

Small-scale gold mining erodes fish assemblage structure in small neotropical streams

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Abstract The current gold rush experienced by the Guiana shield is profoundly disturbing freshwater ecosystems. Indeed, streams act as receptors for the water that drains gold mining sites and that contain a high load of sediment and toxicants. We here investigated how gold mining activities affect the structure of fish assemblages in small forest streams in French Guiana. We selected six streams subjected to different types of gold mining impact (reference, former gold mining and currently exploited sites) to measure the impact of gold mining on downstream fish assemblages, but also to determine the resilience of fish assemblages after stopping mining activities. Although overall descriptors of the assemblage such as species richness and fish biomass were not sensitive to gold mining, the fish taxonomic composition was strongly influenced. Furthermore, we showed that the functional structure of fish assemblages was significantly affected by the mining activities favouring smaller and ubiquitous fish at the expense of bigger and habitat specialist species. Even in areas where mining activities had stopped for some time, site resilience was incomplete.

Keywords Compositional similarity · Fish species traits · Functional diversity · NMDS · Resilience · Turbidity

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Introduction

The Guiana shield is experiencing an unprecedented gold rush, with a 60-fold increase in gold production reported in only two decades (Hammond et al. 2007). Such exponential growth of mining activities has detrimental environmental consequences due to deforestation, road construction, erosion, and heavy metal pollution (Cleary 1990). Freshwater ecosystems are particularly impacted by these mining operations, as the alluvial gold deposits are mined by washing the soils adjacent to the streams with high pressure water jets. In addition to the deforestation on the mining sites that can impact the aquatic ecosystem, the streams also acts as the receptors for the draining water containing a high load of sediments but also accumulating toxicants, such as mercury, which is used to recover gold from the sediment (Watts et al. 2003).

Although much concern has been given to mercury bioaccumulation through the trophic chain (Boudou et al. 2005; Durrieu et al. 2005), less attention has been devoted to the other environmental impacts of gold mining. Among these, soil erosion is probably the most pervasive, as it strikingly increases sediment load in the streams located downstream from the gold mines. It is indeed recognized that mining-related stream sediment levels largely exceed those generated by other land-use changes, such as deforestation, agricultural intensification, road-building, or urbanization (Bruijnzeel 1993; Krishnaswamy et al. 2006). This tendency is triggered in the Guiana shield where the rivers are characterized by some of the lowest levels of natural suspended sediment loads of the world (Hammond et al. 2007). The adverse impacts on stream ecosystems caused by increasing sediment load have been fairly well documented in temperate environments (Wood and Armitage 1997) and there is experimental evidence that the turbidity generated by fine sediment can affect species survival by clogging of gills, abrasion and adsorption of heavy metals. It also affects stream productivity by smothering of algae and benthic invertebrates (Parkhill and Gulliver 2002) and species interactions by reducing water clarity (Utne-Palm 2002; Pekcan-Hekim and Lappalainen 2006). Gold mining activities are therefore prone to affect aquatic ecosystem structure and function in various ways among which heavy metal pollution and sediment load have been studied in most detail to date (Tarras-Whalberg et al. 2001). There is however little information on the overall impact of gold mining activities (Mol and Ouboter 2004; Mendiola 2008; Yule et al. 2010). These data are particularly scarce for the most abundant and widespread type of gold extraction in the Guiana shield, namely small scale gold mines (Hammond et al. 2007). Indeed, at least 900 small scale gold mines have been recorded in the Guiana shield, and probably many more are unrecorded and under development (Hammond et al. 2007; Coppel et al. 2008). These open-cast mines usually involve small groups of workers (mostly 5–50 people) who settle on small forest streams. The remoteness of the sites, as well as the illegal nature of these mining operations may explain our limited knowledge of the environmental impact of these small-scale gold mining activities. Hence, the works of Mol and Ouboter (2004), Mendiola (2008) and Yule et al. (2010) are, to our knowledge, the only studies reporting the overall impact of small-scale gold mines in the tropics. Although these works demonstrated that fish assemblages are affected by mining activities with substantial changes in species composition and abundance, they did not clearly disentangle the relative effects of anthropogenic and environmental factors. Separating the effects of gold mining from other environmental factors as yet remains to be thoroughly considered, and is a prerequisite to a realistic understanding of the impact of gold mining on neotropical biodiversity.

