The effects of land use on environmental features and functional organization of macroinvertebrate communities in Patagonian low order streams

M. Laura Miserendino [a,*], Carolina I. Masi [b]

[a] *Laboratorio de Investigaciones en Ecología y Sistemática Animal (LIESA), Universidad Nacional de la Patagonia, Sede Esquel, Sarmiento 849, 9200 Esquel, Chubut, Argentina*
[b] *IADIZA - Instituto Argentino de Investigaciones de las Zonas Àridas, CRICYT, Mendoza, Argentina*

A B S T R A C T

Benthic invertebrates, water quality variables, chlorophyll *a* and particulate organic matter (POM) were studied in 18 sites of mountain streams in Patagonia (Argentina) subjected to six different land uses: native forest, pine plantation, pasture, harvest forest, urban and reference urban. Three streams of each land use were studied in March 2006. Macroinvertebrates were sampled in three riffles and three pools (*n* = 108) and biomass of detrital fractions of POM were quantified. Overall benthic POM biomass was higher at native and harvest forest than pastures, whereas fine fraction (FPOM) was higher in harvest forest than in pastures. Regarding to autotrophic subsidies bryophytes were the only that changed consistently among uses. These differences in energy resources were correlated with changes in community attributes. Shredder richness was clearly higher at native and harvest forest than exotic pine plantations, collector gatherers density was consistently high at harvest sites, and total density was significantly higher at urban and harvest forest. Multidimensional scaling ordination based on macroinvertebrate density data showed a clear separation of forested (either native or exotic species) from riparian modified areas (pasture, urban and harvest sites). Environmental variables having explanation power on macroinvertebrate assemblages were mostly related with: detritus availability (wood and leaves biomass) and impairment (total phosphorous and % sand). These results confirm that macroinvertebrate assemblage structure in Patagonian low order streams can be altered by land use practices. Among guild structure measures, those indicators based on benthic community functional attributes, shredders richness and collectors density, resulted good candidates to assess land use impacts. On account of riparian corridor management may be critical to the distribution of benthic taxa, the maintenance of good conditions of vegetation adjacent to rivers will enhance water quality and the environment for highly endemic headwater communities of Patagonian streams.

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1. Introduction

Land use changes have resulted in profound modifications of running waters in mountain streams of temperate areas (Allan et al., 1997). In developed basins the conversion of riparian margins from forest to grassland has been one of the most widespread phenomena. Moreover, shifts in land use practices (agriculture to silviculture) have also determined the replacement of native by exotic vegetation as for example pine plantations, sometimes affecting riparian corridors (Baillie and Davies, 2002; Albariño and Balseiro, 2002). Changes in the adjacent vegetation can alter the complex biotic and abiotic processes that govern the function and attributes of aquatic communities including benthic macroinvertebrates (Roy et al., 2003; Thompson and Towsend,

2004; Utz et al., 2009). Early works revealed the consequences that the loss of riparian vegetation can have on physical properties of streams (Newbold et al., 1980). The extirpation of riparian corridors can alter the light and temperature regimes and lead to increases in nutrients and augment primary production (Scarsbrook and Halliday, 1999; Kratzer et al., 2006). The establishment of pastures for agricultural grazing can significantly reduce the amount of allochthonous leaf material entering the river, which consecutively produces a significant effect on trophic structure (Abelho and Graça, 1998; Miserendino, 2005). Scrimgeour and Kendall (2003) found that shredder biomass was significantly lower in Canadian streams under intense pasture development than in those without pastoral use. Loss of insect species (Collier et al., 2001) and higher invertebrate densities have been documented in New Zealand streams in which forest was converted to pasture (Quinn et al., 1997). Thompson and Towsend (2004) observed significant differences between shredders and collector filterer biomass in basins under different land uses such as pasture conversion, native, or exotic vegetation. In a recent

* Corresponding author. Tel.: +54 2945451165; fax: +54 2945452271.
E-mail addresses: lmiserendino@liesa.org.ar (M.L. Miserendino),
cmasi@lab.cricyt.edu.ar (C.I. Masi).

study of 60 sites in streams subjected to different land uses Kratzer et al. (2006) found that macroinvertebrate communities responded considerably to land use and most species variation was accounted by variables related to watershed and riparian conditions. Similarly, Compin and Céréghino (2007) detected significant responses of functional feeding groups to landscape alterations. They recognized macroinvertebrate trophic measurements useful for assessing river health with development along riparian zones with collector gatherers showing higher percentages in urban landscapes and scrapers and predators increasing in agricultural landscapes.

The functional structure along streams in Patagonian mountains seems to correspond well with the predictions of the RCC (River Continuum Concept, Vannote et al., 1980) with biomass of shredders decreasing and collector's biomass increasing downstream from the headwaters (Miserendino and Pizzolón, 2003; Miserendino, 2007). Some differences in the functional attributes of benthic communities between rivers bordered by the native Nothofagus forest and those dominated by Salix spp. have been

observed, the former showed higher biomass of shredders while Salix bordered rivers had dominance of collector and filterers (Miserendino and Pizzolón, 2003, 2004). In recent experimental studies, Albariño and Balseiro (2002) suggested that the litter processing mechanisms would be affected as a consequence of the substitution of native Nothofagus forest by exotic pine plantations.

Streams in Patagonian mountains are subjected to different land use practices; the most common are pasture conversion, exotic plantations, and wood collection/deforestation. The consequences that these activities are having on functional structure of the local, highly endemic biota are not explicitly known. A better understanding of these relationships may contribute to the development of improved land use management practices and to purpose mitigation plans on freshwater ecosystems (Morley and James, 2002; Danger and Robson, 2004).

