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Influence of ecological factors and of land use on mercury levels in fish in the Tapajós River basin, Amazon ☆

D. Sampaio da Silva [a,b,*], M. Lucotte [b], S. Paquet [b], R. Davidson [c]

[a] CEMAGREF, UMR G-EAU, Montpellier, France
[b] GEOTOP, Université du Québec à Montréal, Canada
[c] Biodôme de Montréal, Canada

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ABSTRACT

Mercury (Hg) contamination of riparian communities and of environmental compartments of the Amazon can be directly related to the occupation of the territory. The objective of this study was to identify the characteristics of aquatic environments that are associated with high levels of Hg in ichthyofauna. Our research aimed at determining the influence of variables related to fish ecology, types of aquatic environment, fishing activities by local riparian populations, and watershed use on the levels of contamination of ichthyofauna. Six sites were sampled during two distinct periods of the hydrological cycle: at the beginning of descending waters and during low waters. We focused on ten dominant fish species representing four trophic levels: *Curimata inornata*, *Geophagus proximus*, *Schizodon vittatum*, *Leporinus fasciatus*, *Anostomoides laticeps*, *Hemiodus unimaculatus*, *Caenotropus labyrinthicus*, *Hoplias malabaricus*, *Plagioscion squamosissimus*, *Acestrorhynchus falcirostris*. The study sites, which included lotic and lentic habitats, are exploited year-round by local riparian communities. Spatial variations in Hg contamination in ichthyofauna were determined by factorial analysis of variance taking into account fish diets, seasons, and sampling sites. Multiple regressions were used to check the influence of ecological and anthropogenic variables and variables related to watershed uses, on Hg levels in key species representing the four trophic groups. Each variable was checked independently. Next, multiple regressions were used to verify the concomitant influence of selected variables. Independently of the study site and the phase of the hydrologic cycle, fish Hg contamination followed the trend piscivores > omnivores > herbivores > detritivores. In all the aquatic study sites, Hg levels measured in predatory species were often higher than the 500 ng/g fresh weight threshold. Mean Hg levels in key species were significantly higher during descending waters in lotic environments, and during low waters in lentic environments. Data from this study demonstrated that simple models based on watershed use and on easily obtained variables such as the suspended particulate matter (SPM) load and SPM Hg concentrations, number of inhabitants, habitat types, and the stage in the hydrological cycle enable very good prediction of Hg levels in fish. Our cartographical data clearly showed that the watershed site with the highest aquatic vegetation cover (6% of the open water body) and with the lowest forest cover (62% of the land) corresponded to the highest Hg concentrations in fish. Conversely, the watershed site with 94% forest cover and 1% aquatic vegetation corresponded to the lowest levels Hg concentrations in fish. These results suggest that land uses of watersheds play a key role in the level of Hg contamination of local ichthyofauna.

1. Introduction

The Brazilian Amazon is presently undergoing profound disruption, and current deforestation is not merely a consequence of demographic growth. Gillis and Repetto (1988) demonstrated that land-use models and their associated deforestation reflect the demographic structure of local households as well as the impacts of credit policies and of other macro-economic forces that apply locally. Globally, large pastures, logging activities, family agriculture and more recently, the introduction of large-scale soya

☆ This paper is dedicated to the memory of Dr. Marc Roulet, who made the fascinating discovery that deforestation followed by soil erosion is the main source of mercury in the riparian Tapajós region, and opened a new era of mercury biogeochemistry research in the Amazon. To this outstanding scientist, all our respect and acknowledgments.

* Corresponding author at: 361 rue Jean-François Breton – BP 5095, F-34196, Montpellier cedex 5, France.
E-mail addresses: delaine.sampaio@montpellier.cemagref.fr, sampaio.delaine@gmail.com (D. Sampaio da Silva).

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plantations are a threat to the forest cover in this Brazilian region (Parayil and Tong, 1998; Weinhold, 1999; Margulis, 2004; Fearnside, 2001; Scouvart and Lambin, 2006). Mercury (Hg) contamination of riparian communities and of environmental compartments can be directly related to the occupation of the Amazonian territory. Indeed, previous studies suggested that erosion of cultivated soil contributed to the leaching of natural soil Hg to aquatic environments (Roulet et al., 1998; Farella et al., 2001). Recently, Farella et al. (2006, 2007) showed that soils under any type of local cultivation (pasture, fallow, orchards, banana plantations) were characterized by cation enrichment associated with slash-and-burn activities leading to loss of Hg, compared to levels measured in soils under forest cover.

Hg levels in fish muscular tissues can vary tremendously from one aquatic environment to another one located nearby. These fluctuations result from a combination of biotic and abiotic factors that interact in a complex manner in a specific environment. Most of our knowledge aimed at identifying the variables that influence Hg levels in ichthyofauna comes from studies that were not conducted in tropical regions. Indeed, many researchers have tried to determine key factors that would enable the prediction of Hg concentrations in fish in specific environments. However, most attempts to integrate different factors did not enable prediction of Hg concentration in fish tissues or failed to identify Hg hot spots effectively, particularly in large territories (EPRI, 2003; Roué-Legall et al., 2005). Studies that attempted to explain these fluctuations focused manly on physico-chemical or environmental factors (Bodaly et al., 1993; Cope et al., 1990; Driscoll et al., 1994; Evans et al., 2005; Wiener et al., 2003).

In the Amazonian context, Belger and Forsberg (2006) studied the influence of many variables including pH, dissolved organic carbon (DOC), fish size and the presence of potential methylation sites on Hg levels in carnivorous species (*Cichla.* sp. and *Hoplias malabaricus*) in the River Negro. These authors observed that in *H. malabaricus*, Hg levels increased with fish size and with DOC, whereas in *Cichla* sp., Hg levels increased only with an increase in seasonally flooded areas. Bastos et al. (2007) recently found a relation between Hg bioaccumulation in predatory and non-predatory fish in the River Madeira and food adaptations by the species concerned after seasonal modifications in aquatic ecosystems. This agrees with previous results obtained by Dórea et al. (2006) for ichthyofauna in the River Negro. Furthermore, Sampaio da Silva et al. (2005), conducted a study using C and N stable isotopes, and came to the same conclusions concerning ichthyofauna in three different environments in the Tapajós River basin. Other authors also studied the influence of environmental and ecological factors and human activities on Hg concentrations in Amazonian fish (Barbosa et al., 2003; Kehrig et al., 2008).

Trophic level is of capital importance among the ecological factors that control Hg levels in different fish species. According to Cabana and Rasmussen (1994), trophic level and mean weight explain a large proportion of variation in Hg levels in fish. An isotopic approach ($^{15}N/^{14}N$ ratios) is commonly used to determine trophic level in aquatic organisms. This approach enables measurement of the assimilation of ingested food taking tissue renewal into account (Kling et al., 1992), as well as the degree of omnivory of the organisms (Vander Zanden et al., 1997). Correlations between $^{15}N/^{14}N$ ratios, fish lengths and Hg levels in several species in the Tapajós River have already been identified by Sampaio da Silva et al. (2005). These correlations indicate that these species modify their feeding habits and consequently their trophic levels while they are growing. However, this can vary because isotopic signatures of fish of similar size or age can vary from one environment to another depending on differences at the base of the food chains (Peterson and Fry, 1987) and/or possible modifications in fish diet (Sampaio da Silva et al., 2005). In a study aimed at understanding MeHg bioamplification within trophic levels of the Sinamary River continuum in French Guyana, Dominique et al. (2007) recently highlighted many possible food chains, each based on different local primary producers such as biofilms/periphyton, macrophytes, and terrestrial riparian plants. Thanks to exhaustive sampling, taxonomic characterization, and the identification of prey/predator relationships as well as analysis of the isotopic composition of the principal plants and animals, these authors traced back food relationships linking the organisms that are present in the Sinamary river. The authors also revealed that biofilms at the base of the trophic chain, were not only able to accumulate large quantities of Hg (thereby identifying the way this metal enters trophic chains), but were also important MeHg production sites via their bacterial composition.

It has already been demonstrated that many watershed characteristics, such as land use, vegetation cover, and soil types can have a strong influence on Hg, and particularly on MeHg, production and exportation in these environments (St-Louis et al., 1994; Hurley et al., 1995; Babiarz et al., 2001, 1998; Farella et al., 2006; Roulet et al., 2000; Oliveira et al., 2007; Serudo et al., 2007). These characteristics can have an influence on Hg speciation, partition, and biological availability in water (Hurley et al., 1995, 1998). In recent decades, anthropogenic activities have had major impacts on biological community dynamics and on the physical and chemical structure of Amazonian ecosystems (Salati, 1983; Silveira, 1993), as well as reducing habitat and aquatic biodiversity losses (Schubart, 1993). Medium and large-scale soya production and the replacement of tropical forest by small-scale farming, the latter often leading to the prevalence of pastures, are actually the principal causes of deforestation in the study region. To our knowledge, no information has been published that would enable the identification of a relationship between the health of the aquatic ecosystem and human activities in the study area. However, in a recent study Krusche et al. (2005) demonstrated that changes in the structure and functioning of aquatic environments in the Ji-Paraná basin (Rondônia) occurred at micro- and meso-scales. According to these authors, this could be attributed to significant changes in nutrient concentrations in aquatic environments resulting from the replacement of forests by pastures.

The Tapajós watershed is characterized by an historical process of mostly disorganized occupation both by human and environmental heterogeneity, which probably explains the existing diversity in the ecosystems (terrestrial and aquatic) and local uses of natural resources. In this region, Hg concentrations in fish are often high and as such are the first cause of human exposure to the contaminant (Lebel et al., 1997; Passos et al., 2007). This study thus aimed to help identify environmental characteristics and understand the impact of different local human activities on Hg contamination of local fish resources. During fishing activities at our study sites, specimens representing five orders, 21 families and 96 species of fish were characterized by Sampaio da Silva (2008). Fishing activities enable identification of dominant species at both lentic and lotic study sites. From the compilation and the integration of these data with those characterizing fishing activities and yields obtained by local fishermen (Sampaio da Silva, 2008), it seems that dominant species are also those that are the most frequently captured and consumed by riparian populations. In this study, we mapped Hg contamination in 10 of the dominant fish species from different aquatic ecosystems in the Tapajós River basin. In addition, we attempted to explain spatio-temporal variation(s) in Hg levels for these fish by analyzing the influence of ecological characteristics of fish and of the environment, number of inhabitants, captures by local fishers, and existing land use and vegetation cover in varied watersheds.

