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Ecological Status of a Patagonian Mountain River: Usefulness of Environmental and Biotic Metrics for Rehabilitation Assessment

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Abstract This work evaluates the consequences of anthropogenic pressures at different sections of a Patagonian mountain river using a set of environmental and biological measures. A map of risk of soil erosion at a basin scale was also produced. The study was conducted at 12 sites along the Percy River system, where physicochemical parameters, riparian ecosystem quality, habitat condition, plants, and macroinvertebrates were investigated. While livestock and wood collection, the dominant activities at upper and mean basin sites resulted in an important loss of the forest cover still the riparian ecosystem remains in a relatively good status of conservation, as do the in-stream habitat conditions and physicochemical features. Besides, most indicators based on macroinvertebrates revealed that both upper and middle basin sections supported similar assemblages, richness, density, and most functional feeding group attributes. Instead, the lower urbanized basin showed increases in conductivity and nutrient values, poor quality in the riparian ecosystem, and habitat condition. According to the multivariate analysis, ammonia level, elevation, current velocity, and habitat conditions had explanatory power on benthos assemblages. Discharge, naturalness of the river channel, flood plain morphology, conservation status, and percent of urban areas were important moderators of plant composition. Finally, although the present land use in the basin would not produce a significant risk of soil erosion, unsustainable practices that promotes the substitution of the forest for shrubs would lead to severe consequences. Mitigation efforts should be directed to protect headwater forest, restore altered riparian ecosystem, and to control the incipient eutrophication process.

Keywords Land use · Habitat quality · Riparian ecosystems · Plants · Macroinvertebrates · Forest conservation

Introduction

In the last decades, the consequences of different land-use practices on stream integrity have received major attention (Allan 2004). Agriculture, livestock grazing, deforestation, urbanization, and other human modification of the landscape alter and degrade stream ecosystems (Paul and Meyer 2001; Castela et al. 2008; Nelson 2010). Disturbances such as soil erosion, sedimentation, nutrient enrichment, changes in light irradiance and water temperature, allochthonous organic matter, and input of toxic substances can dramatically change aquatic habitats and their biological communities (Scrimgeour and Kendall 2003; Lussier et al. 2008; Richardson 2008; Theodoropoulos and Iliopoulou-Georgudaki 2010). Although such changes have been well documented across varied landscapes, the pathways between the terrestrial and the lotic aquatic

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environments are usually complex and of multidimensional nature (Ward 1989). The selection of appropriate ecological measures to quantify the magnitude of disturbances on river condition is not always an easy task.

The influence of land use on the ecological integrity of streams has proved to be scale-dependent (Buck et al. 2004; Richardson 2008; Kail and Wolter 2013). Some stream characteristics are strongly affected by the relative proportion of different land use across the entire basin. Landscape attributes at large spatial scales are good predictors of ecological integrity in rivers. For example, disturbance of forest land cover of the catchment (forestry practices, fires, etc.) can result in substantial loss of soil, excessive sedimentation, and in changes of the frequency and intensity of flooding. All these phenomena have been demonstrated to affect species distribution patterns (Buck et al. 2004). On the other hand, local-scale anthropogenic impacts, such as riparian forest clearing, can affect the amount and source of primary production in streams through changes in shading and leaf litter entering the stream from the surrounding forest (Snyder et al. 2003). In this scenario, all species depending on these energy subsidies will be affected.

Indicators employed to assess the ecological status of rivers should pursue objectivity, effectiveness, and consistency (Buendia et al. 2013; Feio et al. 2014). Traditionally, environmental measures to detect impairment in aquatic ecosystems focused primarily on physical and chemical variables. However, the concept of ecological integrity of rivers incorporates biological metrics based on aquatic (bacteria, periphyton, macroinvertebrates, and fish) and riparian communities (plants, birds) (Rosenberg and Resh 1993; Munné et al. 1998; McCormick et al. 2001; Navarro et al. 2011; Paul and Meyer 2001). These more complex approaches that incorporate diverse communities are very useful because they help identify problems of different spatial scales, sources, and intensity. For example, the ecological effects of fine sediments in basins affected by natural sedimentation have been better understood by using macroinvertebrates, where insect species displaying polyvoltinism, short live cycles, and small body sizes had higher survival rates (Buendia et al. 2013). The same disturbance (elevated exposure to fine sediment) dramatically decreased sensitive invertebrate groups in agricultural (Sutherland et al. 2012) and other human-altered landscapes (Relyea et al. 2012). Fish assemblages seem to be more sensitive to flow variability and adjacent land uses than other aquatic communities (Lammert and Allan 1999). The response of bird assemblages to reach-scale land-use effects has been mixed (Miserendino et al. 2011; Navarro et al. 2011). Measurements of riparian plant community quality have been used effectively to test for ecological consequences of urbanization, logging, pasture conversion, livestock, and pine plantations (Kutschker et al. 2009; Sirombra and Mesa 2012). These riparian metrics also have shown predictive value for invertebrate functional assemblages in subtropical rivers, with lower invertebrate diversity and richness in degraded reaches versus native forested sites (Mesa 2014). Ultimately, multiple metrics drawn from different biological communities are very valuable to craft best management plans, propose mitigation actions, and to appropriately direct restoration efforts (Stoddard 2004).

Frequently, the land-use practices and anthropogenic actions in a basin vary in type, time, extent, and spatial location (Richardson 2008; Goldstein et al. 2007). This is the most common scenario in catchments having altitudinal gradients such as in the steppe-mountain ecotone in Patagonia. In the Percy River basin (Chubut, Argentina), cattle grazing have been the dominant land use in upland areas for the last 100 years. Planned forest fires and conversion of the native *Nothofagus* forest ($\sim$60–70 years ago) to pastures has resulted in fragmentation processes and patchy forest cover. Overgrazing, livestock trampling, and deforestation affected basin coverage and riparian buffers, these probably produced remarkable sedimentation inputs on the river and modified the bar development in the main channel. Many disturbances currently affecting lower reaches (flooding, high runoff, excessive sedimentation) likely reflect anthropogenic activities such as forest removal and overgrazing that are concentrated in upland areas (Quinteros et al. 2010). Reduction in forest cover is often associated with stream water quality degradation, especially in rapidly developing mountainous areas (Price and Leigh 2006; Richardson 2008). As occurs elsewhere, the urbanization process has been mostly associated with the valleys in the lowland areas due to natural geographical constrictions.

From an historical perspective, the territorial organization and development of the region has been at some point unplanned, with overlapped land-use practices, an unsustainable forest management (Carabelli and Scoz 2008), in addition to accelerating urbanization. There is a great concern about the rates at which these changes are occurring and how this will affect the ecological integrity of aquatic resources.

Although much scientific research has been conducted in the area (Miserendino et al. 2008, 2011), little work has been done in the Percy River. At present, there is no information on soil erosion risk or ecological status of the river, or evaluation of how these features could change under future land-use scenarios. This multidisciplinary study will attempt to (1) evaluate the consequences of anthropogenic pressures at different sections of the Percy River using a suite of environmental and biological measures including macroinvertebrates, riparian plants, and habitat condition, (2) quantify present soil loss and analyze future soil erosion risk under intensified land-use practices, and (3) propose mitigation and rehabilitation measures to help guide basin management. We hypothesize that longitudinal variation in biotic assemblages is primarily due to human impacts rather than by altitudinal zonation.

Methods

Study Area and Sampling Program

The Percy River (catchment area 1003.8 km^2) is located between 42°37′33″ and 43°09′27″S and 71°40′14″ and 71°9′49″W in the Chubut Province of Argentina (Fig. 1). It is a fifth order (Strahler method), high-gradient (maximum and mean height, 2145 and 792.73 m above sea level, respectively), perennial river (mean discharge 11.99 m^3 s^{-1}) running through predominantly granitic areas typical of Patagonia mountains (Coronato and del Valle 1988), and draining to west toward the Pacific Ocean trough of the Futaleufú River.

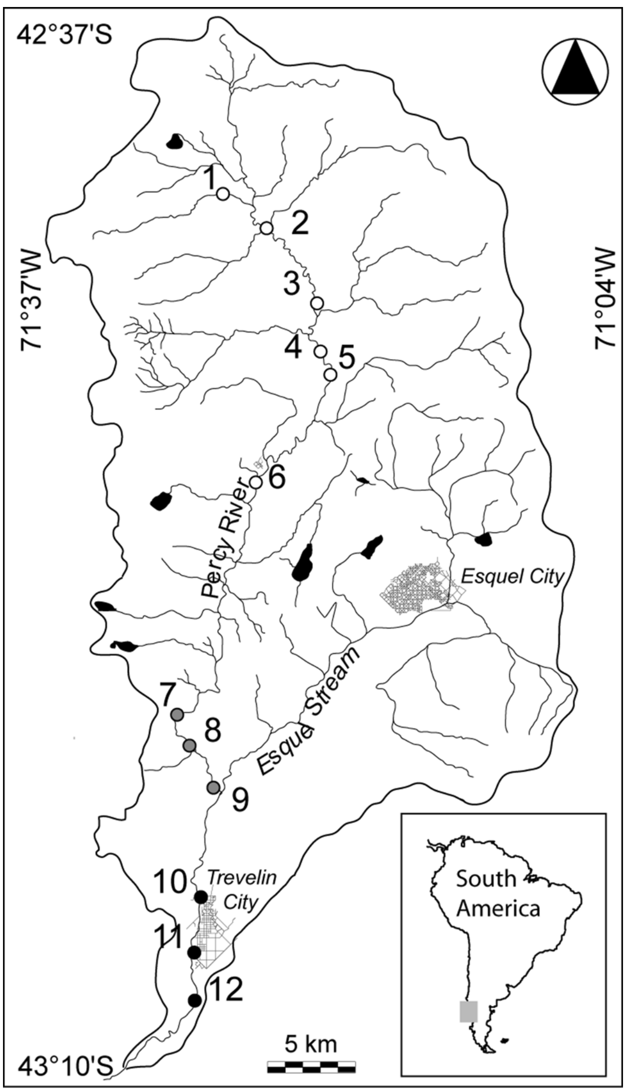

Fig. 1 Location of sampling sites at the Percy River (Chubut, Patagonia) during the study (February 2011). Sites distribution according to basin sections: *1–6* upper, *7–9* middle, and *10–12* lower

Marine and continental sedimentary rocks from the Oligocene constitute the dominant substrate for the Percy and its major tributaries, with recent fluvial, glacial, and glaciofluvial deposits (Cucchi 1980). The landscape and soils of the study area are characterized by a marked west–east pluviometric gradient. Geomorphic action of the Pleistocene glaciers, and the subsequent deposition of volcanic ashes contributed to the fertility of the soils. Two major soil types present in the Percy catchment are the Udic moisture regime dominated by udivitrands, and the xeric moisture regime, dominated by vitraxerands and haploxerolls (Irisarri et al. 2000).

