A preliminary investigation of the concentration of selected metals in the tissues and organs of the tigerfish *(Hydrocynus vittatus)* from the Olifants River, Kruger National Park, South Africa

HH du Preez* and GJ Steyn
Research Unit for Aquatic and Terrestrial Ecosystems, SUEC, Department of Zoology, Rand Afrikaans University, PO Box 524, Johannesburg 2000, Republic of South Africa

Abstract

The axial muscle, gill, stomach, intestine, liver, gonads and body fat from tigerfish, *Hydrocynus vittatus* collected during October 1990 from the Olifants River, Kruger National Park, were analysed for Fe, Zn, Pb, Ni, Cu, Cd and Mn by atomic absorption spectrophotometry. The metals were detected in all the tissues examined, but in variable concentrations indicating differences in the accumulation of the metals by the fish. Stomach tissue had the highest mean concentration of Cd (4,5 ± 0,8 $\mu g \cdot g^{-1}$ wet mass), Mn (7,0 ± 17,7 $\mu g \cdot g^{-1}$ wet mass), Ni (10,0 ± 1,4 $\mu g \cdot g^{-1}$ wet mass) and Pb (14,7 ± 3,6 $\mu g \cdot g^{-1}$ wet mass). The highest mean concentration of Zn (43,7 ± 74,2 $\mu g \cdot g^{-1}$ wet mass) was recorded in the gonads. Liver tissue, however, had the highest mean concentrations of Cu (16,6 ± 12,1 $\mu g \cdot g^{-1}$ wet mass) and Fe (1 634,1 ± 589,6 $\mu g \cdot g^{-1}$ wet mass). The derived bioconcentration factors were generally low (lower than 100) suggesting low bioavailability of these metals.

Introduction

The increase in the demand for water by the various user sectors in South Africa has focused attention on the water required for the management of the environment. This has led to the realisation that the environment should also be recognised as one of the user sectors to which water ought to be allocated. The water requirements (quantity and quality) for the management of the environment will, however, depend on the specific ecosystem and situation under consideration.

When evaluating the demand for water of the Kruger National Park, it is evident that this unique nature reserve requires freshwater for potable use, game watering, ecosystem maintenance and to supply downstream users in Mozambique (Department of Water Affairs, 1986; Moore, 1990). During the past two decades the quantity and quality of the water in the rivers of the Park have increasingly been affected by developments to the west of the Park. The catchment areas of the main rivers, namely the Shingwedzi, Luvuvhu, Letaba, Olifants, Sabie and Crocodile Rivers that flow through the Park, for the greater part lie outside the Park and are subject to an increase in urbanisation, irrigation, mining, industries and poor land-use practices (Gertenbach, 1989; 1991).

This study investigates the concentrations of metals (Fe, Zn, Pb, Ni, Cu, Cd and Mn) in tissue samples of the tigerfish (*Hydrocynus vittatus*) from the Olifants River. The total catchment area of the Olifants River is the second largest (54 805 km^2) of all the rivers that flow through the Park, but has areas which have been exploited extensively by the above-mentioned activities. This study was therefore undertaken to evaluate the extent of contamination of tigerfish by metals which may have been introduced into the river system due to mining, industrial and related activities in the catchment area.

*To whom all correspondence should be addressed.
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Materials and methods

Tigerfish were collected with rod and reel during October 1990 from the Olifants River, opposite the Olifants Wilderness Trail Base Camp (Site A, Fig. 1). After capture the live fish were placed immediately into polyethylene buckets containing river water and carried to the field laboratory. Fish length, (fork length), weight and sex were recorded for all specimens captured. In order to prevent contamination the dissections were carried out using stainless steel tools on a polyethylene work surface (Heit and Klusek, 1982). The following tissue samples were removed from each fish for analysis: axial muscle, gill, stomach, intestine, liver, gonads and body fat. After dissection, tissue samples were placed in washed polyethylene Nunc cryotubes and stored in liquid nitrogen for later analysis.

