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Assessing effects of change in land use on size-related variables of fish in subtropical streams

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Manuscripts

1 **Assessing effects of change in land use on size-related variables of fish**
2 **in subtropical streams**

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Abstract

Transformation of the natural land cover is one of the most important global changes. Changes in land use may strongly affect ecosystem functions and biodiversity by directly or indirectly modifying key structural properties. Here, we examined the effects of land use on the size structure of fish communities in subtropical Uruguayan streams. We analysed the suitability of non-taxonomic, size-related variables as a tool to elucidate shifts in stream fish assemblages along a gradient in land use change. We also tested some taxonomic variables (e.g. species richness and species diversity) to compare their response with size-related variables. In the more anthropogenic disturbed basins (i.e. dominance of urban and agricultural land use), we found higher size diversity, slope of size spectrum (flatter slopes) and total range of fish length, whereas higher species richness, fish abundance and species diversity were detected in more pristine stream basins (i.e. dominance of rangelands). In combination with traditional taxonomic variables, size-related variables of fish communities in streams may be effective tools in evaluating the changes occurring in freshwater ecosystems associated with anthropogenic changes in land use.

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Key words: size structure, body size, size diversity, land cover, rangelands

55 **Introduction**

56 A key driver of global change is the transformation of the natural land cover, with half
57 of the terrestrial surface of the Earth already being transformed by human activities
58 (MEA 2005; Prieto-Benítez and Méndez 2011). Land use change has intensified and
59 accelerated with the urban development of the so-called “emerging economies”, like
60 Brazil, India and China (McGrabham and Martine 2014), as well as the
61 implementation of modern intensive farming practices involving heavy application of
62 fertilizers and high densities of livestock (Tilman et al. 2002; Foley et al. 2005).
63 Landscape conversion from forest or prairie to pasture, agriculture, or urban settlements
64 may strongly affect ecosystem functions and biodiversity (Allan 2004), among other
65 processes through a series of biotic and abiotic alterations (Blevins et al. 2013).
66 Therefore, land use data can be used as a holistic and inclusive approach to summarise
67 the complex and more causative environmental alterations occurring as a whole (Allan
68 2004; Feld et al. 2013).

69
70 Previous studies have analysed the effect of land use on the presence/absence or
71 abundance of key species and functional diversity. Species richness often decreases with
72 land use modification, as seen for mammals (Bhagabati et al. 2014), birds (Herremans
73 1998), herpetofauna (Hartel et al. 2010) and fish (Blevins et al. 2013). Moreover, Flynn
74 et al. (2009) demonstrated changes in functional diversity in bird and mammal
75 communities at varying levels of agricultural land use intensity, and Pool et al. (2010)
76 detected that land use in the catchments explained the variation in fish functional
77 composition. However, little attention has so far been paid to the effects of land use on
78 community or population size structure.

79

Population and community size structures are considered important ecosystem health indicators (Petchey and Belgrano 2010; Emmrich et al. 2014) and may therefore be used to assess the effects of land use changes on natural ecosystems. Nevertheless, only few studies have analysed the impact of land use on size-related variables. In European lakes, Brucet et al. (2013) revealed that human land use (% agricultural land) had a negative effect on fish size, whereas it positively influenced fish species richness and diversity. Similarly, but using eutrophication as a factor (and not land use), negative effects on fish body size were detected from high indirect effects of nutrient concentrations in the water (Jeppesen et al. 2000; Emmrich et al. 2011). In rivers, some studies have shown changes in arithmetic mean length (Murphy et al. 2013; Benejam et al. 2014) or size class structure (Aparicio et al. 2011; Logez and Pont 2011) related to anthropogenic perturbations. However, more complex size-related variables, like size spectrum [log abundance - log body size regression] (Kerr and Dickie 2001) or the size diversity index (Brucet et al. 2006), have not been used to study land use effects, or any specific anthropogenic impact, on running waters. The only previous study on such systems where size spectrum has been applied was in the large St. Lawrence River (New York State, USA) where Murry and Farrel (2014) found that food-web capacity (i.e. y-intercept of size spectra) decreased over time concurrently with a declining concentration of total phosphorus.

The aim of our study was to examine relationships between land use and community size structure. More specifically, we aimed to unravel the following objectives i) analysis of size-related variables (e.g. size spectra, size diversity) in stream ecosystems ii) assessment of the effects of land use on size-related variables of stream fish and comparison of the results with the effects in other ecosystems (i.e. lakes, large rivers).