The aim of this study was to assess the impacts of different land use practices (native forest, exotic, pasture, wood collection, reference urban and urbanizations) on physicochemical characteristics and macroinvertebrate community, and also to evaluate
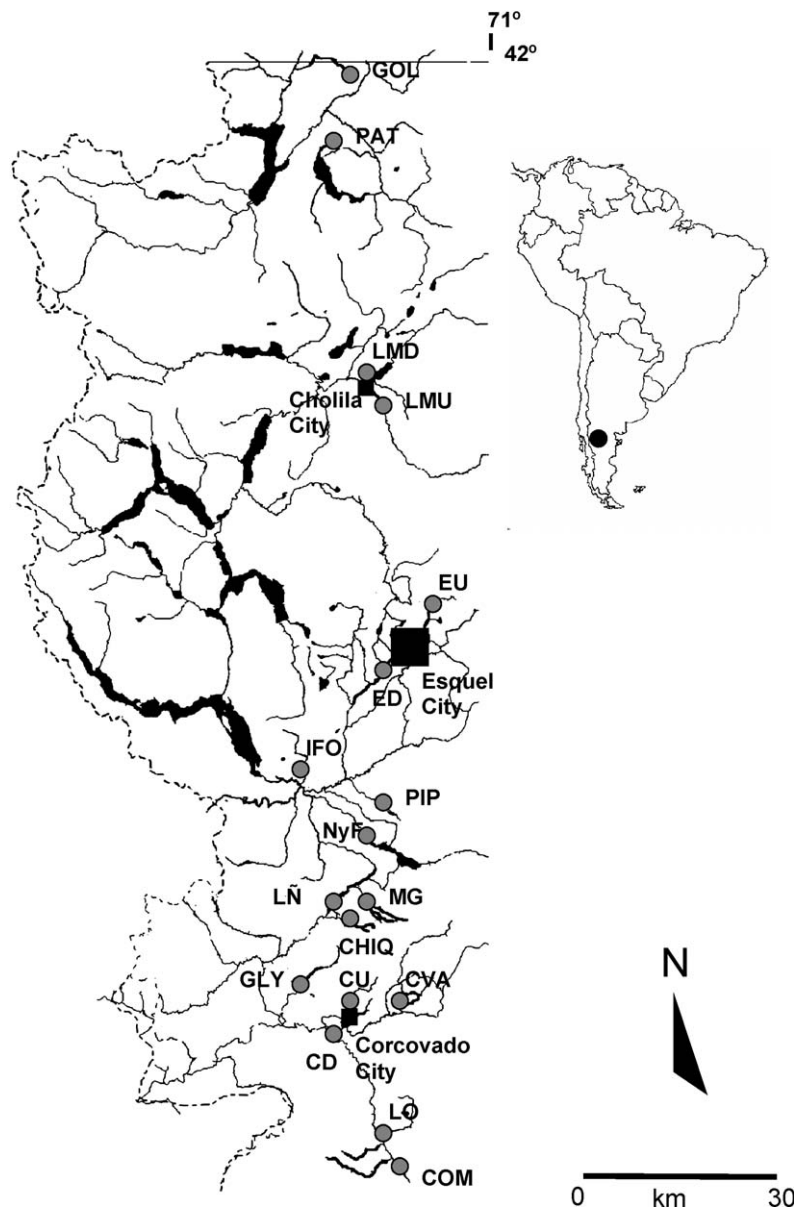

Fig. 1. Map of study area with locations of sampling points, Chubut, Argentina. Site codes are shown in Table 1.

the usefulness of guild structure indicators in mountain streams of northwest Patagonia (Argentina).

2. Materials and methods

2.1. Study area and site selection

The study area is located in the Northwest of Chubut Province (Fig. 1) in the ecotone between the Subantarctic forest and the Patagonian steppe, and exhibits a marked altitudinal gradient. Perennial (Austrocedrus chilensis, Nothofagus dombeyi and Maitenus boaria) and deciduous tree species (Nothofagus pumilio, Nothofagus anctarctica) constitute the Subantarctic forests. The shrub and herbaceous strata are characterized mainly by Chusquea culeou, Berberis buxifolia, and Fuchsia magellanica. The valleys are dominated by the herbaceous, shrub-like steppe vegetation (Tell et al., 1997). Exotic vegetation is composed mainly by the implanted forest of Pseudotsuga menziesii, Pinus ponderosa, Pinus radiata and Pinus lambertiana, in many cases species are included in the riparian corridor and replacing most of the native species.

The survey was conducted on March 2006 on 18 sites in 15 rivers. We choose this month because most streams have stable flow conditions (Coronato and del Valle, 1988). Three sampling sites were placed on non-impacted native Nothofagus forest (Chiquito, Comisario, and Loro), these sites were considered as reference sites to test for impacts: pasture, pine plantation and wood collection.

Three sites on streams with native N. antarctica and N. pumilio forest with wood collection or extraction (Glyn, Cabeza de vaca, Pipo) were selected. At these sites the harvest was achieved during previous spring (3 months before the survey). The logging operations imply a selective cut of trees, which should exclude specimens from the river corridor. Operators also have to keep free the streams of debris and branches once the tree is logged. However during the survey we observed some river corridors having logged trees, and some watercourses obstructed with debris.

For pasture sites we selected three sites that had been cleared from the native Nothofagus forest and had been grazed intensively (Nant y Fall, Manguera and Los Ñires). At the present pasture sites sustain cattle livestock mainly cow and sheep. The pine plantations sites corresponded to streams running (950–1200 m) through plantations of Pseudotsuga menziesii, Pinus ponderosa, P. radiata and P. lambertiana (Golondrinas, Patriada and Ifona). Three urban rivers (Las Minas, Carbón and Esquel) plus their respective reference sites (Las Minas ref, Carbón ref and Esquel ref) were also included in the survey (Fig. 1).

