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Review

# Bioaccumulation of Metals in Some Fish Species from the Romanian Danube River: A Review

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**Abstract:** The Danube is the second-largest river in Europe and has been subject to pollution in the past. Additionally, in the last few years, the rapid pace of industrialization and urbanization has led to the inevitable pollution of this aquatic ecosystem by certain metals (essential and non-essential elements). This issue is considered the central problem of pollution in the Danube and is gaining increasing attention. Fish is a good source of proteins, polyunsaturated fatty acids (especially omega-3 fatty acids), essential vitamins, and minerals. Fish are often exposed to metals present in their aquatic environment through direct contact with contaminated water or by consuming organisms that have accumulated metals in their tissues; therefore, the elevated concentrations of metals in water and sediments are reflected in the fish flesh. In this context, the safety of fish and fishery products for human consumption is a public health concern. In the last two decades, more and more reports have shown that Danube River fish are contaminated with metals, causing great concern among consumers. The negative perception continues, although recent scientific studies show that metal levels in the edible parts of the fish are below acceptable limits. The objective of this study was to put together a multitude of scientific research studies that investigate the levels of some metals in various tissues of some fish species with high economic value in the Romanian market, as well as the levels of metals in the water and sediments. The collected data were then utilized to assess the potential health risks posed to humans.

**Keywords:** metals; tissue bioaccumulation; commercial fish species; Danube River

**Key Contribution:** This article provides a comprehensive review of data regarding the concentrations of some important metals in the Romanian Danube River water and sediments and the bioaccumulation of these metals in fish tissues. By critically analyzing the available research, this article aims to strengthen the scientific community's and the general public's understanding of the magnitude of fish contamination with metals and its potential implications. The conclusion of our study revealed that in the Danube River, the levels of metals detected in fish meat generally remain below the maximum residue limits (MRLs) proposed in the Official Journal of the European Communities (2001). Nevertheless, monitoring metal concentrations in fish meat is necessary, particularly considering its consumption in the human diet.

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## 1. Introduction

The increase in pollution with metals has emerged as a critical environmental concern that has gathered more attention in recent decades. This is primarily due to the alarming levels of metal deposition in various ecosystems and the subsequent detrimental impact on human health and the environment [1,2]. Metals are stable, non-biodegradable, and tend to persist in the environment for prolonged periods [3–5]. Once the environmental levels of these metals surpass a certain threshold and accumulate in living organisms, they pose a significant threat to their health and well-being and lead to long-term adverse effects on humans [1,2]. Some metals, such as copper (Cu), zinc (Zn), cobalt (Co), or iron (Fe), are crucial for many biochemical processes in living organisms, being essential elements for aquatic plants and animals [6,7]. Cadmium (Cd), arsenic (As), and lead (Pb) are non-essential elements that can cause harmful effects even at trace concentrations [8].

Anthropogenic sources of metals include fuel combustion, industrial effluents, smelting, mining, the leaching of drill cuttings, and overburden leaching (Figure 1). The metals do not undergo degradation once they enter the water ecosystem; instead, they persist and can accumulate on solid surfaces or interact with aquatic organisms and plants through absorption, dissolution, suspension, or uptake [9–14].

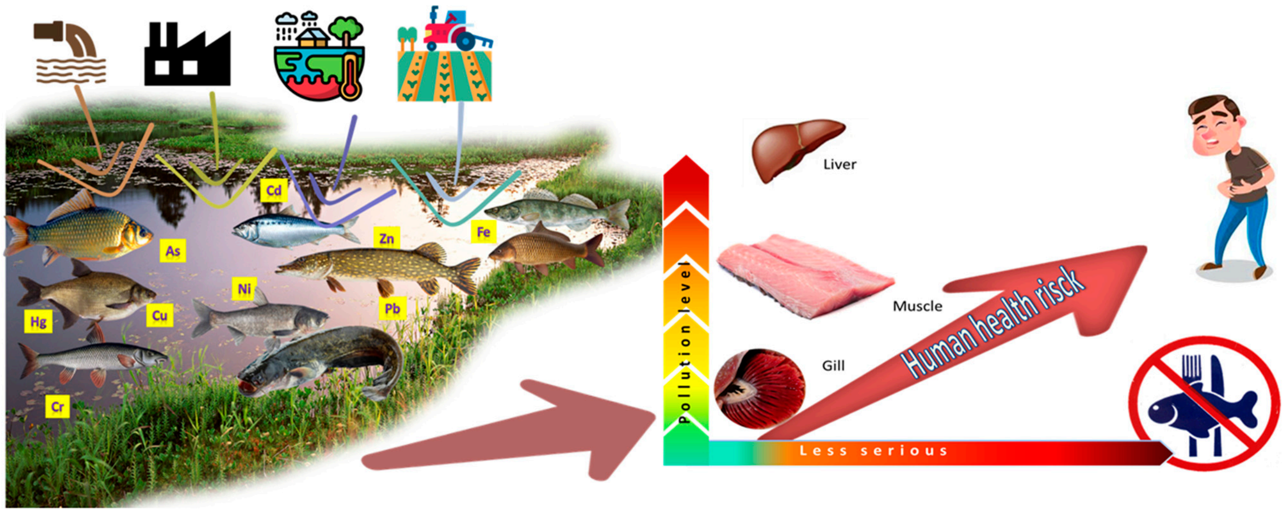


**Figure 1.** The process of metal bioaccumulation in the aquatic trophic chain.

Moreover, various fish species can serve as effective biological indicators for assessing the extent of exposure to metals and other pollutants [15,16].

### 1.1. Effect of Metals on Fish due to Exposure

Metals exert significant influences on fish following exposure. Upon uptake, metals can accumulate in fish tissues and disrupt physiological processes, leading to adverse health effects. Metal-induced pathophysiological changes in fish can include oxidative stress [17], alteration of the hematological profile [18], impaired immune function [19], disruption of ion regulation, and alterations in tissue morphology and cellular function [20].

Accordingly, metals can impair immune function, resulting in decreased resistance to pathogens and heightened susceptibility to infections [21]. Imbalances in ion regulation can modify the body’s homeostasis, resulting in disturbances in both osmoregulation and electrolyte balance. Moreover, metals can induce morphological changes in tissues, such as gill damage and liver abnormalities, while also interfering with cellular functions, including enzyme activity and gene expression [22]. Several other studies have indicated that fish exposed to metals experience a decline in various reproductive parameters, including the gonadosomatic index (GSI), fecundity, hatching rate, fertilization success, abnormal shape of reproductive organs, and ultimately reproductive failure [23–25].

Understanding the complex interactions between metals and fish pathophysiology is crucial for comprehending the consequences of metal pollution on aquatic ecosystems and developing effective mitigation strategies.

### 1.2. Effect of Metals on the Human Population

To safeguard human health, it is crucial to assess the levels of contaminants present in fish meat. Therefore, these pieces of information are important for making informed decisions and implementing measures to minimize human exposure to these contaminants [26,27].

However, the toxicity of the trace metals is correlated with the dose of exposure, pollutant concentration, duration of exposure, the individual's susceptibility, and other relevant variables. Some studies have provided evidence of the potentially severe complications that can arise from human exposure to high doses of mercury (Hg) and Pb [28]. For example, Hg exposure can have detrimental effects on the nervous system, leading to symptoms such as neurological impairment, cognitive deficits, tremors, and, in severe cases, even paralysis. Prolonged exposure to high levels of Hg has been associated with Minamata disease, a neurological disorder characterized by symptoms such as ataxia, sensory disturbances, and visual impairments [29,30].

Pb exposure, on the other hand, can cause damage to multiple organ systems, including the nervous system, gastrointestinal tract, or kidneys [31,32]. The central nervous system is particularly vulnerable to Pb toxicity, and it can result in developmental delays and cognitive impairments in children. In adults, Pb poisoning can lead to abdominal colic pain, anemia, hypertension, and impaired renal function [33]. Additionally, metal poisoning causes damage to cells of the heart, liver, blood composition, and other important organs [34–36].

Consuming fish and seafood contaminated with Cd, Co, Cr, Ni, and Pb can cause neurological disorders, kidney damage, circulatory system problems, and an increased risk of cancer [31].

The target organ effects of inorganic As vary depending on the dose, mode of exposure, and duration of exposure. Consuming high doses (0.04 mg/kg/day) orally, either as a single exposure or repeatedly over weeks or months, can lead to nonspecific effects such as gastrointestinal issues (diarrhea, cramping), hematological effects (anemia, leucopenia), peripheral neuropathy, and cardiovascular effects. While these effects are generally reversible, they can cause permanent damage to the affected organ systems. Chronic exposure to small doses of As (0.01 mg/kg/day or higher) through inhalation or oral ingestion for 3 to 5 years can result in skin hyper-pigmentation (diffuse or spotted) and, over time, benign skin lesions (hyperkeratosis) and skin cancer. Prolonged exposure can also lead to liver disease, as indicated by abnormal porphyrin metabolism [37].

Cd, particularly in its inorganic form, can have various health effects; acute effects from oral exposure to Cd are uncommon [38]. Long-term exposure to low doses of Cd over several years can result in kidney tubular dysfunction and osteoporosis in susceptible populations, particularly elderly women with Fe deficiency [37,39].

Exposure to inorganic Pb compounds can have adverse effects on multiple organ systems. Infants and young children, particularly during the neonatal period and early childhood, are highly susceptible to lead exposure and may experience impaired motor function and cognitive development, along with the possibility of developing anemia. Chronic exposure to high levels of Pb in older children can also lead to anemia as well as central nervous system effects such as impaired motor function and cognitive function, and in severe cases, seizures, coma, and even death, especially when blood Pb levels exceed 80 μg/dL [37]. In adults, elevated blood Pb levels above 40 μg/dL can result in impaired heme synthesis and chronic kidney disease, while sustained levels above 80 μg/dL can lead to lethargy and cognitive impairment. Epidemiological studies indicate a slight dose-effect relationship between Pb exposure and blood pressure, with blood levels up to 30 to 40 μg/dL. Although Pb has been found to cause tumors in experimental animals, there is currently insufficient evidence to classify Pb as a human carcinogen [40,41].

Hg exists in three forms that are of toxicological concern: elemental Hg, inorganic Hg, and methyl Hg. The target organs affected by Hg exposure vary depending on the specific form. The central nervous system and the kidneys are the primary target organs for elemental Hg toxicity. Renal toxicity from elemental Hg exposure may involve an immunological mechanism that can lead to glomerulonephritis, potentially progressing to renal failure [37]. Methyl Hg exposure occurs through the consumption of fish that have accumulated methyl Hg in the aquatic food chain. The brain is the primary target organ in the case of methyl Hg exposure. The most vulnerable population is the developing fetus. Methyl Hg easily crosses the placenta, exposing the developing brain to its toxicity. Even low levels of exposure can result in impaired motor and language skill development during neonatal life and early childhood. Higher exposures can cause severe cognitive effects, including paresthesia, blindness, deafness, and, in the most severe cases, fetal death and abortion. Methyl Hg in the brain slowly transforms into inorganic Hg, raising questions about whether the actual toxic species of Hg in the brain is methyl Hg, inorganic Hg, or elemental Hg [41,42].