Climate corresponds to cold-temperate humid (Paruelo et al. 1998). The study area is located in the ecotone between Subantarctic forest and the Patagonian steppe, and exhibits a marked altitudinal gradient. At least 40 % of the basin is covered by forest and secondary scrublands of *Nothofagus antarctica* (altitudinal distribution range at this latitude <300–1200 m a.s.l.), and to a lesser extent by *Nothofagus pumilio* and *Austrocedrus chilensis* (León et al. 1998).

The historical land use at the upper and middle sections of the Percy basin is rural, consisting mostly of extensive livestock operations and fuel wood collection. Some occurrences of forest fires and pasture conversion have resulted in losses and replacement of the native vegetation, and in damage of forest regeneration due to browsing and trampling (CFI 2006; Quinteros et al. 2012, 2013). Observed impacts include sedimentation, floodplain modification, riparian clearing, and flooding of bridges and roads mostly in the lower basin. Agricultural practices (pastures, traditional and ornamental horticulture) and urbanization are the dominant land uses in the lower basin (CFI 2006). The Esquel tributary flows through the town of Esquel (32,000 inhabitants) before joining the Percy River shortly upstream of the town of Trevelin (10,000 inhabitants). In the lower basin, water is extracted from the river for irrigation purposes through a network of small artificial channels. Dredging activities have resulted in bar and bank movement and riparian modification. Trevelin has a wastewater treatment plant that processes the effluents of approximately 80 % of the population; all treated effluents are discharged into the river.

At present, the regulatory framework that applies to the Percy Basin development is the Federal law (No. 26.331) referred to: Environmental Protection for Native Forest, where three categories of forest preservation have been defined (conservation level: 1: high; 2: moderate, and 3: not protected). The law attempts to promote conservation measures by regulating and controlling the extent of Native Forest affected by management plans in Patagonian basins. As part of a Federal program, watersheds having native forest in the Patagonia Mountain range have been classified and assigned to these three categories of conservation. The assignation process resulted from a series of inter-disciplinary and participatory workshops involving different local experts and actors from governmental administrations, universities, and NGOs (CFI 2007; DGBP 2008).

A total of 29.2 % of the vegetation in the Percy basin falls in the level 1 (high) where the forest is fully protected and no extractive or commercial activity is allowed. This category comprises mostly the headwater forest represented by *Nothofagus pumilio*. Only 2.3 % is considered to have a low value for conservation purposes (status 3: not protected) and can be modified or transformed totally or partially by different practices, and represent the nearby section to urban areas. The rest (68.5 %) is considered of moderate value of conservation (level 2), and according to the regulatory framework, all this area can be subjected to sustainable practices (pastures, forestry, tourism, etc.). The most abundant species at the section is the native *Nothofagus antarctica*, a non-commercial value species, used mostly for fuel. As best pasture for forage is often associated with *N. antarctica*, cattle grazing is developed there (Quinteros et al. 2010). The history of forest fires at these silvopastoral systems seems to enhance plant-level fuel flammability (Blackhall et al. 2015), and then the regeneration process can result compromised for the livestock load due to browsing and trampling. At present, the majority of the Percy basin falls into the non-fully protected category, which means that several sustainable activities are being promoted by government agencies.

Field data were collected during summer (February 2011) from 12 sites (a 100-m reach) (Fig. 1) whose selection was done according to their location and accessibility in the basin. To minimize the influence of diel cycles, most measurements were carried out within same hour range (midday to afternoon) in consecutive days. Six sites were located in the upper basin (U: 1–6), three in the middle (M: 7–9), and three in the lower (L: 10–12) Percy River. Site 1 was settled on a headwater tributary. Four sites (2–5) were established on the Percy River main channel, in an area where the main land use is agricultural (wood collection, pasture development, and livestock). Site 6 was established downstream of the rural town Aldea del Río Percy (140 inhabitants). Sites 7–9 were located upstream of where the Esquel Stream joins the Percy River, recreation and some rural activities being the dominant land uses in this area. In the lower urbanized section, site 10 was selected just upstream of major urban areas, site 11 adjacent to the town of Trevelin, and site 12 downstream of the main facilities of the waste treatment plant of the town.

Soil Erosion Quantification and Status of Conservation at a Basin Scale

Soil loss in the Percy basin was assessed by the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978). Risk factors (rainfall-runoff-erosivity, soil cover, slope length, and slope gradient) were estimated according to FAO (1980) and using the most accurate available local information about precipitation, soils, digital elevation models (DEMs), and vegetation. Practice factor was set to 1 because there are no erosion control practices in the study area. Digital information was processed with ArcView 3.3 software. Layers were converted to grids with 30×30 m^2 cells to match the DEM resolution.

Cover factor was estimated for two conditions: current cover and a modified non-sustainable future scenario in which native forest areas, classified as medium and low conservation value by the forest law, were converted from forest to shrubbery. This is the most common post disturbance response at *Nothofagus* forested areas. For example, after fire episodes (Kitzberger 2003) or in areas with mixed management (harvest and livestock) where usually tree regeneration is disrupted by cattle (Quinteros et al. 2012; Blackhall et al. 2015).

The percent of forest conservation according to the level of preservation (high, moderate, low) and the percent of urban areas upstream each sampling site were estimated using the Tabulate Area tool of Arc View software. This tool allows the user to summarize the area corresponding to the intersection of layer categories.

Land Use and In-stream Habitat Condition

A set of field measurements were taken to assess environmental features adjacent to and within the active channel at each study site. Dominant land use at each site was classified either as native forest, agricultural, pasture, or urban. Morphology and naturalness of the floodplain in the 100-m reach was assessed as in Kutschker et al. (2009) by assigning a score of 0–25 that reflects level of channel modification.

Substrate composition was visually estimated as the percentage of boulder, cobble, gravel, pebble, and sand within a 1-m^2 grid (Gordon et al. 1994). Contribution of algae and aquatic plant coverage was also estimated. Average depth was calculated from five measurements taken along a channel-spanning transect. Surface current speed was obtained by timing a bobber (average of three releases) over a distance of 10 m. Wetted and bankfull widths were also measured, and discharge calculated based on depth, wetted width, and surface velocities as in Gordon et al. (1994).

In order to assess the in-stream habitat available to the biota, we calculated the habitat condition index (HCI) using the assessment procedure for high gradient streams of Barbour et al. (1999). This method ranks 10 river channel features (e.g., epifaunal substrate availability, frequency of riffles, etc.) from 0 to 20. A score of 200 points indicates the river is natural and pristine and in its best possible condition (range 150–200). This index evaluates the ability of the stream's physical habitat to support a given fauna, and is a measure of the spatial heterogeneity of the stream (Castela et al. 2008).

Water Chemistry and Temperature

Mid-channel at each site, specific conductivity, pH, turbidity, and dissolved oxygen were measured with a sensION 156 multiparameter probe. For nutrient analyses, water samples were collected below the water surface, kept at 4 °C and transported to the laboratory for analysis. Total nitrogen (TN) and total phosphorus (TP) were determined on unfiltered samples digested with persulfate, whereas nitrate plus nitrite nitrogen ($NO_3 + NO_2$), ammonia (NH_4), and soluble reactive phosphate (SRP) were analyzed using standard methods (APHA 1994). Total alkalinity was determined by gran titration (Wetzel and Likens 1991). Air temperature and evapotranspiration data (2000–2014) (subset size: 1×1 km^2) were obtained from ORNL DAAC (2014).

Riparian Plant Community Condition

We conducted a riparian plant species inventory at each study site that included aquatic macrophytes (mostly emergent in the flooded area), herbs, shrubs, and trees (Correa 1978, 1999). From this inventory, we calculated total taxa richness, species diversity (Shannon Wiener Index), species cover, and the contribution of native and exotic species (Matteucci and Colma 1982). The complexity and the attributes of the riparian vegetation were also analyzed using an adaptation of the QBR index (Riparian corridor quality index, Munné et al. 1998) for Patagonian streams: the QBRp (Kutschker et al. 2009). This index combines information from four additive metrics: total cover (proportion of the riparian area covered by trees and shrubs), structure (proportion of riparian vegetation composed by trees and shrubs separately), complexity and naturalness of vegetation (number of trees or shrub species and absence of introduced species, and other human impacts in riparian vegetation), and the degree of channel naturalness (e.g., bank modifications, dredging, etc.). It also takes into account differences in the geomorphology of the river from its headwaters to the lower reaches. The total QBRp score ranges from 0 (extreme degradation) to 100 points (excellent quality, natural riparian forest).

Macroinvertebrate Collection and Metric Calculations

Six benthic macroinvertebrate samples were collected from run/riffles ($n = 3$) and pools ($n = 3$) along a 50- to 100-m-long stretch at each site with a Surber sampler (frame area 0.09 m^2, 250-μm mesh). This protocol was designed based on previous studies conducted in the river and in other watercourses having same magnitude (stream order) and characteristics (substrate type) (Miserendino and Pizzolón 2000; Miserendino 2007). Samples were kept separate for quantitative analysis, fixed in a 4 % formaldehyde solution and in the laboratory were washed, sieved, sorted, and identified using a stereo microscope. All individuals were picked from the samples and stored in 70 % ethyl alcohol. Species were identified using available keys (Domínguez and Fernández 2009). A full taxon list with abundance data and species codes is given in Appendix Table 4.

We calculated 22 macroinvertebrate community descriptors for each site. Some metrics were determined from pooled samples from individual sites (e.g., Richness measures), whereas others were averages for the site based on the individual samples (Barbour et al. 1999). Richness measures employed were: taxa richness (SR), which measures the overall variety of the macroinvertebrate assemblage; Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa, which counts the number of mayfly, stonefly, and caddisfly taxa. Other measures employed were species diversity (Shannon Wiener Index), which incorporates measures of both richness and abundance, and Pielou evenness, both were calculated from averaged macroinvertebrate densities at each site. Metrics based on relative abundance were % Plecoptera, % Ephemeroptera, % Trichoptera, % Coleoptera, % Diptera, and % Anelida (Rosenberg and Resh 1993). Tolerance measures included total macroinvertebrate density (ind m^{-2}), whereas trophic measures employed were richness and percentage of functional feeding groups: shredders, scrapers, predators, collector-gatherers, and collector-filterers. We also calculated a score at each site for the BMPS (Biotic Monitoring Patagonian Streams), a biotic index adapted for use in the Patagonian region (Miserendino and Pizzolón 1999; Miserendino 2007). The BMPS is an adaptation of the BMWP (Biological Monitoring Working Party index; Armitage et al. 1983) and is computed by adding pollution sensitivity scores (1–10) for all invertebrate families present at a site for a potential range of 0 (most disturbed) to >150 (least disturbed).

