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ORIGINAL PAPER

Seasonal variations of polycyclic aromatic hydrocarbons in coastal sediments of a marine resource hot spot: the case of pars special economic energy zone, Iran

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Abstract Polycyclic aromatic hydrocarbons (PAHs) are an important group of compounds of major environmental concern, which are in the class of persistent organic pollutants. Therefore, the key purpose of this research was to analyze seasonal fluctuations and to determine the probability of polycyclic aromatic hydrocarbons in coastal sediments of the Iranian Marine Resource Center based on the evaluation of 16 US-EPA important PAH compounds. These compounds have been collected from intertidal sediments located in the marine resources of southern Iran in different seasons. These samples of the surficial sediment were collected at the PSEEZ area using a stainless steel grab sampler in four seasons, from depths between 0.5 and 30 m. Surface sediment samples were removed by spoons and carefully placed in an aluminum foil; they were taken to the laboratory on ice and held at 20° C until their study. After extraction, by using a rotary evaporator apparatus, samples were condensed. The assay was added to roughly 2 g of activated copper flasks in the refrigerator for 36 h for desulfurization. Among different seasons, the highest concentration was observed in winter, with a mean of 281.3 ng g^{-1}. According to ecological risk assessment (concentrations of possible effects, low effect range, degree of threshold effects, and median effect range), PAH risks in surface sediments of PSEEZ were lower than the

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threshold results levels (TEL), possible effects levels (PEL), low range of effects (ERL), and median range of effects (ERM), indicating that a biological effect would rarely occur. The dry weight scale of the concentration of $\sum$PAHs ranges from 145.7 to 348.42 ng g^{-1} with a mean quantity of 260.52 ng g^{-1}. Therefore, according to the amount of $\sum$PAH concentration, the sediments in the PSEEZ area indicated moderate to heavy pollutions. In this way, the sedimentary surface ecosystems of the Persian Gulf were considered as moderately polluted compared with other ecosystems worldwide. Our study highlighted some of the research gaps in PAH contamination studies and the level of PAH contamination. Therefore, this study will provide a scientific background, planning, and policies for PAH pollution control and environmental protection in Iran and similar regions around the world.

Keywords Environmental risk assessment · Diagnostic ratios · Sediments pollution · Petroleum industries

Introduction

The Persian Gulf is a low, deepwater region with a mean depth of 35 m and an area of 240,000 km^2 (Agah et al., 2009; Akhbarizadeh et al., 2019), which is located in the south of Iran and is connected to the international waters via Hormuz strait. Intense human activity is observed in this region due to the extremely rich reserves of oil, gas, and minerals. Rapid industrialization, urbanization, transportation, and development of oil refineries and petrochemical industries, along with tourism and aquatic activities, are commonplace activities in the region (Gevao et al., 2016; Lübeck et al., 2016). As a result of such an extensive development, various types of pollutants, including organic and inorganic ones like nutrients, toxic metals, and volatile organic compounds (VOCs (PAHs)), have been massively released into marine sediments (Davoodi et al., 2017). The activities may lead to an increase in the amounts of PAHs in coastal areas. According to Zhang, Wang, et al. (2020), Zhang, Pan, et al. (2020) and Mehr et al. (2020), PAHs commonly come to the coastal zone as a result of municipal and industrial wastewater dump, urban runoff, combustion processes associated with petroleum industries projects, and urban activities (including transportation and other municipal activities) (Mehr et al., 2020; Zhang, Pan, et al., 2020; Zhang, Wang, et al., 2020).

As a result of the widespread development of human activities, various types of pollutants, including organic and inorganic pollutants such as nutrients and PAHs, have been significantly released in coastal sediments, which have destructive effects. This release will affect human health and ecosystems and other organisms (Sharma & Chatterjee, 2017; Smith et al., 2018). Combustion of multiple fossil fuels and penetration of refined products and oils are widely known as mutants, carcinogens, and teratogens, and therefore endanger human health. The physicochemical properties of PAHs make them highly transportable materials in the environment and allow them to be distributed through water, soil, and air ecosystems. As a result, humans, as living organisms that depend on other living organisms for their survival, including aquatic organisms, will be affected by the adverse effects of this phenomenon. Therefore, awareness of the dangerous effects of these toxic substances on deep-sea creatures is essential to maintain the well-being of human beings (Karbalaei et al., 2018; Laffoley et al., 2019; Lu et al., 2019). PAHs are thus an essential category of compounds of considerable environmental significance. PAHs are highly mutagenic and humanly carcinogenic. The main cause of human susceptibility to PAHs is food exposure. As a result of food processing methods such as curing, drying, burning, roasting, grilling, barbecuing, and refining, PAHs are found in food (Adeyeye, 2020; Tongo et al., 2017). It is understood that these food processing steps produce and increase the amount of PAHs in the food. Sea food is one major food source of PAHs. In developed countries, most diets contain fish and other aquatic foods as a common source of protein. Smoke not only provides a special flavor and fragrance to these foods but also increases preservation because of its dehydrating and bactericidal effects. Smoke, however, includes PAHs, many of which are carcinogenic (Bauer et al., 2018; Malarut & Vangnai, 2020; Sun et al., 2019). Smoking is the most common process used to conserve seafood in developing countries such as Iran (south of Iran), and smoked food items are the most available source of food for consumption. Thus, the risk of pollution of PAHs by ingestion has increased. Food protection is of rising concern worldwide and PAH residues could raise significant public health issues if they are found in sea foods above the prescribed thresholds. The high concentration of benzo[a]pyrene in C. gariepinus and E. fimbriata above the guideline value of 0.05 mg/kg calls for serious health concern (Yusuf et al., 2015). As a result, if the PAH4 levels in food are higher than recommended, there will be serious concerns for human health. The findings of this study reveal that there are significant toxicity and potential carcinogenic risk to human health from the ingestion of Southern Iranian species of seafood.

PAH compounds are a type of organic chemical materials that comprise two to seven molten aromatic carbon rings with an angular linear and cluster arrangement. These substances are categorized into high molecular weight (HMW) and low molecular weight compounds, depending on their molecular weight (LMW) (containing 4–6 combined rings) (containing two or three benzene rings) compounds (Kim et al., 2013; Konstantinova et al., 2020; Oracz & Zyzelewicz, 2019; Tolosa et al., 2005). PAH compounds are also one of the most substantial groups of hazardous pollutants that are generated from both anthropogenic and natural origins (Darılmaz et al., 2013). The burning of numerous fossil oils and line oil and refined products penetration provide the anthropogenic basis (Liu et al., 2012; Yun et al., 2016). Some of these compounds are widely known as mutagens, carcinogens, and teratogens, and therefore endanger crucially the health of living organisms (Taghvaee et al., 2018). The physicochemical traits of PAHs make them highly moveable materials in the environment, permitting them to distribute through water, soil, and air ecosystem where their presence is ubiquitous (Baklanov et al., 2007; McCubbin et al., 2019; Qiao et al., 2006; Sverdrup et al., 2002). In aquatic ecosystems, PAH compounds bind to suspended solids and are deposited (Hu et al., 2014; Kesavvarzifard et al., 2017). Consequently, the impurity of hydrocarbons in sediments will endanger deep-sea creatures. Therefore, as a result of consuming them as a food source, it may lead to acute or chronic poisoning in humans (Bellasi et al., 2020; Ramzi et al., 2017; Zeng et al., 2013; Zhang, Pan, et al., 2020a, b).

Human actions increase the concentration of PAHs in waters and sediments in aqueous environments. The general global issues resulting from the growth of industrial and agricultural operations are water pollution and the lack of access to safe water (Wang et al., 2017). In recent decades, chemical compounds such as polycyclic aromatic hydrocarbons (PAHs) have been detected extensively in aquatic environments. Moreover, the number of new organic compounds entering the world market is growing significantly every year, and most of these compounds, including pharmaceuticals, pesticides, personal care goods, and PAH surfactants, are used in vast amounts in farming operations worldwide, after which they are released into various water sources where they can live, creating extreme heaven (Zambianchi et al., 2017). Recent studies have stated the occurrence of organic pollutants, such as PAHs, in various aquatic systems such as influent and effluent from wastewater treatment plants, groundwater, surface waters, or seawater (Grandclément et al., 2017). The discovery of organic compounds has been confirmed by researchers in America (Gilliom, 2007), Africa (Edokpayi et al., 2017), and Asia (Lin et al., 2018). PAHs have been recognized as soil mixtures, water sources, and crustacean drainage and usually co-occur with other contaminants (Jaward et al., 2012; Ozaki et al., 2019). PAHs are mostly released into the atmosphere as byproducts of the combustion of fuels, but agricultural explosions, industrial waste, and cooking can increase the discharge of these dangerous chemicals. The solubility of PAHs in water typically reduces as the molecular weight increases as their boiling and melting point increases (Adeniji et al., 2018). Adeniji et al. (2018) said that four-ring and five-ring PAHs, such as chrysene and benzo[a]pyrene, are practically insoluble in water. Because of this characteristic, particulate matter can be applied to the surface, and this mechanism defines the primary transport path of PAHs from land and air to marine habitats as well as from the water surface to lower depths (Vagge et al., 2018). Karyab et al. (2013) proposed that PAHs usually infiltrate water sources by dry and wet deposition, road runoff, leaching from creosote impregnated wood, industrial waste water, chemical spills, and fossil fuel combustion. Many researchers have thus verified PAHs in the contamination of marine ecosystems. In regions with a manifold of urban and industrial activities, sediments comprise high levels of these pollutants (Abdollahi et al., 2013; Birch, 1996; de León-Gómez et al., 2020). Sediments contain the highest amount of PAH accumulation in aquatic ecosystems (Gagnon & Fisher, 1997; Yan et al., 2014). The major resources of organic and inorganic contaminants in the coasts of the Persian Gulf are numerous industrial operations such as shipping, fishing, sewage, and wastewater discharges, urban runoff, and landfill leachate (Abdollahi et al., 2013; Chen et al., 2013; Huang et al., 2012; Mostafa et al., 2009; Readman et al., 2002).

According to the above-mentioned explanations, massive industrial activities (e.g., oil industries, transportation, fishing, sewage and wastewater discharges, urban runoff, and landfill leachate) in the Persian Gulf coast can result in two major hypotheses:

i) The coastal sediments of the Gulf have been polluted by PAHs and their main source is petrogenic.

ii) There is a difference in the amount of PAHs in various seasons.

Therefore, the key aim of this study is the identification of the concentrations and distribution of sixteen main PAHs and sediments of PSEEZ in the Persian Gulf, as well as identifying their potential sources and effects on the bio-resources.

Materials and methods

In the present study, PSEEZ was the study area (Fig. 1). This site is an industrial area containing petrochemical industries (Asaluyeh) and human activities such as fishing and oil transport. Consequently, the effluent of untreated municipal wastewaters and hospital sewages are discharged into this site. In this study, point stations were selected, first, to cover all case study (PSEEZ) coasts, and second, to distinguish all probable local sources, industries, and human activities that cause high pressures with respect to the levels of PAHs in this region. Samples were collected within different seasons in 2013 from 16 sampling sites in the coastal area of the Persian Gulf at the PSEEZ (Fig. 1); three subsamples were taken at each site.

Samples of the surficial sediment were collected at the PSEEZ area using a stainless steel grab sampler in four seasons, from depths between 0.5 and 30 m (Fig. 1). Surface sediment samples were removed by spoons and carefully placed in an aluminum foil; they were then taken to the laboratory on ice and held at 20° C until their study. In the laboratory, each sample was homogenized and freeze-dried (− 20 °C) for PAH analysis. Before the extraction, with 300 ng.g^{-1} of deuterated surrogates, 20 g of homogenized sediment until extraction, the sample was spiked to analyze the methodological recovery (acenaphthene-d10, phenanthrene-d10, naphthalene-d8, chrysene-d12, and perylene-d12). Soxhlet extraction was conducted for the extraction of the sixteen essential and priority PAHs of the US-EPA: (Naphthalene = Naph), (Fluorene = Fl), (Acenaphthene = Ace), (Acenaphthylene = Acy), (Phenanthrene = Phen), (Fluoranthene = Fluo), (Anthracene = Anthr), (Aenzo[a]pyrene = BaP), (Benzo [a] anthracen = BaA), (Pyrene = Py), (Benzo[b]fluoranthene = BbF), (Chrysene = Chry), (Dibenz [a,h] anthracene = DahA), (Benzo [g,h,i] perylene =

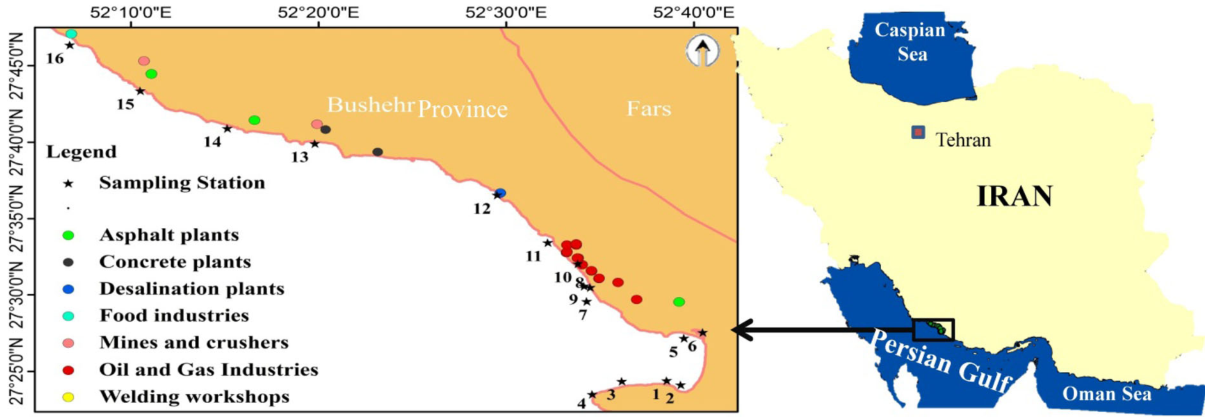

Fig. 1 Place and sampling site of the case study area in PSEEZ, Persian Gulf, Iran

