



## Rainbow Trout diets and macroinvertebrates assemblages responses from watersheds dominated by native and exotic plantations



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### ABSTRACT

Over the past few decades, land-use changes through conversion of global forest cover to exotic plantations is contributing to both habitat and biodiversity loss and species extinctions. To better understand human influences on ecosystem, we use diet composition from introduced Rainbow Trout *Oncorhynchus mykiss* as indicator of potential changes in the composition of stream-macroinvertebrates due to land use changes from native to exotic vegetation (eucalyptus plantations) in southern Chile. Water quality variables, aquatic macroinvertebrates and Rainbow Trout diet were studied in 12 sites from mountain streams located in two watersheds including one dominated by native riparian vegetation and the other dominated by exotic vegetation. As expected, richness and abundance of macroinvertebrates were clearly higher at sites in native forest than in those with exotic vegetation. Collector-gatherer was the most abundant functional feeding group, but there was no statistical difference in the functional composition between the two watersheds. Differences in in-stream macroinvertebrate availability was more higher correlated with changes in Rainbow Trout diets. Specifically, taxa consumed from the watershed dominated by native forests was higher than from the watershed with exotic vegetation. Additional environmental variables showed statistical differences between watersheds. The exotic vegetation sites had the highest concentrations of dissolved solids, suspended solids, nitrates, chlorides and sulphates. Our findings show that macroinvertebrate assemblage structure and trout diets can be altered by changes in riparian vegetation. The absence of specific macroinvertebrate taxa in streams with exotic vegetation was captured by the composition of trout diets. This suggest that Rainbow Trout diets can be a good biological indicator of land use practices and thus, diet can be used as a rapid and effective tool for evaluate environmental quality. Our findings provide insights about the design of aquatic monitoring programmes to improve detection of anthropogenic impacts in streams in South America and elsewhere.

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### 1. Introduction

Freshwater ecosystems are among the most seriously threatened in the world (Saunders et al., 2002; Barletta et al., 2010). During the recent decades, the loss of freshwater biodiversity has been accentuated mainly due to changes in land use from human-related activities (e.g., forestry and livestock or arable farming) that have resulted in habitat destruction, fragmentation and eutrophication (e.g., Encalada et al., 2010; Miserendino et al., 2011; Lunde

and Resh, 2012). In particular, because of the economic benefit from the cellulose industry (Valdovinos, 2006), the replacement of native forest by plantations of exotic species (i.e., monocultures of conifers and eucalyptus) has been a widespread forestry practice all over the world (Hartley, 2002).

In headwaters of forested watersheds, riparian vegetation is a major source of energy and nutrients for stream food webs through the introduction of dead leaves and large woody debris (Vannote et al., 1980). In these environments, the relatively high velocity of water and extensive shade from the canopy limit the autochthonous production (Vannote et al., 1980; Wallace et al., 1997). Therefore, modifications of riparian vegetation can modify the quality of leaf-litter inputs and alter processes in aquatic

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ecosystems such as the trophic structure and composition of aquatic communities (Abelho and Graça, 1996; Martínez et al., 2013).

Since the beginning of the 19th century, varying degrees of anthropogenic disturbance along coast of southern Chile (southern South America) have occurred (Peña-Cortés et al., 2011a). This includes an over-exploitation of the soil and the replacement of the native forest by agriculture, urbanisation, and plantations of exotic tree species (Sala et al., 2000; Peña-Cortés et al., 2006; Aguayo et al., 2009). The consequences of these activities upon aquatic food webs are still not well understood. Recently, it has been reported that among the most threatened communities by such changes in land use are the benthic aquatic macroinvertebrates (Fierro et al., 2015). A few studies conducted in headwaters (e.g., Larrañaga et al., 2009; Miserendino and Masi, 2010) have shown higher shredder richness in streams dominated by native forest compared to streams dominated by exotic plantations. More recently, Fierro et al. (2015) showed higher invertebrate densities and richness in streams dominated by native forest. Because macroinvertebrates assemblages represent intermediate trophic links between primary and tertiary consumers (Jensen et al., 2012; Bertrán et al., 2013; Cornejo-Acevedo et al., 2014; Fierro et al., 2014) as fish food sources, their availability can affect fish carrying capacity of these low-to-medium order streams populations. If prey availability is limiting, prey fish would be affected (Pequeño et al., 2010). Therefore, any change in the assemblage of macroinvertebrates, would result in changes in the functioning of aquatic ecosystems and restructuring of food chains (Richards et al., 1996; Vargas-Chacoff et al., 2013; Tiziano et al., 2014).