In order to know basin features, vegetation types and coverage, a LandSat image was analyzed and processed in the Dirección General de Bosques y Parques. Per each surveyed site we obtained sub-basin area, % of native vegetation coverage, % of exotic vegetation coverage, % pasture and % of urban areas. We included percentage of the basin corresponding to lakes and areas above the timberline.

2.2. Environmental characterization and macroinvertebrate analysis

Percentages of boulder, cobble, gravel, pebble, and sand in the reach were estimated at each site using a 1-m^2 grid. Current speed was measured in mid-channel on three occasions by timing a float (average of three trials) as it moved over a distance of 10 m (Gordon et al., 1994). Average depth was estimated from five measurements along one transversal profile across the channel with a calibrated stick. Discharge was obtained by combining depth, wet width and current velocity as in Gordon et al. (1994). At each site, air and water temperature were measured with a mercury thermometer.

Specific conductance (μS cm^{-1}), pH, dissolved oxygen (DO, mg O$_2$ l^{-1}) were measured with a Horiba U2-probe. For nutrient analyses water samples were collected below the water surface and kept at 4 °C prior to analysis and transported to the laboratory. Total suspended solids (TSS), total nitrogen (TN), total phosphorus (TP), nitrate plus nitrite nitrogen (NO$_3$ + NO$_2$), ammonia (NH$_4$), and soluble reactive phosphate (SRP) were analyzed following standard methods (APHA, 1994).

Quantitative macroinvertebrate samples were taken with a Surber sampler (0.09 m^2; 250 μm mesh size). On each reach, three samples were taken from riffles and three from pools (n = 6). Samples were fixed in situ with 4% formaldehyde, and sorted in the laboratory under at least 5× magnification. Macroinvertebrate species were identified using available keys (Fernández and Domínguez, 2001). Functional feeding groups (FFGs) were assigned using available references, knowledge of feeding modes (mouthpart morphology and behaviour), and analysis of gut contents (Merritt and Cummins, 1996; Albariño and Balseiro, 2002).

Macroinvertebrate total density (TD), total richness of taxa (TR), diversity (H: Shannon Weaver diversity index) and dominance (BPD; Berger–Parker dominance index) were calculated per site. Species richness, percentage and total density of shredders (S), grazer/scrapers (SG), collector filterers (CF), collector gatherers (CG) and predators (P) were also obtained (Ludwig and Reynolds, 1988).

2.3. Particulate organic matter and chlorophyll analysis

Detritus from benthic samples was divided into fine (250–1000 μm) and coarse (>1000 μm) particulate fractions (FPOM and CPOM, respectively). CPOM was separated into wood, leaves (mainly entire leaves), macrophytes, bryophytes and others (fragments of leaves, grass, seeds, roots, buds, etc.) (Miserendino, 2007). All fractions were dried (105° C for 4 h) and weighed on an electronic balance (Scout Pro) to ±0.003 g. Total benthic particulate organic material (BPOM) was obtained by adding FPOM and CPOM.

Algal biomass (as chlorophyll a) was determined by scraping algae from five randomly selected rocks (length range: 5–16 cm) collected within a 20 m run/riffle reach at each site. Scrapings were brought back to the laboratory and drawn on to GF/FF filters. Chlorophyll a was extracted from pulverized filters in 90% acetone and the extract was measured, spectrophotometrically as described by Wetzel and Likens (1991).

2.4. Statistical analyses

Land use differences in physicochemical parameters and functional groups attributes were tested using one-factor analysis of variance, or where assumption of normality was violated, Kruskal–Wallis tests. Location of significant differences was tested using Tukey's post hoc test for ANOVA and Duncan test for Kruskal–Wallis (Sokal and Rohlf, 1995). Variability within and among groups of selected streams was assessed by means of a similitude matrix based on taxa percentage composition (Bray Curtis matrix) (Ludwig and Reynolds, 1988), this was performed using the program Bio Diversity Pro. A Pearson correlation matrix based on quantitative macroinvertebrate data was performed, this matrix was employed to produce a non-linear ordination using the Multidimensional Scaling method. This is an alternative to the factorial method and the objective is to display significant distances among investigated objects (Ludwig and Reynolds, 1988). The associations between the MDS ordination scores and environmental variables were tested by mean Spearman rank correlations.

3. Results

3.1. Environmental features

Stream shade was lower at Pas and Ur than at forested sites, as a consequence lower water temperatures were recorded in canopied sites (Table 1) thus NF sites had significantly lower water temperature than Pas and Ur sites (ANOVA, Tukey test $p < 0.01$). Both wet (range 0.9–21 m) and channel width (range 1.8–31.5 m) varied considerably between sites, having Pas and Ur the most expanded channels, however differences were only significant when comparing pasture dry width with all types of forested channels (ANOVA, Tukey test $p < 0.01$). At most sites the substrate particle size was mainly boulder, cobble and pebble. However, a higher proportion of sand was observed in some particular sites (LÑ, PAT, PIP) having sedimentation symptoms (Table 1). Current velocity was comprised between 0.1 and 3 m seg^{-1} but did not vary in a systematic manner among different uses.

There were also evident differences in physico-chemistry, with dissolved oxygen contents higher ($F_{(5,12)} = 4.0$, $p < 0.02$) and conductivity lower (Kruskal–Wallis, $p < 0.02$) at NF sites than the rest. Most chemical variables including SRP, TP, NO$_3$ and NH$_4$ showed higher mean values at Ur use than at the rest, including R-ur sites. However, values were highly variable, possibly explained by extreme values at ED urban site. A comparison of the same variables, excluding Ur sites from the analyses, highlighted that SRP was slightly lower at Pn sites, TP increased at Pas sites, NO$_3$–NO$_2$, TN and NH$_4$ values were higher at HF, but none of these results was statistically significant ($p > 0.05$). Nonetheless, the trend was that lowest values of NH$_4$, TN, and TP were recorded at reference native forest sites (Table 1).