BghiP), and (Indeno [1,2,3-cd] pyrene = IDP) (Benzo [a] anthracene = BaA). Five grams of sediment sample were weighed and placed in the extractor, and a 250 ml blend of dichloromethane and n-hexane (1:1, v/v) was obtained for 72 h by Soxhlet. Samples were condensed to a volume of 5 ml after extraction using a rotary evaporator apparatus. In the sample flasks for desulfurization (removal of elemental sulfur) for 36 h in the refrigerator, 2 g of activated copper was applied. Samples were then purified and condensed until the volume of the rotary evaporator method was reduced to 2 ml and the main volume was reduced to 2 ml. The solvent was substituted for hexane. With solid phase extraction (column of alumina/silica gel (300 mm? 10 mm i.d)), the hexane extract was distilled and then reduced to 1 ml using a purified nitrogen stream for instrumental analysis. 100 µl of terphenyld14 was put as an internal norm on each sample prior to the instrumental analysis (GC–MS). By the internal standard procedure, target analytes were quantified. Organic material content was detected through (LOI) according to the guidelines (Heiri et al., 2001). For data analyses, mean and standard deviation levels of PAHs were measured and the correlation analysis test was executed using SPSS 21.0 software package.

Sediments are an inherent aspect of the marine ecosystem and, due to their inadequate nitrogen loading, in nurturing floodplains and farm fields; they have been seen to be helpful. Nevertheless, the harmful impacts of the sediments are very alarming: (a) they include benthic marine population smothering (Hull et al., 2014; Mocq & Hare, 2018); (b) the reduction in the capability of shipping channels impedes navigation and thus has a negative impact on the country's economy; (c) they cause the capacity reduction of dams and reservoirs; (d) their presence leads to hydroelectric turbine obstruction (Yun et al., 2016); (e) they cause a reduction in water treatment plant and lagoon capability (Collivignarelli et al., 2018; de Magalhães et al., 2019; Fabião et al., 2016); and (f) the scenic and leisure importance of the rivers is declining. This shows that sediments pose a significant threat to experts and scientists interested in the sectors of water supply, wastewater, environmental cleaning, and dredging (de Magalhães et al., 2019). Sediments will serve as a drain or outlet for pollutants situated at the bottom or in the bed of the water body. This is an ongoing challenge for the contemporary society and needs particular consideration from planners, scientists, engineers, and other players of society. This is a continuing challenge for mainstream society and needs relevant consideration from planners, scientists, engineers, and other social actors. However, only a few attempts by civil engineers have been made to characterize these sediments as a potential raw material for successful mining, safe disposal, and use for sustainable growth.

More specifically, in order to meet the aims of using sediments as a resource, they should be defined on the basis of their general characteristics as an intrinsic part of the body of water (i.e., physical, morphological, geological, geotechnical, bacteriological, and toxic characteristics) which can be changed by the presence of benthic fauna. It should be remembered here that their sampling may be very intricate and advanced for such an elaboration of the classification of sediments (Collivignarelli et al., 2018; de Magalhães et al., 2019).

Quality control and quality assurance data over the years

The rigorous process-oriented quality assurance (QA) model has been developed and applied by ONC in combination with a product-oriented data quality management (QC) model. This model regularly intercepts and analyzes the instruments and the data source at various points of the instrument operation to ensure long-term data quality control and assurance. Five separate phases have been established by the QAQC model data that include data quality management and assurance measures during the life cycle of an instrument (Cowger and Gray 2020; Loh et al., 2017; D'Amore et al., 2015; Porter et al., 2012). Pre-deployment tracking covers all QAQC data/metadata checks conducted up to final deployment at the pre-deployment testing stage of an instrument. Post-deployment commissioning requires all QAQC data/metadata checks, from actual deployment to commissioning of the data as effective or compromised. Automated quality checking covers all real-time or delayed QAQC-related data checks performed when the tool is implemented through automated quality control procedures.

Manual quality management methods contain all QAQC data checks conducted by detailed manual data assessments and routines for annotation. Post-recovery monitoring includes all post-calibration checks undertaken during the post-recovery and restoration phases of the instrument. These phases describe a workflow of QAQC that minimizes individual and/or systemic errors during the life cycle of an instrument to ensure high-quality data. QC and QA monitor the efficiency of measurement devices that essentially lead to the scheduling of maintenance expeditions and instrument platform calibrations. The research and development of enhanced and emerging control systems were complementary to these two processes.

Quality assurance data

Data QA procedures are proactive steps to reduce data source problems and inaccuracies while restricting corrective measures to increase the accuracy of data. The QA data portion requires procedures to ensure that the instrument sensor network's protocols are correctly specified and adhered to. Periodic manual data processing by ONC data professionals, incorporation of annotations for data validation, and end-to-end workflow execution activities are examples of QA procedures currently in operation (Abatemarco et al., 2018; Abeysirigunawardena et al., 2015; Integrated, 2015).

Quality control data

QC data quality management is a product-oriented method for the identification and flagging of questionable information after it has been produced. QC requires both automatic and manual procedures to verify whether the information satisfies the consistency specifications outlined by the end users.

Three components compose the QC of information. The first part dynamically tests real-time data before parsing the data into the database. Using delayed-mode automatic testing, the second factor tests near-real-time or archived results. Manual inspection is the third part, where a specialist inspects the consistency data for problems (Abeysirigunawardena et al., 2015).

The method detection limit (MDL) was measured by the analysis of five sediment samples through the injection of low concentrations (lower than the concentration level on the calibration curve resulting in a detectable peak) of the target analytes until a small peak at ~ 3:1 (signal: noise ratio) was achieved. The MDL was calculated by the standard deviation of the five replicates and the value varied between 0.02 and 0.14 ng g^{-1} (mean 0.06 ng g^{-1}). Concentrations lower than MDLs have been defined as non-detected (N.D.). Target analytes were also quantified using the internal standard method. The practical quantification limits (PQL) for all the sixteen PAH compounds were lower than 0.15 ng g^{-1}.

Results

The values, mean ± SD, and range of sixteen essential and priority PAHs from the US-EPA compounds in sediment samples are represented in Tables 1, 2, 3, and 4 (ng.g^{-1}-dry weight). As illustrated in Table 1, the maximum $\sum$PAH content of 297.1 ng g^{-1}-dry weight was found at station 9 in the spring season. Based on the results of Table 2, the maximum $\sum$PAH content of 328.28 ng g^{-1}-dry weight was found at station 10 in the summer season. As shown in Table 3, the maximum $\sum$PAH content of 343.63 ng g^{-1}-dry weight was found at station 10 in the autumn season. As shown in Table 4, the maximum $\sum$PAH content of 348.42 ng g^{-1}dry weight was found at station 10 in the spring season. Therefore, the maximum $\sum$PAH content was found at station 9 in the spring season. This station is located near petrochemical industries and refinery areas (near the phases of the South Pars petrochemical plant), with heavy traffic of ships, loaded trains, and urban and industrial wastewater discharge. The maximum levels of total PAH concentrations were measured at station 10 in summer, autumn, and winter, respectively. This subject can be attributed to their location near industrial activities, active oil locations, petrochemical activities, and refined oil products, which lead to high transportation of vehicles such as ships, boats, and petroleum transfer buoys and crossing of oil tankers and human activities.

The average of $\sum$PAH content in each season is estimated to be 244.59 ng g^{-1} for spring (145.7–289.8 ng g^{-1}), 260.11 ng g^{-1} for summer (179.22–328.28 ng g^{-1}), 276.10 ng g^{-1} for autumn (199.62–343.63 ng g^{-1}), and 281.3 for winter (205.85–348.43 ng g^{-1}) (Tables 1, 2, 3, and 4).

Based on ANOVA test (Fig. 2), total concentrations in winter were 246 ng g^{-1}. In autumn and summer, the overall concentrations were between 246 and 296 ng g^{-1}, and total concentrations in spring

Table 1 Concentrations of 16 US-EPA important and priority PAHs in sediment samples (ng g^{-1}-dw) from the PSEEZ of Persian Gulf in spring

Station	NAP	ACL	AC	FLA	ANT	PHE	Flu	PY	BaA	CHR	BbFA	BkFA	BaP	DBahA	IP	BghiP	∑PAHs	LWM/HWM
1	8.89	5.01	3.14	4.15	3.45	40.32	19.82	18.79	19.45	8.71	12.51	11.34	27.34	2.54	5.32	4.13	194.91	0.833
2	7.81	4.91	3.16	3.73	4.98	37.56	21.34	20.65	18.19	9.45	13.24	11.87	28.12	2.91	6.18	4.18	198.28	0.761
3	7.51	6.05	2.15	4.92	4.18	38.19	18.76	17.01	19.23	7.66	13.88	10.24	28.79	3.82	5.47	4.57	192.43	0.811
4	5.8	4.9	1.14	4.53	1.81	22.89	17.23	15.39	16.54	6.52	8.5	8.74	20.2	2.65	4.66	4.2	145.7	0.654
5	9.15	7.11	2.17	5.12	3.42	37.85	17.98	16.49	17.91	7.48	10.35	9.69	25.64	4.51	5.46	3.98	184.31	0.904
6	9.82	6.14	2.16	4.98	3.16	37.25	20.71	20.34	19.31	7.99	11.27	10.46	26.57	4.57	5.19	4.12	194.04	0.811
7	10.23	8.56	2.89	5.25	4.16	38.79	21.63	20.27	19.87	8.34	12.49	11.35	28.96	5.26	6.92	4.87	209.84	0.832
8	12.62	9.11	3.67	5.96	4.57	89.68	19.8	24.42	22.65	8.49	13.06	14.76	29.18	5.94	7.15	3.56	274.62	1.405
9	13.96	9.89	6.57	6.62	4.06	99.51	19.5	25.85	23.38	8.85	14.96	15.99	29.71	6.84	7.85	3.56	297.1	1.498
10	13.46	8.76	5.12	6.29	5.03	94.57	20.86	24.92	23.17	8.86	14.95	14.63	29.81	6.93	8.32	4.12	289.8	1.418
11	13.32	8.98	4.63	6.18	4.26	89.21	20.65	24.89	22.34	8.28	15.11	14.58	28.86	6.71	8.77	4.38	281.15	1.365
12	10.98	7.15	3.29	6.03	4.49	62.11	21.57	20.15	22.07	7.93	14.99	14.47	28.92	6.72	8.96	4.93	244.8	1.04
13	9.43	6.24	2.96	5.11	5.1	54.36	22.3	20.12	21.36	7.59	14.55	13.68	27.89	5.89	8.43	5.21	230.22	0.943
14	8.12	6.32	2.64	4.29	4.65	53.23	23	19.41	20.97	8.04	14.2	13.16	27.65	5.45	7.07	4.92	223.12	0.918
15	8.42	7.2	1.46	4.47	5.28	49.87	23.5	17.18	20.09	7.97	13.6	12.76	27.43	5.11	7.89	5.08	217.31	0.909
16	7.68	6.16	1.62	3.45	6.98	51.36	21.4	17.65	21.12	7.49	13.6	11.31	27.86	5.34	8.06	4.76	215.84	0.929
Mean	9.83	7.03	3.05	5.07	4.35	56.05	20.63	20.22	20.48	8.1	13.2	12.44	27.68	5.07	6.98	4.41	224.59	1.002
SD	2.43	1.6	1.42	0.95	1.13	24.09	1.72	3.26	1.99	0.7	1.85	2.12	2.29	1.48	1.42	0.51	40.138	0.249
TEL	34.57	5.87	6.71	21.2	469	86.7	112.82	153	74.8	108	–	–	88.8	6.22	–	–	–	–
PEL	391	128	88.9	144	254	544	1494	1398	693	846	–	–	763	137	–	–	–	–
ERL	16	44	16	19	85.3	240	600	665	261	384	320	280	430	63.4	–	430	3912.2	
ERM	2100	640	500	540	1100	1500	5100	2500	1600	2800	1880	1620	1600	260	–	1600	25,430	

Table 2 Concentrations of 16 US-EPA important and priority in sediment samples, PAHs (ng g^{-1}-dw) from the PSEEZ of Persian Gulf in summer