Further, non-native fish introductions represent one of the greatest threats to freshwater ecosystems in southern Chile (Arismendi et al., 2014). In this region, salmonids have been introduced into freshwaters, mainly for recreational fisheries and aquaculture purposes (Arismendi et al., 2014). Rainbow Trout (*Oncorhynchus mykiss*, Walbaum) is one of the most successful introduced species, and currently it is widely distributed in southern South America, reaching higher abundances than native fishes (Arenas, 1978; Soto et al., 2006; Arismendi et al., 2012, 2014). Like other salmonids in the region, Rainbow Trout is known as generalist and largely opportunistic feeder (e.g., Arenas, 1978; Campos et al., 1984; Ruiz, 1993; Berrios et al., 2002; Palma et al., 2012; Arismendi et al., 2012; Vargas-Chacoff et al., 2013). Most of these studies have related the diet of Rainbow Trout with the availability of macroinvertebrates in the environment in summer, but few of them have investigated this across seasons (Buria et al., 2009; Di Prinzio et al., 2013).

The first goal of this study is to characterise macroinvertebrate assemblages and functional feeding groups from two land use types (native forest and exotic plantations). The second goal is to examine whether diets of Rainbow Trout can be used as predictors of macroinvertebrate assemblage composition from these two land use types. Collectively, our study provides an assessment of the influences of eucalypt plantations on macroinvertebrate functional feeding groups and fish diets. This could help to clarify how land use change may impact aquatic food webs, contribute to the development of management practices on freshwater ecosystems, and serve as a baseline for future investigations of ecological processes in streams under human-related disturbances.

## 2. Materials and methods

### 2.1. Study area

Field sampling was conducted seasonally during 2010, in summer (10–13 January), autumn (10–13 May), winter (10–13 August)

and spring (10–13 November) at the coastal zone of the Araucanía Region (Fig. 1). We sampled water quality, macroinvertebrates and stomach contents from streams between 2nd and 4th order ( $n=12$ ; Table 1, Fig. 1). The climate in this area is maritime with a Mediterranean influence; the average annual precipitation is between 1200 mm and 1600 mm (Di Castri and Hajek, 1976). The landscape geomorphology varies from mountain systems to marine abrasion platforms, with elevations ranging between 870 masl and –2 masl (Peña-Cortés et al., 2009, 2011b). Our sites encompassed two watersheds with varying land uses: the Moncul River located in the northern part of the region is dominated by forest practices on exotic species – mainly *Eucalyptus globulus* (Labill); the Queule River, located in the southern part of the region is dominated by forest practices on native forest, the dominant species being *Nothofagus dombeyi* (Oersted), *Nothofagus obliqua* (Oersted) and *Drimys winteri* (Forster & Forster). The study sites within each watershed were selected according to the proportion covered by riparian vegetation type, including up to 60% of exotic vegetation in the Moncul watershed, dominated by *Eucalyptus* spp., and up to 60% of native forest in the Queule watershed, dominated by *Nothofagus* spp. (Vargas-Chacoff et al., 2013). The eucalyptus plantations have mostly been planted during the last 20–25 years, while the native forest sections have been present for over 50 years.

### 2.2. Sampling

#### 2.2.1. Environmental characteristics

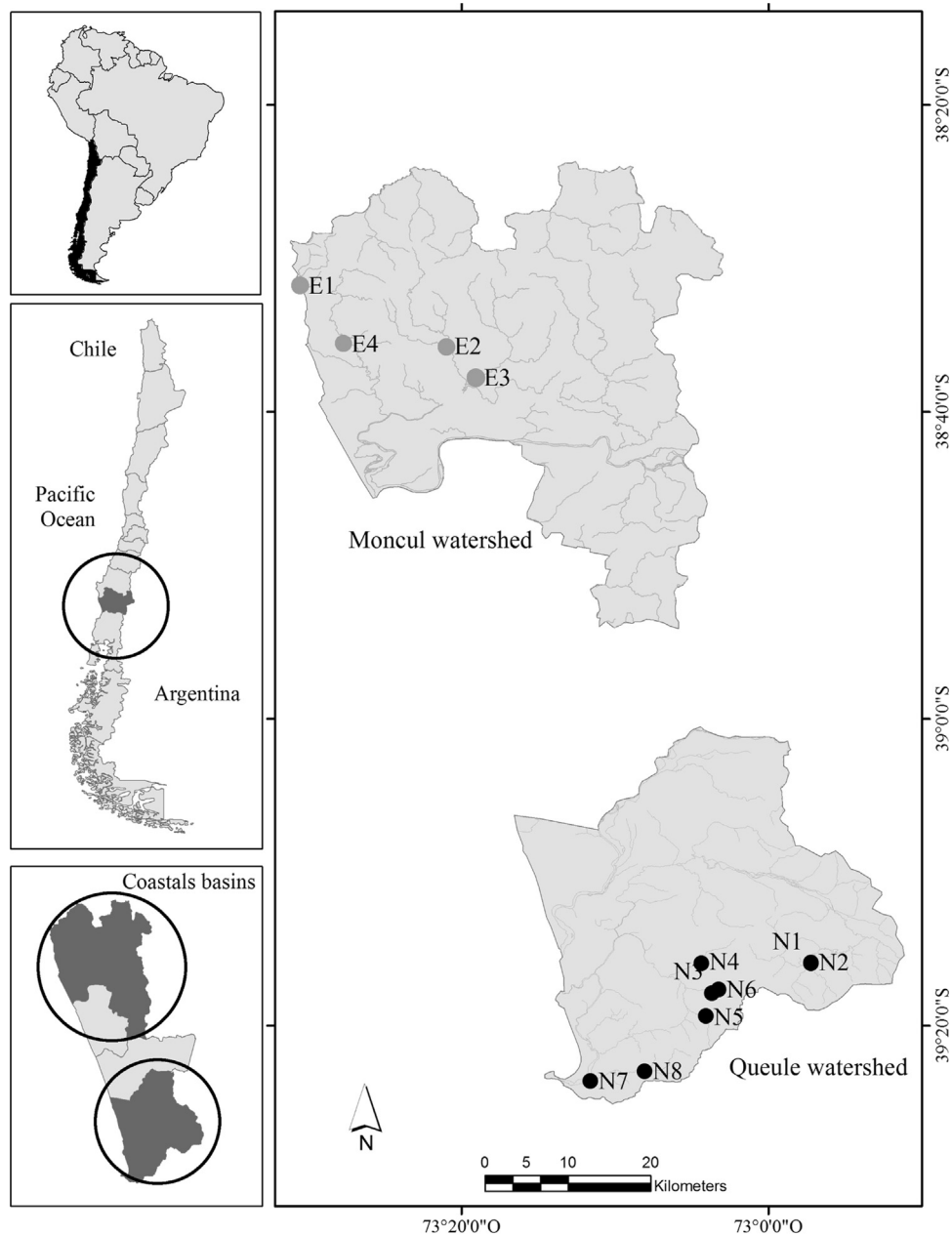
The water samples were collected in duplicate in the morning (8–11 AM) from the centre of the active channel, deposited in bottles and taken to the Analytical Chemistry Laboratory of the Institute of Chemistry and Natural Resources, Universidad de Talca, for the following parameters to be determined: bio-chemical oxygen demand, suspended solids, dissolved oxygen, chlorides, sulphates, dissolved solids, apparent colour, nitrates and phosphates. All the analyses were carried out following standard methods for water and wastewater (APHA, 2005). The temperature, pH and conductivity were measured in situ with a pH meter (WTW pH model 330i/SET), and a conductivity meter (WTW cond. Model 330i/SET).

