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RESEARCH ARTICLE

Assessment of heavy metal contamination in *Hediste diversicolor* (O.F. Müller, 1776), *Mugil cephalus* (Linnaeus, 1758), and surface sediments of Bafa Lake (Eastern Aegean)

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Abstract In the present study, the bioaccumulation of six heavy metals (Cd, Cr, Cu, Hg, Pb, and Zn) in *Hediste* (*Nereis*) *diversicolor* (O.F. Müller, 1776) and also in the muscle and liver of *Mugil cephalus* (Linnaeus, 1758) collected from seven stations in the Bafa Lake was investigated. Sediment samples were also collected in each site to assess heavy metal levels and to provide additional information on pollution of the lake. The mean concentrations of heavy metals in sediment, *H. diversicolor*, and muscle and liver of the fish were found to be in the magnitude of Cr>Pb>Zn>Cu>Cd>Hg, Zn>Cu>Cr>Pb>Hg>Cd, Zn>Cu>Pb>Cr>Hg>Cd, and Cu>Zn>Cr>Cd>Pb>Hg, respectively. Hg, Cu, and Zn in *H. diversicolor* and Hg and Zn in muscle and also Hg, Cd, Cu, and Zn in liver of fish accumulated in a higher degree than in sediment. There was no clear relationship between metal concentrations in sediments, polychaetes, and fish, except Cr. According to international criteria and Turkish regulations, Pb and Zn values in edible muscle of the fish collected from stations S6 and S5 exceeded the food safety limits, respectively. The results of this study suggest that these sentinel species can be considered as good anthropogenic biological indicators for heavy metal pollution along the Bafa Lake.

Keywords *Hediste* (*Nereis*) *diversicolor* · *Mugil cephalus* · Sediments · Heavy metal · Bafa Lake (Eastern Aegean)

Introduction

An excessive increase in pollution of lakes and rivers, originating from domestic wastewaters and agricultural and industrial effluents, is a matter of great concern throughout the world. Among the various pollutants, heavy metals pose a serious risk for the environment and human, and they are persistent, bioaccumulative, and toxic contaminants in aquatic habitats.

During the last few decades, deposits of anthropogenic origin have raised the heavy metal levels and accelerated the risk of irreversible pollution in lakes. The study of the spatio-temporal variation of heavy metals in lake sediments and aquatic biota is of great importance because of its toxicity, bioaccumulation processes, and biomagnification in the food chain (Gochfeld 2003).

Bafa is a brackish residual lake which was formed as the result of delta progradation of Büyük Menderes (Maeander) River and is located in the southeastern part of the Büyük Menderes Delta in the Aegean Region of Turkey.

This ecosystem has great importance with regards to its biodiversity. In addition, the lake offers a suitable habitat for breeding and nursing ground for fish and birds. Bafa Lake Natural Park gained its protection status in July 8, 1994 as being a wetland with national significance. It is one of Turkey's most important protected coastal lakes. Moreover, it is being considered as a "Ramsar Site" by The International Union for the Conservation of Nature and Natural Resources (IUCN) (Gürer and Yıldız 2008). The lake receives urban and industrial wastes from multiple sources. Furthermore, it is subject to environmental disturbance caused by human activities, and this area is prone to contamination.

Lake sediments act as a trap for heavy metals originated from anthropogenic disturbance, and these toxic metals may be taken up by deposit-feeding invertebrates ingesting the sediment (Schropp and Windrom 1988). The study of metal concentrations in several indicator species with different life patterns and at different levels in the trophic chain can give more reliable information on the bioavailability (Cajaraville et al. 2000).

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A search on published literature reveals that information on the bioavailability of heavy metals in sediments, and their accumulation in organisms distributed along the Bafa Lake is scanty. In this regard, we selected the sediment-dwelling rag worm *Hediste* (*Nereis*) *diversicolor* (O.F. Müller 1776) and *Mugil cephalus* (Linnaeus 1758) inhabiting in Bafa Lake as our study organisms.

Deposit and filter feeder *H. diversicolor* and edible fish species *M. cephalus* are brackish and euryhaline ecologically sentinel species (Diéz et al. 2000; Yilmaz 2009). The prominent role of these rag worms functioning as an important bioturbator, as a major component of the benthic biomass of estuarine mudflats, and in remobilizing and transfer of contaminants in the food chain for crustaceans, fish, and birds (Durou et al. 2007) makes them crucial species in biomonitoring studies. This burrowing species has an ability to inhabit metal-rich sediments and pollutants (Moreira et al. 2006). Because of the characteristics of *H. diversicolor*, it was proposed as an efficient indicator species of metal contamination in sediments (Saiz-Salinas and Francés-Zubillaga 1997; Scaps 2002). In addition, *M. cephalus* is considered as one of the fish species which the local population commonly uses for consumption and also a good indicator of heavy metal pollution (Yilmaz 2009).

In the course of this study, we compared the spatio-temporal distribution of metal contamination in the surface sediments of seven stations selected from Bafa Lake and verify the heavy metal levels (Cd, Cr, Cu, Hg, Pb, and Zn) in tissues of different biota samples such as *H. diversicolor* and *M. cephalus* in terms of bioaccumulation. In addition, relationships between metal concentrations, sediment and physicochemical variables, biota, and sediment are also investigated. Moreover, the purpose of the present study was to evaluate the food safety of the edible fish tissue of *M. cephalus*. The results of this study provide useful complementary data for environmental quality assessment and can be used by authorities for directing environmental monitoring and lake management.

Material and methods

Study area

The study area, Bafa Lake (between 37° 28′ N, 27° 22′ E and 37° 32′ N, 27° 32′ E) is situated within the boundaries of Milas district of Muğla Province and Aydın Province's Söke district. This lake has a total area of about 68.6 km^2, a catchment area of 315 km^2, and a shoreline length of 73 km, and its maximum depth changes seasonally between 20 and 23 m. Beginning in Late Antiquity (around the fifth century), the west part of the lake has acted as a sediment trap and the alluvial deposits of the Büyük Menderes River have silted up the entrance of the Latmos Gulf up until end of the Middle Ages. As a result of this, Bafa Lake finally lost its connection to the open sea in 50–300 AD (Kasparek 1988).

The lake is mainly fed by Büyük Menderes River. Moreover, some small-volume saline springwaters in some parts of the lake feed the lake. The lake was connected to the Aegean Sea through the Büyük Menderes River by means of a man-made channel, 3 km long. To control the lake's water level and salinity according to the flows of Büyük Menderes River, a flood prevention barrier and two associated regulators were built at the northwestern shore of the lake by State Hydraulic Works (DSI) in 1985.

Due to anthropogenic activities in the region, the lake may be affected directly or indirectly. The point and non-point sources of pollution are mainly from several raw sewage, streams, algae bloom, sediment transportation, untreated domestic, agricultural, and industrial outfalls (olive oil factory), and fishery.

Seven sampling stations having different contamination levels were chosen in shores of the Bafa Lake (Fig. 1). Stations S1 and S2 are situated in the eastern part of the lake and receive untreated urban and agricultural wastes. Stations S3 and S4 are located in the southern part of the lake, influenced by recreational activities and olive oil factories. Station S5 is found in the southwestern part of the lake. Station S6 is situated at the northwestern part of the lake and close to the inlet regulator receiving wastes derived from fish hatchery. Station S7 is located near Lake Serçin which is situated in the northwest area of the lake, and it receives urban wastes.

Sample collection

H. diversicolor, *M. cephalus*, and sediment samples were collected four times between August 2010 and July 2011. Moreover, the physicochemical variables of the lake such as temperature (°C), salinity (psu), pH, and electrical conductivity (EC, mS/cm) were measured in situ with Wissenschaftlich-Technische Werkstatten (WTW, WTW GmbH, Weilheim, Germany). Winkler method was used to measure dissolved oxygen (DO) (mg/l) (Strickland and Parsons 1972).

Due to constraints on *H. diversicolor* abundances, they were not found in all the selected stations and seasons. For this reason, individuals were collected at only four stations (S1, S3, S6, and S7) in September 2010, four stations (S2, S4, S6, and S7) in May 2011, and only two sampling stations (S3 and S6) in July 2011. In order to analyze heavy metals, 20–30 individuals of *H. diversicolor* in uniform size were carefully handpicked at each station. Then, they were immediately transported to laboratory in cool boxes. Specimens were examined and only whole, undamaged ones were chosen in the laboratory. Polychaeta species were identified by using stereomicroscopes. The polychaetes were placed for 24 h in clean, aerated seawater from the site of origin to allow them to

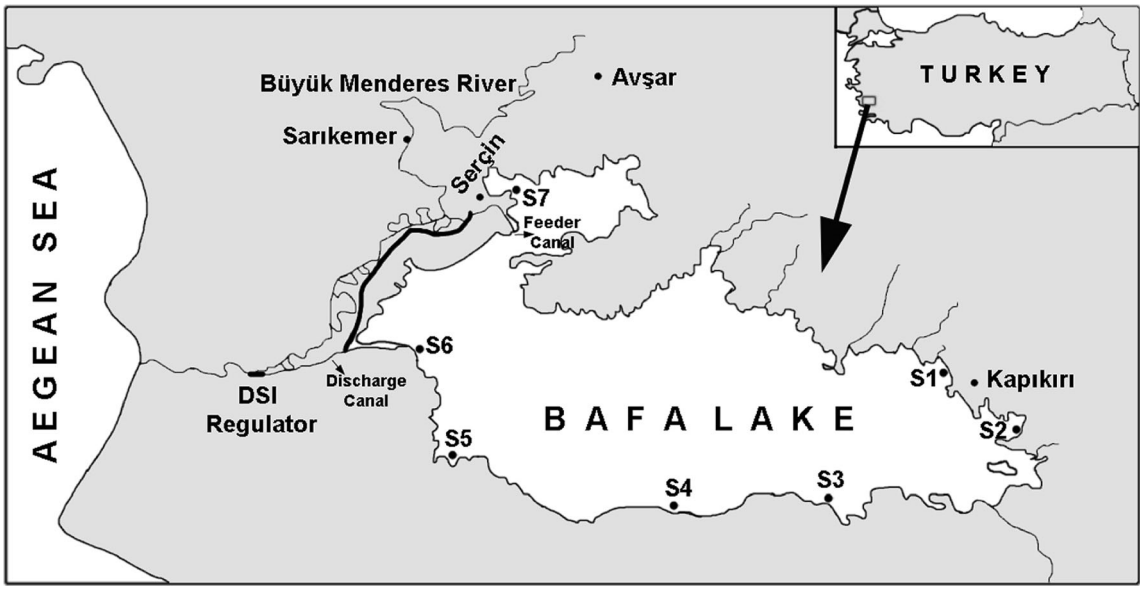


Fig. 1 Map showing locations of the stations and major tributaries in the Bafa Lake

purge of their gut contents. And then, samples were placed in polyethylene plastic bags and kept frozen prior to further analysis.