At 9:00 on October 12, the surface water temperature (±10 cm below surface) (WTW microprocessor, Model OXT 96), conductivity (Jenway, Model 4070), dissolved oxygen (WTW microprocessor, Model OXT 96) and pH (ORION, Model SA 250) were measured at Site A. Surface water samples for metal analysis were collected during April, June, August and October 1990 at Site B for metal analysis. The bottles were rinsed several times with river water before being filled with this surface water (2 ℓ) which was then frozen.

In the laboratory the tissue and water samples were thawed for further analysis. One gram of tissue was weighed into a 100 mℓ Erlenmeyer flask to which 10 mℓ concentrated nitric acid and 5 mℓ concentrated perchloric acid were added. The acid digestion was performed on a hot plate (200 to 250°C) for at least 4 h during which total digestion and clearing of the samples were achieved. Each of the digested samples was then filtered using a millipore 6 µm paper filter and a vacuum pump. After filtration the filter system was rinsed with double distilled water to remove all traces of the dissolved metal from the filter paper and the filter system. Each sample was then made up to 100 mℓ with double distilled water and stored in clean storage bottles for analysis of the different heavy metals. Due to small sample weight, it was necessary to pool some of the stomach, intestine,

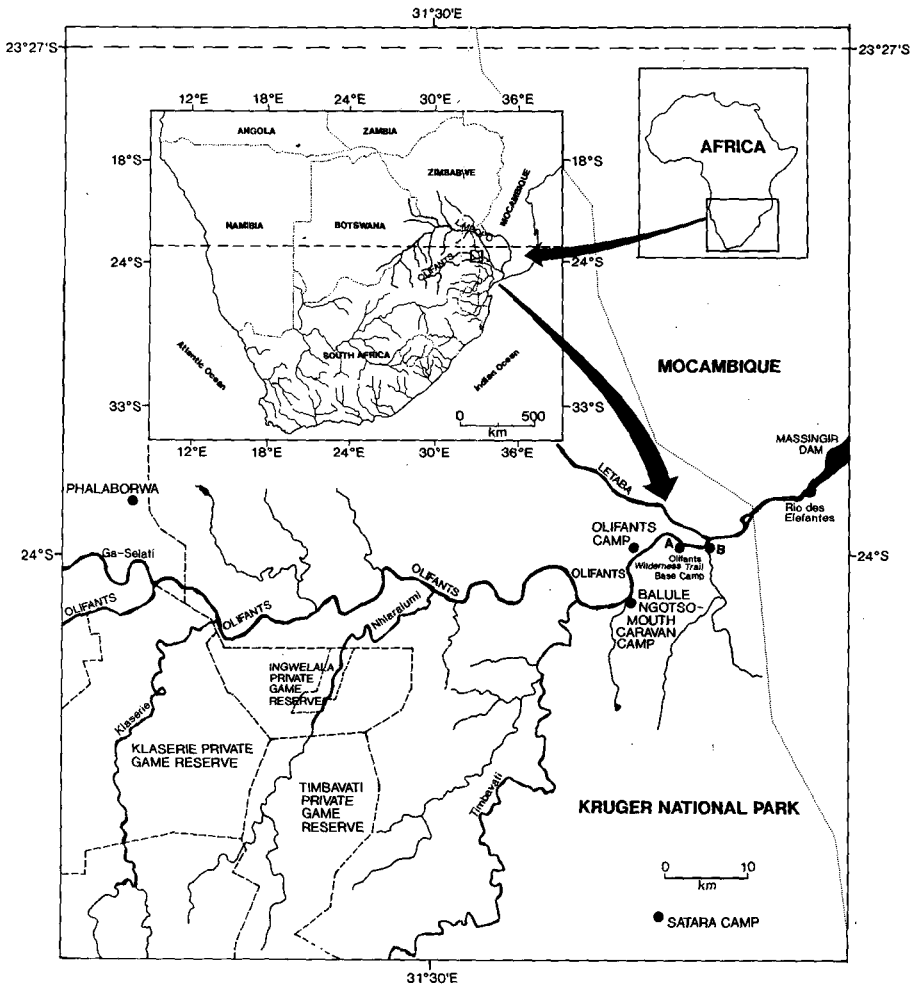

Figure 1
Sampling localities in the Olifants River, South Africa, during October 1990