Based on this knowledge, we investigated here how small-scale gold mining activities affect the structure of fish assemblages in small forest streams of French Guiana. We paid

particular attention to selecting similar streams differently affected by gold mining (reference, former gold mining and currently exploited sites), and to control for environmental variability. Our aim was to measure the impact of gold mining on downstream fish assemblages, but also to determine the resilience of fish assemblages after stopping gold mining activities. We first investigated how gold mining affects overall descriptors of the assemblage such as species richness and fish biomass. Second, we tested if gold mining can be considered as a major driver of the taxonomic composition of fish assemblages. Third, we investigated how gold mining affects the functional structure of fish assemblages by considering biological traits related to trophic level, body size and habitat selection. We hence hypothesized that (i) reducing water transparency increases the proportion of generalist feeders in the assemblage and in turn reduces the proportion of phytophagous and predator species; (ii) reducing stream productivity and increasing toxicant concentration increases the mortality rate for long lived and hence large bodied species, therefore favouring smaller fish; (iii) habitat specialists (i.e. species restricted to small headwater streams) are prone to be more sensitive to environmental changes than ubiquitous species (i.e. species found in both streams and large rivers), and are therefore less represented in disturbed sites.

Methods

Study sites

The study was conducted in November 2008 in the Nouragues Nature Reserve (French Guiana). Six small tributaries of the Arataï River, which then flows into the Approuague River, were selected as sampling sites (Fig. 1). The six streams were all low-order forest streams that flow on the same geological formation (Delor et al. 2001) and share similar physical characteristics (Table 1). Three levels of gold mining activities were considered: (i) reference sites (sites R1 and R2) which were never exploited; (ii) formerly exploited sites (F1 and F2) where gold mining activity was recorded during the year before sampling, but where extraction had stopped for at least 6 months; and (iii) currently exploited streams (C1 and C2) which were being mined at the time of sampling. The current and former mines were located in the upstream part of the waterways, and were small, illegal exploitations. The sampling sites were located in the lower part of the stream and all at a similar distance (c.a. 200 m) from their confluence with the Arataï River. Sampling sites were therefore located several hundred meters downstream from the gold mines, and collect all the effluents produced by the mining activity.

Fish sampling and habitat characterization

In each site we sampled three contiguous stream sections. The length of each section (20–50 m) was set to encompass similar habitat characteristics with one pool (deep area with low current velocity), one riffle (shallow and rapid flowing zone) and one flat (medium depth and medium current velocity). For all the sites the vegetation cover was dense primary forest and hence the streams were shaded with only a few illuminated patches. The overall physical characteristics of each section were summarized by measuring stream width, water depth and current velocity along three transects. Turbidity was measured on each site with a WTW field turbidimeter. According to Vigouroux et al. (2005), turbidity was used as a proxy of gold mining intensity. On each stream the three

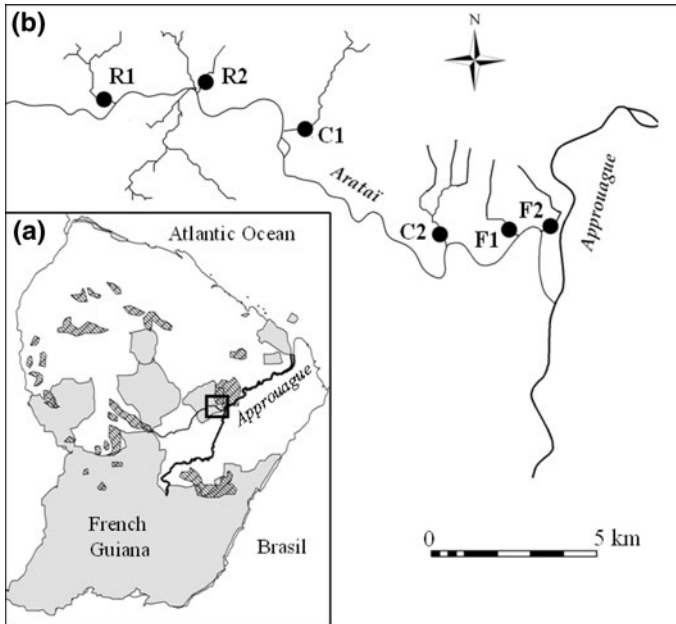


Fig. 1 Location of sampling sites. **a** Map of French Guiana indicating the location of the study area (black square). The biodiversity reserves (grey) and the gold mining areas (hatched) are also indicated. Note that some mining areas are located inside the biodiversity reserves. **b** Detailed map of the study area indicating the sampling sites: R1 and R2: reference sites; F1 and F2: formerly exploited sites; C1 and C2: currently exploited sites