Detritus obtained from benthic samples was divided into fine (250–1000 µm) and coarse (>1000 µm) particulate fractions (FPOM and CPOM, respectively). All fractions were dried (105 °C for 4 h) and then weighed on an electronic balance (Scout Pro) to ±0.003 g.

Statistical analysis

We used Kruskal–Wallis ANOVA and Mann–Whitney *U* tests to evaluate differences between upper, middle, and lower basin sites on environmental and biological metrics (Gotelli and Ellison 2004). To examine the relationship between macroinvertebrate assemblage data and environmental metrics, we used detrended correspondence analysis (DCCA) with down-weighting of rare species in the statistical software package CANOCO (Ter Braak and Smilauer 1999). This model was chosen because previous inspection of the data revealed a linear mode rather than a unimodal response in the biotic variables (Ter Braak and Smilauer 1998). Weighted average species density values (based on six Surber samples per site) were used in the analysis. All relevant environmental variables (that could affect benthos distribution) included in Tables 1 and 2 and Fig. 3, Appendix Table 6, were used, initially to evaluate the response of species and sites to environmental gradients. Variables (except pH) and species density were transformed (log $x + 1$), prior to analysis. Variables that were strongly intercorrelated with others (those with an inflation factor >20) in the initial analysis were removed and a further analysis was carried out with the nine remaining environmental variables (NH_4, alkalinity, water velocity, HCI, elevation, wet channel width, CPOM, silt, and algae percentage/plant coverage). Significance of the axes was proved by Monte Carlo permutation test (Ter Braak and Smilauer 1998).

To analyze the predictive value of environmental variables on riparian plant composition and distribution, a canonical correspondence analysis (CCA) was performed. CCA was chosen because initial analysis of species data revealed a unimodal response (Ter Braak and Smilauer 1998). Data employed were species coverage at the reach scale. Data (biological and environmental) were previously transformed (log x + 1). The option down-weighting rare species of the program was applied. Environmental variables considered in CCA ordination were those thought to most affect vegetation distribution (e.g., geographical location, altitude, attributes of the channel, air temperature, discharge, precipitation, and evapotranspiration). Moreover, variables related to land cover attributes were also included (e.g., percent of conservation and urban areas, Appendix Table 6).

Table 1 Environmental features of sites located at the upper ($n = 6$), middle ($n = 3$) and lower ($n = 3$) Percy River basin during the study period (February 2011)

	Upper	Middle	Lower
Elevation (m a.s.l.)**	867.3[a] (98.2)	449.3[b] (16.7)	370.0[c] (11.1)
Air temperature (°C)	13.8 (1.2)	14.7 (0.8)	15.2 (1)
Floodplain morphology type*	2.7[a] (0.5)	1.7[b] (0.6)	3[a] (0)
Naturalness of the river channel*	25[a] (0)	23.3[a] (2.9)	13.3[b] (5.8)
Wet channel wide (m)	16.0 (7.4)	18.8 (7.0)	16.3 (6.4)
Depth (m)	0.22 (0.08)	0.34 (0.15)	0.26 (0.02)
Water velocity (cm s^{-1})	0.67 (0.26)	0.84 (0.05)	0.91 (0.31)
Discharge (m^3 s^{-1})	2.38 (2.03)	4.91 (0.81)	3.71 (1.18)
Evapotranspiration (kg m^{-2} Day^{-1})	10.1 (0.7)	8.9 (0.7)	10.8 (2.8)
Water temperature (°C)	19.8 (5.7)	22.0 (2.4)	16.7 (1.5)
% Boulder	32.5 (9.8)	25 (10)	23.3 (12.6)
% Cobble	27.5 (6.1)	38.3 (7.6)	46.7 (12.6)
% Gravel	22.5 (12.5)	21.7 (2.9)	16.7 (5.8)
% Sand	14.2 (5.8)	11.7 (2.9)	13.3 (7.6)
% Silt	3.3 (4.1)	3.3 (2.9)	3.3 (2.9)
pH**	7.35[a] (0.17)	7.99[b] (0.17)	7.89[b] (0.10)
Total alkalinity (meq L^{-1})**	1.04[a] (0.09)	1.64[b] (0.39)	1.42[b] (0.08)
Dissolved oxygen (mg L^{-1})**	7.78[a] (0.29)	9.23[b] (0.53)	10.02[b] (0.69)
Oxygen Saturation (%)*	88.0[a] (12.2)	109.0[b] (1.7)	105.2[b] (13.1)

Data are mean values (±SD). Significant differences according to Kruskal–Wallis non parametric ANOVA test are marked as: * $p < 0.05$; ** $p < 0.01$. Same letter indicates no differences among groups

Table 2 Variation of the primary producers coverage, biomass of detrital fractions, values and judgment classes of quality of Patagonian riparian ecosystem index, and of habitat assessment index at sites located at the upper ($n = 6$), middle ($n = 3$) and lower ($n = 3$) Percy River basin during the study period (February 2011)

	Upper	Middle	Lower
Riparian ecosystem			
Riparian species richness	26.5 (3.02)	29 (2)	19.3* (4.9)
Richness of exotics plant	11.3 (3.4)	14 (3)	16.7 (3)
Coverage of exotics plan	44 (9.5)	50 (9.6)	92.7* (8)
Riparian diversity (H)	1.11 (0.1)	1.20 (0.02)	0.92* (0.11)
QBRp index	73.7 (13.3)	71.5 (13.8)	42.2 (11.2)
	77–95	57–84.5	34.5–55
Quality judgment	Riparian ecosystem lightly to moderately disturbed	Riparian ecosystem lightly to moderately disturbed	Riparian ecosystem disturbed
	Good quality	Good quality	Poor quality
Dominant land use	Agricultural	Recreation	Urban
	Pasture	Agricultural	Recreation
In-stream features			
% algae/aq. plant coverage	9.3 (12.3)	20 (34.6)	26.7 (22.5)
CPOM (g m^{-2})	7.74 (9.37)	1.92 (0.38)	3.20 (0.87)
	1.56–27.7	1.49–2.05	2.39–4.12
FPOM (g m^{-2})	3.98 (3.83)	2.58 (0.41)	2.84 (1.35)
	1.53–11.67	2.11–2.83	1.45–4.15
Habitat condition index	160.6 (19.0)	170.3 (16.0)	124.7 (31.5)
	143.5–189	141.5–186	97–159
Quality judgment	Optimal	Optimal	Suboptimal

Data are mean values (±SD), ranges are expressed for indexes and detrital fractions. Significant differences according to Kruskal–Wallis non parametric ANOVA test are marked as: * $p < 0.05$

Results

Environmental Characterization

The analysis of current soil erosion risk revealed that 8 % (ca. 7950 ha) of the Percy basin suffers moderate (6.6 %) or high (1.4 %) erosion risk. These areas occupied headwaters where high-to-moderate gradient low-order tributaries drain naturally forested *N. pumilio* lands. Moreover, some lands with moderate gradients associated with the urban and suburban area of Esquel were categorized into moderate to high risk of erosion. A total of 92 % of the basin was categorized as having little to no erosion risk (<10 Mg ha^{-1} $year^{-1}$), within this category rock outcrops occupied 11 % (Fig. 2a).

These results differed greatly under the non-sustainable future forest use scenario (Fig. 2b). In this case, the estimated area with moderate and high erosion rates rise to 20.7 % (20,650 ha) and 8.1 % (8060 ha), respectively. The spatial distribution of risk will vary at different sections of the basin. Based on the model results, the upper basin, occupied partly by the natural forest of *N. anctarctica*, will be highly impacted. A similar situation is predicted to the area located on the western side of the lower basin, at present occupied with *N. antarctica*, *Austrocedrus chilensis*, and grasslands. This section is being rapidly urbanized, with Trevelin City suffering an active expansion to rural areas. In this context, the situation of suburban areas at the town could change dramatically (0.03 % of moderate risk to 52 % moderate/high risk). At the urbanized area where the erosion risk at present is low, an increase of 18 % in the category moderate risk could occur. The model also predicted an increase (26.4–31.6 %) in the category of moderate/high risk of erosion for Esquel City.

The gradient of environmental conditions in the longitudinal dimension of the river was evidenced in the variables pH, alkalinity, dissolved oxygen (increased U < M, L), and saturation percentage (M > U, L) (Table 1). Air temperature did not differ significantly among basin sites during the study period. Most sites were comparable in terms of features of the channel and flow as revealed in the variables wet channel wide, depth, water velocity, and discharge ($p > 0.05$). Instead, significant differences (Kruskal–Wallis) were documented for variables related with anthropogenic pressures: conductivity ($p = 0.03$), which was higher at L than U section, and nutrients, NO_3