Moreover, we used some classic taxonomic variables (e.g. species richness, abundance of some tolerant and intolerant species) to contrast and complement their response with size-related variables. We worked on Uruguayan streams that, unlike the better-known temperate streams, are rich in fish species (Teixeira-de Mello et al. 2012, 2014) and overall dominated by small-sized fish occurring in high abundances (Teixeira-de Mello et al. 2012, 2014). In accordance with studies of size-related variables of freshwater fish in other aquatic ecosystems (lakes and large rivers) (Jeppesen et al. 2000; Brucet et al. 2013), we tested the hypothesis that at more degraded sampling sites (with higher degree of agricultural and urban land use in the basins), fish community structure would be characterised by low values of size-related variables (e.g. size diversity, mean, range and maximum length), species richness and fish abundance.

2. Methods

2.1 *Study area and fish sampling*

Uruguay is the second smallest country in South America (area: 176,215 km², inhabitants: 3,2 million), it is located in the southern subtropical region and has a mild climate (from a winter mean of 12 °C to a summer mean of 25 °C and an average precipitation of 1,200 mm year⁻¹) (Agudo 1990) (Fig. 1). The country is relatively homogeneous in terms of rainfall, temperature and altitude. Most of the natural grasslands were transformed to grazing land after the introduction of cattle ca. 450 years ago, and nowadays pasture occupies 77% of Uruguay's surface, whereas agriculture occupies 8.7% (MGAP 2012). Extensive husbandry of cattle and sheep (with only 1.4 heads per hectare, hereafter referred to as "rangelands") is the most natural condition currently found across the country. Agriculture (e.g. crop of soybean, cereals, winery and deciduous fruits) (hereafter "agricultural") has undergone an intensification process

130 with use of high quantities of fertilizers and herbicides. Less important geographically,
131 but of great importance in terms of impacts on water quality, is the urban surface
132 (hereafter “urban”). Besides industrial wastewaters, many households are still not
133 connected to urban sewage systems or only have septic tanks and, consequently,
134 produce a notable input of nutrients to surface and underground water. Therefore, the
135 agricultural and urban land uses imply an increase of phosphorus and nitrogen
136 concentrations in running waters (Table 1).

137

138 Wadeable streams are adequate to test our hypothesis because, as all other inland fresh
139 waters, they are directly or indirectly influenced by the land use in the drainage basin
140 (Allan 2004), and they hold comparative advantages since a representative sample of
141 the size distribution of fish can be easily collected with active fish gear, such as
142 electrofishing (Teixeira-de Mello et al. 2014). Moreover, the Uruguayan streams are
143 particularly appropriate to test our hypothesis due to: i) a high fish species richness (ca.
144 220 spp.) with different life histories (Teixeira-de Mello et al. 2011) and ii) a wide land
145 use modification gradient, especially regarding rangelands, agriculture and urbanisation.

146

147 The 26 wadeable stream reaches included in our study were sampled in the summers
148 between 2005 and 2011 (see Table 1 for details). They were widely distributed in the
149 country and covered an important gradient of land use (Fig. 1, Table 1). The area of
150 sub-basins upstream of sampling sites and the different kinds of land use (i.e.
151 rangelands, agricultural or urban) were determined with ArcGIS 10.2 (ESRI Inc.,
152 Redlands, CA, USA) (Table 1). The sampling sites were 50 m long, and on average ca.
153 5 m wide and 36 cm deep (Table 1).

154

155 Fishes were sampled by electrofishing (100–600 V, 2–3 A, pulse DC), without block
156 nets, using two different electrofishing methods: point or multiple pass, and the same
157 team leader undertook the sampling at all sites to ensure standardisation. Point
158 electrofishing (P) was performed by applying one electric pulse per metre, i.e. in total
159 50 electric pulses along each reach. With the multiple-pass (M-P) technique, we applied
160 electricity in the entire area on two occasions (two electrofishing passes). Teixeira-de
161 Mello et al. (2014) compared the results of these electrofishing techniques in similar
162 streams and found that they yield the same size distribution and relative species
163 composition, but show different richness and fish abundance. In their study, reliable
164 transfer equations were presented for richness and abundance to transform catch data of
165 P into M-P estimations. We used these equations to obtain comparable richness and
166 abundance data between P and M-P, and both variables (species richness and fish
167 abundance) were calculated for each sampling site (Table 1). Size data was used directly
168 without any transformation following Teixeira-de Mello et al. (2014). All sampled
169 fishes were identified to species level, counted and measured (standard length in mm).
170 For each sampling site, wetted width, depth and water total phosphorus and total
171 nitrogen concentrations were also measured (Table 1).