Human activities in the past centuries have left metal contamination in terrestrial and aquatic ecosystems [43]. Because of this historical metal contamination, metal concentrations in sediments and surface waters can remain above natural levels and potentially threaten the health of aquatic ecosystems [44–46].

In order to ensure the preservation of a healthy ecosystem for both food security and human safety, it is crucial to thoroughly investigate the negative consequences of heavy metal exposure on fish, which can ultimately impact humans. In this context, the objective of our research was to gather relevant information regarding nine different metals (Cd, Cu, Cr, Fe, Hg, Ni, Pb, and Zn) and the metalloid As in both water and sediment samples, as well as to overview the bioaccumulation of these metals in various organs (such as the gills, liver, kidneys, digestive tract, and muscles) of some fish species with high economic value in Romania. By comprehensively examining these factors, we can gain a better understanding of the potential risks associated with metal contamination and make informed decisions to mitigate their detrimental effects.

## 2. Materials and Methods

The Danube is the second-longest river in Europe, with a total length of 2826 km. Because of the very different geographical regions through which it flows as well as the different hydrobiological characteristics along its course, the Danube is divided into three sectors: the Upper Danube (Alpine sector) from its sources to Bratislava, with a length of 1021 km; the Middle Danube (Pannonian sector), running between Bratislava and Baziaş, with a length of 764 km; and the Lower Danube (Carpathian-Balkan sector) between Baziaş and the Black Sea, with a length of 1075 km [47].

The Romanian Danube is divided into five sectors: the Carpathian sector between Baziaş (km 1075) and Drobeta Turnu Severin (km 931); the upstream sector of Iron Gate II (km 862) to Calafat (km 795); the sector between Calafat (km 795) and Călăraşi (km 370); the sector between Călăraşi (km 370) and Brăila (km 170); and the sector between Brăila (km 170) and the Black Sea [47] (Figure 2).

In this paper, we have conducted a comprehensive review of published research focusing on the Pannonian sector of the Danube and the Romanian section. We aimed to provide an overview of the metal levels found in water, sediments, and some fish species within these regions, from entering Romania to reaching the Black Sea.

Taking into account the subdivision of the lower course of the Danube and the Pannonian sector, we have named five sectors of investigation: sector I includes the km from Bratislava to the entrance to Romania, sector II from Baziaş to the Iron Gate, sector III to Călăraşi, sector IV to Brăila, and sector V from Brăila to the Black Sea.

Our study is based on a systematic search of the Web of Science (Clarivate Analytics), Google Scholar, and ResearchGate for articles published between 1950 and 2023. We focused on selected metals, using keywords such as "metals in the Danube", "bioaccumulation in commercial fish", and "Danube pollutants". Publications that included seasonal monitoring, the same sampling sites in different years, and various locations in the same period were taken into account to ensure a comprehensive perspective.

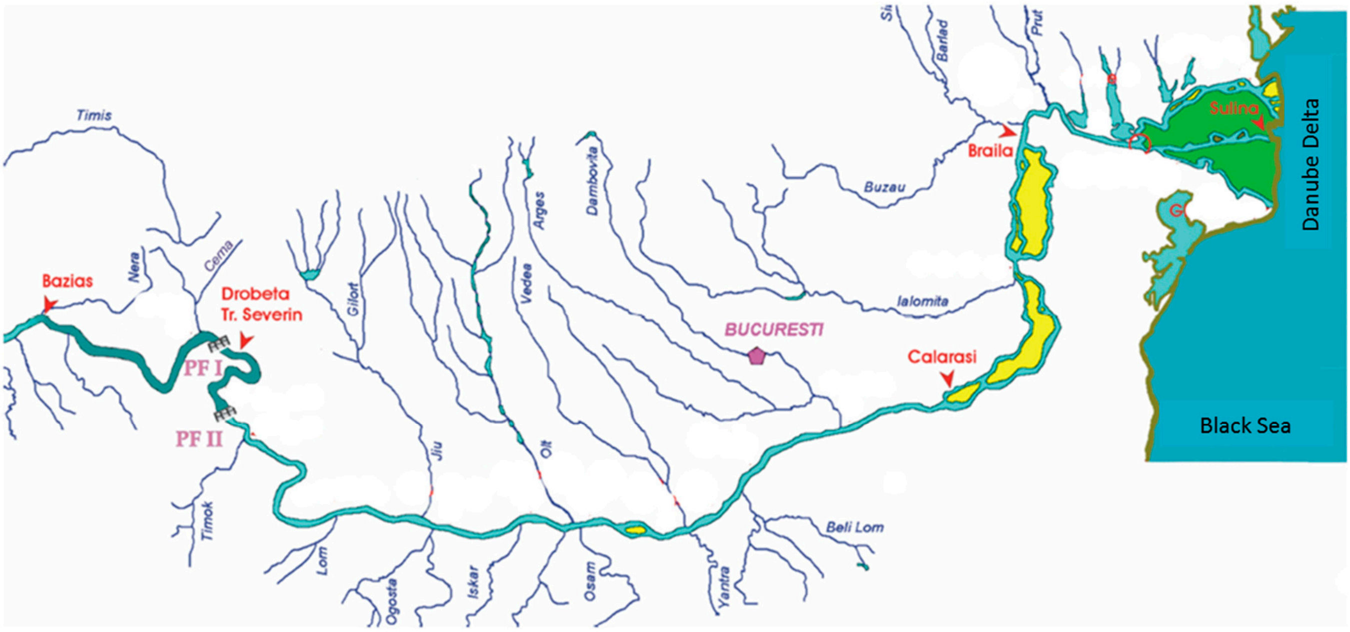


**Figure 2.** Romanian part of the Danube River sector, between Baziaş and the Black Sea [33].

The assessment and interpretation of metal concentrations in the Danube were conducted based on the findings of previous studies [3–5,48,49] and international treaties and agreements such as the International Convention for the Protection of the Danube River [50] (Table 1).

**Table 1.** National water quality criteria and standards for heavy metals in surface water [50].

| Category | Class I | Class II | Class III | Class IV | Class V |
|---|---|---|---|---|---|
| Cd (µg/L) | 0.5 | 1 | 2 | 5 | >5 |
| Co (µg/L) | 10 | 20 | 50 | 100 | >100 |
| Cr total (µg/L) | 25 | 50 | 100 | 250 | >250 |
| Cu (µg/L) | 20 | 30 | 50 | 100 | >100 |
| Ni (µg/L) | 10 | 25 | 50 | 100 | >100 |
| Pb (µg/L) | 5 | 10 | 25 | 50 | >50 |
| Zn (µg/L) | 100 | 200 | 500 | 1000 | >1000 |

Water quality indices are valuable tools for assessing the quality of water. Their origins can be traced back to 1965, when Horton introduced the initial version of the Water Quality Index (WQI). Based on the WQI value, surface water can be classified into five quality classes, as described in Table 2 [51].

**Table 2.** Water quality status based on the Water Quality Index (WQI).

| WQI Values | Status |
|---|---|
| 0–25 | Excellent (I) |
| 26–50 | Good (II) |
| 51–75 | Poor (III) |
| 76–100 | Very poor (IV) |
| >100 | Unsuitable for drinking (V) |

The metal profile in the sediments of the Danube River has a particular configuration. Table 3 presents the standard values of some metal concentrations in the sediment according to Romanian Order 161/2006 [50].

**Table 3.** The standard value of some metal concentrations in the sediment according to Romanian Order 161/2006 [50].

| | **Standard Value (mg/kg)** |
|---|---|
| Cd | 0.8 |
| Cu | 40 |
| Pb | 85 |
| Zn | 150 |
| Ni | 35 |

The consumption of fish meat significantly influences the risk of metal accumulation in humans.

In order to assess the levels of metals in fish and compare them with the maximum allowable concentrations (MACs) established by the European Union [52], the Food and Agriculture Organization of the United Nations [53], and national legislation [54] for safe consumption of fish meat by humans, the concentrations are expressed in milligrams per kilogram (mg/kg) of wet tissue weight (WW). This enables a comprehensive evaluation of the potential risks associated with the presence of these substances in fish destined for human consumption.

The maximum limits for metals in fish muscle (mg/kg) according to international standards are listed in Table 4. The comparison of metal concentrations in fish was also carried out according to the guidelines of the Council of Ministers of the Environment of Romania (Decree No. 356/2001), with maximum limits for metal accumulation in fish tissue (fillets) of 0.2 mg/kg for Pb and 0.05 mg/kg for Cd, but these guidelines exclude Zn and Cu.

**Table 4.** Maximum metal limits in fish muscles according to international standards [55].

| **Organization** | **Metals (mg/kg)** | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | **As** | **Cd** | **Cr** | **Cu** | **Fe** | **Hg** | **Ni** | **Pb** | **Zn** |
| FAO (1983) | 1.0 | 0.05 | 0.15–1.0 | 30 | 100 | 0.5 | 80 | 0.2 | 30 |
| FAO/WHO limit (1989) | | 0.5 | | - | - | | | 0.5 | |
| European Commission Regulation EC No. 1881/2006 | | 0.05 | | - | - | 0.5 | | 0.3 | |
| Decree No. 365/2001 | | 0.05 | | - | - | | | 0.2 | |
| Heavy Metals Regulations Legal Notice No. 66/2003 | | 0.05 | | | | 0.5 | | 0.2 | |

According to EU regulations, the maximum allowable Cd content in fish muscle is 0.05 mg/kg body weight, except in certain marine animals, because Cd is an element that can cause chronic poisoning at a minimum concentration of 1 mg/kg [56]. The literature indicates that Cd concentrations in freshwater fish range from 0.002 g/g in farmed carp to 0.011 g/g in wild fish. The primary source of Cd exposure is the food supply, i.e., highly contaminated fish and fish products. Higher Cd concentrations lead to kidney failure and lung cancer [57,58]. For Pb, fish meat's maximum allowable concentration (MAC) is 0.3 mg/kg body weight. The limit for Zn recommended by the FAO of the United Nations is 30 mg/kg body weight [59]. According to FAO 1983, Heavy Metals Regulations Legal Notice No. 66/2003, and European Commission Regulation EC No. 1881/2006, the established limit for Hg is 0.5 mg/kg [55].

## 3. Results

Factors affecting metal concentration and accumulation in aquatic organisms include metal bioavailability, sampling season, environmental hydrodynamics, size, sex, tissue composition, reproductive cycle [60], and dietary habits [61]. As metabolically active organs, the liver and gills are the target organs for metal accumulation [62], while accumulation in muscle tissue is lower [63–65].

Fish species have different levels of metals in their tissues (muscle, liver, or digestive tract). Pollutants can enter fish through five pathways: feed, non-food particles, gills, oral intake, and skin [66].

Numerous research results show that the distribution of these contaminants depends on the affinity of fish tissues for metals, the degree of uptake and accumulation, and the ability of the organism to excrete them. A major problem with metals is their long biological half-life in living organisms [67]. Accumulated metal concentrations were higher in the gills, liver, and kidneys but lower in muscles, consistent with the essential functions of these tissues. The accumulation of metals in fish tissues, resulting from environmental pollution, poses a potential risk to human health due to their toxic effects on various organ systems. Given the worldwide consumption of fish as a protein source, the ingestion of contaminated fish can lead to adverse health effects associated with metal toxicity [6].