M. cephalus were caught from two stations in September 2010 (S5 and S7) and in May 2011 (S4 and S5), from four stations (S1, S4, S5, and S7) in February 2011, and from five stations (S3, S4, S5, S6, and S7) in July 2011. Eight individuals were caught in each station. Edible muscle tissue and liver of fish samples were dissected and weighed and preserved in polyethylene bags.

Sediment samples were also collected by using Van Veen grab from sampling stations for heavy metal analysis. Then, the samples were transported to the laboratory in polyethylene bags and all studied samples were stored at −20 °C until treatment.

Analytical procedure

Sampling and lab treatments were done using non-metal tools. Twenty-five polychaeta specimens were used to prepared a pooled sample to reduce individual variations in heavy metal concentrations and freeze-dried (−40 °C) (Labconco Freeze Dry Systems, Kansas City, USA) to constant weight, and then they were pulverized and homogenized in a teflon mortar. Approximately 0.5-g aliquots of dried samples were digested with 5:1 $HNO_3/HClO_4$ in a microwave digestion system (Milestone Inc., USA) (UNEP 1982, 1984, 1985a).

The muscle and liver of fish were pooled, homogenized separately, and then freeze-dried (−40 °C). Approximately 0.5 g of tissues was digested by microwave using nitric and perchloric acid (5:1) (UNEP 1982, 1984, 1985a).

Sediment samples were freeze-dried (−40 °C) to constant weight and homogenized and reduced to a fine powder by using a sieve (63 μm) for grain-size correction. For the analysis of total heavy metal in sediment, 0.1–0.2 g of finely powdered and dried sample was digested in microwave digestion system with a mixture of HNO_3-HF-$HClO_4$-HCl (UNEP 1985b, c, d, e).

Grain-size analysis was performed using standard sieving and settling procedures (Häkanson and Jansson 1983). The amount of organic matter (%) was determined spectrophotometrically in dried sediment samples following the sulfochromic oxidation method. The accuracy of this method is ±0.017 % organic matter (HACH Publication 3061 1988). All metal concentrations were expressed on a dry tissue weight basis. Residual water content of all samples was determined by oven drying (105 °C for 24 h).

All analyses were performed by flame, graphite furnace, and cold vapor technique in AAS (Varian AA280FS and Varian AA280Z Zeeman) by using the manufacturer's conditions and with Zeeman background correction (UNEP 1982; UNEP 1985a, b, c, d, e, f). The detection limits for heavy metals were as follows: Cd, 0.10 μg/kg; Cr, 0.06 μg/kg; Cu, 0.03 μg/kg; Hg, 0.05 μg/kg; Pb, 0.10 μg/kg; and Zn, 0.01 μg/kg. The quality assurance of metal analyses was checked using two certified reference materials, the IAEA-407 (biota) (μg/g dw) and IAEA-433 (sediment) (μg/g dw) (Table 1).

Statistical analysis

Normality and homogeneity of variances were tested using Kolmogorov-Smirnov and Levene tests respectively, and transformation of data was conducted when necessary. If data were found to be non-normally distributed, non-parametric tests were preferred for the comparisons of changes in metal

Table 1 Certified and observed values of elemental concentrations in reference materials IAEA-407 and IAEA-433 (mean±SD)

Metal	IAEA-407 (biota) (μg/g dw)		IAEA-433 (sediment) (μg/g dw)	
	Certified	Observed	Certified	Observed
Cd	0.189±0.019	0.187±0.003	0.153±0.033	0.140±0.025
Cr	0.73±0.22	0.75±0.040	136±10	136.7±2.0
Cu	3.28±0.40	3.52±0.18	30±2.6	30.8±2.3
Hg	0.222±0.024	0.231±0.014	0.168±0.017	0.167±0.012
Pb	0.12±0.06	0.137±0.006	26.0±2.7	27.0±3.1
Zn	67.1±3.8	65.5±0.7	101±80	103±1.9

Number of replicates: IAEA 407 (n=6), IAEA 433 (n=6)

dw dry weight

concentrations. One-way analysis of variance (ANOVA) (parametric data) and the Kruskal-Wallis test (non-parametric data) were applied to find any significant differences in metal concentrations among different stations and seasons. The Tukey (parametric data) and the Mann–Whitney U tests (non-parametric data) were used to discriminate significant differences.

Spearman correlation analysis was applied to verify existing relationships (between sediment metal concentrations and sediment metal levels with physicochemical variables, worm metal concentrations, muscle tissue of fish metal values, and liver of fish metal concentrations). The statistical analysis was performed using the STATISTICA v.7.1 (STATSOFT) software package, and statistical significance was defined at the $p<0.05$ level.

Biota-sediment accumulation factor (BSAF) is the ratio between the metal concentration in biota and that found in the sediment (Abdallah and Abdallah 2008). The formula is $BSAF=C_{org}/C_{sed}$, where C_{org} is the heavy metal concentration in the organism (μg/g dw) and C_{sed} is the heavy metal level in sediment (μg/g dw). BSAF was calculated for concentrations recorded for polychaetes and sediment, sediment and muscle, and sediment and liver of fish.

Results

Physicochemical variables

The physicochemical variables as well as biotic factors are known to influence the bioavailability of heavy metals. The results of physicochemical variables in water and also heavy metal concentrations, organic matter, and grain-size values in sediment evaluated in all sampling periods are displayed in Tables 2 and 3, respectively.

Concerning the physicochemical variables, temperature reached its minimum level (11.7 °C) at station S3 in February 2011, while its maximum value (32.0 °C) was observed at station S1 in September 2010 (Table 2). Furthermore, temperature did not vary significantly between sampling stations. However, it showed temporal variation during sampling periods ($p<0.001$), and measured values during February 2011 differed significantly from other seasons.

Recorded pH levels were rather consistent and presented slightly alkaline conditions in all stations. The measured levels varied between a narrow range of 8.0 at station S6 in September 2010 and 9.2 at station S7 in July 2011 (Table 2). Although, the spatial variation of pH levels ($p<0.001$) was significant, no significant temporal differences was detected among seasons. In addition, all pairwise comparisons showed that station S7 was significantly different from other stations.

Conversely, a broad range of values was observed for conductivity. Moreover, the highest conductivity level (28.1 mS/cm) was determined at station S7 in September 2010, and this concentration was four times greater than the lowest value (7.3 mS/cm) recorded at the same station in February 2011 (Table 2). On the other hand, no remarkable differences in all stations were observed for conductivity levels, while statistically significant seasonal differences were found among sampling periods ($p<0.01$). In September 2010, obtained values differed significantly compared to those on other sampling periods.

In terms of dissolved oxygen concentration, the minimum value (3.1 mg/l) was noticed at station S5 in July 2011 and reached its highest level (14.1 mg/l) at station S3 in May 2011 (Table 2). Furthermore, for this variable, no significant differences were detected among stations and seasons in the sampling area during the study period.

For measured salinity levels, there was a gradual decrease from September 2010 to February 2011. In addition, a drastic drop in salinity was observed at station S7 (4.0 psu) and station S6 (4.5 psu), especially in February 2011. Moreover, the highest value was recorded at station S7 (17.4 psu) in September 2010 (Table 2). Salinity levels differed significantly among seasons ($p<0.001$) when all sites were considered. However, they did not display differences among sampling sites.

Table 2 The physicochemical variables at each sampling sites

Stations	Periods	Temperature (°C)	Salinity (psu)	pH	Conductivity (mS/cm)	DO (mg/l)
S1	Sep 2010	32.0	14.2	8.4	23.2	6.5
	Feb 2011	14.1	12.4	8.5	20.9	10.7
	May 2011	27.2	12.6	8.4	21.0	9.7
	Jul 2011	25.5	12.1	8.3	20.3	10.8
	Mean±SE	24.7±3.8	12.8±0.5	8.4±0.03	21.4±5.3	9.4±1.0
S2	Sep 2010	28.5	14.7	8.6	23.9	9.3
	Feb 2011	15.7	12.2	8.5	20.6	12.8
	May 2011	31.4	13.0	8.4	21.3	10.3
	Jul 2011	25.3	12.3	8.3	20.5	7.4
	Mean±SE	25.2±3.4	13.1±0.6	8.4±0.05	21.6±5.4	10.0±1.1
S3	Sep 2010	28.6	14.1	8.7	23.2	11.9
	Feb 2011	11.7	12.5	8.4	21.1	6.7
	May 2011	25.1	12.6	8.6	21.0	14.1
	Jul 2011	30.8	12.0	8.7	19.6	11.8
	Mean±SE	24.1±4.3	12.8±0.5	8.6±0.08	21.2±5.3	11.1±1.6
S4	Sep 2010	28.7	14.0	8.4	23.0	8.8
	Feb 2011	12.1	12.3	8.4	20.9	8.6
	May 2011	23.0	12.2	8.3	20.2	9.6
	Jul 2011	30.3	12.1	8.6	19.8	11.7
	Mean±SE	23.5±4.1	12.7±0.5	8.4±0.07	21.0±5.2	9.7±0.7
S5	Sep 2010	24.5	14.3	8.2	23.7	11.3
	Feb 2011	12.4	12.2	8.1	20.7	10.6
	May 2011	22.3	12.4	8.2	20.8	6.1
	Jul 2011	30.2	12.2	8.3	19.9	3.1
	Mean±SE	22.4±3.7	12.8±0.5	8.2±0.05	21.3±5.3	7.8±1.9
S6	Sep 2010	26.8	16.3	8.0	26.6	3.7
	Feb 2011	14.6	11.3	8.5	19.2	12.2
	May 2011	25.6	12.4	8.5	20.6	10.7
	Jul 2011	31.0	4.5	8.7	18.1	6.2
	Mean±SE	24.5±3.5	11.1±2.5	8.4±0.15	21.1±5.3	8.2±2.0
S7	Sep 2010	29.7	17.4	9.1	28.1	6.9
	Feb 2011	12.6	4.0	9.1	7.3	13.0
	May 2011	29.4	8.9	9.1	15.1	10.2
	Jul 2011	27.9	6.5	9.2	11.3	4.3
	Mean±SE	24.9±4.1	9.2±2.9	9.1±0.03	15.4±3.9	8.6±1.9