Station	NAP	ACL	ACE	FLA	ANT	PHE	Flu	PY	BaA	CHR	BbFA	BkFA	BaP	DBahA	IP	BghiP	∑PAHs	LWM/HWM
1	9.8	6.72	3.8	5.06	6.9	52	21.19	19.28	21	10.06	14.4	14.4	28.43	3.43	5.6	4.27	226.34	0.989
2	9.21	7.01	3.8	5.39	7.2	50.1	24	20.87	20.7	10.52	15.1	14	28.93	3.97	6.58	4.27	231.65	0.926
3	8.56	8	3.25	5.86	6.25	47.2	23.22	18.7	21	10.15	15.9	13.6	30.8	4.57	5.67	4.84	227.57	0.888
4	7.9	6.72	1.92	5.71	4.3	29.3	19.34	15.98	19.3	9.05	10.4	11.1	24.7	3.6	5.46	4.44	179.22	0.755
5	10.2	8.64	3.25	6.16	5	49.1	20.29	18.2	20.5	9.69	12.2	12.2	28	5.05	6.27	4.27	219.02	1.004
6	10.6	8.93	3.84	6.87	5.55	42.5	24.38	21.5	21.6	9.6	12.5	13.4	28.6	5.35	6	4.4	225.62	0.886
7	12.5	9.7	3.84	7.12	5.94	49.1	24.87	21.5	23	9.78	13.9	15.1	29.7	5.95	7.6	5.29	244.89	0.938
8	14.3	9.89	5.7	7.29	5.6	96	21.4	26.3	25.6	10	15	16.6	31	6.68	8.06	3.91	303.33	1.406
9	15.5	10.8	7.36	7.96	5.8	105.8	21.6	27.5	25.8	10	16.6	17.1	30.8	7.46	8.15	3.94	322.17	1.511
10	14.4	10.8	6.66	7.84	5.8	110.6	23.3	27	27	10	16.3	17.1	30.73	7.6	8.69	4.46	328.28	1.511
11	14.8	10.66	5.5	7.32	5	103	22.6	25.6	25.8	9.78	17.3	16.6	31.6	7.28	9.13	4.56	316.53	1.432
12	13.2	10.08	5.38	7.41	5.86	72.8	23.3	22.6	26.3	10.71	18.2	17.6	34.2	7.52	10.3	5.64	291.1	1.084
13	11.3	9.7	4.74	7.42	7.31	68	25.29	22.6	25.3	10.4	19	17	32.3	7.16	9.76	5.44	282.72	1.037
14	9.8	8.26	4	7.4	7.5	64	26.19	20.76	24.9	10.06	17.3	16.5	30	6.8	8	5.29	266.76	1.015
15	10.3	8.4	3.01	7.35	7.78	56	25.3	18.46	23.5	9.32	16	15	28	5.95	8.12	5.44	247.93	0.998
16	8.98	8.16	2.91	7.38	8.4	58.6	24.29	18.5	24	8.95	15.3	14.5	29	5.89	8.75	5.04	248.65	1.021
Mean	11.33	8.9	4.31	6.85	6.26	65.88	23.16	21.58	23.46	9.88	15.34	15.11	29.8	5.89	7.63	4.72	260.1	1.087
SD	2.44	1.39	1.46	0.9	1.14	24.93	1.95	3.48	2.47	0.49	2.28	1.93	2.17	1.43	1.54	0.57	43.21	0.231
TEL	34.57	5.87	6.71	21.2	469	86.7	112.82	153	74.8	108	–	–	88.8	6.22	–	–	–	–
PEL	391	128	88.9	144	254	544	1494	1398	693	846	–	–	763	137	–	–	–	–
ERL	16	44	16	19	85.3	240	600	665	261	384	320	280	430	63.4	–	430	3912.2	
ERM	2100	640	500	540	1100	1500	5100	2500	1600	2800	1880	1620	1600	260	–	1600	25,430	

Table 3 Concentrations of 16 US-EPA important and priority in sediment samples, PAHs (ng g^{-1}-dw) from the PSEEZ of Persian Gulf in autumn

Station	NAP	ACL	AC	FLA	ANT	PHE	Flu	PY	BaA	CHR	BbFA	BkFA	BaP	DBahA	IP	BghiP	∑PAHs	LWM/HWM
1	9.5	6.95	4.1	5.49	8.11	53.22	22.51	20.71	22.46	11.52	15.82	16.42	29.22	4.12	6.18	4.86	241.19	0.947
2	9.45	6.82	4.22	5.42	8.41	51.31	25.31	22.3	22.19	11.98	16.53	15.7	29.54	4.56	7.24	4.72	245.7	0.892
3	9.32	7.46	4.64	5.84	7.46	48.4	24.53	20.13	22.47	11.61	17.34	15.32	31.39	5.32	6.26	5.43	242.92	0.867
4	9.85	8.51	4.95	5.62	5.51	30.52	20.65	17.41	20.75	10.51	11.85	12.74	25.25	4.21	6.05	5.24	199.62	0.804
5	10.34	8.92	5.24	6.23	6.21	50.33	21.6	19.63	21.92	11.15	13.68	13.96	28.58	5.64	6.86	4.86	235.15	0.984
6	10.62	9.49	5.62	6.58	6.76	43.72	25.69	22.93	23.09	11.06	13.99	15.13	29.29	5.94	6.59	4.93	241.43	0.87
7	11.98	9.97	6.16	6.95	7.15	50.34	26.18	22.93	24.43	11.24	15.37	16.82	30.25	6.54	8.21	5.88	260.4	0.919
8	13.45	10.15	6.79	7.91	6.81	97.25	22.71	27.73	27.07	11.46	16.49	18.43	31.53	7.27	8.65	4.5	318.2	1.349
9	15.1	10.48	7.13	8.24	7.01	107.01	22.91	28.93	27.26	12.48	18.14	18.87	31.37	8.05	8.74	4.53	336.25	1.425
10	14.51	10.56	7.27	8.16	7.01	111.82	24.61	28.43	28.42	12.42	17.72	18.85	31.33	8.19	9.28	5.05	343.63	1.441
11	14.2	10.06	7.31	8.21	6.21	104.23	23.91	27.03	27.27	12.24	18.73	18.35	32.19	7.87	9.73	5.15	332.69	1.372
12	13.21	9.13	6.87	8.2	7.07	74.03	24.61	24.03	27.71	12.17	19.61	19.37	34.75	8.11	10.87	6.25	305.9	1.054
13	12.4	8.54	6.27	8.06	8.52	69.24	26.6	24.03	26.74	11.83	20.4	18.75	32.84	7.75	10.35	6.03	298.35	1.017
14	12.13	8.38	5.46	8.02	8.71	65.22	27.5	22.19	26.33	11.52	18.78	18.23	30.55	7.39	8.55	5.88	284.84	1.017
15	11.87	8.2	5.1	7.92	8.99	57.23	26.61	19.89	24.96	10.78	17.46	16.43	28.59	6.54	8.71	6.31	265.59	0.995
16	11.14	8.17	4.43	7.94	9.61	59.82	25.6	19.93	25.21	10.41	16.79	16.38	29.37	6.28	9.13	5.51	265.72	1.024
Mean	11.82	8.86	5.72	7.17	7.47	67.11	24.47	23.01	24.89	11.52	16.79	16.86	30.38	6.49	8.21	5.32	276.1	1.06
SD	1.9	1.2	1.12	1.12	1.14	24.93	1.95	3.48	2.46	0.64	2.27	1.95	2.16	1.41	1.52	0.6	39.88	0.19
TEL	34.57	5.87	6.71	21.2	469	86.7	112.82	153	74.8	108	–	–	88.8	6.22	–	–	–	–
PEL	391	128	88.9	144	254	544	1494	1398	693	846	–	–	763	137	–	–	–	–
ERL	16	44	16	19	85.3	240	600	665	261	384	320	280	430	63.4	–	430	3912.2	
ERM	2100	640	500	540	1100	1500	5100	2500	1600	2800	1880	1620	1600	260	–	1600	25,430	

Table 4 Concentrations of 16 US-EPA important and priority in sediment samples, PAHs (ng g^{-1}-dw) from the PSEEZ of Persian Gulf in winter

Station	NAP	ACL	AC	FLA	ANT	PHE	Flu	PY	BaA	CHR	BbFA	BkFA	BaP	DBahA	IP	BghiP	∑PAHs	LWM/HWM
1	5.45	6.77	4.71	29.81	9.62	9.62	7.12	4.2	6.7	54.43	8.41	22.62	20.81	22.54	11.55	16.73	241.09	0.628
2	5.31	7.83	5.15	30.13	9.3	16.2	7.34	4.3	6.63	52.52	8.51	25.43	22.4	22.27	12.01	17.44	252.18	0.689
3	6.02	6.85	6.85	31.98	9.45	15.8	7.52	4.8	7.05	49.61	7.56	24.65	20.23	22.59	11.64	18.25	250.85	0.737
4	5.83	6.64	4.8	25.84	9.46	13.3	7.95	4.8	6.83	31.73	5.61	20.77	17.51	20.85	10.54	12.76	205.22	0.788
5	5.45	7.45	6.23	29.17	10.58	14.44	8.46	5.45	7.44	51.54	6.31	21.72	19.73	22.03	11.18	14.59	241.77	0.725
6	5.52	7.18	6.53	29.88	11.12	15.67	8.92	5.62	7.79	44.93	6.86	25.81	23.3	23.17	11.09	14.6	247.99	0.735
7	6.47	8.8	7.13	30.84	11.94	17.53	9.37	6.47	8.16	51.55	7.25	26.3	23.3	24.58	11.27	16.28	267.24	0.747
8	5.28	9.24	7.86	32.12	12.38	18.8	9.93	7.12	9.12	98.46	6.91	22.83	27.83	27.16	11.49	17.4	323.93	0.599
9	5.19	9.33	8.64	31.96	12.97	19.33	10.43	7.64	9.45	108.22	7.11	23.03	29.03	27.35	11.49	19.05	340.22	0.576
10	5.64	9.87	8.78	32.92	13.5	19.32	10.34	7.61	9.37	113.03	7.11	24.73	28.53	28.55	11.49	18.63	348.42	0.572
11	5.74	10.32	8.46	32.78	13.11	18.79	10.32	7.59	9.42	105.44	6.31	24.03	27.13	27.38	11.27	19.64	337.73	0.598
12	6.86	11.46	8.7	34.34	12.41	19.8	10.33	7.15	9.41	75.24	7.17	24.13	24.73	27.13	12.2	20.52	312.6	0.723
13	6.86	10.94	8.34	33.43	12.24	19.2	10.08	6.19	9.23	66.43	8.81	17.6	24.13	26.89	11.89	21.31	293.57	0.749
14	6.47	9.14	7.98	31.14	11.71	18.75	8.54	6.19	9.23	66.43	8.81	27.62	22.2	26.45	11.55	19.69	291.9	0.687
15	6.76	9.3	7.13	29.18	11.24	17.2	8.14	6.05	9.13	58.44	9.09	26.73	19.99	25.18	10.81	18.37	272.74	0.702
16	5.82	9.52	6.67	29.76	11.09	16.74	7.98	6.12	9.15	61.03	9.71	25.72	20.03	25.46	10.44	17.7	272.94	0.686
Mean	5.92	8.79	7.12	30.96	11.38	16.91	8.92	6.08	8.38	68.06	7.58	23.98	23.18	24.97	11.37	17.69	–	–
SD	0.58	1.52	1.38	2.16	1.39	2.73	1.19	1.16	1.12	24.93	1.15	2.53	3.5	2.42	0.49	2.29	–	–
TEL	34.57	5.87	6.71	21.2	469	86.7	112.82	153	74.8	108	–	–	88.8	6.22	–	–	–	–
PEL	391	128	88.9	144	254	544	1494	1398	693	846	–	–	763	137	–	–	–	–
ERL	16	44	16	19	85.3	240	600	665	261	384	320	280	430	63.4	–	430	3912.2	
ERM	2100	640	500	540	1100	1500	5100	2500	1600	2800	1880	1620	1600	260	–	1600	25,430	

were 296 ng g^{-1}. Therefore, the total concentrations in winter were higher compared with other seasons. Significant differences in $\sum$PAH content were observed between winter, summer, and spring (per ANOVA tests) (Fig. 2). In winter, the samples presented the highest PAH concentrations (Fig. 2). As a result, the risk of PAHs on the coast of South Pars in winter is higher than in other seasons. These studies suggest that the winter concentration of contaminants would be higher due to increased combustion due to fuels as a result of domestic and industrial activities.

The results of Pearson correlation coefficient between total of sediments PAH concentrations in spring was (sig = 1); therefore, there is no significant relationship between sediment concentrations in spring. As given in Table 5, Pearson correlation coefficient between total of sediments PAH concentrations in summer was (sig = 0.987**); thus, there is a significant relationship between sediment concentrations in spring at the level of 0.001. Furthermore, the Pearson correlation coefficient in autumn was (sig = 0.986**), and in winter, it was (sig = 0.987**). Thus, there is a significant relationship between sediment concentrations in autumn and winter at the level of 0.001. Investigating every season exclusively, no significant correlation was found in all seasons. On the other hand, the $\sum$PAH values were not correlated with sediment's total organic carbon concentration across all seasons ($r = 0.230$; $P < 0.05$). However, analyzing the correlations among total PAH concentrations in different seasons showed positive results in each season ($P < 0.05$) (Table 5).