### *3.1. Sediments*

To provide a comprehensive overview of the presence of selected metals in sediments, we have analyzed each sector of the Danube River (Table 5). Following the analyzed scientific works, the maximum concentration of non-essential elements reported in the Pannonian sector of the Danube was declared in 2004 [3] for As (14.73 mg/kg), Cd (4.03 mg/kg), Hg (0.30 mg/kg), and Pb (43.6 mg/kg). Similarly, the study [68] stated higher concentrations of metals in the sediments collected in 2013, with Hg and Pb values of 0.8 $\pm$ 0.09 mg/kg and 64.92 $\pm$ 2.39 mg/kg, respectively, from the Belgrade Region of the Danube River, near Vinča in Grocka. Additionally, in the same studies in this sector, the tendency to accumulate more essential elements was observed in the last decade compared to the previous twenty years.

At the entrance of the Danube to Romania, in terms of As and Cr concentrations, they have increased in the last two decades from 12.68 mg/kg in 2002 [3] to a value of 17.8 mg/kg in 2022 [69] for As, while for Cr, the values increased from 105.9 mg/kg [48] to 183 mg/kg in 2022 [69].

Concerning the concentrations of metals and metalloids in the sediments collected in the sector between Iron Gate and Călăraşi, As concentrations registered values between 7.3 and 12.9 mg/kg [70]. Milenkovic et al. [3] reported, for the same sector, lower values for As concentrations (3.16 mg/kg). Additionally, in the same study, higher values of Cd, Cr, Cu, Hg, Ni, Pb, and Zn were recorded compared to those found in the SIMONA Project [70]. Regarding the Fe concentrations, the studied articles provided no records for the last two decades.

Sector IV, from Călăraşi to Brăila, was covered by studies starting in 2011, making it difficult to obtain an overview of the entire period before Order 161/2006. From 2011 to 2020, the levels of Cd remained relatively consistent, with readings ranging between 0.3 mg/kg [50] and 0.59 mg/kg [50]. According to national standards, these readings were discovered to be lower than the maximum permissible limits.

**Table 5.** The values of metal concentrations in the sediment (mg/kg).

| | Years | As | Cd | Cr | Cu | Fe | Hg | Ni | Pb | Zn | Ref. |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Sector I | 2002 | 5.08 | 2.84 | 51.8 | 23.9 | - | 0.18 | 46.8 | - | - | [3] |
| | | 14.73 | 4.03 | 112.5 | 36.8 | - | 0.3 | 116.4 | - | - | |
| | 2012 | 8.9 | 0.61 | - | 35.95 | 16,104 | 0.69 | - | 32.58 | 139.4 | [66] |
| | 2013 | 13.89 | 1.69 | - | 50.93 | 17,530 | 0.8 | - | 64.92 | 270.4 | [68] |
| Sector II | 2002 | 12.68 | 3.2 | 105.9 | 41 | - | 0.27 | 99.9 | 40.9 | 389.5 | [3] |
| | | 3.15 | 3.79 | 68 | 45.3 | - | 0.19 | 69.9 | 25.8 | 285.7 | |
| | | 9.24 | 2.98 | 93.3 | 57.6 | - | 0.23 | 74.5 | 43.6 | 307.8 | |
| | 2020 | - | - | - | - | 15.36 | - | - | - | - | [71] |
| | 2022 | 17.8 | - | 183 | 56 | - | - | 97 | - | 328 | [69] |
| Sector III | 2002 | 3.16 | 2.12 | 71.1 | 31.6 | - | 0.19 | 59.2 | 28 | 197.5 | [3] |
| | | 0.99 | 2.91 | 30.6 | 17.8 | - | <0.06 | 23.7 | 2.85 | 49.4 | |
| | 2020 | 7.3–8 | 1 | 60–96 | 12.9–29.4 | - | 0.04–0.005 | 38.6–21.4 | 15.3–17.9 | 59.2–59.7 | [70] |
| | | 8.49–12.9 | 0.24–0.3 | 64–65.9 | 23.3–29.2 | - | 0.16–0.25 | 37.4–38.4 | 14.2–18.4 | 60–78.8 | |
| | | - | 0.27–0.4 | 59.4–74.9 | 26.3–29.8 | - | 0.05–0.06 | 32.4–39 | - | - | |
| Sector IV | 2011–2017 | - | 0.38 | 38.94 | 35.18 | - | - | 37.9 | 20.2 | 94.91 | [72] |
| | | - | - | 42.28 | 38.56 | - | - | 33.83 | - | 98.37 | [73] |
| | | - | - | 42.77 | 27.27 | - | - | 32.54 | - | 95.63 | |
| | | - | - | 44.72 | 38.76 | - | - | 40.11 | - | 103.39 | |
| | | - | - | 40.46 | 40.98 | - | - | 35.83 | - | 105.46 | |
| | | - | - | 33.12 | 34.39 | - | - | 34.73 | - | 100.85 | |
| | | - | - | 36.69 | 26.43 | - | - | 32.46 | - | 95.72 | |
| | | - | - | 37.42 | 31.61 | - | - | 35.1 | - | 98.71 | |
| | | - | - | 32.1 | 31.42 | - | - | 34.6 | - | 106.35 | |
| | | - | - | 27.12 | 24.85 | - | - | 32.26 | - | 89.33 | |
| | | - | - | 30.47 | 27.73 | - | - | 27.48 | - | 88.87 | |
| | | - | - | 28.83 | 30.69 | - | - | 30.4 | - | 97.55 | |
| | | - | - | 29.77 | 34.12 | - | - | 34.08 | - | 103.9 | |
| | | - | - | 32.16 | 35.96 | - | - | 34.28 | - | 104.72 | |
| | | - | - | 24.89 | 30.03 | - | - | 33.89 | - | 92.85 | |
| | | - | - | 43.76 | 26.16 | - | - | 33.49 | - | 89.83 | |
| | | - | - | 41.71 | 25.58 | - | - | 31.7 | - | 83.99 | |
| | | - | 0.434 | 29.44 | 34.24 | - | - | 36.52 | 14.45 | 104.3 | [74] |
| | | - | 0.368 | 30.64 | 32.17 | - | - | 35.93 | 14.47 | 98.3 | |
| | | - | 0.396 | 33.55 | 33.23 | - | - | 42.18 | 16.94 | 112 | |
| | | - | 0.401 | 29.8 | 31.93 | - | - | 34.41 | 17.37 | 101 | |
| | 2018 | - | 0.3 | - | 4.3 | - | - | 16.03 | 5.9 | 58.84 | [50] |
| | 2019 | - | 0.59 | - | 10.31 | - | - | 20.17 | 6.02 | 78.69 | |
| | 2020 | 5.5–7.68 | - | 30–41.7 | 28.9–52.2 | - | 0.067–0.1 | 32.1–47.7 | 20.1–26.9 | 70.1–102 | [70] |
| Sector V | 1950 | - | <0.5 | 50 | 38 | - | - | 56 | 36 | 90 | [75] |
| | 1995–1997 | - | 0.5–10 | 18–101 | 2.0–51.0 | - | - | 6.0–78 | 5.0–68 | 6–119 | [76] |
| | 2003–2009 | - | <0.5–1.5 | 7.5–61.9 | 14.8–194 | - | - | 19–111 | 7.5–51.3 | 29.8–218 | [77] |
| | | - | 0.115–1.9 | 20–124 | 3.5–94 | - | - | 11–72.0 | 1.0–73 | 17–202 | [76] |
| | 2007–2012 | - | 6.12–8.26 | 88.1–134.2 | - | - | - | 54.6–79.1 | 7.16–13.99 | 164.2–204.7 | [78] |
| | | - | 4.16–7.81 | 67.2–93.1 | - | - | - | 31.2–67.1 | 6.22–7.61 | 131.2–171.2 | |
| | | - | 4.97–7.88 | 71.5–117.6 | - | - | - | 48.1–69.1 | 6.81–7.89 | 158.2–187.5 | |
| | | - | 4.26–6.81 | 44.9–58.6 | - | - | - | 33.3–54.6 | 7.14–12.2 | 148.2–197.9 | |
| | | - | 2.12–4.92 | 29.4–47.9 | - | - | - | 27.1–46 | 5.18–7.31 | 122.1–168.3 | |
| | | - | 3.47–4.99 | 38.5–52.4 | - | - | - | 28.3–49.5 | 6.02–8.99 | 139.1–178.5 | |
| | 2012–2013 | - | BDL | 7.62–32.5 | 4.65–45.9 | - | - | 10.8–49.8 | 4.76–41.3 | 17.7–93.1 | [53] |
| | | - | 0.59 | - | 10.72 | - | - | 29.12 | 8.96 | 118.54 | [50] |
| | | - | 0.74 | - | 17.39 | - | - | 27.88 | 12.57 | 120.76 | |
| | | - | 0.5 | - | 12.58 | | | 22.31 | 8.49 | 87.43 | |
| | | - | 0.5 | - | 12.54 | - | - | 20.53 | 7.93 | 84.15 | |
| | | - | 0.46 | - | 7.6 | - | - | 14 | 5.17 | 62.39 | |
| | | - | 0.57 | - | 11.81 | - | - | 19.99 | 7.28 | 84.65 | |
| | | - | 0.54 | - | 10.24 | - | - | 22.09 | 21.14 | 77.64 | |
| | | - | 0.54 | - | 7.89 | - | - | 16.28 | 4.84 | 64.48 | |
| | | - | 0.76 | - | 13.42 | - | - | 23.9 | 7.55 | 146.23 | |
| | | - | 0.53 | - | 6.68 | - | - | 24.83 | 7.83 | 85.4 | |
| | | - | 0.63 | - | 9.79 | - | - | 24.76 | 6.84 | 96.11 | |
| | | - | 0.75 | - | 16.64 | - | - | 28.35 | 8.29 | 121.38 | |
| | | - | 0.57 | - | 15.17 | - | - | 38.81 | 8.11 | 117.01 | |
| | | - | 0.53 | - | 9.42 | - | - | 16.94 | 5.34 | 69.97 | |
| | 2019 | - | 0.65 | - | 11.65 | - | - | 19.33 | 6.05 | 84.21 | |
| | | - | 0.99 | - | 25.01 | - | - | 35.8 | 13.78 | 177.33 | |
| | | - | 0.63 | - | 8.97 | - | - | 16.04 | 6.41 | 73.57 | |
| | | - | 0.57 | - | 10.08 | - | - | 24.58 | 5.7 | 95.67 | |
| | | - | 0.41 | - | 7.55 | - | - | 19.09 | 4.17 | 71.27 | |
| | | - | 0.46 | - | 7.47 | - | - | 17.65 | 5.68 | 63.21 | |
| | | - | 0.78 | - | 19.47 | - | - | 28.49 | 10.35 | 131.5 | |
| | | - | 0.46 | - | 9.29 | - | - | 17.4 | 4.31 | 66.06 | |
| | | - | 0.65 | - | 17.18 | - | - | 27.41 | 8.33 | 121.05 | |
| | | - | 0.72 | - | 27.5 | - | - | 50.46 | 14.64 | 161.24 | |
| | | - | 0.77 | - | 20.75 | - | - | 32.55 | 9.87 | 146.53 | |
| | | - | 0.82 | - | 23.29 | - | - | 39.03 | 9.9 | 154.34 | |
| | | - | 0.48 | - | 9.29 | - | - | 25.85 | 6.76 | 81.26 | |
| | | - | 0.52 | - | 10.07 | - | - | 28.13 | 8.01 | 86.43 | |

Also, some authors reported Cu concentrations in the sediments ranging from 26.16 to 40.98 mg/kg for this sector, which falls within the acceptable limits set by national regulations [50,70,73,74]. However, there were a few exceptions to this trend, in particular the minimum value of 4.3 mg/kg [50] and the maximum value of 52.2 [70]. The value of 52.2 mg/kg exceeded the allowable limits, according to international standards.

