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Effects of local land-use on riparian vegetation, water quality, and the functional organization of macroinvertebrate assemblages

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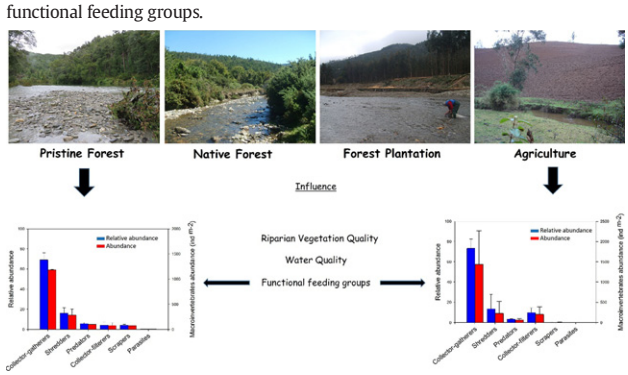
HIGHLIGHTS

- Land-use changes diversely affected river biodiversity.
- Land-use altered riparian vegetation, water quality & macroinvertebrate assemblages.
- No land-use differences found for functional feeding groups.
- Agricultural streams had worst water quality and macroinvertebrate diversity.

GRAPHICAL ABSTRACT

Land-use changes effectively impacted riparian vegetation, water quality, and macroinvertebrate assemblages, but not functional feeding groups.



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ABSTRACT

Land-use change is a principal factor affecting riparian vegetation and river biodiversity. In Chile, land-use change has drastically intensified over the last decade, with native forests converted to exotic forest plantations and agricultural land. However, the effects thereof on aquatic ecosystems are not well understood. Closing this knowledge gap first requires understanding how human perturbations affect riparian and stream biota. Identified biological indicators could then be applied to determine the health of fluvial ecosystems. Therefore, this study investigated the effects of land-use change on the health of riparian and aquatic ecosystems by assessing riparian vegetation, water quality, benthic macroinvertebrate assemblages, and functional feeding groups. Twenty-one sites in catchment areas with different land-uses (i.e. pristine forests, native forests, exotic forest plantations, and agricultural land) were selected and sampled during the 2010 to 2012 dry seasons. Riparian vegetation quality was highest in pristine forests. Per the modified Macroinvertebrate Family Biotic Index for Chilean species, the best conditions existed in native forests and the worst in agricultural catchments. Water quality and macroinvertebrate assemblages significantly varied across land-use areas, with forest plantations and agricultural land having high nutrient concentrations, conductivity, suspended solids, and apparent color. Macroinvertebrate assemblage diversity was lowest for agricultural and exotic forest plantation catchments, with notable non-insect representation. Collector-gatherers were the most abundant functional feeding group, suggesting importance independent of land-use. Land-use areas showed no significant differences in functional feeding groups. In conclusion, anthropogenic land-use changes were detectable through riparian quality, water quality, and

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macroinvertebrate assemblages, but not through functional feeding groups. These data, particularly the riparian vegetation and macroinvertebrate assemblage parameters, could be applied towards the conservation and management of riparian ecosystems through land-use change studies.

1. Introduction

Land-use changes have ecological consequences for the health of terrestrial and aquatic ecosystems (Bruno et al., 2014; Lammert and Allan, 1999). Although the effects of land-use changes are well documented for aquatic ecosystems, few studies have specifically assessed riparian vegetation in this context in South America (Hepp and Santos, 2009; Moraes et al., 2014). Changes in riparian vegetation can affect water quality and aquatic communities. Indeed, agriculture, pasture, and urban land-use changes often result in decreased diversity and shifts in relative abundance among the functional feeding groups (FFGs) of aquatic macroinvertebrate assemblages. In contrast, sites of good riparian quality present higher densities of scrapers, predators, and collector-gatherers (Mesa, 2014; Miserendino et al., 2011).

Global deforestation, forest degradation, and the replacement of native forests with exotic forest plantations or agricultural lands are ongoing, growing trends (Manuschevich and Beier, 2016). For example, south-central Chile is globally recognized as a biodiversity hotspot with a high degree of endemism (Myers et al., 2000). Nevertheless, increasing rates of land-use change over recent decades threaten this region, particularly resulting from native forest conversion for agriculture, forest exotic tree plantations, and urbanization (Fierro et al., 2017; Hernández et al., 2016). Riparian environments, which are ecotones between terrestrial and aquatic ecosystems (Valdovinos, 2001), are heavily degraded due to the aforementioned human activities. This degradation limits the normal role of riparian environments as biological buffers against contaminant, sediment, and nutrient influxes from adjacent land-use areas. Riparian habitat destruction can also increase the quantity of particulate matter in water, which can affect light penetration, increasing water temperature, and contribute to excessive eutrophication (Baillie et al., 2005). Therefore, any deterioration of riparian vegetation can directly and indirectly affect water quality and associated aquatic communities (Fernández et al., 2009; Foley et al., 2005).

In addition to influencing the physicochemical conditions of waterways, riparian vegetation also creates habitats and provides nutrients to terrestrial and aquatic organisms, especially in headwater systems, where allochthonous inputs form a principal energy source. A decrease in riparian vegetation will alter trophic food webs, thereby affecting the composition of primary and secondary producers in rivers, in addition to having knock-on impacts for higher tropic levels (Cummins and Klug, 1979). In effect, changes in land-use and, consequently, riparian vegetation can alter the normal distribution of aquatic fauna, which naturally varies within rivers (Vannote et al., 1980).

To evaluate human impacts on aquatic ecosystems, many studies use biological indicators that can provide integrated information about the cumulative or synergistic temporal effects caused by environmental alterations (Alba-Tercedor, 1996). Aquatic organisms are often used to evaluate river health, particularly as these organisms can serve as a record for various perturbations and multiple pollutants (Karr, 1981; Hughes et al., 1998; Allan, 2004; Macedo et al., 2016). Macroinvertebrate assemblages are excellent indicators of water quality due to varied sensitivities to organic pollution (Fierro et al., 2012). Therefore, these assemblages can be used to evaluate aquatic ecosystem responses to adjacent land-use changes.

Basic metrics such as abundance, taxa richness, and diversity are commonly used. However, a variety of other metrics are affected by distinct disturbance gradients (Hughes et al., 1998; Pont et al., 2009). High macroinvertebrate diversity and taxa richness exist in rivers surrounded by natural pristine forests, whereas rivers impacted by human perturbations show altered communities with dominating taxa groups (e.g. dipteran or oligochaetes families) (Brand and Miserendino, 2015). Bioindices developed over the last three decades integrate other measures that basic metrics alone cannot establish. For example, the Hilsenhoff Biotic Index (HBI; Hilsenhoff, 1988) is widely adapted in many countries and is based on quantities of pollution-tolerant macroinvertebrates at different locations. Since water quality is closely related to land-use, related indicators not only indicate river health, but also the health of adjacent riparian areas (Sweeney et al., 2004). Indeed, riparian vegetation quality in itself can serve to describe the biological or habitat conditions of rivers. This parameter can be determined by several methods, including basic methods such as the presence or biomass of native species. One simple field method for evaluating the status of riparian areas is the Riparian Quality Index (QBR, "*Qualitat del Bosc de Ribera*"; Munné et al., 2003). This index assesses differences in the grade and quality of vegetative coverage, the structure thereof, and the habitat naturalness for the river channel.

In addition to these ecological indices, many biomonitoring programs assess macroinvertebrate FFGs (Callisto et al., 2001; Gebrehiwot et al., 2017). The concept of FFGs is based on associations of (i) feeding adaptations by and food-resource categories for aquatic invertebrates with (ii) longitudinal changes in allochthonous and autochthonous organic matter sources from the headwaters of large rivers (Vannote et al., 1980). Consequently, any human perturbations that impact organic matter sources can modify FFG compositions. For example, shredder abundances decline as native forests and riparian vegetation decline, while collector-filterers and scrapers increase in abundance in rivers with diminished or degraded riparian vegetation quality (Brand and Miserendino, 2015; Moraes et al., 2014).