2. Material and methods

2.1. Study area

This study was carried out in the lower Tapajós basin, located in the State of Pará, showed that the dominant source of Hg in the aquatic ecosystem of this river is from erosion of natural soils in the catchment rather than from anthropogenic pollution (Roulet et al., 2001). The Tapajós basin is an active colonization front that has been the subject of important demographic, cultural and environmental transformations related to the intensification of the exploitation of the territory in the last 40 years (Farella, 2005). Today, family agriculture is the main form of farming in rural areas in the region, and is one of the principal agents of deforestation (Farella, 2005). Sampling took place along a 150 km segment on the river, between the municipalities of Aveiro and Itaituba.

2.2. Sampling methods

Fish were captured with gill nets at six aquatic sites located in the Tapajós River basin (Table 1 and Fig. 1). The local names of the lotic sites are Itapacurazinho (ITA), Paraná (PAR) and Cupari (CUP) and of the lentic sites: Jacaré (JAC), Restinga (RES) and Capituã (CAP). Fishing methods and materials are described in detail in Sampaio da Silva (2008). Sampling was carried out during fishing in two distinct periods of the year (at the beginning of descending waters in 2003 and during low waters in 2004).

In order to achieve the objectives of this study, we focused on 10 key species: *Curimata inornata*, *Geophagus proximus*, *Schizodon vittatum*, *Leporinus fasciatus*, *Anostomoides laticeps*, *Hemiodus unimaculatus*, *Caenotropus labyrinthicus*, *Hoplias malabaricus*, *Plagioscion squamosissimus*, *Acestrorhynchus falcirostris*. *A. falcirostris* is the dominant species in the Tapajós basin and is frequently consumed by local populations. It occupies different trophic levels. The species of fish of the specimens were identified, total length (cm) and mass (g) were measured; a section of the dorsal muscle tissue free of skin and bones was removed for analysis of total Hg and finally the muscle tissues were frozen until laboratory analyses.

Suspended particulate matter (SPM) was collected using a vacuum filtration method. During SPM collection, at each sampling site, water was pre-filtered in a patalas trap with a 64 and 210 μm mesh filter. A volume of 5–10 l of water was collected in tanks for transport to the field laboratory. Then, a known volume of water was filtered using pre-weighted GF/F filters. In order to obtain a better representation of the amount of particles per unit volume, three GF/F filters were used at each station. At each study site, we chose two stations that were then sampled on three consecutive days. After filtration, the filters were frozen in Petri dishes until laboratory analysis. Water samples were collected with a manual peristaltic pump. Two bottles were filled and kept in a cooler with freeze packs until transported to the field laboratory, where the samples were frozen until laboratory analyses.

In addition to a total of 1500 Hg measurements in fish, the data analyzed included information concerning watershed use obtained with satellite maps, SPM loads, and Hg levels contained in SPM, ecological data (dominance index, Shannon–Wiener index and fish size classes) and normalized fishing yields presented in Sampaio da Silva (2008).

2.3. Analyses

Hg levels in fish samples were measured by atomic fluorescence according to the method described by Pichet et al. (1999). These analyses were performed at the GEOTOP laboratory of the *Université du Québec à Montréal*. The analytical procedure firstly consisted in acid digestion of subsamples of wet muscle. The digest was brought to a final volume of 30 ml by the addition of water, and Hg was then analyzed by atomic fluorescence. Results are expressed in wet weights. This method has a 5 ng/g detection limit for 1 mg of dry sample. Statistical analyses were performed using the JMP 5.1 program and geographical analyses were performed with the GRASS 6.2.1 program (Geographic Resources Analysis and Support Systems, GRASS Development Team, ITCC-irst, Trento-Italy, 2006). Watershed surfaces were estimated with NASA's digital elevation models (DEMs) from the Shutter Radar Topographic Mission program (SRTM, www.srtm.csti.cgiar.org). These DEMs cover surfaces of 5×5 degrees with a resolution of 90 m. The four DEMs that were used to cover the study region were the following: SRTM_25_13; SRTM_26_14_1; SRTM_26_13; SRTM_25_14. GRASS (6.2.1, r.watershed) geographical modelling tools were used to delimit basins and sub-basins from the four above-mentioned maps. Landsat 7 ETM+ satellite maps were obtained from the Global Land Cover Facility website (www.glcf.umiacs.umd.edu). Three series of images were used with the following image codes p227r063 (July 30, 2001), p228r062 (August 6, 2001), p228r063 (August 6, 2001). Each series of images has a resolution of 30 m and is composed of eight layers. The seven spectral layers (1, 2, 3, 4, 5, and 7) were used in the territorial analysis presented in this article. The study area was first divided into 10 classes using the MLC classification method (Maximum Likelihood Classification Discriminant Analysis) modified by Neteler (1999) and Neteler and Mitasova (2002). Then, classes were compiled based on similarities in landscape composition, which led to a total of seven classes: water surface free of aquatic vegetation (referred to as open water in this article), aquatic surface covered by emerged macrophytes and emerging vegetation, inundated forests (in poorly drained areas, local name *igapós*), upland forests (local name *terra firme*), agricultural land (crops, pasture, fallows), savannas (Amazonian *caatinga*: the term savannas refers to «open *campinarana* forest», a tropical vegetation growing on a sandy oligotrophic substrate (Anderson, 1981; Pires and Prance, 1985)) and bare soil (sand, roads, human constructions). Some natural herbaceous formations were included in the agricultural land class when their presence was not globally significant. The validity of the classes was

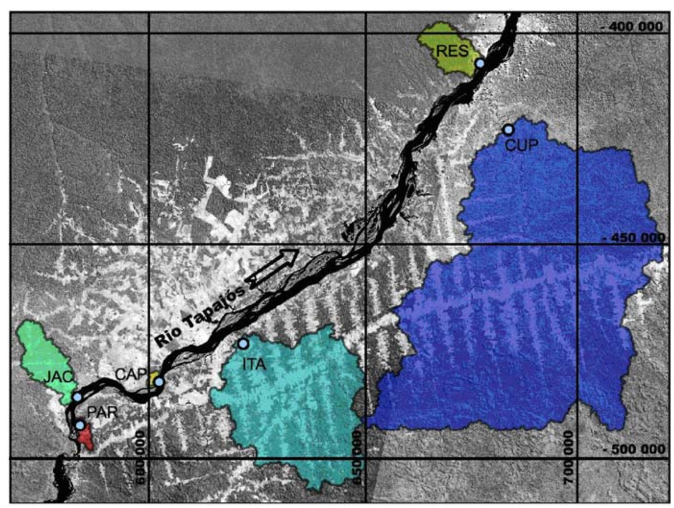

Fig. 1. Localization of 6 studied sites and delimitation of their watershed (RES = Restinga; CUP = Cupari; ITA = Itapacurazinho; CAP = Capituã; JAC = Jacaré and PAR = Paraná). Small dots found in each watershed represent sampling sites. The arrow indicates the flow of the Tapajós River.

Table 1
Summary of location and environmental data of the study sites.

Site	Type of site	Sampling location		Descending waters 2003			Low waters 2004		
				Mean depth of the site (m)	SPM[a] (mg/L)	T^b (°C)	Mean depth of the site (m)	SPM (mg/L)	T (°C)
ITA	River (lotic)	04°16′07.1″S	55°54′36.1″W	7.7	8.7	28.0	4.0	12.1	27.5
PAR	Channel (lotic)	04°27′29.2″S	56°14′59.8″W	6.0	5.3	n.d.	4.1	3.4	31.0
RES	Bay (lentic)	04°41′07.6″S	55°24′40.5″W	4.0	3.9	31.5	1.25	13.9	31.0
JAC	Lake (lentic)	04°22′16.9″S	56°14′36.7″W	3.5	5.2	30.0	1.25	13.7	n.d.
CUP	River (lotic)	03°49′42.6″S	55°21′21.2″W	5.8	n.d.	28.5	2.6	3.6	28.0
CAP	Lake (lentic)	04°19′55.9″S	56°04′44.3″W	4.7	12.6	n.d.	2.4	5.5	n.d.

n.d. = data not available.
[a] SPM = Suspended particulate matter.
[b] T = Water temperature.

confirmed by aerial pictures and by check points recorded during field work. Finally, surface areas of land occupation were recorded for each watershed concerned and values were transformed into percentages.

2.4. Statistical analyses

Fish Hg levels were firstly transformed ($\log_{10}$) because the normality and homoscedasticity conditions were not met. A *t*-test was applied to check if seasons had an influence on Hg concentrations in SPM. Mean Hg levels were compared by analysis of variance (ANOVA) with two classification criteria (season and site) to test for spatio-temporal differences between Hg levels. Because of the small number of *H. malabaricus* and *L. fasciatus* samples, we used an ANOVA with one criterion of classification to test the site effect on Hg concentrations in these two species. For mean comparisons, we used a Student *t*-test to determine the highest mean. When there were three values or more, H.S.D. of Tuckey's method was used (Scherrer, 1984). Geographical variations in Hg contamination of ichthyofauna were determined by factorial analysis of variance. This analysis took into account fish diets, seasons, and sampling sites. Finally, multiple regressions were used to measure the influence of the following variables (considered independently from one another): ecological, and anthropogenic variables and variables related to watershed uses, on Hg levels in key species representing the four trophic groups. Then, multiple regressions were used to measure the concomitant influence of selected variables. The first group of variables examined included fish size, ecological diversity, SPM (mg/L), $Hg_{(SPM)}$ (ng/g), habitat types (lentic or lotic) and hydrologic cycle (descending waters or low waters). The ecological diversity of the fish communities at each site was expressed using the Shannon–Wiener diversity index ($H'_{(BPUE)}$, normalized fisheries yield (kg/day)) (Magurran, 1988). In the second group of variables, the number of inhabitants of the communities studied and normalized fishing yields (kg/day) were examined. In the third group of variables, watershed use classes were obtained using the geographical information system (GIS) described above. Explanatory variables for linear regression models were selected or withdrawn using the progressive method (Scherrer, 1984).

3. Results

3.1. Spatial and temporal characterization of watersheds and of Hg levels in water and in SPM

The ITA and JAC watersheds had the largest areas occupied by anthropogenic activities and consequently the smallest surface areas of upland and inundated forests (taken together) (Table 2).