Regarding organic matter (OM) (%) in sediment, measured values are quite similar across stations (0.11–4.7 %) (Table 3). The average concentrations of OM (%) in sampling stations exhibited the following order: S5>S6>S7>S2>S1>S3>S4. In addition, the stations situated in the west and south parts of the lake presented with relatively higher percentages of organic matter levels compared to stations located in the eastern area. No significant temporal variability was determined among seasons. Furthermore, the non-parametric Kruskal-Wallis test indicated that there were significant variations among stations ($p=0.0090$, $p<0.05$).

The textural distribution of sediments revealed that stations S1, S3, and S4 were primarily composed of sand (33.8–59.4 %, 35.1–86.2 %, and 43.1–65.8 %) and gravel (37.9–66.0 %, 9.3–57.0 %, and 30.0–52.9 %), respectively. Moreover, stations S2 and S5 were characterized by higher percentage of sand (62.4–71.5 % and 53.3–63.6 %) and silt+clay (15.6–33.1 % and 9.7–41.4 %), respectively. In addition, station S6 was covered by silty clay (fine fraction) (23.6–66.2 %) with moderate percentages of sand (26.2–47.5 %) and also station S7 was predominantly composed of silty clay (91.2–93.9 %) containing up to 64.5–69.3 % clay (Table 3). There were remarkable spatial differences for gravel ($p<0.01$), sand

Table 3 The heavy metal concentrations (μg/g dry weight; mean±*SE*), organic matter (%) and grain-size (%) values in sediments during sampling periods

Station	Periods	Cd	Cr	Cu	Hg	Pb	Zn	Org. Mat.	Gravel	Sand	Silt+clay
S1	Sep 2010	0.078	93.7	5.6	0.015	73.2	16.6	3.3	37.9	59.4	2.4
	Feb 2011	0.037	96.7	2.1	0.018	63.5	4.1	0.11	43.3	54.5	2.2
	May 2011	0.075	90.5	11.6	0.012	71.6	23.3	0.62	40.9	56.5	2.6
	Jul 2011	0.25	99.1	8.6	0.017	66.7	15.7	3.1	66.0	33.8	0.2
	Mean±SE	0.11±0.048	95±1.9	7.0±2.0	0.016±0.001	69±2.2	15±4.0	1.8±0.83	47.0±6.4	51.1±5.8	1.8±0.55
S2	Sep 2010	0.14	15.0	11.1	0.028	111.2	50.0	3.7	4.54	62.4	33.1
	Feb 2011	0.10	22.3	25.4	0.063	105.6	55.1	1.6	11.1	67.6	21.3
	May 2011	0.06	7.8	16.9	0.011	86.7	24.3	1.4	4.59	71.5	23.9
	Jul 2011	0.05	18.2	17.2	0.008	69.9	24.5	1.0	15.8	68.6	15.6
	Mean±SE	0.088±0.022	16±3.1	18±2.9	0.028±0.013	93±9.4	39±8.2	1.9±0.60	9.0±2.7	67.5±1.9	23.5±3.7
S3	Sep 2010	0.21	48.7	29.4	0.023	122.3	65.2	3.0	9.34	86.2	4.4
	Feb 2011	0.22	151.9	22.5	0.031	75.5	58.5	0.54	26.6	64.4	9.0
	May 2011	0.36	69.5	50.1	0.025	151.8	101.5	2.3	47.9	41.2	10.9
	Jul 2011	0.08	75.3	10.1	0.010	70.3	22.8	0.68	57.0	35.1	8.0
	Mean±SE	0.22±0.057	86±23	28±8.4	0.022±0.004	105±20	62±16	1.6±0.60	35.2±10.7	56.7±11.7	8.1±1.4
S4	Sep 2010	0.21	66.1	10.4	0.016	59.9	14.0	0.95	32.6	63.2	4.2
	Feb 2011	0.18	60.2	10.1	0.020	42.2	10.9	1.4	49.9	45.9	4.2
	May 2011	0.18	61.5	37.3	0.092	72.7	27.0	1.0	30.0	65.8	4.2
	Jul 2011	0.17	41.4	36.1	0.013	64.9	79.3	1.9	52.9	43.1	4.0
	Mean±SE	0.18±0.009	57±5.5	24±7.6	0.035±0.019	60±6.5	33±15.9	1.3±0.22	41.4±5.9	54.5±5.8	4.2±0.05
S5	Sep 2010	0.25	97.6	14.0	0.023	107.5	45.5	4.0	26.7	63.6	9.7
	Feb 2011	0.38	87.1	13.5	0.029	103.5	39.5	4.3	6.1	55.2	38.8
	May 2011	0.22	69.6	13.1	0.011	90.3	39.9	4.7	26.2	55.2	18.6
	Jul 2011	0.22	122.0	14.8	0.017	95.0	40.9	3.7	5.4	53.3	41.4
	Mean±SE	0.27±0.037	94±11	14±0.4	0.020±0.004	99±3.9	42±1.4	4.2±0.21	16.1±6.0	56.8±2.3	27.1±7.7
S6	Sep 2010	0.18	231.2	25.2	0.027	45.2	64.7	3.7	31.9	40.0	28.1
	Feb 2011	0.34	47.5	45.5	0.045	136.9	74.5	3.7	7.6	26.2	66.2
	May 2011	0.16	201.6	23.9	0.023	110.6	68.1	4.0	35.8	40.6	23.6
	Jul 2011	0.22	202.2	17.9	0.026	87.8	45.1	1.4	1.5	47.5	51.0
	Mean±SE	0.22±0.042	171±42	28±6.0	0.030±0.005	95±19	63±6.3	3.2±0.62	19.2±8.6	38.6±4.5	42.2±10.0
S7	Sep 2010	0.22	220.8	26.7	0.035	101.0	71.0	3.3	0.06	6.7	93.2
	Feb 2011	0.20	272.9	31.1	0.037	83.9	83.6	3.5	0.44	7.2	92.4
	May 2011	0.18	235.5	27.3	0.017	94.5	69.8	2.0	0.03	6.0	93.9
	Jul 2011	0.14	199.9	26.6	0.025	94.1	63.7	3.7	0.17	8.7	91.2
	Mean±SE	0.19±0.017	232±15	28±1.1	0.029±0.005	93±3.5	72±4.2	3.1±0.38	0.18±0.09	7.2±0.55	92.7±0.59

($p<0.05$), and clay-silt fractions ($p<0.001$). Furthermore, no significant seasonal differences were determined.

Heavy metal concentrations in sediments

Based on the results of this study, the contents of investigated heavy metals in sediment were found to be in the range of 0.037–0.38 μg/g for Cd, 7.8–272.9 μg/g for Cr, 2.1–50.1 μg/g for Cu, 0.008–0.092 μg/g for Hg, 42.2–151.8 μg/g for Pb, and 4.1–101.5 μg/g for Zn (Table 3). Moreover, the mean metal concentrations in sediment generally decreased in the following order: Cr>Pb>Zn>Cu>Cd>Hg.

In case of site variation, the highest mean concentrations of Hg, Cd, and Cu were recorded at stations S4, S5, and S6, respectively. Furthermore, the maximum mean level of Pb was found at station S3, while the highest average contents of Cr and Zn were recorded at station S7. Seasonally, the highest mean levels of Cd and Hg were detected in February 2011, Cu, Pb, and Zn were recorded in May 2011, and Cr was found in July 2011.

Spatial and temporal patterns were not statistically significant for Hg, Pb, and Zn concentrations in sediment samples. In terms of Cd and Cr, there were statistically significant differences ($p<0.05$ and $p<0.01$) across the seven sites, respectively. On the other hand, both Cd and Cr concentrations in sediment did not exhibit meaningful differences for sampling periods. Cu values in sediment were significantly different at each station (Kruskal-Wallis test, $p<0.05$). However, determined levels did not show significant variations among seasons.

Spearman's rank correlation analysis was performed to evaluate the relationships between metal concentrations in sediment and also between sediment and physicochemical variables. In terms of Hg, there were positive correlations with Cd ($r=0.4012$, $p<0.05$), Cu ($r=0.5340$, $p<0.01$), Pb ($r=0.4026$, $p<0.05$), and Zn ($r=0.4920$, $p<0.05$). Concerning Pb in sediment, positive relation was found with Zn ($r=0.5297$, $p<0.01$). Moreover, Pb content was negatively correlated with gravel fraction ($r=-0.4678$, $p<0.05$). Based on the metal data, a significant strong relationship existed between Cu and Zn ($r=0.8490$, $p<0.001$) and also a negative correlation was obtained for Cu in sediment and sand fraction ($r=-0.5071$, $p<0.05$). In terms of clay-silt fraction, there were significantly positive relations with Cu ($r=0.5588$, $p<0.01$), Pb ($r=0.5221$, $p<0.05$), Zn ($r=0.6047$, $p<0.01$), and organic matter ($r=0.5163$, $p<0.05$). However, clay-silt particles showed negative correlations with gravel ($r=-0.8709$, $p<0.001$) and sand ($r=-0.3851$, $p<0.05$). Organic matter was directly related to Pb ($r=0.5446$, $p<0.01$). Furthermore, sand was inversely correlated with Cu ($r=-0.5071$, $p<0.05$), Cr ($r=-0.6513$, $p<0.001$), and Zn ($r=-0.5954$, $p<0.01$).

