Assessment of metal levels and pollution indices of the Songor Wetland, Ghana

Emmanuel Klubi [a,*], Dennis Kpakpo Adotey [b,d], Samuel Addo [a], José M. Abril [c]

[a] *Department of Marine and Fisheries Sciences, School of Biological Sciences, College of Basic and Applied Sciences, University of Ghana, Ghana* [b] *School of Nuclear and Allied Sciences, College of Basic and Applied Sciences, University of Ghana, Ghana*
[c] *Departmento de Fisca Aplicada I, ETSIA, Universidad de Sevilla, Spain*
[d] *Ghana Atomic Energy Commission, P. O. Box LG 80, Legon, Ghana*

ABSTRACT

Sediment, soil and fish samples collected from the Songor Wetland were assessed for the spatial distributions of metal (Al, Fe, Zn, Cr, Ni, Cu, Pb, As, Cd and Hg) levels and contamination loads, using pollution indices and multivariate analysis. The samples were processed through microwave digestions (Soil and Sediment: [HNO_3, HCl, HF, H_2O_2]; Fish: [HNO_3, H_2O_2]), followed by the analysis of extracts using ICP-MS. The results displayed patchiness of metal levels in the sediment and soil samples, and in some cases, defying the established trend that levels of metals in sediments are generally higher than in soils. The differences in the results were ascribed to geological dominance and anthropogenic impacts. The finfish species displayed relatively higher bioaccumulation patterns of the metal levels than crustaceans. Aluminium (Al) and Zn levels were moderately enriched in sediment and soil samples. Overall enrichment factors ([EF {Al} and EF {Fe}]) suggested low to minimal enrichment except in a few cases. Pollution Load Index (PLI) based on Contamination Factor (CF) suggested that metal loads were less than baseline levels. On the other hand, Pearson Correlation Coefficient demonstrated that the metals present in the wetland were more lithogenic with remarkable inputs of biogenic and anthropogenic components. Principal Component Analysis revealed an association of Zn to the western section of the wetland, Cd, and Hg to the eastern part while the remaining metals were concentrated at the mid-section of the wetland. Lead (Pb) levels (1.10 $\pm$ 0.70 mg/kg) in finfishes exceeded the EU Regulation 1881/2006/EU (0.05 mg/kg) for fish tissues and could pose public health concerns.

1. Introduction

Probabilistic assessments for pollution evaluations in most environmental studies provide information on the original states of the pristine environments as well as knowledge on the evolving changes due to pollution (Romano et al., 2015; Ruiz-Fernández et al., 2019). On the other hand, the lack of appropriate baseline information makes explanation and cross border inter-comparison of probabilistic studies largely unattractive. This has led to the introduction of recommended and permissible levels of environmental contaminants for monitoring purposes (Costa-Böddeker et al., 2018; Jarapala et al., 2014; McCauley et al., 2000; Twining et al., 2008). Notwithstanding, there are still issues of analytical concerns. Various problems are encountered in an attempt to estimate dissolved metal levels in water columns primarily due to low levels of dissolved metals in aquatic ecosystems (Rauch and Pacyna, 2009). Accordingly, pre-concentration and acid digestion as well as back-extraction techniques, are employed (Lo et al. 1982; Koesmawati et al., 2017). In bulk sediments, the total metal contents are affected by biogeochemical interactions that subsequently impact the available aqueous phases of dissolved metals in the water column and in the biota (Rauch and Pacyna, 2009). The amounts of exchangeable metals held in the sediments are also affected by particle sizes and organic fractions (natural or anthropogenic through sewages) of the bulk sediment. Physical and chemical parameters such as pH, salinity and redox potential also play differential roles in metal's availability and bioaccumulation (Xu et al., 2014). Benthic and nekton communities also influence the distributions of metal levels in aquatic ecosystems through diverse bio-accumulation pathways (Oyekanmi et al., 2017; Tepe et al., 2008). Thus, the exchangeable fractions of metals could be in transition and may affect the evaluation of metal concentrations for pollution load estimations. However, the introduction of mathematical models has proven that the exchangeable fractions and the species of environmental contaminants could be linked through a multivariate analysis. This procedure presents pictorial interlinkages

between elements of concern, and is capable of displaying the interdependence and sorption processes of the pollutants in the environment (Gustafsson and Van-Schaik, 2003).

The principles of pollution indices and multivariate analysis have been documented across different ecosystems, estuaries, river systems, lakes, marine, freshwater and wetlands (Klubi et al., 2018; Kouassi et al., 2015; Oketola et al., 2013). As such, the focus of this study was to apply the principles of pollution indices and multivariate analysis [the Pearson Coefficient Correlation and Principal Component Analysis (PCA)] in understanding the biogeochemical interactions of metals in the Songor Wetland, a coastal wetland ecosystem in southeastern Ghana.

Wetlands are identified as a natural remedy for pollution control through passive removal of excess contaminants from aquatic ecosystems; thereby minimizing the introduction of pollutants into the food chain (Carter, 1991). This process is associated with the biogeochemical interplay which occurs between the physical and chemical properties of water; in addition to the uptake of environmental contaminants by organisms and absorbed into the sediment bed. Sequestration of metals in wetlands through biogeochemistry processes is also linked to sulphate reduction to sulphide by sulphur-reducing bacteria (SRB) under high anoxic conditions and low oxidation–reduction potentials [ORP] (Xu and Mills, 2018). The SRB precipitates metals as sulphides in most cases and is not limited by environmental factors such as pH and dissolved oxygen levels. The SRB biogeochemical process is a substitute for ferric iron electron transfer mechanisms in relatively low pH conditions existing in wetlands (Xu and Mills, 2018). As such, the natural and passive removal of excess metals enhances water quality and reduces the risk of aquatic food web contaminations (Xu et al., 2014). Anthropogenic interactions with the ecosystems and natural bio-perturbation may lead to remobilization and release of excess metals into overlying waters and pose health implications for biota and humans.

The use of pollution indices requires a proper definition of geo-chemical background levels (Loring and Rantala, 1992). As such, in this study, upper crustal metal levels were used as background concentrations for the pollution indices evaluations (Rudnick and Gao, 2003). Additionally, for comparison purposes, the content of the metals of a wide set of surficial sediments in the Volta Estuary, described by Klubi et al. (2018), was also used as a local reference.

2. Materials and methods

2.1. Study area

The Songor Wetland (located between latitude 05° 51′ 48″ N and 05° 48′ 30″ N and between longitude 00° 31′ 07″ E and 00° 37′ 15″ E, respectively) is a low-lying gentle slope area covering about 1100 hectares of a depressional land of the Volta deltaic system characterized by parallel and isolated dunes next to the Atlantic Ocean at the eastern coast of Ghana (Fig. 1). It is a natural wetland occupying the western part of the Volta deltaic system. The wetland is separated from the lower section of the Volta River (near the Volta Estuary to the east) by a strip of mudflats on which the Ada Township is built. The northern and western parts are bounded by elevated uplands; while the southern part is bounded by the Gulf of Guinea.

The Songor Wetland receives freshwater inflows from the lower sections of the river by multiple tributaries that cut through the Ada Township into the wetland to the east. Groundwater recharges and surface water runoffs enter through the north and seawater via the southwestern corner. The seawater is believed to be toppling over the sand bars separating the wetland and the Gulf of Guinea during high tide peaks and spring tides. There is

also an intermittent breaching of the sand bar (Dankwa et al., 2004). The inflow of freshwater into the Songor Wetland is assumed to be reduced by dams built upstream on the Volta River at Akosombo and Kpong in 1964 and 1982, respectively (Bollen et al., 2011; Finlayson et al., 2000; Nyarko et al., 2016). The inflow of freshwaters into the wetland currently depends on the tidal surges through the estuary near Ada-Foah, which "pushes" diverted waters into the wetland. Also, the surface runoffs from the northern sections of the wetland are collected in low-lying meadows which are mainly used for subsistence vegetable farms at the immediate peripheries of the wetland.