The aims of this study were to investigate the effects of land-use changes in catchment areas, i.e. drainage basins, on aquatic and riparian health in south-central Chile using a variety of indicators, namely water quality (via physicochemical parameters), riparian vegetation (via QBR), and aquatic macroinvertebrate assemblages (via modified HBI and FFGs). The assessed sites were within a region recognized as a biodiversity hotspot with high levels of endemism. However, this region has been greatly impacted by deforestation, exotic forest plantations (namely pine and eucalyptus), and intensive agricultural land conversion. We hypothesized that land-use changes in catchment areas would negatively affect riparian vegetation and water qualities. We further hypothesized that land-use changes would alter aquatic invertebrate communities, which are often used as bioindicators of water quality, as well as impact invertebrate diversity and FFG compositions. The generated information represents a valuable tool that could aid in policy-making and the establishment of biomonitoring and management programs, namely by identifying anthropogenic factors and biological indicators that can alter the health of aquatic ecosystems in southern South America.

2. Materials and methods

2.1. Geographical traits and land-use categorizations of assessed catchment areas

2.1.1. General geography of the study area

All assessed catchment areas and respective study sites were located in the Araucanía and Los Ríos Regions of Chile (38°54′0″S, 72°40′0″W). This area, situated in the south-central part of the country, was selected due to its noted natural biodiversity and growing incidence of land-use

changes (see Introduction). The catchment areas chosen for study were, more specifically, located on the western side of the Nahuelbuta Range (37°43′S 73°02′W), which forms part of the Chilean Coast Range. The slopes of the assessed catchments were generally steep (>20% slope), principally composed of sedimentary rock, and had annual hydrographs dominated by rainfall. The climate of the Nahuelbuta Range is maritime Mediterranean, with wet winters and dry hot summers (Di Castri and Hajek, 1976). Average yearly precipitation is between 1200 mm and 1600 mm, and average annual temperature is 12 °C. Most rainfall occurs between May and August, with June and July having the highest precipitation rates.

2.1.2. Land-use categories considered for catchment area assessments

Research focused on four land-use categories: pristine forest, native forest, exotic forest plantation, and agricultural land. The pristine forest was located in the Valdivian Coastal Reserve, a private protected area for nature conservation characterized by a temperate rainforest with rich floral diversity (Romero-Mieres et al., 2014). This protected area was principally composed of a pristine *Nothofagus* forest that has rarely/never been harvested for wood. In turn, the native forest was dominated by roble (*Nothofagus obliqua*), rauli (*Nothofagus alpina*), and coihue (*Nothofagus dombeyi*), with native trees aged >50 years old. The exotic forest plantation included pine and eucalyptus trees that, over the course of 40 years, had displaced native forests; this plantation will soon be harvested for wood (Peña-Cortés et al., 2014). The exotic forest plantation started roughly 5–10 m from the stream edge, leaving a riparian corridor comprised of mixed native and exotic species. Finally, the agricultural land was primarily dedicated to potato farming (*Solanum tuberosum*). Riparian plant coverage on both sides of the stream within this land-use category presented exotic species, including willow (*Salix viminalis*, *S. caprea*, *S. babylonica*), poplars (*Populus nigra*), and flax (*Phormium tenax*).

2.1.3. Catchment areas and respective sampling sites

All catchment areas were similar in gradient and geology. The assessed streams were free-flowing and perennial. The surrounding territory consisted of a landscape highly transformed by human activities (Peña-Cortés et al., 2011, 2014), with most human populations located near the stream mouths. As detailed (see Section 2.1.2), the assessed catchments were affected by anthropogenic land-use changes, including intensive farming and exotic forest plantations, principally of pine and eucalyptus (Fierro et al., 2015; Huber et al., 2010; Valdovinos, 2001).

Five catchment areas, with varied land uses, were defined (Peña-Cortés et al., 2011). A total of 21 sampling sites were then established within these five catchment areas (Fig. 1, Table 1). The Chaihuin River catchment area was within a pristine forest and had two sampling sites. The Queule River catchment area was assigned eight sampling sites. The surrounding land-use of the Queule River catchment area was >60% native forests, with the remaining land-use principally being small-scale pine and eucalyptus plantations. The Moncul River and Lingue River catchment areas had eight sampling sites and were within an area where exotic pine or eucalyptus plantations accounted for >60% of total land-use, with native forest remnants accounting for the remaining land-use. Finally, the Budi River catchment area was on agricultural land and had three sampling sites.

2.2. Field-data collection

Water physicochemical parameters, riparian vegetation, and macroinvertebrate samples were collected during spring 2010 and 2011 (i.e. November) and summer 2011 and 2012 (i.e. January). These seasons were selected for sampling due to river-flow stabilities (i.e. during low-flow periods) and evidence that aquatic macroinvertebrate abundances in headwaters are greatest during these periods (Fierro et al., 2015, 2016). Sampling was not conducted in the fall or winter (i.e. high-flow periods) so that land-use analyses were unaffected by any hydrological stress to faunal communities.

2.2.1. Physicochemical parameters and water sampling

At each sampling site, physicochemical parameters (e.g. water temperature, pH, and conductivity) were measured in situ. Boulder, cobble, gravel, pebble, and sand distributions (%) were visually estimated at each sampling site using a 1 m^2 grid. A tape measure was used to record the width and average depth of active channels. River flow velocities at each site were calculated as an average of three cross-sectional measurements (i.e. river surface, middle, and bed) recorded using a handheld flow meter (Kaufmann et al., 1999).

Water samples were collected in duplicate during the morning, between 8:00 and 11:00 AM. Pre-washed glass bottles (1 L) were placed below the water surface in the center of the active channel. Water samples destined for dissolved oxygen analysis were immediately fixed in the field with reagents according to the Winkler Method (APHA, 2005). All collected water samples were kept at 4 °C and transported to the Analytical Chemistry Laboratory of the Institute of Chemistry and Natural Resources at the Universidad de Talca for chemical analyses. In the laboratory, water samples were first filtered through a membrane (0.45 μm pore size) and then subjected to posterior assessments to determine: biochemical oxygen demand; total suspended and dissolved solids; dissolved oxygen concentration; apparent color; and chlorides, sulfates, nitrates, and phosphates contents. All analyses were completed using standard methods for water and wastewater (APHA, 2005).

2.2.2. Macroinvertebrates sampling

Macroinvertebrates were sampled from a riffle zone, the most common habitat, at each sampling site using a Surber sampler (0.09 m^2, 250 μm mesh, 3 replicates per site). The collected macroinvertebrates were preserved in the field with 90% ethanol and then transported to the Benthos Laboratory of the Institute of Marine and Limnological Sciences at the Universidad Austral de Chile. In the laboratory, samples were sorted, and macroinvertebrates were counted and identified to the lowest possible taxonomic resolution using current keys (Domínguez and Fernández, 2009). Following criteria established in Fierro et al. (2015), the identified taxa were also classified within one of seven FFGs: shredders, collector-gatherers, collector-filterers, grazers, predators, detritivores, or parasites. All aquatic invertebrates were identified by the first author to maintain consistency among sample sets.

2.2.3. Modified Hilsenhoff Biotic Index for Chilean species

The sampled macroinvertebrate assemblages were assessed using the HBI modified for Chilean species (Fierro et al., 2012). This family biotic index assigned a pollution-tolerance value to each taxon as per macroinvertebrate assemblage composition. The final index value was determined by Eq. (1):

$$HBI = \frac{\sum n_i a_i}{N} \quad (1)$$

where n is the number of specimens in taxon i, a is the tolerance value of taxon i, and N is the total number of specimens in the sample. The final value corresponded to one of seven water quality classes, ranging from excellent (HBI = 0.00–3.75; organic pollution unlikely) to very poor (HBI = 7.26–10.00; severe organic pollution likely).

