the source? Org Geochemisty, **46,** 12-37.

Yunker, M. B., Macdonald, R. W. (1995). Composition and origin of polycyclic aromatic hydrocarbons in the Mackenzic River and on the Beaufort Sea shelf. Arctic, **48,** 118-129.

Yunker, M. B., Macdonald, R. W., Vingarzan, R., Mitchell, R. H., Goyette, D., Sylvestre, S. (2002). PAHs in the Fraser River basin: A critical appraisal of PAH ratios as indicators of PAH source and composition. Organic Geochemistry, **33(4),** 489-515.

Zhang, X. L., Tao, S., Liu, W. X., Yang, Y., Zuo, Q., Liu, S. Z. (2005). Source diagnostics of polycyclic aromatic hydrocarbons based on species ratios: A multimedia approach. Environmental Science and Technology, **39,** 9109-9114.