2.2. Sampling

Sediment, soil and fish specimens were collected along the main channel connecting the Songor wetland and the Songor lagoon (Fig. 1). Sediment samples were collected at the eastern sections (AD-11, AD-7, and AD-5) using a Birge-Ekman grab (Hydro-Bios Ltd.), deployed from a fibreglass boat. The grab allowed for a maximum penetration depth of 15 cm in a soft-bottom substrate. At the western end (AD-1, AD-3, and AD-4) a UWITEC PVC corer was manually used to collect the surface sediment due to relatively low water levels. The core diameter was 87 mm and the top $\approx$10–15 cm was retrieved for analysis following the same procedure as for the Ekman grab.

Soil samples were obtained, using an Auger sampler on dried coastal sand bars located at the seaward (AD-U) and landward (AD-L) locations of the main watercourse; and on the banks of the channel at locations AD-8, AD-3, and AD-2 within the wetland, respectively (Fig. 1). The soil samples were heterogeneous mixtures of interbedded sands, clays and silts.

Fish specimens were obtained by traps made from coconut plant fronds, set in between barricades overnight, and supported by active fishing using a cast net between locations AD-4 and AD-6 (Fig. 1). The samples (sediment, soil and fish) were placed in hermetically-sealed polyethene bags and kept in thermally insulated receptacles with ice cubes to minimize enzymatic activities of microbes (organisms). The samples were then conveyed to the laboratories of the Department of Marine and Fisheries Sciences, University of Ghana for analysis.

2.3. Sample preparations

At the laboratory, sediment and soil samples were oven-dried to constant weight at 80 °C, using the Kallen Camp Plus II Oven. Fish samples were sorted, identified and grouped into similar size ranges. All the fish samples were rinsed with distilled water and oven-dried (80 °C) to a constant weight. The dried fish tissues were crushed in a mortar using a pestle and then processed further into a fine powdery form using an electric blender with stainless blades. The fish samples were de-scaled and then pulverized by selecting twelve (12) individuals of similar size ranges (9 cm and 15 cm of total length [TL]) of Cichlidae (*Sarotherodon melanotheron and Hemichromis fasciatus*). Nine (9) individual crustaceans (*Macrobrachium sp.*), measuring between 8 and 14 cm total lengths (telson to the rostrum) were also prepared similarly as the finfish.

The dried sediment and soil samples were, however, re-packaged into hermetically-sealed polyethene bags. The milled fish samples were stored similarly, and shipped for the metal levels analysis at the Czech Republic's Masaryk University Research Centre for Toxic Compounds in the Environment (RECETOX), Brno.
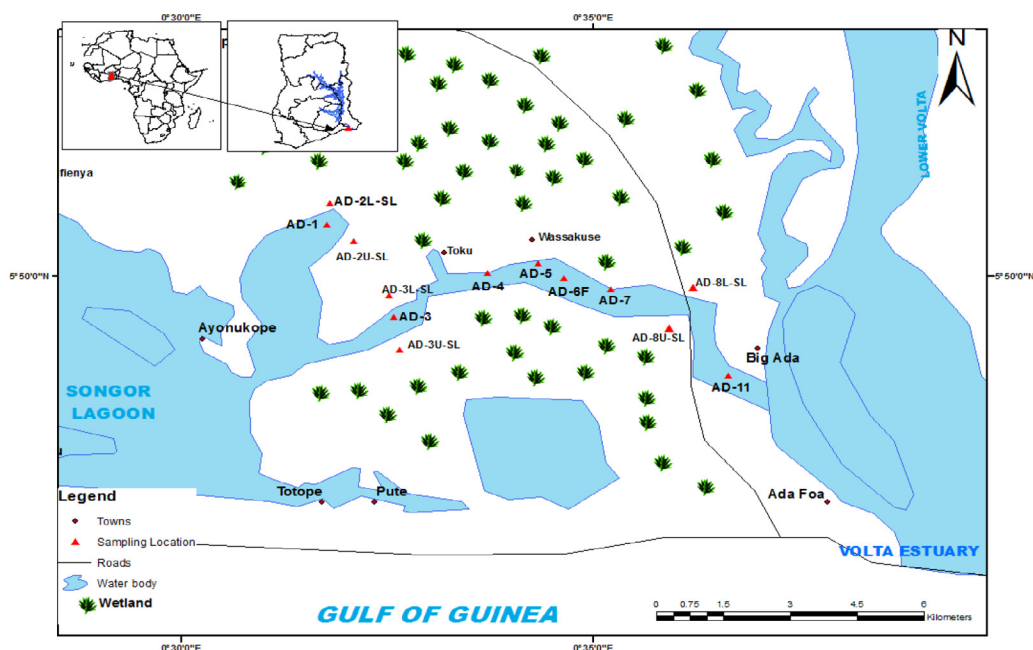
Fig. 1. Map of Songor Wetland showing the studied areas (marks in red). Insert: Africa Map and Ghana map. AD-1 up to AD-11 marks are sediment collection points, AD-1U and AD-1L marks are the soil collection points, and AD-4 to AD-6F marks are fish collection points. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

2.4. Laboratory analysis

At RECETOX, the dried sediment and soil samples were sieved and the larger grains were excluded by being retained on a 125 μm sieve. Micro-wave digestions were then performed on the fine particle (< 125 μm) samples by weighing approximately 0.3 g and mixed with aqua-regia (HNO_3 and HCl), plus hydrofluoric acid (HF) [1:3:1.5 ratio] (Gaudino et al., 2007; Bettinelli et al., 2000). The methods were adopted from the USEPA Method 3052 (USEPA, 1996).

For the fish samples, 0.2 g was weighed and mixed with a 2:1 ratio of 69% HNO_3 and 30% H_2O_2. All the reagents were of Supprapur grades (Merck, Darmstadt, Germany). The closed-vessel digestion was performed using a microwave system, MWS 3+ Berghof, Germany. The digests were allowed to cool to room temperature, and diluted with 18 MΩ cm deionized water to 100 mL volume, and allowed to stand and settle for 48 h at 4 °C; followed by filtration (Klubi et al., 2018).

The extracts were analysed for metal (Al, Fe, Zn, Cr, Ni, Cu, Pb, As and Cd) levels using the Agilent ICP-MS 7700 series Inductively Coupled Plasma Mass Spectrometer (Agilent Technologies, Inc., Japan). The ICP-MS 7700 series was equipped with a triple and single quadrupole ion trap, electron, chemical ionization and multichannel detector, and coupled with ASX-500 Series Auto Sampler. Samples were analysed in triplicates. For each sample, a full spectrum was acquired with the instrument in the survey mode (10 channels per mass, with 10 sweeps and dwell time of 0.6 s). Long pre-acquisition and washing times of 90 s each were enabled to minimize memory effects. The net areas of the peaks for the m/z ratio were estimated by subtraction of the nearest reagent blank and quantified through the calibration standards.

Total-mercury (THg) levels in the sediment, soil and fish samples were, however, analysed using a one-purposed atomic absorption spectrophotometer AMA-254 (Advanced Mercury Analyser) with direct solid sampling automation (Száková et al., 2004). About 250 mg of the sediment and soil samples and 200 mg of the fish samples were (respectively) transferred into separate analytical boats after the performance of the zero calibration and

a three-point calibration (linear calibration) for THg at 253.7 nm wavelength. The samples were pre-heated at 120 °C for 200 s and decomposed at 750 °C for 150 s; followed by post-combustion flushing at 700 °C for 45 s (Cizdziel et al., 2002).

2.5. Quality assurance (QC)

The analytical methods were validated, using certified reference materials (CRM) MET-31 (from ANALYTIKA®, spol. s r.o.) and internal CRM (from an inter-lab collaboration). DORM-2 (Dogfish Muscle) [from the National Research Council of Canada (NRCC)] was used as the CRM for the fish samples. The CRMs were analysed together with the samples under the same conditions. The results obtained are presented in Tables 1 and 2. The tables depict the arithmetic mean values and the standard deviations as well as the minimum detection limits (MDL) of the three replicates of the CRM-MET-31 and DORM-2 dogfish muscle, NRCC, respectively. The CRM-MET-31 results compare well with the certified values, and show percentage (%) recovery values between 94% (for Hg) and 107% (for Fe) and an arithmetic mean of 98.6 ± 2.1%. The acid digested method produced a good recovery of the mean of the certified values within the second and the third order of standard deviations.