#### 2.2.2. Availability of prey

Together with the water samples in each sampling station, three separate samples were taken in a zone of riffles (the most common habitat type) using a Surber net with 500  $\mu\text{m}$  mesh (0.09  $\text{m}^2$  area). The samples were fixed in situ with 90% ethanol and then taken to the Benthos Laboratory of the Institute of Marine and Limnological Sciences, Universidad Austral de Chile, where they were separated, identified and counted under stereo microscope (Olympus, model SZ 51, 40 $\times$ ) and optical microscope (Olympus, model CX 31, 100 $\times$ ) at lowest possible taxonomic resolution following Domínguez and Fernández (2009). The taxa identified were assigned to seven functional feeding groups (FFG): shredders, collector-gatherers, collector-filterers, grazers, predators, detritivores and parasites, following the criteria of Merritt and Cummins (1996) and Fierro et al. (2015).

#### 2.2.3. Fish sampling

Individuals of Rainbow Trout were captured using an electrofishing equipment (EFKO, model FEG 1000, 1 kW, 150–600 V) at the same sampling sites where the invertebrates were collected. The electrofishing method was carried out on a 100 m stretch of stream for 15 min. The fish captured were fixed and preserved in ethanol 90% and then transported to the Benthos Laboratory of the Institute of Marine and Limnological Sciences, Universidad Austral de Chile, where the individuals were measured (standard length, 0.1 mm) and weighed (0.001 g accuracy).



**Fig. 1.** Map of the study area and study sites ( $n = 12$ ) from two watersheds (Moncul watershed – exotic vegetation; Queule watershed – native vegetation) of the Araucanía Region in southern Chile.

#### 2.2.4. Diet of Rainbow Trout

The stomach contents extracted from each fish were emptied into a Petri dish. The prey organisms were removed and identified to the same taxonomic level as the benthic organisms. The contribution of each prey type was assessed using two methods following Hyslop (1980) including the frequency of appearance (%F), corresponding to the number of stomach samples containing each taxon, expressed as a percentage of the total stomach samples, and abundance of occurrence (%N), corresponding to the total number of individuals of each taxon expressed as a percentage of the total stomach samples.

To assess the particular contribution of each prey, the Index of Relative Importance (IRI) established by Pinkas et al. (1971), and used by other studies in the region (e.g., Bertrán et al., 2013;

Cornejo-Acevedo et al., 2014; Fierro et al., 2014), was applied. The relative importance of each food item was calculated as follows:

$$IRI = \%F \times \%N \times 100^{-1}$$

#### 2.2.5. Statistical analyses

Associations between the physical, chemical and biological data for each site within the two watersheds were examined using multivariate statistical procedures within the software package PRIMER V.6.1.2 software (Clarke and Gorley, 2006) and PERMANOVA v.1 software (Anderson et al., 2008). The physical–chemical data were first transformed (square root) and normalised, and a matrix of Euclidian distance was constructed. These transformed and normalised data were subjected to principal