Hg concentrations measured in water samples from the aquatic environments we studied ranged from 0.46 to 1.47 ng/L (Table 2). Hg contents in SPM ranged from 64 to 200 ng/g and from 199 to 421 ng/g during descending and low waters, respectively. SPM collected during low waters was significantly more contaminated than SPM collected during descending waters ($p = 0.0064$). Moreover, during descending waters, the Hg contents in SPM collected at ITA and PAR were remarkably higher than those collected at other sites. During low waters, mean Hg content measured in SPM of ITA was the highest, and in SPM at RES was the lowest (Fig. 2).

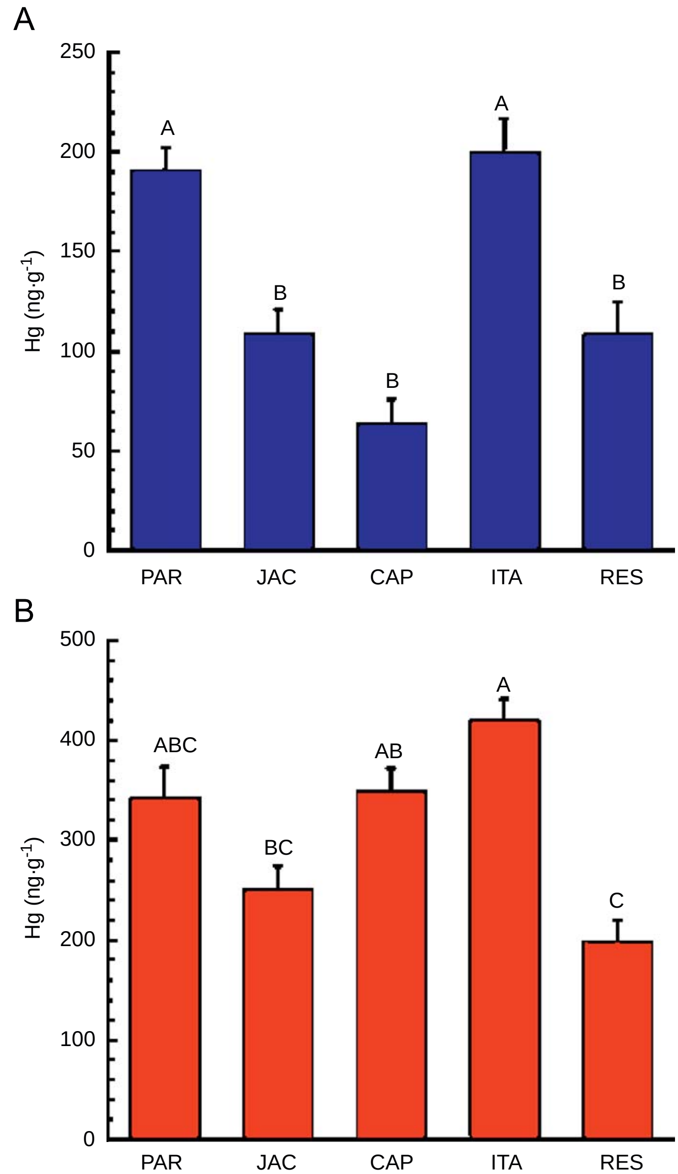

Fig. 2. Spatial variation of Hg measured in SPM collected in sampling sites during descending waters (A) and low waters (B). Mean Hg concentration and standard deviations (RES = Restinga; ITA = Itapacurazinho; CAP = Capituã; JAC = Jacaré and PAR = Paraná). Different letters indicate significant differences between corresponding means ($\alpha \leqslant 0.05$, Tukey test for multiple mean comparisons).

Table 2
General characterization of watersheds anthropization and Hg concentrations in water samples collected in studied sites.

Site	Number of inhabitants	Anthropization of the watershed (%)[a]	Forest cover in the watershed (%)[b]	Open water (%)	Aquatic vegetation (%)	Total dissolved Hg (ng/L)
ITA	122	31	62	1	6	1.47 ± 0.22
PAR	545	23	71	1	5	0.46 ± 0.05
RES	281	3	94	2	1	0.71 ± 0.06
JAC	172	30	68	1	1	0.75 ± 0.04
CUP	140	14	83	1	2	1.45 ± 0.12
CAP	0	4	80	12	4	n.d.

n.d. = data not available.
[a] In addition to the anthropogenic uses that were observed in watersheds, which are represented by crops, pastures, fallows and bare soils, this percentage also includes relatively open natural areas consisting of herbaceous and Amazonian *caatingas*. This percentage of natural open areas is relatively low comparatively to anthropogenic uses, thus it does not significantly influence global results.
[b] This percentage includes remaining intact inundated and upland forests.

3.2. Descriptive statistics of Hg concentrations in the flesh of sampled fish

Hg concentrations in fish ranged from 14 to 2330 ng/g during descending waters and from 9 to 3502 ng/g during low waters. Fish that were captured at RES had the lowest Hg concentrations while those captured at ITA had the highest. Table 3 presents mean Hg concentrations for the 10 key species captured during descending waters and low waters at our six study sites.

3.3. Influence of fish trophic level and size on Hg levels

Mercury contamination of fish followed the trend piscivores > herbivores > omnivores > detritivores, in both the seasons sampled. During descending and low waters, piscivorous fish presented on average five times and eight times more Hg than detritivorous fish. Fish size did not appear to be a key variable influencing Hg contamination levels and its influence varied with the species. A positive correlation between Hg contents and the size of the specimen was found only in *C. inornata* ($n = 183$; $R^2 = 0.24$; $p < 0.0001$) *C. labirinthyncus* ($n = 188$; $R^2 = 0.13$; $p < 0.0001$; and *H. malabaricus* ($n = 64$; $R^2 = 0.34$; $p < 0.0001$). In *P. squamosissimus*, a positive correlation was observed ($n = 199$; $R^2 = 0.26$; $p < 0.0001$), but only with the logarithmized data. In *S. vitattum*, a negative correlation was found between Hg contents and fish size ($n = 71$; $R^2 = 0.13$; $p = 0.0019$).

3.4. Concomitant influence of season and sampling site on Hg levels in key fish species (ANOVA)

In both seasons studied, fish captured in lotic habitats generally had higher levels of Hg than fish captured in lentic habitats ($p < 0.0001$). However, the concomitant influence of hydrologic cycle and site characteristics on Hg levels varied with the species.

In *C. inornata* specimens captured at JAC, ITA, and RES, we found that Hg levels were higher during descending waters than during low waters ($p = 0.0032$; $N = 159$). Moreover, we observed that in both periods of the hydrological cycle, fish captured at JAC presented higher Hg contents than those found in fish captured at RES ($p < 0.0001$), but did not differ from those found in fish at ITA. In *G. proximus*, Hg levels were only significantly influenced by the site ($p < 0.0001$; $n = 155$). Indeed, fish captured at ITA, JAC, and CUP showed higher Hg levels than those found in fish captured at CAP, PAR, and RES. In contrast, only the season had a significant effect on Hg levels ($p = 0.0093$; $N = 38$) in another herbivorous species, *S. vitattum*. Fish captured at JAC and PAR showed higher Hg levels during descending waters than during low waters. We tested the site effect in *L. fasciatus* specimens captured during low waters at CUP and RES, and during descending waters at PAR and JAC. Site did not have a significant influence on Hg levels. In *H. unimaculatus*, only one variation related to fish origin was observed ($p < 0.0001$; $n = 259$), i.e. fish captured at ITA were more contaminated than those captured at CUP, PAR, JAC, and RES. The same trend was observed in the second omnivorous species, *C. labyrinthicus* ($p < 0.0001$; $n = 162$), which was more contaminated when captured at ITA, JAC, and CAP than at RES. However, in the third omnivorous species, *A. laticeps*, site and season also significantly influenced Hg levels ($p = 0.0008$ and $p < 0.0001$; $n = 56$ respectively), i.e. Hg levels were higher during low waters and in fish captured at ITA, than in fish captured at RES. In the three predators, the influence of season and of site also varied with the species of fish.

In *P. squamosissimus*, no significant variation related to season or to sampling site was observed in fish captured at CUP, JAC, PAR, and RES (season: $p = 0.3418$ and site: $p = 0.6542$; $n = 184$). In *A. falcirostris* captured at ITA and JAC, only the effect of season was significant ($p = 0.0007$; $n = 54$) and fish captured during low waters were more contaminated than those captured during descending waters. However, in *H. malabariscus* captured at ITA, JAC, CAP, RES, and PAR, only the site had a significant effect on Hg levels ($p < 0.0001$; $n = 60$). Indeed, during descending waters, fish caught at the first three sites were more contaminated than those caught at the last two sites.

Finally, statistically significant variations in fish Hg levels could be linked to the three previously analyzed factors, that is, fish diet, stage of the hydrological cycle and sampling sites (Fig. 3).

3.5. Multiple regression models

3.5.1. Influence of anthropization of the watershed on levels of Hg in fish

The influence and importance of watershed uses on levels of Hg in fish were highlighted by multiple regression analyses. The first analyses showed that the seven predetermined classes that spatially characterized the watersheds all had an influence on Hg levels in omnivorous and piscivorous fish. Concerning the two other trophic groups (detritivorous and herbivorous), only the percentage of upland forest and of open water of the territory significantly influenced Hg levels in these fish.

In detritivorous fish, the four models that resulted from multiple regressions explained more than 40% of the variations in Hg levels measured in fish flesh. In herbivorous, omnivorous and piscivorous fish, the models identified by multiple regression only explained between 16% and 19% of the variations in Hg levels. Details on these analyses can be found in Sampaio da Silva (2008).

Bare soil surfaces contributed positively to the accumulation of Hg in the four trophic groups studied. Agricultural lands also contributed to an increase in Hg levels in detritivorous and herbivorous fish, whereas the opposite was observed in omnivorous and piscivorous fish. Aquatic vegetation was related to an increase in Hg levels in herbivorous and omnivorous fish. However, the opposite was observed in detritivorous and piscivorous fish. The percentage of the territory covered by inundated and upland forests as well as by savannas contributed to an increase in Hg concentrations in the flesh of herbivorous and piscivorous fish. However, inundated forest surfaces contributed negatively to Hg accumulation in detritivorous and omnivorous fish. Furthermore, the percentage of upland forest cover in the territory appeared to be related to an increase in Hg levels in omnivores, while this variable did not have a significant influence in detritivorous fish.

