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Impact of sediment releases on water chemistry and macroinvertebrate communities in clear water Andean streams (Bolivia)

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With 6 figures, 2 tables and 1 appendix

Abstract: The impact of sediment releases due to road construction on water chemistry and invertebrate communities was studied in a clear water river system in the tropical humid Bolivian Andes. Eight sites (2 reference, 1 source and 5 impacted along the main river) were sampled during the 1997 low flow season. Suspended sediment concentrations exhibited a 500-fold increase downstream from the source of pollution compared to the reference site, but recovered to natural levels within 90 km in the main river. Suspended solids had only a minor influence on other chemical parameters, but had a clear negative effect on invertebrate density (200-fold decrease in abundance) and diversity (6-fold decrease in number of taxa) in the main river. The most affected insects were epibenthic gatherers (e. g. Ephemeroptera: Leptohyphidae, Coleoptera: Elmidae), swimmers (Ephemeroptera: Leptophlebiidae), and scrapers (Coleoptera: Psephenidae, Trichoptera: Hydroptilidae). These families are therefore considered to be the best potential bio-indicators of sediment release impact in clear-water Andean rivers.

Key words: River, South America, Bolivia, suspended solids, water chemistry, macroinvertebrates, neotropical.

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Introduction

Sediment releases in rivers are caused by various types of human activity (e. g. agriculture, urban development, mining and road construction) and induce many modifications in physical habitats and aquatic communities downstream from the sediment source (e.g. ELLIS 1936, CORDONE & KELLY 1961, RIER & KING 1996, TRAYLER & DAVIS 1998). Sediment releases have been noted as one of the threats to imperiled freshwater fauna (RICHTER et al. 1997). However, their impact may vary depending on the natural characteristics of the affected river system.

The impact of road construction on invertebrate communities has been described in other continents (see FORMAN & ALEXANDER 1998, SPELLENBERG 1998) but never in a tropical South American system. The release of sediments near or within river channels increases both sediment deposition and SS (suspended sediments) concentrations. This can affect invertebrate communities directly through habitat modification, and indirectly through disturbance of primary production and through modifications in available trophic resources (HYNES 1960, DAVIES-COLLEY et al. 1992). This generally results in a reduction in invertebrate densities with or without modifications to community structure (CHUTTER 1969, GAMMON 1970, NUTTALL 1972, MCCLELLAND & BRUSVEN 1980, LENAT et al. 1981, DOEG & KOEHN 1994, HUBERT et al. 1996, WANTZEN 1998), or even in the total elimination of the flora and fauna (CORDONE & KELLY 1961). Our hypothesis is that these negative effects vary depending on the biological and ecological traits of the different taxa, mainly mobility and feeding strategies.

In the Bolivian Andes, some streams and rivers are turbid ail year round. They have extremely high concentrations of SS during the wet season (GUYOT 1993) and impoverished invertebrate communities (WASSON et al. 1998). However, during periods of low flow, the streams and rivers of the humid mountain range known as "Yungas" have naturally clear water. Some watersheds are also affected by deforestation and poor agricultural practices, resulting in turbid waters during the wet season. Tropical Andean freshwater invertebrate communities are still poorly understood (see ROCABADO & WASSON 1999, ROCABADO et al., in press). Attention has been drawn to the effect of substrate disturbance on the annual dynamics of invertebrate communities (FLECKER & FEIFAREK 1994, WASSON et al. 1998), but the impact of anthropic sediment releases in clear water rivers has only been briefly described (SALINAS et al. 1999) despite the tact that it is one of the major problems affecting streams and rivers in the whole Yungas region.

The Rio Coroico, which flows through the Yungas about 80 km to the north of the city of La Paz, was recently affected by huge sediment releases due to the construction of a major road, which resulted in permanently turbid water.

The Rio Coroico, which is a typical Yungas river, thus provided an appropriate site to study the problem rince the sediment release occurred at a single point source, and there was a regular decrease in SS concentration downstream without any other major anthropogenic impact. The aims of this study were 1) to present evidence of the impact of sediment release on water chemistry and on the invertebrate community in a clear water river, 2) to link these impacts to ecological traits of the affected fauna, and 3) to identify taxa that could be used as bio-indicators of SS impacts in clear water rivers.

Study sites

The Bolivian Amazonian basin is part of the high Rio Madeira basin and covers 65 % of the Bolivian territory (724,000km^2, Fig. 1). The Andean part of this basin has highly varied relief and climate. The Yungas region, facing the Amazon basin on the eastern slope of the cordillera between 500 and 3,500 m above sea level, lies in the tropical humid mountain range. The relief is very accentuated with V-shaped valleys incised through consolidated Ordovician rocks. In the Rio Coroico basin, the main valleys are partially deforested for pasture and agriculture (coca, coffee, citrus fruit, banana), leading to erosion problems in some locations, mainly in the coca fields. The mainstem (Rio Coroico) is consequently turbid during the wet season (from November to April)