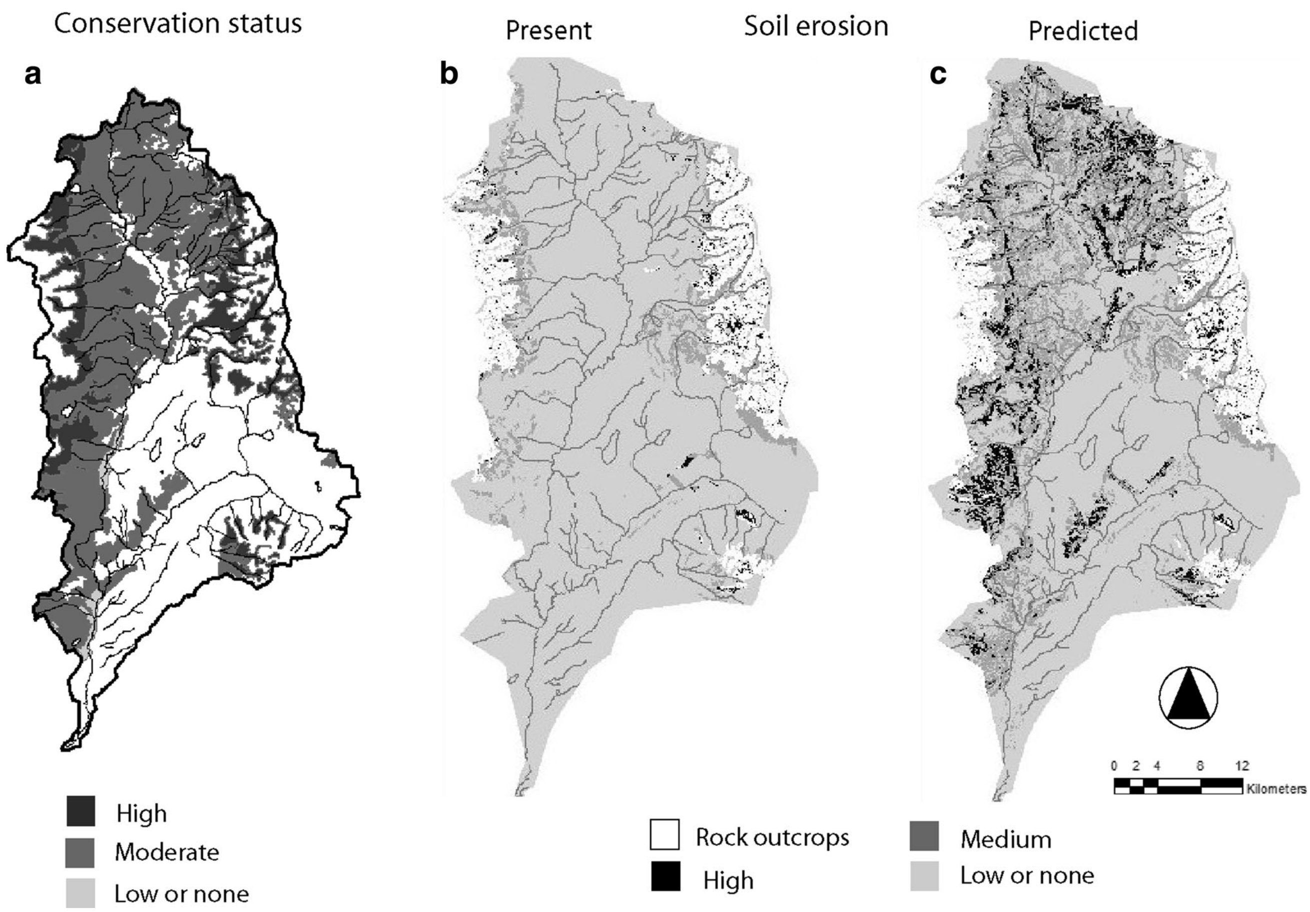

Fig. 2 **a** Map of conservation of the basin under the current territorial planning law of forest protection, in *white color*: non-forested areas. Soil loss according to USLE for two scenarios: **b** current cover, **c** forest land converted to shrubbery in areas not considered relevant for conservation according to the current law. Erosion rates classes: *light grey* low or none (<10 Mg ha^{-1} year^{-1}); *dark grey* medium (10–50 Mg ha^{-1} year^{-1}); *black* high (50–200 Mg ha^{-1} year^{-1}). *White* rock outcrops

($p = 0.002$), NH_4 ($p = 0.02$), and SRP ($p = 0.0012$) that were found to be significantly higher at L than at U and M sections (Fig. 3).

Plant Community, Riparian, and In-Stream Habitat Condition

Regarding plant composition, a total of 81 taxa were identified in the riparian zone across all sites, of which over 40 % were exotics (35 taxa). Total species richness per site ranged from 16 to 31, with a mean of 22.8 taxa (Appendix Table 4).

Riparian ecosystem metrics showed that plant assemblages were significantly richer and more diverse at U and M, than at L (Table 2). The contribution of exotics, primarily *Salix fragilis* and *S. alba,* was marked at lower basin sites, in several cases being the dominant species. Conversely, the relative abundance of the native *Nothofagus antarctica,* diminished and was practically absent at M and L sites ($p < 0.05$).

Sites at the upper and middle sections of the basin were lightly to moderately disturbed according to the QBRp (mean scores 73.7 and 71.5 respectively), compared to lower sites where riparian quality was poor (Table 2, Appendix Table 6).

Distribution and biomass of coarse and fine particulate organic matter was quite similar among basin sections (Table 2, $p > 0.05$) and very variable within sites at sections (e.g., CPOM ranged from 1.6 to 27.71 g m^{-2} at U sites). The contribution of algae and aquatic plant from the headwater to lower sites increased but not markedly ($p > 0.05$). The HCI was optimal as both L (score 160.6) and M (score 170), but suboptimal at L (score 124.7) (Table 2, Appendix Table 6).

Macroinvertebrate Metrics

Benthic macroinvertebrates were abundant organisms (total mean density 2844 in d m^{-2}) and rich in terms of species (58 taxa). Diptera (24), Plecoptera (8), Trichoptera (8), Ephemeroptera (6), and Coleoptera (4) were the best represented

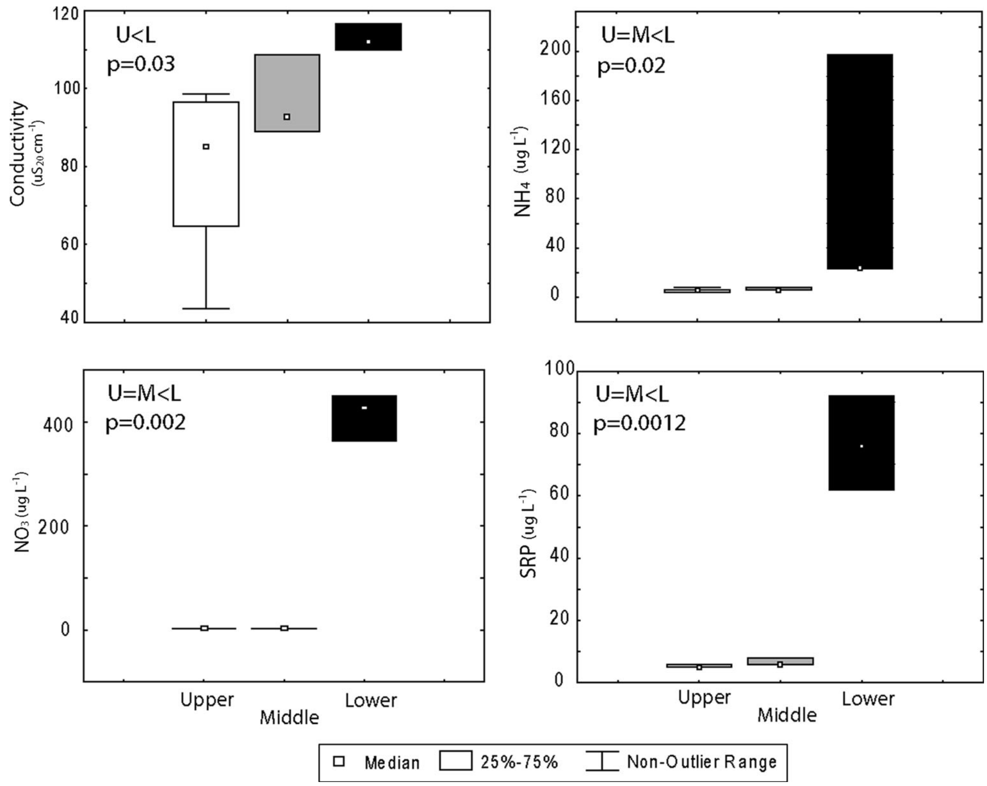

Fig. 3 Physicochemical features at the upper (*white*), middle (*grey*), and lower (*black*) basin sites at Percy River during the study (February 2011). Significant differences are expressed after Kruskal–Wallis test

orders. Oligochaeta, Crustacea, and Platyhelminthes comprised the rest of the benthic community. Total richness per site varied between 18 and 30 at sites 4 and 11, respectively, whereas mean density was between 983 ind m^{-2} (site 1) and 9003 ind m^{-2} (site 11) (Appendix Table 4).

Of the 22 invertebrate metrics we tested, 10 showed significant differences between river sections (Fig. 4). Total taxa richness and density and EPT taxa richness were significantly higher at L than U and M sites. In terms of density, worms (*Limnodrilus* sp. and *Nais communis*), midges (*Paratrichocladius* and *Parapsetrocladius*), and baetids (*Andesiops torrens*) were the groups that best explained this pattern. The relative contribution of Trichoptera was higher at M and L than U. Annelida responded as predicted, with significantly higher relative abundance at L sites compared to M and U. The opposite pattern was detected for % Plecoptera of decreasing abundance moving from upstream to downstream sites (Figs. 4, 5). Among the functional metrics, richness and relative abundance of collector gathered responded as predicted with higher values at L than at M, but not significantly different from U. A significant increase in predator richness from U to lower sites ($U < M, L$) was observed. The contribution of collector-filterers (%) increased from U to M sites. We did not observe any significant differences by river section for any of the species diversity or evenness measures, nor did we detect differences in the BMPS.

Natural and Anthropogenic Predictors of Biotic Assemblages

Results of detrended CCA based on macroinvertebrate data are summarized in Table 3 and shown in Fig. 6. The environmental variables selected in the analysis are represented in the biplot by arrows pointing in the direction of maximum change in the value of the associated variable (Fig. 6a). The species–environmental correlations were 0.99 and 0.97 for the first and second axes, respectively (Table 3a), indicating strong relationships with the environmental variables selected. Nevertheless, the Monte Carlo test was only significant for the first axis, highlighting one marked environmental gradient.

The strongest explanatory factors were related with physicochemical features and with the ecological integrity of sites, and 41.1 % of variation in the species abundance data was accounted by the environmental variables measured (Table 3). DCCA1 reflects the distribution of species along a gradient of elevation, ammonia, habitat condition, and water velocity highlighting natural and anthropogenic factors. Secondary variables associated with this axis were percentage of algae, aquatic plant total alkalinity, and silt percentage. An assemblage of taxa typical of clean mountain rivers: *Klapopteryx kuscheli*, *Cura* sp., *Rheochorema lobuliferum*, and *Podonomus* sp. was placed on the lower left quadrant. *Limnodrilus* sp. and Glossiphoniidae spp. taxa were frequently associated with disturbed or