2.2.4. Riparian quality index

The status and quality of riparian vegetation at all samples sites were evaluated using the QBR, following the methodology established in Carrasco et al. (2014). Four fundamental aspects of each riparian system were analyzed via the QBR, including total coverage, coverage structure, coverage quality, and channel alterations. The resulting quality values were classified within one of six ranges: very good quality (QBR ≥ 95;

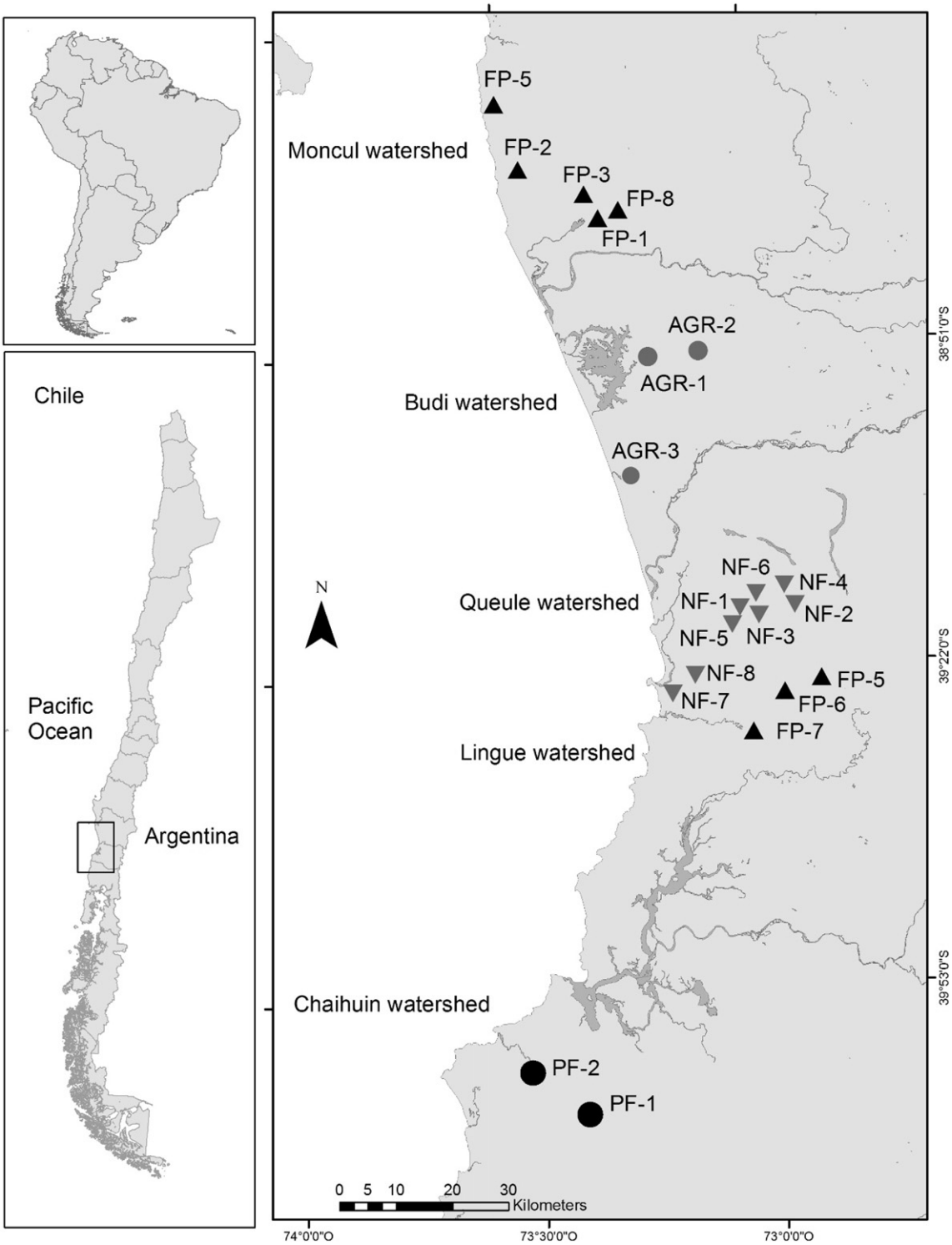

Fig. 1. Map of the study area, showing location of the 21 sampling sites in South-Central Chile across four different land-uses: pristine forest (PF, ●), native forest (NF, ▼), forest plantation (FP, ▲) and agriculture (AGR, ●).

riparian habitat in natural condition) to very bad quality (QBR ≤ 25; extreme degradation).

2.3. Data analysis

Associations between physicochemical parameters, macroinvertebrate assemblages, and FFGs were tested using multivariate statistical procedures within the PRIMER v6.1.2 software package (Clarke and Gorley, 2006) and through permutational multivariate analysis of variance (PERMANOVA; Anderson et al., 2008). Physicochemical data were first square-root transformed and normalized, after which a Euclidian distance matrix was constructed. Patterns among sampling sites and environmental gradients were assessed through a principal components analysis (PCA) of physicochemical data, QBR scores, and HBI values. Macroinvertebrate diversity was assessed using the Shannon-Weaver Index, and dominance was analyzed using Simpson's Diversity Index. Dissimilarities in the taxonomic composition of macroinvertebrate assemblages between land-use areas were assessed by

Table 1
Summary of catchments characteristics at the study sites ($n = 21$) across four different land-use in South-Central Chile. Site codes: PF, pristine forest; NF, native forest; FP, forest plantation; AGR, agriculture.

Watershed	Site code	Altitude (masl)	Stream order	Active channel width (m)	Water velocity (m s^{-1})	Depth (m)	Substrate type[a]
Chaihuín	PF-1	195	3	10	2.1	0.5	Bou/cob/peb
	PF-2	11	3	45	2.5	0.8	Bou/cob/peb
Queule	NF-1	101	3	3	1.7	0.3	Bou/cob/peb
	NF-2	99	4	20	0.9	0.5	Bou/cob/peb
	NF-3	45	3	6	2.5	0.6	Bou/cob/peb
	NF-4	17	4	7	1.1	0.5	Cob/peb
	NF-5	20	4	7	0.8	0.5	Cob/peb
	NF-6	25	4	6	1.8	0.7	Cob/peb
	NF-7	5	3	9	1.5	0.9	Cob/peb
	NF-8	30	3	7	1.1	0.5	Cob/peb/gra
Moncul	FP-1	25	4	30	0.7	0.5	Cob/peb
	FP-2	125	4	15	0.6	0.9	Cob/peb
	FP-3	36	3	25	1.6	0.6	Cob/peb
	FP-4	19	3	3	0.5	0.8	Peb/gra
Lingue	FP-5	60	2	6	0.8	0.4	Peb/gra
	FP-6	66	2	4	0.9	0.4	Peb/gra
	FP-7	12	2	22	1.1	0.5	Peb/gra
	FP-8	15	3	24	0.7	0.5	Peb/gra
Budi	AGR-1	13	2	5	0.2	0.3	Gra/san
	AGR-2	27	2	5	0.2	0.2	Gra/san
	AGR-3	19	2	4	0.2	0.3	Gra/san

[a] Bou: boulders, Peb: Pebbles, Cob: Cobbles, Gra: Gravel, San: Sand.

square-root transforming the biological data, macroinvertebrate abundances, and FFGs, constructing three Bray-Curtis distance matrices, and, finally, running non-metric multidimensional scaling ordinations. Significant differences ($p < 0.05$) in physicochemical data, diversity indices, macroinvertebrate assemblages, and FFGs were established using a one-way Permutational Multivariate Analysis of Variance (PERMANOVA; 9999 permutations), with land-use as a fixed factor. The contribution (%) of each taxon and FFG to the average dissimilarity between land-use areas was determined by similarity percentage analysis (SIMPER). Box- and whisker-plots were constructed in the R statistical software (R Development Core Team, 2016).

3. Results

3.1. Physicochemical parameters

Substrate sizes varied between boulders and sand. Sand was more common at agricultural land-use sites than at other types of land-use sites (Table 1). Channel widths/depths and flow velocities were significantly lower at agricultural land-use sites at other types of land-use sits (one-way PERMANOVA, $p < 0.001$, Table 2). Most sites had good water quality (Table 3), with most parameters falling in the expected range for coastal rivers in south-central Chile. Nevertheless, significant differences were found across the four land-use types (one-way PERMANOVA, $p < 0.001$, Table 2). Mean water temperatures were fairly similar despite different land-uses, although the highest recorded temperatures were at agricultural land-use sites. In turn, sampling sites within the catchment areas affected by agricultural and exotic forest plantation land-uses had significantly higher values of nitrates, conductivity, suspended solids, and apparent color than sampling sites within the pristine and native forest catchment areas. The highest pH levels and biological oxygen demand were found at the pristine forest land-use sites, while the highest concentrations of dissolved oxygen were associated with the native forest land-use sites (Fig. 2).