body fat, gonads and liver samples to achieve the required 1 g wet tissue sample weight.

The total metal concentration in the river water (total = dissolved plus suspended; Wiener and Giesy, 1979) was determined by adding 10 mℓ concentrated nitric acid, 5 mℓ concentrated perchloric acid and 20 mℓ of well mixed river water into a 100 mℓ Erlenmeyer flask, and then evaporated to 2 to 5 mℓ on a hot plate until it cleared (Standard Methods, 1989). Each sample was then made up to 20 mℓ with double distilled water and stored as previously described. All glassware which was used, was soaked for 24h in 2% Contrad concentrate solution (Merck chemicals), rinsed in double distilled water, acid-washed in 1M HCl for 24h and rinsed again in double distilled water (Giesy and Wiener, 1977).

A Varian atomic absorption spectrophotometer (Spectra AA-10) was used to measure the concentrations of the metals in each sample. Analytical standards for Fe, Zn, Cu, Mn, Ni, Cd and Pb were prepared from stock solutions of Holpro chemicals. The metal concentrations in the river water are expressed as $\mu g \cdot \ell^{-1}$ while the metal concentrations in the tissues were recalculated and expressed as µg metal per gram wet tissue mass.

The bioconcentration factors (B.F.) between river water and fish tissue were calculated using the formula of Wiener and Giesy (1979):

$$B.F. = \frac{Co}{Cw}$$

where Co = wet mass concentration of a given metal in the tissue and Cw = average total concentration of the metal in water.

Statistical analyses were performed on the data and in cases of significant analysis of variance ($P<0,05$), multiple comparisons of means were made with the Tukey test (Zar, 1984).

Results

During the sampling period (11 to 12 October 1990) the Olifants River was low flowing and formed a few calm pools in the vicinity of the Wilderness Trail Base Camp (Site A, Fig. 1). The selected physical and chemical variables of the river water are summarised in Table 1. Surface water at 9:00 (12 October) was well oxygenated (8,2 $mg \cdot \ell^{-1}$), alkaline (pH = 8,4) and had a temperature of 26°C. Total metal concentrations varied with the mean concentrations of iron ($2\,285 \pm 643\ \mu g \cdot \ell^{-1}$) and cadmium (lower than detectable) being the highest and lowest

TABLE 1
PHYSICAL AND CHEMICAL VARIABLES OF SURFACE WATER IN THE OLIFANTS RIVER, KRUGER NATIONAL PARK

Variable			
Temperature (°C)*	26,8		
Electrical conductivity (mS·m^{-1} @ 25°C)*	85,0		
Dissolved oxygen (mg·ℓ$^{-1}$)*	8,2		
Dissolved oxygen (percentage saturation)*	101,0		
pH*	8,4		
Metal concentrations (µg·ℓ$^{-1}$)**	**Range**	**$\bar{X}$±S.D.**	**C.V.(%)**
Iron	1 830-2 140	2 285±643	28
Zinc	360-1740	1 075±573	53
Copper	40-120	70±35	49
Manganese	30-60	45±17	38
Nickel	130-240	185±58	31
Cadmium	<0,02***	N.D.	N.D.
Lead	230-440	355±89	25

* = data collected during October 1990 at Locality A
** = data collected during April 1990, June 1990, August 1990 and October 1990 at Locality B
*** = lowest detectable concentration
$\bar{X}$ = mean value
S.D. = standard deviation of the mean
C.V. = coefficient of variation
N.D. = non-detectable concentration

respectively (Table 1). The order of abundance of mean metal concentrations in the river water for the period April to October 1990 was Fe > Zn > Pb > Ni > Cu > Mn > Cd.