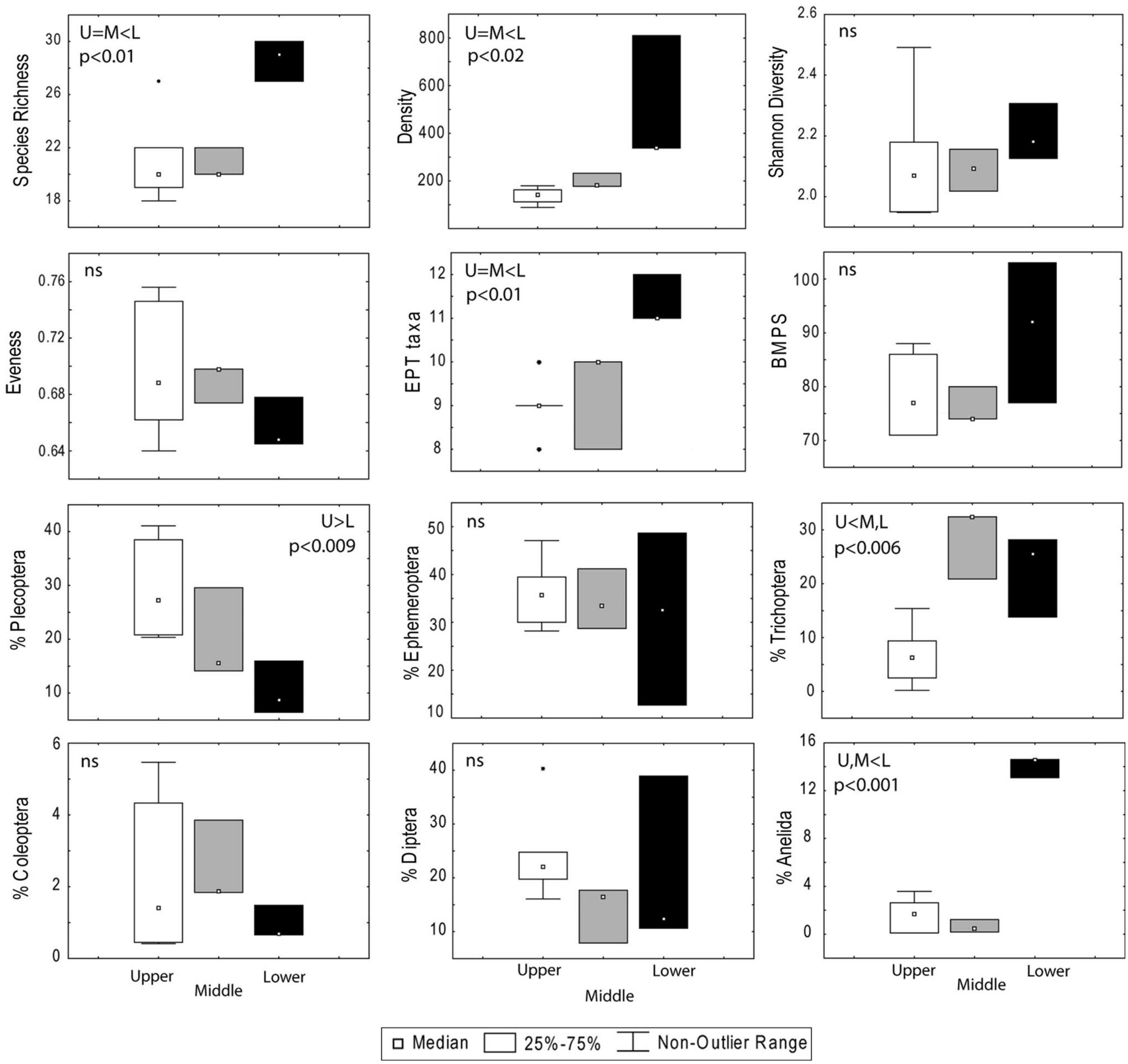

Fig. 4 Macroinvertebrate metrics based in measures of richness, tolerance, relative abundance, and the Biotic Monitoring Patagonian Stream index, at the upper (*white*), middle (*grey*), and lower (*black*) basin sites at Percy River during the study (February 2011). Significant differences are expressed after Kruskal–Wallis test

polluted environments, and were positioned to the positive end of DCCA1 (Fig. 6b). A group of species that appeared at nearly all sample sites (*Andesiops peruvianus*, *A. torrens*, *Cailloma pumida*, *Potamoperla myrmidon*, *Paratrichocladius* spp., Elmidae spp., and *Meridialaris laminata*) were located on the center of the ordination diagram.

Results of the CCA analysis based on riparian plant species revealed that the main variables influencing species distribution were related to natural gradients defined by discharge and air temperature, but also by the proportion of the basin upstream having high level of conservation (CCA1). Percent of urban area, naturalness of the river channel, and floodplain morphology were significant features along CCA2 (Fig. 6c). All variables explained 46.7 % of species abundance and the model was highly significant ($p = 0.003$, Table 3). Native species (*Maytenus chubutensis*, *Discaria articulata*, *Vicia nigricans*, *Calceolaria* sp., and *Ranunculus* sp.) characteristic of upper river sites were grouped on the positive end of CCA1 (Fig. 6d). A plant assemblage composed mostly of exotics (*Polygonum aviculare*, *Cytisus scoparius*, *Chenopodium album*, *Capsella bursa-pastoris*, *Saponaria officinalis,* and

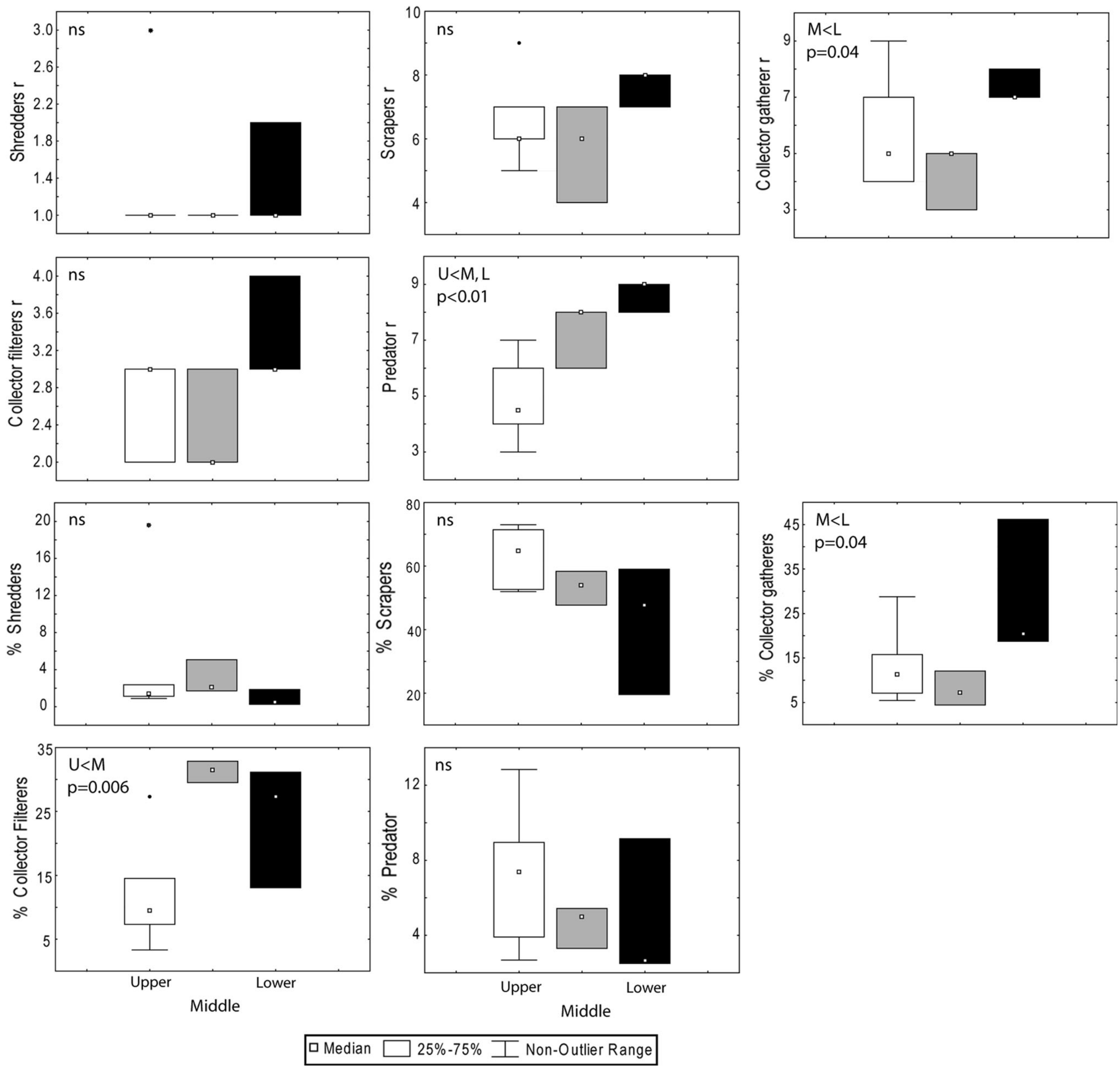

Fig. 5 Macroinvertebrate metrics following functional feeding groups at the upper (*white*), middle (*grey*), and lower (*black*) basin sites at Percy River during the study (February 2011). Significant differences are expressed after Kruskal–Wallis test

Dysphania multifida) characterized urbanized river sites and was located on the top of the left corner of the plot.

Discussion

Anthropogenic Disturbance and Natural Gradients in the Basin

In this research, we examined how environmental features affected a diverse assemblage of plants (81 taxa) and macroinvertebrates (58 taxa) in an urbanizing river basin of Patagonia. This work is novel for the region, and helps establish background levels of ecological integrity across the river basin. When proposing restoration measures it is important to identify how biological assemblages respond along different environmental gradients (Hughes et al. 2010; Nelson 2010). The Percy River displays an ecological complexity that reflects natural environmental gradient of elevation, temperature, and precipitation, but which is also affected by multiple anthropogenic impacts. As in other related studies, disentangling the interacting effects of natural and human gradients on the biotic components of fluvial networks is challenging (Ward 1989).

Table 3 Weighted intraset correlation of environmental variables with the axes of (a) Detrended Canonical Correspondence Analysis (DCCA) based on macroinvertebrate species density (b) Community correspondence analysis based on riparian plant coverage species in the Percy River system, Patagonia, Argentina. Most explanatory variables are marked in bold

	DCCA	
	Axis 1	Axis 2
(a) Macroinvertebrates		
Eigenvalue	0.330	0.141
Species-environmental correlations	0.997	0.970
Cumulative percentage variance		
Of species data	36.0	51.5
Of species-environmental relation	41.1	58.8
Correlations		
Habitat condition	**−0.65**	−0.09
Elevation	**−0.81**	0.36
Wet channel	−0.12	0.23
Ammonia	**0.91**	0.02
Alkalinity	**0.47**	**−0.41**
% Algae and Aq. plants	**0.52**	0.12
Silt	**0.30**	0.14
Coarse Particulate organic Matter	−0.23	**0.72**
Current velocity	**−0.63**	0.01
F value and significance, Monte Carlo test		
First axis 1.126, *P* value = 0.0037		

	CCA	
	Axis 1	Axis 2
(b) Plant species from the riparian		
Eigenvalue	0.435	0.256
Species-environmental correlations	0.987	0.973
Cumulative percentage variance		
Of species data	26.9	46.7
Of species-environmental relation	38.7	61.4
Correlations		
Discharge	**−0.85**	−0.19
Floodplain morphology	**0.31**	**0.64**
Naturalness of the river channel	**0.40**	**−0.75**
Air temperature	**−0.60**	0.28
% Conservation I	**0.52**	**−0.31**
% Urban	**−0.40**	**0.90**
F value and significance, Monte Carlo test		
First axis 1.840, *P* value = 0.003		
All axes *F* ratio = 1.905 *P* value = 0.0005		

Although at present the risk of soil erosion for pluvial precipitation at a basin scale is low (less than 10 % of the basin suffering moderate to high risk of soil erosion), this has the potential to change greatly under current forest protection laws. As we demonstrated, if forest conversion rates increase at current rates due to agricultural and livestock practices, the risk of soil erosion will increase to encompass 28 % of the total river basin. These results highlight the vulnerability of the Percy watershed and the potential ecological and economic implications of unsustainable land use. In this scenario, the increase in the severity of fine sediment pulses on macroinvertebrates survival and sensitive species extirpation could be a major threat. Although total suspended solids were not measured, records during flooding after rains at low part of Percy River can be very high (>1200 mg L^{-1} Miserendino, personal observation). These values are much higher than those proposed as critical to protect stream invertebrate populations in other countries (<5 mg L^{-1} in New Zealand, 30 up to 158 mg L^{-1} USA) (Harding et al. 2000; EPA 2003). Sedimentation effects on benthic communities have been widely recorded at urban streams (Miserendino et al. 2008). Moreover, this phenomenon was also associated with decreased areas for fish spawning and foraging (Di Prinzio et al. 2009).