The CRM-MET-31 (Metranal 31) information values for Al and Fe concentrations were inferred from the percentages of the oxides as 81.6 g/kg and 33.1 g/kg, respectively. This validates the good performance of the digestion method for Fe, while the low recovery for Al in this sample (31%) may be attributed to the likely formation of insoluble fluorides (such as AlF_3). This, however, does not limit the use of the method for the other elements since their recoveries were validated directly against the certified values (Table 1). Moreover, the method performed well for Al in the case of internal CRM (Table 1).

On the other hand, the DORM-2 dogfish muscle showed a better percentage (99.89 ± 10.33%) of recovery than the sediment CRM-MET-31 (Tables 1 and 2, respectively). This could be attributed to differences in the matrices. Fish muscles (organic) are more susceptible to acid dissolution than solid particles of a highly compacted and complex matrix like silica.

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Table 1
Results of quality assurance (QA)[a] and the method of detection limits (MDL) for the studied certified reference material MET-31 and internal reference material (Internal CRM).

Element	m/z	MDL	MET-31			Internal CRM	
			Measured mgkg^{-1}	Certified	Recovery %	Measured mgkg^{-1}	Certified
Al[c]	27	10	25 ± 2		100 ± 5	45 ± 2	45 ± 1
Fe[c]	57	80	31 ± 2		107 ± 1	35 ± 0.6	34 ± 1
Cr	52	0.1	80 ± 6	71.9 ± 5.9	98 ± 0.4	109 ± 3	104 ± 4
Ni	60	0.06	31 ± 2	31.8 ± 1.2	94.6 ± 0.4	37.2 ± 0.7	33 ± 2
Cu	65	0.09	27 ± 3	28.9 ± 0.8	94.7 ± 0.8	28.2 ± 0.6	26 ± 2
Zn	66	1	104 ± 7	108 ± 3.5	96.6 ± 1.3	226 ± 5	200 ± 8
As	75	0.1	12 ± 1	10.4 ± 1	103 ± 1	28.8 ± 0.5	25 ± 2
Cd	111	0.004	0.27 ± 0.02	0.29 ± 0.04	97.2 ± 1.1	1.02 ± 0.02	0.96 ± 0.08
Pb	208	0.07	26 ± 2	24.1 ± 1.7	101 ± 1	84 ± 2	82 ± 5
Hg[b]	202	0.001	0.082 ± 0.005	0.087 ± 0.006	94 ± 9	0.084 ± 0.005	

[a]Mean and standard deviation of CRM (n = 3).
[b]Analysis based on direct mercury analyser (DMA).
[c]Values in g/kg.

Table 2
Results of quality assurance (QA)[a] and the method of detection limits (MDL) for the certified reference material DORM-2 (Dogfish Muscle) [NRCC].

Element	MDL	DORM-2 Dogfish Muscle		
		Measured μgg^{-1}	Certified	Recovery %
Al	–	NA	NA	–
Fe	–	139.3 ± 5	142 ± 10	98 ± 8
Cr	–	33.8 ± 2.5	34.7 ± 5.5	97 ± 17
Ni	0.06	20.5 ± 1.7	19.4 ± 3.1	106 ± 18
Cu	0.08	2.17 ± 0.08	2.34 ± 0.16	93 ± 8
As	0.004	17.3 ± 0.8	18.0 ± 1.1	96 ± 8
Zn	2.0	26.4 ± 2.3	25.6 ± 2.3	103 ± 13
Cd	0.0006	0.043 ± 0.003	0.043 ± 0.008	100 ± 2
Pb	0.3	0.070 ± 0.005	0.065 ± 0.007	108 ± 13
Hg[b]	0.0003	4.48 ± 0.08	4.64 ± 0.26	97 ± 6

NRCC (National Research Council of Canada).
[a]Mean and standard deviation of CRM (n = 3).
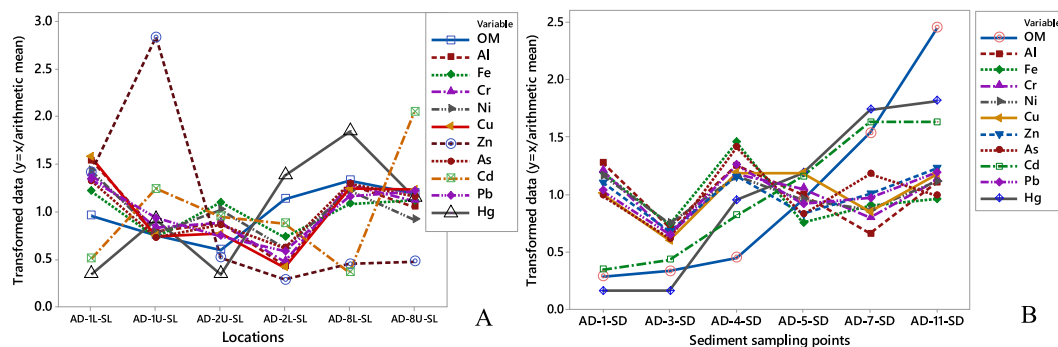[b]Analysis based on direct mercury analyser (DMA).

Fig. 2. Spatial distribution of the elements studied in the soil **(A)** and sediment **(B)** samples of the wetland. For the sake of comparison, and due to scale differences, the raw data were transformed using the mean values.

2.6. Organic matter content

The organic matter (OM) contents of the sediment and soil samples were estimated by the Lost-On-Ignition (LOI) method. The LOI is the most widely used procedure for a quick appraisal of OM contents in soil. However, there is no available universal standardized protocol. For this study, 5 g of the bulk dried sediment and soil samples were pre-heated at 105 °C overnight and weighed after cooling in a desiccator, followed by combustion at 360 °C for two (2) hours (Konen et al., 2002). The combusted samples and the sample holders (crucibles) were cooled at 105 °C for four hours before cooling to room temperature in a desiccator; and weighing. The differences in the pre-and post-weights were attributed to the organic composition of each sample and reported in percentage (%) terms as OM.

2.7. Statistical analysis

Statistical analyses of the data were performed using Minitab® 18 and PRIMER® 6 Computer Software as shown in Figs. 2 to 6, respectively. The data were reported by computing the arithmetic means and standard deviations [SD] (Table 1). The propagation of error and testing of homogeneity were performed, using One-Way Analysis of Variance (ANOVA) and inter-comparison of the means by Least Significant Differences (LSD) tests at 95% confidence levels (CL). Provenance analysis was achieved through the

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Pearson Coefficient Correlation and Principal Component Analysis (PCA).

Natural and human impacts were established using geoaccumulation index ($\mathbf{I_{geo}}$) and enrichment factor (EF); followed by the pollution load index (PLI).

The $\mathbf{I_{geo}}$ is a quantitative measure (Muller, 1969; Magesh et al., 2011) for estimating the extent of contamination in the environment per Eq. (1);

$$I_{geo} = \log_2\left(\frac{Cn}{1.5*Bn}\right) \qquad (1)$$

where C_n is the measured concentration in the sediment for the metal n; Bn, the corresponding geochemical background value (upper earth crustal values were used as the background for the sediment and soil samples); the 1.5 is a constant introduced to account for background variations as a result of geological differences in the sample (sediment and the soil) basins.

The enrichment factor *(EF)* is an estimator for evaluating anthropogenic influences on sediments or soils using either Al or Fe as a normalizing element. It was estimated based on the formula in Eq. (2) (Magesh et al., 2011):

$$EF = \frac{C_a \cdot Fe_b}{C_b \cdot Fe_a} \qquad (2)$$

where: C_a and C_b are the examined metal contents in the sample and the background reference; while Fe_a and Fe_b are the Fe contents in the sample and the background reference, respectively.