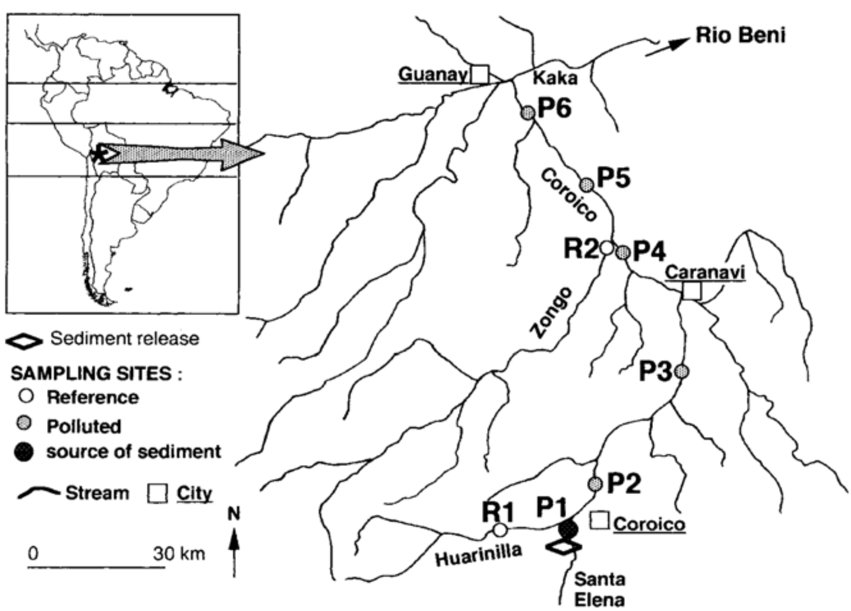

Fig. 1. The High Kaka River basin, with sampling sites and main source of sediment pollution indicated.

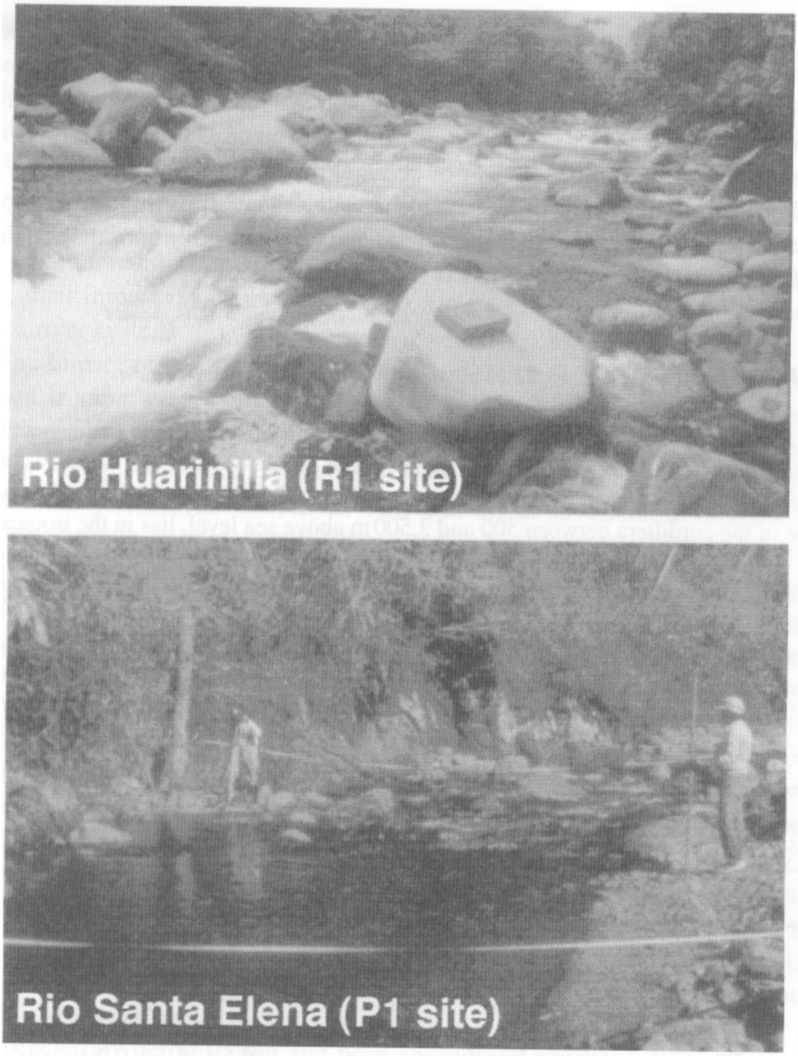

Fig. 2.

due to sediment inputs from nome tributaries, but is normally clear during the dry season.

During the years 1995-1998, the Rio Santa Elena, a small tributary of the mainstem near Coroico, served as spillway for the sediments eroded as the result of the construction of a main road. The effect on the turbidity of the Rio Coroico during the dry season was detectable up to its confluence with the Rio Kaka, 90 km downstream. Eight study sites were selected to study the impact (Fig. 1 and 2): two unimpacted reference or control sites (Rio Huarinilla and Rio Zongo, Rl and R2), one site in the stream that received the sediment input, close to its confluence with the Rio Huarinilla

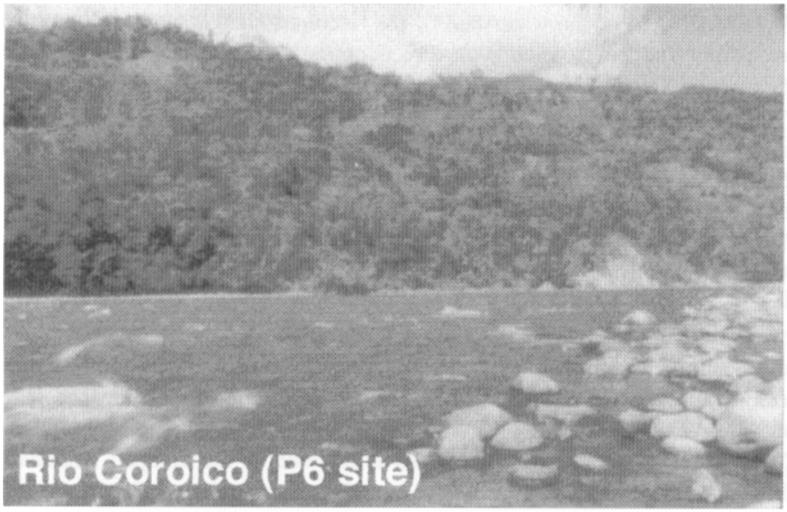

Fig.2. Photographs of three sampling sites: a references site (Rio Huarinilla, RI), the source site (Rio Santa Elena, Pl) and a dowstream site (P6). Photographs: J. G. WASSOSON.