From the headwater to downstream, the environmental changes documented along the system were linked to impoverished habitat condition and riparian ecosystem status. As anticipated in earlier reports (Pizzolon et al. 2001; Bauer 2010), the urbanized section displayed incipient eutrophication (higher levels of nutrients) and marked alteration of banks due to human intervention. Modification of the riparian vegetation at lower sites was practically complete, but at sites located in the upper basin section (sites: 3–6) was also striking, with *Salix fragilis* and *S. alba,* notoriously invasive species for the region, replacing the native *Nothofagus antarctica*. A recent study reported an unusual ability of the invasive willows (*S. alba-fragilis* complex) to replace native species, with a downstream-directed invasion process in the Negro River (Northern Patagonia) (Thomas and Leyer 2014). This phenomenon is of great concern in the riverine landscapes in Patagonia because the effects of this invasion are not completely understood.

Riparian plant distribution was strongly linked to discharge and air temperature but also with attributes of the floodplain and river channel affected by anthropogenic interventions (dredging, channelization, bank disruption, etc.) (Appendix Table 6). From a landscape perspective, upstream catchment areas having urbanization and high level of conservation of the forest appeared also as strong predictors. Our results suggest that riparian communities change in structure and composition along natural downstream gradients, but that elevation and land use interact to influence the invasion of non-native species along the river corridor (Pauchard and Alaback 2004).

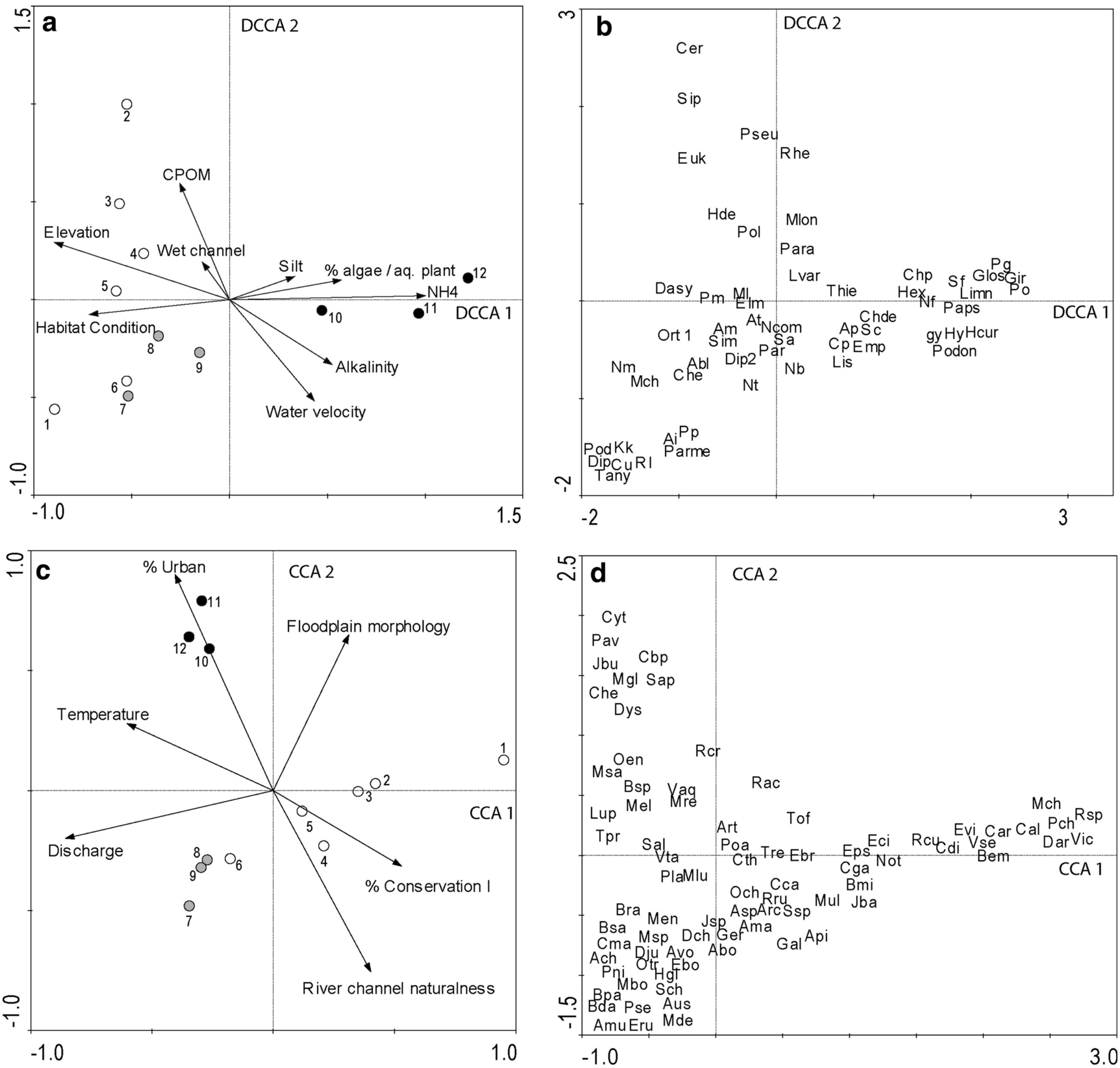

Fig. 6 **a** Biplot of the Percy River sampling sites and environmental variables in relation to the first ordination axes of DCCA based on abundance data of benthos community (58 species). **b** Ordination of macroinvertebrate species with regards to the first environmental axes of DCCA. **c** Canonical correspondence analysis displaying studied site and environmental variables based on coverage data of plants species. **d** Ordination of plant species with regards to the first environmental axes of CCA. Code for sites at Percy basin as follows: *open circles*: upper, *grey circles*: middle, and *black circles*: lower. Codes for plants and macroinvertebrates species are listed in Appendix Tables 4 and 5, respectively

The replacement of native species by exotic vegetation in the riparian corridors of Patagonian mountain streams has influenced changes in the functional attributes of the benthic communities (Miserendino and Pizzolón 2004; Miserendino and Masi 2010; Miserendino et al. 2011) and in the quality and quantity of detrital fractions (Masi and Miserendino 2009). According to Hughes et al. (2010), the riparian vegetation in river systems has a multifunctional role (e.g., flood prevention, sediment retention, seed recruitment, provision of shelter, habitat heterogeneity, allochthonous input) and is vital to the integrity of river ecosystems. The removal or disturbance of riparian vegetation can also facilitate the invasion of terrestrial and aquatic alien species (Richardson et al. 2010).

Disturbed areas in the Percy River, as indicated by soil alteration in the river margins, were dominated by the exotics *Carduus thoermeri*, *Trifoliums repens*, *Melilotus alba, Medicago sativa*, *Rumex acetossella*, and *Rosa rubiginosa* in the herbaceous and shrub stratum. These species have several dispersal mechanisms that allow them to propagate and to colonize heavily modified environments (Mack et al. 2000).

Biological Measures and Anthropogenic Pressures

Patterns observed in several macroinvertebrate metrics corresponded more to changes in the longitudinal dimension (altitudinal gradient) than to disturbance or anthropogenic causes (Miserendino and Pizzolón 2000). Even those metrics that showed significant variation and expected patterns to disruption did not show strong relationships (e.g., collector, gatherers, and Annelida). Assemblages of sensitive species (Barbour et al. 1999; Lussier et al. 2008), Plecoptera (*Notoperlopsis femina*, *Antarctoperla michaelseni*, and *Potamoperla myrmidon*), Ephemeroptera (*Meridialaris laminata*, *Andesiops peruvianus*, and *A. torrens*), and Trichoptera (*Mastigoptila longicornuta*, *Cailloma pumida*, *Smicridea annulicornis,* and *S. frequens*), were still found at urbanized sites. Although nutrient levels were several times higher in lower river sites compared to upper, it is likely that the river still has an adequate water flow, good levels of oxygen saturation, and heterogeneous habitats available given the aquatic plants and algae present. Probably the urban impacts at Trevelin town are mostly local and relatively small in comparison to the status of the overall watershed. This situation could potentially be described as of moderate or intermediate impact, resulting in increases of macroinvertebrate richness and diversity. The intermediate disturbance hypothesis (Connell 1978; Ward and Stanford 1983) predicts the response of species richness patterns along a gradient of natural disturbance varying in intensity or frequency. In this sense, taxon richness peaks at intermediate levels of disturbance, in which the presence of refugia and food sources (detrital, algae), each likely to be used by different sets of species, must be responsible for species resilience (Townsend and Scarsbrook 1997).

For riparian plant communities, the metrics of total species richness, proportional coverage by exotics, and species diversity discriminate disturbed sites from undisturbed. These results are in line with Bonanno and Lo Giudice's (2010) findings, who documented a decline in floristic index values (based in riparian plant composition) at the Imera River (Sicily) as a response to increased anthropogenic activities (e.g., grazing, tourism). The higher contribution of exotic species at the lower section of Percy River support the idea that urbanization is one of the main factors that favors the distribution of non-native species on riverine landscapes (Lopez and Fennessy 2002). Some detrimental interventions such as dredging and hard structure construction can severely affect the floristic integrity in a river, even when other measures indicate good water quality at each reach. This suggests that although disturbance occurred within the watershed, water quality and riparian plant community structure responded to this broad scale disturbance independently.

Conclusions and Recommendation for Rehabilitation Measures

Multidisciplinary assessment approaches that consider the interactive components of healthy river ecosystems are critical for selecting and implementing appropriate restoration measures (Hughes et al. 2010). In this context, our work attempted to provide a more integrative picture of the ecological status of the Percy River basin. The scientific information obtained in this study has application to a much larger regional area. Species distribution patterns and human development pressures are likely very similar at several basins in the mountainous area in Patagonia.