The Pollution Load Index (PLI) was used to assess site-specific contaminations of the analysed elements from the Songor Wetland. The PLI values were calculated by the estimations of the contamination factors (CF), according to Tomlinson et al. (1980) using Eqs. (3) and (4), respectively.

$$PLI = \sqrt[n]{(CF_1 * CF_2 * CF_3 \ldots CF_n)} \qquad (3)$$

$$CF = \frac{C_m}{C_b} \qquad (4)$$

CF is the contamination factor; C_m, the concentration of metal from each sampling point; C_b, the background level of the same metal in the reference site; n is the number of metals studied. The PLI provides an estimation of metal contamination status and the probable action that needs to be taken. PLI-value < 1 depicts no sample contamination, whereas PLI = 1 suggested a pollutant at a baseline level and a PLI > 1 an indication of site deterioration (Tomlinson et al., 1980). However, Likuku et al. (2013) suggested a proactive definition for PLI values. Accordingly, a PLI value ≥ 1 indicates an immediate intervention needed to ameliorate the contamination, 0.5 ≤ PLI < 1 suggests monitoring of the site, and PLI < 0.5 implies no need for drastic measures. Additionally, PLI-values less than one (<1) are considered as no pollution whereas greater than one (>1) is termed polluted Harikumar and Nasir (2010).

3. Results and discussion

3.1. Organic matter content

The organic matter (OM) levels (2.28 ± 1.95%) in the sediment were relatively higher than in the soil samples (1.81 ± 0.50%) [Table 3]. This could be a confirmation of multiple sources of organic matter reaching the sediment compared to soil. Actually, the sediment lining the water bed comprised depositions of particle grains coated with organic matter of different origins (water column, terrestrial [leaf-litter] and epibenthic fauna as well as detritus pools). Fisher's Least Significant Differences (LSD) at 95% Confidential Levels (CL) assessment, however, revealed no significant differences between the OM values in the sediment and the soil (Table 3).

3.2. Metal levels

3.2.1. Fish

Generally, the metal concentrations in the Cichlidae (*S. melanotheron and H. fasciatus*) [fin fish] were relatively higher than the crustacean concentrations [*Macrobrachium sp.*] (Table 3). This may be linked to their different feeding characteristics (Oyekanmi et al., 2017). The Cichlidae are mostly omnivores and detritivores as compared to *Macrobrachium* sp. that feeds on phytoplankton and zooplankton and switch to being omnivores in their adult stages (Oyekanmi et al., 2017). The concentrations of metals in the fish were within the EU Regulation 1881/2006/EU maximum limits (Hg, Cd, and Pb of 0.5 mg/kg, 0.5 mg/kg and 0.3 mg/kg, respectively) established for fish tissues. The exception was the Pb levels which could be issues of public concern.

3.2.2. Sediment and soils

The metal levels in sediment and soil samples are reported in Table 3 together with Sediment Toxicity Reference Values (STRV), after Jones and Suter II (1997) and some metal levels from the sediment of the Volta Estuary (Volta E. Ref.), after Klubi et al. (2018) [as a local reference value]. The Volta Estuary could be currently considered as a pristine environment within the coastal belt of Ghana (Addo et al., 2011; Asare-Donkor and Adimado, 2016; Biney, 1982; Donkor et al., 2006; Klubi et al., 2018; Tay et al., 2010).

The concentrations of the metals were in the order Al>Fe>Cr>Zn>Ni>Cu>Pb>As>Cd>Hg for the sediment; and for soil, the order is Al>Fe>Zn>Cr> Ni>Cu>Pb>As>Cd>Hg. The metal loads in the sediment were slightly higher than in the soil except in a few cases (Table 3). The LSD test at 95% CL indicated that the assessed metals showed significant differences between their mean values within the sediment and the soil, respectively (Table 3). However, on the level-by-level basis between sediment and the soil samples, there were no significant differences as indicated by "a", "b" and "c" in Table 3. On the other hand, it can also be inferred from the mean and standard deviation values (Table 3) that there were instances of significant differences between the elements in the sediment and the soil. The apparent and significant differences in the metal levels may be the variations in their sources reaching the sediment and soils of the Songor Wetland, respectively. It also suggested that the biogeochemical processes in the sediment and the soil are likely to be different, though, the soil and sediment samples were collected from adjacent areas (Fig. 1). This agrees with the findings of Dankwa et al. (2004), that there is a limited supply of waters into the Songor Wetland and that most parts of the wetland are dry during the greatest part of the year.

The dominance of Al levels in the sediment and soil samples and their mean values (6.0 ± 1.6% [sediment], 6.2 ± 2.4% [soil]) were comparable with the Global mean concentrations in soil (6.2%) and in loess (6.7%) according to the International Atomic Energy Agency (IAEA) Database of Natural Reference Materials, as cited by Rauch and Pacyna (2009); and Poulton and Raiswell (2000), respectively. Iron (Fe) levels in both matrices (3.6 ± 0.8% [sediment], 2.5 ± 0.5% [soil]) were also within the Global mean levels in soil (3.3%) and in loess (2.4%) from the above database. Iron (Fe) and Al are considered conservative elements in the earth crustal, as such, their levels are not expected to change significantly.

Chromium (Cr) levels in the sediment (66 ± 15 mg/kg) and the soil (60 ± 20 mg/kg) samples were comparable to the levels in the Cichlidae (62 ± 38 mg/kg) and the crustacean (58.93 mg/kg) [Table 3]. The levels were also in close agreement with the Global mean concentrations for loess (59 mg/kg), continental sediment (74 mg/kg) and oceanic sediment (79 mg/kg), as indicated in

Table 3

Elemental concentrations in the sediment, soil, cichlidae and crustacean from the Sondor Wetland (mg/kg *dry wt.* unless specified otherwise).

Elements	Sediment	Soil	Cichlidae	Crustacean	*Volta E. Ref.*	*STRV*
OM[k]	2.28 ± 1.95 (c)	1.82 ± 0.50 (c)	NA	NA	*ND*	*ND*
Al[k]	6.0 ± 1.6 (a)	6.2 ± 2.4 (a)	NA	NA	*4.2 ± 2.8*	*ND*
Fe[k]	3.0 ± 0.8 (b)	2.5 ± 0. 5 (b)	0.273 ± 0.182	0.047	*2.9 ± 1.3*	*ND*
Cr	66 ± 15 (c)	60 ± 20 (c)	62 ± 38	58.93	*36 ± 16*	*81*
Ni	25.63 ± 4.41 (c)	23.98 ± 7.11 (c)	BDL	BDL	*12.2 ± 6.6*	*20.9*
Cu	21.80 ± 5.25 (c)	22.09 ± 9.41 (c)	35.43 ± 41.46	98.10	*6.4 ± 4.3*	*34*
Zn	40 ± 8 (c)	79 ± 78 (c)	71 ± 13	177.00	*21 ± 10*	*150*
As	4.25 ± 1.18 (c)	3.70 ± 1.12 (c)	1.01 ± 0.75	0.66	*5.7 ± 3.0*	*8.2*
Cd	0.023 ± 0.013(c)	0.014 ± 0.008(c)	0.013 ± 0.006	0.015	*0.06 ± 0.12*	*1.2*
Pb	11.17 ± 2.42 (c)	11.09 ± 3.25 (c)	1.10 ± 0.70	BDL	*6.2 ± 3.6*	*46.7*
Hg	0.013 ± 0.009(c)	0.009 ± 0.005(c)	0.090 ± 0.042	0.027	*0.075 ± 0.057*	*0.15*

Classes a, b, c (in brackets) according to the LSD test at the 95% confidence level
Mean and standard deviation of the (sediment and soil) samples (n = 6)
Cichlidae = *Serotherodon melanotheron* and *Hemichromis fasciatus*
Crustacean = *Machrobrachium sp*
Volta E. Ref. = Volta Estuary Reference values, after Klubi et al. (2018)
STRV = Sediment Toxicity Reference Values, after Jones and Suter II (1997)
OM = organic matter
k = value in percentage
NA = Not analysed
ND = No data available
BDL = Below detection levels.

the IAEA Database of Natural Reference Materials (Rauch and Pacyna, 2009; Poulton and Raiswell, 2000; Plank and Langmuir, 1998). The Cr levels in the sediment, soil and fish samples could be described as uniformly distributed in the wetland. Elevated chromium concentrations are mainly associated with industrial activities. No commercial or industrial activities were observed within the study catchment area. As such, the relatively higher levels of Cr, and for the Songor Wetland, could be ascribed as geological in origin.