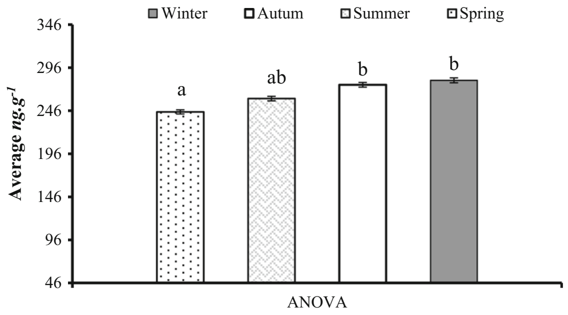

Fig. 2 Comparison of cumulative sediment PAH concentrations in various seasons using ANOVA studies

Table 5 Pearson correlation coefficient between PAH concentrations of total sediments in various seasons

	Spring	Summer	Autumn	Winter
Spring	1			
Summer	0.987^{**}	1		
Autumn	0.986^{**}	0.999^{**}	1	
Winter	0.987^{**}	0.996^{**}	0.997^{**}	1

** Correlation is significant at level 0.01 (two-tailed)

PAH composition and source identification

The isomeric ratios Ant/(Ant + Phe), BaA/(BaA + Chry), IP/(IP + BghiP), and Flua/(Flua + Pyr) were calculated as diagnostic indicators to identify the potential PAH origin. For instance, the common ratio FLA/(FLA + PYR) < 0.4 typically (indicates predominantly petrogenic sources) defines petroleum contamination; a ratio between 0.40 and 0.50 (indicates predominantly petrogenic sources) shows petroleum fuel combustion; a ratio > 0.50 indicates coal or biomass combustion (combustion origin of biomass) (e.g., coal, wood, grass); a BaA/(BaA + Chr) ratio < 0.20 is commonly taken as an indication of a predominant petroleum origin; a ratio between 0.20 and 0.35 indicates either combustion or petroleum; and $N > 0.35$ shows only combustion (Katsoyiannis & Breivik, 2014; Tobiszewski & Namieśnik, 2012). The ratio An/(An + Phe) < 0.10 demonstrates the predominance of the petroleum source, and $N > 0.10$ shows the dominance of combustion (Baumard et al. 1999).

Based on results, a precise reconstruction of the biogeochemical characteristics of pollutants at various periods and spatial scales is needed to understand the real influence of polluted sediments on the aquatic environment and their possible effects on the ecosystem. Moreover, a stagnant understanding of the effect of contaminated sediments on the coastal marine environment is questioned by normal and personal interactions between land-to-coast and coast-to-deep-sea systems. In addition, approaches to monitoring aquatic sediments that have historically been limited to unique and restricted areas (where pollutants simply tend to be more abundant in surface-to-surface sediments) hinder an accurate understanding of their effects on the marine ecosystem. Indeed for the aquatic ecology and the atmosphere, polluted sediments must be treated as dynamic time- and space-modulated active sources of contaminants. For holistic and appropriate management of contaminated aquatic sediments, a current view must take into account the chemical and physical behavior and the complex dynamics of pollutants in various ecosystems. The authors therefore suggest and propose that a larger buffer zone be established as an external and systemic aspect of the surveillance action committed to the heavily contaminated marine coastal regions.

Discussion

The concentrations of the 16 PAHs in samples of surface sediments from various stations in different seasons are represented in Tables 1, 2, 3, and 4. The $\sum$PAH concentration ranged from 145.7 to 348.42 ng^{-1}dw, and the mean concentration is 260.52 ng g^{-1}. The maximum levels of PAH content of 348.42 ng g^{-1} – dw were found at station 10. This station is located near the first and second phases of the petrochemical industrial zone, which had the highest industrial activity and was the major discharging area for the petrochemical industrial influent stream to the coastal region. In addition, this zone was affected by the effluent discharge from ballast water due to ship traffic, petroleum transfer, etc. Specifically, this station demonstrated the highest range of $\sum$PAHs in all seasons. In addition, this zone was affected by the effluent discharge from ballast water due to ship traffic, petroleum transfer, etc. Specifically, this station demonstrated the highest range of $\sum$PAHs in all seasons. CHR was detected to have the highest concentration (113.03 ng g^{-1}) in the winter season, followed by Phe, FL, and Bap, with concentrations of 111.82, 34.75, and 34.34 ng g^{-1}, in autumn and winter, respectively. In addition, sediment properties play a key role in the distribution of PAHs. Grain size is modified for different seasons and this is crucial for PAHs fixation. Under all conditions, i.e., seasons, time, temperature, and pH, the distribution of the slopes of the combined leaching curves is larger for some metals than for others. In addition to the environmental driving factors, it can be postulated that the release activities of any metal are contingent on the metal/sediment mix. The aggregates are then dissolved and the framework in an extremely acidic environment is broken. Free and soluble metal compounds and the difference in concentration between the sample matrix and the leachate induce the gradual degradation of metals in the washing liquids in the early phases under acidic and alkaline conditions. Diffusion (0.35 < slope ?? 0.65) and depletion (slope ?? 0.35) are the prevailing processes of the metals over the entire leaching cycle under acidic and alkaline conditions.

This outcome is in line with the results of Vijayan et al. (2019). Their findings showed that laboratory snowmelt experiments with actual snow samples in late winter, just before the final snowmelt, in two similar cities in northern Sweden, were related to CHR with the highest concentrations. The $\sum$PAH concentrations in sediments can be grouped into four categories: $\sum$PAHs < 100 ng g^{-1} indicated low contamination, 100–1000 ng g^{-1}showed moderate pollution, 1000–5000 ng g^{-1} represented heavy pollution, and $\sum$PAHs > 5000 ng g^{-1} referred to very heavy pollution (Chen et al., 2013; Neşer et al., 2012). Therefore, according to the amount of $\sum$PAH concentration, the sediments in the PSEEZ area indicated moderate to heavy pollution. Tables 2 and 5 show that all stations were moderately polluted. Therefore, the sedimentary surface ecosystems of the Persian Gulf were considered as moderately polluted, compared with other ecosystems worldwide (Table 6). The higher concentrations at stations 8, 9, 10, and 11 can be explained by different input sources in winter, such as discharge of wastewater and runoffs, petrochemical industrial effluent, and oil spills from different vehicles like boats and ships. This is because these sites are located in a more industrialized and urbanized region and near the Asaluyeh phases. In addition, there were lots of ships and boats traveling in this area, which might contribute to the higher amount of PAH concentrations. In fact, the PAHs might be caused by the local sources of petroleum hydrocarbons, such as ferries and fishing boats. These results are consistent with the findings of Gune et al. (2019), Sarti et al. (2017), and Liu et al., (2012*)*.Urban wastewater and runoff also tend to have different energies during the seasons, resulting in significant changes in grain size, organic matter, and mineralogy. Therefore, these factors are also very important in creating seasonal changes and assessing the aromatic polycyclic hydrocarbons in the aquatic resource center's coastal sediments. In order to forecast PAH

Table 6 Comparison of the sum of 16 PAH concentrations in sediment samples with other aquatic ecosystems worldwide

Case study	Number of PAHs studied	PAH concentrations (ng g^{-1})	Pollution level	References
PSEEZ, Iran	16	145.7–348.43	Moderate	This study
Cochin estuary, India	16	194–14,149	Moderate–heavy pollution	(1)
Intertidal zone of Bohai Bay, China	16	37.2–206.6	Low–moderate	(2)
Mediterranean coastal zone, Spain	14	5.3–2627.4	Low–heavy	(3)
Liaodong Bay, China	16	144.5–291.7	Moderate	(4)
Western Taiwan coastal plain	16	2.41−218.54	Low–moderate	(5)

emissions, knowledge of the PAHs partitioning activities and their differences are critically significant in gaseous and particulate phases and in size-segregated phases. Previous studies have shown that the partitioning of PAHs depends primarily on their volatility between the gaseous and particulate phases and that the particulate partitioning of PAHs depends to a large degree on the scale, the unburned carbon content, and the species of metal particulates.

The average concentration of PAH for spring (145.7–289.8 ng g^{-1}) was calculated to be 244.59 ng g^{-1}; for summer, 260.11 ng g^{-1} (179.22–328.28 ng g^{-1}); for autumn, 276.10 ng g^{-1} (199.62–343.63 ng g^{-1}); and for winter, 281.3 (205.85–348.43 ng.g^{-1}) (Figs. 1, 2, 3, and 4). Total concentrations in winter were higher compared with other seasons, followed by fall, and the lowest levels of $\sum$16 PAHs occurred in spring (Tables 1, 2, 3, and 4). The highest $\sum$PAH concentration was observed in winter (348.42 ng g^{-1}) because of the increase in fuel usage and the high rate of precipitation in winter. Furthermore, enhanced photodegradation in summer is proposed as a probable explanation for higher PAHs in winter (Dimashki et al., 2001; Kim and Chae, 2016). It could be explained as a result of the intensified use of oil products and could carry the petroleum and its derivatives by urban runoff and rivers, industrial and municipal effluent discharge, and atmospheric deposition in winter (Guo et al., 2007; Keshavarzifard et al., 2014). Kim and Chae (2016) indicated that PAHs have a negative temperature dependency and that temperature and atmospheric PAHs have a negative relationship. In addition, at a warm period of time, photolysis, volatilization, and the lower PAH number can be explained by algal blooming, along with sorption into suspended particle matter. Sorption of gases from air to water, accompanied by elevated levels of PAHs in the air, may be another cause for increasing levels of PAH in seawater during cold times. In other beach habitats such as the Khambhat Gulf, India (Reddy et al., 2005) and European coastal mussel waters, higher concentrations of PAH were also found in winter relative to warm periods of time (Olenycz et al., 2015). The maximum PAHs were found in the monsoon by Ramzi et al. (2017), Szopińska et al. (2019), and Zhang, Wang, et al. (2020), Zhang, Pan, et al. (2020)) (Ramzi et al., 2017). For all seasons, the highest amounts of $\sum$PAH concentrations were, respectively, observed at stations 10, 9, and 11. These stations were the major discharging zones for the oil and petroleum industry wastewaters outflowing to coastal areas. Among all tested PAHs in sediments, PHE and BaP were the predominant species of all stations. BaP is considered as an indicator for combustion-originated PAHs and intensely supports the predominantly incomplete combustion derived (pyrogenic) from PAH in sediments (Magi et al., 2002; Neira et al., 2017). The results of the parametric Pearson correlation analysis showed no significant relationship between TOC and PAHs ($P > 0.05$, $n = 16$), and this was in compliance with the literature where the TOC content of sediments had no correlation with PAH concentration as a main controlling factor (Barhoumi et al., 2014; Chen et al., 2013; Sany et al., 2014).

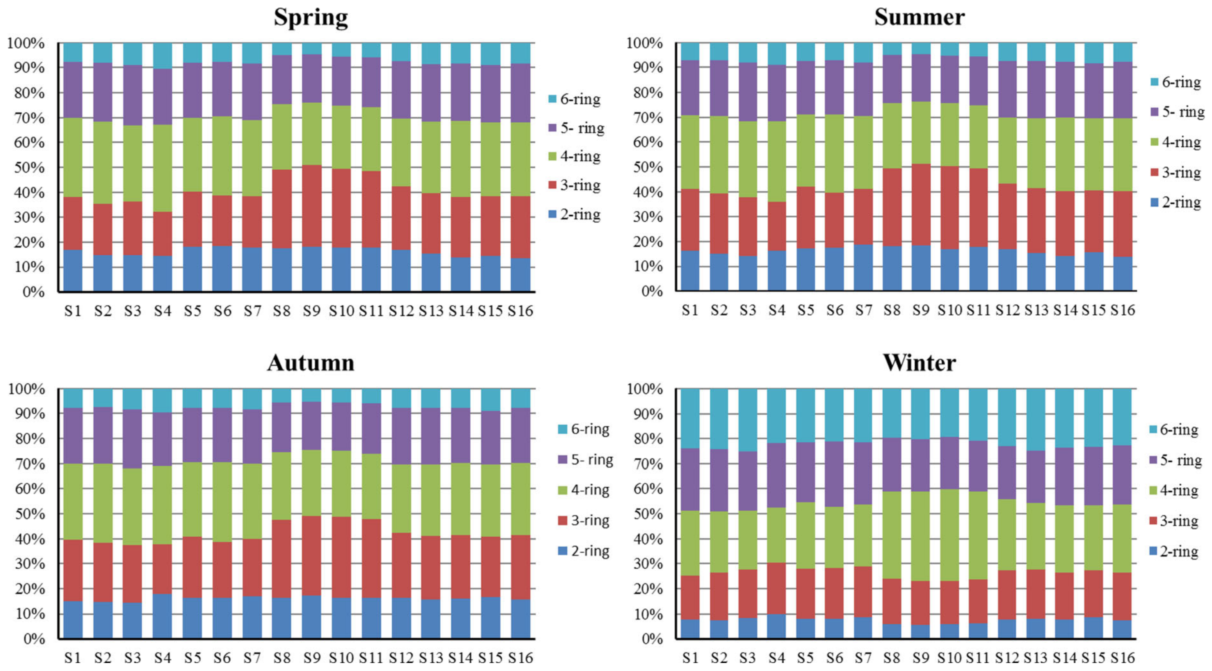

Fig. 3 Distribution of 2–6-ring ∑16PAH compounds in sediments from the PSEEZ (Persian Gulf)

Seasonal deviations in PAH composition

In spring, the four-ring PAHs were the most prevailing compounds, with an average value. The 2- and 3-ring PAHs were the second dominant compounds (Fig. 2). The 4-ring PAHs were the most prevailing compounds in all seasons (Fig. 2). The principal compound in the winter was inhabited by medium molecular weight (MMW, 4-ring) PAHs. The compounds of the 5-ring PAHs became the second dominant PAHs. The key compound was occupied in autumn by the 4-ring PAHs.