The land coverage with exotic (pine) plantation at selected sites ranged between 3.1 and 7.3% and land coverage with pastures primarily dedicated to sustain livestock varied between <1% and 25%. Urbanization affected between 0.4% and 9% of the basins however, rural/agricultural areas affected near of 18.7% of some basins as was the case of ED.

3.2. Particulate organic matter and chlorophyll a

Differences in energy availability with land uses were also evident particularly in the POM fractions: wood, leaves, CPOM, BPOM, and FPOM (Fig. 2). HF streams contain significantly higher leaves than Ur sites (Mann–Whitney U test, $p < 0.04$) and NF showed more wood than Pas and Ur sites (Kruskal–Wallis, $p < 0.04$). As expected, BPOM biomass was higher at NF and HN than Pas (Mann–Whitney U test, $p < 0.05$). Although FPOM did not varied in a consistent manner among land uses (ANOVA, $p = 0.30$), HF sites had higher FPOM biomass than Pas sites (Mann–Whitney, $p < 0.04$, $n = 6$, Fig. 2).

Regarding to autotrophic subsidies differences observed in chlorophyll a (range 2.3 and 17.9 mg m^{-2}) and macrophytes biomass (0 and 15.56 g m^{-2}) were not significant among land uses ($p = 0.84$, 0.77, respectively). However, bryophytes biomass was significantly higher at native forest sites than at Pn, Ur, Pas and R-ur sites (Kruskal–Wallis, $p < 0.01$, Fig. 2).

3.3. Functional metrics and land use

A total of 112 taxa were identified in the survey, being 22 shredders, 27 collector gatherers, 13 collector filterers, 23 scrapers/grazers and 27 predators. Taxa richness ranged between 20 and 48 taxa, and as expected the lowest value (20 taxa) corresponded to ED an urban site (Table 2).

Significant differences in shredder richness were observed among land uses with more shredders species found in HF and NF

Table 1

Environmental features of 18 sampled sites of streams in Patagonia (Argentina) during March 2006. Land use codes—Pas: pasture, R-ur: reference for urban site, Ur: urban site, Pn: pine plantation, HF: harvest forest, NF: native forest reference site.