3.5.2. Influence of ecological and anthropogenic variables on Hg levels in fish

The influence and importance of the eight predetermined ecological and anthropogenic variables (fish size, site ecological diversity, SPM load and SPM Hg levels, habitat types (lentic or lotic), and season (descending waters or low waters) varied with the trophic group. In detritivorous fish, the eight variables always had an influence (positive or negative) on Hg levels. In omnivorous and piscivorous species, all the variables except fishing yield in the first group and fish size in the second group, significantly influenced (+ or −) Hg levels. In herbivorous fish, only habitat type (higher Hg level in fish captured in lotic habitats than in those captured in lentic habitats), site specific diversity (−), SPM load (+) and number of inhabitants in the watershed (−) significantly influenced Hg levels in fish flesh.

The models that resulted from multiple regressions were composed of a maximum of six variables and explained from 18%

Table 3
Hg concentration in muscular tissue of 10 key species from the 6 sites of the Tapajós river basin: Cupari (CUP), Itapacurazinho (ITA), Jacaré (JAC), Paraná (PAR), Restinga (RES) and Capituã (CAP).

	Cur ino Hg (ng/g)[a]	Geo pro Hg (ng/g)	Lep fas Hg (ng/g)	Sch vit Hg (ng/g)	Ana lat Hg (ng/g)	Cae lab Hg (ng/g)	Hem uni Hg (ng/g)	Hop mal Hg (ng/g)	Acefalci Hg (ng/g)	Pla squ Hg (ng/g)
CUP										
Fall	99 ± 1.4 (2)[b]	108 ± 24.8 (7)	68 ± 39.6 (2)	175 ± 77.6 (3)	n.d.	n.d	67 ± 32.4 (42)	400 ± 94.4 (4)	740 ± 308.0 (11)	392 ± 225.8 (17)
Low water	204 ± 134 (9)	200 ± 100.6 (11)	144 ± 119.1 (6)	11 (1)	95 ± 55.9 (2)	390 ± 205.7 (10)	125 ± 85.1 (19)	n.d	421 (1)	662 ± 337 (6)
ITA										
Fall	90 ± 24.4 (5)	137 ± 57.7 (4)	164 ± 84.6 (2)	156 ± 14.8 (2)	186 ± 114.1 (13)	334 ± 188.4 (18)	141 ± 109.8 (48)	n.d	867 ± 535.3 (15)	600 ± 279.5 (9)
Low water	100 ± 32.5 (17)	256 ± 85.5 (18)	295 ± 24.7 (2)	300 ± 18.5 (3)	460 ± 214.9 (7)	315 ± 122.7 (5)	169 ± 117.0 (20)	1398 ± 690.8 (12)	1659 ± 861.7 (24)	1367 ± 301.9 (2)
JAC										
Fall	175 ± 89.6 (63)	181 ± 110.9 (16)	145 ± 83.4 (18)	232 ± 138.8 (11)	228 ± 130.2 (6)	279 ± 85.9 (27)	84 ± 56.6 (25)	274 (1)	672 ± 384.2 (3)	567 ± 229.7 (31)
Low water	111 ± 47.9 (28)	160 ± 70.3 (7)	153 ± 58.7 (2)	91 ± 66.5 (15)	232 ± 57.3 (2)	257 ± 86.8 (13)	60 ± 41.7 (31)	810 ± 260.8 (5)	1182 ± 571.3 (12)	528 ± 150.6 (23)
PAR										
Fall	n.d.	183 ± 117.1 (4)	180 ± 106.5 (25)	187 ± 142.3 (10)	255 ± 178.0 (12)	n.d.	88 ± 81.1 (46)	n.d.	757 ± 266.6 (2)	649 ± 333.7 (75)
Low water	117 ± 58.4 (15)	99 ± 39.2 (11)	125 (1)	134 ± 89.6 (5)	n.d.	252 ± 108.0 (19)	67 ± 26.6 (9)	295 ± 79.0 (15)	497 ± 136.6 (12)	468 ± 164.9 (8)
RES										
Fall	49 ± 17.4 (28)	70 ± 27.4 (25)	n.d.	109 ± 67.6 (23)	116 ± 53.7 (3)	156 ± 144.7 (6)	39 ± 28.7 (66)	n.d.	198 (1)	452 ± 169.0 (12)
Low water	32 ± 18.0 (20)	79 ± 36.6 (16)	108 ± 65.5 (17)	211 ± 132.9 (2)	119 ± 50.1 (33)	92 ± 54.9 (38)	115 ± 63.6 (2)	501 ± 228.5 (8)	475 ± 176.7 (45)	550 ± 379.5 (14)
CAP										
Fall	n.d.	180 ± 93.4 (6)	n.d.	n.d.	153 ± 60.1 (2)	258 ± 81.8 (52)	463 ± 8.5 (2)	n.d.	n.d.	553 ± 73.5 (2)
Low water	n.d.	164 ± 106.0 (32)	n.d.	355 ± 68.6 (2)	273 ± 133.4 (16)	325 ± 149.4 (4)	851 ± 400.5 (17)	n.d.	887 ± 289.1 (27)	164 ± 45.9 (6)

n.d. = data not available.
Cur ino = *Curimata inornata*, Geo pro = *Geophagus proximus*, Sch vit = *Schizodon vitattum*, Lep fas = *Leporinus fasciatus*, Ana lat = *Anastomoides laticeps*, Hem uni = *Hemiodus unimaculatus*, Cae lab = *Caenotropus labyrinthicus*, Hop mal = *Hoplias malabaricus*, Pla squ = *Plagioscion squamosissimus*, Acefalci = *Acestrorhynchus falcirostris*.
[a] Values are presented as the mean ± standard deviation.
[b] Values in parentheses represented the number of samples.

to 57% of the variability observed in Hg levels measured in these four groups of fish. Details of these analyses can be found in Sampaio da Silva (2008).

SPM load had a negative influence on Hg accumulation in detritivorous and omnivorous fish and a positive influence in herbivorous and piscivorous fish. Moreover, Hg levels in SPM negatively influenced Hg levels in detritivorous, omnivorous and piscivorous fish flesh. Fishing yield did not exert a significant influence on Hg levels, but nevertheless had a positive influence on Hg levels in detritivorous fish and a negative influence on piscivorous fish. Site diversity also contributed positively (in detritivorous and omnivorous species) or negatively (in herbivorous and piscivorous species) to the accumulation of Hg in fish. The number of inhabitants in the watershed contributed negatively to the level of Hg observed in the four trophic groups studied.

3.5.3. *Multiple regression models including all variables*

Table 4 presents the details of the main multiple regression analyses and of prediction equations that arose from the four trophic groups studied. For detritivorous fish species, 12 variables significantly positively or negatively influenced levels of Hg in fish. More specifically, eight significant models explained more than 50% of variability of Hg levels. These models included a maximum of four variables. In herbivorous fish, only eight variables significantly influenced (+ or −) Hg accumulation and four models with a maximum of four variables were significant. These models explained from 22% to 24% of variability of Hg level in fish flesh. In omnivorous fish, 13 variables significantly influenced (+ or −) Hg accumulation in fish. Four models including four to six variables resulted. These models explained up to 31% of the variability of Hg levels. In piscivores, four significant models including a maximum of four variables explained more than 20% of Hg variability in fish in this group (Table 4).

Hg concentrations measured in dominant species were linearly related to one or more of the independent variables of this study. Our data also suggest that globally, all the selected variables have a significantly different influence on Hg level in the flesh of the

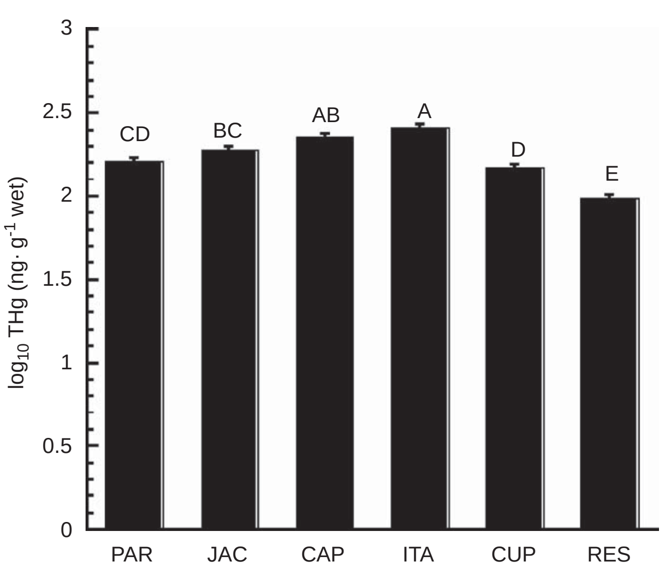

Fig. 3. Spatial variation of Hg levels in 10 key fish species captured in 6 sites of the Tapajós River basin. Factorial analysis of variance taking in account fish diets, seasons and sampling sites. THg data are expressed in least square means (ng/g). Different letters indicate significant differences between corresponding means ($\alpha \leqslant 0.05$, Tukey test for multiple mean comparisons).

species studied. This is particularly true for the following variables: number of inhabitants living in a given watershed, ichthyologic diversity at the study site, SPM load, Hg level in SPM, and the percentage of open water in the watershed. Furthermore, in the analysis by species, it was shown that fish Hg levels were not related to specimen size or to fishing yield.

Regarding key species, our analyses clearly showed that the spatial characteristics of the watershed had a significant effect on Hg levels in fish (Table 5). For instance, bigger watersheds consisting of agricultural and bare soil surfaces were associated with higher Hg levels in fish flesh. Finally, the first variable was positively correlated with an increase in Hg in *H. unimaculatus*, *P. squamosissimus*, *H. malabaricus*, *L. fasciatus*, *C. inornata*, and *A. falcirostris.* The second variable positively influenced Hg accumulation in the three other key species and also in *C. labyrinthicus* and *G. proximus.* Higher percentages of the watershed covered by savannas were associated with higher Hg levels in *H. unimaculatus*, *A. laticeps*, *L. fasciatus*, *S. vitattum*, *H. malabaricus*, *A. falcirostris*, *G. proximus*, and *P. squamosissimu*. A similar correlation was found between the percentage of watershed covered by upland forest and Hg levels in *H. unimaculatus*, *A. laticeps*, *L. fasciatus*, *S. vitattum*, *H. malabaricus* and *A. falcirostris*.