**Table 1**  
Summary of watershed characteristics at the study sites ( $n = 12$ ) in southern Chile.

Land use (%)	Site code	Basin/Sub-basin	Watershed size (km <sup>2</sup> )	Stream order	Altitude (masl)	Active channel width (m)	Water velocity (m s <sup>-1</sup> )	Depth (m)	Substrate type <sup>a</sup>
<i>Moncul</i>									
>60% Exotic forestry species	E1	Danquil	19.23	2	19	3.50	0.49	0.80	Peb/Gra
	E2	Cabrero	26.15	3	36	5.00	1.55	0.60	Peb/Cob
	E3	El Peral	67.93	4	30	25.00	0.66	0.50	Peb/Cob
	E4	Puyanhue	108.53	4	125	15.00	0.55	0.90	Peb/Cob
<i>Queule</i>									
>60% Native forest	N1	Boldo River	308.31	4	99	20.00	0.90	0.50	Bou/Peb/Cob
	N2	Boldo River	308.31	3	101	5.00	1.70	0.30	Bou/Peb/Cob
	N3	Lovera stream	24.28	3	25	18.00	1.14	0.45	Peb/Cob
	N4	Ramírez stream	21.89	3	27	8.00	0.80	0.50	Peb/Cob
	N5	Boroa River	82.93	3	66	10.00	2.51	0.60	Bou/Peb/Cob
	N6	Lovera stream	24.28	3	25	6.00	1.78	0.70	Peb/Cob
	N7	Piren stream	48.15	3	26	8.00	1.50	0.90	Peb/Cob
	N8	Piren stream	48.15	3	25	18.00	1.08	0.50	Peb/Cob/Gra

<sup>a</sup> Bou: boulders, Peb: pebbles, Cob: cobbles, Gra: gravel.

component analysis (PCA) to order the sampling sites along the environmental gradient. To assess the degree of similarity between the sampling sites of the two watersheds by season, the biological data (abundance of macroinvertebrates, abundance of functional feeding groups and index of relative importance) were transformed (square root) in order to construct three Bray–Curtis similarity matrixes. To test significant differences ( $P < 0.05$ ) between the two watersheds by season, two-way fixed factors were used: watershed and season, which were tested using a Permutational multivariate analysis of variance (PERMANOVA; 9999 permutations). This nonparametric method is similar to the analysis of variance, using the permutations method to test the difference between groups (Anderson et al., 2008). A RELATE analysis (Clarke and Gorley, 2006) was used to determine the significance of the correlation between the Bray–Curtis similarity matrixes of the benthic data and the index of relative importance. A Spearman correlation ranking was used to determine the coefficient level between the two matrixes (benthic data and index of relative importance).

### 3. Results

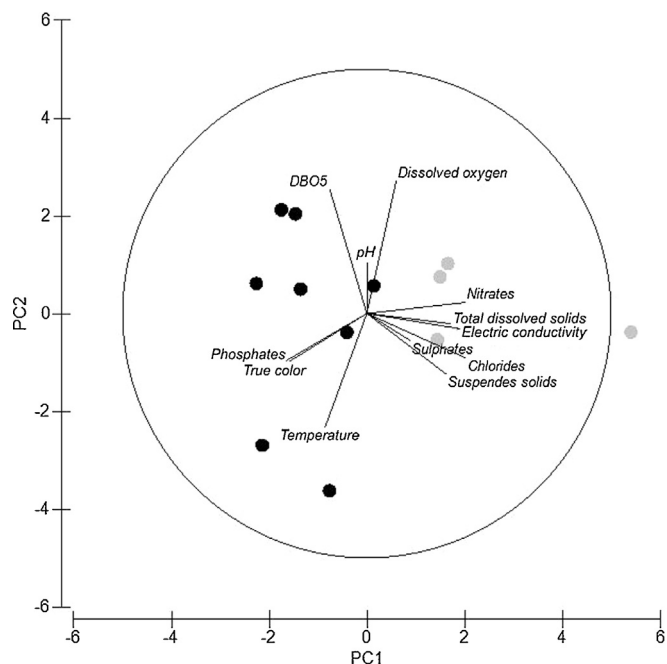
#### 3.1. Environmental characteristics

Chemical and physical data provided a clear distinction between native watershed sites and exotic watershed sites (Table 2, Fig. 2). Significant statistical differences between the watersheds was found (PERMANOVA:  $F = 1.010$ ,  $P = 0.001$ ). Of all the variables measured, in the PCA analysis, it was established that the strongest relationships with the sites in watersheds dominated by exotic vegetation were with total dissolved solids, suspended solids, nitrates, chlorides and sulphates (Fig. 2). These variables showed higher mean values at exotic watershed sites than at native watershed sites.

However, when the physical and chemical data were compared among seasons, we found non-significant statistical differences (PERMANOVA:  $F = 21.779$ ,  $P = 0.416$ ). A synthesis of physical and chemical data of the sites is presented in Table 2.