The Pb values achieved in 2011–2020 are identified as being lower than the 85 mg/kg limit. Regarding Ni, the concentrations have significantly increased over time; the maximum value that exceeded the allowed limit was declared in 2020 [70].

No specific study was identified that reported concentrations of Zn exceeding the permissible limit of 150 mg/kg. The highest value of Zn (106.35 mg/kg) was observed between 2011 and 2017, as reported by [73].

The studies on the last sector of the Romanian Danube, from Brăila to the Black Sea, spanned a more extensive timeframe from 1950 to 2019. As per the records examined in the relevant articles, it was found that most of the concentrations identified in the studies were within the acceptable limit. However, the values reported between 2005 and 2012 exceeded the permissible limits in most cases [58], while other researchers reported acceptable levels of metals [50,53,75–77].

The concentration of metals (Zn, Hg, and Cu) increased until 1989 due to industrialization in Central and Eastern Europe [52], including Romania. However, in the last five decades, a decrease in metal concentrations has been observed in the Danube River due to the management of intensive agricultural programs, which has led to a decline in Cd concentrations as this compound is a component of fertilizers [53].

### 3.2. Water

Between 2007 and 2021, several authors conducted analyses of metal levels in the Danube water, both upstream of the Baziaş entry point and in the sections before Brăila leading to the Black Sea. The results indicated that the Danube contained acceptable levels of essential and non-essential elements, which met the water quality criteria outlined in Class I of the national standards (Table 6).

After passing through ten countries and being subjected to agricultural pollution, industrial pollution, and human impact, the Danube flows into the Black Sea. The metal concentrations in this region showed significant variations, but the water quality was classified as Class II (indicating a good ecological status) [78,79]. The only exception was Zn, where excessive levels were detected in most cases.

**Table 6.** The value of each metal concentration in the water (mg/L).

| | Years | As | Cd | Cr | Cu | Fe | Hg | Ni | Pb | Zn | Ref. |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Sector I | 2012 | 0.004 | ND | - | 0.004 | 0.33 | ND | - | ND | 0.032 | [66] |
| | 2013 | 0.006 | ND | - | 0.004 | 0.41 | ND | - | ND | 0.063 | [68] |
| | 2011–2013 | 0.001<br>0.0006 | 0.00004<br>0.00002 | 0.002<br>0.009 | 0.006<br>0.006 | 0.26<br>0.21 | 0.0001<br>0.0001 | 0.0014<br>0.003 | 0.0006<br>0.0005 | 0.02<br>0.02 | [80] |
| | 2010–2012 | -<br>-<br>- | 0.002<br>-<br>0.008 | -<br>-<br>- | 1.8<br>3.17<br>1.46 | -<br>-<br>- | -<br>-<br>- | -<br>-<br>- | -<br>-<br>2.76 | 1.82<br>1.35<br>0.78 | [81] |
| | 2020 | 0.09<br>0.12<br>0.12<br>0.14<br>0.1<br>0.11<br>0.09<br>0.11<br>0.14 | 0.004<br>0.009<br>0.008<br>0.012<br>0.014<br>0.008<br>0.011<br>0.042<br>0.088 | -<br>-<br>-<br>-<br>-<br>-<br>-<br>-<br>- | -<br>-<br>-<br>-<br>-<br>-<br>-<br>-<br>- | 0.766<br>0.821<br>0.685<br>0.801<br>0.804<br>0.792<br>0.803<br>0.911<br>2.193 | 0.011<br>0.017<br>0.012<br>0.014<br>0.011<br>0.009<br>0.01<br>0.009<br>0.015 | -<br>-<br>-<br>-<br>-<br>-<br>-<br>-<br>- | 0.21<br>0.24<br>0.28<br>0.21<br>0.21<br>0.22<br>0.22<br>0.22<br>0.31 | 21.4<br>18.5<br>20.1<br>18.5<br>17.9<br>19.1<br>19.7<br>19.4<br>18.1 | [71] |
| | 2011–2013 | 0.002 | 0.00002 | 0.007 | 0.006 | 0.08 | 0.00001 | 0.0009 | 0.0005 | 0.009 | [80] |

**Table 6.** *Cont.*

| | Years | As | Cd | Cr | Cu | Fe | Hg | Ni | Pb | Zn | Ref. |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Sector IV | 2010 | - | 18.4 | - | 112.3 | - | - | - | 21.44 | 47.14 | [82] |
| | | - | - | 1.13 | 4.14 | - | - | 2.78 | - | 12.5 | [73] |
| | | - | - | 1.09 | 4.12 | - | - | 2.5 | - | 11.95 | |
| | | - | - | 1.12 | 4.18 | - | - | 2.44 | - | 12.18 | |
| | | - | - | 1.21 | 4.79 | - | - | 2.6 | - | 13.14 | |
| | | - | - | 0.77 | 3.52 | - | - | 1.75 | - | 15.62 | |
| | | - | - | 0.71 | 3.49 | - | - | 1.76 | - | 15.94 | |
| | | - | - | 0.64 | 3.92 | - | - | 1.96 | - | 16.4 | |
| | | - | - | 0.63 | 3.26 | - | - | 1.7 | - | 15.12 | |
| | | - | - | 0.61 | 3.43 | - | - | 1.84 | - | 16.07 | |
| | | - | - | 0.63 | 3.14 | - | - | 1.63 | - | 14.03 | |
| | | - | - | 0.69 | 3.27 | - | - | 1.9 | - | 15.76 | |
| | | - | - | 0.64 | 3.29 | - | - | 1.62 | - | 15.11 | |
| | | - | - | 0.52 | 3.06 | - | - | 1.78 | - | 15.76 | |
| | | - | - | 0.6 | 3.51 | - | - | 1.61 | - | 16.45 | |
| | | - | - | 1.13 | 5.2 | - | - | 2.57 | - | 11.28 | |
| | | - | - | 0.93 | 4.03 | - | - | 2.38 | - | 10.6 | |
| | | - | 0.067 | 0.718 | 3.1 | - | - | 1.82 | 0.784 | 23 | [74] |
| | | - | 0.067 | 0.89 | 3.31 | - | - | 1.94 | 0.781 | 22.1 | |
| | | - | 0.065 | 0.867 | 3.2 | - | - | 1.74 | 0.884 | 23.7 | |
| | | - | 0.063 | 0.848 | 3.26 | - | - | 1.7 | 0.917 | 23.3 | |
| Sector V | 2007–2012 | - | 6.62–10.5 | 43.1–81.2 | - | - | - | 49.6–78.9 | 7.76–11.2 | 161.2–209.8 | [78] |
| | | - | 4.26–8.41 | 31.0–64.2 | - | - | - | 41.2–59.1 | 6.11–7.52 | 138.2–188.7 | |
| | | - | 6.26–8.41 | 35.2–73.1 | - | - | - | 43.4–71.2 | 6.83–8.91 | 146.9–179.6 | |
| | | - | 4.71–8.46 | 30.1–48.2 | - | - | - | 40–59.1 | 6.81–8.96 | 124.5–181.9 | |
| | | - | 3.5–6.46 | 20.6–37.0 | - | - | - | 27.9–46.6 | 6.24–7.21 | 100.2–164.0 | |
| | | - | 3.6–7.32 | 29.4–44.6 | - | - | - | 36.2–52.8 | 6.23–7.81 | 117.7–179.3 | |
| | | - | 8.3 | 81.24 | - | - | - | 86.18 | 42.61 | 333.78 | [79] |
| | | - | 7.11 | 26 | - | - | - | 104.28 | 21.39 | 144.56 | |
| | | - | 7.8 | 28.59 | - | - | - | 67.97 | 24.33 | 230.59 | |
| | | - | 5.82 | 56.8 | - | - | - | 64.05 | 9.29 | 175.95 | |
| | | - | 8.67 | 72.56 | - | - | - | 38.16 | 48.04 | 249.58 | |
| | | - | 6.33 | 32.67 | - | - | - | 31.67 | 34.67 | 124.5 | |
| | | - | 6.27 | 47.67 | - | - | - | 27.15 | 36.02 | 165.54 | |
| | | - | 5.8 | 28.65 | - | - | - | 47.9 | 8.23 | 167.95 | |
| | | - | 9.13 | 71.81 | - | - | - | 41.1 | 32.52 | 310.79 | |
| | | - | 9.33 | 35 | - | - | - | 89.67 | 28.22 | 178.89 | |
| | | - | 7.69 | 47.67 | - | - | - | 39.5 | 29.35 | 187.15 | |
| | | - | 9.26 | 29.4 | - | - | - | 35.85 | 6.81 | 128.35 | |
| | | - | 10.47 | 55.39 | - | - | - | 98.55 | 35.99 | 312.02 | |
| | | - | 9.5 | 56.67 | - | - | - | 92.33 | 11.33 | 197.33 | |
| | | - | 10.09 | 61.27 | - | - | - | 88.82 | 23.21 | 182.27 | |
| | | - | 6.6 | 56.23 | - | - | - | 48.7 | 7.31 | 125.58 | |
| | | - | 11.05 | 73.12 | - | - | - | 54.95 | 39.37 | 209.12 | |
| | | - | 8 | 47.33 | - | - | - | 41.17 | 17 | 172 | |
| | | - | 7.28 | 41.72 | - | - | - | 85.38 | 32.11 | 161.4 | |
| | | - | 4.21 | 58.5 | - | - | - | 47.8 | 6.93 | 150.6 | |
| | | - | 15.7 | - | 93.5 | - | - | - | 14.31 | 32.58 | [82] |
| | | 2 | 0.2 | 1.4 | - | - | 0.21 | 2 | 1.6 | 1.2 | [83] |
| | | 2.1 | 0.19 | 1.3 | - | - | 0.22 | 2.1 | 1.5 | 1.3 | |
| | | 1.9 | 0.21 | 1.2 | - | - | 0.2 | 2.3 | 1.76 | 1.1 | |
| | | 2.2 | 0.055 | - | - | - | 0.2 | 2.1 | 1.5 | 1.2 | |
| | | 2.4 | 0.056 | - | - | - | 0.16 | 2.1 | 1.5 | 1.3 | |
| | | 2.3 | 0.054 | - | - | - | 0.18 | 2.3 | 1.5 | 1.1 | |
| | | 2.9 | 0.28 | - | - | - | 0.2 | 2 | 1.5 | - | |
| | | 2.8 | 0.11 | - | - | - | 0.16 | 2.1 | 1.5 | - | |
| | | 2.7 | 0.21 | - | - | - | 0.18 | 2.3 | 1.5 | - | |
| | 2018 | - | 0.243 | - | 5.7 | 722.65 | - | 7.2 | 3.67 | 16.27 | [84] |
| | | - | 0.158 | - | 9.59 | 1244.7 | - | 5.65 | 2.76 | 38.9 | |
| | | - | 0.4 | <1.3 | <1–2.9 | - | - | <1 | <0.75 | <2.1–14.9 | [85] |
| | 2020 | - | 0.4–0.9 | <1.3 | 0.9–1.3 | - | - | <1–1.2 | <0.75 | <2.1–6.3 | |
| | 2021 | - | 0.4–0.8 | <1.3 | 0.9–2.8 | - | - | 0.9–1.8 | <0.75–1.1 | 3.3–11.6 | |

ND—not detected.