Table 1. Data table of the physico-chemical parameters at the sampling sites (temperature, nitrates and chlorides excluded).

	pH	Cond. (µS/cm)	S.S. (mg/l)	HCO_3^- (mg/l)	SO_4^{--} (mg/l)	PO_4^{3-} (µgP/l)	Na^+ (mg/l)	K^+ (mg/l)	Ca^{++} (mg/l)	Mg^{++} (mg/l)	Elevation (m a.s.l.)	Slope (m/km)
R1	7.14	22	2	3.1	4.7	5	2.2	0.4	1.9	0.5	1260	33.90
R2	7.12	29	4	0	12.0	5	2.4	0.3	1.6	1.0	520	22.22
P6	6.63	48	5	3.1	16.0	5	3.5	0.5	2.6	2.8	420	2.50
P5	7.08	42	33	3.1	14.0	5	2.7	0.5	2.6	2.8	480	1.82
P4	5.16	48	25	6.1	16.0	5	2.9	0.5	3.4	2.8	520	6.00
P3	6.53	34	106	6.1	13.0	20	2.8	0.4	1.8	2.1	625	5.56
P2	6.90	63	267	9.2	13.0	10	2.8	0.5	2.0	2.4	960	6.37
P1	6.83	20	2574	4.3	5.0	5	1.4	0.9	0.8	0.8	1060	35.09
Mean	6.67	38	377	4.4	11.7	8	2.6	0.5	2.1	1.9	731	14.18
S.D.	0.65	14	892	2.8	4.5	5	0.6	0.2	0.8	0.98	316	14.05

(Rio Santa Elena, Pl), and five sites (P2 to P6) downstream from the sediment source, in the Rio Coroico mainstem.

The watershed area of the Rio Coroico ranges between 691 km^2 at Santa Barbara (site P2) and 5,840 km^2 at the confluence with the Rio Kaka (downstream P6). The mean interannual discharge near site P5 is 243 m^3/s for a specific discharge of about 521/s . km (Santa Rita de Buenos Aires, period 1974-83, GUYOT 1993). Between Coroico and Caranavi, the river flows in a steep V-shaped valley, through a succession of gorges interspersed with some broadened stretches where sampling sites P2 and P3 were located. At Caranavi, the valley turns left following the western edge of the first sub-Andean range and widens conspicuously. The river slope decreases from about 0.6% (sites P2 to P4) to about 0.2% downstream (Table 1).

The surface area of the watershed of the reference sites is similar to those of the impacted sites. The Rio Coroico is the direct continuation of the Rio Huarinilla and the watershed at the R1 site (378 km^2) is slightly more than half the size of the watershed at the P2 site. The watersheds of the Rio Zongo at the R2 site and of the Rio Coroico at Caranavi are almost the came size (1573 and 1675 km^2, respectively) and have similar geo-climatic features (see WASSON & BARRÈRE 1999). However, the river slope at both reference sites (3.4% and 2.2 %, respectively for R1 and R2) is greater than in the Rio Coroico. The median diameter (D50) of the superficial riverbed sediment decreases regularly along the Huarinilla—Coroico mainstem, varying from 187mm at the RI site, to 144—84 mm between P2 and P3, and to 72—63 mm downstream (P4 and P5 sites) (GUYOT et al. 1999). The corresponding bankfull width of the Rio Huarinilla (RI) river is about 25 m; of the Rio Coroico upstream from Caranavi (P2—P3), 45 m; and varies between 60 and 200 m downstream (sites P4—P6) depending on the local morphology of the valley. The watersheds of both reference sites are much less anthropized than the Coroico valley. The RI site was located at the outlet of the Cotapata National Park, and there is no main road in the Zongo valley. However the Zongo river is used for hydroelectric production in its upper reaches, although the effect on discharge regime was barely perceptible at the R2 site. Before the road construction, the Rio Santa Elena was a nice yungean stream, with a bankfull width of about 10 m. The bottom of the valley is now filled with raw sediment and the channel is braided. The stream slope is within the range of the reference sites.

Methods

Each site was sampled once during the dry season, either in July (R2 and Pl to P6) or in September, 1997 (R1). Samples were taken only under base flow conditions for two reasons: first, quantitative invertebrate sampling is almost impossible in this kind of river during high flow due to hydraulic conditions, and second, the purpose of the study was to evidence the impact of increasing SS concentration during low flow, when the natural condition of the water is clear.