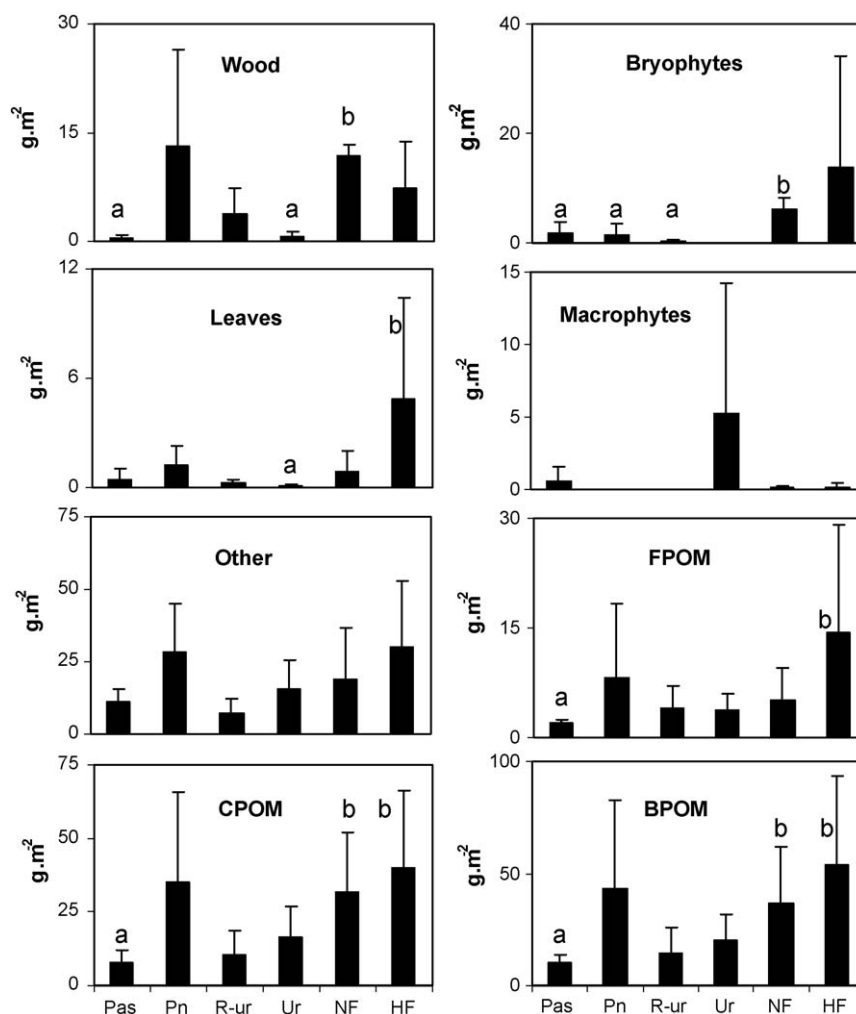| Site	Site code	Land use	Elevation (m.a.s.l.)	% Native forest	% Pasture wetlands	% Exotic forest	% Urban areas	% Timberline and lakes	Water temperature (°C)	Wet and dry width (m)	Depth (cm)	W. velocity (m seg^{-1})	% Sand	Conductivity (μS cm^{-1})	DO (mg l^{-1})	TSS (mg l^{-1})	SRP (μg l^{-1})	TP (μg l^{-1})	NO$_3$ (μg l^{-1})	TN (μg l^{-1})	NH$_4$ (μg l^{-1})	Chlorophyll a (mg m^{-2})
Nant y Fall	NyF	Pas	690	56.3	25.9	0.2	0	14.6	13.6	21–31.5	19	1.5	15	113	7.7	1.7	0.4	8	0.0	410	0.7	2.3
E. Los ñires	LÑ	Pas	667	81.0	7.9	0	0	11.0	9.4	4.3–22.5	20.6	0.9	40	98	9.8	1.1	0.5	18	0.1	<10	0.5	5.4
Manguera	MG	Pas	699	76	<1	0	0	23	8.8	5.9–14.2	13.2	1.5	5	87	10.2	1.3	0.7	13	0.0	<10	0.4	10.4
Esquel	EU	R-ur	721	11.3	0	0.7	0	88.0	13.1	2.8–10.8	13.1	3	10	105	7.8	0.1	0.4	4	0.0	467	0.5	4.5
Las Minas	LMU	R-ur	573	93.1	1.3	0	0	5.6	10.2	3.25–5	17.9	0.9	10	164	9.2	0.8	0.4	2	nd	228	0.8	3.5
Carbón	CU	R-ur	384	97.0	1.4	1.4	0.2	0	9.3	7.57–10.5	18.9	1.7	10	93	9.8	1.7	0.7	14	0.1	319	0.5	9.3
Esquel	ED	Ur	491	11.0	18.7	5.6	9.0	55.7	11.9	9.8–10.5	23.1	1.9	20	366	8.3	4.1	21.0	816	69.0	5464	271.5	3.7
Las Minas	LMD	Ur	555	92.3	1.3	0	0.7	5.7	12.4	4.3–9	10.5	3	5	72	8.3	1.1	0.3	6	0.1	55	0.6	4.2
Carbón	CD	Ur	403	96.6	1.5	1.1	0.4	0.4	10.1	15.8–23.5	27.6	1.3	10	84	9.3	0.7	0.4	13	0.3	45	0.6	5.6
Golondrinas	GOL	Pn	361	57.1	1.3	4.7	0	38.2	8.7	1.6–4.2	20.5	2.6	30	126	10.1	1.5	0.2	12	0.1	<10	0.4	5.7
Patriada	PAT	Pn	495	79.1	0	7.3	0	13.6	8.4	1.2–1.8	7.8	0.3	35	76	10.3	0.5	0.6	8	0.5	354	0.6	4.1
Ifona	IFO	Pn	368	69.5	0	3.1	0	27.4	10.5	8.8–10.5	33.2	1.3	5	38	9.1	4.1	0.2	7	0.0	<10	1.6	15.5
Cab. De vaca	CVA	HF	699	90.4	2.8	0	0	6.8	9.4	4.6–6.9	21.6	1.1	15	63	9.7	0.7	0.7	13	0.2	10	0.6	3.7
Pipo	PIP	HF	728	99.0	0	0	0	1.0	10.6	0.9–2.3	7.7	0.1	35	84	8.9	0.6	0.9	13	0.2	280	4.0	15.5
Glyn	GLY	HF	615	98.8	0	0	0	1.2	8	3.5–5.7	15.6	1.5	15	69	10.6	0.7	0.2	2	0.4	524	0.5	3.7
Comisario	COM	NF	770	60.8	0	0	0	37.2	7.4	8.7–9	24.1	0.8	5	29	11.0	0.7	0.2	4	0.0	<10	0.5	5.0
Loro	LO	NF	820	43.7	0	0	0	56.3	6.1	3.4–4	22.1	1.1	5	26	11.9	4	0.2	3	nd	153	0.5	4.7
Chiquito	CHI	NF	670	97.1	0	0	0	2.9	6.8	2.4–3.5	10.4	3	10	9	11.3	5.2	0.9	26	0.2	<10	0.5	17.9

Fig. 2. Mean dry biomass of different energy supplies across different land uses at 18 sampling sites, Chubut, Argentina. Items: wood, entire leaves, other (mainly fragments), macrophytes, bryophytes, fine particulate organic matter (FPOM), total coarse benthic organic matter (CPOM) and total benthic organic matter (BPOM) ($\pm$SD, g m^{-2}). Superscript letters show significant differences among sites ($p < 0.05$) as indicated by Kruskal–Wallis test.

Table 2
TR: total richness, TD: total density (ind. 0.09 m^{-2}), D: density (S: shredders, CG: collector gatherers, CF: collector filterers SG: scrapers/grazers and P: predators), H: Shannon Weaver diversity index, BPD: Berger–Parker dominance (%) ($n = 6$). Data from 18 sampled streams in Patagonia Argentina (March 2006).