### 3.3. Fish Tissues

The accumulation of metals in different tissues varies depending on their physiological functions. Fish gills and digestive tracts have a high capacity for metal accumulation, with levels influenced by the concentrations of metals in the water and food. In comparison, muscle tissue accumulates lower levels of metals and is commonly used to assess water pollution and associated health risks related to fish consumption [86].

Gills are important entry points for essential elements, such as Cu, Zn, Se, Mn, and Fe, and non-essential elements, including Al, As, Cd, Cr, and Pb [87]. Furthermore, the analysis of gills can be utilized to evaluate bioaccumulation levels. For example, the carp gills can accurately reflect metal pollutants in water, as the negatively charged gill attracts positively charged metal species in water [87–90]. Therefore, monitoring metal content in carp gills can be important for assessing water quality and potential risks to public health.

However, some studies on common carp showed that Pb, Cd, and As concentrations in the digestive tract and liver were higher than in muscle [68].

The liver and kidney are commonly studied in bioaccumulation research [91,92]. It is important to note that the liver is a vital organ responsible for detoxification and is particularly prone to metal accumulation [54].

Studies conducted on various fish species have shown that elements accumulate mainly in metabolic organs such as the liver, which produces metal-binding proteins [93] (Table 7).

**Table 7.** The value of each metal concentration in the fish tissue (μg/g).

| Tissues | As | Cd | Cr | Cu | Fe | Hg | Ni | Pb | Zn | Years | Ref. | Sect |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Common carp (*Cyprinus carpio*) | | | | | | | | | | | | |
| M | 0.01 | 0.01 | - | - | - | 0.24 | - | 0.048 | - | 2013 | [68] | I |
| | 0.258 | 0.059 | - | 0.688 | 9.38 | 0.393 | - | 0.059 | 6.16 | 2013 | [94] | I |
| | 0.333 | 0.082 | - | 0.757 | 9.68 | 0.466 | - | 0.084 | 6.17 | 2013 | [94] | I |
| | 0.66 | 0.005 | 0.01 | 1.3 | 19.62 | 0.89 | - | - | 59.01 | 2010 | [95] | I |
| | 0.395 | - | - | - | 7.42 | - | - | - | 54.70 | 2010 | [96] | I |
| | 0.66 | - | - | - | - | 0.89 | - | - | 54.23 | 2010 | [48] | I |
| | 0.055 | 0.016 | - | - | - | 0.234 | - | 0.014 | - | 2010 | [97] | I |
| | 0.013 | 0.014 | - | - | - | 0.207 | - | 0.036 | - | 2012 | [66] | I |
| | - | 0.01 | - | - | - | 0.5 | - | 0.16 | - | 2018 | [96] | I |
| | - | 0.084 | - | 5.10 | - | - | - | 0.58 | 42.2 | 2008 | [98] | IV |
| | - | 0.010 | - | 3.22 | - | - | - | 0.38 | 39.20 | 2008 | [98] | V |
| | 0.412 | - | 1.1435 | 1.291 | - | 0.2315 | 1.6855 | 0.1445 | - | 2013–2014 | [99] | V |
| | - | - | - | - | - | 0.054 | - | - | - | 2014 | [85] | V |
| Int | 0.02 | 0.10 | - | - | - | 0.22 | - | 0.21 | - | 2013 | [68] | I |
| | 0.016 | 0.103 | - | - | - | 0.207 | - | 0.266 | - | 2012 | [66] | I |
| | - | - | - | 13.5 | - | - | 0.9 | - | 644 | 2003–2013 | [100] | V |
| G | 0.29 | 0.03 | 0.01 | 1.90 | 139.26 | 0.89 | - | - | 1186.3 | 2010 | [97] | I |
| | - | - | - | - | 261.97 | - | - | - | 1773.76 | 2010 | [96] | I |
| | - | - | - | 4.33 | - | - | <0.03 | - | 674 | 2003–2013 | [101] | V |
| | - | - | - | 6.13 | - | - | 9.5 | - | 872 | 2003–2013 | [101] | V |
| Skin | - | - | - | 3.98 | - | - | <0.03 | - | 263 | 2003–2013 | [101] | V |
| | - | - | - | 3.15 | - | - | 12.9 | - | 485 | 2003–2013 | [101] | V |
| L | 0.49 | 0.28 | 0.01 | 33.49 | 141.44 | 1.63 | - | - | 325.377 | 2010 | [95] | I |
| | - | - | - | 21.97 | 418.36 | - | - | - | 582.79 | 2010 | [96] | I |
| | 0.02 | 0.13 | - | - | - | 0.22 | - | 0.06 | - | 2013 | [68] | I |
| | 0.018 | 0.132 | - | - | - | 0.206 | - | 0.047 | - | 2012 | [66] | I |
| | 0.48 | - | - | - | - | 0.63 | - | - | 325.37 | 2010 | [48] | I |
| | - | - | - | 73.3 | - | - | 4.41 | - | 450 | 2003–2013 | [101] | V |
| | - | - | - | 110 | - | - | 8.17 | - | 243 | 2003–2013 | [101] | V |
| | - | - | - | 89.9 | - | - | 2.46 | - | 189 | 2003–2013 | [101] | V |
| Gon | - | - | - | - | 6.28 | - | - | - | 74.53 | 2010 | [96] | I |
| Crucian carp (*Carassius carassius*, *Carassius gibelio*) | | | | | | | | | | | | |
| M | 0.031 | 0.017 | - | - | - | 0.087 | - | 0.052 | - | 2010 | [97] | I |
| | 0.139 | 0.057 | - | 0.809 | 8.05 | 0.094 | - | 0.030 | 11.16 | 2013 | [94] | I |
| | 0.172 | 0.051 | - | 0.824 | 7.25 | 0.139 | | 0.040 | 10.26 | 2013 | [94] | I |
| | 0.6045 | - | 1.345 | 1.4115 | - | 0.3 | 2.2465 | 0.1835 | - | 2013–2014 | [99] | V |
| | 0.5985 | - | 1.2085 | 1.5005 | - | 0.267 | 1.994 | 0.1675 | - | 2013–2014 | [99] | V |
| | - | - | - | - | - | 0.0255 | - | - | - | 2014 | [79] | V |
| | - | - | - | - | - | 0.073 | - | - | - | 2014 | [79] | V |
| | - | - | - | 0.715 | - | 0.15 | - | - | 11.72 | 2007 | [102] | V |

**Table 7.** *Cont.*

| Tissues | As | Cd | Cr | Cu | Fe | Hg | Ni | Pb | Zn | Years | Ref. | Sect |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Freshwater bream (*Abramis brama*) | | | | | | | | | | | | |
| M | 0.16 | 0.004 | 0.23 | 0.2 | 2.31 | 0.16 | 0.02 | 0.21 | 3.15 | 2011–2013 | [80] | I + II |
| | 0.21 | 0.004 | 0.2 | 0.14 | 2.32 | 0.08 | 0.02 | 0.25 | 3.9 | 2011–2013 | [80] | I + II |
| | 0.15 | 0.004 | 0.26 | 0.17 | 1.66 | 0.15 | 0.05 | 0.23 | 4.64 | 2011–2013 | [80] | I + II |
| | 1.73 | - | 0.26 | 1.13 | 9.72 | - | - | 0.08 | - | 2014 | [103] | I |
| | 0.46 | - | 0.30 | 1.41 | 14.23 | - | - | ND | - | 2014 | [103] | I |
| | ND | - | 0.33 | 0.66 | 27.64 | - | - | ND | - | 2014 | [103] | I |
| | ND | - | 0.19 | 1.49 | 15.40 | - | - | ND | - | 2014 | [103] | I |
| | ND | - | - | - | 1.31 | - | - | - | 23.84 | 2010 | [96] | I |
| | 0.035 | 0.018 | - | - | - | 0.237 | - | 0.030 | - | 2010 | [97] | I |
| | 0.109 | 0.021 | - | 0.707 | 13.6 | 0.110 | - | 0.019 | 9.06 | 2013 | [94] | I |
| | 0.154 | 0.027 | - | 0.717 | 13.54 | 0.161 | - | 0.028 | 9.02 | 2013 | [94] | I |
| | - | 0.01 | - | - | - | 0.17 | - | 0.08 | - | 2018 | [96] | I |
| | - | 0.053 | - | 2.77 | - | - | - | 0.27 | 33.27 | 2008 | [98] | IV |
| | - | - | - | 2.15 | - | - | - | 0.29 | 35.77 | 2008 | [98] | V |
| | 0.2375 | - | 0.962 | 1.889 | - | 0.2395 | 2.6735 | 0.1685 | - | 2013–2014 | [99] | V |
| | - | - | - | - | - | 0.02–0.035 | - | - | - | 2014 | [79] | V |
| L | 0.22 | - | 0.25 | 44.25 | 177.56 | - | - | 0.20 | - | 2014 | [103] | I |
| | 0.14 | - | 0.18 | 22.07 | 225.11 | - | - | 0.05 | - | 2014 | [103] | I |
| | ND | - | 0.20 | 44.06 | 190.35 | - | - | 0.47 | - | 2014 | [103] | I |
| | ND | - | 0.21 | 64.66 | 177.74 | - | - | 0.11 | - | 2014 | [103] | I |
| | ND | - | - | 14.14 | 213.53 | - | - | - | 66.77 | 2010 | [96] | I |
| G | ND | - | 1.32 | 2.11 | 428.22 | - | - | 0.23 | - | 2014 | [103] | I |
| | 3.2 | - | 0.91 | 0.85 | 158.45 | - | - | 0.12 | - | 2014 | [103] | I |
| | ND | - | 3.71 | 1.76 | 167.64 | - | - | ND | - | 2014 | [103] | I |
| | ND | - | 0.76 | 0.98 | 117 | - | - | ND | - | 2014 | [103] | I |
| | ND | - | - | ND | 369.21 | - | - | - | 59.85 | 2010 | [96] | I |
| Gon | 1.75 | - | 0.08 | 1.16 | 20.67 | - | - | 0.08 | - | 2014 | [103] | I |
| | 1.21 | - | 0.18 | 1.54 | 33.80 | - | - | 0.08 | - | 2014 | [103] | I |
| | 1.61 | - | 0.10 | 1.31 | 28.12 | - | - | 0.09 | - | 2014 | [103] | I |
| | 1.04 | - | 0.21 | 0.81 | 29.26 | - | - | ND | - | 2014 | [103] | I |
| | 0.15 | - | - | ND | 19.24 | - | - | - | 137.76 | 2010 | [1] | I |
| Grass carp (*Ctenopharyngodon idella*) | | | | | | | | | | | | |
| M | 0.039 | 0.018 | - | - | - | 0.367 | - | 0.034 | - | 2010 | [97] | I |
| W.B. | 48.5 | - | 0.03 | - | 11.6 | <7.3 | 109 | - | 15.4 | 2011 | [104] | I |
| Silver carp (*Hypophthalmichthys molitrix*) | | | | | | | | | | | | |
| M | ND | - | - | ND | 12.53 | - | - | - | 31.90 | 2010 | [96] | I |
| | 0.018 | 0.015 | - | - | - | 0.441 | - | 0.015 | - | 2010 | [97] | I |
| | 0.036 | 0.014 | - | - | - | 0.140 | - | 0.048 | - | 2012 | [66] | I |
| | 0.04 | 0.01 | - | - | - | 0.16 | - | 0.056 | - | 2013 | [68] | I |
| L | 0.08 | 0.21 | - | - | - | 0.20 | - | 0.14 | - | 2013 | [68] | I |
| | 0.21 | - | - | 188.78 | 511.72 | - | - | - | 222.40 | 2010 | [96] | I |
| | 0.073 | 0.191 | - | - | - | 0.185 | - | 0.125 | - | 2012 | [66] | I |
| G | ND | - | - | ND | 211.88 | - | - | - | 73.72 | 2010 | [96] | I |
| Int | 0.065 | 0.062 | - | - | - | 0.253 | - | 1.518 | - | 2012 | [66] | I |
| | 0.07 | 0.07 | - | - | - | 0.26 | - | 1.3 | - | 2013 | [68] | I |
| W.B. | 58.7 | - | 0.24 | - | 51.4 | <13 | 284 | - | 82.5 | 2011 | [105] | I |
| Pontic shad (*Alosa immaculata*) | | | | | | | | | | | | |
| M | 7.725 | 0.433 | - | 4.074 | 40.346 | - | - | - | 66.098 | 2007–2008 | [65] | III |
| | 12.6 | 0.17 | - | 3.45 | 143.26 | - | - | 0.13 | 58.4 | 2010 | [106] | III |
| | 6.53 | 0.31 | - | 15.5 | 269 | - | - | 0.35 | 44.1 | 2010 | [106] | III |
| | - | 0.091 | - | 5.34 | - | - | - | 0.65 | 44.55 | 2008 | [98] | IV |
| | - | 0.012 | - | 3.3 | - | - | - | 0.45 | 41.45 | 2008 | [98] | V |
| L | 9.4 | 20.8 | - | 34.2 | 1225 | - | - | 0.27 | 83.5 | 2010 | [106] | III |
| | 6.389 | 0.714 | - | 20.003 | 751.814 | - | - | - | 99.759 | 2007–2008 | [65] | III |
| G | 1.63 | 0.219 | - | 2.987 | 289.506 | - | - | - | 80.507 | 2007–2008 | [65] | III |