172

173 *2.2 Calculations of size-related variables and statistical analyses*

174 In total, five size-related variables, and other metrics like species richness and fish
175 abundance, were calculated for each sampling site for the entire fish community (Table
176 2, Figs. 2 and 3, Appendix A and B). We calculated: (1) the arithmetic mean [L_{mean}
177 (mm)] and (2) total range [TR (mm)] of fish length. The (3) maximum length [L_{max}
178 (mm)] was selected from the largest individual caught at each sampling site. (4) Size
179 diversity [SD] was also calculated for each sampling site following the nonparametric

approach of Quintana et al. (2008), which is related to the Shannon diversity index but adapted for continuous variables (herein fish standard length). Size diversity integrates the amplitude of the size range and the evenness in the same way as the Shannon species diversity integrates the number of species and their relative abundance (Brucet et al. 2006; Quintana et al. 2008). The size diversity of different community organisms has been found to reflect environmental and human disturbance gradients (Brucet et al. 2006, 2010; Emmrich et al. 2011; García-Comas et al. 2013). The advantage of using size diversity is that it does not require statistical fitting (in contrast to the biomass size spectrum) and that it is represented by a single value (Brucet et al. 2006). High values of size diversity were obtained in cases of equally distributed fish lengths and when the catch consisted of many different fish lengths, the opposite being found for low values of size diversity (Fig. 2). Finally, for each sampling site we obtained (5) the slope [b] of the linear size spectrum. Analyses of size spectrum [log abundance - log body size regression] have been applied to suspended particles, plankton and fish in marine ecosystems (Sheldon et al. 1972; Blanchard et al. 2005), wetlands and lakes (Sprules and Munawar 1986; Brucet et al. 2005; Gamble et al. 2006; Emmrich et al. 2011) and occasionally in terrestrial ecosystems to soil invertebrates and vegetation patterns (Mulder and Elser 2009; Petchy and Belgrano 2010; Borthagaray et al. 2012). We calculated the linear size spectrum according to Emmrich et al. (2011). Numbers of fish were accumulated into $\log_2$ size classes, thus obtaining 5 size classes. The smallest size classes were extremely rare, for this reason we accumulated in the first class: 1st class < 32 mm; 2nd class: $2^5 = 32$ mm to $2^6 = 64$ mm; 3rd class: $2^6 = 64$ to $2^7 = 128$ mm; 4th class: $2^7 = 128$ mm to $2^8 = 256$ mm; 5th class: $2^8 = 256$ mm to $2^9 = 512$ mm. The linear size spectrum was calculated as linear least-square regressions between $\log_2$ numbers accumulated per size class and $\log_2$ midpoint of size classes (Kerr and Dickie, 2001).

205 Intermediate empty size classes were replaced by zeroes to calculate the linear size
206 spectrum (Emmrich et al. 2014). The slope of the linear size spectrum serves as a rough
207 estimation of food-web trophic or ecological efficiency (i.e. rate of decreased fish
208 abundance with increasing body size). Steeper slopes from the linear size spectrum
209 reflect relatively higher abundance of small organisms, whereas flatter slopes (less
210 negative) indicate a high proportion of large fish (Emmrich et al. 2011). We additionally
211 estimated species diversity according to Shannon-Wiener (Pielou 1969) and richness as
212 the total number of species in a stream (Table 2).

213

214 We conducted a Principal Component Analysis (PCA) including the different kinds of
215 land use, chemical and mesohabitat features and area of sub-basin for each sampling
216 site. Kaiser–Meyer–Olkin’s measure of sampling adequacy (KMO) was used to assess
217 the usefulness of a PCA. KMO ranges from 0 to 1 and should be >0.5 if variables are
218 sufficiently interdependent for PCA to be useful. Then, we used the first axis of PCA as
219 a proxy for variation in land use (hereafter referred as “gradient of land use”). To test
220 the effects of the gradient of land use on each of the size-related and taxonomic
221 variables, we used generalized linear models (GLMs), which provide more appropriate
222 error structures for data such as counts and proportions. For fish abundance and number
223 of species, we used GLMs with Poisson errors and log-link functions. For fish lengths
224 and ranges, and variables of the linear size spectrum, we used gamma errors and identity
225 link functions. For diversity indices we used normal errors and identity link functions.
226 We also analysed, using GLMs with Poisson errors and log-link functions, the effects of
227 the gradient land use on the abundance of the three most common fish species with
228 contrasting tolerance to water and ecosystem quality: *Bryconamericus iheringii* (tetra)
229 and *Cheirodon interruptus* (tetra), both classified as intolerant species (Teixeira-de

230 Mello 2007), and *Cnesterodon decemmaculatus* (ten spotted live-bearer), tolerant to
231 poor water and ecosystem conditions (Teixeira-de Mello 2007).