Table 1 Geographical location, gold mining activity, turbidity and mean (\pm s.e.) physical characteristics of the six streams

Stream	Goldmining	Geographical location	Turbidity (NTU)	Depth (cm)	Width (m)	Current velocity (m s^{-1})
R1	No	22314007E–446293N	3.5	48.9 ± 6.5	4.03 ± 0.16	0.05 ± 0.02
R2	No	22316256E–446272N	3.6	36.1 ± 12.0	2.36 ± 0.03	0.09 ± 0.01
F1	Former	22325809E–441751N	5	10.6 ± 1.5	1.28 ± 0.10	0.13 ± 0.03
F2	Former	22326385E–442213N	4.3	21.7 ± 1.7	1.93 ± 0.13	0.10 ± 0.10
C1	Current	22319356E–445037N	41	22.7 ± 4.7	4.86 ± 0.48	0.41 ± 0.30
C2	Current	22323802E–441536N	201	20.3 ± 2.0	1.48 ± 0.07	0.19 ± 0.02

contiguous sections were separated each by four fine-meshed stop nets (mesh size 4 mm) to avoid fish escape from the sampling area and fish displacement between sections. Fish sampling was achieved using Rotenone. At least two subsequent doses of PREDATOX well mixed with water (6.6% emulsifiable solution of rotenone extracted from *Derris elliptica* by Saphyr, Antibes, France) were introduced immediately upstream from the sampling area. Fish were then collected with fine meshed dip nets (2-mm mesh). The sampling method has been described in detail by Mérioux et al. (1998), and has been proved efficient to collect quantitative samples of fish assemblages in small neotropical streams (Mérioux et al. 1998). Fish species were then identified according to Planquette et al. (1996) and Keith et al. (2000) and some specimens of each species were collected and

fixed in 5% formaldehyde solution for taxonomic confirmation. For each species, all the fish were counted and individually measured to the nearest millimetre. Fish sizes were then transformed into body mass using length-weight relationships provided by Fishbase (www.fishbase.org).

Three species traits were considered and informed by crosschecking the bibliography. Three classes of fish were determined based on their trophic status: fish predators corresponding to piscivorous fish, phytophagous species and omnivorous species. The body size was the maximal body size of the species as specified in Fishbase. Fish exhibited three major habitat preferences: small stream specialists (i.e. species inhabiting only headwater streams), large river specialists, and ubiquitous species known to inhabit both small streams and large rivers (Table 2).

Comparing compositional similarity among sites

We evaluated similarity in fish species assemblages for both taxonomic and functional composition. For taxonomic similarity, we evaluated similarity between all pairs of possible combinations using the Bray-Curtis distance. For functional similarity, the Euclidean distance was used.

To illustrate pairwise similarities among sites, we performed non-metric multidimensional scaling (NMDS). This iterative method, which is based on the rankings of distance between points, is of considerable theoretical interest since it circumvents the linearity assumption of metric ordination methods (Fasham 1977; Kenkel and Orlóci 1986). In NMDS, a stress parameter measures the degree to which between-site distances from the original data are preserved in lower-dimensional ordination model (Clarke 1993). We used the function ‘metaMDS’ of the package ‘VEGAN’ (Oksanen et al. 2008) in the R statistical software (R Core Team Development 2008), which finds a stable solution using several random starts.

We chose the number of dimensions equal to 2 to minimize the stress parameter (maximize the rank correlation between the calculated similarity distances and the plotted distances).

To assess the effects of environmental characteristics and gold mining intensity on variation in fish composition, we performed permutational MANOVA (McArdle and Anderson 2001), an analysis of variance using distance matrices performed by the function ‘adonis’ of the package ‘VEGAN’. This function partitions sums of squares using metric or semimetric distance matrices. The significance of the test was given by *F*-tests based on sequential sums of squares from 1000 permutations of the raw data. As the order of non-orthogonal variables can impact the outcomes of significance testing in such procedure, the overall physical descriptors of each section (depth, width, current velocity) were first introduced in the analysis, and the gold mining activity (represented by turbidity) was the last variable considered. That procedure allowed measuring the pure effect of gold mining after controlling for the natural environmental variability.