To obtain abiotic parameters, water samples were taken in a fast-flowing current vein as close as possible to the center of the river. During the months preceding the sampling period, road construction activity was almost continuous, so that SS concentrations remained relatively stable in the mainstem. Field measurements of conductivity, pH, and temperature were made using a WTW device. Laboratory measurements were made of SS (100ml filtration, W-42 filter), bicarbonates (HCI titration), chlorides ($AgNO_3$ titration, 0.01 M), sulfates (spectrophotometry), nitrates and phosphates (colorimetry), calcium, magnesium, sodium and potassium (atomic absorption). Temperature (variable during daytime), nitrates (undetectable), and chlorides (close to detection level) were omitted from data treatment. The data matrix was completed with the measurement of valley slope and elevation at the sampling sites (from 1/50,000 topographic maps), resulting in a 12 parameters x8 sampling sites matrix (Table 1). Invertebrate sampling comprised six Surber samples (0.1 m^2, mesh size 0.25 mm) distributed to represent physical habitat area and heterogeneity of a reach comprising two riffle/

pool sequences. Invertebrates were sorted after washing through a 0.63 mm sieve in the laboratory, identified to order or family level, and counted. For data treatments, rare taxa (only one specimen encountered in all the samples collected in all 8 sites) were excluded, data were then transformed using a log (n + 1) resulting in a 8 sites x39 taxa matrix (Appendix: data before log (n + 1) transformation and mean densities per m^2).

Regressions and correlations were computed using Systat® v.5.2. Correlations were tested using the Spearman rank test. Multifactorial analysis and graphs were constructed using the ADE-4 package (THIOULOUSE et al. 1997). Habitat data were treated through a correlation matrix Principal Component Analysis (nPCA, 8 sites x 12 parameters) and invertebrates through a covariance matrix Principal Component Analysis (nPCA, 8 sites x39 taxa).

Results

Water chemistry

Concentrations of suspended solids were less than 4 mg/1 at the reference sites (R1 and R2, Table 1), but increased to 2,574 mg/1 at the source site (Rio Santa Elena, Pl), before gradually declining from 267 to 5 mg/1 downstream (Rio Coroico, P2 to P6). With the exception of SS, the chemical composition of the water at the eight study sites was relatively similar, with low ionic content, except for a slight increase in phosphates and bicarbonates immediately downstream from the sediment source (Pl and P2). None of the other chemical parameters exhibited a distinct pattern of variation along the Rio Coroico. This resulted in low conductivities (20–48 µS/cm) even at impacted sites.

Only the first two nPCA axes are of interest (Fig. 3 a). The first axis represents 45 % of the total inertia of the analysis and demonstrates the importance of slope, as oppose to conductivity and sulfates (Fig. 3b). This axis describes the differences between the source and reference sites (Pl, RI and R2 and the right side of the Cl axis, Fig. 3 c) and the Rio Coroico mainstem (P2 to P6). The second axis represents 11 % of total inertia and distinguishes the sites with low levels of potassium and bicarbonates (R1 and R2) from sites with high levels (mainly Pl, Fig. 3c). The SS are thus only of secondary significance for the ordination, and are linked to differences in potassium and bicarbonates.

Invertebrates

A total of 39 non-rare invertebrate taxa were identified, generally at the family level (Appendix). The greatest taxonomie diversity was recorded at the reference sites (RI and R2), with 33 and 31 taxa (Fig. 4 a), whereas only 5 taxa were recorded at the sediment source (Pl). In the Rio Coroico mainstem, the most impacted site had only 13 taxa, but the number increased rapidly downstream, with 20 to 25 taxa recorded at sites P3 to P6. Invertebrates were al most absent from the source site (P1, 7 ind./0.6 m^2) but

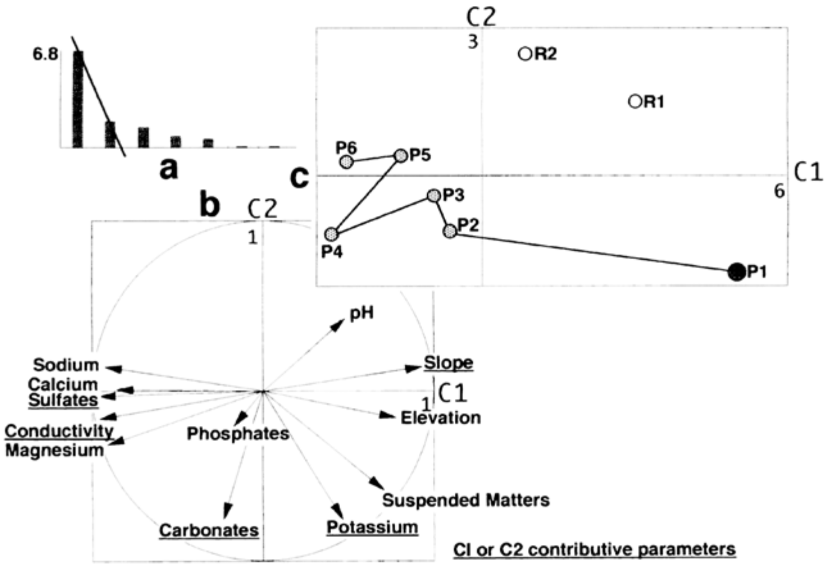

Fig.3. nPCA of the physico-chemical analysis of the water. (a) Eigenvalues graph. **(b)** C1xC2 correlation circle of the 12 parameters. The parameters underlined contribute most to these axes. (c) ClxC2 map of the sampling sites.

most absent from the source site (P1, 7 ind./0.6 m^2) but density increased rapidly downstream (Fig. 4 a). Density at the site closest to the sediment source was only 0.4 % of the average density at the reference sites, but increased to 78 % at the impacted site furthest downstream (P6).