232

233 As described in more detail by Teixeira-de Mello et al. (2014), we used established
234 equations between two different electrofishing methods to obtain comparable richness
235 and abundance. However, in order to control for potential effects of fishing method, we
236 used sampling method as a categorical covariate in GLMs. We ran three models for
237 each size-related variable with the following independent variables: 1) gradient of land
238 use; 2) gradient of land use and sampling method; and 3) gradient of land use, sampling
239 method and their interaction. We selected the best model comparing the AICc. As some
240 sampling sites were spatially segregated (from the centre of the country to the sea, Fig.
241 1), we applied the Durbin Watson statistic using the residuals of GLMs to test for
242 spatial autocorrelation, which, if present, indicates non-independence of residuals and
243 the potential for inflated statistical error type 1. Durbin-Watson test ranges from 0 to 4
244 and should be around 2 if variables are sufficiently independent. All statistical analyses
245 were performed with SPSS 15 (SPSS Inc., Chicago, IL, USA, 1989–2006).

246

247 **3. Results**

248 Most of the variables (land use, chemical and mesohabitat features and area of sub-basin
249 for each sampling site) were significantly correlated (Appendix A) and KMO (= 0.53)
250 indicated the usefulness of a PCA, which explained 45.5% of the variation on axis one
251 and 26.7% on axis two (Fig. 4). The first axis reflected a gradient of land use, being
252 positively related to rangelands and negatively to urban land and concentrations of
253 phosphorus and nitrogen. Thus, in the subsequent statistical analyses, the first axis of
254 PCA was used as an indicator of the gradient of land use.

255

256 In total, 9.800 fish were caught by electrofishing in the 26 Uruguayan stream reaches,
257 corresponding to 51 species, mainly belonging to Characiformes and Siluriformes
258 orders. Differences in range and coefficient of variation (CV) were substantial between
259 the size-related variables tested (Table 2). The highest variability occurred in species
260 richness and fish abundance. In contrast, size diversity and slope of spectrum exhibited
261 low variability. Other variables like species diversity, arithmetic mean length, total
262 range length and maximum length presented intermediate values with an around 42% of
263 coefficient of variation. Several size-related variables were correlated (Table 3). For
264 example, size diversity was positively correlated with mean length and slope but
265 negatively with fish abundance. Species richness was positively correlated with fish
266 abundance but negatively with slope. Total range, maximum length and slope were all
267 positively correlated with each other.

268

269 We found significant effects of the gradient of land use on some size-related variables
270 (Table 4, Fig. 5, Appendix B). Generalized linear models showed an increase of species
271 richness, fish abundance and species diversity along the gradient of land use, with
272 maximum values in well preserved areas where rangelands was the predominant land
273 use (positive values of first axis of PCA). In the more degraded areas, where urban land
274 use was dominant (negative values of first axis of PCA), we found a significantly higher
275 fish size diversity, slope (flatter slopes) and total range of fish length. The values
276 obtained in the Durbin-Watson test for the residuals of GLMs were around 2 (Tables 5
277 and 6), which means that residuals are independent.

278

Some significant changes in fish abundance at species level were found. The three most common species showed changes in abundances along the transformation gradient of land use (Table 5). The abundances of two intolerant tetra species, *Bryconamericus iheringii* and *Cheirodon interruptus* were higher in sampling sites dominated by rangelands land use. In contrast, the tolerant species *Cnesterodon decemmaculatus*, increased in abundance in streams with higher presence of urban and agricultural land use in their basins.

4. Discussion

Our study showed a gradient in land use modification in Uruguay from rangelands, corresponding to the most natural environmental conditions, to urban land use, that was highly related to concentrations of phosphorus and nitrogen. The changes in fish community size structure along this environmental gradient were reflected in some size- and taxonomic-related variables. In the more anthropogenic disturbed basins (i.e. dominance of urban and agricultural land use), we found higher size diversity, slope of size spectrum (flatter slopes) and total range of fish length. By contrast, we found an increase of species richness, fish abundance and species diversity in basins where rangelands was the most abundant land use. The abundance of the three most common species changed along the gradient in land use, with an increase of the two intolerant species in basins dominated by rangelands and higher presence of the tolerant species in altered basins.