Heavy metal concentrations in *H. diversicolor*

H. diversicolor could not be sampled at all stations during the study. Table 4 presents the measured metal concentrations in *H. diversicolor* during sampling periods. According to the results, accumulation of heavy metals in *H. diversicolor* was varied as follows: 0.025–0.103 μg/g for Cd, 1.07–20.4 μg/g for Cr, 7.5–30.6 μg/g for Cu, 0.050–0.15 μg/g for Hg, 2.8–6.0 μg/g for Pb, and 96.1–209.5 μg/g for Zn (Table 4). The relative dominance of heavy metals in *H. diversicolor* was observed in the following sequence: Zn>Cu>Cr>Pb>Hg>Cd.

Table 4 Heavy metals concentrations detected in *Hediste diversicolor* during sampling periods (μg/g dry weight)

Stations	Periods	Cd	Cr	Cu	Hg	Pb	Zn
S1	Sep 2010	0.025	3.4	9.9	0.050	5.5	170.1
S3	Sep 2010	0.041	1.7	11.6	0.050	2.8	142.0
S6	Sep 2010	0.049	20.4	23.5	0.15	4.1	121.1
S7	Sep 2010	0.10	2.4	30.6	0.050	4.8	186.2
S2	May 2011	0.034	1.4	21.4	0.050	6.0	209.5
S4	May 2011	0.032	1.1	10.9	0.050	5.5	147.5
S6	May 2011	0.052	5.1	10.8	0.001	4.0	97.3
S7	May 2011	0.039	3.0	14.3	0.024	2.9	119.8
S3	July 2011	0.029	1.1	7.5	0.050	4.4	96.1
S6	July 2011	0.034	12.4	11.3	0.050	5.6	114.3

Concerning the site variation of metals, maximum concentrations of Cd and Cu were determined at station S7. In the case of Cr and Hg, the highest levels were found at station S6 compared to other stations. And, Pb levels also showed its maximum value at station S2 where the highest Zn concentration was recorded.

The majority of polychaetes collected from eastern part of the lake were generally presented with high levels of Pb and Zn concentrations. Although, no significant spatial and temporal variations were detected for all studied heavy metal levels in polychaetes, the highest Cd, Cr, Cu, and Hg levels were measured in September 2010, while Pb and Zn concentrations were determined in May 2011.

A considerable number of significant correlations were observed between sediment and *H. diversicolor*. Cr in *H. diversicolor* showed a moderate positive correlation ($r=0.6606$, $p<0.05$) with the values recorded Cr in sediment. A strong positive relation was found between Cd in *H. diversicolor* and Zn in sediment ($r=0.8936$, $p<0.05$). Furthermore, Cd in *H. diversicolor* showed a positive relationship with Cu in *H. diversicolor* ($r=0.6687$, $p<0.05$).

Heavy metal concentrations in *M. cephalus*

During the study, fish samples could be sampled once from stations S1, S3, and S6. Thus, the concentrations of metals in the muscle and the liver of *M. cephalus* are reported in Tables 5 and 6, respectively. With regard to the results, accumulation levels in the edible muscle of the examined fish were in the range of 0.009–0.028 μg/g for Cd, 0.030–0.58 μg/g for Cr, 0.53–1.9 μg/g for Cu, 0.020–0.22 μg/g for Hg, 0.60–1.2 μg/g for Pb, and 12.7–31.6 μg/g for Zn (Table 5). Moreover, metal concentrations in the liver of *M. cephalus* fluctuated as follows: Cd, 0.32–1.1 μg/g; Cr, 0.24–1.5 μg/g; Cu, 78.0–371.5 μg/g; Hg, 0.11–0.47 μg/g; Pb, 0.26–0.91 μg/g; and Zn, 84.2–234.0 μg/g (Table 6). The average metal levels in the muscle and the liver of the fish follow in descending order: Zn>Cu>Pb>Cr>Hg>Cd and Cu>Zn>Cr>Cd>Pb>Hg, respectively.

Concerning the heavy metal levels in the muscle of *M. cephalus*, the maximum Cd was recorded at station S4, while the highest Cr, Hg, and Zn values in the muscle were found at station S5. Furthermore, the highest concentration of Pb was determined at station S6 while maximum level of Cu was measured at station S7. Moreover, the highest values of Cd and Cr in the liver were determined at station S4, while maximum level of Hg, Cu, and Pb were recorded at station S5.

Table 5 Heavy metals levels detected in muscle of *M. cephalus* during sampling periods (μg/g dry weight)

Stations	Periods	Cd	Cr	Cu	Hg	Pb	Zn
S1	Feb 2011	0.022	0.36	1.2	0.020	0.62	13.9
S3	Jul 2011	0.012	0.19	0.92	0.11	0.75	13.0
S4	Feb 2011	0.028	0.41	1.8	0.062	0.98	12.9
S4	May 2011	0.028	0.13	1.4	0.055	0.71	16.4
S4	Jul 2011	0.010	0.30	0.53	0.072	0.69	15.2
S5	Sep 2010	0.010	0.15	1.2	0.22	0.71	31.6
S5	Feb 2011	0.022	0.048	1.8	0.035	0.60	15.0
S5	May 2011	0.024	0.58	1.1	0.041	0.61	12.7
S5	Jul 2011	0.011	0.030	1.0	0.15	0.75	16.9
S6	Jul 2011	0.013	0.078	0.70	0.15	1.2	15.5
S7	Sep 2010	0.009	0.10	1.1	0.10	0.63	16.5
S7	Feb 2011	0.023	0.040	1.9	0.025	0.62	14.6
S7	Jul 2011	0.012	0.090	0.71	0.070	0.66	13.9

And, maximum value of Zn was determined at station S7 (Table 5). Regardless of all studied metals, only Pb in liver showed statistically significant variation ($p<0.01$) among seasons. Furthermore, statistically significant differences were found for Hg ($p<0.05$), Cd ($p<0.001$), and Cu ($p<0.01$) in muscle tissue.

Comparison between sampling seasons indicated that the highest concentrations of Cd and Cr in the muscle tissue of *M. cephalus* were recorded in May 2011, while maximum level of Cu was observed in February 2011, Hg and Zn values were measured in September 2010, and the Pb concentration was determined in July 2011. On a seasonal scale, Cd in liver showed its highest level in May 2011 whereas the highest concentrations of Cr and Pb were exhibited during February 2011. Moreover, the highest Cu, Hg, and Zn levels in liver were detected during July 2011 (Table 6).

Table 6 Heavy metals levels detected in liver of *M. cephalus* during sampling periods (μg/g dry weight)

Stations	Periods	Cd	Cr	Cu	Hg	Pb	Zn
S1	Feb 2011	0.47	0.63	103.7	0.11	0.52	89.1
S3	Jul 2011	0.32	0.60	121.0	0.13	0.53	172.8
S4	Feb 2011	0.62	1.5	78.0	0.17	0.81	121.1
S4	May 2011	1.1	0.75	243.4	0.14	0.65	199.7
S4	Jul 2011	0.35	0.62	209.3	0.30	0.46	189.3
S5	Sep 2010	0.46	0.38	144.6	0.20	0.27	182.2
S5	Feb 2011	0.54	1.1	144.5	0.15	0.91	157.0
S5	May 2011	0.55	1.2	302.8	0.17	0.64	191.9
S5	Jul 2011	1.0	0.86	371.5	0.47	0.65	203.6
S6	Jul 2011	0.38	0.78	95.7	0.15	0.46	88.9
S7	Sep 2010	0.35	0.24	120.0	0.14	0.26	84.2
S7	Feb 2011	0.59	0.78	286.2	0.21	0.74	170.2
S7	Jul 2011	0.94	0.61	257.4	0.34	0.54	234.0

Based on the results of this study, there were several significant relationships between measured metal levels in *M. cephalus* tissues and also between metal concentrations in fish and sediment. The highest coefficients were found between Cd and Cu in muscle ($r=0.6188$, $p<0.05$). However, Cr in muscle was inversely correlated with Cr in sediment ($r=-0.6868$, $p<0.05$). Hg in muscle exhibited positive relation with Pb in muscle ($r=0.7098$, $p<0.05$). Moreover, Zn in muscle revealed negative relationship with Cd in muscle ($r=-0.5671$, $p<0.05$).

In terms of Cr in liver, a significant relationship was found with Pb in liver ($r=0.7582$, $p<0.01$). On the other hand, Cr in liver was indirectly related with Cr in sediment ($r=-0.7363$, $p<0.01$) and Pb in sediment ($r=-0.8022$, $p<0.01$). A high degree of correlation was found between Zn in liver and Cu in liver ($r=0.8131$, $p<0.01$). However, significant negative relation was established between Pb in liver and Zn in sediment ($r=-0.7418$, $p<0.01$).

Sediment and tissue metal comparison

Bioaccumulation of metals by organism occurs if the BSAF is >1. In this study, the BSAF for Cu in *H. diversicolor* generally showed high values at stations S1 and S7 in September 2010 and also at station S2 in May 2011. Furthermore, the studied stations generally represented with high BSAF levels for Hg, except at stations S4 and S6 in May 2011. The BSAF for Zn was more than 1 unit at all stations, and higher values were found at station S1 in September 2010, at stations S2 and S4 in May 2011, and at station S3 in July 2011.

In this study, the highest BSAF levels were generally found for Hg and Zn in the muscle tissue of *M. cephalus*. In terms of Hg, station S3 in July 2011 and station S5 during September 2010 and July 2011 were represented with higher BSAF values. Regarding Zn, BSAF showed high values at stations S1 and S4 during February 2011.