The four-ring PAHs dominated the primary compound of PAHs throughout the autumn. The second prevalent PAH compounds were the 3-ring PAHs. In conclusion, the four-ring PAHs in both seasons were the most prevalent PAH compounds. These results are consistent with the findings of Khaksar et al. (2019), and Škrbić et al. (2019).

PAH source diagnosis

The PAHs of the 2–3 (low molecular weight)- and 4–6 (high molecular weight)-ring diagnostic ratio fractions can be applied as a marker to determine the PAH origins; LMW/HMW ratio >1 is usually considered as a petrogenic origin, and a ratio <1 represents a pyrolytic origination (Wang et al., 2011). Petrogenic mechanism is responsible for PAH pollution distinguished by 2–3-ring PAHs, while pyrogenic mechanism is responsible for PAH contaminations distinguished by 4–6-ring PAHs (Baumard et al., 1998; Ramzi et al., 2017). The number of PAHs ratio in the present analysis (2–3 rings/4–6 rings) in different seasons in stations 8, 9, 10, 11, and 13 was higher than one, indicating the petrogenic origin of pollution. On the other hand, in stations 2 and 3, 4–6-ring PAHs (LMW/HMW) ratios were lower than one (values were between 0.78 and 0.99), showing a pyrolytic origin. These results confirmed the petrogenic source of pollution for most stations. The existence of 2 and 3 rings in PAHs represents that the source of PAHs is from shipping discharges and oil spills. Since some sediment samples contained LMW PAHs, engine missions from port transportation and vehicles exhausts were also a source of contamination. Among all tested PAHs in sediments, PHE and BaP were the predominant species in all stations. Moreover, BaP can be used as an indicator for combustion-originated PAHs because the value of BaP is

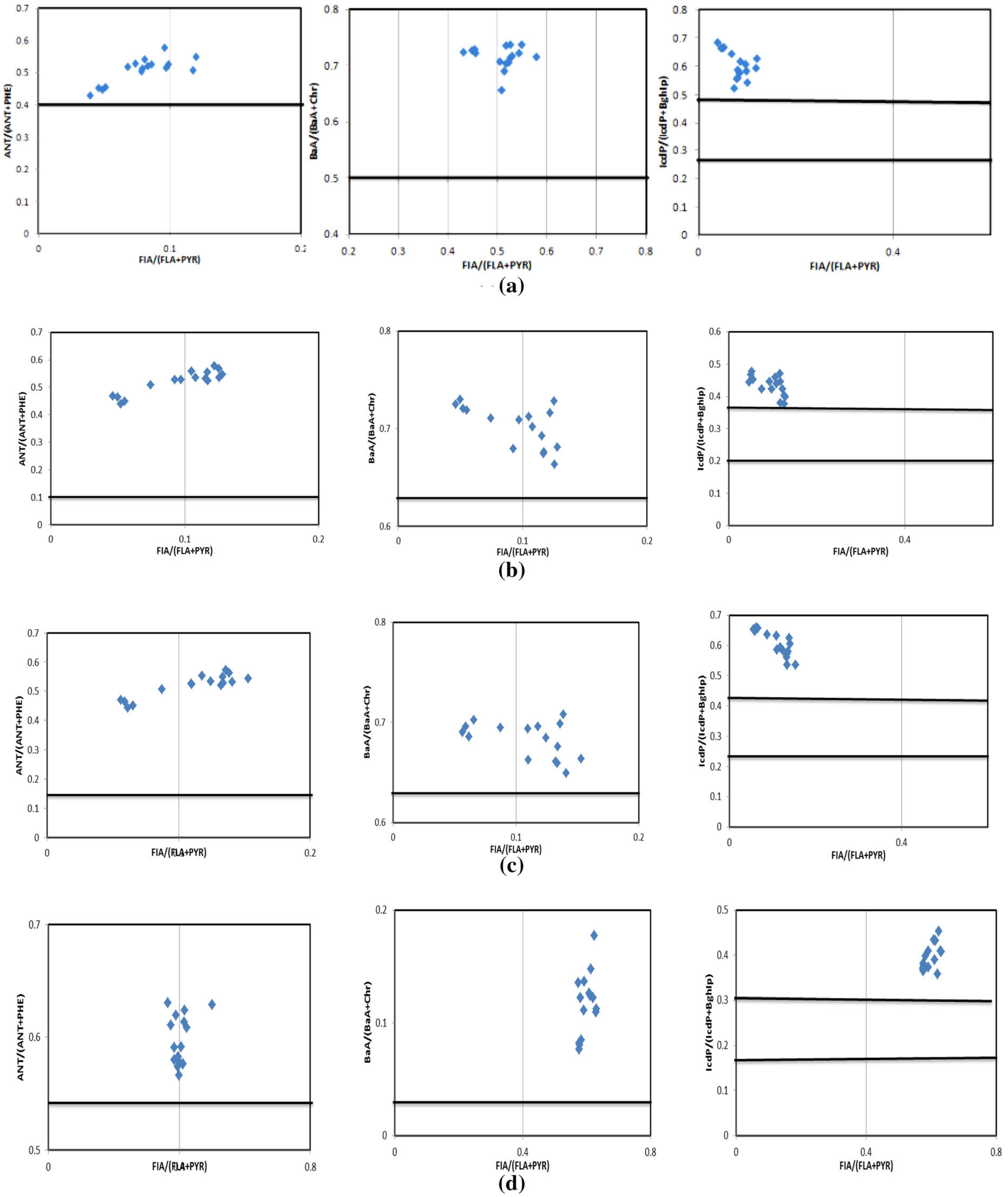

Fig. 4 Scattering plots of PAH diagnostic ratios, FLA/(FLA + PYR) versus. (BaA + CHR), ANT/(ANT + PHE), and (IcdP + BghiP), for (**a**) spring, (**b**) summer, (**c**) autumn, and (**d**) winter

approximately insignificant in petroleum (Magi et al., 2002). Engine vehicles and their exhaust systems were considered as the main origin of potential carcinogenic PAHs, including BaA, BbkF, and BaP, exclusively in urban zones (Neira et al., 2017; Jakovljević et al., 2018; Ciarkowska et al., 2019; Siemering and Thibodeaux, 2020).

In order to realize the transport and fate of PAHs in the ecosystem, the assessment of the dominant origin of them is important. Thus, diagnostic ratio values of various groups of PAH contents have been widely calculated for identifying the potential origins of PAHs. Some diagnostic ratios, such as Ant/Ant + Phen, Flua/Flua + Pyr, IP/qIP + BghiP, and BaA/(BaA + Chry), were widely chosen as an accepted index to distinguish between PAHs from different sources (such as pyrolytic and petrogenic origins). Pyrogenic compounds are typically a class of composition resulting from fossil fuel, and petrogenic combustion compounds are the product of petroleum operations. Higher molecular weight PAHs (4–6 benzene rings) are compounds from pyrogenic backgrounds, whereas those from petrogenic sources have an LMW of 2 or 3 rings of benzene (Neff, 1979; Zakaria et al., 2002). It is proved that the prevalent source is petrogenic when the Ant/Ant + Phen diagnostic ratio is less than 0.1, whereas a higher ratio (> 0.1) suggests that the root of PAHs is pyrolytic. Based on the Fluo/(Fluo + Pyr), the values lower than 0.4 suggest a petrogenic source of contamination of PAHs, and those between 0.4 and 0.5 suggest petroleum combustion sources, whereas ratios higher than 0.5 suggest that the contamination is caused by grass, wood, and coal burning. BaA/BaA + Chry values less than 2 demonstrate petrogenic sources; values between 2 and 0.35 show either a petroleum or combustion origin. Finally, (BaA/BaA + Chry) values higher than 0.35 imply a pyrogenic origin (Li et al., 2016; Yunker et al., 2002). Meanwhile, for IP/(IP + BghiP), the ratios less than 0.2 and 0.4 imply that the main source is petroleum, and if the ratios are both over 0.5, coal, wood, and grass combustion origins are suggested. Other internal ratios are seemed to correspond to petroleum combustion (particularly crude oil, vehicle, and liquid fossil fuel) (Yunker et al., 2002).

In order to assess the source of PAHs in all seasons, these distinguished ratios as indicators of PAH compound’s origin were plotted in Fig. 2. In winter, diagnostic values of Ant/(Ant + Phe) versus Flu/(Flu + Pye) implied that the ratios of Flu/(Flu + Pye) were 0.56–0.63, all of which are higher than 0.4; the proportions of Ant/(Ant + Phe) is greater than 0.1, which means that pyrogenic sources tainted the sediment samples. The values of BaA/(BaA + CHR) in sediment samples ranged from 0.07 to 0.17, all of which are lower than 0.4, demonstrating that the coastal sediments were contaminated by petroleum input. The values of IP/(IP + BghiP) were 0.35–0.45 (between 0.4 and 0.5), suggesting that the origins of PAHs in sediments can be associated with pyrogenic and petrogenic sources. Considering the combined diagnostic ratios in other seasons, including spring, summer, and autumn, the PAHs were mainly originated alike (Fig. 2). In autumn, the distinguish ratios of Ant/(Ant + Phe) were 0.056–0.15, and the Ant/Ant + Phen levels were greater than 0.1 at stations 1, 2, 3, 4, 6, 7, 13, 14, 15, and 16, suggesting ignition origins at these sites. The ratios of Flu/(Flu + Pyr) in sediment samples were between 0.44 and 0.57, and most varied between 0.40 and 0.50. These findings revealed that both pyrogenic and petrogenic origins were correlated with the primary source of PAHs. The values of IP/(IP + BghiP) and BaA/(BaA + CHR) were 0.53–0.66 and 0.44–0.5, respectively, both greater than 0.5 and 0355 (Fig. 3). These data suggest that PAHs are produced from the combustion of grass, petroleum, tar, and wood in coastal sediments. In summer, the quantity of Ant/(Ant + Phe) was 0.04–0.13, denoting that the PAH compositions in this season were associated with pyrogenic and petrogenic sources. The quantities of Flu/(Flu + Pyr) and BaA/(BaA + CHR) were 0.44–57 and 0.67–0.72, respectively, denoting that the PAH compositions in these stations were mainly derived from incomplete burning of organic matter sources (pyrogenic). Finally, the quantity of IP/(IP + BghiP) was 0.37–0.47, all of which were lower than 0.5, which indicated that the coastal sediments were contaminated by petroleum input. However, in spring, the quantity of Ant/(Ant + Phe) was 0.04–0.12, indicating that the PAH compositions in this season were derived from pyrogenic and petrogenic sources. The quantity of Fluresul/(Flu + Pyr) was 0.43–57, and the values of BaA/(BaA + CHR) were 0.65–0.74, indicating that in these stations, the PAH compositions were primarily extracted from incomplete burning of organic matter sources (pyrogenic). Finally, the distinguished ratios of IP/(IP + BghiP) were 0.52–0.68, all of which were higher than 0.5, which indicated that the coastal sediments were polluted by pyrogenic input.

PAH risk assessment

The rate of threshold effects, medium impact range, and median effect range are efficient methods for assessing the ecological risk effects of inorganic and organic pollution in aquatic sediments (CCME, Long et al., 1995; Macdonald et al., 1996; 2002). SQGs are clarified by comparing the multiple sediment toxicity responses of aquatic species with the sediment content of PAHs considered. Thus, it is recommended as an efficient method for assessment of risk levels and protection of water ecosystems (Hu et al., 2011; Long et al., 1995). Sediment quality guidelines (SQGs) are classified into three categories according to PAH concentrations ranges. The values of PAH contaminants can be less than ERL and TEL (biological effects are rare), between ERM/PEL and ERL/TEL ratios (occasional), and finally, greater than ERM and PEL (representing that biological effects are expected frequently). The $\sum$PAH concentrations in all seasons were importantly less than ERL and ERM quantities (Tables 2–5). These results indicated that biological effects would seldom happen. These results were in agreement with other studies such as those by Manneh et al. (2016) and Kucuksezgin et al., 2013) (Kucuksezgin et al., 2013; Manneh et al., 2016; Wang et al., 2017). According to our findings, in spring, the amount of PHE (Wu et al., 2020) at stations 8, 9, 10, and 11; the amount of Acl at stations 3 and 5–16; and the amount of DBahA at stations 9, 10, 11, and 12 were more than TEL, but less than PEL, ERL, and ERM. In summer, the concentrations of PHE at stations 8, 9, 10, and 11; Acl at all sixteen stations; Ace at station 9; and DBahA at stations 8 to 14 were lower than PEL, ERL, and ERM, but greater than TEL. In autumn, the concentrations of PHE at stations 8, 9, 10, and 11; Acl at all sixteen stations; Ace at stations 8, 9, 10, 11, and 12; and DBahA at stations 7–16 were less than PEL, ERL, and ERM, but greater than TEL. In winter, the concentrations of AC at stations 3 and 7–16, as well as those of FLA, ACL, and DBahA at all sixteen stations, were greater than TEL, but less than PEL, ERL, and ERM. This indicated that adverse toxicological effects resulting from these compounds might occasionally happen in each season, and potential risks were relatively low. These stations were located near the first and second phases of the petrochemical industrial zone, which had a large number of industrial activities, and were the main zones for depletion of petrochemical industrial wastewater effluent into the coastal areas. In addition, this zone was affected by the effluent depletion of ballast water due to the heavy traffic of national and foreign ships.