Although many studies on streams and lakes have shown changes in the arithmetic mean of fish length as a response to anthropogenic perturbations, we did not detect a significant change in mean length caused by the gradient of land use. Our hypothesis

304 was that in more degraded basins we would detect low values of mean length, this being
305 based on the findings of Jeppesen et al. (2000) and Brucet et al. (2013), who reported a
306 decrease of mean body size of fish assemblages in lakes with increasing phosphorus
307 concentrations. However, the pattern for stream fish communities is not so clear.
308 Although, in general, decreased average sizes of fish in streams have been reported as a
309 response to anthropogenic perturbations (Walters and Post 2008; Benejam et al. 2014),
310 some studies have revealed an increase in the average size of populations under
311 increasingly disturbed conditions (Murphy et al. 2013). The authors of the latter study
312 suggested that the increase in size could reflect failure to recruit under disturbed
313 conditions, this being based on the finding that the increase in length appeared to be
314 related to a trend of decreasing abundance. Therefore, in streams, the different fish size
315 classes could be affected by several and diverse human factors. For example, substrate
316 alteration and siltation may affect the survival of fry or preclude reproduction, reducing
317 abundance and removing young age classes (Logez and Pont 2011; Aparicio et al.
318 2011); while water abstraction, pollution and exploitation may increase adult mortality
319 (Matthews and Marsh-Matthews 2003; Benejam et al. 2014).

320

321 We found that fish abundance increased with the proportion of rangelands, indicating
322 that these areas were more productive for fish than the areas dominated by urban or
323 agricultural land use. In streams in the state of Maryland (USA), Utz et al. (2010) also
324 found that reductions in fish population abundances were stronger in areas dominated
325 by urban land use. With a different approach, using nutrient data on the same area
326 covering 29 years, Murry and Farrell (2014) showed that, in a large river ecosystem, the
327 y-intercept of size spectra (i.e. fish abundance) decreased concurrently with declining
328 total phosphorus levels. This difference compared to our results may be explained by

the fact that they worked with a large river instead of streams (indicating different pattern of responses depending on ecosystem type) and/or that they focused on nutrients, while we used land use data that is a more comprehensive variable.

332

Contrary to our hypothesis, we found significantly higher size diversity, slope (flatter slopes) and total range of fish length in the more degraded basins. Although maximum length did not vary significantly, the tendency showed an increase in altered basins. Previous studies on lakes (Jeppesen et al. 2000; Emmrich et al. 2011; Brucet et al. 2013) have demonstrated loss of large-bodied fishes with eutrophication and, as a consequence, we expected a lower total range of fish length, size diversity and slope of size spectrum in altered basins. The higher ranges of length in more altered basins were due to the presence of large tolerant species, such as catfish *Hypostomus commersoni* and marbled swamp eel *Synbranchus marmoratus*, being, respectively, facultative and obligated air-breathers (Teixeira-de Mello et al. 2011). With these large tolerant species, the slope of the size spectrum was flatter and size diversity was higher than in more natural basins where large tolerant species were absent. Thus, several size-related variables reflected the changes in community size structure of stream fish due to degradation of habitat quality, and the combination with taxonomic information may be useful to understand these changes.

348

In more altered basins, where urban land use was dominant, we detected lower species richness, lower species diversity and lower abundance of intolerant species, but higher abundance of tolerant species. Our results are consistent with those of previous studies. Streams draining agricultural land areas typically support fewer intolerant or sensitive fish species than streams draining forested catchments (e.g. Lenat and Crawford 1994;

354 Wang et al. 1997; Allan 2004; Feld 2013); in the same way, urban streams typically
355 support lower densities of intolerant species (e.g. Brewer and Rabeni 2011; Chalar et al.
356 2013). However, such effects of land use on fish species composition have not been
357 detected in species-poor regions, for example in Patagonian northwest streams
358 (Miserendino et al. 2011) and New Zealand (Clapcott et al. 2012). Similarly, in areas
359 with relatively low richness of freshwater fishes (e.g. Mediterranean basins, headwaters,
360 cold waters), it has been difficult to develop enough taxonomic metrics to formulate a
361 reliable fish Index of Biotic Integrity (IBI) with sufficient metrics for a broad spectrum
362 of anthropogenic impacts (Miller et al. 1988; Moyle and Marchetti 1999; Benejam et al.
363 2008). In such cases, a non-taxonomic approach (focusing on size-related variables)
364 such as the one presented here may help to develop complementary metrics at
365 population or community level for the purpose of ecological assessment. Some projects
366 with this aim have been conducted in Mediterranean (Aparicio et al. 2011; Murphy et al.
367 2013) and cold water streams (Logez and Pont 2011; Benejam et al. 2014). On the other
368 extreme, in remote areas where species richness is high (with some species likely not
369 yet discovered or classified) and where the guilds of fishes are still unknown (e.g. sub-
370 tropical and tropical areas), preventing the development of classic metrics for IBI, a
371 non-taxonomic approach with size-related variables seems very useful.