With regard to BSAF values calculated for liver of *M. cephalus* and sediment, high levels were recorded for Hg, Cd, Cu, and Zn. Moreover, recorded BSAF levels for Hg at stations S4, S5, and S3 during July 2011 and station S5 in May 2011 were generally higher compared to those in other seasons. Furthermore, station S1 in February 2011 represented with the highest BSAF values for Cd, Cu, and Zn.

Discussion

The physicochemical variables characterizing water quality for any aquatic ecosystem influence the amount and distribution of water and sediment-associated contaminants

(Chapman and Wang 2001) and also limit the survival of aquatic organisms (Lawson 2011).

The accumulation of heavy metals from water column to the sediment depends on some physicochemical variables such as salinity, pH, dissolved oxygen, conductivity, and the variation in grain-size distribution (Chen and Jiao 2008). Moreover, temperature has a noticeable influence on the chemical and biochemical reactions that occur in water. An increase in temperature may enhance the toxicity of heavy metals and also increases the sensitivity of living organisms to toxic wastes (Momba et al. 2006). In this study, water temperature changed between 11.7 and 32.0 °C (Table 2) and showed an expected seasonal variation, gradually decreased from September 2010 through February 2011, and then increased up to the end of July 2011. Furthermore, the temporal variation of temperature was significant ($p<0.001$).

According to Abowei (2010), pH range between 7 and 8.5 is ideal for biological productivity, while pH at <4 has detrimental effect to aquatic life. In the present study, with the exception of station S7, recorded pH values generally fell within the accepted ranges. In contrast, measured pH levels at station S7 were always higher than 9 (Table 2) and differed significantly among stations ($p<0.001$). However, no significant temporal variation was observed. Yılmaz and Koç (2014) conducted a study on water quality of Bafa Lake, and they analyzed the water samples taken from eight different stations located inside and at the shores of the lake. According to the results, they found that the pH values (7.84–9.0) were in agreement with the findings of this study. The results of this study showed that pH may not adversely affect aquatic organisms in Bafa Lake.

Conductivity (EC) is an indirect measure of ionized species and inorganic dissolved solids in the water column and reflects the status of major ions derived from inorganic pollution. Conductivity levels varied during the study period and fluctuated between 7.3 mS/cm in February 2011 and 28.1 mS/cm in September 2010 (Table 2). However, no clear spatial difference was observed for conductivity levels, while it changed significantly among sampling seasons ($p<0.01$). Moreover, recorded EC levels in September 2010 were significantly higher compared to the measured values in other sampling periods. Measured conductivity levels in this study were lower compared to given values (670–35,500 μS/cm) by Yılmaz and Koç (2014). EC reaches high levels in warmer seasons due to increase in inorganic substances as a result of evaporation in water (Başyiğit and Tekin-Özan 2013). According to Boyd and Lichktoppler (1979), conductivity shows significant relationships with temperature, pH, and salinity. In this study, EC showed a strong positive correlation with salinity ($p<0.001$) and a negative relation with pH ($p<0.01$).

The main threat to water quality in Bafa Lake is changing salinity levels. The lake is slightly brackish and its salinity levels range between 4.0 and 17.4 psu (Table 2). During the study period, selected stations represented with similar salinity levels, except stations S6 and S7. These stations showed fluctuations for salinity through the sampling periods, and salinity peaks stand out at the beginning of September 2010. Moreover, temporal variations also noted for salinity ($p<0.001$) and September 2010 differed significantly from those in other seasons. However, a sharp decrease in salinity was recorded at the western part of the lake, i.e., station S7 (4.00 psu) particularly in February 2011 because of high amount of rainfall (TUIK 2010; 2011). In addition, the influence of discharge from the feeder channel which connects the Büyük Menderes River to Bafa Lake could presumably account for decreasing salinity in Serçin inlet (Koç 2008). Salinity has an important effect on the solubility of some metals, and an increase in salinity causes a decrease in the metal concentrations in sediment (Williams et al. 1994; Chen and Jiao 2008). It is worth noting that, in this study, there were negative correlations between salinity levels and all the studied metal concentrations, but with low levels of confidence ($p>0.05$).

Concerning DO, the standard for sustaining aquatic life is stipulated at 5 mg/l (Chapman 1992). However, DO levels less than 3 mg/l are stressful to most aquatic organisms, while concentration below 2 mg/l may cause death for most fishes. In the present study, DO concentrations changed between 3.1 and 14.1 mg/l and did not tend to vary spatially and seasonally. Moreover, recorded values were within acceptable levels for survival, growth, and metabolic activity of aquatic organisms. In contrast, recorded DO levels at station S5 in July 2011 (3.1 mg/l), station S6 (3.7 mg/l) in September 2010, and station S7 (4.3 mg/l) in July 2011 were below the permissible limit of 5 mg/l. Therefore, this variable gives cause for concern in this part of the lake.

Our measured DO levels were higher than the recorded values (5.04–11.6 mg/l) in water samples taken from the shore of Lake Bafa (Yılmaz and Koç 2014). Oxygen solubility decreases slightly as salinity increases (Lawson 2011). However, in this study, measured salinity values showed a weak correlation with DO ($r=-0.1837$, $p>0.05$). Furthermore, oxygen solubility decreases as temperature goes up (Lawson 2011). For this reason, measured DO levels were generally found higher in February 2011 because of decreasing temperature.

Lake sediments are often sampled to assess the level of environmental pollution, because they have ability to bind contaminants which enter the aquatic environment. Moreover, heavy metals in sediments can cause problems when the contaminated sediments are resuspended and such metals are taken up by filter feeders (Karthikeyan et al. 2007).

In order to evaluate the degree of contamination, some heavy metals were measured in sediment samples during the study period. According to results of this study, Cd showed significant variations among stations ($p<0.05$), and its maximum level (0.38 μg/g) was detected at station S5 in February 2011. The levels of cadmium in the sediment samples were

below the permissible value of 6 μg/g prescribed by WHO (2004).

Comparing the results with recent studies showed that the obtained results (0.037–0.38 μg/g) in this study were lower than those measured in stations in a polluted estuary situated in Ria de Bilbao, Spain (0±01–7.8±3.4 μg/g mean±SD) by Saiz Salinaz and Francés Zubillaga (1997); in Venetian lagoon (Italy) (0.25–17.8 μg/g) by Volpi Ghirardini et al. (1999); in metal-rich Seine estuary (1.13±0.05 μg/g) and in Restronguet Creek (2.76±0.20 μg/g) by Berthet et al. (2003); in Moroccan lagoons (Oualidia, 0.6±0.1 μg/g) by Idardare et al. (2008); and in Venice lagoon (Italy) (0.23±0.04–3.61±0.58 μg/g) by Frangipane et al. (2005). Moreover, Cd concentrations in sediments collected from Homa lagoon (Izmir Bay) (0.03–0.43 μg/g) (Dora et al. 2007) and in Boulogne harbor (0.38 ±0.01 μg/g) (Berthet et al. 2003) were similar to those found in the present investigation. The results for Cd concentrations in this study were higher than those found in Bay of Somme (0.05±0.00 μg/g), in Blackwater (0.19±0.01 μg/g) (Berthet et al. 2003), in Moroccan lagoons (Khnifiss, 0.37±0.08 μg/g) (Idardare et al. 2008), and in Homa lagoon (0.059±0.01–0.062±0.00 μg/g) (Taş et al. 2009).

Chromium in sediment showed a wide range of 7.8–272.9 μg/g, and its highest level was recorded at station S7 in February 2011. In contrast to the temporal variability, Cr showed spatial differences for sampling periods ($p<0.01$). A study on the sediment characteristics of Bafa Lake to evaluate the sediment quality obtained similar findings for Cr levels (63.1–277 mg/kg) (Yilgor et al. 2012) compared to our results. The findings of this study showed that Cr concentrations in sediment highly exceeded the permissible limit (25 μg/g) given by WHO (Radojevic and Bashkin 1999). These high values of Cr in the study area could be originated from the geochemical structure of the Menderes massif and also human activities (Yilgor et al. 2012).

The recorded Cr values in the present study were considerably higher than those previously reported results by Saiz Salinaz and Francés Zubillaga (1997) (14±1.6–71±22 μg/g), Dora et al. (2007) (33.4–84.9 μg/g), Idardare et al. (2008) in Oualidia (68±8 μg/g) and in Khnifiss (62±3 μg/g), Taş et al. (2009) (0.78±0.11–1.14±0.26 μg/g), and Frangipane et al. (2005) (2.80±0.41–11.21±3.33 μg/g).

The level of Cu in sediment varied significantly among sites (Kruskal-Wallis test, $p<0.05$) and did not exhibit significant variations among the seasons. Moreover, the minimum Cu concentration was detected at station S1 (2.1 μg/g) in February 2011 while the maximum level (50.1 μg/g) was recorded at station S3 in May 2011. Similar findings for Cu (5.79–55.1 mg/kg) were reported by Yilgor et al. (2012). This elevated value of copper was probably indicating the anthropogenic influence caused by the input of untreated domestic sewage. According to WHO (2004), Cu values in the sediment samples from the sampling points were above the standard value (25 μg/g). Copper is probably controlled by sedimentary features such as organic matter and grain-size (Hirner et al. 1990). In this study, Cu showed a moderate relation with clay-silt fraction ($r=0.5588$, $p<0.01$) and a negative relationship with sand ($r=-0.5071$, $p<0.05$).

The highest Cu concentrations reported in this study were, however, lower than those noted in Ria de Bilbao, Spain (21± 3.4–262±98 μg/g) by Saiz Salinaz and Francés Zubillaga (1997); in Blackwater (53±16 μg/g), in Boulogne harbor (153±10 μg/g), and in Restronguet Creek (4413±262 μg/g) by Berthet et al. (2003); and in Venice lagoon (1.66±1.59–55.15±29.93 μg/g) by Frangipane et al. (2005). The obtained results were higher than the findings in Bay of Somme (0.64± 0.11 μg/g) and in Seine estuary (48±7 μg/g) (Berthet et al. 2003), in Homa lagoon (17.2–41.0 μg/g) (Dora et al. 2007), in Moroccan lagoons (Oualidia, 17±2 μg/g; Khnifiss, 17±2 μg/g) (Idardare et al. 2008), and in Homa lagoon (3.32±0.22–3.86±0.19 μg/g) (Taş et al. 2009).