For copper (Cu), the concentrations in the sediment (21.80 ± 5.25 mg/kg) and the soil (22.09 ± 9.41 mg/kg) were within the Global mean level (21 mg/kg) for loess as cited by Rauch and Pacyna (2009) and Poulton and Raiswell (2000). The levels in the sediment and the soil samples were, however, lower than the concentrations obtained for the fish samples (Table 3). Copper is naturally abundant in the earth's crustal, and forms complexes with other elements. It is an essential metal for physiological processes and is abundant in plants, molluscs, and arthropods. As such, higher levels of Cu in the fish samples than in the sediment and the soil (Table 3) were expected. However, there were further discrepancies in the levels of Cu in the different fish species. For the Cichlidae, *S. melanotheron* Cu levels were between 8.88 and 14.2 mg/kg and its concentration in *H. fasciatus* was 83.2 mg/kg. For the crustacean (*Macrobrachium sp.*), Cu level was 98.1 mg/kg. This might be demonstrative of a bio-magnification of Cu levels in fish species as being niche-specific. *Hemichromis fasciatus* is a carnivore (piscivore) whereas *Macrobrachium sp.* feed on plankton and *S. melanotheron* is an omnivore (Oyekanmi et al., 2017). Additionally, reported levels of Cu in invertebrates are in-part due to its usage in hemocyanin, an oxygen transporter. This may also account for concentrations of Cu in the *Macrobrachium sp.* being higher than in the finfish species.

Zinc (Zn) levels (79 ± 78 mg/kg) in the soil samples showed large variabilities and suggests likely anthropogenic pollution of localized impacts. On the other hand, Zn levels in sediment samples (40 ± 8 mg/kg) were relatively similar across the sampling spots, and they compared well with the global mean concentration (48 mg/kg) for soils, as cited by Rauch and Pacyna (2009) in Han et al. (2002). For the fish samples, Zn levels could be a source of public concern, since high concentrations of Zn are believed to be toxic for fish species (Alabaster and Lloyd, 1980).

Nickel (Ni) levels were relatively uniform in soil (23.98 ± 7.11 mg/kg) and sediment (25.63 ± 4.41 mg/kg) samples. These levels

are comparable to the global mean concentrations in loess (27 mg/kg) and soils (25 mg/kg), as reported in the IAEA Database of Natural Reference Materials (Rauch and Pacyna, 2009; Poulton and Raiswell, 2000).

For lead (Pb), the levels in sediment (11.17 ± 2.42 mg/kg) and soil (11.09 ± 3.25 mg/kg) samples were within the global mean concentrations of the bulk continental crust (11 mg/kg) and loess [13 mg/kg] (Rauch and Pacyna, 2009; Rudnick and Gao, 2003; Poulton and Raiswell, 2000). These levels are lower than the STRV but higher than the local reference values reported for the Volta Estuary [Table 3] (Klubi et al., 2018; Jones and Suter II, 1997).

Arsenic (As) levels (Table 3) were similar in the sediment and soil samples, and they compared well with the local reference of the Volta Estuary. Occurrences of As in soils are generally geological in origin, with higher background levels found in clayey soils (Ursitti et al., 2004). Although anthropogenic sources of As have been documented (ATSDR, 1999), present results suggest geological fingerprints.

Concerning their spatial distribution across the wetland, there were overall higher metal concentrations at the extreme western and the eastern sections. The inter- and intra-comparisons will be considered in Section 3.4 by using the Pearson Correlation Coefficient and the Principal Component Analysis (PCA).

For studying their spatial distributions, the concentrations of the elements in soils and sediments were standardized by dividing each raw data value by its respective mean value for the whole set of the samples. This was to achieve an overall unit and do away with the scale differences. The standardized concentrations for the different sampling points are presented in Fig. 2, which shows different patterns of metal distributions within the wetland. The element concentrations in the soil demonstrated the highest forms of disparities compared to the sediment. For example, Zn displayed peak levels at locations AD-1L-SL and AD-1U-SL (Fig. 2A), which are close to the Songor Lagoon, and they might have been impacted by anthropogenic activities such as excavation and dyke constructions for salt mining (Dankwa et al., 2004). The standardized concentrations of Hg, Cd, and OM in soils, however, increased towards the eastern section of the study area (Fig. 2A). This suggests possible anthropogenic impacts, especially at locations AD-8U and AD-8L that are close to the Big Ada community.

In sediments, the standardized concentrations of Cd, Hg, and OM showed a common trend of increase from the west to the east

in the wetland (Fig. 2B). This is also an indication of the continuous inflows and depositions of organic matter with accompanying Cd and Hg, likely conveyed by the rising tides from the Big Ada community into the wetland. The other metals showed a relatively uniform spatial pattern of distributions in sediments, with a relative minimum at location AD-3-SD and a maximum at AD-4-SD (Fig. 2B). This last can be attributable to gross contamination from the Toku and Wassakuse communities, located west and east of the sampling point, respectively (Fig. 1).

3.3. Pollution indices

3.3.1. Enrichment factor and geoaccumulation index

The enrichment factor (EF), geoaccumulation index (I_{geo}), Contamination Factor (CF) and the Pollution Load Index (PLI) were estimated, using the upper earth crustal values as reference background, following the same principles as Turekian and Wedepohl (1961).

The mean, minimum and maximum I_{geo} values for Al, Fe, Cr, Ni, Cu, Zn, As, Cd, Pb, and Hg were mostly less than zero (Igeo < 0) [Table 4] for sediments and soils. Thus, the use of upper-crustal values indicated that the levels of the studied metals in the wetland were below the background concentrations. Although I_{geo} index values were relatively higher in sediments (Table 4), there were no issues for public concern.

The use of Al as a normalizing element for the chemical tracing of Al-silicates, particularly the clay minerals (Loring and Rantala, 1992) showed general higher scores of EF (Al) in soils than in sediments (Table 4). The EF (Al) values ranged between 0.12 ± 0.08 (Cd) and 0.86 ± 0.97 (Zn) for soils and between 0.14 ± 0.10 (Hg) and 0.53 ± 0.11 (Fe) for sediments (Table 4). Nevertheless, EF (Al) was below 2.0 in all cases, which can be interpreted as the absence of pollution or only a minimal enrichment (Sutherland, 2000).

The use of iron (Fe) as a normalizing element for the chemical tracing of Fe-rich clay minerals (Loring and Rantala, 1992) showed general higher scores than the use of aluminium [Table 4]. This is partially due to the relatively higher Al concentrations in the background reference and the samples (Tables 3 and 4). The general scores of the EF (Fe) were higher in soils than in sediments (Table 4). The EF (Fe) values in sediments ranged from 0.36 ± 0.28 for Hg to 1.18 ± 0.21 for As, while in soils they ranged from 0.25 ± 0.15 for Cd to 2.26 ± 0.18 for Al (Table 4). Most of the EF (Fe) scores were below 2.0, suggesting no to minimal enrichment (Sutherland, 2000). According to Zhang and Liu (2002), EF scores between 0.5 and 1.5 suggest that the metals are from crustal materials or natural processes, and scores greater than 1.5 could be more of anthropogenic origin. With this last criterion, the levels of Al and Zn could be due to human impacts.

Additionally, the anthropogenic impacts on the soil are likely to be localized compared to sediment. The focus is on sediments because of continuous interactions between the waters, biota, and the exhibition of a complex but stable system. Wetland geochemistry is considered multifaceted and largely controlled by changes in the chemistry of the available elements. The sequestering of metals into the sediment beds in the wetlands also varies depending on the availability of iron or sulphur. As such, the receptor and donors of electrons switch to favour the prevailing conditions to achieve an overall precipitation of metals into the sediment, unlike the soil which is limited by biogeochemical interaction and processes.