The fork length of the captured male tigerfish (n=25) ranged from 269 to 318 mm (mean: 287 ± 13 mm) with a mass range of 390 to 560 g (mean: 451 ± 58g). Only four female fish with a mean fork length of 294 ± 28 mm (range: 250 to 318 mm) and a mean mass of 472 ± 70g (range: 450 to 550g) were captured. The testes of the males were well-developed with a few individuals being ripe running. The ovaries of the females were less developed.

Means and ranges of concentrations (µg·g^{-1} wet mass) of Fe, Zn, Cu, Mn, Ni, Cd and Pb in the selected tissues are presented in Table 2. Only four females were caught and their data were therefore not subjected to statistical analysis (e.g. analysis of variance and means). The highest mean concentrations of Fe (1 634,1 ± 589,6 µg·g^{-1} wet mass) and Cu (16,6 ± 12,1 µg·g^{-1} wet mass) were recorded in the liver tissue (Table 2). Stomach tissue had the highest mean concentrations of Mn (7,0 ± 17,7 µg·g^{-1} wet mass), Ni (10,0 ± 1,4 µg·g^{-1} wet mass), Cd (4,5 ± 0,8 µg·g^{-1} wet mass) and Pb (14,7 ± 3,6 µg·g^{-1} wet mass). Gonads, however, had the highest mean concentration of Zn (43,7 ± 74,2 µg·g^{-1} wet mass).

Concentrations of metals in axial muscle tissue, gonads and body fat occurred in the order Fe > Zn > Pb > Ni > Cu > Cd > Mn (Table 2). In the gill, stomach and intestine tissue, however, the concentration order of Mn, Cu and Cd was different, Fe > Zn > Pb > Ni > Mn > Cu > Cd. The concentration order of the metals in the liver (Fe > Zn > Cu > Pb > Mn > Ni > Cd) deviates from both these metal concentration orders. One-way analysis of variance followed by the Tukey's multiple comparison test demonstrated that the mean concentration of Fe in a tissue differed significantly from the other metals. The mean concentration of Zn in a tissue was also significantly different ($P<0,05$) for the mean concentrations of other selected metals, except in the body fat ($P>0,05$ for Ni, Pb), muscle and intestinal tissue ($P>0,05$ for Pb). When statistical comparisons were made between the mean Pb concentration and the mean Ni, Cu, Cd and Mn concentrations in different tissues, no significant differences ($P>0,05$) were generally observed, the only significant differences ($P<0,05$) being for the Cu, Cd and Mn concentration in stomach tissue. The Tukey's multiple comparison test also revealed no significant differences in the mean concentrations of Ni, Cu, Cd and Mn. In the muscle tissue, however, the Ni concentrations were significantly higher than the Cu, Cd or Mn concentrations.

Bioconcentration factors for the metals found in tigerfish tissues were generally lower than 100, with higher values being calculated for Fe, Cu and Mn in liver tissue and also for Mn in stomach and gill tissue. The bioconcentration factors for Fe (range: 17 to 715) and Cu (range: 36 to 237) were the highest in the liver and the lowest in the axial muscle tissue. In contrast, the bioconcentration factors for Mn (range: 31 to 156), Ni (range: 23 to 54) and Pb (range: 28 to 40) were the highest in the stomach tissue and the lowest in body fat, liver and gill tissue respectively. Zinc bioconcentration factors (range: 14 to 40) were the highest in the gonads, (as for Fe) and the lowest in muscle tissue.