#### 3.2. Availability of prey invertebrates

A total of 103 taxa of macroinvertebrates were identified during the study time period (Appendix I), with the most represented



**Fig. 2.** Principal component analysis (PCA) of environmental variables at sites dominated by exotic vegetation (grey circles) and at sites dominated by native vegetation (black circles).

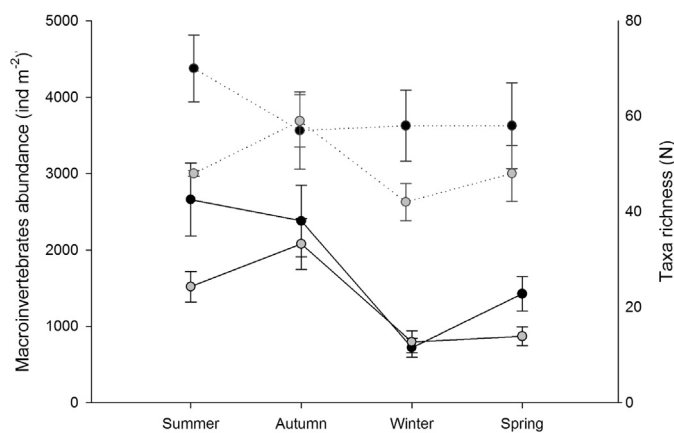
orders being Diptera (26%), Ephemeroptera (16%), Plecoptera (16%) and Trichoptera (16%). There was a significant statistical difference in the composition of macroinvertebrate communities between the two watersheds (Fig. 3) (PERMANOVA:  $F = 2.545$ ,  $P = 0.002$ ) and among seasons (PERMANOVA:  $F = 3.075$ ,  $P < 0.001$ ). In general, taxa richness and total density were lowest in the exotic vegetation watershed streams (richness: 42–59, density: 796–2.079 individuals/m<sup>2</sup>, respectively) and highest in the native vegetation watershed streams (richness: 56–70, density: 722–2.660 individuals/m<sup>2</sup>) (Fig. 3 and Appendix I). The lowest abundance of macroinvertebrates occurred in winter, while the highest abundances were recorded in summer and autumn at both watersheds.

Species diversity of benthic macroinvertebrates were different between watersheds, some species were recorded on native



**Table 2**  
Physical and chemical characteristics of streams across seasons. Values represent average ± SD.

Basin	Sites	Temperature (°C)	Electrical conductivity (µS cm <sup>-1</sup> )	Total dissolved solids (mg L <sup>-1</sup> )	pH	Suspended solids (mg L <sup>-1</sup> )	Dissolved oxygen (mg L <sup>-1</sup> )	DBO5 (mg L <sup>-1</sup> )	Phosphates (µg L <sup>-1</sup> )	Nitrates (mg L <sup>-1</sup> )	Apparent colour (Pt Co l <sup>-1</sup> )	Chlorides (mg L <sup>-1</sup> )	Sulphates (mg L <sup>-1</sup> )
Exotic	E1	10.35 ± 3.1	57.80 ± 1	49.38 ± 17.6	6.79 ± 0.3	7.50 ± 3.9	10.86 ± 1.1	1.80 ± 0.8	43.10 ± 22.3	1.73 ± 0.3	27.75 ± 15	16.55 ± 2.9	2.36 ± 2.7
	E2	10.80 ± 3.2	40.20 ± 2.6	35.00 ± 12.9	6.9125 ± 0.3	5.80 ± 1.2	10.98 ± 1.2	2.25 ± 0.6	47.60 ± 52.1	1.25 ± 0.6	32.00 ± 8.9	13.53 ± 1.6	0.83 ± 0.2
	E3	10.98 ± 3.4	44.23 ± 9.3	27.13 ± 13.5	6.82 ± 0.3	6.99 ± 8.6	11.10 ± 1.2	2.38 ± 0.4	41.98 ± 31.9	1.45 ± 0.3	36.97 ± 11.7	11.98 ± 2.1	1.13 ± 0.2
	E4	10.30 ± 3.0	40.88 ± 0.3	27.13 ± 13.5	6.65 ± 0.3	8.57 ± 5.4	10.65 ± 1.1	2.10 ± 0.7	58.03 ± 30.9	1.38 ± 0.4	42.43 ± 7.8	14.30 ± 1.6	1.42 ± 0.7
Native	N1	10.40 ± 4.0	24.70 ± 0.8	29.38 ± 14.9	6.95 ± 0.3	1.83 ± 0.8	11.15 ± 1.1	2.55 ± 1	59.88 ± 26.7	0.75 ± 0.2	39.10 ± 14.3	9.65 ± 0.6	1.87 ± 1.4
	N2	10.48 ± 3.6	22.78 ± 1.3	29.38 ± 14.9	6.74 ± 0.3	3.02 ± 1.3	10.93 ± 1.3	2.25 ± 0.7	76.48 ± 15.1	1.10 ± 0.2	37.95 ± 13.2	11.28 ± 1.7	1.27 ± 0.3
	N3	12.25 ± 3.2	34.08 ± 2.1	37.88 ± 19.5	6.70 ± 0.1	5.55 ± 4.9	8.20 ± 5	1.78 ± 0.5	72.93 ± 27.8	0.90 ± 0.4	44.35 ± 19.4	12.58 ± 0.9	1.77 ± 0.9
	N4	12.35 ± 3.8	33.35 ± 1.3	18.00 ± 9.3	6.80 ± 0.2	4.25 ± 1.6	8.33 ± 5.1	1.78 ± 0.1	71.70 ± 31.6	0.90 ± 0.2	40.58 ± 16.9	11.18 ± 2.3	1.02 ± 0.3
	N5	11.28 ± 2.7	33.85 ± 0.9	16.13 ± 8.3	6.76 ± 0.1	2.45 ± 1.9	10.88 ± 1	2.35 ± 0.6	61.93 ± 44.4	0.70 ± 0.1	39.28 ± 15.7	10.83 ± 1.1	1.06 ± 0.3
	N6	10.95 ± 2.7	37.28 ± 1.9	24.13 ± 12.4	6.83 ± 0.1	6.29 ± 3.3	10.60 ± 0.8	2.05 ± 0.4	54.88 ± 22.5	1.08 ± 0.2	45.38 ± 19.2	10.95 ± 1.6	1.37 ± 0.3
	N7	10.38 ± 2.6	40.50 ± 1.6	28.88 ± 14.9	6.71 ± 0.2	6.82 ± 1.6	10.85 ± 1	2.60 ± 0.8	71.30 ± 53.6	0.98 ± 0.2	35.60 ± 11.7	12.53 ± 2.4	1.06 ± 0.1
	N8	10.38 ± 2.2	36.43 ± 0.7	25.63 ± 12.3	6.65 ± 0.2	1.44 ± 1.2	11.25 ± 1.2	2.93 ± 1.1	62.10 ± 35.8	1.05 ± 0.3	39.80 ± 13.9	9.98 ± 1.4	1.12 ± 0.3