Our data showed that the percentage of surface water covered by aquatic vegetation positively influenced Hg accumulation in most of the key species (*H. unimaculatus*, *C. labyrinthicus*, *A. laticeps*, *S. vittatum*, *G. proximus*, *L. fasciatus*, *H. malabaricus* and *A. falcirostris*). However, this independent variable was negatively correlated with Hg levels measured in *P. squamosissimus*. The percentage of water also influenced Hg accumulation, positively in *L. fasciatus*, *S. vittatum*, *A. falcirostris*, and *G. proximus*, and negatively in *C. inornata* and *P. squamosissimus*.

Our data highlighted the fact that the influence of ecological and anthropogenic variables indeed varied with the fish species. More precisely, the number of inhabitants negatively influenced Hg levels in *L. fasciatus*, *H. malabaricus*, and *A. falcirostris* and positively influenced Hg levels in *P. squamosissimus* and *A. laticeps.* More site specific diversity was associated with higher levels of Hg in *H. unimaculatus*, *P. squamosissimus*, and *A. falcirostris*, whereas the opposite was found in *C. labyrinthicus* and *C. inornata.* SPM load positively influenced Hg levels in *L. fasciatum*, *S. vitattum*, *H. malabaricus*, *A. falcirostris*, and *G. proximus*, but had a negative effect in *A. laticeps* and *C. inornata.* Hg concentrations measured in SPM did not have a significant influence on Hg levels in all the species studied. Even when this influence was significant, it also varied with the species. Finally, a positive correlation with an increase in Hg levels in fish flesh was observed in *C. labyrinthicus* and *C. inornata* while a negative correlation was observed in *H. unimaculatus, A. laticeps*, and *P. squamosissimus*.

4. Discussion

According to the results of the present study, variations in Hg concentrations in the flesh of fish captured in lentic and lotic environments of the Tapajós River basin could be partially explained by environmental variables (Tables 6 and 7). Indeed, multiple regressions of the four trophic groups of fish suggested that a higher number of inhabitants were associated with less contaminated fish. Higher fishing yields by local fishermen were also linked with lower levels of Hg in fish (except for piscivores). Outside the Amazonian context, studies using different approaches reported that Hg concentrations were lower in slow-growing fish (Doyon et al., 1998; Essington and Houser, 2003; Harris and Bodaly, 1998; Olsson, 1976; Simoneau et al., 2005; Stafford and Haines, 2001; Stafford et al., 2004). Lower Hg concentrations have also been observed in fish that have higher

Table 4
Multiple regression models including all variables. Concomitant influence of watershed use and bio-ecological factors (independent variables) on Hg levels in four groups of fish (detritivores, herbivores, omnivores and piscivores) of the Tapajós basin in the Brazilian Amazon.

Model	Independent variables	Adjusted R^2	R^2	SSE	F	p	N
	Detritivores (*C. inornata*)						
1	Agricultural land ($R^2=0.42$), $H'_{(BPUE)}$ ($R^2=0.08$) and $Hg_{(SPM)}$ ($R^2=0.04$) $\log_{10}(Hg_{fish})=-0.030$(agricultural land)$-1.540(H'_{(BPUE)})+0.002(Hg_{(SPM)})+7.71$	0.53	0.54	10.2	67.7	<0.0001	176
2	Open water ($R^2=0.31$), inundated forest ($R^2=0.15$) and $H'_{(BPUE)}$ ($R^2=0.08$) $\log_{10}(Hg_{fish})=-0.136$(open water)$-0.008$(inundated forest)-0.370$(H'_{(BPUE)})$+4.00	0.53	0.54	10.3	65.3	<0.0001	186
3	Open water ($R^2=0.31$), agricultural land ($R^2=0.15$) and season ($R^2=0.07$) $\log_{10}(Hg_{fish})=-0.210$(water)+0.018(agricultural land)-0.103(season)+1.95	0.53	0.54	10.3	63.3	<0.0001	186
4	Bare soil ($R^2=0.39$), inundated forest ($R^2=0.07$) and SPM load ($R^2=0.07$) $\log_{10}(Hg_{fish})=-0.125$(bare soil)$-0.005$(inundated forest)-0.024(SPM load)+2.08	0.53	0.53	10.4	65.4	<0.0001	176
5	Upland forest ($R^2=0.18$), aquatic vegetation ($R^2=0.26$), SPM load ($R^2=0.04$) and inhabitants ($R^2=0.05$) $\log_{10}(Hg_{fish})=-0.020$(upland forest)+0.101(aquatic vegetation)-0.024(SPM load)-0.0008 (inhabitants)+1.35	0.52	0.53	10.4	48.8	<0.0001	176
6	Upland forest ($R^2=0.18$), aquatic vegetation ($R^2=0.26$) and $Hg_{(SPM)}$ ($R^2=0.08$) $\log_{10}(Hg_{fish})=-0.020$(upland forest)$-0.102$(aquatic vegetation)-0.0005$(Hg_{(SPM)})$+1.13	0.52	0.53	10.5	69.9	<0.0001	176
7	Bare soil ($R^2=0.39$), inundated forest ($R^2=0.07$) and $Hg_{(SPM)}$ ($R^2=0.06$) $\log_{10}(Hg_{fish})=-0.092$(bare soil)$-0.005$(inundated forest)$-0.0005(Hg_{(SPM)})$+2.10	0.51	0.52	10.6	62.3	<0.0001	176
8	Inundated forest ($R^2=0.42$), $H'_{(BPUE)}$ ($R^2=0.08$) and yield ($R^2=0.02$) $\log_{10}(Hg_{fish})=-0.013$(inundated forest)$-0.260(H'_{(BPUE)})-0.013+3.64$	0.51	0.52	10.7	57.9	<0.0001	186
	Herbivores (*S. vittatum, L. fasciatus and G. proximus*)						
1	Amazonian savanna ($R^2=0.18$), yield ($R^2=0.04$), inhabitants ($R^2=0.02$) and agricultural land ($R^2=0.02$) $\log_{10}(Hg_{FISH})=0.019$(Amazonian savanna)$-0.020$(yield)$-0.0003$(inhabitants)+0.010+2.08	0.24	0.25	15.6	20.3	<0.0001	306
2	Upland forest($R^2=0.10$), habitat ($R^2=0.07$) and inhabitants ($R^2=0.08$) $\log_{10}(Hg_{FISH})=-0.003$(inundated forest)$-0.143$(habitat)$-0.0006$(inhabitants)+2.39	0.24	0.25	15.8	19.4	<0.0001	306
3	aquatic vegetation ($R^2=0.18$) and bare soil ($R^2=0.06$) $\log_{10}(Hg_{FISH})=-0.041$(aquatic vegetation)+0.0004(bare soil)+1.92	0.22	0.23	15.4	35.5	<0.0001	306
	Omnivores (*C. labyrinthicus*, *A. laticeps* and *H. unimaculatus*)						
1	Bare soil ($R^2=0.11$), open water ($R^2=0.09$), $Hg_{(SPM)}$ ($R^2=0.04$), SPM load ($R^2=0.06$), season ($R^2=0.009$) and $H'_{(BPUE)}$ ($R^2=0.01$) $\log_{10}(Hg_{FISH})=0.29$(bare soil)+0.095(open water)+0.001$(Hg_{(SPM)})-0.81$(SPM load)+0.154(season)+0.675$(H'_{(BPUE)})-1.37$	0.31	0.32	49.6	36.3	<0.0001	469
2	Upland forest($R^2=0.11$), yield ($R^2=0.06$). $H'_{(BPUE)}$ ($R^2=0.11$), inhabitants ($R^2=0.03$) and Amazonian savanna ($R^2=0.007$) $\log_{10}(Hg_{FISH})=-0.014$(inundated forest)$-0.043$(yield)+0.572$(H'_{(BPUE)})-0.0004$(inhabitants)+0.007(savanna)+0.47	0.31	0.32	49.8	32.5	<0.0001	542
3	Upland forest($R^2=0.11$), SPM load ($R^2=0.02$), $H'_{(BPUE)}$ ($R^2=0.002$) and yield ($R^2=0.16$) $\log_{10}(Hg_{FISH})=-0.016$(inundated forest)$-0.017$(SPM load)+0.380$(H'_{(BPUE)})-0.045$(yield)+1.15	0.28	0.29	51.8	47.4	<0.0001	469
4	Aquatic vegetation ($R^2=0.12$), upland forest ($R^2=0.06$), $H'_{(BPUE)}$ ($R^2=0.05$) and inhabitants ($R^2=0.02$) $\log_{10}(Hg_{FISH})=0.092$(aquatic vegetation)+0.012(upland forest)+0.317$(H'_{(BPUE)})-0.00003$(inhabitants)+0.06	0.25	0.26	54.2	31.4	<0.0001	542
	Piscivores (*A. falcirostris, P. squamosissimus and H. malabaricus*)						
1	aquatic vegetation ($R^2=0.10$), SPM load ($R^2=0.08$) and $Hg_{(SPM)}$ ($R^2=0.04$) $\log_{10}(Hg_{FISH})=0.045$(aquatic vegetation)+0.037(SPM load)$-0.0005(Hg_{(SPM)})$+2.49	0.22	0.22	22.2	31.6	<0.0001	381
2	Amazonian savanna ($R^2=0.11$), open water ($R^2=0.05$) and SPM load ($R^2=0.05$) $\log_{10}(Hg_{FISH})=0.030$(Amazonian savanna)+0.025(open water)+0.016(SPM load)+2.39	0.21	0.22	22.3	33.9	<0.0001	381
3	Bare soil ($R^2=0.11$), open water ($R^2=0.05$), SPM load ($R^2=0.03$) and habitat ($R^2=0.03$) $\log_{10}(Hg_{FISH})=0.086$(bare soil)+0.029(water)+0.019(SPM load)-0.063(habitat)+2.40	0.21	0.22	22.3	25.0	<0.0001	381
4	Aquatic vegetation ($R^2=0.10$), bare soil ($R^2=0.04$), SPM load ($R^2=0.05$) and inhabitants ($R^2=0.02$) $\log_{10}(Hg_{FISH})=0.037$(aquatic vegetation)+0.033(bare soil)+0.013(SPM load)-0.0002(inhabitants)+2.55	0.21	0.21	22.5	25.0	<0.0001	381

weight to size ratios (Cizdziel et al., 2002; Greenfield et al., 2001; Suns and Hitchin, 1990). Moreover, intensive fishing in five small experimental lakes in northern Quebec (in 1997 and 1998) was associated with a decrease in Hg concentrations in walleye (*Sander vitreus*) (Doire et al., 2002; Surette et al., 2006). These authors observed a decrease of 33% in Hg levels after intensive fishing activities in two of the lakes, in parallel with increased fish growth rates. Furthermore, this decrease could not be explained by a change in walleye diet or in trophic structure or by a reduction in circulation of methylmercury (MeHg) in the system or of MeHg concentrations in specimens (Surette, 2005).