Discussion

The water of the Olifants River entering the Park is highly mineralised, with no significant change in the composition of the salts as the water flows through the Park (Van Veelen, 1991). Water quality in the Olifants River represents a relatively stable system with no significant trend (excluding a decrease in chloride) over a period of six years (1983 to 1989) in either the composition or concentration of the salts (Moore et

TABLE 2
CONCENTRATIONS (μg.g^{-1} WET MASS) OF HEAVY METALS IN TISSUE OF TIGERFISH *(HYDROCYNUS VITTATUS)* COLLECTED IN THE OLIFANTS RIVER, KRUGER NATIONAL PARK

Tissue	Sex of fish	Number of samples		Iron	Zinc	Copper	Manganese	Nickel	Cadmium	Lead
MUSCLE	Male	25	Range	19,0-72,0	6,0-50,0	1,0-4,0	1,0-6,0	6,0-11,0	0,8-4,4	3,0-20,0
			$\bar{X}$±S.D.	38,1±12,2	14,6±9,5	2,5±1,0	2,0±1,1	8,4±1,6	2,3±0,8	12,8±4,9
			C.V.(%)	32,0	65,1	380,0	55,0	19,0	34,8	38,3
			B.F.	16,7	13,6	35,7	44,4	45,4	-	36,1
	Female	4	Range	26,0-54,0	8,0-14,0	2,0-14,0	1,0-2,0	8,0-10,0	1,6-2,2	2,0-12,0
			$\bar{X}$±S.D.	35,3±13,2	11,8±2,6	6,0±5,4	1,3±0,5	9,0±0,8	1,9±0,3	9,0±4,7
			C.V.(%)	37,4	22,0	90	38,5	8,9	15,8	52,2
			B.F.	15,4	11,0	85,7	28,9	48,6	-	25,4
GILL	Male	25	Range	60,0-384,0	20,0-185,0	2,0-13,0	3,0-7,0	4,0-8,0	0,6-2,2	3,0-17,0
			$\bar{X}$±S.D.	134,8±70,1	36,9±32,5	4,2±2,8	4,7±1,2	6,4±1,1	1,5±0,4	9,9±3,9
			C.V.(%)	52,0	88,1	66,7	25,5	17,2	26,7	39,4
			B.F.	59,0	34,4	60	104,4	34,6	-	27,9
	Female	4	Range	96,0-153,0	17,0-39,0	2,0-4,0	4,0-5,0	5,0-9,0	0,6-1,6	4,0-10,0
			$\bar{X}$±S.D.	115,8-25,5	28,3±9,2	3,0±9,2	4,8±0,5	7,0±1,8	1,1±0,4	7,3±2,5
			C.V.(%)	22,0	32,5	26,7	10,4	25,7	36,4	34,2
			B.F.	50,7	26,4	42,9	106,7	37,8	-	20,7
STOMACH	Male	21	Range	50,0-123,0	6,0-196,0	2,0-8,0	1,0-84,0	9,0-12,0	2,0-6,6	7,0-21,0
			$\bar{X}$±S.D.	76,1±23,0	31,5±39,4	6,1±1,6	7,0±17,7	10,0±1,4	4,5±0,8	14,7±3,6
			C.V.(%)	30,2	125,6	26,2	252,9	14,0	17,8	24,5
			B.F.	33,3	29,4	87,1	156,6	54,1	-	41,4
	Female	4	Range	38,0-60,0	7,0-55,0	3,0-8,0	2,0-3,0	6,0-9,0	3,7-4,4	2,0-11,0
			$\bar{X}$±S.D.	48,5±9,2	28,8±19,8	5,3±2,1	2,3±0,5	8,0±1,4	4,0±0,3	8,3±4,3