Conclusion

One of the world's strategic regions is the Persian Gulf. In the countries that lay on the Persian Gulf coast, about 60% of the world's oil reserves are located. In addition, in this area, about 25 percent of the world's total soil is created. During the war between Iran and Iraq and the war between Iraq and Kuwait, in the Persian Gulf, roughly 2–4 and 6–8 million barrels of crude oil were poured, respectively, and a large portion of the coastline in this area was severely polluted.

Dramatic industrial development and population growth have been triggered by the abundance in the Persian Gulf and its coastal areas, along with rich capital, resulting in intensive emissions of toxins into marine environments. In addition, through residential and industrial towns in the area, the rivers of Karun, Zohreh, and other large rivers snake and bring industrial, agricultural, and domestic waste to the Persian Gulf. The site of the Pars Special Energy Economic Zone is Asaluyeh, on the northern side of the Persian Gulf (APSEZ). Therefore, in this study, seasonal variations, source identification, and potential ecological assessment of sixteen PAH compounds were investigated based on a quantitative investigation of $\sum$16PAH compounds in intertidal sediments in PSEEZ, Iran. Qualitative and quantitative evaluations found that PAHs with mild pyrogenic superiority came from both petrogenic and pyrogenic origins. Source distribution using key component analysis found that the key causes of PAHs have been fossil fuel combustion, traffic-related PAHs, biomass and coal combustion, and petrogenic PAHs. Hence, it seems the consumption of energy is the main reason for PAH contamination in the intertidal zone of Asaluyeh. Intertidal environments play an important role in the earth's biogeochemical processes. In order to prevent/reduce the detrimental health impacts on the environment and public health, more controls and changes can also be made in the intertidal ecosystem of Asaluyeh. This study has in fact investigated the adverse effects on water bioavailability. Therefore, this study will provide a scientific background, planning, and policies for PAH pollution control and environmental protection in Iran and similar regions around the world. Finally, in order to support the understanding of seasonal and spatial distribution of PAHs in sediments, physical, chemical and mineralogical properties of the sediments should be investigated in future studies. It is also proposed that future studies should rely on the use of statistical methods in addition to the Pearson correlations of principal component analysis (PCA)/factor analysis (FA). This approach can support some novel and relevant conclusions regarding sediment complex matrix and geochemistry of PAHs.

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Availability of data and materials All data generated or analyzed during this study are included in this published article. Also, the datasets used and/or analyzed during the current study are available from the corresponding author on reasonable request.

Declaration

Conflict of interest The authors declare no conflict of interest.

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References

Abatemarco, D., Perera, S., Bao, S. H., et al. (2018). Training augmented intelligent capabilities for pharmacovigilance: Applying deep-learning approaches to individual case safety report processing. *Pharmaceutical Medicine, 32,* 391–401.

Abbasi, S., Keshavarzi, B., Moore, F., et al. (2019). Geochemistry and environmental effects of potentially toxic elements, polycyclic aromatic hydrocarbons and microplastics in coastal sediments of the Persian Gulf. *Environmental Earth Science, 78,* 492. https://doi.org/10.1007/s12665-019-8420-z

Abdel-Shafy, H. I., & Mansour, M. S. (2016). A review on polycyclic aromatic hydrocarbons: Source, environmental impact, effect on human health and remediation. *Egyptian Journal of Petroleum, 25*(1), 107–123. https://doi.org/10.1016/j.ejpe.2015.03.011

Abdollahi, S., Raoufi, Z., Faghiri, I., Savari, A., Nikpour, Y., & Mansouri, A. (2013). Contamination levels and spatial distributions of heavy metals and PAHs in surface sediment of Imam Khomeini Port, Persian Gulf. *Iran. Marine Pollution Bulletin, 71*(1–2), 336–345. https://doi.org/10.1016/j.marpolbul.2013.01.025

Abeysirigunawardena, D., Jeffries, M., Morley, M. G., Bui, A. O., & Hoeberechts, M. (2015). Data quality control and quality assurance practices for Ocean Networks Canada observatories. In OCEANS 2015-MTS/IEEE Washington (pp. 1–8). IEEE.

Adeniji, A. O., Okoh, O. O., & Okoh, A. I. (2018). Analytical methods for polycyclic aromatic hydrocarbons and their global trend of distribution in water and sediment: a review. Recent insights in petroleum science and engineering, 10.

Adeyeye, S. A. Q. (2020). Polycyclic aromatic hydrocarbons in foods a critical review. *Current Nutrition and Food Science, 16*(6), 866–873.

Agah, H., Elskens, M., Fatemi, S. M. R., Owfi, F., Baeyens, W., & Leermakers, M. (2009). Mercury speciation in the Persian Gulf sediments. *Environmental Monitoring and Assessment, 157*(1–4), 363–373. https://doi.org/10.1007/s10661-008-0541-x

Akhbarizadeh, R., Moore, F., & Keshavarzi, B. (2019). Polycyclic aromatic hydrocarbons and potentially toxic elements in seafood from the Persian gulf Presence, trophic transfer, and chronic intake risk assessment. *Environmental Geochemistry and Health, 41,* 2803–2820.

Amos-Tautua, B. M. W., Inengite, A. K., Abasi, C. Y., & Amirize, G. C. (2013). Evaluation of polycyclic aromatic hydrocarbons and some heavy metals in roasted food snacks in Amassoma. *Niger Delta Nigeria, 7*(10), 961–966. https://doi.org/10.5897/AJEST2013.1545

Baklanov, A., Hänninen, O., Slørdal, L. H., Kukkonen, J., Bjergene, N., Fay, B., Finardi, S., Hoe, S. C., Jantunen, M., Karppinen, A., & Rasmussen, A. (2007). Integrated systems for forecasting urban meteorology. Air Pollution and Population Exposurehttps://doi.org/10.5194/acp-7-855-2007

Baltrėnas, P., & Baltrėnaitė, E. (2020). Sustainable Natural Materials and Their Importance for Waste Management and Stabilization of Soil Pollution. *Sustainable Environmental Protection Technologies* (pp. 93–141). Cham: Springer.

Barhoumi, B., LeMenach, K., Devier, M. H., Ameur, W. B., Etcheber, H., Budzinski, H., Cachot, J., & Driss, M. R. (2014). Polycyclic aromatic hydrocarbons (PAHs) in surface sediments from the Bizerte Lagoon, Tunisia Levels, sources, and toxicological significance. *Environmental Monitoring and Assessment, 186*(5), 2653–2669. https://doi.org/10.1007/s10661-013-3569-5

Bauer, A. K., Velmurugan, K., Plöttner, S., et al. (2018). Environmentally prevalent polycyclic aromatic hydrocarbons can elicit co-carcinogenic properties in an in vitro murine lung epithelial cell model. *Archives of Toxicology, 92,* 1311–1322.

Baumard, P., Budzinski, H., Michon, Q., Garrigues, P., Burgeot, T., & Bellocq, J. (1998). Origin and bioavailability of PAHs in the Mediterranean Sea from mussel and sediment records. *Estuarine, Coastal and Shelf Science, 47*(1), 77–90. https://doi.org/10.1006/ecss.1998.0337

Baumard, P., Budzinski, H., Garrigues, P., Dizer, H., & Hansen, P. D. (1999). Polycyclic aromatic hydrocarbons in recent sediments and mussels (Mytilus edulis) from the Western Baltic Sea Occurrence, bioavailability and seasonal variations. *Marine Environmental Research, 47*(1), 17–47. https://doi.org/10.1016/S0141-1136(98)00105-6

Bellasi, A., Binda, G., Pozzi, A., Galafassi, S., Volta, P., & Bettinetti, R. (2020). Microplastic contamination in freshwater environments A review, focusing on interactions with sediments and benthic organisms. *Environments, 7,* 30.

Birch, G. F. (1996). Sediment-bound metallic contaminants in Sydney's estuaries and adjacent offshore, Australia. *Estuarine, Coastal and Shelf Science, 42*(1), 31–44. https://doi.org/10.1006/ecss.1996.0003

CCME (Canadian Council of Ministers of the Environment), 2002. Canadian sediment quality guidelines for the protection of aquatic life. Summary tables. Canadian Environmental Quality Guidelines.

Chen, C. F., Chen, C. W., Dong, C. D., & Kao, C. M. (2013). Assessment of toxicity of polycyclic aromatic hydrocarbons in sediments of Kaohsiung Harbor. *Taiwan. Science of the Total Environment, 463,* 1174–1181. https://doi.org/10.1016/j.scitotenv.2012.06.101

Ciarkowska, K., Gambus, F., Antonkiewicz, J., & Koliopoulos, T. (2019). Polycyclic aromatic hydrocarbon and heavy metal contents in the urban soils in southern Poland. *Chemosphere, 229,* 214–226.

Collivignarelli, M. C., Abbà, A., Benigna, I., Sorlini, S., & Torretta, V. (2018). Overview of the main disinfection processes for wastewater and drinking water treatment plants. *Sustainability, 10,* 86.

Cowger, W., & Gray, A. (2020). Critical review of processing and classification techniques for images and spectra in microplastic research. *Applied Spectroscopy, 74*(9), 989–1010.

D'Amore, F., Bencardino, M., Cinnirella, S., Sprovieri, F., & Pirrone, N. (2015). *Data quality through a web-based QA/QC system: implementation for atmospheric mercury data from the global mercury observation system* (p. 8). Processes and Impacts.

Darılmaz, E., Kontaş, A., Uluturhan, E., Akçalı, İ, & Altay, O. (2013). Spatial variations in polycyclic aromatic hydrocarbons concentrations at surface sediments from the Cyprus (Eastern Mediterranean): relation to ecological risk assessment. *Marine Pollution Bulletin, 75*(1–2), 174–181. https://doi.org/10.1016/j.marpolbul.2013.07.042

Davoodi, H., Gharibreza, M., Negarestan, H., Mortazavi, M. S., & Lak, R. (2017). Ecological risk assessment of the Assaluyeh and Bassatin estuaries (northern Persian Gulf) using sediment quality indices. *Estuarine, Coastal and Shelf Science, 192,* 17–28. https://doi.org/10.1016/j.ecss.2017.05.003

de León-Gómez, H., Martin del Campo-Delgado, M. A., Esteller-Alberich, M. V., et al. (2020). Assessment of nitrate and heavy metal contamination of groundwater using the heavy metal pollution index case study of Linares. *Mexico. Environmental Earth Science*. https://doi.org/10.1007/s12665-020-09164-3

de Magalhães, L., Noyma, N. P., Furtado, L. L., et al. (2019). Managing eutrophication in a tropical brackish water lagoon Testing lanthanum-modified clay and coagulant for internal load reduction and cyanobacteria bloom removal. *Estuaries and Coasts, 42,* 390–402.

Dimashki, M., Lim, L. H., Harrison, R. M., & Harrad, S. (2001). Temporal trends, temperature dependence, and relative reactivity of atmospheric polycyclic aromatic hydrocarbons. *Environmental Science and Technology, 35*(11), 2264–2267. https://doi.org/10.1021/es000232y

Edokpayi, J. N., Odiyo, J. O., & Durowoju, O. S. (2017). Impact of wastewater on surface water quality in developing countries: a case study of South Africa. Water quality, 401–416.

Etats-Unis. Environmental Protection Agency (UEP), 1997 The incidence and severity of sediment contamination in surface waters of the United States. United States environmental protection agency. EPA-823-R-007, Washington, DC.

Fabião, J. P. F., Rodrigues, M. F. G., Fortunato, A. B., et al. (2016). Water exchanges between a multi-inlet lagoon and the ocean The role of forcing mechanisms. *Ocean Dynamics, 66,* 173–194.

Gagnon, C., & Fisher, N. S. (1997). The bioavailability of sediment-bound Cd Co, and Ag to the mussel Mytilus edulis. *Canadian Journal of Fisheries and Aquatic Sciences, 54*(1), 147–156. https://doi.org/10.1139/f96-256

Gevao, B., Boyle, E. A., Carrasco, G. G., Ghadban, A. N., Zafar, J., & Bahloul, M. (2016). Spatial and temporal distributions of polycyclic aromatic hydrocarbons in the Northern Arabian Gulf sediments. *Marine Pollution Bulletin, 112*(1–2), 218–224. https://doi.org/10.1016/j.marpolbul.2016.08.016

Gilliom, R. J. (2007). Pesticides in US streams and groundwater. Environmental Science & Technology. 3408–3414.

Grandclément, C., Seyssiecq, I., Piram, A., Wong-Wah-Chung, P., Vanot, G., Tiliacos, N., & Doumenq, P. (2017). From the conventional biological wastewater treatment to hybrid processes, the evaluation of organic micropollutant removal: a review. *Water research, 111,* 297–317.

Gune, M. M., Ma, W. L., Sampath, S., Li, W., Li, Y. F., Udayashankar, H. N., & Zhang, Z. (2019). Occurrence of polycyclic aromatic hydrocarbons (PAHs) in air and soil surrounding a coal-fired thermal power plant in the south-west coast of India. *Environmental Science and Pollution Research, 26*(22), 22772–22782.