The results of the present study showed that the highest levels of Hg were found in predatory fish. These data are in agreement with studies previously carried out at other sites in the Tapajós basin (Malm et al., 1995; Lebel et al., 1997; Santos et al., 2000; Uryu et al., 2001; Sampaio da Silva et al., 2006). We compared our mean Hg concentrations per species with those found by Uryu et al. (2001) and Sampaio da Silva et al. (2006). In the first study,

Table 5
Multiple regression models including all variables. Concomitant influence of watershed use[§] and bio-ecological factors[§§] (independent variables) on Hg levels in 10 dominant species of the Tapajós basin in the Brazilian Amazon.

Model Diet[a]	Independent variables	Adjusted R^2	R^2	SSE	F	p	N
D	*Curimata inornata*						
1	% of upland forest($R^2 = 0.42$), $H'_{(BPUE)}$ ($R^2 = 0.08$) and % of open water ($R^2 = 0.03$)	0.53	0.54	10.3	65.3	<0.0001	186
	$\log_{10}(Hg_{FISH}) = -0.008$(inundated forest)$-0.370(H'_{(BPUE)})-0.136$(% of water)+4.00						
2	% of agricultural land ($R^2 = 0.42$), $H'_{(BPUE)}$ ($R^2 = 0.08$) and $Hg_{(SPM)}$ ($R^2 = 0.04$)	0.53	0.54	10.2	67.7	<0.0001	176
	$\log_{10}(Hg_{FISH}) = -0.030$(% of agricultural land)$-1.536(H'_{(BPUE)})+0.002(Hg_{(SPM)})+7.71$						
3	% of bare soil ($R^2 = 0.39$), SPM load ($R^2 = 0.11$) and yield ($R^2 = 0.02$)	0.52	0.53	10.5	64.2	<0.0001	176
	$\log_{10}(Hg_{FISH}) = -0.174$(% of bare soil)$-0.035$(SPM load)+0.012(yield)+1.80						
H	*Schizodon vittatum*						
1	yield ($R^2 = 0.09$) and SPM load ($R^2 = 0.06$) and % of upland forest (($R^2 = 0.12$)	0.24	0.27	4.3	8.2	<0.0001	70
	$\log_{10}(Hg_{FISH}) = -0.052$(yield)+0.043(SPM load)+0.015(% of upland forest)+1.60						
2	yield ($R^2 = 0.09$). % of open water ($R^2 = 0.01$) and % of Amazonian savanna ($R^2 = 0.14$)	0.20	0.24	4.5	5.9	$= 0.0012$	74
	$\log_{10}(Hg_{FISH}) = -0.015$(yield)+0.045(% of open water)+0.030(% of Amazonian savanna)+1.98						
3	% of aquatic vegetation ($R^2 = 0.12$)	0.10	0.11	5.2	7.3	$= 0.0084$	74
	$\log_{10}(Hg_{FISH}) = 0.051$(aquatic vegetation)+1.98						
H	*Leporinus fasciatus*						
1	% of Amazonian savanna ($R^2 = 0.15$)	0.14	0.15	3.4	10.4	<0.0001	75
	$\log_{10}(Hg_{FISH}) = 0.026$(% of Amazonian savanna)+1.98						
2	% of aquatic vegetation ($R^2 = 0.15$)	0.14	0.15	3.4	8.9	<0.0001	75
	$\log_{10}(Hg_{FISH}) = 0.050$(% of aquatic vegetation)+2.01						
H	*Geophagus proximus*						
1	% of bare soil ($R^2 = 0.27$) and % of open water ($R^2 = 0.16$)	0.43	0.44	6.2	23.2	<0.0001	157
	$\log_{10}(Hg_{FISH}) = 0.156$(% of bare soil)+0.030(% of water)+1.77						
2	% of aquatic vegetation ($R^2 = 0.27$), % of agricultural land ($R^2 = 0.11$) and inhabitants ($R^2 = 0.07$)	0.42	0.43	6.2	32.4	<0.0001	157
	$\log_{10}(Hg_{FISH}) = 0.048$(% of aquatic vegetation)+0.016(% of agricultural land)-0.0004(inhabitants)+1.89						
3	% of Amazonian savanna ($R^2 = 0.28$), % of upland forest($R^2 = 0.10$) and inhabitants ($R^2 = 0.06$) $\log_{10}(Hg_{FISH}) = 0.024$(% of Amazonian savanna)$-0.004$(% of inundated forest)$-0.0005$(inhabitants)+2.20	0.42	0.43	6.2	32.1	<0.0001	157
4	% of Amazonian savanna ($R^2 = 0.28$), % of upland forest ($R^2 = 0.10$), yield ($R^2 = 0.03$) and SPM load ($R^2 = 0.03$)	0.42	0.43	6.2	25.2	<0.0001	138
	$\log_{10}(Hg_{FISH}) = 0.037$(% of Amazonian savanna)+0.011(% of upland forest)-0.015(yield)+0.015(SPM load)+1.46						
O	*Caenotropus labyrinthicus*						
1	% of agricultural land ($R^2 = 0.37$), $H'_{(BPUE)}$ ($R^2 = 0.15$) and $Hg_{(SPM)}$ ($R^2 = 0.05$)	0.56	0.57	5.3	56.3	<0.0001	129
	$\log_{10}(Hg_{FISH}) = 0.024$(% of agricultural land)$-1.133(H'_{(BPUE)})-1.133(Hg_{(SPM)})+6.48$						
2	% of upland forest ($R^2 = 0.45$) and % of aquatic vegetation ($R^2 = 0.12$)	0.56	0.57	5.4	90.0	<0.0001	191
	$\log_{10}(Hg_{FISH}) = -0.010$(% of inundated forest)+0.046(% of aquatic vegetation)+2.531						
3	% of bare soil ($R^2 = 0.42$), $H'_{(BPUE)}$ ($R^2 = 0.09$), % of upland forest ($R^2 = 0.03$) and $Hg_{(SPM)}$ ($R^2 = 0.03$)	0.56	0.57	5.4	41.2	<0.0001	129
	$\log_{10}(Hg_{FISH}) = 0.091$(% of bare soil)$-0.792(H'_{(BPUE)})+0.008$(% of upland forest)$+0.0009(Hg_{(SPM)})+4.96$						
4	$Hg_{(SPM)}$ ($R^2 = 0.40$), % of upland forest ($R^2 = 0.10$) and habitat ($R^2 = 0.06$)	0.55	0.56	5.5	71.3	<0.0001	181
	$\log_{10}(Hg_{FISH}) = -0.0003(Hg_{(SPM)})-0.008$(% of inundated forest)$-0.085$(habitat)+2.71						
5	% of upland forest($R^2 = 0.45$) and $H'_{(BPUE)}$ ($R^2 = 0.09$) $\log_{10}(Hg_{FISH}) = 0.007$(% inundated forest)$-0.344(H'_{(BPUE)})+4.04$	0.53	0.54	5.7	70.0	<0.0001	139
O	*Anastomoides laticeps*						
1	$Hg_{(SPM)}$ ($R^2 = 0.13$), season ($R^2 = 0.17$), % of Amazonian savanna ($R^2 = 0.04$) and inhabitants ($R^2 = 0.03$)	0.35	0.37	5.0	13.4	<0.0001	95
	$\log_{10}(Hg_{FISH}) = -0.001(Hg_{(SPM)})+0.323$(season)+0.017(of Amazonian savanna)+0.0005(inhabitants)+2.50						
2	% of bare soil ($R^2 = 0.08$), SPM load ($R^2 = 0.08$) and season ($R^2 = 0.19$) $\log_{10}(Hg_{FISH}) = -0.110$(bare soil)$-0.050$(SPM load)+0.213(season)+2.52	0.34	0.36	5.1	13.8	<0.0001	95
3	% of aquatic vegetation ($R^2 = 0.21$) and % of upland forest ($R^2 = 0.08$)	0.27	0.29	5.7	16.3	<0.0001	97
	$\log_{10}(Hg_{FISH}) = 0.055$(% of aquatic vegetation)+0.007(% of upland forest)+1.81						
O	*Hemiodus unimaculatus*						
1	% of bare soil ($R^2 = 0.27$), yield ($R^2 = 0.03$) and % of upland forest ($R^2 = 0.02$)	0.31	0.32	20.6	29.1	<0.0001	325
	$\log_{10}(Hg_{FISH}) = 0.141$(% of bare soil)$-0.021$(yield)+0.006(% of upland forest)+1.44						
2	habitat ($R^2 = 0.17$), inhabitants ($R^2 = 0.09$) and % of upland forest($R^2 = 0.04$)	0.30	0.30	21.0	11.8	<0.0001	325
	$\log_{10}(Hg_{FISH}) = -0.155$(habitat)$-0.0006$(inhabitants)$-0.004$(% of inundated forest)+2.15						
3	% of Amazonian savanna ($R^2 = 0.29$)	0.29	0.29	21.4	103.8	<0.0001	325