			C.V.(%)	19,0	68,8	39,6	21,7	17,5	7,5	51,8
			B.F.	21,2	26,9	75,7	51,1	43,2	-	23,4
INTESTINE	Male	18	Range	37,0-110,0	18,0-30,0	2,0-9,0	1,0-81,0	5,0-9,0	2,0-8,9	5,0-26,0
			$\bar{X}$±S.D.	70,1±23,9	23,3±4,5	4,7±2,5	5,7±18,8	6,3±1,1	3,7±1,9	13,0±1,3
			C.V.(%)	34,1	19,3	53,2	329,8	17,5	51,4	10,0
			B.F.	30,7	21,7	67,1	126,7	34,1	-	36,6
	Female	4	Range	36,0-130,0	14,0-21,0	3,0-10,0	1,0-2,0	5,0-8,0	2,0-4,0	7,0-10,0
			$\bar{X}$±S.D.	76,5±40,0	18,5±3,1	4,8±3,5	13±0,5	6,0±1,4	2,8±0,9	8,3±1,3
			C.V.(%)	52,3	18,9	72,9	38,5	23,3	32,1	15,7
			B.F.	33,5	17,2	68,6	28,9	32,4	-	23,4
LIVER	Male	18	Range	500,0-2620,0	11,0-104,0	2,0-61,0	1,0-34,0	3,0-6,0	1,3-10,0	7,0-16,0
			$\bar{X}$±S.D.	1634,1±589,6	35,2±20,1	16,6±12,1	6,0±10,3	4,2±0,9	2,3±2,0	11,2±3,2
			C.V.(%)	36,1	57,3	72,9	171,7	21,4	87,0	28,6
			B.F.	715,0	32,8	237,1	133,3	22,7	-	31,5
	Female	4	Range	782,0-1634,0	26,0-47,0	16,0-23,0	1,0-2,0	3,0-4,0	2,2-12,2	7,0-14,0
			$\bar{X}$±S.D.	1183,8±404,5	39,3±9,2	18,8±4,7	1,5±0,6	3,3±0,5	4,9±4,9	10,3±3,8
			C.V.(%)	36,1	23,4	25,0	40,0	15,2	100,0	36,9
			B.F.	518,1	36,6	268,6	33,3	17,8	-	29,0
GONADS	Male	23	Range	15,0-184,0	12,0-342,0	2,0-10,0	1,0-3,0	2,0-12,0	1,3-2,4	5,0-58,0
			$\bar{X}$±S.D.	44,6±33,0	43,7±74,2	3,7±1,8	1,8±0,3	5,0±2,4	1,9±0,5	14,3±11,1
			C.V.(%)	74,0	169,8	48,6	15,8	48,0	26,3	77,6
			B.F.	19,5	40,7	52,9	42,2	27,0	-	40,3
	Female	4	Range	43,0-64,0	19,0-192,0	3,0-5,0	2,0-3,0	3,0-7,0	1,7-3,4	1,0-12,0
			$\bar{X}$±S.D.	55,3±9,4	89,5±73,5	3,8±1,0	2,3±0,5	5,5±1,9	1,9±0,3	7,8±4,8
			C.V.(%)	17,0	82,1	26,3	21,7	34,5	15,8	61,8
			B.F.	24,2	83,4	54,3	51,1	29,7	-	22
BODY FAT	Male	18	Range	26,0-129,0	5,0-126,0	1,0-9,0	1,0-2,0	3,0-7,0	1,0-2,6	2,0-16,0
			$\bar{X}$±S.D.	55,5±42,0	17,7±27,5	3,3±1,6	1,4±0,6	5,3±1,2	1,8±0,4	10,9±3,4
			C.V.(%)	75,7	155,4	48,5	42,9	22,6	22,2	31,2
			B.F.	24,4	16,5	47,1	31,1	28,6	-	30,7
	Female	4	Range	62,0-153,0	7,0-41,0	3,0-4,0	2,0-3,0	5,0-6,0	1,7-2,7	3,0-11,0
			$\bar{X}$±S.D.	94,0±42,0	21,0±14,1	3,5±0,6	2,5±0,6	5,5±0,8	2,2±0,4	7,8±3,4
			C.V.(%)	44,7	67,1	17,1	24,0	14,5	18,2	43,6
			B.F.	19,6	19,6	50,0	55,6	29,7	-	22,0