Results

A total of 1243 fishes of seventy species (i.e., 22 families from 6 orders) were caught in the six streams surveyed (Table 2). Characidae and Cichlidae were the most represented families and accounted for 30.0 and 11.4% of the total number of species, respectively. No significant difference was apparent in either fish species richness (mean = 15.4 ± 5.8)

Table 2 Occurrence and functional attributes of each species

	Sites						Functional attributes		
	R1	R2	F1	F2	C1	C2	Length (cm)	Trophy	Habitat
<i>Acestrorhynchus falcatus</i>					X		27.2	Predator	Streams & rivers
<i>Aequidens tetramerus</i>			X				16.0	Omnivorous	Streams & rivers
<i>Apistogramma gossei</i>				X			4.4	Omnivorous	Streams
<i>Apteronotus aff. albifrons</i>					X		50.0	Omnivorous	Streams & rivers
<i>Astyanax abramoides</i>		X			X		12.0	Omnivorous	Streams
<i>Astyanax keithi</i>	X				X	X	10.0	Omnivorous	Streams & rivers
<i>Astyanax validus</i>		X	X	X	X		15.0	Omnivorous	Streams
<i>Brachyhypopomus beebei</i>			X	X			35.0	Omnivorous	Streams & rivers
<i>Bryconamericus aff. stramineus</i>		X	X	X			11.4	Omnivorous	Streams
<i>Bryconops affinis</i>	X	X			X		12.0	Omnivorous	Streams & rivers
<i>Bryconops melanurus</i>	X			X	X		12.0	Omnivorous	Streams & rivers
<i>Callichthys callichthys</i>				X			16.5	Omnivorous	Rivers
<i>Characidium zebra</i>		X					6.5	Omnivorous	Streams
<i>Cleithracara maronii</i>			X	X	X		7.1	Omnivorous	Streams
<i>Copella carsevennensis</i>	X	X	X	X		X	3.6	Omnivorous	Streams
<i>Corydoras aeneus</i>			X	X			7.5	Omnivorous	Streams
<i>Corydoras geoffroy</i>	X						6.6	Omnivorous	Streams
<i>Creagrutus planquettei</i>					X		6.7	Omnivorous	Streams & rivers
<i>Crenicichla albopunctata</i>	X	X	X	X	X		14.0	Predator	Streams
<i>Crenicichla johana</i>	X	X					28.3	Predator	Streams
<i>Cyphocharax (juvenile)</i>	X				X	X	17.0	Omnivorous	Streams & rivers
<i>Dysichthys coracoideus</i>				X			11.0	Omnivorous	Streams
<i>Eigenmania virescens</i>	X				X	X	35.8	Omnivorous	Streams
<i>Electrophorus electricus</i>	X						250.0	Predator	Streams
<i>Erythrinus erythrinus</i>				X			20.0	Predator	Streams
<i>Gasteropelecus sternicla</i>					X	X	3.8	Omnivorous	Streams & rivers
<i>Guianacara geayi</i>	X				X		8.5	Omnivorous	Streams
<i>Gymnotus anguillaris/carapo</i>	X	X	X	X	X	X	30.0	Omnivorous	Streams
<i>Helogenes marmoratus</i>			X	X		X	7.3	Omnivorous	Streams
<i>Hemibrycon surinamensis</i>				X			7.1	Omnivorous	Streams
<i>Hemigrammus ocellifer</i>			X	X	X		4.4	Omnivorous	Streams & rivers
<i>Heptapterus bleekeri</i>				X			15.5	Omnivorous	Streams
<i>Hoplerythrinus unitaeniatus</i>			X	X			25.0	Predator	Streams
<i>Hoplias aimara</i>	X				X		100.0	Predator	Streams & rivers
<i>Hoplias malabaricus</i>				X			55.2	Predator	Streams & rivers
<i>Hyphessobrycon copelandi</i>	X						3.5	Omnivorous	Streams
<i>Hypopomus artedi</i>	X				X		50.0	Omnivorous	Streams
<i>Hypopygus lepturus</i>	X						10.0	Omnivorous	Streams
<i>Hypostomus gymnorhynchus</i>					X		17.0	Phytophagous	Rivers
<i>Ituglanis amazonicus</i>				X			7.5	Omnivorous	Streams
<i>Ituglanis nebulosus</i>				X			7.5	Omnivorous	Streams