	Site (land use)																	
	NyF (Pas)	LÑ (Pas)	MG (Pas)	EU (R-ur)	LMU (R-ur)	CU (R-ur)	ED (Ur)	LMD (Ur)	CD (Ur)	GOL (Pn)	PAT (Pn)	IFO (Pn)	CVA (HF)	PIP (HF)	GLY (HF)	COM (NF)	LO (NF)	CHI (NF)
TR	32	37	28	31	31	36	20	31	42	38	27	31.5	39	32	48	33	34	45
S	3	6	5	5	4	4	1	7	6	4	5	5	7	6	7	6	9	8
CG	9	11	4	7	8	9	9	6	12	8	6	6	11	7	13	10	9	13
CF	5	2	3	5	4	2	1	3	5	2	1	3	4	1	2	2	2	1
SG	9	9	10	9	8	11	7	7	11	10	6	8	9	7	13	7	4	10
P	6	9	6	5	7	10	2	8	8	14	9	9.5	8	11	13	8	10	13
% S	48.3	12.8	42.1	15.2	23.2	12.2	0.1	20.7	17.2	9	35.1	13.5	11.1	17.5	62.6	26.4	42.8	21.1
%CG	18.5	16.2	21	18.1	36.8	34.8	93.1	42.4	26.8	8.3	9.9	24.4	39.8	43.6	16.1	28.4	14	20.3
%CF	16.5	42.1	5.9	7.7	8	5.8	0.1	16.6	14.3	36.8	14.9	5	25.4	0.2	3.5	2.8	2.7	35.7
% SG	15.5	25.6	27.9	54.7	26.1	40.3	0.2	14.5	37.6	37.6	33.5	52.9	17.9	16.3	14.6	37.4	26.1	19.6
% P	1.2	3.2	3.1	4.3	5.9	6.9	6.5	5.8	3.9	8.2	6.7	4.2	5.8	22.4	3.2	5	14.4	3.2
TD	1119	548.5	629.6	273	229.8	494.5	3497	2126	1051	457.6	141.8	83.9	1192.2	1488.5	1866.8	206.8	108.9	614.3
DS	540.8	70.2	265.5	41.5	53.3	60.3	1	440.8	181.1	41	49.6	11.3	132.2	260	1168.5	55.2	46.6	129.8
D CG	207.6	89	132	49.3	84.5	172.1	3254.6	901	282.2	38	14	20.5	474	649	299	58.6	15.2	124.6
D CF	184.6	231	37.2	21.2	18.3	28.5	5.3	353.2	150.8	168.6	21.2	4.2	302.3	3.7	66.3	5.8	3	219.5
D SG	172.8	140.5	175.5	149.3	60	199.5	8.7	308.2	395.5	172.3	47.5	44.4	213.8	242.6	272.8	77.3	28.3	120.6
D P	13	17.8	19.5	11.6	13.6	34	227.3	122.8	41.3	37.6	9.5	3.5	69.8	333.2	60.2	9.8	15.7	19.6
H	2.33	2.04	2.12	2.51	2.58	2.81	1.89	2.07	2.93	2.34	2.11	2.28	2.75	2.26	1.86	2.7	2.89	2.42
BPD	27.4	41.9	36.7	22.5	20.8	17.4	39.3	20.9	13.2	33.1	31.1	30.2	14.9	32.3	60.5	20.5	13.6	35.7

Table 3
Results of Kruskal–Wallis (H, $n = 6$) or Mann–Whitney U test (pair site comparison) for effects of land use on macroinvertebrates functional attributes.

	$H_{(5,18)}$	p	Post hoc test	Mann–Whitney U test ($p < 0.05$)
Total richness	4.58	0.46		
Richness				
Shredders	9.7	**0.026**	HF, NF > Pn, R-ur	
Collector gatherers	5.89	0.32		
Collector filterers	4.48	0.48		
Scraper/grazers	2.56	0.76		
Predators	8.02	0.15		
Relative abundance				
% Shredders	4.32	0.50		
% Collector gatherers	7.90	0.16		
% Collector filterers	2.77	0.73		
% Scraper/grazer	9.98	0.07		Pn > HF
% Predator	6.49	0.26		
Density				
Total density	13.8	**0.016**	Ur, HF > Pn, NF, R-ur	
Shredders	9.6	0.08		HF, Pas > Pn
Collector gatherers	14.5	**0.012**	HF > Pas, NF, R-ur > Pn	
Collector filterers	3.30	0.65		
Scraper/grazer	7.61	0.18		HF > Pn, NF, Pas
Predator	11.7	**0.04**	Ur > Pas, NF	

Significant differences are in bold. Significant post hoc comparisons (Duncan, Kruskal–Wallis). Land use codes—Pas: pasture; R-ur: reference for urban site; Ur: urban site; Pn: pine plantation; NH: native harvest; NF: native forest reference site.

than Pn and R-ur sites (Kruskal–Wallis, $p < 0.026$, Table 3, Fig. 3). Collector gatherers richness was highly variable (range: 4–13 taxa) and collector filterers (range: 1–5 taxa) were poorly represented mostly at Pas, Pin and NF sites.

As expected total density of macroinvertebrates was significantly higher at Ur and HF than at Pn, NF and R-ur sites (Kruskal–Wallis, $p < 0.016$). Collector's gatherers were significantly more abundant at HF sites than at the rest of uses, moreover, Pas, NF and R-ur had higher density of collector's gatherers than Pn (Kruskal–Wallis, $p < 0.012$). Abundance of shredders was significantly lower at Pn sites than at Pas and HF. Density of scrapers/grazer was higher at HF compared to Pn, Pas and NF (Mann–Whitney U test, $p < 0.05$) (Table 3, Fig. 3). More predators were recorded at Ur sites than at Pas and NF sites.

Diversity values ranges between 1.86 and 2.93 and lowest values were found at Pas and Pn sites. Dominance values were high at HF, Pn and Pas uses. However, some particular sites showed extreme values as GLY by recruitment of the stonefly *Antarctoperla michaelseni* (Gripopterygidae) and ED by high density of *Hyalella araucana*, *Limnodrilus* spp. and Orthocladiinae species.