372

373 The present study is the first to assess complex size-related variables (i.e. size spectra,
374 size diversity) of stream fish communities. Moreover, this work is the first attempt to
375 examine relationships between land cover and the community size structure of stream
376 fish. We provide evidence that size-related variables of fish communities, in
377 combination with traditional taxonomic variables, may be effective tools to evaluate the
378 impacts of such land use changes on freshwater ecosystems. However, more research is

needed to unravel the variation of size-related variables on stream fish, our results suggesting that patterns may be different than from those of other freshwater ecosystems such as lakes or large rivers. The need to combine approaches and build complementary indicators is crucial considering the acceleration of transformations at the landscape level worldwide.

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Table 1. Location, limnological features and additional information of sampling sites, including the hectares of different land uses in each river basin upstream of the sampling site. Sampling methods: point electrofishing (P) or multiple-pass (M-P).

Site name	Site code	Basin	Coordinates	Sampled year	Method	Width (m)	Depth (m)	Total P ($\mu\text{g L}^{-1}$)	Total N ($\mu\text{g L}^{-1}$)	Species richness	Fish abundance	Area basin (hectares)
A. Sopas	1	Uruguay River	31°33'15"S, 56°24'52"V	2007	M-P	10.6	0.33	54	215	19	1599	42.0
Tacuarembó chico	2	Negro River	31°48'09"S, 55°12'56"V	2007	M-P	6.8	0.41	32	233	17	509	98.7
C. Honda	3	Uruguay River	31°32'55"S, 56°24'04"V	2007	M-P	12.5	0.38	41	215	15	284	35.6
California	4	Negro River	31°35'51"S, 56°16'48"V	2007	M-P	8.7	0.70	38	233	14	382	55.7
C. del Paso	5	Negro River	31°38'37"S, 56°17'52"V	2007	M-P	4.4	0.52	16	184	17	685	15.0
Carpintería	6	Negro River	31°48'09"S, 55°12'56"V	2007	M-P	8.4	0.23	84	605	21	247	185.3
A. del Sauce	7	Negro River	31°55'56"S, 55°49'52"V	2007	M-P	7.6	0.37	101	460	36	1319	137.3
Monzón	8	Negro River	33°15'43"S, 55°20'16"V	2008	P	3.3	0.49			51	558	96.2
Marincho	9	Negro River	33°15'28"S, 57°06'35"V	2008	P	3.3	0.35			33	357	234.8
Molles del pescado	10	Negro River	33°21'35"S, 55°25'13"V	2008	P	6.3	0.40			51	320	156.0
Mansavillagra	11	Negro River	33°32'48"S, 55°43'30"V	2008	P	8.7	0.34			46	215	1060.7
Puntas pantanoso	12	Santa Lucía River	33°49'30"S, 56°16'55"V	2012	M-P	2.5	0.22	1650	1186	15	195	8.4
Chal chal 1	13	Santa Lucía River	33°54'13"S, 56°00'23"V	2012	M-P	3.1	0.26	1058	2372	24	749	18.8
Chal chal 2	14	Santa Lucía River	33°54'13"S, 56°00'23"V	2011	M-P	2.1	0.15	1058	2372	21	548	18.8
Conchillas 2	15	Santa Lucía River	34°41'16"S, 56°17'27"V	2005	P	1.9	0.36	477	4769	19	202	15.7
Colorado 3	16	Santa Lucía River	34°40'28"S, 56°16'38"V	2005	P	5.3	0.52	925	3119	17	212	41.1
Cañada de Colorado	17	Santa Lucía River	34°40'44"S, 56°14'46"V	2005	P	2.5	0.29	783	1179	9	72	6.3
Dragón 2	18	Santa Lucía River	34°42'33"S, 56°18'35"V	2005	P	3.0	0.40	514	1501	44	329	13.4
Colorado 2	19	Santa Lucía River	34°41'32"S, 56°14'16"V	2005	P	6.2	0.20	1029	9795	14	91	19.4
Colorado 1	20	Santa Lucía River	34°41'43"S, 56°11'59"V	2005	P	1.9	0.32	310	1159	33	226	4.0
Las Piedras 1	21	Santa Lucía River	34°43'19"S, 56°11'03"V	2005	P	1.8	0.42	453	1253	23	2518	2.4
Juncal	22	Santa Lucía River	34°43'16"S, 56°18'59"V	2005	P	3.0	0.23	645	795	35	430	8.6
Las Piedras 3	23	Santa Lucía River	34°44'18"S, 56°16'34"V	2005	P	6.1	0.27	853	6383	17	179	37.8
Conchillas 1	24	Santa Lucía River	34°43'42"S, 56°14'59"V	2005	P	2.3	0.44	894	4001	17	1320	3.6
Las Piedras 2	25	Santa Lucía River	34°44'32"S, 56°12'39"V	2005	P	4.3	0.28	913	3143	17	141	12.8
Dragon 1	26	Santa Lucía River	34°46'30"S, 56°16'07"V	2005	P	2.3	0.47	118	833	7	23	1.6