Table 2 continued

	Sites						Functional attributes		
	R1	R2	F1	F2	C1	C2	Length (cm)	Trophy	Habitat
<i>Krobia aff. guianensis</i>	X	X	X	X	X	X	12.8	Omnivorous	Streams & rivers
<i>Leporinus gosseii</i>	X	X					25.0	Omnivorous	Streams & rivers
<i>Leporinus granti</i>	X				X		20.0	Omnivorous	Streams & rivers
<i>Loricaria gr. cataphracta</i>					X		29.5	Phytophagous	Streams & rivers
<i>Microcharacidium eleotrioides</i>		X					2.1	Omnivorous	Streams
<i>Moenkhausia aff. grandisquamis</i>	X	X		X	X	X	10.0	Omnivorous	Streams & rivers
<i>Moenkhausia aff. intermedia</i>	X				X		8.0	Omnivorous	Streams & rivers
<i>Moenkhausia collettii</i>	X						5.0	Omnivorous	Rivers
<i>Moenkhausia hemigrammoides</i>		X	X	X	X	X	4.0	Omnivorous	Streams & rivers
<i>Moenkhausia inraii</i>	X						6.5	Omnivorous	Rivers
<i>Moenkhausia oligolepis</i>	X	X	X	X	X		10.0	Omnivorous	Streams
<i>Moenkhausia surinamensis</i>	X			X			10.0	Omnivorous	Streams & rivers
<i>Nannostomus bifasciatus</i>				X			3.4	Omnivorous	Streams
<i>Parauchenipterus galeatus</i>		X					22.0	Omnivorous	Streams & rivers
<i>Phenacogaster nsp C</i>					X		3.7	Omnivorous	Streams
<i>Pimelodella cristata</i>	X	X	X	X	X		34.0	Omnivorous	Streams & rivers
<i>Poptella brevispina</i>	X				X		7.6	Omnivorous	Streams
<i>Potamotrygon hystrix</i>					X		40.0	Omnivorous	Rivers
<i>Pseudopimelodus ranius</i>			X			X	20.0	Predator	Streams & rivers
<i>Rhamdia quelen</i>			X	X			47.4	Omnivorous	Streams & rivers
<i>Rineloricaria aff. stewarti</i>	X						10.0	Phytophagous	Streams
<i>Rivulus gaeyi</i>		X	X	X	X	X	5.0	Omnivorous	Streams
<i>Rivulus lungi</i>		X			X		6.5	Omnivorous	Streams
<i>Rivulus xiphiidus</i>	X	X	X			X	3.8	Omnivorous	Streams
<i>Satanoperca aff. jurupari</i>	X						18.5	Omnivorous	Streams & rivers
<i>Sternopygus macrurus</i>	X				X		14.1	Omnivorous	Streams & rivers
<i>Tatia cf. intermedia</i>	X						12.0	Omnivorous	Streams
<i>Thayeria ifati</i>	X						3.5	Omnivorous	Streams

The length is the maximal standard length of the species reported in Fishbase. Trophic and habitat categories were derived from a literature survey (see [Methods](#)). R1 and R2: reference sites; F1 and F2: formerly exploited sites; C1 and C2: currently exploited sites

or fish biomass (mean = $16.0 \pm 15.7 \text{ g m}^{-2}$) among site status (Kruskal–Wallis test, $P > 0.05$, Fig. 2).

Fish compositional patterns

Although patterns of dissimilarity among fish assemblages were more pronounced for taxonomic than for functional composition, the NMDS clearly distinguished between site status in both cases (Fig. 3). For taxonomic and functional compositional patterns, a two-dimensional solution provided final stress values of 13.4 and 14.2, respectively, indicating

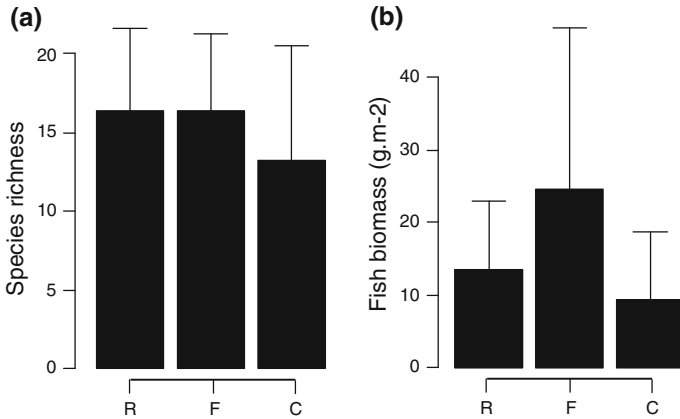


Fig. 2 **a** Mean (+sd) fish species richness and **b** biomass (g m⁻²) in reference (R), formerly exploited (F), and currently exploited (C) sites