**Table 7.** *Cont.*

| Tissues | As | Cd | Cr | Cu | Fe | Hg | Ni | Pb | Zn | Years | Ref. | Sect |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Wels catfish (*Silurus glanis*) | | | | | | | | | | | | |
| M | 0.1 | 0.09 | 0.145 | 0.07 | 0.95 | 0.33 | 0.074 | 0.17 | 7.62 | 2011–2013 | [80] | I + II |
| | 0.09 | 0.001 | 0.13 | 0.07 | 1.33 | 0.2 | 0.016 | 0.18 | 2.97 | 2011–2013 | [80] | I + II |
| | 0.11 | 0.004 | 0.14 | 0.07 | 0.55 | 0.62 | 0.03 | 0.16 | 3 | 2011–2013 | [80] | I + II |
| | ND | ND | ND | ND | ND | ND | ND | ND | 7.91 | 2010 | [96] | I |
| | 0.034 | 0.02 | - | - | - | 0.235 | - | 0.032 | - | 2010 | [97] | I |
| | 0.22 | 0.01 | 0.08 | 1.42 | 27.06 | 1.63 | - | - | 20.81 | 2011 | [97] | I |
| | 0.003 | 0.008 | - | - | - | 0.327 | - | 0.014 | - | 2012 | [66] | I |
| | 0.131 | 0.004 | 0.138 | 0.949 | 19.46 | 1.598 | 0.120 | 0.006 | 19.62 | 2013 | [100] | I |
| | 0.003 | 0.01 | - | - | - | 0.53 | - | 0.06 | - | 2013 | [68] | I |
| | 0.160 | 0.068 | - | 1.55 | 8.32 | 0.208 | - | 0.058 | 7.06 | 2013 | [94] | I |
| | 0.211 | 0.069 | - | 1.62 | 8.17 | 0.260 | - | 0.069 | 6.68 | 2013 | [94] | I |
| | 0.7635 | - | 3.638 | 1.9455 | - | 0.235 | 1.823–4.089 | 0.145–0.382 | - | 2013–2014 | [99] | V |
| | - | - | - | - | - | 0.014–0.042 | - | - | - | 2014 | [79] | V |
| G | ND | - | - | 160.11 | 54.93 | - | - | - | 53.77 | 2010 | [96] | I |
| | 0.16 | 0.07 | 0.06 | 1.98 | 74.88 | 1.50 | - | - | 58.05 | 2010 | [97] | I |
| | 0.008 | - | 0.270 | 4.460 | 163.0 | 0.328 | 0.170 | 0.236 | 80.42 | 2013 | [107] | I |
| | 0.117 | 0.005 | 0.090 | 0.412 | 43.98 | 0.071 | 0.211 | 0.387 | 69.81 | 2013 | [100] | I |
| L | ND | - | - | ND | 412.29 | - | - | - | 38.69 | 2010 | [96] | I |
| | 0.24 | 0.02 | 0.04 | 8.37 | 396.16 | 1.90 | - | - | 41.52 | 2010 | [97] | I |
| | 0.004 | 0.064 | - | - | - | 0.143 | - | 0.034 | - | 2012 | [66] | I |
| | 0.096 | - | 0.010 | 17.77 | 745.7 | 0.639 | 0.060 | 0.067 | 93.14 | 2013 | [107] | I |
| | 0.005 | 0.12 | - | - | - | 0.23 | - | 0.10 | - | 2013 | [68] | I |
| Int | 0.005 | 0.101 | - | - | - | 0.167 | - | 0.036 | - | 2012 | [66] | I |
| | 0.006 | 0.14 | - | - | - | 0.30 | - | 0.10 | - | 2013 | [68] | I |
| Skin | 0.360 | 0.005 | 0.154 | 1.902 | 25.86 | 0.657 | 0.244 | 0.009 | 58.95 | 2013 | [100] | I |
| W.B | < 29 | - | 0.062 | - | 13.7 | <3.1 | 84 | - | 12.4 | 2011 | [105] | I |
| Gon | 0.960 | - | 0.100 | 2.200 | 99.29 | 0.114 | 0.220 | 0.018 | 134.3 | 2013 | [107] | I |
| Pike-perch (*Sander lucioperca*) | | | | | | | | | | | | |
| M | 0.11 | 0.003 | 0.18 | 0.09 | 0.81 | 0.15 | 0.01 | 0.25 | 2.74 | 2011–2013 | [80] | I + II |
| | 0.13 | 0.04 | 0.18 | 0.11 | 2.35 | 0.3 | 0.016 | 0.23 | 3.76 | 2011–2013 | [80] | I + II |
| | 0.15 | 0.002 | 0.11 | 0.11 | 4.63 | 0.28 | 0.08 | 0.18 | 3.07 | 2011–2013 | [80] | I + II |
| | 0.032 | 0.018 | - | - | - | 0.173 | - | 0.043 | - | 2010 | [97] | I |
| | 0.17 | 0.005 | 0.043 | 0.75 | 17.97 | 1.32 | - | - | 15.14 | 2010 | [95] | I |
| | 0.199–0.219 | - | 0.708–1.587 | 0.979–1.244 | - | 0.205–0.223 | 2.084–3.543 | 0.091–0.143 | - | 2013–2014 | [99] | V |
| | - | - | - | - | - | 0.016–0.041 | - | - | - | 2014 | [79] | V |
| L | 0.50 | 0.02 | 0.04 | 6.18 | 241.07 | 1.66 | - | - | 58.37 | 2010 | [95] | I |
| G | 0.25 | 0.01 | 0.02 | 1.01 | 73.01 | 1.52 | - | - | 40.11 | 2010 | [95] | I |
| Barbel (*Barbus barbus*) | | | | | | | | | | | | |
| M | 0.84 | - | - | 4.8 | 11.12 | 2.15 | - | - | 10.06 | 2010 | [108] | I |
| | 1.4 | - | - | ND | ND | - | - | - | 12.89 | 2010 | [109] | I |
| | 1.57 | - | 0.41 | 1.90 | - | 0.27 | 0.19 | 0.11 | 18.37 | 2012 | [110] | I |
| | 0.189 | 0.052 | - | 0.826 | 12.22 | 0.222 | - | 0.048 | 5.2 | 2013 | [94] | I |
| | 0.239 | 0.062 | - | 0.839 | 11.91 | 0.325 | - | 0.062 | 6.02 | 2013 | [94] | I |
| L | 1.488 | 0.014 | - | 19.63 | 74.81 | - | 14.88 | - | 25.65 | 2010 | [108] | I |
| | 0.54 | - | - | 27.49 | 78.82 | - | - | - | 47.08 | 2010 | [109] | I |
| | 1.74 | - | 0.37 | 25.85 | - | 0.09 | 0.16 | 0.12 | 59.50 | 2012 | [110] | I |
| G | 0.001 | - | - | 12.15 | 106.22 | 1.187 | 0.96 | - | 40.59 | 2010 | [108] | I |
| | 0.59 | - | - | ND | 120.91 | - | - | - | 47.85 | 2010 | [109] | I |
| | 0.85 | - | 0.64 | 2.68 | - | 0.02 | 0.22 | ND | 68.33 | 2012 | [110] | I |
| Gon | 1.41 | - | - | ND | ND | - | - | - | 71.69 | 2010 | [109] | I |
| M | - | 0.01 | - | - | - | 0.09 | - | 0.15 | - | 2018 | [111] | I |
| Int | 1.80 | - | 0.57 | 5.81 | - | ND | 0.33 | 0.37 | 49.76 | 2012 | [110] | I |