Table 2. Characteristics and variability of five size-related variables from fish assemblages sampled in 26 Uruguayan streams including range, mean and coefficient of variation (CV). For a detailed description of variables, see Methods.

Variables	Code	Range		Mean	CV (%)
		Lower	Upper		
Fish abundance		23.00	2518.00	527.31	108.27
Species richness		7.00	36.00	24.00	52.07
Species diversity (Shannon)		0.29	2.87	1.74	39.11
Size-related variables					
Arithmetic mean length (mm)	Lmean	16.70	84.70	36.00	36.65
Total range (mm)	TR	71.00	458.00	227.00	49.92
Maximum length (mm)	Lmax	80.00	468.00	238.00	47.20
Size diversity	SD	1.60	3.36	2.35	19.05
Slope spectra	b	-2.53	-0.31	-1.83	27.14

Table 3. Correlation matrices of the five size-related variables. The white matrix contains Spearman's r values and the grey matrix the corresponding P -values. For the full names of the size-related variables, see Table 2.

Variables	Fish abundance	Species richness	Species diversity	Lmean	TR	Lmax	Size diversity	Slope b
Fish abundance	-	0.452	0.212	-0.291	-0.051	-0.059	-0.538	-0.825
Species richness	0.020	-	0.572	0.074	-0.372	-0.370	-0.205	-0.505
Species diversity	0.298	0.002	-	0.353	-0.171	-0.167	0.003	-0.335
Lmean	0.149	0.721	0.077	-	0.109	0.117	0.507	0.272
TR	0.805	0.061	0.404	0.595	-	0.999	0.232	0.433
Lmax	0.773	0.063	0.416	0.569	0.000	-	0.236	0.439
Size diversity	0.005	0.316	0.988	0.008	0.254	0.246	-	0.508
Slope b	0.000	0.008	0.094	0.178	0.027	0.025	0.008	-

Table 4. Generalised linear models selected, with lowest AICC, to test the effect of the gradient of land use (first axis of PCA) on size-related variables. The expected direction of response to each land use is given for each variable (Beta coeff.). Durbin-Watson test informs about spatial autocorrelation.

Variables	Model	Beta coeff.	P-value	Likelihood ratio test	Durbin-Watson
Species richness	First axis PCA	0.50	0.000	78.35	1.82
	Method	-0.43	0.000		
	First axis PCA*Method	-0.52	0.000		
Fish abundance	First axis PCA	0.49	0.000	709.35	1.87
	Method	-0.35	0.000		
	First axis PCA*Method	-0.34	0.000		
Species diversity (Shannon)	First axis PCA	0.69	0.000	15.72	2.08
	Method	0.23	0.341		
	First axis PCA*Method	-0.75	0.001		
Size-related variables					
Size diversity	First axis PCA	-0.16	0.047	3.67	1.90
Slope length spectra	First axis PCA	-0.21	0.020	4.91	1.61
Arithmetic mean length (mm)	First axis PCA	1.53	0.563	0.34	1.68
Maximum length (mm)	First axis PCA	-19.55	0.310	0.97	1.86
Total range (mm)	First axis PCA	-0.34	0.000	199.50	1.89
	Method	0.31	0.000		
	First axis PCA*Method	0.24	0.000		

Table 5. Generalised linear models selected, with lowest AICC, to test the effect of the gradient of land use (first Axis of PCA) on the abundance of three tolerant fish species. The expected direction of response to each land use is given for each variable (Beta coeff.). Durbin-Watson test informs about spatial autocorrelation.

Fish abundance	Model	Beta coeff.	P-value	Likelihood ratio test	Durbin-Watson
<i>Bryconamericus iheringii</i>	First axis PCA	0.69	0.000	469.53	1.96
	Method	0.88	0.000		
<i>Cheirodon interruptus</i>	First axis PCA*Method	-1.16	0.000	709.35	1.09
	First axis PCA	1.15	0.000		
<i>Cnesterodon decemmaculatus</i>	Method	-0.87	0.000	409.23	2.34
	First axis PCA*Method	-0.68	0.000		
	First axis PCA	-1.34	0.000		
	Method	-0.24	0.000		
	First axis PCA*Method	1.69	0.000		

Fig. 1. Location of the 26 sampling sites in Uruguay. See site codes in Table 1 for further details.