Table 2
Results from multivariate PERMANOVA analyses for differences in physical features habitat, physicochemical features water and macroinvertebrates.

Source	df	Pseudo-F	*P* (perm)	Perms
Physical				
Channel width - water velocities- depth	3	6,9648	***0,0001***	9939
Physicochemical				
Water quality	3	3,7898	***0,0001***	9922
Macroinvertebrates				
Assemblage	3	2,0246	***0,0012***	9880
Diversity index	3	3,5054	***0,0343***	9947
Functional feeding groups	3	1,3148	*0,2246*	9946
HBI	3	1,2239	*0,3276*	9958
Riparian vegetation				
QBR	3	4,7247	***0,0164***	1943

3.2. Macroinvertebrates and functional feeding groups

A total of 82,106 individuals, belonging to 113 taxa, were identified across the sampling sites. Taxa richness across sites varied between 22 and 56. In ascending order, taxa richness values were as follows for the assessed sites: agricultural sites (22–30 taxa); exotic forest plantation sites (26–54 taxa); native forest sites (31–54 taxa); and pristine forest sites (53–56 taxa). Mean individual densities differed between land-uses. While the highest average density was found for native forest sites, the highest overall density (2878 ind m^{-2}) occurred at an agricultural site (Table 4). Macroinvertebrate assemblages differed between land-use catchment areas (Fig. 3), as initially determined via non-metric multidimensional scaling (based on benthic macroinvertebrate compositions), and as subsequently confirmed by PERMANOVA ($p < 0.001$, Table 2). SIMPER analysis demonstrated an average species similarity of 68% at pristine forest sites, of 57% at native forest sites, of 53% at exotic forest plantation sites, and of 41% at agriculture sites. The species that contributed most to similarities were *Andesiops peruvianus* (Ephemeroptera), *Smicridea* sp. (Trichoptera), and *Eukiefierella* sp. (Diptera) at the agricultural land-use sites, and *A. peruvianus*, *Meridialaris diguillina* (Ephemeroptera), and *Eukiefierella* sp. (Diptera) at the remaining three land-use catchment areas.

Independent of land-use, all of the analyzed sites generally displayed high macroinvertebrate diversity and little dominance, with the most abundant species being ephemeropterans, plecopterans, and dipterans (Table 4). However, both the diversity and dominance parameters showed statistically significant differences between the distinct land-uses catchments (one-way PERMANOVA, $p < 0.001$, Table 2). More specifically, agricultural sites had the highest density and a dominance of chironomids, in addition to presenting the lowest Shannon Index value (1.98) and lowest richness (22 taxa) among the studied catchment areas. Moreover, many sensitive insects species (e.g. plecopterans *Udamocercia* sp., *Diamphipnosis samali*, ephemeropterans *Chiloporter eatoni*, *Murphyella needhami*, and trichopterans *Parasericostoma* sp., *Dolophilodes* sp.) were absent at agricultural sites but present in large numbers at pristine forest sites.

Looking at FFGs, 39 taxa were identified as collector-gatherers, 18 as shredders, 34 as predators, 7 as collector-filterers, 13 as scrapers, and 2 as parasites (Fig. 4). According to SIMPER analysis, the collector-gatherers made up the largest proportion (>50%) across all land-use catchment areas, followed by shredders (≈19%) and predators (≈13%). In contrast, collector-filterers, scrapers, and parasites were found in low proportions. Collector-gatherers, characterized by ephemeropterans and chironomids, were found in abundance across all land-use sites. Shredders, dominated by the austroperlid *Klapopteryx armillata* and gripoterygid *Limnoperla jaffueli*, were most represented within the pristine and native forest sites. Collector-filterers, mainly the hydropsychid *Smicridea* sp., were most abundant at agriculture sites. Scrapers were abundant at the pristine forest sites, with the majority classified as the coleopteran *Tychepsephenus felix* and gastropods

Table 3
Physical and chemical characteristics of 21 sampled sites across four different land-uses in South-Central Chile. Values represent averages of four seasons (November 2010, 2011, and January 2011, 2012) ± SD. Sites codes: PF: Pristine forest, NF: Native forest, FP: Forest plantation, AGR: Agriculture.

Sites	Temperature (°C)	pH	Electrical conductivity (μs cm^{-1})	Suspended solids (mg L^{-1})	Dissolved oxygen (mg L^{-1})	DBO5 (mg L^{-1})	Total dissolved solids (mg L^{-1})	Chlorides (mg L^{-1})	Sulfates (mg L^{-1})	True color (Pt Co^{-1})	Nitrates (mg L^{-1})	Phosphates (μg L^{-1})
PF-1	14.5 ± 5.2	7.5 ± 0.1	27.4 ± 3.1	0.5 ± 0	9.5 ± 2.4	4.2 ± 1.4	27.5 ± 3.5	10.5 ± 0.7	0.7 ± 0.1	10.5 ± 6.4	1.4 ± 1.2	51.3 ± 61.6
PF-2	15.4 ± 4.3	6.9 ± 0.1	30.6 ± 0.8	0.5 ± 0	9.1 ± 1.4	3.4 ± 1.6	30.5 ± 0.7	11.5 ± 0.7	1.4 ± 0.6	9.0 ± 4.4	1.7 ± 1.4	47.1 ± 50.2
NF-1	14.8 ± 2.1	6.7 ± 0	23.1 ± 0.1	2.9 ± 1.3	9.6 ± 0.2	1.6 ± 0.4	9.0 ± 0.2	13.7 ± 1.0	1.2 ± 0.4	19.4 ± 10.0	1.2 ± 0.2	96.1 ± 40.1
NF-2	14.9 ± 2.0	6.8 ± 0.2	25.1 ± 1.3	1.9 ± 0.3	9.4 ± 1.0	1.5 ± 0.1	10.0 ± 0.2	10.3 ± 0	3.9 ± 0.4	19.4 ± 10.1	0.8 ± 0	65.1 ± 39.9
NF-3	14.7 ± 0.9	6.9 ± 0.4	34.8 ± 1.8	2.1 ± 2.6	9.7 ± 0.5	1.9 ± 0.4	24.0 ± 12.2	13.0 ± 1.0	1.2 ± 0.6	26.6 ± 11.1	0.5 ± 0.3	45.3 ± 46.0
NF-4	15.5 ± 0.8	7.2 ± 0.5	38.0 ± 4.0	3.2 ± 1.6	9.6 ± 0.2	3.0 ± 2.0	27.5 ± 15.1	12.3 ± 1.4	1.2 ± 0.3	20.6 ± 4.8	1.2 ± 1.0	48.1 ± 45.2
NF-5	15.6 ± 1.0	6.6 ± 0.3	34.6 ± 0.4	6.5 ± 2.4	9.7 ± 0.1	2.2 ± 0.8	14.0 ± 0.3	12.0 ± 1.0	1.0 ± 0.2	19.4 ± 9.9	0.9 ± 0.1	65.1 ± 44.3
NF-6	14.7 ± 2.5	6.7 ± 0.1	35.6 ± 1.3	4.1 ± 2.7	9.6 ± 0.5	1.7 ± 0.2	14.5 ± 0.7	12.0 ± 0.9	1.2 ± 0.1	19.4 ± 10.0	1.0 ± 0.1	68.2 ± 42.3
NF-7	13.2 ± 1.8	6.7 ± 0	42.2 ± 2.6	4.6 ± 4.2	10.1 ± 0.4	2.3 ± 0.4	17.0 ± 1.4	12.0 ± 1.0	0.9 ± 0.1	19.4 ± 9.0	1.1 ± 0	127.0 ± 56.5
NF-8	13,7 ± 2.1	6.7 ± 0.4	37.9 ± 1.1	2.4 ± 0.4	10.0 ± 0	3.1 ± 0	15.0 ± 0.3	8.6 ± 0.5	0.9 ± 0.1	19.4 ± 10.1	0.9 ± 0	83.7 ± 40.3
FP-1	14.1 ± 2.1	6.9 ± 0.6	42.2 ± 3.5	3.9 ± 1.6	10.1 ± 0.5	2.4 ± 0.5	30.0 ± 16.8	13.8 ± 2.9	1.0 ± 0.1	29.0 ± 7.0	2.9 ± 1.4	49.1 ± 44.3
FP-2	13.1 ± 1.6	6.3 ± 0	41.5 ± 1.2	9.2 ± 8.3	9.5 ± 0.3	1.7 ± 0.5	16.5 ± 0.7	16.3 ± 0	1.0 ± 0	31.1 ± 12.1	1.3 ± 0.1	91.5 ± 45.5
FP-3	14.6 ± 2.8	7.1 ± 0.5	42.1 ± 2.1	5.9 ± 3.1	10.1 ± 1.0	2.1 ± 0.7	30.0 ± 15.0	12.8 ± 2.0	0.9 ± 0.2	21.7 ± 8.8	3.2 ± 2.3	60.1 ± 48.6
FP-4	12.9 ± 0.8	6.5 ± 0.2	59.0 ± 3.3	6.3 ± 1.1	9.8 ± 0.2	2.1 ± 0.4	24.0 ± 1.4	20.0 ± 1.5	0.8 ± 0	24.8 ± 11.9	1.6 ± 0.1	29.8 ± 20.2
FP-5	11.9 ± 2.5	7.1 ± 0.1	22.8 ± 0.5	5.7 ± 1.6	10.9 ± 1.6	2.9 ± 0.8	24.0 ± 1.4	18.2 ± 4.5	1.9 ± 1.5	28.0 ± 11.3	1.9 ± 0.1	48.2 ± 57.3
FP-6	13.1 ± 3.3	7.1 ± 0	30.6 ± 2.3	2.5 ± 0.8	10.0 ± 0.8	2.8 ± 0.5	22.0 ± 14.1	14.5 ± 7.8	1.0 ± 0.1	18.9 ± 1.5	1.9 ± 0.5	53.6 ± 41.1
FP-7	15.7 ± 4.0	7.0 ± 0.3	31.9 ± 3.3	3.1 ± 2.1	7.6 ± 0.8	5.3 ± 2.1	32.5 ± 3.5	11.5 ± 2.1	1.1 ± 0.2	42.2 ± 10.3	2.2 ± 1.3	96.4 ± 21.1
FP-8	14.8 ± 1.4	6.8 ± 0.1	74.7 ± 2.3	18.2 ± 0.4	9.5 ± 0.2	1.9 ± 0.8	30.0 ± 1.4	17.2 ± 1.4	1.1 ± 0	56.7 ± 14.4	2.8 ± 0.5	121.0 ± 50.5
AGR-1	15.5 ± 1.3	6.5 ± 0.2	65.2 ± 0.4	20.0 ± 1.2	8.5 ± 0	1.4 ± 0.6	26.0 ± 0	16.3 ± 1.4	1.1 ± 0	47.0 ± 13.2	2.7 ± 0.3	62.0 ± 34.3
AGR-2	14.5 ± 1.6	6.6 ± 0.2	56.0 ± 1.1	14.9 ± 0.7	9.2 ± 0.4	1.6 ± 0.5	22.5 ± 0.7	13.7 ± 1.3	1.2 ± 0.1	54.7 ± 15.4	2.0 ± 0.2	58.9 ± 32.2
AGR-3	15.8 ± 2.3	7.0 ± 0.2	121.9 ± 3.9	11.6 ± 5.4	7.7 ± 0.7	1.4 ± 0.4	49.0 ± 1.4	18.0 ± 2.2	2.8 ± 0.4	45.0 ± 12.2	5.6 ± 0.1	55.8 ± 30.1