3.3.2. Pollution Load Index (PLI) and Contamination factors (CF)

The metal pollution based on the contamination factor (CF) and the pollution load index (PLI) shows an absence or moderate contamination of the Songor wetland (Figs. 3 and 4).

The computed CF for the studied metals has values between 0.25 ± 0.15 (Hg) and 1.47 ± 0.39 (Al) [Fig. 3] for the sediment samples, indicating low to minimal enrichment [CF < 1.5] (Muzerengi, 2013), except a moderate enrichment for aluminium (Al). The CF were in the order Al > As > Cu > Fe > Cr > Pb > Zn > Ni > Cd > Hg for the sediment samples.

For soils, the CF scores ranged from 0.15 ± 0.09 (Cd) to 1.50 ± 0.58 (Al) [Fig. 3]. Again, these CF values confirmed low to minimal enrichment (CF < 1.5), except for Al, which showed a moderate enrichment. The CF values were in the order Al > Zn > Cu > As > Cr ≥ Fe ≥ Pb > Ni > Hg > Cd for the soil samples. The dissimilarities in the order of the metal enrichments in the sediment and the soil could be attributable to natural and human activities. According to Dankwa et al. (2004), there are sections of the Songor Wetland that suffer a limited supply of water. As such, the sediment and soil chemistry are likely to vary due to differences in inundation periods. The anthropogenic impacts are the salt extraction activities at the western section and domestic and sewage-related contaminations at the central and eastern sections of the study area.

The PLI estimated based upon the CF results showed higher values for sediments than for soils [Fig. 4A and B]. The PLI values for sediments ranged from 0.34 (AD-3-SD) to 0.74 (AD-11-SD) [Fig. 4A], with an arithmetic mean of 0.59 ± 0.15. The highest PLI values were found at locations AD-4-SD (0.72) and AD-11-SD (0.74), and they could be attributed to anthropogenic impacts. Thus, site AD-4-SD is located between two communities, Toku and Wasakuse, while site AD-11-SD is closer to the Big-Ada community. Nevertheless, the PLI values were generally less than unity (PLI < 1), and thus the sediments can be considered as uncontaminated.

For soils, PLI values were between 0.34 (AD-2L-SL) and 0.63 (AD-1L-SL) [Fig. 4B], with an arithmetic mean of 0.53 ± 0.12 across the study area. The relatively higher values of PLI at locations AD-1L-SL and AD-1U-SL (0.63 and 0.55) and AD-8U-SL and AD-8L-SL (0.63 and 0.59) reflected some anthropogenic impacts. Locations AD-1L-SL and AD-1U-SL, at the western end of the study area, were reported as salt extraction areas (Dankwa et al., 2004). Generally, the PLI values were less than unity (PLI < 1) and the soils can also be described as uncontaminated.

3.4. Multivariate analysis

3.4.1. Pearson correlation coefficient

The Pearson correlation coefficient matrices based on complete-linkage for the organic matter (OM) and the metals are reported in Table 5a (for sediments) and Table 5b (for soils). The range of correlations (poor to weak, moderate and strong) suggested diverse potential sources of metal contaminations within the wetland. This is in agreement with results from I_{geo}, EF, and PLI, as above discussed.

The levels of the studied elements in the sediments showed two groupings with internal strong positive correlations: (i) the group of {Al, Fe, Cr, Ni, Cu, Zn, Pb} (**0.68 ≤ r ≤ 0.96**, p < 0.01), (ii) the group of {Cd, Hg, OM} (Table 5a). The element As correlated with those of group (i), but with r > **0.7** only for Fe and Zn. The close association of group (i) could be an indicator of lithogenic or inorganic complexing. According to Ghrefat and Yusuf (2006), the distinct behaviour of the elements of group (ii) is an indication of anthropogenic or different natural sources. Indeed, the strong positive correlations among Cd, Hg, and OM in sediments (Table 5a), could be attributed to an organic material-based association (Tanner and Leong, 1995) and, therefore, it

Table 4
Reference levels (mg/kg) of the Earth's crust, and the calculated geoaccumulation index (I_{geo}) and enrichment factors (EF) estimated for the two normalization elements (Al and Fe) for the studied analytes in the sediment and the soil samples.

Element	Ref. Earth Crust	Igeo		EF (Al)		EF (Fe)	
		Sediment	Soil	Sediment	Soil	Sediment	Soil
Al	4.1¥	−0.08 (−0.64,0.32)	−0.11(−1.20, 0.62)			1.97 ± 0.41	2.26 ± 0.58
Fe	3.9¥	−1.03 (−1.42,−0.44)	−1.24 (−1.65,−0.93)	0.53 ± 0.11	0.47 ± 0.14		
Cr	92	−1.11 (−1.69,−0.80)	−1.29 (−2.27,−0.72)	0.38 ± 0.09	0.44 ± 0.03	0.96 ± 0.21	0.99 ± 0.22
Ni	47	−1.48 (−1.88,−1.24)	−1.61 (−2.25,−1.04)	0.29 ± 0.05	0.36 ± 0.07	0.74 ± 0.12	0.78 ± 0.12
Cu	28	−0.99 (−1.69,−0.70)	−1.06 (−2.19,−0.27)	0.42 ± 0.10	0.52 ± 0.05	1.06 ± 0.32	1.18 ± 0.34
Zn	67	−1.36 (−1.90,−1.04)	−0.83 (−2.13,1.16)	0.32 ± 0.07	0.86 ± 0.97	0.80 ± 0.14	2.04 ± 2.49
As	4.8	−0.81 (−0.26, 1.47,)	−1.02 (−1.65,−0.56)	0.47 ± 0.13	0.54 ± 0.10	1.18 ± 0.21	1.18 ± 0.18
Cd	0.09	−2.77 (−4.08,−1.83)	−3.53 (−4.75,−2.72)	0.14 ± 0.08	0.12 ± 0.08	0.37 ± 0.23	0.25 ± 0.15
Pb	17	−1.22 (−1.82,−0.87)	−1.26 (−1.97,−0.78)	0.35 ± 0.08	0.46 ± 0.09	0.88 ± 0.15	1.01 ± 0.22
Hg	0.05	−3.15 (−5.23,−1.71)	−3.38 (−4.64,−2.23)	0.14 ± 0.10	0.14 ± 0.12	0.36 ± 0.28	0.29 ± 0.18

Upper crust metal levels after Rudnick and Gao (2003).
¥ levels in percentages
For *Igeo* mean values and range (in brackets) are reported, while for EF reported values are the mean and standard deviation of the mean.
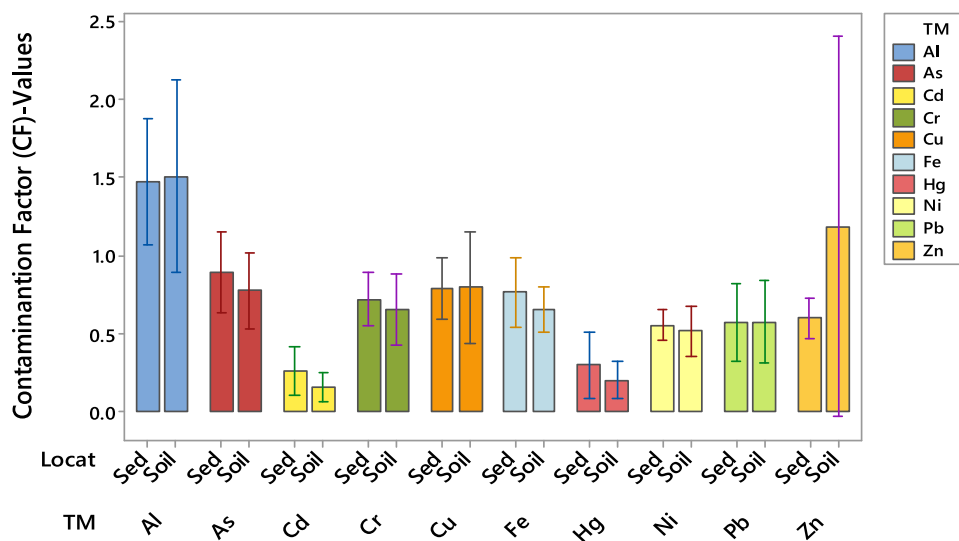