Guo, W., He, M., Yang, Z., Lin, C., Quan, X., & Wang, H. (2007). Distribution of polycyclic aromatic hydrocarbons in water, suspended particulate matter and sediment from Daliao River watershed. *China. Chemosphere, 68*(1), 93–104. https://doi.org/10.1016/j.chemosphere.2006.12.072

Guzzella, L., & De Paolis, A. (1994). Polycyclic aromatic hydrocarbons in sediments of the Adriatic Sea. *Marine Pollution Bulletin, 28*(3), 159–165. https://doi.org/10.1016/0025-326X(94)90392-1

Habibullah-Al-Mamun, M., Ahmed, M. K., Hossain, A., & Masunaga, S. (2019). Distribution, source apportionment, and risk assessment of polycyclic aromatic hydrocarbons (PAHs) in the surficial sediments from the coastal areas of Bangladesh. *Archives of Environmental Contamination and Toxicology, 76*(2), 178–190. https://doi.org/10.1007/s00244-018-0571-5

Heiri, O., Lotter, A. F., & Lemcke, G. (2001). Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. *Journal of Paleolimnology, 25*(1), 101–110. https://doi.org/10.1023/A:1008119611481

Hoepner, T. (1999). A procedure for environmental impact assessments (EIA) for seawater desalination plants. *Desalination, 124*(1–3), 1–12. https://doi.org/10.1016/S0011-9164(99)00083-1

Hu, N., Shi, X., Huang, P., Mao, J., Liu, J., Liu, Y., & Ma, D. (2011). Polycyclic aromatic hydrocarbons (PAHs) in surface sediments of Liaodong Bay, Bohai Sea. *China. Environmental Science and Pollution Research, 18*(2), 163–172. https://doi.org/10.1007/s11356-010-0359-2

Hu, N. J., Huang, P., Liu, J. H., Ma, D. Y., Shi, X. F., Mao, J., & Liu, Y. (2014). Characterization and source apportionment of polycyclic aromatic hydrocarbons (PAHs) in sediments in the Yellow River Estuary. *China. Environmental Earth Sciences, 71*(2), 873–883. https://doi.org/10.1007/s12665-013-2490-0

Huang, W., Wang, Z., & Yan, W. (2012). Distribution and sources of polycyclic aromatic hydrocarbons (PAHs) in sediments from Zhanjiang Bay and Leizhou Bay. *South China. Marine Pollution Bulletin, 64*(9), 1962–1969. https://doi.org/10.1016/j.marpolbul.2012.05.023

Hull, S. L., Oty, U. V., & Mayes, W. M. (2014). Rapid recovery of benthic invertebrates downstream of hyperalkaline steel slag discharges. *Hydrobiologia, 736,* 83–97.

Jakovljević, I., Pehnec, G., Vađić, V., Čačković, M., Tomašić, V., & Jelinić, J. D. (2018). Polycyclic aromatic hydrocarbons in PM 10, PM 2.5 and PM 1 particle fractions in an urban area. *Air Quality, Atmosphere & Health*, *11*(7), 843–854.

Jaward, F. M., Alegria, H. A., Galindo Reyes, J. G., & Hoare, A. (2012). Levels of PAHs in the waters, sediments, and shrimps of Estero de Urias, an estuary in Mexico, and their toxicological effects. The Scientific World Journal, 2012.

Karbalaei, S., Hanachi, P., Walker, T. R., et al. (2018). Occurrence, sources, human health impacts and mitigation of microplastic pollution. *Environmental Science and Pollution Research, 25,* 36046–36063.

Karyab, H., Yunesian, M., Nasseri, S., Mahvi, A. H., Ahmadkhaniha, R., Rastkari, N., & Nabizadeh, R. (2013). Polycyclic aromatic hydrocarbons in drinking water of Tehran. *Iran. Journal of Environmental Health Science and Engineering, 11*(1), 1–7.

Katsoyiannis, A., & Breivik, K. (2014). Model-based evaluation of the use of polycyclic aromatic hydrocarbons molecular diagnostic ratios as a source identification tool. *Environmental Pollution, 184,* 488–494. https://doi.org/10.1016/j.envpol.2013.09.028

Keshavarzifard, M., Zakaria, M. P., Hwai, T. S., Mustafa, S., Vaezzadeh, V., Magam, S. M., Masood, N., Alkhadher, S. A., & Abootalebi-Jahromi, F. (2014). Baseline distributions and sources of polycyclic aromatic hydrocarbons (PAHs) in the surface sediments from the Prai and Malacca Rivers. *Peninsular Malaysia. Marine Pollution Bulletin, 88*(1–2), 366–372. https://doi.org/10.1016/j.marpolbul.2014.08.014

Keshavarzifard, M., Zakaria, M. P., & Hwai, T. S. (2017). Bioavailability of polycyclic aromatic hydrocarbons (PAHs) to short-neck clam (Paphia undulata) from sediment matrices in mudflat ecosystem of the west coast of Peninsular Malaysia. *Environmental Geochemistry and Health, 39,* 591–610. https://doi.org/10.1007/s10653-016-9835-z

Kim, K. H., Jahan, S. A., Kabir, E., & Brown, R. J. (2013). A review of airborne polycyclic aromatic hydrocarbons (PAHs) and their human health effects. *Environment International, 60,* 71–80. https://doi.org/10.1016/j.envint.2013.07.019.

Kim, S. K., & Chae, D. H. (2016). Seasonal variation in diffusive exchange of polycyclic aromatic hydrocarbons across the air–seawater interface in coastal urban area. *Marine pollution bulletin, 109*(1), 221–229.

Konstantinova, E., Minkina, T., Sushkova, S., et al. (2020). Levels, sources, and toxicity assessment of polycyclic aromatic hydrocarbons in urban topsoils of an intensively developing Western Siberian city. *Environmental Geochemistry and Health, 42,* 325–341.

Kucuksezgin, F., Pazi, I., Gonul, L. T., & Duman, M. (2013). Distribution and sources of polycyclic aromatic hydrocarbons in Cilician Basin shelf sediments (NE Mediterranean). *Marine Pollution Bulletin, 71*(1–2), 330–335. https://doi.org/10.1016/j.marpolbul.2013.01.026

Kucuksezgin, F., Pazi, I., Gonul, L. T., & Duman, M. (2016). Organochlorine compounds in surface sediments from the northern coast of Cyprus, Eastern Mediterranean: Levels, possible sources and potential risk. *Marine pollution bulletin*, *109*(1), 591–596.

Kuppusamy, S., Maddela, N. R., Megharaj, M., & Venkateswarlu, K. (2020). Ecological Impacts of Total Petroleum Hydrocarbons. *Total Petroleum Hydrocarbons* (pp. 95–138). Cham: Springer.

Laffoley, D., Baxter, J. M., et al. (2019). Eight urgent, fundamental and simultaneous steps needed to restore ocean health, and the consequences for humanity and the planet of inaction or delay. *Aquatic Conservation, 30*(1), 194–208.

Larsson, P. (1985). Contaminated sediments of lakes and oceans act as sources of chlorinated hydrocarbons for release to water and atmosphere. *Nature, 317*(6035), 347–349. https://doi.org/10.1038/317347a0

Li, T., Sun, G., Ma, S., Liang, K., Yang, C., Li, B., & Luo, W. (2016). Inferring sources of polycyclic aromatic hydrocarbons (PAHs) in sediments from the western Taiwan Strait through end-member mixing analysis. *Marine Pollution Bulletin, 112*(1–2), 166–176. https://doi.org/10.1016/j.marpolbul.2016.08.024

Lima, A. L. C., Eglinton, T. I., & Reddy, C. M. (2003). High-resolution record of pyrogenic polycyclic aromatic hydrocarbon deposition during the 20th century. *Environmental Science and Technology, 37*(1), 53–61. https://doi.org/10.1021/es025895p

Lin, C., Weng, C. E., Yuan, C. S., Lee, C. W., Hung, C. H., Bui, X. T., & Lin, J. X. (2018). Vertical stratification of volatile organic compounds and their photochemical product formation potential in an industrial urban area. *Journal of environmental management, 217,* 327–336.

Liu, L. Y., Wang, J. Z., Wei, G. L., Guan, Y. F., & Zeng, E. Y. (2012). Polycyclic aromatic hydrocarbons (PAHs) in continental shelf sediment of China: implications for anthropogenic influences on coastal marine environment. *Environmental Pollution, 167,* 155–162. https://doi.org/10.1016/j.envpol.2012.03.038

Loh, M., Sarigiannis, D., Gotti, A., Karakitsios, S., Pronk, A., Kuijpers, E., Annesi-Maesano, I., Baiz, N., Madureira, J., Oliveira Fernandes, E., Jerrett, M., & Cherrie, J. W. (2017). How Sensors Might Help Define the External Exposome. *International Journal of Environmental Research Public Health, 14*(4), 434.

Long, E. R., Macdonald, D. D., Smith, S. L., & Calder, F. D. (1995). Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management, 19*(1), 81–97. https://doi.org/10.1007/BF02472006

Lu, L., Luo, T., Zhao, Y., Cai, C., Fu, Z., & Jin, Y. (2019). Interaction between microplastics and microorganism as well as gut microbiota A consideration on environmental animal and human health. *Science of The Total Environment, 667,* 94–100.

Lübeck, J. S., Poulsen, K. G., Knudsen, S. B., Soleimani, M., Furbo, S., Tomasi, G., & Christensen, J. H. (2016). Source apportionment of polycyclic aromatic hydrocarbons (PAHs) in sediments from Khuzestan province. *Iran. Marine Pollution Bulletin, 110*(1), 584–590. https://doi.org/10.1016/j.marpolbul.2016.05.057

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*(4), 253–278. https://doi.org/10.1007/BF00118995

Magi, E., Bianco, R., Ianni, C., & Di Carro, M. (2002). Distribution of polycyclic aromatic hydrocarbons in the sediments of the Adriatic Sea. *Environmental Pollution, 119*(1), 91–98. https://doi.org/10.1016/S0269-7491(01)00321-9

Malarut, J., & Vangnai, K. (2020). Influence of wood types on quality and carcinogenic polycyclic aromatic hydrocarbons (PAHs) of smoked sausages. *Food Control, 85,* 98–106.

Mandalakis, M., Gustafsson, Ö., Alsberg, T., Egebäck, A. L., Reddy, C. M., Xu, L., Klanova, J., Holoubek, I., & Stephanou, E. G. (2005). Contribution of biomass burning to atmospheric polycyclic aromatic hydrocarbons at three European background sites. *Environmental Science and Technology, 39*(9), 2976–2982. https://doi.org/10.1021/es048184v

Manneh, R., Ghanem, C. A., Khalaf, G., Najjar, E., El Khoury, B., Iaaly, A., & El Zakhem, H. (2016). Analysis of polycyclic aromatic hydrocarbons (PAHs) in Lebanese surficial sediments A focus on the regions of Tripoli, Jounieh, Dora, and Tyre. *Marine Pollution Bulletin, 110*(1), 578–583. https://doi.org/10.1016/j.marpolbul.2016.05.058

McCubbin, F. M., Herd, C. D. K., Yada, T., et al. (2019). Advanced curation of astromaterials for planetary science. *Space Science Reviews, 215,* 48. https://doi.org/10.1007/s11214-019-0615-9

Mehr, M. R., Keshavarzi, B., Moore, F., Fooladivanda, S., Sorooshian, A., & Biester, H. (2020). Spatial distribution, environmental risk and sources of heavy metals and polycyclic aromatic hydrocarbons (PAHs) in surface sediments-northwest of Persian Gulf. *Continental Shelf Research, 193,* 104036. https://doi.org/10.1016/j.csr.2019.104036

Mocq, J., & Hare, L. (2018). Influence of Acid Mine Drainage, and Its Remediation, on Lakewater Quality and Benthic Invertebrate Communities. *Water, Air, and Soil pollution, 229,* 28.

Mostafa, A. R., Wade, T. L., Sweet, S. T., Al-Alimi, A. K. A., & Barakat, A. O. (2009). Distribution and characteristics of polycyclic aromatic hydrocarbons (PAHs) in sediments of Hadhramout coastal area, Gulf of Aden. *Yemen. Journal of Marine Systems, 78*(1), 1–8. https://doi.org/10.1016/j.jmarsys.2009.02.002

Neff, J. M. (1979). *Polycyclic aromatic hydrocarbons in the aquatic environment: sources, fates and biological effects*. Applied Science Publishers.

Neira, C., Cossaboon, J., Mendoza, G., Hoh, E., & Levin, L. A. (2017). Occurrence and distribution of polycyclic aromatic hydrocarbons in surface sediments of San Diego Bay marinas. *Marine Pollution Bulletin, 114*(1), 466–479. https://doi.org/10.1016/j.marpolbul.2016.10.009

Neşer, G., Kontas, A., Ünsalan, D., Altay, O., Darılmaz, E., Uluturhan, E., Küçüksezgin, F., Tekoğul, N., & Yercan, F. (2012). Polycyclic aromatic and aliphatic hydrocarbons pollution at the coast of Aliağa (Turkey) ship recycling zone. *Marine Pollution Bulletin, 64*(5), 1055–1059. https://doi.org/10.1016/j.marpolbul.2012.02.019

Nowak, P., Kucharska, K., & Kamiński, M. (2019). Ecological and health effects of lubricant oils emitted into the environment. *International Journal of Environmental Research and Public Health, 16,* 3002.