**Fig. 3.** Average ± SD of macroinvertebrate abundances (solid lines – in d m<sup>-2</sup>) and number of taxa (dotted lines) across seasons at the watershed dominated by native vegetation (black circles) and the watershed dominated by exotic vegetation (grey circles).

vegetation watershed but not on exotic vegetation watershed. Moreover, 91 taxa were recorded in native vegetation watershed, while only 81 taxa were recorded in exotic vegetation watershed. The macroinvertebrates only present in sites in the native vegetation watershed were principally immature stages of Trichoptera and Diptera.

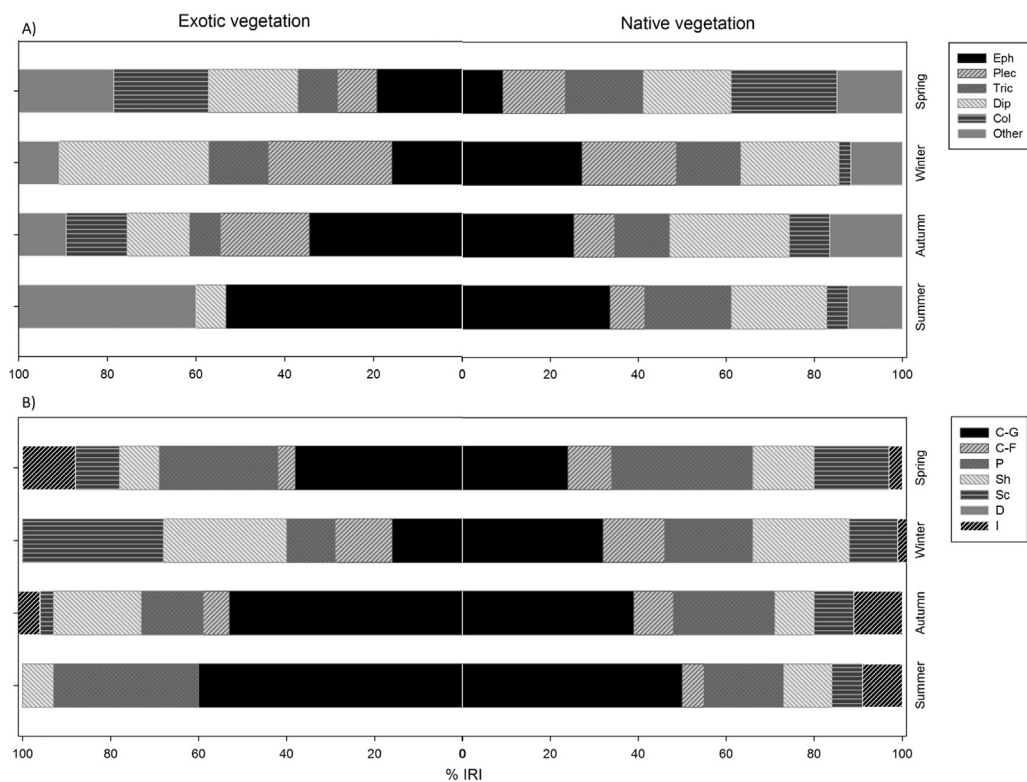
From total of 103 taxa, being 38 collector-gatherers, 27 predators, 15 shredders, 10 scrapers, 6 collector-filterers, 5 detritivores and 2 parasites (Appendix I). Non-significant statistical differences between the two watersheds were observed (PERMANOVA:  $F = 2.028, P = 0.258$ ). Collector-gatherers were the most abundant group in both cases (44–77% relative abundance respectively), followed by shredders (13–35%) and predators (2–20%). Other functional feeding groups were poorly represented in both watersheds. However, the relative abundance of each functional feeding group showed seasonal changes (PERMANOVA:  $F = 4.647, P = 0.037$ ). Although the collector-gatherers had the highest proportion at both watersheds year round, the shredders increased in winter while the predators increased in spring.