Fig. 2. Examples of contrasting fish-length distributions: high size diversity appeared with more equally distributed fish lengths (e.g. Conchillas 2), while low size diversity occurred when the catches included many different fish lengths (i.e. a wide size range) (e.g. Las Piedras 3). See Table 1 for details on land use and water chemical features at each sampling site.

Fig. 3. Examples of two contrasting tendencies in linear size spectra. Left: sites (1, 3) with high values of rangelands and low values of slope and high intercept. Right: sites (17, 23) sites dominated by urban and agricultural land use and exhibiting high values of slope and low intercept.

Fig. 4. Principal Component Analyses (PCA) of the factor loadings of land use, chemical and mesohabitat variables in Uruguayan streams. The two axes explained, respectively, 45.5% and 26.7% of the variance. Land uses are shown by filled circles. Area is the sub-basin area.

Fig. 5. Conceptual diagram depicting the effects of different kinds of land use on fish community size structure.

Appendix A. Correlation matrix of the chemical and use land variables. The white matrix contains Spearman's r values; the grey matrix contains the corresponding P -values. Significant correlations are highlighted in bold.

Variables	Total P (μm)	Total N (μm)	rangelands	urban	agricultural	silviculture	quarry	sub-basin area
Total P (μm)	-	0.799	-0.489	0.559	0.482	-0.155	0.327	-0.315
Total N (μm)	0.000	-	-0.484	0.747	0.446	-0.189	0.515	-0.267
rangelands	0.021	0.022	-	-0.606	-0.938	0.294	-0.267	0.682
urban	0.007	0.000	0.001	-	0.468	-0.272	0.468	-0.472
agricultural	0.023	0.037	0.000	0.016	-	-0.450	0.282	-0.737
silviculture	0.491	0.399	0.145	0.180	0.021	-	-0.093	0.626
quarry	0.138	0.014	0.188	0.016	0.164	0.652	-	0.008
sub-basin area	0.153	0.230	0.000	0.015	0.000	0.001	0.968	-

Appendix B. The linear size spectrum was calculated as linear least-square regressions between $\log_2$ numbers accumulated per size class and $\log_2$ midpoint of size classes in each sampling site (see Figures of size spectrum in Appendix C). In this table *p*-value of linear regression, y-intercept, slope and coefficient of determination are showed.

Site name	Site code	y-intercept a	Slope b	R ²	P-value
A. Sopas	1	-2.38	12.40	0.95	0.005
Tacuarembó chico	2	-2.28	10.87	0.84	0.028
C. Honda	3	-1.85	9.28	0.92	0.009
California	4	-1.79	9.33	0.74	0.060
C. del Paso	5	-2.21	11.42	0.92	0.010
Carpintería	6	-1.95	9.54	0.92	0.009
A. del Sauce	7	-2.53	13.16	0.94	0.006
Monzón	8	-2.11	11.09	0.81	0.036
Marincho	9	-1.95	10.23	0.83	0.030
Molles del pescado	10	-2.03	10.15	0.96	0.003
Mansavillagra	11	-1.81	8.51	0.76	0.054
Puntas pantanoso	12	-1.41	8.07	0.81	0.036
Chal chal 1	13	-2.18	11.56	0.91	0.013
Chal chal 2	14	-2.17	11.20	0.98	0.002
Conchillas 2	15	-1.20	7.80	0.84	0.029
Colorado 3	16	-1.99	9.06	0.95	0.005
Cañada de Colorado	17	-1.16	6.50	0.73	0.065
Dragón 2	18	-2.20	10.37	0.91	0.011
Colorado 2	19	-1.61	7.31	0.93	0.008
Colorado 1	20	-1.34	8.27	0.86	0.023
Las Piedras 1	21	-2.52	12.15	0.87	0.021
Juncal	22	-2.10	10.58	0.97	0.002
Las Piedras 3	23	-1.54	7.16	0.61	0.119
Conchillas 1	24	-1.74	9.34	0.62	0.116
Las Piedras 2	25	-1.35	7.55	0.78	0.045
Dragon 1	26	-0.31	2.85	0.11	0.581

Appendix C. Size spectrums (between $\log_2$ abundance and $\log_2$ of size classes) for each sampling site. See appendix B for p -value of linear regression, y-intercept, slope and coefficient of determination.

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