In the Percy basin, mitigation efforts should be directed to:

(1) Promote a territorial planning directed to conservation of certain areas: not only to protect the headwater forest, but supervise the practices allowed in the basin area having moderate conservation of the forest, at present mostly occupied by *N. antarctica.* Our analysis of soil erosion risk evidenced that this area appears as the most vulnerable to change in the future, where a significant loss of soils is predicted according to the model. Most previous management plans have failed because wood collection practices were not always well controlled. Distribution of these areas (level 2) overlaps with lands subjected to pasture practices, very susceptible to forest fires, and the recolonization process is usually compromised by grazing. Another emergent problem is the rapid urban expansion at rural and suburban areas at the lower basin (Trevelin), which could result in dramatic changes in the vegetation cover.

(2) Restore altered riparian ecosystem and enhance habitat condition: by establishing some control actions on the invasive *Salix,* re-implanting natives in some key segments of the river, and by limiting or avoiding the dredging action especially in the middle and lower basin. The aesthetical value of these areas is a matter of concern too, because some sections used as recreational areas in the past are missing due to marked human interventions. The benefits derived

from maintaining the riparian ecosystem in good condition are very valuable for citizens.

(3) Control the eutrophication process in the lower part of the basin: these actions should involve the water authority commission from the adjacent urban centers: Esquel, which contributes with their treated effluents on the Esquel Stream, and Trevelin, both towns are experiencing an accelerated process of urban expansion.

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Appendix

See Tables 4, 5 and 6.

Table 4 Distribution and relative abundance (%) of riparian plants per site, total richness, and coverage of exotics at Percy River basin during the study (February 2011), Chubut, Argentina

Taxa	Code	Percy River basin											
		Upper				Middle				Lower			
		1	2	3	4	5	6	7	8	9	10	11	12
Equisetopsida													
Equisetaceae													
Equisetum bogotense	*Ebo*	0	0	0	0	0	1	0	0	0	0	0	0
Gymnospermae													
Cupressaceae													
Austrocedrus chilensis	*Aus*	0	0	0	0	0	2	20	0	0	0	0	0
Monocotyledonae													
Cyperaceae													
Carex gayana	*Cga*	0	0	1	0	0	0	0	0	0	0	0	0
Eleocharis pseudoalbibracteata	*Eps*	0	0	5	0	0	0	0	0	0	0	0	0
Juncaceae													
Juncus balticus	*Jba*	0	2	0	0.5	0	0	0	0	0	0	0	0
Juncus burkartii	*Jbu*	0	0	0	0	0	0	0	0	0	0	0	5
Juncus sp.	*Jsp*	0	0	8	0	0	0	0.5	8	0	0	0	0
Dicotyledoneae													
Anacardiaceae													
Schinus patagonicus	*Sch*	0	0	0	1	2	10	8	2	1	0	0	0
Apiaceae													
*Conium maculatum**	*Cma*	0	0	0	0	0	0	0	1	0	0	0	0
Mulinum spinosum	*Mul*	5	1	1	2	2	5	0	2	0	0	0	0
Osmorhiza chilensis	*Och*	0	0	0	0	1	0	0	0	0	0	0	0
Asteraceae													
*Achillea millefolium**	*Ach*	0	0	0	0	0	0	0	1	1	0	0	0
*Arctium minus**	*Arc*	0	0	0	0	1	0	0	0	0	0	0	0
*Artemisia absinthium**	*Art*	0	5	2	1	1	0	0	2	1	2	2	0
Baccharis patagonica	*Bpa*	0	0	0	0.5	0	0	10	10	0	0	0	0
Baccharis racemosa	*Bra*	0	0	0	0	1	0	0	15	6	1	0	0
Baccharis salicifolia	*Bsa*	0	0	0	0	0	5	0	0	10	1	0	0
*Carduus thoermeri**	*Cth*	0	2	2	2	0	0	0	5	0	3	0	0
Chiliotrichum diffusum	*Cdi*	3	6	0	0	0	0	0	0	0	0	0	0
*Crepis capillaris**	*Cca*	2	6	3	3	2	0	2	2	1	0	2	0
Haplopappus glutinosus	*Hgl*	0	0	0	1	1	1	5	4	4	0	0	0
*Matricaria recutita**	*Mre*	0	3	4	8	5	4	0.5	2	2	15	15	25
Mutisia decurrens	*Mde*	0	0	0	0	0	0	1	0	0	0	0	0

Table 4 continued

Taxa	Code	Percy River basin											
		Upper				Middle				Lower			
		1	2	3	4	5	6	7	8	9	10	11	12
Mutisia spinosa	*Msp*	0	0	0	0	0	2	0	0	0	0	0	0
Senecio sp.	*Ssp*	2	3	1.5	1	2	1	0.5	2	1	0	0	0
*Taraxacum officinalis**	*Tof*	0	2	0.5	1	0	0	0	0	0	0	1	0
Berberidaceae													
Berberis darwinii	*Bda*	0	0	0	0	0	0	10	0	2	0	0	0
Berberis empetrifolia	*Bem*	2	2	0	0	0	0	0	0	0	0	0	0
Berberis microphylla	*Bmi*	5	25	10	5	8	0	0	1	0	0	0	0
Boraginaceae													
Phacelia secunda	*Pse*	0	0	0	0	0	1	5	0	0	0	0	0
Brassicaceae													
Brassica sp.*	*Bsp*	0	0	0	0	0	0	0	2	0	1	1	0
*Capsella bursa-pastoris**	*Cbp*	0	0	0	0	0	0	0	0	0	1	0	0
Calceolariaceae													
Calceolaria sp.	*Cal*	0.5	0.1	0	0	0	0	0	0	0	0	0	0
Caryophyllaceae													
*Cerastium arvense**	*Car*	1	0.5	0	0	0	0	0	0	0	0	0	0
*Saponaria officinalis**	*Sap*	0	0	0	0	0	0	0	0	0	2	0	0
Celastraceae													
Maytenus boaria	*Mbo*	0	0	0	0	0	2	1	0	0	0	0	0
Maytenus chubutensis	*Mch*	1	0	0	0	0	0	0	0	0	0	0	0
Chenopodiaceae													
*Chenopodium album**	*Che*	0	0	0	0	0	0	0	0	0	2	0	0
*Dysphania multifida**	*Dys*	0	0	0	0.5	0	0	0	0	1	2	8	1
Escalloniaceae													
Escallonia rubra	*Eru*	0	0	0	0	0	0	15	0	0	0	0	0
Escallonia virgata	*Evi*	30	7	1	0	0	0	0	0	0	0	0	0
Fabaceae													
Adesmia boronioides	*Abo*	0	0	0	1	1	0.5	0	0	1	0	0	0
Adesmia volckmannii	*Avo*	0	0	0	2	3	10	0.5	1	3	0	0	0
*Cytisus scoparius**	*Cyt*	0	0	0	0	0	0	0	0	0	0	2	0
Geraniaceae													
*Erodium cicutarium**	*Eci*	0	7	0	0	0	0	0	0	0	0	0	0
Geranium sp.	*Ger*	0	0	0	0.5	2	0	0.5	0	0	0	0	0
Fabaceae													
*Lupinus polyphyllus**	*Lup*	0	0	0	0	0	0	8	1	15	10	5	5
*Medicago lupulina**	*Mlu*	0	3	0	4	2	2	5	5	0	2	0	5
*Medicago sativa**	*Msa*	0	0	0	0	0	0	5	2	15	15	10	15
*Melilotus albus**	*Mel*	0	0	0	0	20	8	0.5	15	20	25	25	35
*Trifolium pratense**	*Tpr*	0	0	0	0	0	0	0	3	0	0	0	2
*Trifolium repens**	*Tre*	15	5	8	2	1	1	10	10	0	2	0	5
Vicia nigricans	*Vic*	3	0	0	0	0	0	0	0	0	0	0	0
Grossulariaceae													
Ribes cucullatum	*Rcu*	5	5	2	1	0	0	0	0	0	0	0	0
Lamiaceae													
Mentha sp.*	*Men*	0	0	0	0	0.5	0	0	2	0	0	0	0

Table 4 continued

Taxa	Code	Percy River basin											
		Upper				Middle				Lower			
		1	2	3	4	5	6	7	8	9	10	11	12
Nothofagaceae													
Nothofagus antarctica	*Not*	60	40	30	5	10	3	0	0	0	0	0	0
Onagraceae													
*Epilobium brachycarpum**	*Ebr*	2	12	1	2	2	0	1	0	2	2	0	0
Oenothera odorata	*Oen*	0	0	0	0	0	0	0	0	1	3	0	0
Phrymaceae													
Mimulus glabratus	*Mgl*	0	0	0	0	0	0	0	0	0	1	0	0
Plantaginaceae													
*Plantago lanceolata**	*Pla*	0	5	3	3	4	3	5	8	5	3	3	2
*Veronica anagallis-aquatica**	*Vaq*	3	0	0	0	0	2	0	3	2	5	5	2
*Veronica serpyllifolia**	*Vse*	2	0	0	0.5	0.1	0	0	0	0	0	0	0
Poaceae		15	15	20	5	4	5	15	5	2	10	5	10
Polygonaceae													
*Polygonum aviculare**	*Pav*	0	0	0	0	0	0	0	0	0	0	2	2
*Rumex acetosella**	*Rac*	2	15	2	1	2	0	0	0	0	2	3	5
*Rumex crispus**	*Rcr*	0	0	1	0	0	0	0	0	0	2	0	0
Ranunculaceae													
Anemone multifida	*Amu*	0	0	0	0	0	0	0.5	0	0	0	0	0
Ranunculus sp.	*Rsp*	5	0	0	0	0	0	0	0	0	0	0	0
Rhamnaceae													
Discaria articulata	*Dar*	30	0	0	0	0	0	0	0	0	0	0	0
Discaria chacaye	*Dch*	0	0	20	10	0	5	2	20	2	0	0	0
Ochetophila trinervis	*Otr*	0	0	0	0	15	20	25	30	8	0	0	0
Rosaceae													
Acaena magellanica	*Ama*	1	8	5	3	0	2	5	5	0	0	0	0
Acaena pinnatifida	*Api*	0	0	0	1	0	0	0	0	0	0	0	0
Acaena splendens	*Asp*	0	6	2	0	5	0	0	5	2	0	0	0
Potentilla chiloensis	*Pch*	3	0	0	0	0	0	0	0	0	0	0	0
*Rosa rubiginosa**	*Rru*	0	0	0	0	2	0	0	0	0	0	0	0
Rubiaceae													
Galium sp.*	*Gal*	0	0	0	0.5	0	0	0	0	0	0	0	0
Salicaceae													
*Populus nigra**	*Pni*	0	0	0	0	0	0	0	0	1	0	0	0
Salix sp.*	*Sal*	0	0	45	25	50	30	40	40	20	40	50	80
Scrophulariaceae													
*Verbascum thapsus**	*Vta*	0	2	0	2	0.5	3	0.5	0	5	1	1	1
Verbenaceae													
Diostea juncea	*Dju*	0	0	0	0	0	0.5	0	0	0	0	0	0
Species richness		23	26	24	31	29	26	29	31	27	25	17	16
Exotic species richness		7	12	11	15	15	8	11	17	14	20	16	14
Exotic species percentage		32	48	48	50	54	32	39	57	54	84	100	94