Invertebrate communities were dominated by Diptera (61 % of the individuels), Trichoptera (15 %), Ephemeroptera (11%), and Coleoptera (7%). Dipteran densities decreased in sites with SS concentrations of over 100 mg/1 (Fig. 4 b), while densities of Coleoptera, Trichoptera, and Ephemeroptera appeared even more closely related to SS concentrations (Fig. 4 c). Trichoptera were however, abundant at the P3 site impacted by SS. Less abundant invertebrate orders, such as Plecoptera, Heteroptera and Odonata, did not appear to react so clearly to SS concentrations (Fig. 4d).

Significant correlations between invertebrate order densities and SS (log/ log, Spearman test) were found for Coleoptera ($p = 0.01$), Ephemeroptera and Hemiptera ($p = 0.02$), Plecoptera ($p = 0.03$), and Diptera ($p = 0.04$), but not for Trichoptera ($p = 0.06$) or any other order. At the family level, the densities of 13 families or subfamilies show a significant negative relationship with SS concentrations (Table 2). However, the level of significance is not very high for Baetidae, Ceratopogonidae and Chironomidae. A log/log linear regression

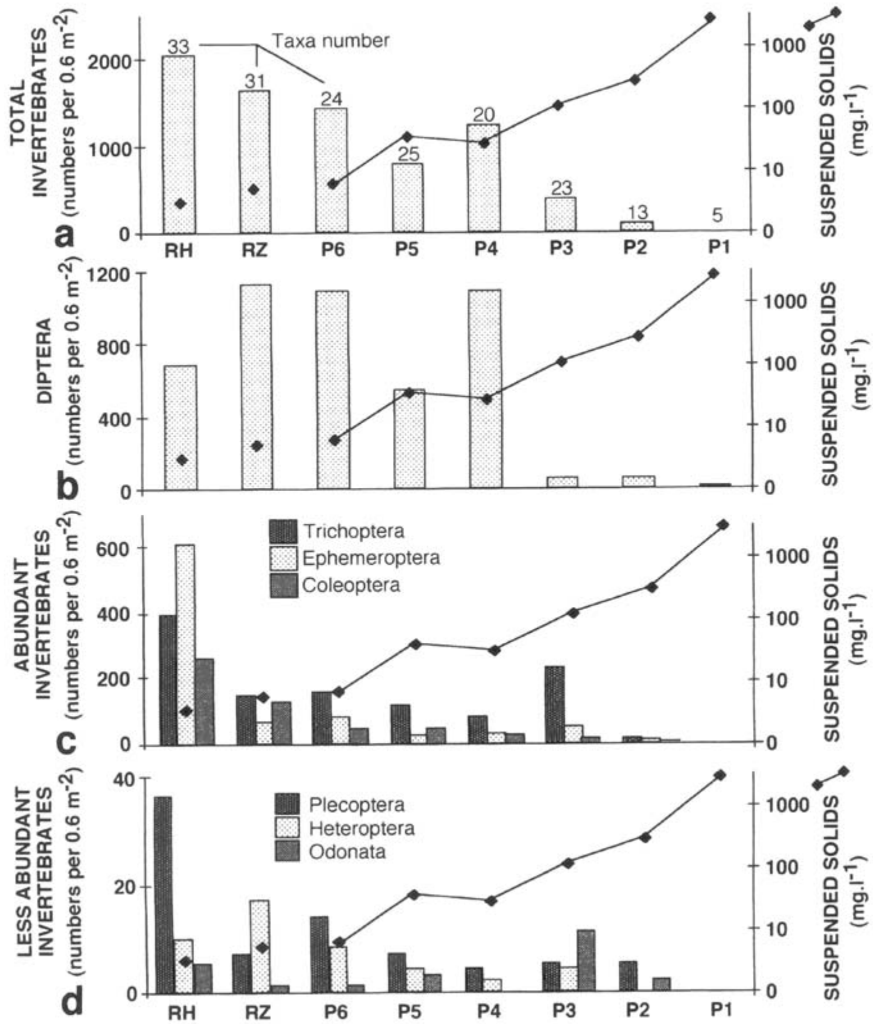

Fig. 4. Invertebrate numbers in relation to suspended solids loads. (a) Total invertebrate numbers per 0.6 m², and numbers of taxa. **(b)** Diptera. (c) Trichoptera, Ephemeroptera, and Coleoptera. **(d)** Plecoptera, Heteroptera, and Odonata.

produced the same results. For Hydropsychidae, the linear regression is not significant, but a second order polynomial regression is significant, illustrating an "arch effect" relationship between density and SS concentration.

Only the first two axes of the cPCA are of interest (Fig. 5 a). The first axis represents 60 % of the inertia of the analysis and clearly separates the reference site and less impacted sites on the right from the source site and more impacted sites on the left (Fig. 5 b). The faunistic ordination of sites Pl to P6 in

Table 2. Expected and observed effects of sediment addition on the insects of the High Kaka River basin. Families present in only 1 or 2 samples omitted. Ecology primilarily from HURLBERT et aI. (1981) and MERRITT & CUMMINS (1996). Significance levels: *p<0.05, — NS.