Chilina dombeyana and *Littoridina cumingi*. Predators were most prevalent at the native forests sites and were characterized by the dipteran genera *Atherix* and *Limonia*. The composition of FFGs was relatively uniform across all land-use catchment areas (Fig. 4). The one exception was that agricultural sites had higher abundances of collector-filterers than the remaining land-use types (Fig. 4). Notwithstanding this difference, there were no significant differences in the overall composition of FFGs among the four land-use catchment areas (one-way PERMANOVA, $p > 0.05$, Table 2).

3.3. Riparian quality and Hilsenhoff biotic indices

Values for the QBR ranged from very good riparian quality (100) to very bad riparian quality (10). The pristine forest sites had very good

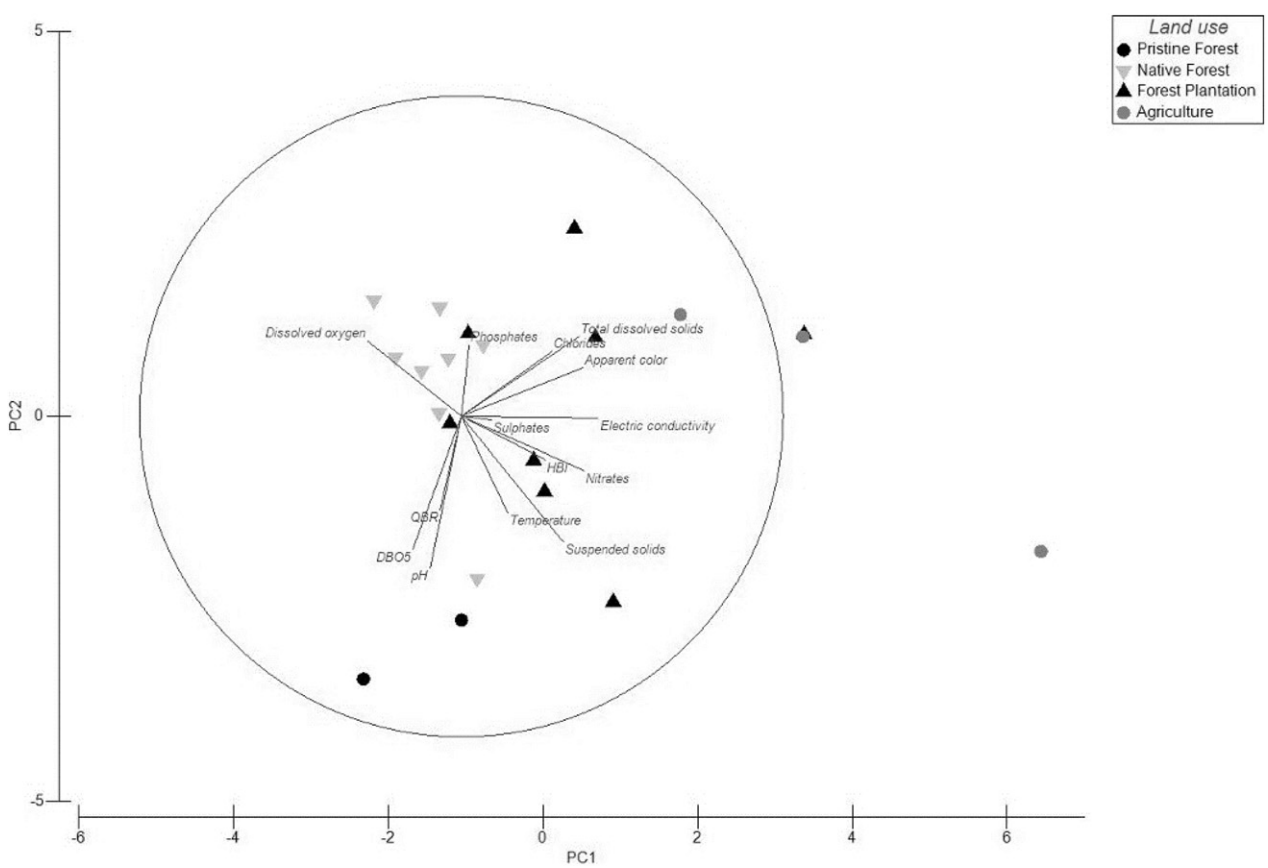

Fig. 2. Principal component analysis (PCA) of environmental variables at 21 sampling sites grouped by land use.

Table 4
Richness (S), mean density (ind·m^{-2}) (N), Shannon diversity index (H′), Dominance (D) and dominant species collected at the 21 sampling sites in South-Central Chile. Dominant species codes: Eph, Ephemeroptera, Plecop: Plecoptera, Dip: Diptera, Coleop: Coleoptera.