	$\log_{10}(Hg_{FISH}) = 0.035$(% of Amazonian savanna)+1.56						
4	% of aquatic vegetation ($R^2 = 0.24$), $H'_{(BPUE)}$ ($R^2 = 0.02$) and $Hg_{(SPM)}$ ($R^2 = 0.02$)	0.26	0.27	21.9	30.5	<0.0001	247
	$\log_{10}(Hg_{FISH}) = 0.108$(% of aquatic vegetation)+1.244($H'_{(BPUE)}$)−0.002($Hg_{(SPM)}$)-3.34						
5	% of aquatic vegetation ($R^2 = 0.24$)+$H'_{(BPUE)}$ ($R^2 = 0.25$)+$Hg_{(SPM)}$ ($R^2 = 0.62$)	0.26	0.27	21.9	30.5	<0.0001	247
	$\log_{10}(Hg_{FISH}) = 0.108$(% of aquatic vegetation)+1.244($H'_{(BPUE)}$)−0.002($Hg_{(SPM)}$)-3.34						
P	*Acestrorhynchus falcirostris*						
1	% of upland forest ($R^2 = 0.20$), SPM load ($R^2 = 0.11$), yield ($R^2 = 0.14$) and habitat ($R^2 = 0.05$)	0.49	0.50	4.9	34.2	<0.0001	140
	$\log_{10}(Hg_{FISH}) = -0.016$(% of inundated forest)+0.060(SPM load)−0.025−0.064(habitat)+3.05						
2	% of aquatic vegetation ($R^2 = 0.26$), yield ($R^2 = 0.11$), inhabitants ($R^2 = 0.08$) and season ($R^2 = 0.02$)	0.47	0.48	5.11	30.8	<0.0001	152
	$\log_{10}(Hg_{FISH}) = 0.067$(% of aquatic vegetation)+0.022(yield)−0.0005(inhabitants)−0.062(season)+2.58						
3	% of aquatic vegetation ($R^2 = 0.26$), yield ($R^2 = 0.11$), inhabitants ($R^2 = 0.08$) and $H'_{(BPUE)}$ ($R^2 = 0.02$)	0.46	0.49	5.2	31.3	<0.0001	152
	$\log_{10}(Hg_{FISH}) = 0.078$(% of aquatic vegetation)+0.022(yield)−0.0006(inhabitants)+0.170($H'_{(BPUE)}$)+1.85						
4	% of agricultural land ($R^2 = 0.14$), inhabitants ($R^2 = 0.24$), season ($R^2 = 0.04$) and habitat ($R^2 = 0.08$)	0.48	0.50	5.0	31.4	<0.0001	152
	$\log_{10}(Hg_{FISH}) = 0.015$(% of agricultural land)+2.864(inhabitants)−0.113(season)−0.094(habitat)+2.86						
5	% of Amazonian savanna ($R^2 = 0.24$), inhabitants ($R^2 = 0.15$), season ($R^2 = 0.08$) and % of upland forest ($R^2 = 0.03$)	0.48	0.49	5.0	31.5	<0.0001	152
	$\log_{10}(Hg_{FISH}) = 0.030$(% of Amazonian savanna)−0.0005(inhabitants)−0.112(season)+0.005(% of upland forest)+2.53						
6	% of bare soil ($R^2 = 0.26$) and % of open water ($R^2 = 0.15$)	0.40	0.41	5.8	39.5	<0.0001	152
	$\log_{10}(Hg_{FISH}) = 0.127$(% of bare soil)+0.029(% of open water)+2.570						
P	*Hoplias malabaricus*						
1	inhabitants ($R^2 = 0.49$), % of Amazonian savanna ($R^2 = 0.11$) and season ($R^2 = 0.03$)	0.61	0.63	2.0	34.9	<0.0001	64
	$\log_{10}(Hg_{FISH}) = -0.001$(inhabitants)+0.0196(% of Amazonian savanna)−0.217(season)						
2	% of bare soil ($R^2 = 0.06$), inhabitants ($R^2 = 0.52$) and season ($R^2 = 0.04$)	0.61	0.63	2.0	33.4	<0.0001	64
	$\log_{10}(Hg_{FISH}) = 0.072$(% of bare soil)−0.001(inhabitants)−0.248(season)+2.68						
3	% of aquatic vegetation ($R^2 = 0.14$), SPM load ($R^2 = 0.29$) and % of upland forest ($R^2 = 0.19$)	0.60	0.62	2.0	18.1	<0.0001	60
	$\log_{10}(Hg_{FISH}) = 0.114$(% of aquatic vegetation)+0.065(SPM load)+0.014(upland forest)						
4	% of aquatic vegetation ($R^2 = 0.14$), SPM load ($R^2 = 0.29$) and inhabitants ($R^2 = 0.18$)	0.60	0.62	2.1	23.7	<0.0001	60
	$\log_{10}(Hg_{FISH}) = 0.055$(% of vegetation)+0.025(SPM load)−0.0007(inhabitants)+2.53						
P	*Plagioscion squamosissimus*						
1	% of open water ($R^2 = 0.17$)	0.17	0.17	8.5	20.0	<0.0001	203
	$\log_{10}(Hg_{FISH}) = -0.049$(% of open water)+2.78						
2	% of bare soil ($R^2 = 0.04$), $Hg_{(SPM)}$ ($R^2 = 0.0002$), $H'_{(BPUE)}$ ($R^2 = 0.008$) and inhabitants ($R^2 = 0.11$)	0.14	0.15	8.7	7.9	<0.0001	178
	$\log_{10}(Hg_{FISH}) = 0.070$(% of bare soil)−0.0006($Hg_{(SPM)}$)+0.482($H'_{(BPUE)}$)+0.0005(inhabitants)						
3	% of Amazonian savanna ($R^2 = 0.06$), inhabitants ($R^2 = 0.03$) and % of aquatic vegetation ($R^2 = 0.06$)	0.14	0.15	8.8	9.8	<0.0001	203
	$\log_{10}(Hg_{FISH}) = 0.033$(% of Amazonian savanna)+0.0004(inhabitants)−0.045(% of aquatic vegetation)+2.50						

Independent variables: territory (the following classes are expressed in % of watershed surface: [§] open water. aquatic vegetation. inundated and upland forests, agricultural land. Amazonian savanna and bare soil) and bio-ecology ([§§] size classes (<20 cm; $\geqslant 20$ cm and >40 cm). site diversity ($H'_{(BPUE)}$). normalized fishing yield (kg/day). habitat (lentic or lotic). season (beginning of fall-2003 or low water-2004). number of inhabitants. SPM load (mg/L) and $Hg_{(SPM)}$ (ppb)).

[a] D = detritivores; H = herbivores; O = omnivores and P = piscivores.

Table 6
Influence of bio-ecological and anthropogenic variables on Hg levels in fish belonging to 4 trophic levels of the Tapajós basin, Brazilian Amazon (Data coming from multiple regressions).

Variables / Trophic group	Nb of inhabitants in the watershed	Fishing yield	Habitat[a]	Season[b]	H′	SPM load	SPM Hg	Bare soil	Agricultural land	Upland forest	Inundated forest	Aquatic vegetation	Amazonian savanna	Open water
Detritivore	–	–	n.s	+slack water	–	–	–	+	+	+	–	+	n.s	–
Herbivore	–	–	+lotic	n.s	n.s.	n.s	n.s	+	+		–	+	+	n.s
Omnivore	–	–	+lentic	+slack water	+	+	+	+	n.s	+	–	+	+	+
Piscivore	–	n.s	+lotic	n.s	n.s	+	–	+	n.s	n.s	n.s	+	+	+

(+) = significantly positive effect.
(–) = significantly negative effect.
n.s = non-significant effect).
[a] Hg levels were higher in fish captured in lotic environments comparatively to those of fish from lentic environments.
[b] Hg levels were higher in fish captured during fall comparatively to those of fish captured during low water.

Table 7
Influence of bio-ecological and anthropic variables on Hg levels in 10 dominant species in aquatic ecosystems of the Tapajós basin, Brazilian Amazon (Data coming from multiple regressions).

Variables / Species	Nb of inhabitants in the watershed	Fishing yield	Habitat[a]	Season[b]	H′	SPM load	SPM Hg	Bare soil	Agricultural land	Upland forest	Inundated forest	Aquatic vegetation	Amazonian savanna	Open water
Cur ino	n.s	+	n.s	n.s	–	–	+	+	+	n.s	–	n.s	n.s	–
Geo pro	–	–	n.s	n.s	n.s	+	n.s	+	+	n.s	–	+	+	+
Sch vit	n.s	–	n.s	n.s	n.s	+	n.s	n.s	n.s	+	n.s	+	+	+
Lep fas	–	–	n.s	n.s	n.s	+	n.s	+	+	+	–	+	+	+
Ana lat	+	n.s	n.s	+	n.s	–	–	n.s	n.s	+	n.s	+	+	n.s
Hem uni	–	–	+	n.s	+	n.s	–	+	n.s	+	–	+	+	n.s
Cae lab	n.s	n.s	+	n.s	–	n.s	+	n.s	+	+	–	+	n.s	n.s
Hop mal	–	n.s	n.s	+	n.s	+	n.s	+	n.s	+	n.s	+	+	n.s
Acefalci	–	+	+	+	+	+	n.s	+	+	+	–	+	+	+
Pla squ	+	n.s	n.s	n.s	+	n.s	–	+	n.s	n.s	n.s	–	+	–

(+) = significantly positive effect.
(–) = significantly negative effect.
n.s = non-significant effect).
[a] Hg levels were higher in fish captured in lotic environments comparatively to those of fish from lentic environments.
[b] Hg levels were higher in fish captured during fall comparatively to those of fish captured during low water.

fish were captured between 1991 and 1996. The six fish species were the same in both studies. We observed that Hg levels in predatory fish were quite similar in the two studies but that Hg contamination levels in non-predatory species appeared to be higher in the present study. In the second comparison, fish were captured in 2000 and 2001 and seven species were common to the two studies. Moreover, the same sampling and analyses protocols were used in the two studies. The mean concentrations observed in the previous study were noticeably lower than in the present study. However, it is important to note that the sampling sites were different. The other available data on Hg in fish were not usable for long-term monitoring of changing trends in Hg contamination in fish in the region. This was principally due to the use of only vernacular names to identify the fish (Lebel et al., 1997), lack of knowledge of the origin of the sample, or the use of very limited categories, for example carnivorous or non-carnivorous fish (Malm et al., 1995; Santos et al., 2000). It is currently recognized that variations in diet during the life cycle of the species can influence Hg concentration in the fish muscle tissue of fish (Sampaio da Silva et al., 2005; Bastos et al., 2007). However, we can assume that, independently of spatio-temporal characteristics, Hg levels follow the trend piscivores > omnivores > herbivores > detritivores.

The concentrations of Hg in the food fish ingest as well as the fish position in the trophic chain are of major importance in explaining the Hg levels in fish flesh (Leaner and Mason, 2002). While the study of the trophic chain structure was not analysed in detail in the present study, we base our interpretations on ecological information gathered in a previous study by Sampaio da Silva (2008). All aquatic environments studied had a rich and diversified ichthyofauna. Diversity indexes were slightly higher during low waters with a greater abundance of predator species. In spite of the results obtained in our previous study which clearly showed that fish communities at our study sites varied from one site to another and/or between seasons, we cannot affirm that there was a difference in the length of the trophic chains. In Sampaio da Silva (2008), we also observed that CAP presented a mix of species that was different from other sites. This did not appear to have an impact on the level of contamination of fish since the level was similar to that at the site with the most contaminated fish (ITA).