$\bar{X}$ = Mean value;
S.D. = Standard deviation of the mean;
C.V. = Coefficient of variation per cent;
B.F. = Bioconcentration factor

al., 1991; Van Veelen, 1991). The pH and electrical conductivity of the water at the time of sampling compares well with the mean pH (8,0 ± 0,3) and electrical conductivity (83,6 ± 30,7 $mS \cdot m^{-1}$) measured over six years at the Wilderness Trail Base Camp (Van Veelen, 1991). Available pH and dissolved oxygen data presented in this study and by Van Veelen (1991) fall within the pH (6 to 9, pH values) and dissolved oxygen (>4 $mg.\ell^{-1}$) ranges suggested for the protection of aquatic life (Kempster et al., 1980). Fish mortalities in the Olifants River were, however, associated with changes in the physical and chemical properties such as decrease in temperature (Deacon, 1990), elevated potassium (Moore, 1990) and high silt loads of between 24 000 and 77 000 $mg.\ell^{-1}$ (Moore et al., 1991; personal observation). Most of these observations lack sound scientific evaluation and need further investigation when observed. Although electrical conductivity may influence the growth rate and life-span of fish (Hellawell, 1986), it is at present not known whether the measured electrical conductivity of the Olifants River water affects the tigerfish or other fish populations.

The concentrations of metals in the Olifants River water must be interpreted with caution because the data are based on only four samples. The mean Fe, Zn, Ni and Pb concentrations are higher than the maximum concentration limits for these metals (Fe = 2 000 $\mu g.\ell^{-1}$; Zn = 100 $\mu g.\ell^{-1}$ Ni = 50 $ug.\ell^{-1}$; Pb = 100 $\mu g.\ell^{-1}$) suggested for the protection of aquatic life (Kempster et al., 1980). In contrast the mean Cu and Mn concentrations of the river water were within the minimum (Cu = 5 $\mu g.\ell^{-1}$; Mn = 45 $\mu g.\ell^{-1}$) and maximum (Cu = 200 $\mu g.\ell^{-1}$; Mn = 1 000 $\mu g.\ell^{-1}$) concentration criteria derived to preserve aquatic life in the riverine system. Cadmium was not detected in the river water with the techniques used in this study and it is therefore assumed that levels lower than the detection limited of 0,02 $\mu g.\ell^{-1}$ are present.

The selected metals (Fe, Zn, Pb, Cu, Mn, Ni, Cd) were detected in the tissues (axial muscle, gill, stomach, intestine, liver, gonads, and body fat) of all the tigerfish that were examined. Concentration differences were noted for the metals in these tissues: e.g. the mean iron concentration was the highest in the liver, which had the lowest mean nickel concentration. Differences in tissue metal concentrations of fish from a specific locality or fish exposed to specific metal concentrations under laboratory conditions have been recorded for some fish species (Matthiessen and Brafield, 1977; Holcombe et al., 1979; Wiener and Giesy, 1979; Buckley et al., 1982; Boyer, 1984; Bezuidenhout et al., 1990). These differences in tissue metal concentrations appear to be related to tissue type and the specific metal in question. It must, however, be stressed that the concentration of metal in a tissue may not always be significantly different from the other metals as indicated for some metals in the present study. Furthermore, metal concentrations in fish are the result of complex processes associated with uptake-excretion kinetics and homeostases in fish (Giesy and Wiener, 1977; Heath, 1987).