* Means exotic origin, rest all others. Codes used in the multivariate analysis are consigned

Table 5 Mean density of macroinvertebrate taxa (ind m^{-2}), total density, and total richness per site, at Percy River basin during the study (February 2011), Chubut, Argentina

Taxa	FFG	Code	Percy River basin											
			Upper				Middle				Lower			
			1	2	3	4	5	6	7	8	9	10	11	12
Platyhelminthes														
Girardia sp.	P	*Gir*	0	0	0	0	0	0	0	0	0	0	0	5.6
Cura sp.	P	*Cu*	1.9	0	0	0	0	0	0	0	0	0	0	0
Annelida														
Lumbriculus variegatus	CG	*Lvar*	0	35.6	38.9	0	2.2	2.2	31.1	2.2	8.9	16.7	96.7	27.8
Nais communis	CG	*Ncom*	35.6	5.6	0	11.1	0	0	0	0	0	535.5	0	0
Limnodrilus spp.	CG	*Limn*	0	0	0	0	0	0	0	1.9	0	0	1079.9	508.8
Glossiphoniidae spp.	P	*Glos*	0	0	0	0	0	0	0	0	0	0	1.9	8.9
Mollusca														
Chilina patagonica	GS	*Chp*	0	1.9	0	0	0	0	1.9	0	0	0	1.9	11.1
Arthropoda														
Crustacea														
Hyalella curvispina	CG	*Hcur*	0	0	0	0	0	0	0	0	0	0	1.9	0
Insecta														
Plecoptera														
Potamoperla myrmidon	GS	*Pm*	0	285.5	635.5	357.7	733.3	427.7	713.3	272.2	166.7	124.4	63.3	7.8
Notoperlopsis femina	GS	*Nf*	0	0	0	0	0	0	0	0	11.1	407.7	470	307.7
Notoperla magnaspina	GS	*Nm*	7.4	1.9	0	0	0	0	0	0	0	0	0	0
Aubertoperla illiesi	GS	*Ai*	116.7	0	0	0	0	0	0	0	0	0	0	3.3
Antarctoperla michaelseni	S	*Am*	63.3	36.7	18.9	11.1	33.3	20	44.4	42.2	100	70	44.4	5.6
Pelurgoperla personata	CG	*Pp*	0	0	0	0	0	0	5.6	0	0	0	0	0
Klapopteryx kuscheli	S	*Kk*	13.3	0	0	0	0	0	0	0	0	0	0	0
Pictetoperla gayi	P	*Pg*	0	0	0	0	0	0	0	0	0	0	0	1.9
Ephemeroptera														
Meridialaris chiloeensis	GS	*Mch*	311.1	24.4	0	0	24.4	27.8	105.5	0	0	0	0	0
Meridialaris laminata	GS	*Ml*	0	88.9	320	152.2	175.5	225.5	233.3	281.1	355.5	66.7	218.9	105.5
Andesiops peruvianus	GS	*Ap*	1.9	1.9	3.3	22.2	5.6	5.6	47.8	0	146.7	57.8	130	397.7
Andesiops torrens	GS	*At*	147.8	307.7	150	281.1	485.5	453.3	353.3	368.9	303.3	1107.7	796.6	1324.3
Chiloporter eatoni	P	*Che*	1.9	0	0	0	0	0	0	24.4	0	0	0	0
Siphlonella sp.	CF	*Sip*	0	16.7	3.3	2.2	0	0	0	0	0	0	0	0
Trichoptera														
Mastigoptila longicornuta	SG	*Ml*	0	30	5.6	2.2	2.2	0	0	0	0	16.7	24.4	2.2
Rheochorema lobuliferum	P	*Rl*	2.2	0	0	0	0	0	0	0	0	0	0	0
Cailloma pumida	P	*Cp*	0	0	2.2	2.2	13.3	11.1	18.9	18.9	20	38.9	64.4	58.9
Neoatopsyche brevispina	P	*Nb*	0	0	0	0	0	0	3.3	7.8	3.3	2.2	7.8	0
Neopsilochorema tricarinatum	P	*Nt*	0	0	0	0	0	0	0	0	1.9	0	0	0
Smicridea annulicornis	CF	*Sa*	0	8.9	75.5	86.7	172.2	266.6	516.6	627.7	613.3	892.1	1461.0	85.5

Table 5 continued

Taxa	FFG	Code	Percy River basin											
			Upper				Middle				Lower			
			1	2	3	4	5	6	7	8	9	10	11	12
Smicridea frequens	CF	*Sf*	0	0	0	0	0	0	0	0	0	13.3	977.7	370
Parasericostoma ovale	S	*Po*	0	0	0	0	0	0	0	0	0	0	0	1.9
Coleoptera														
Hemiosus dejeani	P	*Hde*	0	5.6	2.2	0	0	2.2	0	0	2.2	2.2	0	0
Gyrinidae sp.	P	*Gy*	0	0	0	0	0	0	0	0	0	0	2.2	0
Hydrophilidae sp.	P	*Hy*	0	0	0	0	0	0	0	0	0	0	2.2	0
Elmidae spp.	GS	*Elm*	42.2	80	22.2	5.6	25.6	5.6	47.8	36.7	74.4	24.4	55.6	55.6
Diptera														
Rheotanytarsus sp.	CG	*Rhe*	0	14.4	0	0	0	0	0	0	0	0	5.6	2.2
Chironomus decorus	CG	*Chde*	0	0	0	0	0	0	0	0	0	8.9	0	0
Polypedilum sp.	CG	*Pol*	0	63.3	0	52.2	5.6	7.8	2.2	0	0	42.2	3.3	2.2
Orthocladiinae sp.	CG	*Ort1*	11.1	7.8	8.9	27.8	80	31.1	11.1	42.2	35.6	0	0	0
Parametriocnemus sp.	CG	*Par*	0	0	0	0	0	0	7.8	0	0	0	0	0
Thienemanniella sp.	CG	*Thie*	5.6	5.6	2.2	0	18.9	0	0	0	0	5.6	27.8	14.4
Pseudosmittia sp1	CG	*Pseu*	0	13.3	0	0	2.2	0	0	0	0	0	0	3.3
Eukiefferiella sp.	CG	*Euk*	0	274.4	92.2	75.5	7.8	0	0	3.3	0	0	0	0
Paratrichocladius sp.	CG	*Par*	63.3	30	35.6	30	25.6	57.8	136.7	192.2	42.2	88.9	2319.8	0
Parapsectrocladius sp.	CG	*Parps*	0	0	0	0	0	0	0	0	0	75.5	619.9	142.2
Paramerina sp.	P	*Para*	0	113.3	24.4	27.8	11.1	5.6	2.2	7.8	13.3	18.9	72.2	70
Ablabesmyia sp.	P	*Abl*	0	0	0	0	0	0	0	1.9	0	0	0	0
Tanytarsus sp.	P	*Tany*	2.2	0	0	0	0	0	0	0	0	0	0	0
Podonomus sp.	CF	*Pod*	2.2	0	0	0	0	0	0	0	0	0	0	0
Podonominae sp.	CF	*Podon*	2.2	0	0	0	0	0	0	0	0	0	330	0
Empididae spp.	P	*Emp*	2.2	0	0	0	0	0	0	3.3	2.2	3.3	3.3	5.6
Ceratopogonidae sp.	P	*Cer*	0	5.6	0	0	0	0	0	0	0	0	0	0
Simulium spp.	CF	*Sim*	88.9	25.6	36.7	31.1	116.7	225.5	236.6	33.3	7.8	127.8	35.6	33.3
Hexatoma spp.	P	*Hex*	0	11.1	0	0	0	2.2	3.3	2.2	5.6	24.4	85.5	188.9
Diptera sp.1	P	*Dip*	2.2	0	0	0	0	0	0	0	0	0	0	0
Diptera sp.2	P	*Dip2*	0	0	0	0	2.2	0	2.2	0	0	2.2	0	0
Lispoides sp.	P	*Lis*	0	0	0	0	0	2.2	0	0	0	2.2	2.2	0
Dasyoma sp.	P	*Dasy*	61.1	64.4	114.4	63.3	50	25.6	55.6	44.4	50	0	0	2.2
Sciomyzidae sp.	P	*Sc*	0	0	0	0	0	0	0	0	0	1.9	0	0
Total density			983.2	1557	1592	1241	1992	1805	2581	2015	1968	3776	9003	3755
Total richness			22	27	19	18	21	19	22	20	20	27	30	29

FFG functional feeding groups, *S* shredders, *GS* grazer/scrapers, *CF* collector filterers, *CG* collector gatherers, *P* predators. Codes used in the multivariate analysis are consigned

Table 6 Status of conservation (percentage of basin area) upstream of each site (1: high, 2: moderate, 3: low or none), component values of QBRp and habitat condition indexes at Percy River sites during the study (February 2011), Chubut, Argentina

	Land cover metrics				QBRp metrics			
Sites:	% Conservation 1	% Conservation 2	% Conservation 3	% Urban	Total cover	Structure	Complexity and naturalness of the vegetation	River channel naturalness
1	37	29.8	0	0	25	20	25	25
2	14.5	54.8	0	0	17	15	20	25
3	13.3	56.1	0	0	22	15	17.5	25
4	18.2	49.1	0	0	12	10	10	25
5	18.3	48.4	0	0	17	12	10	25
6	18.8	45.4	0	0	13	14	17.5	25
7	18	44.5	0.7	0	22	20	17.5	25
8	17.7	43.5	0.7	0	18	20	15	20
9	17.5	43.7	0.7	0	17	10	5	25
10	14.0	33	1	0.9	10	12	5	10
11	13.8	32.6	1	1.1	10	12	2.5	10
12	13.7	32.3	1	1.1	15	15	5	20

	Habitat condition index									
Sites:	Availability of substrate	Embeddedness	Velocity and depth combinations	Sediment deposition	Channel flow status	Channel alteration	Frequency of riffles/ Channel sinuosity	Bank stability	Bank vegetative protection	Riparian vegetative zone width
1	18	20	20	18	20	18	20	18	18	19
2	15	14	20	14.5	16.5	14.5	18	11	10	10
3	16	13.5	20	17	19.5	19.5	20	18	17	17
4	19	16	20	18	19.5	20	20	11.5	6	3
5	18	18	20	18	19	18	20	13	9	6
6	16.5	14	20	12	18	14	19	13	9	6
7	20	19	20	19	20	20	20	18	15	15
8	20	18	20	17	19	18	20	16	14	9
9	18	16	20	14	18	18	14	14	9	13
10	14.5	18	19.58	16	17	7	13	2	6	5
11	14	16	13	11	17	7	6	2	6	5
12	11	18	18	15	18	19	14	18	11	17

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