### 3.3. Diet of Rainbow Trout

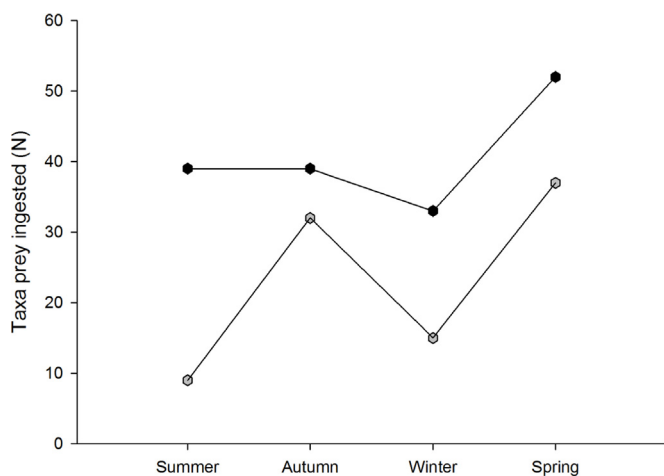
We analysed a total of 244 stomachs from Rainbow Trout that ranged between 3.3 and 19.8 cm SL, and between 0.19 and 252.87 g in mass (Table 3). The diet consisted of 79 taxa of animal origin, from 12 orders. Benthic macroinvertebrates, especially immature insects (Ephemeroptera, Plecoptera, Trichoptera, Diptera and Coleoptera) were the most common diet items (Appendix II, Fig. 4a). However, the diet consumed by Rainbow Trout differed between the two watersheds (PERMANOVA:  $F = 1.870, P = 0.013$ ). The number of taxa consumed in the watershed dominated by native vegetation was higher (76 taxa) than the exotic plantation watershed (56 taxa) (Appendices II and III). Likewise, diets changed across seasons (PERMANOVA:  $F = 2.327, P = 0.001$ ) (Fig. 5, Appendix II).

**Table 3**  
Frequency, standard length and weight of *Oncorhynchus mykiss* in the Araucanía Region (Chile) during the study period.

	N		Length (mm)			Weight (g)		
	Native	Exotic	Min	Max	Average	Min	Max	Average
Summer	29	23	3.3	18.7	7.83	0.19	55.5	7.32
Autumn	69	35	5.4	19.2	9.12	1.82	70.07	10.99
Winter	38	7	5.6	16.5	9.95	1.51	30.12	9.71
Spring	35	8	4.6	19.8	12.57	1.68	252.87	29.76



**Fig. 4.** Index of Relative Importance (IRI) of prey items by taxa (A) (Eph: Ephemeroptera; Plec: Plecoptera; Tri: Trichoptera; Dip: Diptera; Col: Coleoptera; Other: Other taxa) and functional feeding groups (B) (C-G: Collector-gatherer; C-F: Collector-filterer; P: Predator; Sh: Shredder; Sc: Scraper; D: Detritivore; I: Indeterminate) across seasons from stomach contents of trout (*O. mykiss*) at the watershed dominated by native vegetation (right panel) and the watershed dominated by exotic vegetation (left panel).



**Fig. 5.** Seasonal changes in the total number of taxa ingested in trout (*O. mykiss*) at the watershed dominated by native vegetation (black circles) and the watershed dominated by exotic vegetation (grey circles).

The taxa registered in the stomach contents of Rainbow Trout were often found in great abundance in the benthos in both watersheds during the year (Appendices I and II). Nevertheless some taxa present in the benthos were absent in the stomach contents (Appendix III). Using the RELATE analysis showed strong correlation between the matrices of Bray–Curtis similarities of the benthic data and the index of relative importance (RELATE  $R=0.577$ ,  $P=0.001$ ). Similar to the taxonomic analysis, there was a strong association between the benthos and diets at the functional feeding groups level (RELATE  $R=0.486$ ,  $P=0.007$ ). In general, the collector-gatherers were the best represented functional group in the diets at both watersheds across seasons. In winter and spring, shredders,

predators and grazers were consumed in a greater proportion. The other functional groups (collectors-filterers and detritivores) were poorly represented (Fig. 4b).

#### 4. Discussion

This study examined the influence of changes in land use on aquatic food webs in native forest watershed and exotic plantation watershed streams, and explores how these relationships change with seasons. Benthic invertebrate abundance, richness and aquatic prey ingested by Rainbow Trout were higher on native forest sites. This suggests that trophic structure is different between land uses, and provide evidence that diets are representative of taxa from the benthos.