	Ecology		Regression (log/log)	Spearman Test	Suspended Solids	
	Mobility	Feeding			expected effect[1]	observed effect[2]
Collembola	skater	gatherer	–	5.2–	?	NC
Baetidae	swimmer	gatherer	5.03–0.61 S.S.	4.3*	–	–
Leptohyphidae	clinger	gatherer	5.52–0.70 S.S.	1.0*	–	–
Leptophlebiidae	swimmer	gatherer	4.99–0.71 S.S.	1.0*	–	–
Perlidae	clinger	predator	3.89–0.43 S.S.	3.5*	–	–
Hydroptilidae	clinger	scraper/gatherer	4.85–0.73 S.S.	2.1*	–	–
Leptoceridae	variable	variable	–	36.6–	?	NC
Hydropsychidae	clinger	filterer	arch effect	52.9–	–	NC
Glossosomatidae	clinger	scraper	–	9.8–	–	–
Elmidae	clinger	gatherer	6.14–0.77 S.S.	1.5*	–	–
Psephenidae	clinger	scraper	4.01–0.60 S.S.	0.1*	–	–
Naucoridae	clinger/swimmer	predator	–	28.4–	–	NC
Gerridae	skater	predator	2.02–0.36 S.S.	2.1*	–	–
Tipulidae	burrower	gatherer	–	20.2–	–	–
Ceratopogonidae	sprawler	predator	2.71–0.31 S.S.	4.9*	–	–
Simuliidae	clinger	filterer	2.08–0.34 S.S.	3.6*	–	–
Psychodidae	burrower	gatherer	–	13.8–	–	NC
Empididae	sprawler/burrower	predator	3.32–0.45 S.S.	3.0*	–	–
Chironominae	burrower	gatherer	–	5.9–	–	–
Orthocladiinae	burrower/clinger	gatherer	7.11–0.71 S.S.	4.4*	–	–
Tanypodinae	sprawler	predator	4.13–0.57 S.S.	4.6*	–	–
Libellulidae	sprawler	predator	–	56.4–	–	NC
Corydalidae	clinger	predator	–	13.0–	–	NC
Pyralidae	climber	shredder	–	9.8–	–	–

(1) From ecology: – = negative effect, ? = unknown effect.
(2) From the PCAn: – = decrease in density, NC = non contributive invertebrates.

relation to the reference sites is thus opposed to that produced by physico-chemical analysis. Since the ordination of the sites on the first axis is closely linked to their SS concentration, this axis results in a classification of the taxa according to their sensitivity to SS effects. This ordination distinguishes a pole of "impacted sites" on the left and a pole of "clear water sites" on the right. The second axis, which is less significant mathematically (11 % of the inertia), opposes upstream sites (top of the graph, mainly R1) to downstream ones (bottom of the graph).

At the clear-water sites (Fig. 5 d), the most significant taxa were Leptohyphidae, Leptophlebiidae, and Elmidae, Oligochaeta, Nematoda, and Naucoridae downstream, Trichoptera (Philopotamidae, Polycentropodidae) and Coleoptera (Dryopidae, Staphylinidae) at the RI site.

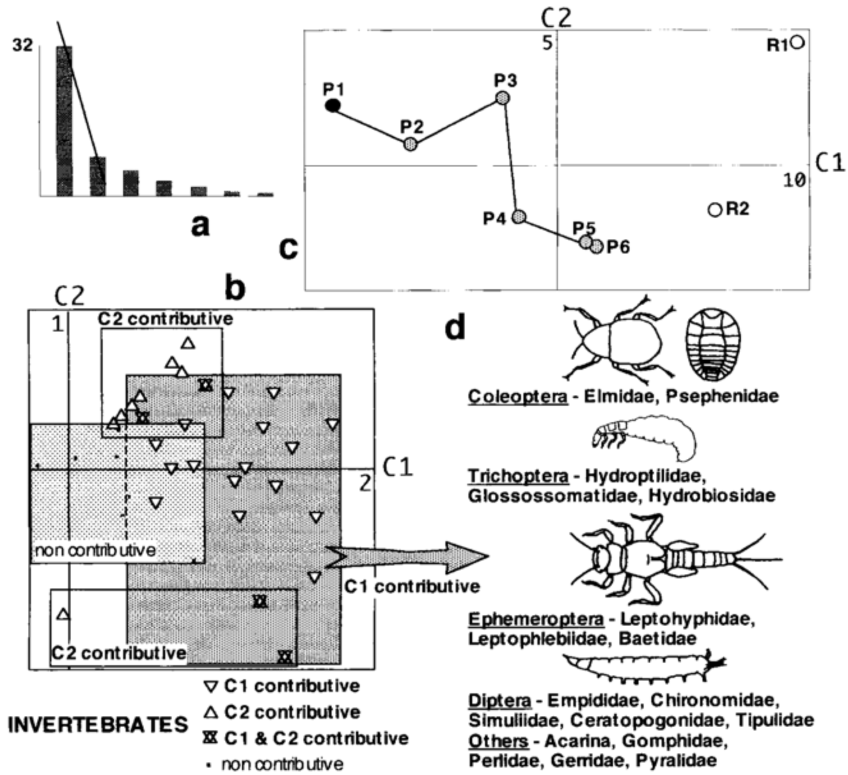

Fig. 5. cPCA of the invertebrate table (log(invertebrates/0.6 m^2) + 1). **(a)** Eigenvalues graph. **(b)** C1xC2 map of the sampling sites. (c) ClxC2 map of the taxa, with Cl contributing taxa indicated. **(d)** Taxa contributing to the clear water pole.

Discussion

Impacts on water chemistry

Although SS concentrations exhibited a 500-fold increase at the source site as compared with the reference sites, no significant change related to the SS effect was detected in the other chemical parameters. Similarly, in Canada, no change in water chemistry was observed following highway construction although SS showed a 300-fold increase and sediment deposition a 10-fold increase (BARTON 1977). The only limited variation in the water chemistry observed may be linked to an upstream-downstream gradient (Fig. 3 b). Thus, the observed impacts on invertebrate fauna are probably related to the direct and indirect effects of SS, and not to other changes in the chemical properties of the water.