**Table 7.** *Cont.*

| Tissues | As | Cd | Cr | Cu | Fe | Hg | Ni | Pb | Zn | Years | Ref. | Sect |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Northern pike (*Esox lucius*) | | | | | | | | | | | | |
| M. | 0.030 | 0.015 | - | - | - | 0.236 | - | 0.036 | - | 2010 | [97] | I |
| | 0.105 | 0.023 | - | 0.548 | 10.10 | 0.106 | - | 0.032 | 5.10 | 2013 | [94] | I |
| | 0.153 | 0.036 | - | 0.574 | 9.97 | 0.162 | - | 0.037 | 5.17 | 2013 | [94] | I |
| | - | 0.044 | - | 2.9 | - | - | - | 0.36 | 23.9 | 2008 | [98] | IV |
| | - | - | - | 1.5 | - | - | - | 0.26 | 21.92 | 2008 | [98] | V |
| | 0.173–1.199 | - | 0.781–2.071 | 0.901–2.696 | - | 0.182–0.428 | 1.892–3.601 | 0.060–0.270 | - | 2013–2014 | [99] | V |
| | - | - | - | - | - | 0.021–0.058 | - | - | - | 2014 | [79] | V |
| European perch (*Perca fluviatilis*) | | | | | | | | | | | | |
| M | 1.00 | ND | 0.09 | 0.45 | 11.85 | 2.72 | ND | - | 18.89 | 2011 | [111] | I |
| | - | 0.034 | - | 3.85 | - | - | - | 0.19 | 32.33 | 2008 | [98] | IV |
| | - | - | - | 1.25 | - | - | - | 0.33 | 32.36 | 2008 | [98] | V |
| | 0.150–0.341 | - | 0.248–3.063 | 1.588–2.319 | - | 0.190–0.417 | 1.252–3.353 | 0.182–0.582 | - | 2013–2014 | [99] | V |
| | - | - | - | - | - | 0.012–0.038 | - | - | - | 2014 | [79] | V |
| | - | - | - | 0.26–0.37 | - | 0.29–0.35 | - | - | 6.13–6.36 | 2007 | [102] | V |
| L | 3.03 | ND | 0.11 | 18.20 | 225 | 2.52 | ND | - | 77.66 | 2011 | [111] | I |
| G | 1.11 | ND | 0.25 | 0.66 | 189.39 | 1.84 | ND | - | 64.82 | 2011 | [111] | I |
| Gon | 1.43 | ND | 0.11 | 2.21 | 53.98 | 1.19 | ND | - | 68.06 | 2011 | [111] | I |

M—muscle, L—liver, Gon—gonads, G—gills, Int—intestine, W.B.—whole body, ND—not detected.

Lenhardt et al. conducted studies on the levels of metals and trace elements in the tissues of freshwater fish in the Danube River and found that carp had higher levels of Zn in their liver, muscle, and gill samples than catfish, while catfish had higher levels of Mn in their gill samples than carp [96]. These differences in metal bioaccumulation between the two species could be due to their feeding habits, physiology, and habitat variations.

Overall, the results confirm the differences in elemental accumulation in different tissues. The highest concentrations of Cu, Fe, Mn, and Zn were found in the liver, which agrees with other studies [112–117].

On the other hand, Hg concentrations are highest in muscles [116–118]. Muscle is not an active tissue for element accumulation, as reported in many studies [112,119,120]. For example, Figure 3 shows the bioaccumulation trends of As, Cd, and Pb in the muscle tissues of certain fish species upon their entry into the river sector of Romania (2010–2013).

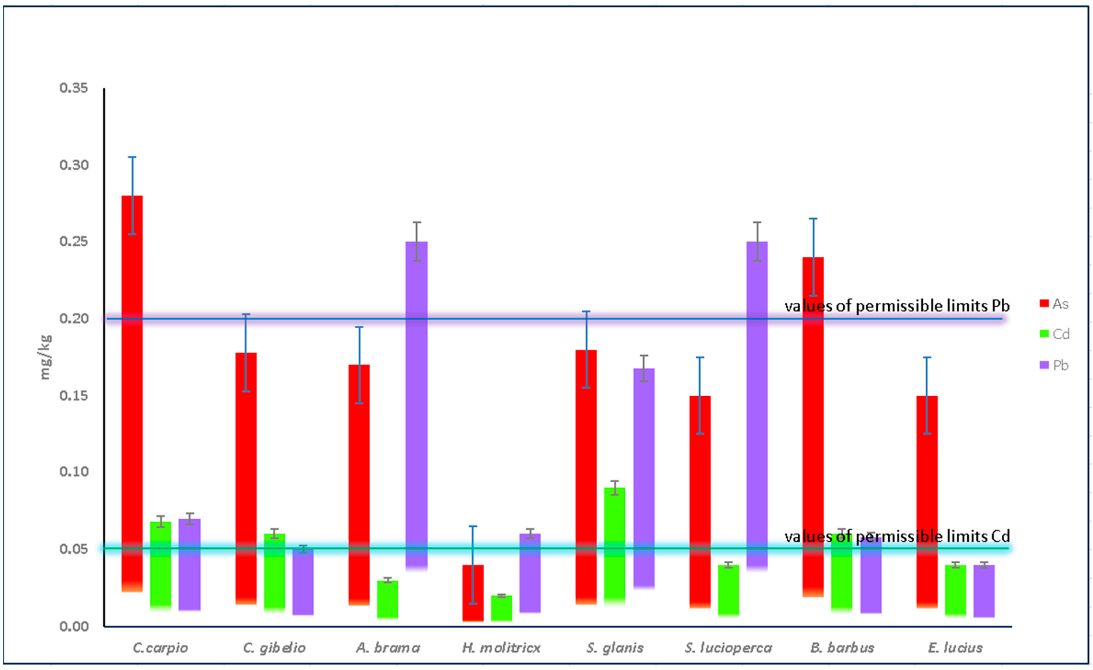


**Figure 3.** Bioaccumulation of As, Cd, and Pb in the muscle of some fish species.

Metal accumulation was lower in muscle than in other fish organs such as the gills, skin, and liver [121]. Although muscle tissue may not always accurately measure metal contamination in fish, it remains a crucial concern as it is part of the fish most commonly consumed by humans.

Lower Cu, Fe, manganese, and Zn concentrations in muscle tissue may be due to lower amounts of binding proteins in muscle tissue [122].

Compared to other species, the lower Hg concentration in carp muscle tissue can be attributed to biomagnification in the food chain [123]. This means that as Hg travels up the food chain, it accumulates in higher concentrations in the bodies of organisms at higher trophic levels.

In Figure 4, the analysis of muscle tissue samples taken from Freshwater bream (*Abramis brama*), Wels catfish (*Silurus glanis*), and Pike-perch *(Sander lucioperca*) collected from sectors I and II reveals some noteworthy findings. Although all values for As, Cr, Cu, and Pb fall within acceptable limits, the *Abramis brama* samples exhibit relatively higher concentrations than the other species. Conversely, regarding Hg and Pb, the muscle tissue samples from *Sander lucioperca* demonstrate slightly elevated levels, which still meet acceptable standards. Notably, the samples taken from *Silurus glanis* in sector I exhibit the highest values for Fe and Zn, measuring 19.46 mg/kg and 19.62 mg/kg, respectively.

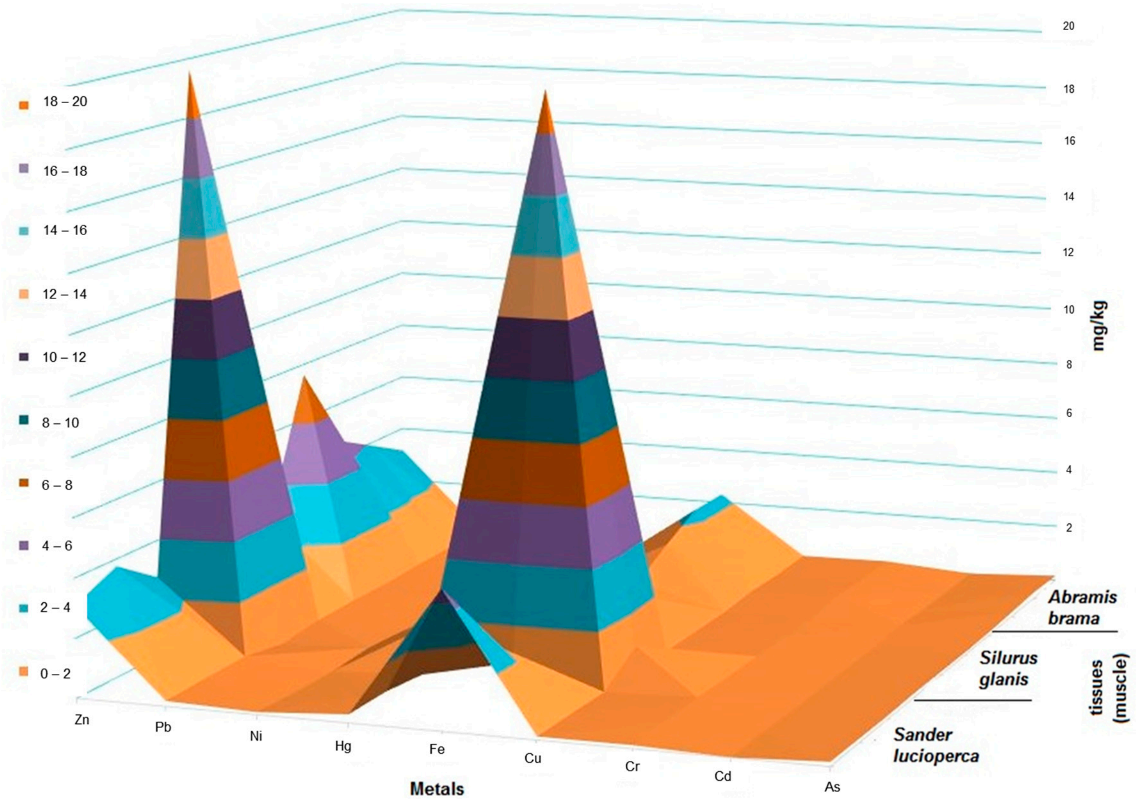


**Figure 4.** Metal bioaccumulation in some fish muscle (sectors I and II).

Metal contamination in fish tissue has been linked to various health issues, including neurological problems, kidney damage, and cancer [124]. Thus, to ensure the safety of fish as a food source and to protect public health, regular monitoring of metal content in fish tissue is crucial [125,126]. Moreover, it is essential to note that the level of metal contamination can vary significantly depending on the fish's species, age, location, and type of metal involved. Therefore, a comprehensive and continuous monitoring system is necessary to identify potential risks and ensure the safety of fish consumption.

## 4. Discussion

Several industrial accidents in the Romanian Carpathian region, where there is a long mining tradition, especially for gold, silver, Pb, Zn, Cu, Cd, and Mn, have led to pollution of the Danube River [127,128]. Toxic elements such as As, Cd, Hg, and Pb, as well as essential trace elements such as Ca, Co, Cu, Fe, Mn, molybdenum (Mo), nickel (Ni), selenium (Se), and Zn, are toxic to organisms at high concentrations, according to Barlas [129] and Lopez Alosno et al. [130].