Hg levels did not show distinct differences between study sites and seasons. It showed positive relations with Cd ($r=0.4012$, $p<0.05$), Cu ($r=0.5340$, $p<0.01$), Pb ($r=0.4026$, $p<0.05$), and Zn ($r=0.4920$, $p<0.05$). The results of correlation analysis show that Hg exhibited positive significant relations with majority of the metals, suggesting a common source or chemical similarity. According to Yilgor et al. (2012), measured Hg values (0.01–0.26 μg/g) were generally higher than those determined (0.008–0.092 μg/g) in the present study.

Pb concentration ranged between 42.2 and 151.8 μg/g, and the results did not display any spatial and temporal significant differences. The highest Pb value in this study was approximately eight times higher than the maximum level (2.3–20 μg/g) recorded from the stations selected from the central part of the lake by Yilgor et al. (2012). The findings of this study showed that Pb exhibited a moderate relationship ($r=0.5446$, $p<0.01$) with organic matter and positive relations with Cu ($r=0.3828$, $p<0.05$) and Zn ($r=0.5297$, $p<0.01$). The high levels of Pb in sediments from sampling stations reflect the presence of polluting activities in the lake. Elevated Pb concentration in coastal stations selected in this study could be related to human activities such as fishing boat paintings, polluted wastes and discharges, municipal runoffs, and atmospheric deposition. In addition, the levels of Pb in the analyzed sediment samples have exceeded the limiting value (10 μg/g) set by USEPA (1999).

Previous studies carried out on Pb levels in sediment revealed that our findings were lower than those determined levels in sediments given by Saiz Salinaz and Francés Zubillaga (1997) (35±6.7–313±121 μg/g) and higher than the results reported by Dora et al. (2007) (6.5–19.1 μg/g), Frangipane et al. (2005) (18.55±2.53–101.05±5.82 μg/g), Idardare et al. (2008) in Oualidia (7±6 μg/g) and Khnifiss (6±4 μg/g), and Taş et al. (2009) (9.13±0.82–9.69± 0.72 μg/g).

The measured concentrations of Zn in this study were lower than the recommended permissible limits (123 μg/g) prescribed by USEPA (2002). Zn ranged between 4.1 and 101.5 μg/g and did not show significant temporal and spatial accumulation. A strong positive correlation was found between Cu and Zn (r=0.8490, p<0.001), and this relationship probably indicated that these elements originated from the same source. Moreover, the significant positive relation between Zn and organic matter (r=0.3765, p<0.05) showed that this may be due to preferential association of Zn and Cu to organic matter (Bardarudeen et al. 1996).

Comparing our results with the previous findings showed that Zn concentrations in sediment samples were found to be lower than those mentioned for Ria de Bilbao, Spain (121±27–951±39 μg/g) by Saiz Salinaz and Francés Zubillaga (1997); in Venetian lagoon (114–3510 μg/g) by Volpi Ghirardini et al. (1999); in Blackwater (141±4 μg/g), in Seine estuary (219±2 μg/g), in Boulogne harbor (1092±87 μg/g), in Restronguet Creek (3650±17 μg/g) by Berthet et al. (2003); and in Moroccan lagoons (Oualidia, 104±28 μg/g) by Berthet et al. (2003). In contrast, measured Zn levels in this study were higher than those reported in Bay of Somme (7.3±0.2 μg/g) by Berthet et al. (2003), in Moroccan lagoons (Khnifiss, 70±7 μg/g) by Idardare et al. (2008), in Homa lagoon (14.19±0.39–14.50±0.68 μg/g) by Taş et al. (2009), and in Bafa Lake (26.6–79.9 μg/g) by Yilgor et al. (2012).

In the present study, considerably higher organic matter levels were generally found at stations S5, S6, and S7 (Table 3). This can be explained by the organic matter flux to sediments due to the effect of anthropogenic and natural inputs. Besides, the source for organic matter within the lake could be due to supply from river runoff. The organic matter content played a role in favoring the accumulation of trace metals in the sediments of lake, as shown by the positive correlations with Cd, Pb, and Zn. Furthermore, our values (0.11–4.72 %) were in agreement with the results found (0.35–3.58 %) by Yilgor et al. (2012).

Concerning the clay-silt fraction of the sediment, a noticeable relation was found with organic matter (r=0.5163, p<0.05) which demonstrates adsorbility of fine-grained particles. The high organic matter content of the sediments and low dissolved oxygen concentration in the water column at station S5 especially in May 2011 and July 2011 reflect a reducing environment conditions (Table 3).

It is well established that the physicochemical characteristics of the sediments, particularly grain size, are important factors governing metal binding. Sediments composed of fine particles bind heavy metals and pollutants due to their high specific surface areas (Keil et al. 1994). In this study, sediment grain size distribution also differed among sites. Stations S1, S2, S3, S4, and S5 were mainly composed of medium and fine sand while stations S6 and S7 have high clay-silt content (fine fraction). Additionally, the results showed that the last two stations generally have higher heavy metal levels especially Cr, Cu, and Zn.

It is clear that sediments in the western and southern parts of Bafa Lake were heavily polluted by heavy metals while the eastern area exhibited lower metal pollution related to the inflow of freshwater runoff from the Büyük Menderes River that drains in this region, and grain-size distribution. This area was exceedingly polluted due to human activities within the lake arising from fishing activities and also such services as boats and restaurants. Moreover, temporal fluctuations in metal concentrations of surficial sediments could be associated with rainfall. The results of this study showed that recorded Cr, Cu, and Pb levels in sediment samples were above the maximum allowable limits set by WHO/USEPA, and the studied stations might face significant risks from heavy metal pollution in the future.

Polychaetes are the mostly used ecologically keystone species in the assessment of bioaccumulation and biomagnification of contaminants within the ecosystem (Díez et al. 2000; Idardare et al. 2008; Frangipane et al. 2005). Regarding Cd in *H. diversicolor*, the minimum value (0.025 μg/g) was detected in station S1 in September 2010, and it was four times less than that of the maximum level (0.10 μg/g) recorded at station S7 during the same period (Table 4). *H. diversicolor* can tolerate higher concentrations of Cd by using two metalloproteins such as myohaemerythrin-like protein and the extracellular hemoglobin (Ruffin et al. 1994). In *H. diversicolor*, a moderate relationship found between Cu and Cd (r=0.6687, p<0.05) may be ascribed to the prevailing same source of these metals. *H. diversicolor* has an ability to accumulate both Cu and Cd proportionally to their concentrations in the surrounding medium (Díez et al. 2000; Berthet et al. 2003). The higher concentration of these two metals could be associated with anthropogenic sources including sewage, industrial effluents, and agriculture runoffs.

Based on the previous data, Cd concentrations in *H. diversicolor* reported in this study were lower than those measured in Venetian lagoon by Volpi Ghirardini et al. (1999) (0.04–0.55 μg/g), in Estuary of Urdaibai by Díez et al. (2000) (0.1–1.7 μg/g), in Blackwater (0.221 μg/g) and Seine estuary (0.33±0.09 μg/g), in Restronguet Creek (0.6 μg/g) by Berthet et al. (2003), in Moroccan lagoons (Khnifiss, 1.0±0.2 μg/g) by Idardare et al. (2008), in Homa lagoon (Izmir Bay) by Taş et al. (2009) (0.31±0.02–0.34±0.03 μg/g ww), in Venice lagoon (Italy) by Frangipane et al. (2005) (0.7–0.30 μg/g), and in Ria Formosa lagoon and Cádiz Bay by Gomez et al. (2013) (0.04±0.01–0.17±0.04 μg/g). This findings agree with earlier reported works conducted in Ria de Bilbao, (Spain) by Saiz Salinaz and Francés Zubillaga (1997) (not detectable (nd)–0.1±0.1 μg/g), in Bay of Somme (0.05±0.03 μg/g) and in Boulogne harbor (0.06 μg/g) by Berthet et al. (2003), in Homa lagoon by Dora et al. (2007) (0.01–0.16 μg/g ww), and in Moroccan lagoons (Oualidia, 0.09±0.6 μg/g) by Idardare et al. (2008).

Peak value of Cr content in *H. diversicolor* was measured (20.4 μg/g) at station S6 in September 2010 and the lowest concentration (1.07 μg/g) was measured at station S3 in July 2011 (Table 4). Metal concentrations in surface sediments and in *H. diversicolor* appear to be generally unrelated. However, there was a significant positive relation for Cr levels in *H. diversicolor* relative to their concentrations in sediment (r=0.6606, p<0.05). The results of this study showed that *H. diversicolor* is the strongest accumulator of Cr in sediment.

A comparison with previous studies showed that Cr concentrations in *H. diversicolor* reported in this study were lower than the levels (50.71±1.64–62.34±1.83 μg/g ww) given by Taş et al. (2009). However, findings were higher than those measured by Díez et al. (2000) (0.1–1.7 μg/g), Dora et al. (2007) (0.02–1.10 μg/g ww), in Oualidia (2.0±1.3 μg/g) and in Khnifiss (2.1±0.5 μg/g) by Idardare et al. (2008); Frangipane et al. (2005) (1.58–4.59 μg/g).

Copper is accumulated and stored in membrane-bound vesicles and in Cu-containing granules in epidermal and nephridia cells of *H. diversicolor* rather than being excreted (Depledge and Rainbow 1990). The minimum level of copper (7.5 μg/g) was recorded in station S3 during July 2011 and maximum level (30.6 μg/g) was determined at station S7 in September 2010 (Table 4).