Site	S	N	H′	D	Dominant species
PF-1	53	1861	2.88	0.21	*Meridialaris diguillina* (Eph)
PF-2	55	1612	3.14	0.14	*Meridialaris diguillina*(Eph)
NF-1	44	2449	2.67	0.20	*Klapopteryx armillata* (Plecop)
NF-2	35	1070	2.87	0.17	*Eukiefferiella* sp.(Dip)
NF-3	54	2628	2.82	0.20	*Hexatoma* sp. (Dip)
NF-4	50	1374	2.53	0.24	*Meridialaris diguillina* (Eph)
NF-5	42	2586	2.48	0.25	*Penaphlebia chilensis* (Eph)
NF-6	32	1722	2.15	0.34	*Rheotanytarsus* sp. (Dip)
NF-7	31	1157	2.40	0.25	*Andesiops peruvianus* (Eph)
NF-8	33	2264	1.96	0.51	*Corynoneura* sp. (Dip)
FP-1	54	1057	2.65	0.38	*Meridialaris diguillina* (Eph)
FP-2	37	646	2.82	0.16	*Meridialaris diguillina* (Eph)
FP-3	52	1587	2.59	0.33	*Meridialaris diguillina* (Eph)
FP-4	26	506	2.38	0.23	*Meridialaris diguillina* (Eph)
FP-5	44	1748	2.72	0.22	*Eukiefferiella* sp. (Dip)
FP-6	53	1655	2.81	0.29	*Austrolimnius* sp.(Coleop)
FP-7	53	1826	2.97	0.18	*Andesiops peruvianus* (Eph)
FP-8	37	1341	2.59	0.17	*Andesiops peruvianus* (Eph)
AGR-1	22	951	2.26	0.24	*Andesiops peruvianus* (Eph)
AGR-2	30	1973	2.41	0.21	*Andesiops peruvianus* (Eph)
AGR-3	22	2878	1.98	0.31	*Eukiefferiella* sp. (Dip)

riparian quality. In contrast, the native forest, exotic forest plantation, and agriculture sites had highly variable QBR values that ranged between fair and very bad riparian conditions. Only one site in the native forest catchment area had a good riparian quality (Fig. 5). Statistically significant differences were recorded for QBR values among the land-use catchment areas (one-way PERMANOVA, $p < 0.01$, Table 2).

In turn, HBI values indicated excellent, very good, and good water qualities. The best water quality values were found in the pristine forest catchment area (2.52), while the lowest water quality values were found at agriculture land-use sites (4.97; Fig. 5). Intermediate values (i.e. very good water quality) were obtained for sites within the native forest and exotic forest plantation catchment areas. Despite the different HBI values obtained between pristine forest and agricultural sites, no significant differences were found among the distinct land-use categories (one-way PERMANOVA, $p > 0.05$, Table 2). Subsequent PCA (Fig. 2) clearly associated high HBI values with higher nitrate values. This result would be expected since sites with high nitrate contents had more organic-pollution-tolerant species (e.g. Chironomidae, Janiridae, Tubificidae), thus leading to higher HBI scores.

A comparison of the QBR and HBI revealed a very low, non-significant correlation ($R^2 = 0.01$, $p = 0.64$), indicating little relationship between riparian quality (i.e. QBR) and water quality (i.e. HBI). Low riparian quality did not translate into low water quality, as measured by bioindicators. Furthermore, even when QBR values were high, HBI results rarely indicated excellent water quality (Fig. 6).

4. Discussion

The results of the conducted research support the negative impact that land-use changes can have on freshwater ecosystems, particularly in catchment areas. Significant alterations were detected in the physicochemical parameters for water quality, in riparian vegetation quality, and in macroinvertebrate assemblages. However, the modified HBI (i.e. adapted to Chilean species) and FFGs were similar between the different land-use catchment areas.

Studies across South America indicate that agricultural practices have intensified in intermediate and lower catchment areas, often extending to river banks where water quality is then negatively impacted (Hunke et al., 2015; Ribbe et al., 2008). While the effects of a reduced or substituted riparian corridor were not evaluated, all sampling sites did contain riparian vegetation, which varied in quantity and quality among the different land-use catchment areas. Streams in the agricultural land-use catchments were significantly more affected than streams in other land-use catchment areas. For example, aquatic ecosystem health was best at sites within the pristine forests. Considering that most land-use changes in the catchment areas of southern South America occur due to exotic forest plantations and agricultural lands, the results obtained by this study can be applied to similar catchment areas within this region of the world.

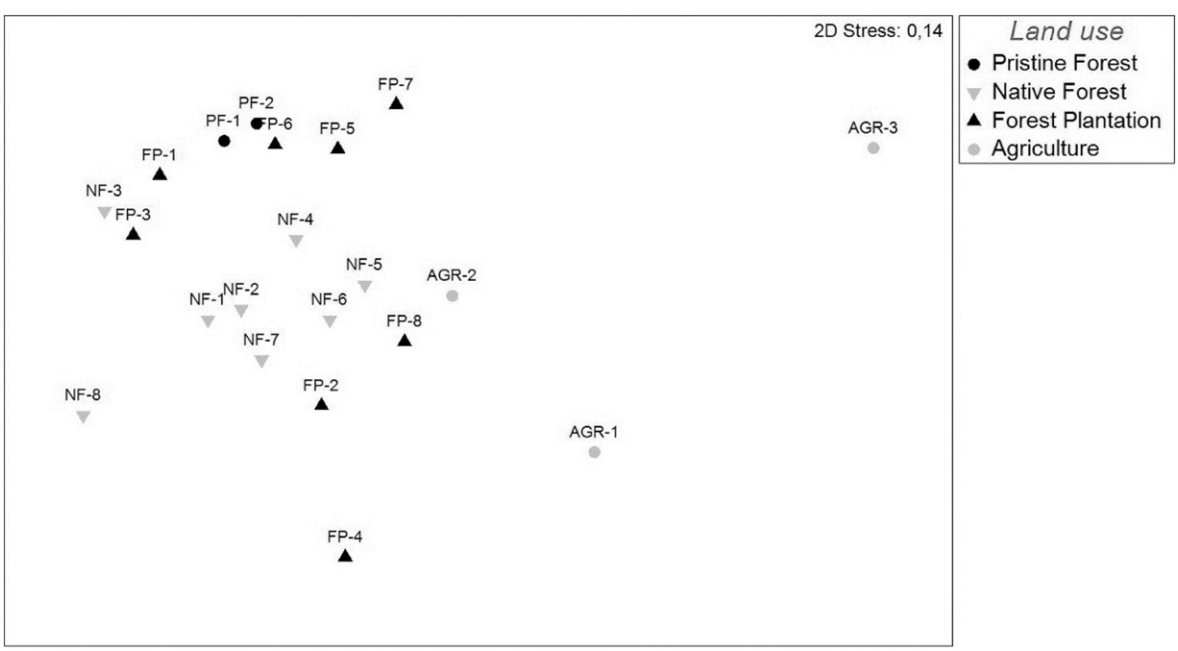

Fig. 3. Non-metric multidimensional scaling (nMDS) plot using Bray-Curtis similarities for macroinvertebrates assemblages from 21 sampling sites, grouped by land use, in South-Central Chile. Axes appear without legend, because they are relative scales.