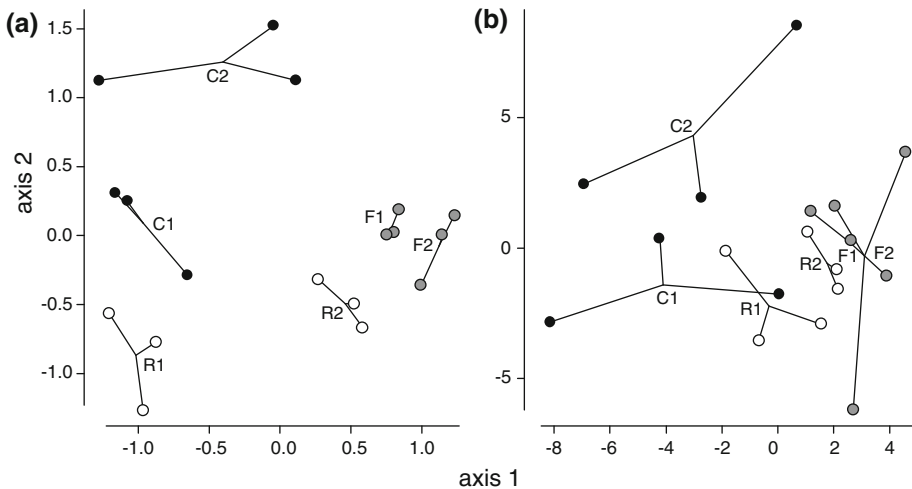


Fig. 3 Non-metric multidimensional scaling ordinations (NMDS) of **a** taxonomic and **b** functional composition of fish assemblages in reference (white dots), formerly exploited (grey dots), and currently exploited (dark dots) sites

a good preservation of the multidimensional between-site similarities in the reduced ordination space.

Differences in fish taxonomic composition between sites were significantly explained by stream width (permutational MANOVA, $P < 0.001$, Table 3a), but not by water depth or current velocity. For functional composition, the effect of stream width was significant ($P = 0.020$), while the effect of water depth and current velocity was not apparent (Table 3b). Finally, after accounting for the effects of stream habitat descriptors, gold mining still explained c.a. 13% of the differences in both taxonomic and functional composition between sites ($R^2 = 0.128$; $P = 0.004$ and $R^2 = 0.134$; $P = 0.009$, respectively; Table 3). Despite the limited number of sampling sites and the high fish abundance

Table 3 Permutational MANOVA assessing the effects of stream width, water depth, current velocity and gold mining intensity (i.e. water turbidity, see Methods) on variations in (a) taxonomic and (b) functional similarity between sites

	<i>MS</i>	<i>F</i>	<i>R</i> ²	<i>P</i>	
(a)					
Stream width	1.166	5.476	0.215	<0.001	***
Water depth	0.393	1.847	0.073	0.057	ns
Current velocity	0.394	1.851	0.073	0.069	ns
Gold mining	0.694	3.257	0.128	0.003	**
Residuals	0.213		0.511		
(b)					
Stream width	18.018	2.869	0.132	0.020	*
Water depth	4.825	0.768	0.036	0.576	ns
Current velocity	13.278	2.114	0.098	0.081	ns
Gold mining	18.220	2.901	0.134	0.009	**
Residuals	6.281		0.601		

The significance of the tests was checked using *F*-tests based on sequential sums of squares from 1,000 permutations of the raw data (ns non-significant, * *P* < 0.05, ** *P* < 0.01, *** *P* < 0.001)

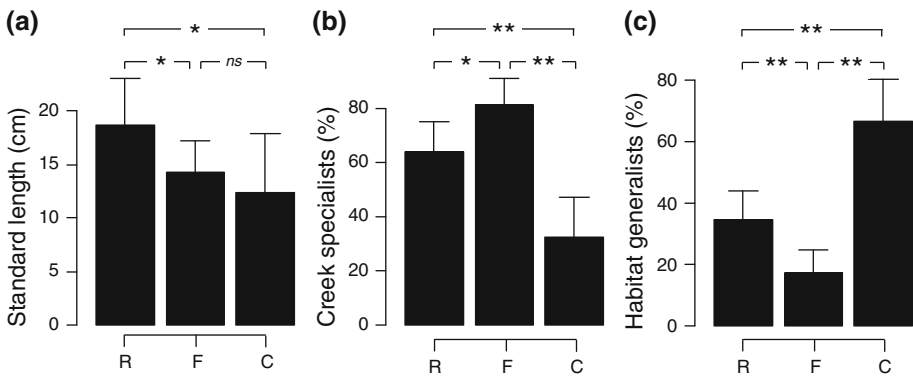


Fig. 4 **a** Mean (+sd) standard length (cm), **b** percentage of creek specialists and **c** percentage of habitat generalist species in reference (R), formerly exploited (F) and currently exploited (C) sites. Pairwise comparisons were done using Mann–Whitney tests (ns non-significant, **P* < 0.05, ***P* < 0.01)

and richness variability among sites, gold mining remained a significant determinant of fish assemblage structure.