Impacts on invertebrate fauna

When comparing physico-chemical and faunistic structures, the most salient result is the opposite ordination of the impacted sites in relation to the reference sites on the factorial maps. The first analysis evidenced the morphological and (limited) chemical differences between the reference and source sites on the one hand, and the Rio Coroico mainstem on the other. Conversely, in the second analysis, the source and most impacted sites were located opposite the reference and less impacted sites. Thus, the faunistic structure observed was not primarily caused by morphological characteristics related to an upstream–downstream gradient, but was clearly related to the SS concentration. The second axis (less important) of the faunistic analysis discriminated the sites upstream and downstream of the town of Caranavi; this could be related either to a "regional" effect (associated with geo-morphological and climatic changes, see WASSON & BARRÈRE 1999), or to the increase in stream size. Nevertheless, this effect is far less important than the impact of SS on the faunistic structure.

Suspended solids had a clear negative effect on the density of invertebrate fauna as a whole (Fig. 4 a), and especially on some orders such as Ephemeroptera and Coleoptera (Fig. 4 c). The reduction in diversity was more apparent at SS concentrations of over 100 mg/1. No positive effect of SS was demonstrated on any invertebrate population. These results are consistent with previous studies. A reduction in benthic faunal densities following SS release in streams has been observed by various authors (CORDONE & KELLY 1961, CHUTTER 1969, GAMMON 1970). Accumulation of fine sediment has had negative effects on hyporheic (RICHARDS & BACON 1994) and benthic fauna, inducing a reduction in benthic densities (NUTTALL 1972, MCCLELLAND & BRUSVEN 1980, LENAT et al. 1981, DOEG & KOEHN 1994, HUBERT et al. 1996, WANTZEN 1998). Sediment release may also induce an increase in drifting invertebrates (DOEG & MILLEDGE 1991). Thus, invertebrate density is an important indicator for the evaluation of sediment release impacts.

Mechanisms of impact

The mechanisms of the negative effects observed appear to be complex, involving physical effects (habitat modifications) and trophic effects (disturbance of trophic resources and feeding mechanisms) (Fig. 6). Sediment releases induced siltation within and on the surface of the substrate. An increase in SS reduced transparency and thus primary production (HYNES 1960, DAVIES-COELEY et al. 1992), but also directly affected some biological processes. Expected impacts on macroinvertebrate communities were derived from knowledge of the ecology of each family (HURLBERT et al. 1981, complemented when neces-

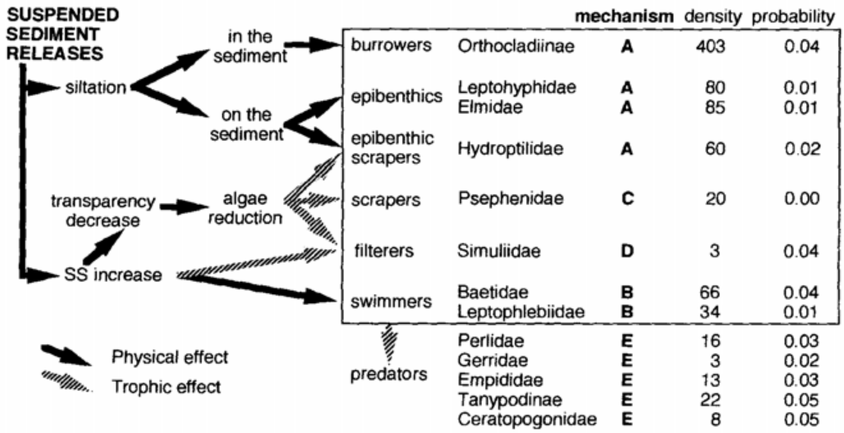

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Fig. 6. Hypothesized effects of sediment addition, and observed affected invertebrates. Mechanisms: **A** – Siltation on the surface and within the sediment. **B** – Perturbation for the swimmers. **C** – Clogging of filtering devices. **D** – Decrease in water transparency. **E** – Decrease in prey densities.

cary with data from North America, MERRITT & CUMMINS 1996, and field observations).

Various direct or indirect negative effects of sediment releases on invertebrate populations were demonstrated (Table 2, Fig. 4 and 6):

A Siltation on the surface and within the substrate reduced habitat suitability and interstitial spaces with a resulting negative effect on burrowers and epibenthic invertebrates, particularly gatherers such as Leptohyphidae, Elmidae, Hydroptilidae, and Orthocladiinae.

B The increase in SS was a source of disturbance for swimmer invertebrates such as most Ephemeroptera. This effect was clear in the whole order as well as in Leptophlebiidae and, to a lesser extent, in Baetidae.

C The clogging of filtration devices through high SS loads was responsible for low densities of Hydropsychidae (using filtering nets) and Simuliidae (using filtering fans). A negative linear effect was significant for the latter, but not for the former. Hydropsychidae were less abundant at reference sites than at sites with moderate SS concentrations (P3 to P6), and there was a marked decline at the most impacted sites (P1, P2). The first effect could be due to a trophic limitation in very clear waters where their food resource (suspended organic matter) was lacking, while the latter was related to the clogging of their nets.

D An increase in SS concentrations reduced water transparency, inducing a decline in epibenthic algal populations. Gatherers and most significantly scrapers (Psephenidae and Hydroptilidae) were affected by this type of modification.

E – reduction in primary consumers (scrapers) and in secondary consumers (gatherers) affected predators like Plecoptera (Perlidae), Heteroptera (Gerridae) and some Diptera (Ceratopogonidae, Empididae, Tanypodinae).