Fig. 3. A plot of contamination factor (CF) values for the studied elements in the sediment and soil samples of the Songor Wetland, Locat; sample type, TM: trace metal.
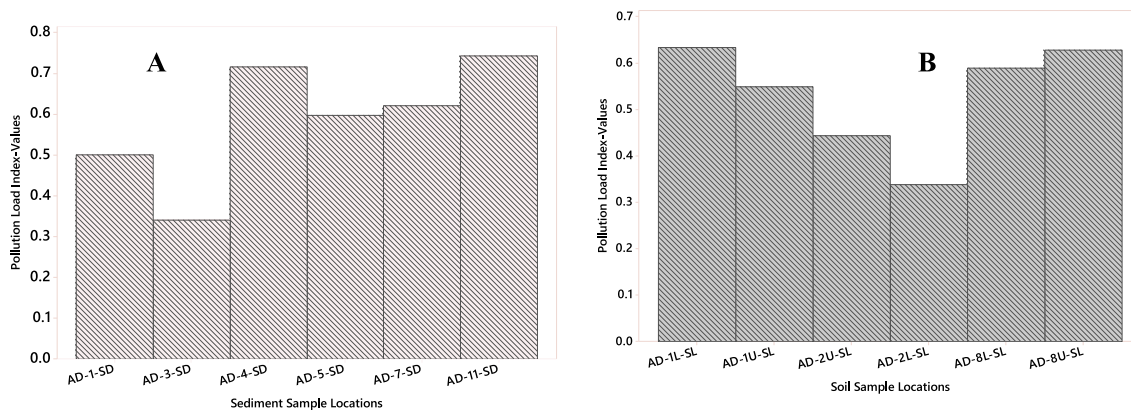

Fig. 4. Plots of the pollution load index (PLI) at the sampling sites for sediments (**A**) and soils (**B**), respectively.

displays biogenic or anthropogenic impacts within the wetland. This will be further supported by the PCA approach.

Concerning Arsenic (As), the present results agree with the findings of Klubi et al. (2018). The observations could be attributed to the possible occurrences of sulphur and their coexistence with As and Fe known as arsenopyrites. This confirms a lithogenic fingerprint of the geological formation of the wetland basin.

The soil samples exhibited strong and significant correlations, but different from those found in the sediment samples (Tables 5a and 5b). Here, the above group (i) includes As but excludes Zn, while group (ii) excludes Cd. Aluminium (Al) correlated strongly (**0.84 ≤ r ≤ 0.99,** $p < 0.01$) with Cr, Cu, Ni, As, Pb and Fe (Table 5b). Aluminium (Al) and Fe are major earth metals; they are considered conservative elements and their levels are not usually affected by anthropogenic activities (Jin et al., 2017).

Table 5a
Pearson correlation coefficient matrix based on complete-linkage for OM and the studied elements in the sediments of the Songor Wetland, Ada, Ghana; $p < 0.05$.

	OM	Al	Fe	Cr	Ni	Cu	Zn	As	Cd	Pb	Hg
OM	1.00										
Al	−0.16	**1.00**									
Fe	−0.26	**0.74**	1.00								
Cr	0.06	**0.95**	**0.68**	1.00							
Ni	0.07	**0.95**	**0.79**	**0.97**	1.00						
Cu	0.32	**0.74**	0.46	**0.88**	**0.79**	1.00					
Zn	0.44	**0.66**	**0.71**	**0.76**	**0.86**	**0.66**	1.00				
As	0.11	0.41	**0.82**	0.51	0.59	0.51	**0.74**	1.00			
Cd	**0.91**	−0.29	−0.24	−0.02	−0.04	0.35	0.35	0.29	1.00		
Pb	0.37	**0.71**	**0.77**	**0.82**	**0.87**	**0.82**	**0.94**	**0.83**	0.36	1.00	
Hg	**0.89**	−0.18	−0.12	0.09	0.07	0.45	0.44	0.40	**0.99**	0.47	1.00

OM = Organic matter
Boldface indicates strong or statistically significant correlation.

Therefore, their associations with metals of group (i) could be of natural origin and not the result of biogenic or anthropogenic effects. However, the strong correlation (r = 0.81) between OM and Hg could be attributed to vehicle emissions as well as the atmospheric transportation and fallout of fly ashes (Pirrone et al., 2001; Xu et al., 2014). Mercury (Hg) is a volatile metal and it is easily transported into the atmosphere, where it can be absorbed by aerosols, and then deposited on the surface, or absorbed by organic matter. The occurrences of high levels of Hg within the Ghanaian coastal sediments and soils were attributed to artisanal gold mining activities which are restricted to the western coast (Asare-Donkor and Adimado, 2016; Bortey-Sam et al., 2015). This, therefore, placed the association of Hg and OM in the soils of the Songor Wetland as a remote source and more likely attributed to vehicles emissions and atmospheric deposition. Zinc and Cd were poorly and negatively correlated with most of the studied elements in the soils (Fig. 5b). The occurrence of Zn levels in the soils was patchy and therefore a possible indication of anthropogenic impacts. This will be considered further using the PCA approach in Section 3.4.2. However, the overall correlations were comparable to the findings of Donkor et al. (2005), Klubi et al. (2018) and Jin et al. (2017), which uses similar approaches for the assessment of provenance of metals in sediments and soils, respectively.

3.4.2. Principal Component Analysis (PCA)

Principal component analysis (PCA) is an algorithm that reduces complex data into linear combinations yet retaining the basic variations in the dataset. Further information on the metal levels in the sediments and the soils of the wetland was ascertained by standardization and normalization of the raw data before applying the PCA. The standardization reduces the influences of outliers and equates the average to zero and the standard deviation to 1, while the normalization centres the data between −2 to +2 (Ringnér, 2008). This is to do away with the scale differences and skewness of the dataset by outliers.

Results of the PCA show a clear grouping of the sample sites and the metals (Fig. 5). It can be inferred without consulting the single value decomposition (SVD), that Zn has a higher bearing on the western section (AD-1L-SL, AD-1-SD, AD-2U-SL, and AD-3-SD) while OM, Hg, and Cd are remarkable at the eastern section (AD-11-SD and AD-7-SD). The overlayed cluster plot and the contour marks (14 distance) on the PCA show that the majority of the metals analysed were concentrated at sampling sites AD-1L-SL, AD-4-SD, AD-1-SD, AD-8L-SL and AD-11-SD (Fig. 5), with overall negative PC1, and accounting for 53% of the total variability (Fig. 6e). PC2 contributed 24.7% of the total variability and it was dominated by the OM, Cd, and Hg (Fig. 6b). It is negatively loaded by sites AD-11-SD and AD-7-SD (eastern section of the study area). Thus, the occurrence of Cd and Hg could be associated with biogenic or anthropogenic impacts emanating from the Big Ada community. Principal component (PC) 3 formed 10.7% of the total variability, dominated by Zn (Fig. 6c), and positively loaded by AD-1L-SL, AD-1-SD, AD-2U-SL and AD-3-SD (western section of the study area). The high levels of the occurrences of Zn at the western section of the study area could be attributed to an anthropogenic activity like salt extraction according to Dankwa et al. (2004). Principal component (PC) 4 and PC5 accounted for 5.2% and 2.5% of the total variability, respectively. Although PC4 and PC5 are both negatively and positively loaded by the other metals (Fig. 6d and e), they are less significant and their eigenvalues were less than one (Ghrefat and Yusuf, 2006).