Fish species traits among site status

No significant difference among site status was apparent in the percentage of each trophic group (Kruskal–Wallis test, *P* > 0.05 for piscivorous, phytophagous and omnivorous groups). Significant differences were observed for fish body size and habitat preference. Fish were significantly smaller in formerly and currently exploited sites (Fig. 4a). Fewer small stream specialists were observed in currently exploited sites (Fig. 4b), where fish assemblages were mainly dominated by ubiquitous fish species (Fig. 4c).

Discussion

Species richness as well as the biomass or abundance of organisms within an assemblage have been extensively used as rough biodiversity assessment metrics. Both have frequently

been combined to give rise to a wide range of diversity measures (Lawton et al. 1998; Gotelli and Colwell 2001). Under harsh disturbances, a given ecosystem is expected to reduce its richness, whereas softer pressures may not push species to extinction and are more likely to affect species biomass and/or abundances and hence modify community evenness (Begon et al. 1996; Heino et al. 2007). Gold mining, like other disturbances can affect both richness and abundance of fish in rivers. Such an impact has been reported for both streams (Mol and Ouboter 2004; Mendiola 2008) and large rivers (Vigouroux et al. 2005), but the results were often limited by the environmental variability between reference and impacted sites. Focussing on small forest streams permitted us, at least partly, to control that aspect of environmental variability. We hence showed that the basic descriptors of the community (i.e. richness and biomass) were not significantly affected by mining activities (Fig. 2), which parallels Yule et al. (2010) results in Indonesian rivers. This could be interpreted in two different ways. It may indicate that the gold mining did not affect fish communities or that these metrics are not suited to detecting an environmental impact. The lack of sensitivity of these general metrics has already been underlined (Diaz and Cabido 2001; Petchey and Gaston 2002), and that tendency is probably triggered for highly diversified communities where species replacements probably compensate for richness decline due to human disturbances. It would hence be more relevant to consider temporal changes in assemblages before commencement and after cessation of mining activities (Roset et al. 2007). This is however hindered by a lack of historical data on fish assemblages in these neotropical areas. A classical way to deal with this problem is to set up spatial comparisons between impacted and reference sites, a design well known as the 'reference condition approach' (Bailey et al. 1998; Hughes et al. 1998). However, regardless of the approach, our capacity to distinguish between natural and human-induced variation of biological conditions at both local and regional scales remains a crucial point (Hughes et al. 1998). With the development of modern statistical tools, such as NMDS and permutational MANOVA, the relative influence of environmental factors can now be quantitatively compared (Kenkel and Orłóci 1986; McArdle and Anderson 2001). Applying these techniques enabled us to discriminate between sites according to their level of mining disturbance, based on the taxonomic composition of their fish assemblages (Fig. 3a). The impact of gold mining was then confirmed by the permutational MANOVA, as gold mining remains significantly influential after accounting for environmental variability. Although the mining activities we considered here were small scale operations involving only a few workers without heavy equipment (the miners cross the forest on foot with all their equipment), the resulting impact was sufficient to exceed the resistance threshold of the stream. We also confirm that stream resilience was still not complete several months after stopping gold mining activities, as the formerly impacted sites still had not recovered entirely from the disturbance (Fig. 3a), and may reach an equilibrium state characterised by an assemblage structure slightly different from that found under the initial conditions. Our results are consistent with those of Yule et al. (2010) on an Indonesian river, where the extent of recovery 10 month after stopping the mining depended on the duration of the mining and on the severity of the perturbation.