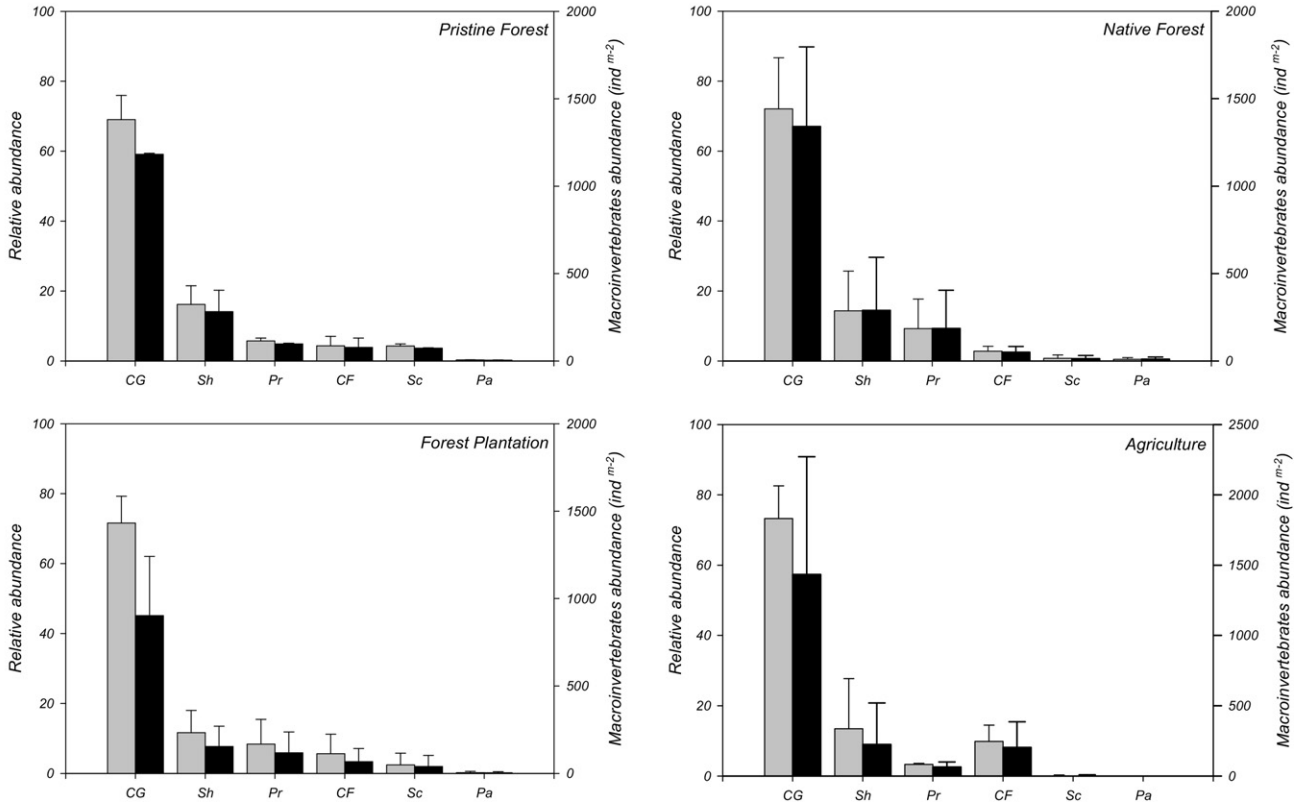

Fig. 4. Relative abundance (grey bar, %) and abundance (black bar, ind·m^{-2}) of functional feeding groups recorded in 21 sampling sites across four different land uses in South-Central Chile. The FFG are, Sh: shredders, CG: collector-gatherers, Pr: predators, CF: collector-filterers, Sc: scrapers, and Pa: parasites.

4.1. Physicochemical factors

Rivers that drain agricultural catchments are characterized by high levels of conductivity, nutrients, and suspended solids (Gerth et al., 2017; Stefanidis et al., 2015). Prior research found these associations in the presently assessed exotic forest plantation catchments (Fierro et al., 2016). The current study further established that high conductivity, nutrient, and suspended solids values were associated with agricultural lands and exotic forest plantations, indicating that both of these land-use types can negatively affect physicochemical water quality. Another characteristic of rivers in agricultural catchments is comparatively elevated water temperatures (Gerth et al., 2017). Even though the present study showed that rivers draining agricultural catchments had higher water temperatures than other land-uses, likely as a result of scarce vegetative coverage, the recorded water temperatures were not excessively higher than the other assessed catchments. Although agricultural sites had low QBR values, the mere presence of riparian vegetation, and associated share, would likely explain the maintenance of reasonable water temperatures.

4.2. Macroinvertebrate assemblages

The change in land-use from native forests to exotic forest plantations is responsible for modifications to the taxonomic composition of macroinvertebrate assemblages, with greater species richness and abundances in native forest catchments versus exotic forest plantations catchments (Fierro et al., 2016). The present study corroborates land-use change impacts on macroinvertebrate assemblages, with significant differences found in taxonomic composition and species richness between the assessed sites. Streams in the pristine forest had greater macroinvertebrate biodiversity than those in the native forest. In turn, macroinvertebrate diversity was higher in native forest streams than in exotic forest plantation streams, and agricultural streams had the lowest biodiversity levels.

One consequence of changes in riparian vegetation is a loss of habitat diversity, with rivers in agricultural catchments lacking macroinvertebrate refuges (Orlinskiy et al., 2015). In the current investigation, agricultural land-use catchments had the lowest species richness and the highest abundances of non-insects. These results are similar to patterns reported for other streams in the Araucanía Region, Chile (Fierro et al., 2015) and for temporary streams in Oregon, USA (Gerth et al., 2017). The collected data reaffirm the idea that agricultural practices appear to be one of the major causes for aquatic ecosystem deterioration (Hunke et al., 2015); streams draining agricultural catchments have low macroinvertebrate diversities, especially of insects, which is related to changes caused by agricultural activities in the surrounding landscape. These findings mirror the larger global issue of land-use conversions to agricultural lands, which can lead to potentially drastic increases in the extinction rates of sensitive aquatic insects (Thomas, 2005).

4.3. Functional feeding groups

Riparian vegetation plays a key role in landscape ecology, providing shade, habitat diversity, and allochthonous nutrients. Changes in adjacent vegetation can therefore alter the amount of organic material entering aquatic systems, subsequently modifying the trophic structure of aquatic food webs (Dosskey et al., 2010). In the present study, FFGs were dominated by collector-gatherers. This pattern aligns with other rivers in South America, thus emphasizing the importance of this group in processing coarse particulate organic matter to make it available to other invertebrate FFGs (Ferru and Fierro, 2015; Miserendino, 2007). Collector-gatherers were abundant across all assessed land-use types. This abundance might be explained by the presence of riparian vegetation at all of the sampling sites, which suggests that food

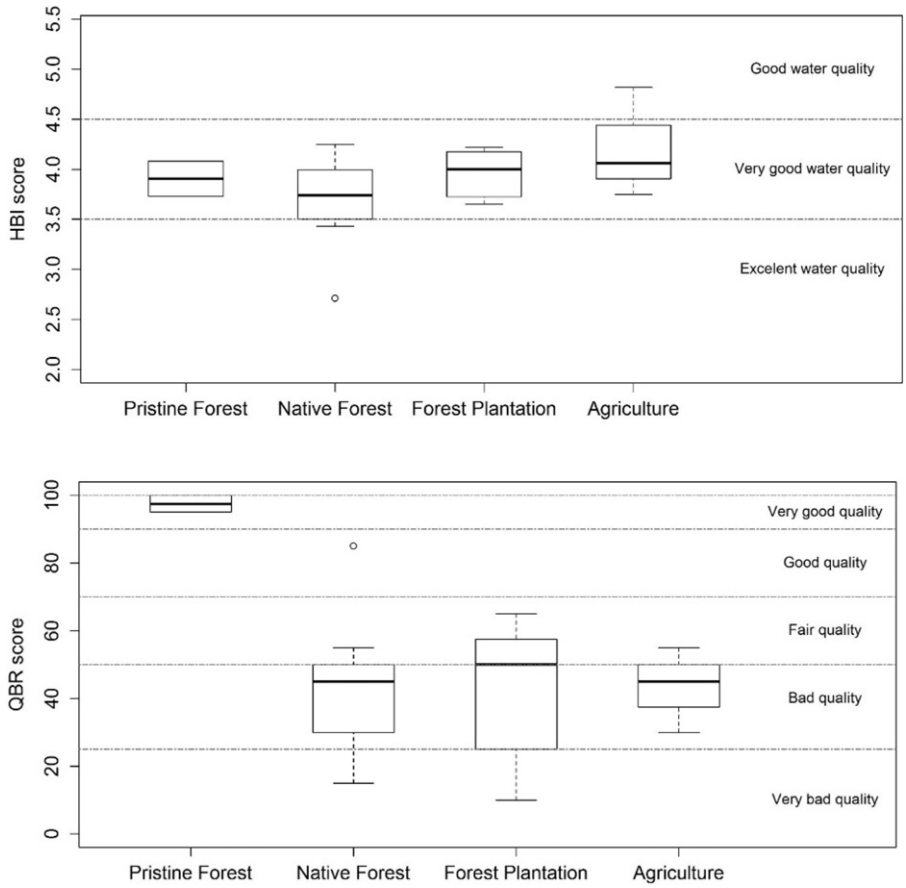

Fig. 5. QBR and HBI score for each sampling site grouped by land use (PF, pristine forest; NF, native forest; FP, forest plantation; AGR, agriculture). Dotted lines indicate QBR and HBI category thresholds.

availability may not be a limiting factor. However, QBR varied between the evaluated land-use categories, with rivers in exotic forest plantations and agricultural catchments mainly composed of exotic tree species.