Ofori, S. A., Cobbina, S. J., & Doke, D. A. (2020). The occurrence and levels of polycyclic aromatic hydrocarbons (PAHs) in African environments—a systematic review. *Environmental Science and Pollution Research, 27,* 32389–32431. https://doi.org/10.1007/s11356-020-09428-2

Ohkouchi, N., Kawamura, K., & Kawahata, H. (1999). Distributions of three-to seven-ring polynuclear aromatic hydrocarbons on the deep sea floor in the central Pacific. *Environmental Science and Technology, 33*(18), 3086–3090. https://doi.org/10.1021/es981181w

Olenycz, M., Sokołowski, A., Niewińska, A., Wołowicz, M., Namieśnik, J., Hummel, H., & Jansen, J. (2015). Comparison of PCBs and PAHs levels in European coastal waters using mussels from the Mytilus edulis complex as biomonitors. *Oceanologia, 57*(2), 196–211. https://doi.org/10.1016/j.oceano.2014.12.001

Oracz, J., & Zyzelewicz, D. (2019). In vitro antioxidant activity and ftir characterization of high-molecular weight melanoidin fractions from different types of cocoa beans. *Antioxidants, 8,* 560.

Ozaki, N., Yamauchi, T., Kindaichi, T., & Ohashi, A. (2019). Stormwater inflow loading of polycyclic aromatic hydrocarbons into urban domestic wastewater treatment plant for separate sewer system. *Water Science and Technology, 79*(7), 1426–1436.

Porter, J. H., Hanson, P. C., & Lin, C. C. (2012). Staying afloat in the sensor data deluge. *Trends in ecology and evolution., 27*(2), 121–129.

Qiao, M., Wang, C., Huang, S., Wang, D., & Wang, Z. (2006). Composition, sources, and potential toxicological significance of PAHs in the surface sediments of the Meiliang Bay, Taihu Lake. *China. Environment International, 32*(1), 28–33. https://doi.org/10.1016/j.envint.2005.04.005

Ramzi, A., Rahman, K. H., Gireeshkumar, T. R., Balachandran, K. K., Jacob, C., & Chandramohanakumar, N. (2017). Dynamics of polycyclic aromatic hydrocarbons (PAHs) in surface sediments of Cochin estuary. *India. Marine Pollution Bulletin, 114*(2), 1081–1087. https://doi.org/10.1016/j.marpolbul.2016.10.015

Ravindra, K., Sokhi, R., & Van Grieken, R. (2008). Atmospheric polycyclic aromatic hydrocarbons: Source attribution, emission factors and regulation. *Atmospheric Environment, 42*(13), 2895–2921. https://doi.org/10.1016/j.atmosenv.2007.12.010

Readman, J. W., Fillmann, G., Tolosa, I., Bartocci, J., Villeneuve, J. P., Catinni, C., & Mee, L. D. (2002). Petroleum and PAH contamination of the Black Sea. *Marine Pollution Bulletin, 44*(1), 48–62. https://doi.org/10.1016/S0025-326X(01)00189-8

Reddy, M. S., Basha, S., Joshi, H. V., & Ramachandraiah, G. (2005). Seasonal distribution and contamination levels of total PHCs, PAHs and heavy metals in coastal waters of the Alang-Sosiya ship scrapping yard, Gulf of Cambay. *India. Chemosphere, 61*(11), 1587–1593. https://doi.org/10.1016/j.chemosphere.2005.04.093

Sany, S. B. T., Hashim, R., Salleh, A., Safari, O., Mehdinia, A., & Rezayi, M. (2014). Risk assessment of polycyclic aromatic hydrocarbons in the West Port semi-enclosed basin (Malaysia). *Environmental Earth Sciences, 71*(10), 4319–4332. https://doi.org/10.1007/s12665-013-2826-9

Sarti, E., Pasti, L., Scaroni, I., Casali, P., Cavazzini, A., & Rossi, M. (2017). Determination of n-alkanes, PAHs and nitro-PAHs in PM2. 5 and PM1 sampled in the surroundings of a municipal waste incinerator. *Atmospheric Environment, 149,* 12–23.

Savvides, C., Papadopoulos, A., Haralambous, K. J., & Loizidou, M. (1995). Sea sediments contaminated with heavy metals: metal speciation and removal. *Water Science and Technology, 32*(9–10), 65–73. https://doi.org/10.1016/0273-1223(96)00077-7

Sharma, S., & Chatterjee, S. (2017). Microplastic pollution, a threat to marine ecosystem and human health: a short review. *Environmental Science and Pollution Research, 24,* 21530–21547.

Siemering, G. S., & Thiboldeaux, R. (2020). Background concentration, risk assessment and regulatory threshold development: Polycyclic aromatic hydrocarbons (PAH) in Milwaukee. *Wisconsin surface soils. Environmental Pollution*, *268*, 115772.

Smith, M., Love, D. C., Rochman, C. M., et al. (2018). Microplastics in Seafood and the Implications for Human Health. *Current Environmental Health Reports, 5,* 375–386.

soil pollution and remediation. (2019). Sakshi, Singh, S.K. & Haritash, A.K. Polycyclic aromatic hydrocarbons. *International Journal of Environmental Science and Technology, 16,* 6489–6512. https://doi.org/10.1007/s13762-019-02414-3

Sun, Y., Wu, S., & Gong, G. (2019). Trends of research on polycyclic aromatic hydrocarbons in food: A 20-year perspective from 1997 to 2017. *Trends in Food Science and Technology, 83,* 86–98.

Sverdrup, L. E., Nielsen, T., & Krogh, P. H. (2002). Soil ecotoxicity of polycyclic aromatic hydrocarbons in relation to soil sorption, lipophilicity, and water solubility. *Environmental Science and Technology, 36*(11), 2429–2435. https://doi.org/10.1021/es010180s

Szopińska, M., Szumińska, D., Bialik, R. J., Dymerski, T., Rosenberg, E., & Polkowska, Ż. (2019). Determination of polycyclic aromatic hydrocarbons (PAHs) and other organic pollutants in freshwaters on the western shore of Admiralty Bay (King George Island, Maritime Antarctica). *Environmental Science and Pollution Research*, *26*(18), 18143–18161.

Taghvaee, S., Sowlat, M. H., Hassanvand, M. S., Yunesian, M., Naddafi, K., & Sioutas, C. (2018). Source-specific lung cancer risk assessment of ambient PM2. 5-bound polycyclic aromatic hydrocarbons (PAHs) in central Tehran. *Environment International, 120,* 321–332. https://doi.org/10.1016/j.envint.2018.08.003

Tobiszewski, M., & Namieśnik, J. (2012). PAH diagnostic ratios for the identification of pollution emission sources. *Environmental Pollution, 162,* 110–119. https://doi.org/10.1016/j.envpol.2011.10.025

Tolosa, I., De Mora, S. J., Fowler, S. W., Villeneuve, J. P., Bartocci, J., & Cattini, C. (2005). Aliphatic and aromatic hydrocarbons in marine biota and coastal sediments from the Gulf and the Gulf of Oman. *Marine Pollution Bulletin, 50*(12), 1619–1633. https://doi.org/10.1016/j.marpolbul.2005.06.029

Tongo, I., Ogbeide, O., & Ezemonye, L. (2017). Human health risk assessment of polycyclic aromatic hydrocarbons (PAHs) in smoked fish species from markets in Southern Nigeria. *Toxicology Reports, 4,* 55–61.

Truskewycz, A., Gundry, T. D., Khudur, L. S., Kolobaric, A., Taha, M., Aburto-Medina, A., Ball, A. S., & Shahsavari, E. (2019). Petroleum Hydrocarbon Contamination in Terrestrial Ecosystems—Fate and Microbial Responses. *Molecules, 24,* 3400.

U. S. Integrated, "Manual for Real-Time Quality Control of Ocean Optics Data: A guide to quality control and quality assurance of coastal and oceanic optics observations", Ocean Observing System 2015, pp. 46.

Vagge, G., Cutroneo, L., Castellano, M., Canepa, G., Bertolotto, R. M., & Capello, M. (2018). The effects of dredging and environmental conditions on concentrations of polycyclic aromatic hydrocarbons in the water column. *Marine pollution bulletin, 135,* 704–713.

Vijayan, A., Österlund, H., Marsalek, J., et al. (2019). Laboratory melting of late-winter urban snow samples: The Magnitude and dynamics of releases of heavy metals and

PAHs. *Water, Air, and Soil pollution, 230,* 182. https://doi.org/10.1007/s11270-019-4201-2

Wang, C., Wang, W., He, S., Du, J., & Sun, Z. (2011). Sources and distribution of aliphatic and polycyclic aromatic hydrocarbons in Yellow River Delta Nature Reserve. *China. Applied Geochemistry, 26*(8), 1330–1336. https://doi.org/10.1016/j.apgeochem.2011.05.006

Wang, B., Zeng, C., Chu, K. H., Wu, D., Yip, H. Y., Ye, L., & Wong, P. K. (2017). Enhanced biological hydrogen production from Escherichia coli with surface precipitated cadmium sulfide nanoparticles. *Advanced Energy Materials, 7*(20), 1700611. https://doi.org/10.1002/aenm.201700611

Wu, K., Huang, J., Yi, J., Liu, X., Liu, Y., Wang, Y., Zhang, J., & Xia, Y. (2020). Recent advances in polymer electrolytes for zinc ion batteries: Mechanisms, properties, and perspectives. *Advanced Energy Materials*. https://doi.org/10.1002/aenm.201903977

Yan, J., Liu, J., Li, Y., & Lang, S. (2014). Effect of water current on the distribution of polycyclic aromatic hydrocarbons, heavy metals and benthic diatom community in sediments of Haihe estuary. *China. Environmental Science and Pollution Research, 21*(20), 12050–12061. https://doi.org/10.1007/s11356-014-3145-8

Yun, X., Yang, Y., Liu, M., Zhang, M., & Wang, J. (2016). Distribution, seasonal variations, and ecological risk assessment of polycyclic aromatic hydrocarbons in the East Lake. *China. CLEAN Soil Air Water, 44*(5), 506–514. https://doi.org/10.1002/clen.201400187

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. https://doi.org/10.1016/S0146-6380(02)00002-5

Zakaria, M. P., Takada, H., Tsutsumi, S., Ohno, K., Yamada, J., Kouno, E., & Kumata, H. (2002). Distribution of polycyclic aromatic hydrocarbons (PAHs) in rivers and estuaries in Malaysia A widespread input of petrogenic PAHs. *Environmental Science and Technology, 36*(9), 1907–1918. https://doi.org/10.1021/es011278+

Zeng, S., Zeng, L., Dong, X., & Chen, J. (2013). Polycyclic aromatic hydrocarbons in river sediments from the western and southern catchments of the Bohai Sea, China Toxicity assessment and source identification. *Environmental Monitoring and Assessment, 185*(5), 4291–4303. https://doi.org/10.1007/s10661-012-2869-5

Zha, D., Li, Y., Yang, C., et al. (2018). Assessment of organophosphate flame retardants in surface water and sediment from a freshwater environment (Yangtze River, China). *Environmental Monitoring and Assessment, 190,* 222. https://doi.org/10.1007/s10661-018-6587-5

Zhang, J. D., Wang, Y. S., Cheng, H., Jiang, Z. Y., Sun, C. C., & Wu, M. L. (2015). Distribution and sources of the polycyclic aromatic hydrocarbons in the sediments of the Pearl River estuary. *China. Ecotoxicology, 24*(7–8), 1643–1649. https://doi.org/10.1007/s10646-015-1503-z

Zhang, P., Pan, X., Wang, Q., et al. (2020). Toxic effects of heavy metals on the freshwater benthic organisms in sediments and research on quality guidelines in Poyang Lake, China. *Journal of Soils and Sediments, 20,* 3779–3792. https://doi.org/10.1007/s11368-020-02700-5

Zhang, L., Wang, Y., Tan, F., Yang, Y., Wu, X., Wang, W., & Liu, D. (2020). Tidal variability of polycyclic aromatic hydrocarbons and organophosphate esters in the coastal seawater of Dalian. *China. Science of The Total Environment, 708,* 134441. https://doi.org/10.1016/j.scitotenv.2019.134441

Zheng, H., Xing, X., & Hu, T. (2018). Biomass burning contributed most to the human cancer risk exposed to the soil-bound PAHs from Chengdu Economic Region, western China. *Ecotoxicology and Environmental Safety, 159,* 63–70.

Yusuf, K. A., Ezechukwu, L. N., Fakoya, K. A., Akintola, S. L., Agboola, J. I., & Omoleye, T. O. (2015). Influence of fish smoking methods on polycyclic aromatic hydrocarbons content and possible risks to human health. *African Journal of Food Science, 9*(3), 126–135.

Zambianchi, M., Durso, M., Liscio, A., Treossi, E., Bettini, C., Capobianco, M. L., & Melucci, M. (2017). Graphene oxide doped polysulfone membrane adsorbers for the removal of organic contaminants from water. *Chemical Engineering Journal, 326,* 130–140.