The relatively large coefficient of variation (C.V.) for metals reflects variation between individual tigerfish. A large C.V. for metal concentration was in some cases (e.g. Mn in stomach and intestinal tissues) the result of a single tissue sample in which a high concentration was detected. These variations must be attributed to factors other than mass since the mass of the tigerfish was within a relatively small range (390 ± 560g). Literature dealing with the relationship between the age of the fish and metal concentration reports conflicting results, with the correlation being either negative, positive or absent (Cross et al., 1973; Bohn and McElroy, 1976; Atchison et al., 1977; Giesy and Wiener, 1977; Bohn and Fallis, 1978; Mears and Eisler, 1977; Chernoff and Dooley, 1979; Vinikour et al., 1980), which leads to the conclusion that it is not of importance to consider age when comparing Pb and Cd concentrations in fish. It is, however, recommended that due to the discrepancies found in the literature, age should be considered when interspecific comparisons of metal concentrations are made as suggested by Cross et al. (1973). This would be of particular importance when very little is known about the bioaccumulation of metals in a species (e.g. tigerfish).

Mean zinc and copper concentrations in the axial muscle, gills, gonads and body fat of *Clarias gariepinus* from an industrial and mine polluted lake (Germiston lake, South Africa) were higher than the mean concentrations detected in the tigerfish (Bezuidenhout et al., 1990). The liver tissue of *C. gariepinus* also had a higher zinc concentration than the tigerfish; however, the copper concentration was lower. At present more detailed information on metal concentration levels and bioconcentration factors for South African fish is not available. It is therefore not possible to relate the metal concentrations detected in the tigerfish to concentration levels for other South African fish species. Bioavailability and biological effects of metals on aquatic biota (e.g. fish) are largely dependent on the concentrations of the metals and the physico-chemical properties of the aquatic system (Pagenkopf et al., 1974; Wiener and Giesy, 1979; Heath, 1987). In these systems the metals have a tendency to form complexes with suspended solids or organic and inorganic ligands (e.g. hydroxide, chloride, carbonate), thus reducing their bioavailability (Stiff, 1971; Giesy and Briese, 1977; Giesy et al., 1978; Heath, 1987). If it is assumed that lower bioconcentration factors for a metal indicate lower bioavailability (e.g. B.F. for *Lepomis macrochirus*: Cd=240 vs 71, 83 or 120; Cu=370; Mn=220; Pb=230 vs 33; Wiener and Giesy, 1979), the generally low bioconcentration factors for the metals in tigerfish tissue indicate that metals in the Olifants River system of the Park are less available for uptake by tigerfish. The lower bioavailability of the metals may be attributed to the formation of inorganic complexes (e.g. carbonate) in hard-water systems (total alkalinity: >100 $mg.\ell^{-1}$ $CaCO_3$) such as the Olifants River system (total alkalinity: 131 ± 37 $mg.\ell^{-1}$ $CaCO_3$, Van Veelen, 1991). The absorption of metals and toxicity to aquatic biota such as fish is therefore lower (Brown, et al., 1974; Pagenkopf et al., 1974; Andrew et al., 1977; Giesy et al., 1977; Rodgers and Beamish, 1983; Alabaster and Lloyd, 1980; Heath, 1987; McMurtry et al., 1989). Water hardness further affects the absorption of metals because it is negatively correlated with gill permeability to ions and water, while positively charged molecules are increasingly being repelled by the outside surface of the gills (Heath, 1987). The bioconcentration factors, however, would not give the accurate indication of the relative availability of the metal for uptake if it was regulated by the fish (Wiener and Giesy, 1979). At present it is not known to what extent tigerfish can regulate the uptake of specific metals.

On the basis of this preliminary study it may be concluded:

- the selected metals were detected in all the tissue examined, but in variable concentrations indicating differential bioconcentration of the metals; and

- bioconcentration factors were generally low suggesting low bioavailability of the metals.

The preliminary nature of the present study must, however, be emphasised. A long-term monitoring programme at selected sites in the catchment area should be initiated to obtain more information about the natural and anthropogenic flux of metals in the system. Future research must also include studies on metal speciation and the distribution and interaction of the metals in the ecosystem.

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