In accordance with Fierro et al. (2016), who also assessed streams in southern Chile, no differences in FFGs were recorded among the distinct types of land-use. Coastal Chilean streams are low-order and have a short source-mouth distance, traits that result in a steep gradient.

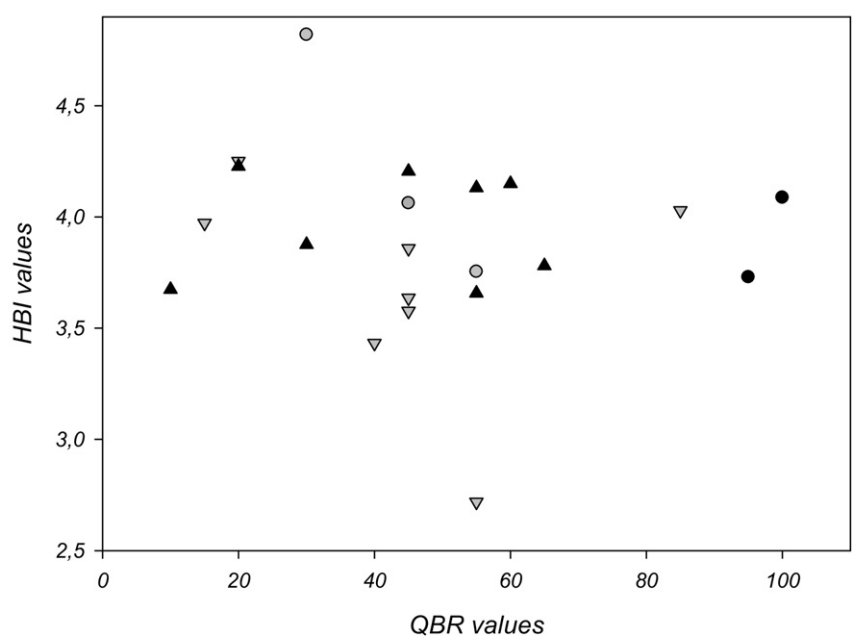

Fig. 6. Relationship of HBI and QBR indices from 21 sampling stations across four different land-uses in South-Central Chile. Sites pristine forest (●), native forest (▼), forest plantation (▲) and agriculture (●).

Therefore, organic matter from riparian vegetation present in headwaters could be transported downstream and could be used to provide refuge or feed to FFGs (Mesa, 2014). Another important factor to consider is that riparian vegetation present within exotic forest plantation and agricultural land-use catchment would contribute coarse woody debris to streams, which can be processed by some macroinvertebrate FFGs (Valdovinos, 2001). The poor riparian quality recorded for sites evaluated within the agricultural catchment area could result in an input of fine particulates. This, in turn, would favor collector-filterers that capture and feed on fine particulates from the flowing water column. Indeed, the present study recorded increased collector-filterer densities at agricultural stream sites. Similar findings have been reported elsewhere for disturbed streams, especially those in agricultural land-use areas (Miserendino and Masi, 2010).

Shredders were the second most represented FFG in the pristine forest and native forest sites. Nevertheless, the representation of this FFG was less than expected. In nearby catchments, the relative abundance of shredders exceeded 20% in some streams (Fierro et al., 2015). Differences between studies within similar geographical areas could be due to variations in sampling periods. Fierro et al. (2015) assessed seasonal samples throughout the year, reporting high relative abundances of shredders during rainy periods, which is a period when organic material inputs to the river increase. In contrast, the present study only considered the dry seasons, when low-order streams generally have peak collector-gatherer abundances (Roeding and Smock, 1989; Stoaks and Kondratieff, 2014).

4.4. Riparian quality and Hilsenhoff biotic indices

As expected, rivers located in pristine forests had the highest QBR values, indicating the best riparian quality. Variations in QBR values among native forest sites were likely due to illegal, small-scale logging at some catchment area sites, which would produce depressed QBR values. Riparian zones in exotic forest plantations presented conditions similar to those in native forests. This could be because of exotic pine and eucalyptus plantations often forming part of the assessed riparian vegetation, leading to low index values. Finally, the low QBR values for agricultural catchment areas can be explained by proximal agricultural activities, which occurred up to the river margins, and by the abundant presence of exotic species, such as willows (*Salix* spp.).

The importance of riparian vegetation corridors in maintaining biodiversity has been demonstrated for both terrestrial and aquatic species (Naiman et al., 1993). In general, rivers with broad vegetative buffers are more effective in maintaining and improving aquatic biodiversity and water quality (Iñiguez-Armijos et al., 2014). Bioindicators are a quick and low-cost tool for evaluating water quality. Specifically, aquatic macroinvertebrates have been widely used to assess water pollution. These organisms are relatively easy to sample, very sensitive, have low mobility levels, and can indicate environmental statuses for preceding months (Alba-Tercedor, 1996; Hilsenhoff, 1988). In this study, the compositions of invertebrate assemblages significantly differed between the assessed land-use categories. In effect, many sensitive insects species were absent at agricultural sites but present at the remaining land-use sites, indicating a possible local extirpation of these species in agricultural zones. However, use of the modified HBI to evaluate water quality did not result in reliable data for environmental monitoring. Even though the HBI detected good to excellent water qualities, the pristine forest sites did not obtain the highest water quality score, contradicting the result that was expected. The modified HBI, as with other indexing tools, can obviously err, particularly when, as in the present study, family-level classifications are used instead of genus- or species-level classifications. Nevertheless, when used with other indices, the modified HBI remains a useful tool.

Indeed, the QBR and HBI have been successfully adapted and applied to research in Chile and across South America, providing valuable insights into riparian vegetation and water qualities (Carrasco et al., 2014; Fernández et al., 2009; Fierro et al., 2012; Kutschker et al., 2009). The present study predicted a strong relationship between these two indices, especially in areas where riparian zone deterioration would lead to increased nutrient influxes and, potentially, increased organic pollution (Strand and Merritt, 1999). However, the indices were not related, indicating that neither index can be used to provide an overall measurement of aquatic ecosystem health in the studied catchment areas.

5. Conclusions

The results of this study strongly support that physicochemical parameters and biological characteristics are significantly modified as the result of anthropogenic land-use changes. The evaluated land-use changes resulted in deteriorated riparian vegetation, which can modify the trophic ecology of rivers and, subsequently, significantly alter the physicochemical and biological characteristics of aquatic ecosystems. This is of great importance in a country such as Chile, where there exists a general tendency to replace native forests with exotic forest plantations and agricultural/grazing lands (Manuschevich and Beier, 2016; Nahuelhual et al., 2012).

Macroinvertebrate assemblages were applied as a powerful tool in establishing the ecological effects of different land-uses. Sampling sites in the pristine forest catchment area had the greatest diversity of macroinvertebrates and sensitive species. In contrast, sites in the agricultural catchment area presented considerably lower macroinvertebrate richness and a complete lack of some sensitive species. Intermediate diversity was recorded for sites within the native forest and exotic forest plantation catchment areas. Collector-gatherers were largely abundant across all assessed land-uses, indicating the importance of riparian vegetation in acting as a buffer against human activities that directly impact aquatic ecosystems. Riparian vegetation also importantly provides nutrition, refuge, and habitats for various species. In turn, differences were not found in functional feeding groups across the distinct land-use catchments, a result suggesting that this may not be a good indicator for ecosystem health.

The application of riparian quality (QBR) and water quality (HBI) are relevant in determining the health of riparian and aquatic ecosystems. Both the QBR and HBI describe ecosystem health and can be used as resource-planning tools. However, each provides a distinct understanding of existing ecological conditions, providing measurements for very different variables. As such, these indices should be used in conjunction with biological and physicochemical assessments to ensure gaining clear understands for the status of aquatic and terrestrial ecosystems. Since both indices work in different ways, the development of a multimetric index able to assess overall ecosystem in a single value would be invaluable for future research.

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