Regarding to species composition, similarity percentages showed that sites in Pas overlapped more with Ur, R-ur, and HF (Table 4), than with Pn and NF. Similarities between sites within land uses were relatively high, although Ur sites were more variable probably reflecting differences in degrees of impairment. NF selected sites showed a reasonable similitude among each other (55.8%), the same as HF sites (57%). However, the within similarity of land cover types ranged 36.9–64.3%. This might show a fairly
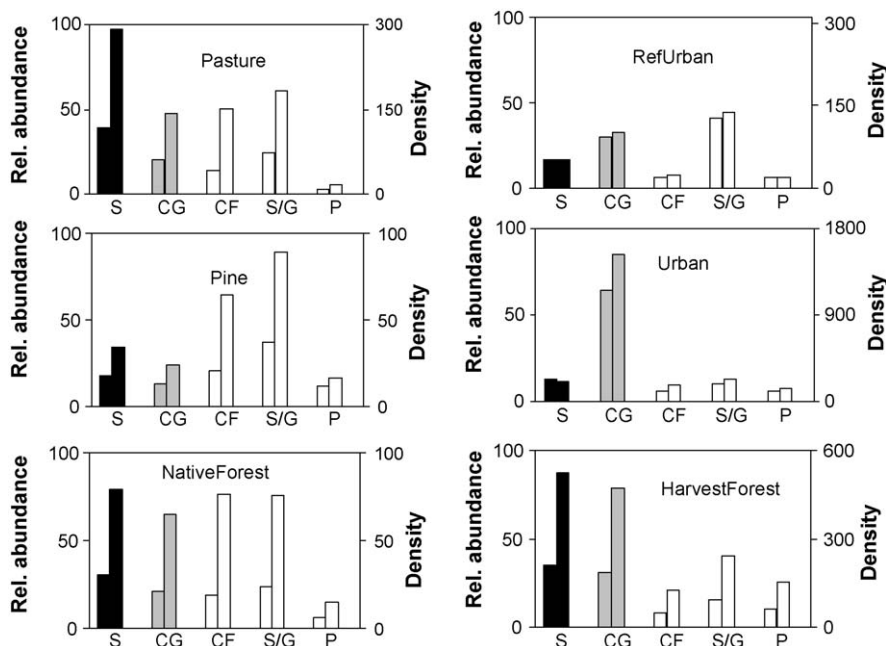

Fig. 3. Relative abundance (left bar) and total density (right bar, ind. m^{-2}) of functional feeding groups at 18 sampling sites subjected to different land uses, Chubut, Argentina. Functional feeding groups are—S: shredders, CG: collector gatherers, CF; collector filterers, S/G: scrapers/grazers and P: predators.

Table 4
Similarity percentages (Bray Curtis matrix) showing similarity in species composition within and between land uses types.

	Pasture	Ref urban	Urban	Pine	Native forest	Harvest forest
Pasture	53.4					
Ref urban	58.9	64.3				
Urban	50.3	49.6	36.9			
Pine	39.8	50.6	37.4	40.7		
Native forest	41.5	51.3	38.1	47.3	55.8	
Harvest forest	53.1	55.6	46.7	47.7	51.6	57
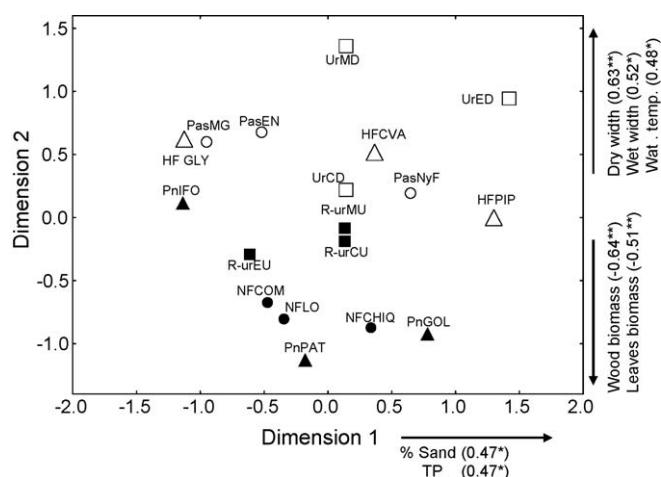
Fig. 4. Results of the multidimensional ordination of macroinvertebrate community data collected from 18 sampling sites subjected to different land uses (Chubut, Argentina). Symbols are: (□) urban sites; (△) harvest forest; (○) pasture; (■) reference urban sites; (●) reference native forest and (▲) pine plantations. Significant correlation coefficients (Spearman rank) with dimension 1 and 2 are given (*$p < 0.05$, **$p < 0.005$).

high within-type variability, which was largest in Ur and Pn forest sites.

3.4. Environmental relationships

The separation of forested either with native or exotic species, plus reference sites (dark symbols) from pasture, urban and harvest sites (open symbols) was clearly evident in the site ordination using density data of macroinvertebrate species (dimension 2, Fig. 4). Thus, NF, R-ur and Pn sites were grouped to the negative end of dimension 2, whereas Pas, HF and Ur were positioned to the positive end of dimension 2. When identifying variables associated with dimension 2, wood and leaf biomass which were negatively correlated, resulted as significant variables explaining sites ordination (Fig. 4). Other significant variables were water temperature and dry and wet width, which were positively correlated with dimension 2. On the other hand, dimension 1 highlighted a water quality impairment gradient or sedimentation; variables that were positively correlated with dimension 1 were TP and sand percentage (Fig. 4).