As suggested by the results obtained in the present study, it is clear that Hg contamination in fish can vary with the season however, this variation is not independent of the species. Moreover, seasonal variability cannot be entirely associated with a particular trophic group, as was observed in a detritivorous species (*C. inornata*), in an herbivorous species (*S. vitattum*), in one of the omnivorous species (*A. laticeps*), and also in one of the piscivorous species (*A. falcistrosis*). On the other hand, our study showed that non-predatory species have higher Hg levels in their flesh during descending waters, while in predatory species, higher Hg levels were observed during low waters. Some authors have suggested that periodical floods related to hydrological cycles could increase the bioavailability and/or production of MeHg in reservoirs and wet areas (Caldwell and Canavan, 1998; Snodgrass et al., 2000).

The low water period is dramatic for the majority of the non-predatory species because the aquatic environment is more restricted, provides less food, and shelter is scarce. However, this period is more favorable for the capture of prey by predatory fish (Lowe-Mcconnell, 1987; Barthem and Goulding, 1997; Jepsen, 1997; Arrington et al., 2002; Hoeinghaus et al., 2003; Arrington et al., 2006). This could be the cause of the higher Hg concentrations that were observed in *A. falcirostris* during low waters. Hylander et al. (2000) also observed higher Hg concentrations during the dry season in three predatory species captured in the Pantanal region. In omnivorous and piscivorous species, *H. unimaculatus* and *H. malabaricus*, the absence of seasonal variability demonstrated in the present study agrees with the data presented by Dórea et al. (2006).

Several land uses at the watershed level of small affluent of the Tapajós River, such as bare soils, agricultural land, aquatic vegetation and Amazonian savannas also contributed to the increase in Hg levels in fish. Some studies also reported correlations between watershed characteristics and Hg levels in water columns, periphyton and fish (Rudd, 1995; Hurley et al., 1995; Desrosiers et al., 2006; Sveinsdottir and Mason, 2005). As fish from the site with the most forest cover (RES) were the least contaminated by Hg (Table 2), it appears from our study that surfaces without forest cover are directly associated with higher Hg levels in fish. Previous studies already demonstrated that deforestation and agricultural practices, which are associated with an increase in sediment load, positively contributed to the presence of Hg in aquatic ecosystems (Maurice-Bourgoin et al., 2000). In the Tapajós region, Roulet et al. (2000) observed that lixiviation could significantly increase Hg transfer from soil to aquatic ecosystems. On the other hand, this information was never directly related to levels of contamination in fish. When combined, our study variables explained more than 50% of variability of Hg levels in detritivorous species, up to 31% in omnivores, and more than 20% in herbivores and piscivores. It also emerged that most classes of land uses had a similar effect on Hg levels in fish while the influence of all other variables appeared to be intrinsically related to fish species.

Although we did not measure the Hg methylation rate in our study environments, geographical data clearly showed that ITA, where fish exhibited the highest Hg levels, had one of the largest aquatic vegetation covers (6% of the watershed surface). It is recognized that wetlands are a significant source of MeHg for nearby lakes and rivers (Rudd, 1995; Grigal, 2002). Indeed, they are a source of dissolved organic matter, which reduces oxygen levels and thus facilitates bacterial Hg methylation (Gilmour et al., 1992; Crisman et al., 1998). It is also known that, in the Amazon, macrophyte mats are privileged sites for Hg methylation (Guimarães et al., 2000). This could be the explanation for the high Hg contents that were found in the ITA ichthyofauna. Our data indeed indicated that the presence of aquatic vegetation surfaces contributes positively to an increase in Hg contamination in almost all fish species that were studied in the present work (Tables 6 and 7). Biberhofer and Rukavina (2002) demonstrated that submerged aquatic vegetation could prevent the resuspension of sediments by turbulence. Other studies indicate that roots of aquatic plants can also absorb Hg present in water, sediments and air (Thompson-Roberts et al., 1999; Skinner et al., 2007). We can thus assume that submerged aquatic vegetation can play a role in the Hg cycle in lakes, rivers, and wetlands through Hg transfer to herbivores, through Hg liberation during decomposition, and through the release of the metal into the water column (Thompson-Roberts et al., 1999).

Uryu et al. (2001) found a gradient of Hg contamination in fish captured at different sites located between Santarém (a city located north of our study area) and Teles Pires (a site located far south of our study area). These authors observed that omnivorous and piscivorous fish captured near gold-mining areas were noticeably more contaminated by Hg than those captured near Santarém. Contrary to the results of these authors, in our study, no geographical gradient was observed in the contamination of ichthyofauna.

A large proportion of the contaminants present as trace elements can be associated with allochthonous SPM, which may be a vector of terrigenous matter and of the transfer of Hg to aquatic ecosystems. Previous research indeed showed that Hg is principally transferred into tropical hydrosystems in its inorganic form mainly complexed to organic matter and/or iron, aluminum and manganese oxy-hydroxides (Roulet et al., 1998, 2000, 2001). In aquatic environments, the diversity of abiotic and biotic conditions influences both the dynamics and bioavailability of the metal. It is recognized that Hg transfer towards aquatic systems is favoured during high water periods (Bisinoti and Jardim, 2004). During this season, the aquatic ecosystems studied here were, to varying degrees, under the direct influence of the Tapajós River. The depth of the aquatic environment varied by almost 4 m between the two seasons. Because of the heavy rainfall that characterizes tropical environments, we expected that the SPM load would be higher at the beginning of high waters and during low waters. However, our data indicated that at most sites, except PAR and CAP, maximum values occurred during low waters. Moreover, differences in SPM loads cannot be explained by the lotic or lentic nature of the study environments. Some authors also observed an increase in SPM loads in deep water during low waters. This has been related to winds, which can cause the resuspension of unconsolidated sediments, to variations in algae biomass as well as to watershed erosion (Marlier, 1965; Geisler, 1969; Boechat-Lopes et al., 1982). Two other factors may also explain the variability of SPM loads: watershed morphology and/or anthropogenic activities. Our results also indicated relatively high SPM Hg concentrations and suggested spatio-temporal variability of the contamination. This variability could thus be explained by a difference in the nature of the SPM (Idlafkih et al., 1995) or by increased remobilisation of Hg contaminated sediments occurring along with the major hydrological event, i.e., the low water period. Seasonal trends in Hg contamination of SPM were identical to those of the ichthyofauna, that is, sites with the most contaminated SPM also had the most contaminated fish (Figs. 2 and 3). However, SPM played a minor role in the models that were revealed by our results. These two observations allow us to conclude that Hg levels in SPM are not a good predictive variable for the state of contamination of the ichthyofauna in lentic and lotic habitats in the Tapajós watershed. Our data also suggest that in spite of similar Hg contamination levels in SPM, such as at RES, JAC, and CAP during descending waters and at ITA, PAR, and JAC during low waters, the specific fate of this metal in different aquatic environments can generate distinct levels of Hg contamination (Figs. 2 and 3). Finally, we suppose that the amount of Hg available in aquatic systems varies right from the beginning and, consequently, that the initial exposure of the lowest trophic groups and/or that specific site characteristics can influence the fate of Hg once it is in the aquatic environment to different degrees.

Hg is found in various forms and at different degrees of toxicity in the environment. However, MeHg is recognized to be the most toxic and, the dominant form in fish flesh (Grieb et al., 1990). MeHg production in the aquatic systems is not correlated to THg but is strongly associated with factors favouring methylation by microbial communities, such as limnological characteristics, presence of macrophytes (Grieb et al., 1990; Guimarães et al., 2000; Watras et al., 1998), anoxic conditions, increased organic matter contents (Driscoll et al., 1995; Hurley et al., 1991), and finally, seasonal floods (Kelley et al., 1997). Hg in fish flesh can thus be affected by seasonal variations in the methylation process. Results of a previous study, Sampaio da Silva et al. (2006) suggested that Hg concentrations in muscles of fish from the Tapajós varied seasonally.

Globally, our results tended to show that during descending waters, fish in lotic environments are more contaminated by Hg than fish in lentic environments. However, habitat type, watershed size or site depth cannot entirely explain the high Hg concentrations observed in fish captured at ITA. This is because (i) fish captured in other lotic environments (PAR and CUP) presented intermediate levels of contamination (Fig. 3); (ii) ITA is not the largest or the smallest site (Fig. 1); and (iii) this site is not the deepest or the shallowest (Table 1). In previous studies, it was demonstrated that fish in small lakes had the highest levels of Hg (Greenfield et al., 2001; Bodaly et al., 1993). In temperate or boreal environments, the negative correlation between lake size and Hg concentrations in fish has been attributed to warmer water temperature in smaller lakes, which could result in enhanced methylation of the metal. Observations concerning higher methylation rates in littoral sediments (Ramlal et al., 1993) indeed indicated that smaller lakes had higher Hg levels in their biota because of enhanced methylation. However, a decrease in the size of lakes can also increase the relative inputs of allochthonous organic matter and thus increase the affluence of derived materials in wet zones, proportionally to the total volume of the lake.

5. Conclusion

In this study, we found that bio-ecological properties, anthropogenic variables and variables related to spatial characterization of watersheds are key elements for the prediction of Hg concentrations in the local ichthyofauna. These groups of variables (taken one by one in multiple regression analyses, without any interaction with others) independently represented very good elements for predicting Hg concentrations, particularly in detritivorous fish. The percentage of the watershed surface under inundated forest cover and open water explained more than 40% of the variability of Hg levels in this group of fish. The same prediction power was observed in a model including only the percentage of the watershed converted to agriculture, and consisting of open water. In other groups of fish, watershed uses provided moderate prediction power for Hg concentrations (from 16% to 19%). On the other hand, for the four fish trophic groups, the explanatory power increased with the addition of eco-biological variables to the models (from 21% to 53%). The new data that resulted from this study allowed us to demonstrate that simple models composed of watershed spatial characterization and easily obtained variables, such as SPM load and Hg levels in SPM, number of inhabitants, habitat type and season have an important prediction power for levels of Hg in fish.

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