4. Conclusions

In summary, sediment, soil and fish samples were collected from the Songor Wetland, South-eastern Ghana, and investigated for the spatial distributions of metal concentrations and pollution loads for assessment of natural and anthropogenic impacts. The occurrence of metal concentrations was in the order Al > Fe > Cr > Zn > Ni > Cu > Pb > As > Cd > Hg in sediments; and Al > Fe > Zn > Cr > Ni > Cu > Pb > As > Cd > Hg in soils. The levels of the studied metals in the fish were in the order Fe > Zn > Cr > Cu > Pb > As > Cd > Hg for Cichlidae, and Fe > Zn > Cu > Cr > As > Hg > Cd for crustaceans. The levels of the studied metals in sediments and soils indicated important spatial variability with local sources within the wetland. Cichlidae exhibited slightly high levels of selective metal accumulation when compared to the crustaceans. The contamination assessments indicate that both sediment and soil samples were not polluted. The enrichment factor (EF) indicated that Al-rich and Fe-rich minerals

Table 5b
Pearson correlation coefficient matrix based on complete-linkage for OM and the studied elements in the soils of the Songor Wetland, Ada, Ghana; $p < 0.05$.

	OM	Al	Fe	Cr	Ni	Cu	Zn	As	Cd	Pb	Hg
OM	1.00										
Al	0.19	1.00									
Fe	0.11	**0.84**	1.00								
Cr	0.24	**0.99**	**0.83**	1.00							
Ni	0.08	**0.96**	**0.86**	**0.92**	1.00						
Cu	0.30	**0.97**	**0.84**	**0.98**	**0.89**	1.00					
Zn	−0.48	0.06	−0.35	0.05	−0.03	0.02	1.00				
As	0.44	**0.95**	**0.87**	**0.96**	**0.88**	**0.96**	−0.18	1.00			
Cd	−0.05	−0.31	−0.12	−0.22	−0.50	−0.13	0.05	−0.21	1.00		
Pb	0.35	**0.91**	**0.70**	**0.94**	**0.77**	**0.97**	0.17	**0.91**	0.00	1.00	
Hg	**0.81**	−0.19	−0.32	−0.12	−0.29	−0.15	−0.31	0.03	−0.05	−0.05	1.00

OM = Organic matter
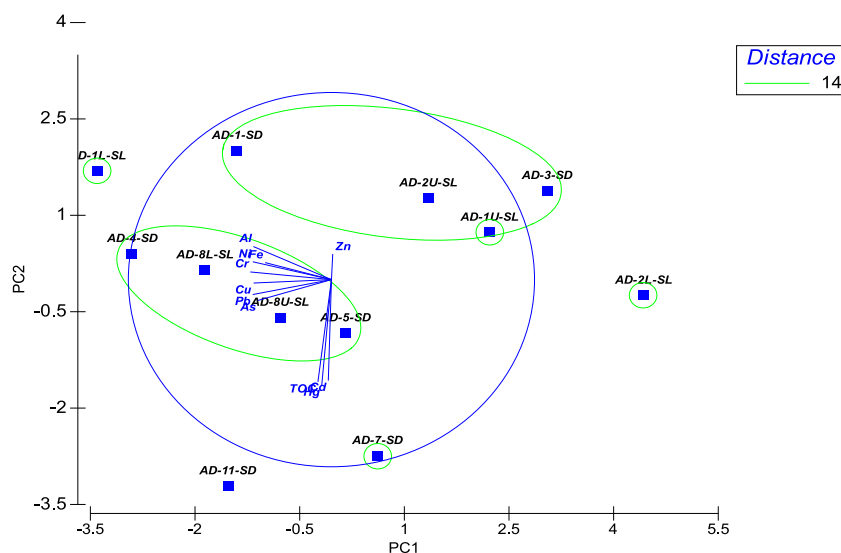Boldface indicates strong or statistically significant correlations.

Fig. 5. Principal component analysis (PCA) plot of metal occurrence with sampling sites.
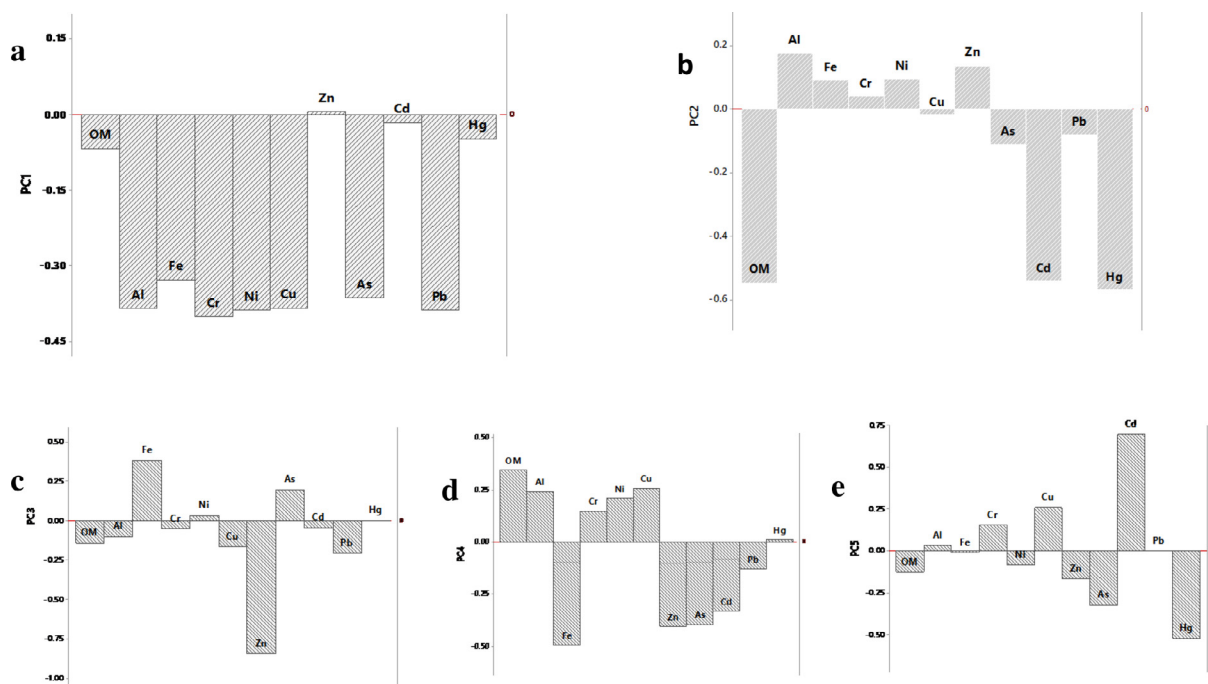

Fig. 6. Result of principal components (PC) for PC1 (a), PC2 (b), PC3 (c), PC4 (d) and PC5 (e) for the studied elements in the sediments and soils of the Songor Wetland.

were more enriched in the soil than in the sediment samples. The overall enrichment ([EF {Al} and EF {Fe}]) showed low to minimal enrichments except for Al and Zn, which were moderately enriched in the soils and the sediments, respectively. The contamination factor suggested moderate contamination for Al and Zn in both sediment and soil samples, whereas pollution load indices suggested that contamination by the studied metals were below the baseline levels when using the reference values of the upper crustal after Rudnick and Gao (2003). Pearson correlation coefficient indicated that the metal loads of the Songor Wetland were mostly lithogenic with some biogenic and anthropogenic fractions. Principal component analysis (PCA) indicated that the western section of the wetland was contaminated with Zn, while Cd and Hg were concentrated at the central and eastern sections. Lead (Pb) levels in fish species exceeded the EU Regulation

1881/2006/EU for fish tissues and could pose a public health concern.

CRediT authorship contribution statement

Emmanuel Klubi: Data collection and analysis of the data with the relevant authority laboratories and drafting of the body of the draft paper, Pollution indices analysis and multivariate statistical analysis and the provenance and geochemistry assessment. **Dennis Kpakpo Adotey:** Assessment in checking the chemistry of trace metal behaviour in the matrixes analysis, Additional supervisor and grammatical correlation of the English usage in the paper. **Samuel Addo:** Supervisor and the proof reading of the manuscript at every stage of the progress of the paper. **José M. Abril:** Responsible for and ensuring that all the method and

the statistics were carefully followed, Recheck and enhancing the introduction and the quality assurance analysis as well as checking the correct usage of the English language, Support with additional literature review and making available of other articles to support the technical findings of the research.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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