4. Discussion

Our results suggest that distribution of detritus and trophic structure were different among land uses, this is in agreement with what has been reported for streams draining in developed catchments (Delong and Brusven, 1993; Riley et al., 2003; Thompson and Towsend, 2004; Danger and Robson, 2004). Changes in physical attributes of streams produced by some land use practices were also documented in this study. The buffering effect of the riparian corridors on water temperature (Collier et al., 2000; Scrimgeour and Kendall, 2003; Leunda et al., 2009) was evident at native forested sites. Similarly the percentage of fine sediment on the streambed varied among uses and explained some differences in benthic assemblages. Agricultural land use and pasture development often results in an increased supply of fine sediments (Scarsbrook and Halliday, 1999; Brisbois et al., 2008). Niyogi et al. (2007) found that fine sediment covered 33% of the stream bottom at agricultural streams in New Zealand. We observed that LÑ showed a high percentage of sand as consequence of the constant supply from the severely eroded banks while post-harvest sedimentation effects were evident at Pipo (HF). At this site several trees from the riparian zone were cut and the channel was obstructed by debris (branches and leaves) after the harvest. Logging operations can produce increases of sand and silt in the streambed and this is more significant in small streams (Death et al., 2003). Some groups favored by the presence of fine sediments such as the collector gatherers *Lumbriculus variegatus* and *Pseudochironomus* sp. were abundant at these sites.

We expected higher values of chlorophyll *a* at pasture sites as a consequence of higher temperatures or nutrients levels. On the contrary, chlorophyll *a* pattern was highly variable as were nutrients, which is consistent with observations of Scrimgeour and Kendall (2003) and Scarsbrook and Halliday (1999). They did not report any significant difference in chlorophyll patterns and land uses, although values increased with distance to forested areas. In our study, even though nutrients did not show any significant differences among land use practices, lower values of NH_4, TN, and TP were recorded at reference native forest sites. Likely we were unable to separate cattle effects from other effects in our sampling design, thus cattle is an all-pervading land use in the area. Then some of the patterns observed at harvest sites could result of the combined effect of wood collection and pasturing.

A noteworthy feature in our study was that bryophyte biomass was significantly higher at native forest than at other land uses. Similarly, Riley et al. (2003) found lower moss coverage of streambeds at developed pasture streams than at undeveloped or lightly grazed native grassland catchments. They concluded that the presence of moss can be influenced by substrate stability (bedrock, boulder vs. gravel, sand), trampling by stock and nutrient concentrations.

As reported in other studies (Delong and Brusven, 1993; Benfield, 1997), forested sites showed consistently more allochthonous material than pastures, thus BPOM was significantly higher at NF and HF than at Pas. However, BPOM varied in a non-systematic manner among Pn sites. Scarsbrook et al. (2001) found more detritus material in pine plantations than in pasture sites and Thompson and Towsend (2004) reported a decreasing gradient of detritus biomass from pine, native forest to pasture grasslands. Likely differences observed in the attributes of the riparian corridors (composition, coverage and age) at our Pn sites, were the cause of high variation in detritus biomass.

The contribution of shredders to benthic community at pasture sites was higher than expected (12.8–48%). Most selected pasture sites have remnant forest upstream and this probably had some positive effects on detritus supply. From all of our pasture sites, the most extreme situation was Los Ñires with benthos community basically dominated by filterers (Hydropsychidae: *Smicridea annulicornis*). Predominance of filtering and gathering collectors in rural and agricultural landscapes has been well documented (Shieh et al., 2003, Ometto et al., 2004; Utz et al., 2009) and as suggested by Strand and Merritt (1999) a high proportion of filterers reflect the relative amount of fine particulate organic matter in transport in the water column. In addition, in our study collectors dominated or co-dominated at all urban sites (Compin and Céréghino, 2007) but also at most harvest forest sites. Thus, the

relative proportion of collectors was higher at HF sites than forest reference sites suggesting a compositional shift from CPOM feeders to FPOM feeders. McCord et al. (2007) observed that increases of collector's relative abundance was a significant response indicating detrimental effects of silviculture operations when comparing harvest from reference sites. As we mentioned the accumulation of logging slash due to the incorrect treatment of debris was evident at least in one site (Pipo).

Whiles and Wallace (1997) stated that conversion of forest to pine monoculture can influence benthic taxonomic composition. In our study, when assessing the possible effects of exotic plantation on functional feeding groups, the most marked finding was that shredder richness was consistently higher at HF and NF than Pn sites. Besides, total density of shredders was significantly lower at Pn than at Pas and HF. Valdovinos (2001) found more density and biomass of shredders in leaf packs of *N. pumilio* (native) than *Pinus radiata* in experimental trials in Chilean streams. In other experiments, Albariño and Balseiro (2002) noticed that large shredders (*Klapopteryx kuscheli* and *Parasericostoma cristatum*) fed only on *N. pumilio* and not on *Pinus ponderosa* leaf packs in Patagonian streams. The most dramatic situation in our study was the site PAT, in which the riparian corridor is completely replaced by *Pseudotsuga menziesii* and the stream runs through 1200 m of the exotic forest. Paradoxically, the shredder *Myotricha murina* was well represented at this site and constitutes almost 35% of the benthic community. However, other common shredders such as *Senzilloides panguipulli*, *Antarctoperla michaelseni*, and *Udamocercia arumifera* that were well represented at HF and HF sites were practically absent at Pn sites.

As was showed in the ordination analysis, pasture sites were clearly separated from forested sites (either native or exotic) in agreement with reports from similar studies (Hall et al., 2001; Death et al., 2003; Thompson and Towsend, 2004; Compin and Céréghino, 2007). Moreover, in our study harvest forest sites were also clearly separated from undisturbed native forest suggesting changes in composition and structure of benthic community. Detritus biomass, sedimentation processes and habitat impoverishment, factors associated with the removal of the forest canopy and harvesting procedure, appear as important variables determining those assemblages. As we tested here there is also a potential usefulness of functional metrics in particular shredders richness and collectors density to indicate the impact of land use change on macroinvertebrates.

This study reinforces the idea that riparian corridor management may be critical to the distribution of benthic taxa as well functional organization in headwater Patagonian streams. For this reason maintenance of good conditions of vegetation adjacent to rivers will enhance water quality and the environment for stream communities.

Acknowledgments

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