These taxonomic changes may also reflect functional consequences for the streams (Hooper et al. 2005), which were confirmed by the functional NMDS analysis combining a set of biological attributes dealing with trophic level, body size and habitat selection (Fig. 3b). Again the permutational MANOVA identified the gold mining activity as a significant driver of the functional structure of the stream fish assemblage (Table 3b). Our prediction that the environmental disturbances arising from gold mining favour small sized species was hence verified (Fig. 4a). This might indicate that gold mining activity

increases mortality rate for long lived and hence large bodied species, thus favouring smaller fish having a shorter life span, which are favoured under disturbed conditions such as reduced stream productivity and increased toxicant concentration (McCoy and Gillooly 2008). We also verified our second prediction postulating that the mining activities favour habitat-generalist species at the expense of habitat specialists (Fig. 4b and c). Such a replacement of species is considered as an overall indicator of stream alteration (Karr 1981; Pont et al. 2006). This was particularly true for currently impacted sites, but the result was much more inconsistent for the formerly disturbed sites, which exhibited a higher proportion of habitat specialists than reference sites. This may indicate a transition phase between disturbance and total recovery state, but may also result from the environmental variability between sites. Indeed, although we paid particular attention in selecting similar sites, the two formerly exploited streams were slightly smaller than currently exploited sites (Table 1). We are nevertheless confident in our results concerning the overall impact of gold mining. Indeed, the multivariate comparisons of the sites by NMDS, based on functional and taxonomic attributes did not reveal any proximity of the sites according to their environmental descriptors (e.g. the sites C2, F1 and F2, which had similar water depth and channel width (Table 1), are not clustered together on Fig. 3). Finally, the trophic structure of the fish community was apparently not affected by gold mining activity; therefore rejecting our hypothesis that disturbance favours omnivorous species. It is indeed generally accepted that omnivorous species are able to adapt their trophic regime in response to an alteration of river food webs (Karr 1981). This lack of trophic response of the mining disturbance is probably due to the low trophic specialisation of the fish in the small streams, indeed, more than 97% of the fish found in each reference site were omnivorous. The trophic level, although commonly used as a functional metric in river quality assessment frameworks (Oberdorff et al. 2002; Pont et al. 2006), is probably not relevant for these small streams.

The overall impact of small scale gold mining on the taxonomic and functional composition of the fish assemblage is therefore contrasting with the use of simple measures such as taxonomic richness or overall biomass. These rough metrics are easy to implement, not expensive and do not require high taxonomic (morpho-species, or local names are often used as a surrogate for real species) and scientific expertise (Kerr et al. 2000; Abadie et al. 2008). They are hence frequently used in rapid biodiversity assessment programs (Kerr et al. 2000; Mendiola 2008; Silvano et al. 2009), but might not be of sufficient sensitivity to detect some disturbances, as shown here for fishes inhabiting small Neotropical forest streams. We hence encourage environmental managers to consider these rough indices with caution, and select more detailed metrics considering both species identity and environmental variability. The ability to disentangle environmental and human disturbance factors here revealed that small stream specialist species are the most strongly impacted by gold mining. This raises a particular conservation concern as replacing these species by more opportunistic ones, coming from the main stream channel, might reduce the spatial species turnover and might hence erode the longitudinal structure of the fish assemblage in the river basin.

Although the pervasive effects of gold mining have already been underlined (see introduction), here, we show that a single small scale gold mine can affect the structure and the function of the recipient stream. In French Guiana, a substantial number of these small gold mines have been established in areas protected for their high biodiversity (e.g. Guiana national park, Arataï natural reserve) (Fig. 1). The exact number of gold mines is unknown but satellite images and aerial photographs show a rapid increase of these sites, even inside the protected areas. Moreover, a recent report of the French National Forest Office reveals

that between 2000 and 2006, the areas exploited in French Guiana increased from 210 to 11,470 ha, i.e. more than a 50-fold increase in only 6 years (CIRAD-ONF 2006; Coppel et al. 2008). This situation is roughly similar for the entire Guiana Shield (Hammond et al. 2007). This means that we can fear that a substantial number of the headwater streams might be already impacted.

The limited duration and geographical extent of our study does not permit an extrapolation of our findings over large areas, and the obvious interconnectedness of the stream samples might affect our results due to possible fish movements from one site to another, hence violating the statistical independence between samples. This however reflects the spatial limitations imposed by sampling within a single catchment. In the same way, annual variation in river flow might also affect fish assemblage structure and the effect of gold mining might be lower during the rain season, so our results are only valuable for the dry season, when water levels are low. Despite these limitations, our results are sufficiently consistent to appeal for more comprehensive studies on the ecological impact of gold mining. Pending these future studies, we ask politicians and environmental managers to adopt precautionary conservation measures. The first measure we suggest to be taken being to ensure the respect of national laws that forbid gold mining settlements without an official permit. Legal gold mining activities should then be regulated after a precise assessment of their environmental impact.

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