While some of these elements are essential for human metabolism [131], they can be divided into potentially toxic (aluminum (Al), As, Cd, Pb, Hg, etc.), possibly essential (Ni, vanadium, and Co), and elemental (Cu, Zn, and Se) [132,133]. When metals are ingested through long-term food consumption, they can accumulate in the body, causing damage to the liver (hepatotoxicity), kidneys (nephrotoxicity), central nervous system (neurotoxicity), and DNA (genotoxicity) [134,135]. Metals have unique physicochemical properties and exhibit variable tissue distribution and bioaccumulation in fish [136,137].

Fish that contain metals can cause serious health problems, such as developmental disorders, neurological disorders, liver and kidney damage, reproductive and hematological effects, cancer, and cardiovascular disease, if the exposure concentrations are exceeded [138–145]. As a result, many researchers have recently focused on the potential risk posed by contaminated fish to consumers health.

Studies conducted by Jarup [146] and Ko [147] have indicated that consuming fish containing elevated levels of metals, including Hg, Cd, As, and Pb, can potentially result in severe skin diseases and autism in children. The disparity in metal exposure may be attributed to variations in dietary habits, with coastal populations consuming more fish than inland populations.

While chromium and Cu are essential for good health, excessive intake can lead to liver and kidney damage. The hexavalent form of chromium is toxic when ingested and has been classified as carcinogenic by the International Agency for Research on Cancer (IARC). The maximum allowable Fe content in fish, as determined by [59], is set at 100 mg/g. Ni is a significant pollutant in aquatic environments, and its toxicity can cause respiratory cancer and harm the immune and reproductive systems.

From 2008 to 2013, a study conducted on fish species such as Common carp (*Cyprinus carpio*), Crucian carp (*Carassius carassius*), Silver carp (*Hypophthalmichthys molitrix*), Wels catfish (*Silurus glanis*), and Barbel (*Barbus barbus*), specifically in sector I, revealed a significant accumulation of Cd, particularly in the muscle tissue. Furthermore, in sectors closer to the point where the Danube River flows into the Black Sea, Cd levels exceeded the optimal accepted threshold between 2007 and 2010. This excessive Cd content was observed not only in the muscle tissue but also in the liver and gonads of the fish.

The elevated concentrations of Cd in the samples of fish in sector I could be caused by the sediment samples, which displayed increased metal values collected between 2002 and 2013 [3,66].

Secondly, fish belonging to the *Cyprinidae* family, such as Common carp (*Cyprinus carpio*) and Crucian carp (*Carassius carassius*), primarily consume phytoplankton during the fry period. As phytoplankton can accumulate Cd, it is plausible that the phytoplanktonophagous nature of these stages of carps led to higher Cd levels in their tissues. The Common carp (*Cyprinus carpio*) is an omnivorous fish that feeds on various food sources, including detritus, chironomids, mollusks, amphipods, zooplankton, and epiphytes [48].

Moreover, fish species such as catfish and barbel, which spend a significant part of their lives dwelling at the bottom of the water, near the shore, and in the sand, exhibited Cd concentrations exceeding the accepted level. This behavior makes them more susceptible to contact with sediments and substances, including Cd, in these environments.

It is worth noting that all other samples detailed in Table 6 showed Cd concentrations below the accepted level, indicating that the bioaccumulation of Cd was predominantly observed in the fish species mentioned above and in specific areas rather than being a widespread issue across all samples.

Between 2010 and 2014, studies reported increased accumulation of Cu in liver samples obtained from Common carp (*Cyprinus carpio*), Freshwater bream (*Abramis brama*), and Silver carp (*Hypophthalmichthys molitrix*) samples in sector I [96,97,103]. Additionally, elevated Cu levels were observed in *Silurus glanis* gonads during 2010 in the same sector. Surprisingly, these accumulations cannot directly correlate with the Cu values measured in water and sediments from the corresponding study area.

Despite the lack of a clear correlation between Cu bioaccumulation in fish tissues and the Cu levels found in water and sediments, several factors could contribute to this disparity. Various complex interactions within the aquatic environment may influence Cu accumulation in fish organs. Factors such as the bioavailability and speciation of Cu in the water column and the fish species' physiology and dietary habits may play significant roles in the differential accumulation patterns observed.

Moreover, variations in the exposure routes and uptake mechanisms of Cu can also contribute to the mismatch between tissue and environmental Cu levels. The fish species studied, including Common carp (*Cyprinus carpio*), Freshwater bream (*Abramis brama*), and Wels catfish (*Silurus glanis*), may exhibit unique physiological and metabolic processes that lead to differential Cu accumulation in their livers and gonads. Additionally, the timing and duration of exposure to Cu in the aquatic environment may further complicate the relationship between environmental Cu concentrations and fish tissue accumulation.

It is essential to consider these complexities and explore additional factors that could influence Cu accumulation in fish tissues, such as other contaminants, inter-species differences in metabolism, and potential interactions between Cu and other trace elements. Further research is necessary to fully understand the mechanisms underlying the observed variations in Cu accumulation in the studied fish species and to ascertain the factors driving these disparities between tissue and environmental concentrations.

Based on the examination of three data consolidation tables obtained from the specialized literature, it is possible to postulate various correlations between elevated metal levels in water and sediments and their presence in the muscle tissue of various fish species.

In the case of the species *Cyprinus carpio*, the slightly elevated Cd values (0.059 and 0.082) reported in the year 2013 [94] can be attributed to the increased Cd levels found in the sediments within the same study area and year (1.69) [68]. Similarly, high Cd values were observed in sector 1 in 2013 for *Carassius carassius* (0.057 and 0.051) and *Silurul glanis*, as documented in the aforementioned study [94].

Pb, due to its lipophilic properties, can be readily absorbed by fish into their blood and bones. Omnivorous fish are more likely to accumulate metals than pelagic species due to the higher metal concentrations in sediment than in the water column [148]. This diverse diet may contribute to their high bioaccumulation of Pb.

The analysis of fish muscle samples from *Cyprinus carpio* and *Alosa immaculata* species obtained from sectors IV and V revealed elevated Pb levels, as reported in 2008 [98]. These higher Pb concentrations in the fish muscles could potentially be associated with the increased values found in water samples, as indicated by Burada in their articles from 2014 and 2015, covering the period between 2007 and 2012.

The findings suggest a potential relationship between the Pb levels detected in the fish muscles and the corresponding Pb concentrations in the water samples. The studies conducted by Burada (2014–2015) likely provide valuable insights into environmental Pb contamination during the specified timeframe [78,79]. By examining the water samples, Burada may have identified elevated Pb values that could have influenced the accumulation of lead in the muscle tissues of carp and mackerel species.

It is essential to consider that Pb accumulation in fish can occur through various pathways, including direct exposure to contaminated water or through the food chain. Fish species such as carp and mackerel exhibit different feeding habits and ecological behaviors, which can affect their susceptibility to Pb contamination. Factors such as the proximity of the sampled sectors to potential pollution sources, the concentration and availability of Pb in the water, and the migratory patterns of the fish species should also be taken into account.

Further research and investigation are necessary to establish a more definitive cause-and-effect relationship between the Pb levels in the fish muscles and the corresponding water samples. This would involve assessing the temporal and spatial correlation, considering other potential sources of Pb contamination, and evaluating the bioaccumulation and biomagnification processes within the aquatic ecosystem.

The analysis of samples collected between 2003 and 2013 indicated that the Zn content in muscle tissue, liver, gonads, and tegument exceeded the accepted limits for *Cyprinus carpio* and *Alosa immaculata* in all studied sectors. These findings suggest a widespread issue of elevated Zn levels in these fish species.

In sector V, there was a notable correlation between the high Zn values observed in water and sediment samples. The elevated Zn levels in the water and sediment samples from sector V may have contributed to the higher Zn concentrations found in the fish species, thus potentially playing a role in the occurrence of diseases.

Regarding catfish, barbel, and perch samples from sector I, higher Zn values were specifically detected in the gonads and liver. This indicates that these organs of the studied fish species in sector I accumulated elevated amounts of Zn. The reasons behind these higher Zn levels in the gonads and liver of *Silurus glanis*, *Barbus barbus*, and *Perca fluviatilis* in sector I may be associated with specific environmental factors or biological characteristics of these fish species, such as their feeding habits or habitat preferences.

Further investigation is necessary to determine the causes of the high Zn content in the various fish tissues and sectors. Factors such as pollution sources, dietary patterns, and species-specific physiological processes should be considered to understand the mechanisms leading to Zn accumulation in different organs. Additionally, assessing the potential health implications of these elevated Zn levels in fish species is essential to evaluating the overall ecological impact and potential risks to human consumption.

The differences in metal concentrations within tissues can be attributed to the tissue's capacity to generate metal-binding proteins such as metallothionein [68].

Various pathways facilitate the entry of metals into the human body, which can occur through multiple sources, such as water, food, air, and even cosmetics. Among these avenues, the most prevalent metal intake method is regularly consuming contaminated food.

Several factors influence the potential risk to human health related to fish consumption. These include the size of the meal, the type of fish consumed, variations in bioaccumulation among different fish species, and the presence of specific chemicals [98]. Monitoring the concentration of substances, including inorganic and organic compounds, is essential for reducing pollution and minimizing metal contamination.

Due to regular consumption of such contaminated food, these metals gradually accumulate in the body over time. Addressing and monitoring this primary pathway of metal intake is important to safeguard public health and minimize the potential risks associated with metal exposure.

## 5. Conclusions

It is crucial to have a comprehensive understanding of metal concentrations in fish to safeguard human health and effectively manage the environment. Metal poisoning can cause damage to the brain, kidneys, liver, and other important organs and even lead to the development of cancer, as exemplified by the carcinogenic properties of As. Symptoms of metal poisoning can range from weakness to headaches. Therefore, continuous monitoring of metal concentrations in fish is of utmost importance.

Fish have long been considered practical pollution biomarkers in aquatic environments. Evaluating the levels of metals in edible fish is essential for ensuring the safety of fish protein for consumers and comprehending its detrimental effects on individuals, populations, or ecosystems.

It is vital to assess the ecological and health risks associated with metal exposure through food, especially when consuming fish contaminated with metals. However, there is a lack of comprehensive studies addressing this issue in Romania. Previous studies have focused on small areas or solely examined sediment contamination without exploring the connection between metals in entire watersheds or sediments and their impact on humans through the food chain.

Specific populations, such as pregnant women, children, and fishermen relying heavily on fish as a protein source, may be disproportionately affected by consuming contaminated fish. Therefore, assessing the potential health risks associated with exposure to metals is essential.

Further research is needed to evaluate the nutrient and metal concentrations in commonly consumed fish to determine acceptable toxicity levels and understand their potential effects on human health. While several studies have already addressed the risks of metal exposure from fish consumption, more research is required to compare different thresholds and better understand the impact on human health.

Although many fish species can absorb metals, in the Danube River, the levels detected in fish meat generally remain below the maximum residue limits (MRLs) proposed in the Official Journal of the European Communities (2001). Nevertheless, monitoring metal concentrations in fish meat is necessary, particularly considering its consumption in the human diet.

Monitoring fish welfare and assessing the quality of aquatic ecosystems in proximity to significant human activities is essential to proactively mitigate potential health risks for consumers.

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