Based on the previous data, Cu concentrations in *H. diversicolor* reported in this study were lower than those measured in Blackwater (44 μg/g) and Restronguet Creek (3940 μg/g) by Berthet et al. (2003), Díez et al. (2000) (6.3–39 μg/g), Frangipane et al. (2005) (15.08–39.44 μg/g), and Gomez et al. (2013) (2.66±0.65–37.47±11.88 μg/g). Both potential detoxification systems of copper in *H. diversicolor* include binding by metallothioneins (sulfur-rich cytosolic proteins) and incorporation into insoluble deposits or granules (Berthet et al. 2003; Mouneyrac et al. 2003). In the present study, no significant enhancement or variation between organisms collected from different sites was detected in terms of Cu despite the high anthropogenic metal levels in the sediments.

The maximum Cu concentrations in *H. diversicolor* reported in this study were higher than those described in Bay of Somme (11±4 μg/g), in Seine estuary (26±11 μg/g), and in Boulogne harbor (15 μg/g) by Berthet et al. (2003); in Homa lagoon by Dora et al. (2007) (1.54–10.10 μg/g ww); in Moroccan lagoons (Oualidia, 6.8±2.5 μg/g; Khnifiss, 8.8±3.8 μg/g) by Idardare et al. (2008); and in Homa lagoon by Taş et al. (2009) (27.0±0.99–29.1±1.0 μg/g ww).

According to Alam et al. (2010), the feeding ecology of polychaetes and their effect on sediment biogeochemistry are more important factors in determining the bioaccumulation of Hg. The highest Hg concentration (0.15 μg/g) in *H. diversicolor* was detected at station S6 in September 2010 and the lowest value (0.001 μg/g) was found at station S6 in May 2011 (Table 4). Our results show that concentrations of Hg exhibited a lower degree of accumulation in polychaete species compared to other studied metals. Comparison of literature data showed that *H. diversicolor* collected from the Pialassa lagoons located along the northern Adriatic Italian coast (Virgilio et al. 2003) had consistently higher mercury concentrations (22.8 to 466.2 ng/g) as compared with our results.

The estimated Pb content of *H. diversicolor* was observed in the range between 2.8 and 6.0 μg/g. The maximum concentration was found at station S2 during May 2011 and the lowest level was measured at station S3 in September 2010 (Table 4). Based on the previous data, Pb concentrations in *H. diversicolor* determined in this study were lower than the given levels by Díez et al. (2000) (0–10 μg/g), Dora et al. (2007) (0.23–15.80 μg/g ww), Taş et al. (2009) (11.27±0.41–12.54±0.39 μg/g ww), and Gomez et al. (2013) (0.30±0.05–6.84±1.50 μg/g).

And, our results were higher than the values measured by Saiz Salinaz and Francés Zubillaga (1997) (nd–0.3±0.5 μg/g); Frangipane et al. (2005) (0.68–1.54 μg/g); in Moroccan lagoons (Oualidia, 1.0±0.5 μg/g; Khnifiss, 3.0±1.1 μg/g) by Idardare et al. (2008). The accumulation of Pb in *H. diversicolor* was found to be lower than those found in the sediments, and the results are in agreement with those reported by Alam et al. (2010) in different polychaeta species collected from the Indian Sundarban mangrove.

The concentration of Zn reached its maximum level (209.5 μg/g) at station S2 in May 2011, while its minimum value (96.1 μg/g) was recorded during July 2011 at station S3 in polychaeta (Table 4). No significant relation was found between Zn level in *H. diversicolor* and sediment. Higher value of Zn which is an essential metal used as a co-factor in numerous enzymes was possibly regulated from the soft tissue of this species (Durou et al. 2007).

Based on the previous data, Zn concentrations in *H. diversicolor* reported in this study were lower than those measured by Volpi Ghirardini et al. (1999) (177–358 μg/g), Díez et al. (2000) (25–307 μg/g), and Gomez et al. (2013) (126.13±4.65–257.50±37.41 μg/g). Conversely, the findings of this study were higher compared to the results obtained from the study in Ria de Bilbao, Spain (136±19–181±26 μg/g–nd) by Saiz Salinaz and Francés Zubillaga (1997); in Bay of Somme (105±3 μg/g), in Restronguet Creek (173 μg/g), in Blackwater (152 μg/g), in Seine estuary (166±37 μg/g), and in Boulogne harbor (88 μg/g) by Berthet et al. (2003); in Moroccan lagoons (Oualidia, 115±30 μg/g; Khnifiss, 94±44 μg/g) by Idardare et al. (2008); and in Homa lagoon (57.05±2.26–58.90±1.83 μg/g ww) by Taş et al. (2009).

Among the six heavy metals examined, generally, Cu, Hg, and Zn concentrations were readily accumulated in *H. diversicolor* much higher than in sediments (Tables 3 and 4). Hateley et al. (1989) observed that Cd, Cu, and Zn tolerance of *H. diversicolor* was inheritable, demonstrating its

genetic basis. The consistent higher values of Cu and Zn in polychaetes reveal that these two metals have a high potential for biomagnifications. Diet or feeding habits on microzoobenthos and macrozoobenthos, diatoms, organic matter, and detritus on the sediment surface are important factors that can explain the variety of metal concentration in this species (Lucas and Bertru 1997).

Concentrations of Cd, Pb, and Cr showed a lower degree of accumulation in *H. diversicolor*. This might be related to the unique adaptive strategies of these worms by secreting mucus in response to these metals which help in reducing metal availability for uptake (Mouneyrac et al. 2003).

This study revealed that heavy metal accumulation in *H. diversicolor* is being influenced by an increasing number of diverse sources such as industrial and domestic effluents and also agricultural runoff. Seasonally, it was noted that the highest Cd, Cr, Cu, and Hg levels in *H. diversicolor* were measured in September 2010. On the other hand, the maximum Pb and Zn values were recorded in May 2011 (Table 4). It can be due to the reproductive cycle and variations in body weight (Idardare et al. 2008). The highest concentrations for all studied metals in *H. diversicolor* tissues were generally found during periods of high gametogenic activity (September 2010). Similar seasonal variations were also observed to those reported by Díez et al. (2000); Idardare et al. (2008). The overall metal concentrations determined in *H. diversicolor* tissues showed different spatial responses, reflecting different sources of contamination, but with no clear spatial and temporal patterns.

Fishes are on the top of the aquatic food web and have the ability to accumulate considerable amounts of metals from water and sediment. The metal accumulation in different organs in fish species also depends on their benthic feeding strategy, ecological needs, metabolism, age, size, length, regulatory ability, swimming, and behavior of the fish (Marcovecchio 2004; Canbek et al. 2007). The results of the present study showed that different metals are accumulated at different concentrations in organ and tissue of *M. cephalus* which is an important species in this region.

Concerning the feeding behavior, this species is a demersal detritivore and feeds preferentially on detritus, zooplankton, fish, crustaceans, and algae (Farrugio and Quignard 1974). It is a local and endemic species, which has the ability to accumulate heavy metals and other pollutants in larger amounts and so it is a suitable indicator of its environment (Ferreira et al. 2005). *M. cephalus* is continually sifting through benthic sediments for food which not only increases exposure to contaminated sediments but also liberates metals from sediments into the water column after disturbance (Waltham et al. 2011). Uptake of contaminants in food or ingested material, including sediment in the case of *M. cephalus*, is reflected by high metal concentrations in the liver (Heath 1995).

According to the results of this study, metal accumulation in examined species was generally higher in liver than that measured in muscle whereas only Pb concentrations were generally higher than those in liver. This is important because the muscle constitutes the greatest mass of the fish that is consumed by human. Liver has a tendency to regulate and reduce metals in a higher degree compared to muscle tissue (Abdel-Moneim et al. 1994). This is probably due to their physiological roles such as strong irrigation and excretion function in fish metabolism. Fish respond to heavy metal exposure by producing metallothionein which appears as a metal detoxification mechanism within the body (Peakall and Burger 2003). In this study, Zn contents were the most abundant element in the muscle of fish species, followed by Cu (Table 5). On the other hand, Cu was the highest in liver of *M. cephalus*, followed by Zn (Table 6). Zn levels in liver were 6–16 times higher than in muscle.

The minimum Cd in muscle (0.009 μg/g) was detected at station S7 in September 2010 (Table 5). Conversely, the lowest Cd in liver (0.32 μg/g) was found in station S3 during July 2011 (Table 6). However, Cd in muscle (0.028 μg/g) was enriched in station S4 in February 2011 and May 2011 (Table 5) and also liver (1.1 μg/g) was represented with the maximum level at the same station during May 2011 (Table 6). The highest coefficient was found between Cd and Cu in muscle (r=0.6188, p<0.05). These findings for Cd in muscle were similar to those in previously recorded levels (0.013–0.025 μg/g) in lagoon of Ghar El Melh (GEM) by Chouba et al. (2007).

In this study, comparing to our results, high levels of Cd have been detected in muscle of *M. cephalus* inhabiting in Tuzla lagoon (0.08±0.01–0.11±0.00 μg/g) (Dural et al. 2007), Mazala Lake (1.08–1.71 μg/g) (Bahnasawy et al. 2009), in Köycegiz Lake (0.09–0.14 μg/g ww) (Yilmaz 2009), and Nokoue Lake (0.05±0.01–0.09±0.01 μg/g) (Degnon et al. 2012). In terms of Cd in liver, recorded concentrations in this study were lower than the levels found by Yilmaz (2009) (2.41–4.19 μg/g ww). And, our results were higher than the concentrations in liver (0.30–0.68 μg/g) found by Chouba et al. (2007), Dural et al. (2007) (0.02±0.02–0.35±0.17 μg/g), and Degnon et al. (2012) (0.11±0.01–0.5±0.05 μg/g).