Some families were collected only in reference sites (Philopotamidae, Hydrobiosidae, Polycentropodidae, ?Rhyacophilidae, Trichoptera undet., Dryopidae, Staphylinidae, and Gomphidae). They were perhaps the most sensitive to SS, but their absence in the other sites might be due to other factors related to changes in stream morphology and habitat. Faunistic analysis demonstrated the negative effects of suspended solids on numerous families (clear water pole taxa in Fig. 5 c). These effects were generally confirmed through a nonparametrical statistical test (Table 2). Although associated with the clear-water sites, Glossossomatidae, Tipulidae, and Pyralidae did not demonstrate any significant correlations with SS, primarily because they were scarce and randomly distributed at the impacted sites. The non-significant correlation between SS and Chironominae is linked to their abundance at site P4. This site had the lowest pH and the highest concentration of sulfates and bicarbonates and the abundance of Chironomidae there might be due to a local effect, such as slight organic pollution generated by the town of Caranavi.

Suitability for bio-indication

The best potential bio-indicators of SS impacts should fulfill three conditions: 1) be well represented at both reference and impacted sites, 2) present a significant linear relationship between density and SS concentrations, and 3) the mechanisms of population reduction must be coherent with existing ecological knowledge. At this stage of our work, the densities of Leptophlebiidae, Leptohyphidae (Ephemeroptera), Psephenidae, Elmidae (Coleoptera), and Hydroptilidae (Trichoptera) have been identified as the best candidate metrics to be included in a bio-indication method to evaluate the impacts of sediment release in clear water rivers in the Yungas region. Hydropsychidae cannot be used as indicators due to their non-linear relationship with SS concentration.

Obviously, this method still requires refinement. The effects of sediment addition may vary through the year, as has been observed in a Canadian river (ROSENBERG & WIENS 1978). Our samples were taken during a stable low flow period, when SS concentrations were naturally low and thus the impact was greatest. During high flow periods, most Andean rivers present high SS concentrations. as can easily be observed from the color of the water. However, even in these periods, anthropogenic releases of sediments may have indirect effects, like greater abrasion of the substrate due to increased sediment transport.

In this study, identification was carried out at the family level, which is consistent with the objective of bio-indication. However, insect ecology and physiology do differ within a family or within a genus. As an example, sand deposition in an English river eliminated *Baetis pumilus* but favored *Baetis rhodani* (Ephemeroptera, Baetidae, NUTTALL 1972). Greater taxonomie resolution would help to identify the most sensitive taxa.

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Appendix. Data table of the 39 invertebrate taxa, in numbers/0.6 m^2, and mean densities in numbers/m^2.

	R1	R2	P6	P5	P4	P3	P2	P1	Density
Oligochaeta	0	6	9	2	0	0	0	0	3.5
Nematoda	0	7	24	10	4	1	7	1	11.2
Microcrustacea	0	10	0	1	0	0	0	0	2.3
Hydracarina	17	95	1	19	1	6	2	0	29.4
Collembola	1	7	0	0	0	9	0	0	3.5
Baetidae	233	15	14	7	8	37	1	0	65.6
Leptohyphidae	288	32	35	8	9	5	5	0	79.6
Caenidae	0	1	0	0	0	2	0	0	0.6
Leptophlebiidae	92	17	33	5	13	4	0	0	34.2
Perlidae	36	7	14	7	4	5	5	0	16.2
Hydroptilidae	223	45	5	12	1	2	0	0	60.0
Leptoceridae	24	20	1	0	0	35	3	0	17.3
Hydropsychidae	46	76	148	106	79	193	12	1	137.7
Philopotamidae	23	0	0	0	0	0	0	0	4.8
Helicopsychidae	29	0	1	0	0	0	0	0	6.2
Glossosomatidae	6	2	1	0	1	0	0	0	2.1
Hydrobiosidae	22	4	0	0	0	0	0	0	5.4
Polycentropodidae	3	0	0	0	0	0	0	0	0.6
?Rhyacophilidae	8	0	0	0	0	0	0	0	1.7
undet. Trichoptera	8	0	0	0	0	0	0	0	1.7
Elmidae	188	110	37	46	17	10	2	0	85.4
Psephenidae	64	19	5	1	4	2	0	0	19.8
Dryopidae	4	0	0	0	0	0	0	0	0.8
Staphylinidae	6	0	0	0	0	0	0	0	1.2
Naucoridae	0	13	7	4	2	4	0	0	6.2
Gerridae	10	4	1	0	0	0	0	0	3.1
Tipulidae	16	25	0	1	1	6	2	0	10.6
Ceratopogonidae	12	10	2	10	1	2	1	1	8.1
Simuliidae	3	3	6	1	0	0	0	0	2.7
Psychodidae	2	2	0	1	2	2	0	0	1.9
Empididae	23	29	2	7	1	2	0	0	13.3
Chironominae	298	455	191	261	1051	1	34	2	477.7
Orthocladiinae	291	506	863	223	8	26	16	2	403.1
Tanypodinae	21	29	15	23	3	14	0	0	21.9
Chironomidae-P.	18	67	12	20	25	0	0	0	29.6
Libellulidae	0	0	1	3	0	11	2	0	3.5
Gomphidae	5	1	0	0	0	0	0	0	1.2
Corydalidae	1	4	0	3	0	0	0	0	1.7
Pyralidae	26	12	0	1	0	1	0	0	8.3
Total	2047	1633	1428	782	1235	380	92	7	1584.2