According to our findings, the least Cr in muscle (0.030 μg/g) was measured at station S5 in July 2011 while the highest level (0.58 μg/g) was found at the same station in May 2011 (Table 5). However, the lowest Cr in liver (0.24 μg/g) was measured in station S7 during September 2010 and it reached its maximum level (1.5 μg/g) at station S4 in February 2011 (Table 6). In terms of Cr in muscle, a significant inverse relationship was found with Cr in sediment (r=−0.6868, p<0.05). However, Cr in liver showed a relationship with Pb in liver (r=0.7582, p<0.01). On the other hand, Cr in liver was inversely related with Cr in sediment (r=−0.7363, p<0.01) and Pb in sediment (r=−0.8022, p<0.01).

Both Cu in muscle (0.53 μg/g) and liver (78.0 μg/g) represented with minimum values at S4 station in July 2011 and February 2011, respectively (Tables 5 and 6). Furthermore, the highest level of Cu in muscle tissue was recorded (1.9 μg/g) at station S7 in February 2011 (Table 5), while the maximum Cu in liver (371.5 μg/g) was detected at station S5 in July 2011 (Table 6). Previous studies conducted by Bahnasawy et al. (2009) (3.56–5.68 μg/g) and Yilmaz (2009) (5.45–7.09 μg/g ww) have shown that Cu concentrations in muscle of *M. cephalus* were somewhat higher than our results. Moreover, determined Cu concentrations in liver were relatively lower than the values given by Yilmaz (2009) (367.90–981.34 μg/g ww). However, in this study, Cu levels in muscle and liver were generally higher than the levels 0.47±0.12–0.62±0.01 μg/g and 4.77±0.03–12.03±0.52 μg/g found by Dural et al. (2007), respectively. A high degree of correlation was found between Zn and Cu in liver (r=0.8131, p<0.01), indicating that they are from the same anthropogenic source.

The lowest concentrations of mercury were measured in the muscle (0.020 μg/g) (Table 5) and the liver (0.11 μg/g) (Table 6) of *M. cephalus* at station S1 in February 2011. However, peak values of Hg levels in muscle (0.22 μg/g) (Table 5) and liver (0.47 μg/g) (Table 6) were detected at station S5 in September 2010 and the same station in July 2011, respectively. Hg in muscle exhibited positive relation with Pb in muscle (r=0.7098, p<0.05). Concerning the muscle and the liver of *M. cephalus*, the Hg levels obtained from this study were lower than earlier reported results for the muscle (0.22–0.41 μg/g) and liver (0.90–0.97 μg/g) of this species (Chouba et al. 2007). And, the detected results for muscle in this study were lower than the given concentrations (0.02±0.00–0.3±0.03 μg/g) by Degnon et al. (2012). And, given levels for liver were higher than the values (0.01±0.00–0.05±0.01 μg/g) reported by Degnon et al. (2012).

In case of Pb in muscle, the lowest level (0.60 μg/g) was obtained from station S5 in February 2011, and the highest level in muscle (1.2 μg/g) was determined at station S6 in July 2011 (Table 5). On the other hand, the lowest Pb in liver was found (0.26 μg/g) at station S7 in September 2010, and its maximum level (0.91 μg/g) was measured at station S5 during February 2011 (Table 6).

In earlier studies, *M. cephalus* from lagoon of Ghar El Melh (GEM), Tunisia, showed higher Pb levels in their muscles in Mazala Lake (1.66–2.98 μg/g) (Bahnasawy et al. 2009) than our findings. However, in this study, measured Pb levels in muscle were in agreement with the findings reported from Tuzla lagoon (0.49±0.15–1.19±0.44 μg/g) (Dural et al. 2007) and the Nokoué Lake (0.29±0.02–1.3±0.01 μg/g) (Degnon et al. 2012). And, our recorded Pb levels in muscle were higher than the findings recorded from the Ghar El Melh Lagoon (Tunisia) (0.05–0.42 μg/g) (Chouba et al. 2007) and Köycegiz Lake (0.021–0.63 μg/g ww) (Yilmaz 2009). Concerning Pb in liver, significant negative relation was established between Pb in liver and Zn in sediment (r=−0.7418, p<0.01). In the present study, Pb levels in liver were relatively lower than those reported from previous studies by Dural et al. (2007) (1.85±0.11–3.12±4.68 μg/g) and Degnon et al. (2012) (1.52±0.1–1.0±0.01 μg/g). However, these Pb concentrations in liver were in the range of values reported by Chouba et al. (2007) (0.548–0.927 μg/g) and Yilmaz (2009) (0.56–0.98 μg/g ww).

Concerning Zn in muscle, the lowest value was (12.7 μg/g) recorded at station S5 in May 2011 while the highest concentration was (31.6 μg/g) detected at the same station in September 2010 (Table 5). Furthermore, the lowest Zn value in liver (84.2 μg/g) was detected at station S7 in September 2010 whereas its highest level in liver (234.0 μg/g) was determined at station S7 in July 2011 (Table 6).

On the other side, the comparison of the present results with other studies showed that detected Zn level in muscle were lower than the results reported by Dural et al. (2007) (8.27±2.18–60.86±56.10 μg/g), Yilmaz (2009) (68.90–124.72 μg/g ww), and Bahnasawy et al. (2009) (13.21–38.42 μg/g). Furthermore, Zn concentrations in liver were lower than the values given by Yilmaz (2009) (259.12–587.23 μg/g ww) while the measured levels in this study were higher than the concentrations reported by Dural et al. (2007) (26.7±4.35–99.8±1.27 μg/g). Moreover, Zn in muscle revealed a negative relationship with Cd in muscle (r=−0.5671, p<0.05).

Seasonally, metal analysis in liver showed that Hg and Cu concentrations were high in July 2011, Cd was maximum during May 2011, and however, those of Cr, Pb, and Zn were high in February 2011. Seasonal changes of metal concentrations in fish may result from intrinsic factors such as growth cycle and reproductive cycle and from changes in water temperature (Kamaruzzaman et al. 2010). On the other hand, the differences noted in the metal concentrations in different tissues could have been originated from local pollution.

The permissible limits proposed by the Turkish legislation in fish muscle is 0.1 mg/kg for Cd, 1.0 mg/kg for Pb, 20.0 mg/kg for Cu, and 50 mg/kg for Zn (Anonymous 1996). In addition, Food and Agricultural Organization establishes maximum levels of 0.5 mg/kg for Cd and Pb, and 30 mg/kg for Cu and Zn (FAO 1983). Comparison of our data with these limits indicates that Cd and Cu levels were generally lower than the levels issued by Turkish legislation and the FAO. Concerning Cr, no levels established by FAO are reported; however, the detected concentrations do not exceed the threshold levels (8.0 mg/kg) established by EPA.

In the case of mercury, levels were below the 0.5 mg/kg wet weight limit recommended by the WHO (1986) in edible parts of the fish samples. In addition, all samples were below the maximum permissible Hg limit of 0.5 mg/kg set by the European Commission (Regulations 1881/2006 and 420/2011) and also 0.5–1 mg/kg limit for different fish products adopted by Turkey (Turkish Alimentarius Codex 2002).

In terms of Pb, the detected level in fish collected from S6 station in July 2011 exceeded the Turkish legislation limit. However, levels were below the allowable limit proposed by FAO. Zn levels was below the allowable limit accepted by Turkish legislation, and fish samples collected from S5 station in September 2010 only exceeded the permissible limit accepted by FAO.

Biota-sediment accumulation factor (BSAF) is generally used to evaluate the efficiency of metal bioaccumulation in the tissues or organs of the organisms (Thomann et al. 1995). Our study indicated that the BSAF for *H. diversicolor* generally showed higher values at stations S1 and S2 for Zn and Cu. BSAF levels revealed that this organism has the ability to accumulate Hg, Cu, and Zn more than in sediments. BSAF values may be ranked in the following order in the polychaeta: Zn>Hg>Cu. However, we noted a different order of accumulation (Hg>Zn and Cu>Hg>Zn>Cd) in muscle and liver of *M. cephalus*, respectively. The differences can be explained by their ecological characteristics of the species.

Conclusions

In Bafa Lake, even though there are no on-site sources of contamination, the major source of heavy metal contaminations in the lake is originated from Büyük Menderes River and some streams adjoining the lake and its proximity to industrial locations, and also domestic and municipal sewage.

The findings of this study showed that the accumulation of Cr, Cu, and Pb in sediment samples was exceeding the maximum permissible limits prescribed by WHO/USEPA and that the studied stations might face high level risk from metal pollution in the future. It is logical to say that the high concentration of metals in the lake becomes gradually accumulated on the sediments and gets transferred to biota. Thus, the study of the different fractions of the heavy metal levels in the sediment was important because the total concentrations do not give accurate information of the real degree of the potential contamination.

According to the results, Cr in *H. diversicolor* showed a moderate positive relationship with sediment whereas Cr in both muscle and liver was inversely correlated with Cr in sediment. *H. diversicolor* can validate its capacity to bioaccumulate several metals from sediments. Zn contents were the highest in both *H. diversicolor* tissue and muscle of fish species in this study, followed by Cu. In terms of liver of *M. cephalus*, Cu was enriched followed by Zn. According to international criteria and Turkish regulations, the level of Pb in edible muscle of the fish collected from S6 station and the level of Zn in the fish from S5 station exceeded the food safety limits of the edible fish tissue of *M. cephalus*. With regards to human health, the muscle of *M. cephalus* is not contaminated and it does not show any danger for consumption.

Comparing the studied species, we can conclude that *H. diversicolor* passes a greater capacity for metal bioaccumulation than muscle tissue of *M. cephalus* with the exception of Hg. Higher concentrations of Hg, Cd, and Cu were determined in liver of *M. cephalus* samples in the present study as compared with those in *H. diversicolor*.

The high level of BSAF for Cu, Hg, and Zn in *H. diversicolor*, Hg and Zn in muscle, and Cd, Cu, Hg, and Zn in liver of *M. cephalus* showed that these species were good bioindicators to monitor pollution in the lake. Understand the uptake pathways and accumulation mechanisms of heavy metals by these species could be investigated with the use of chemical tracers.

In conclusion, the results presented in this study provide important baseline information that can be used to assess future changes in contaminants in the study area. The study showed a need for continuous pollution assessment studies of sediment and aquatic organisms in Bafa Lake.

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