The influence of riparian vegetation on freshwater ecosystems has been widely discussed elsewhere (e.g., Miserendino et al., 2011; Da Silva et al., 2012; Fierro et al., 2015). Most of the studies have concluded that allochthonous organic matter is a key component that sustains food webs in the aquatic systems of mountain streams. Therefore, any alteration in its quality or quantity can affect the aquatic biota (Abelho and Graça, 1996).

We show that exotic vegetation influenced chemical and physical variables (see also Harding and Winterbourn, 1995; Miserendino and Masi, 2010). Streams dominated by exotic vegetation led to higher concentrations of nutrients, minerals and solids. This finding is in agreement with other studies in streams from coastal watershed in southern Chile (Fierro et al., 2012, 2015). Direct effects from forest practices including road-building, fertiliser application and erosion due to forest management increasing find sediment entering streams (Kansagaki et al., 2008; Peña-Cortés et al., 2011a).

As in other studies in southern South America, the diet of Rainbow Trout consists mainly of benthic aquatic macroinvertebrates

and a few terrestrial taxa, confirming the generalist-opportunist diet of this species in southern Chile (Arismendi et al., 2012; Di Prinzi et al., 2013; Vargas-Chacoff et al., 2013). The response of Rainbow Trout to food availability in this study, suggest that the diet is influenced by the type of riparian vegetation. Aquatic invertebrates are less available in the watershed with eucalyptus plantations. Indeed, some taxa that are not found in this watershed may be excluded from these streams, and thus a lower availability of prey to trout (Duffy et al., 2010).

Vegetation effects on prey availability in this study are concordant with previous studies (Romero et al., 2005; Mancilla et al., 2009) that suggests conversion of native forest to monoculture plantations may influence aquatic macroinvertebrate composition. The higher richness and abundance of macroinvertebrates in the native vegetation watershed may be explained by the preference of certain taxa for the organic matter derived from native forest. Abelho and Graça (1996) reports that aquatic macroinvertebrates use fewer leaves in streams dominated by eucalyptus than in streams dominated by native forest. A consequence of exotic plantations is the contribution of particulate organic matter and/or the food quality of the detritus to freshwaters ecosystems (Larrañaga et al., 2009). This low preference of macroinvertebrates, may be related to the characteristics of the eucalyptus leaves, for example the quantity of nutrients, and presence of secondary compounds (e.g. tannins) (Peralta-Maraver et al., 2011). It is difficult to conclude which of these factors accounts for the changes observed in our study, so future work should focus on resolving which are the principal consequences produced by eucalyptus species affecting benthic aquatic macroinvertebrates.

Although prey availability in this study was different between watersheds, we found no evidence for differences in FFG. This supports the hypothesis of Pozo et al. (1998) and Peralta-Maraver et al. (2011), who indicate that the organic matter contributed by eucalyptus (e.g., leaves and branches) is also colonised by collector-gatherers, shredders and predators. These groups wait for a time during which the leaves would be pre-conditioned by fungal and bacterial activity to then can consume. We suggest that the diet of Rainbow Trout based on FFG was not affected by watershed characteristics because the diet was comprise primarily on most abundant functional feeding groups recorded in the study area year round. Other FFG, like grazers or detritivores, had low presence at the sampling sites (prey availability and stomach content), would be limited by the scarce presence of periphyton or macrophytes. Both FFGs feeding on these elements, which have lower abundance and biomass in mountain headwaters, therefore may restrict the presence of this feedings groups.

Temporal patterns of macroinvertebrate availability in these streams were similar to seasonal tendencies observed in other streams of South America (Hollman and Miserendino, 2008; Epele et al., 2011; Fierro et al., 2015). The dependence of Rainbow Trout on this temporal pattern availability may be caused than their diet varied during the year, presenting a significant relation with the presence of prey items. The more numerous terrestrial species prey in the Rainbow Trout stomachs was in spring, compared to other seasons. This difference between seasons in the abundance of terrestrial prey ingested could be explained by higher reproduction of prey during this time, making them more abundant in riparian

habitats and thus more available for consumption (Romero et al., 2005). This situation, together with the emergence of aquatic insects and their reduced presence in the benthos, would influence the diet of fish at this time of year (Buria et al., 2009; Da Silva et al., 2012).

Nevertheless, it is surprising to find that of the total taxa recorded in the benthos during the year, slightly over half were consumed. In fact, some taxa which were present in very low abundance in the benthos were well represented in the Rainbow Trout stomachs (e.g., some species of crustaceans). This difference could be explained since salmonids are mainly visual predators (Eggers, 1978), and prey which are large in size but few in number in the benthos are more exposed to predation (Buria et al., 2007). In addition other taxa recorded in the benthos were not recorded in the stomachs, may be smaller prey species can seek protection in safe refuges among the rocks, where they are at less risk of predation because they are less visible (McCutchen, 2002). It should be noted however, that the individuals examined in the stomach often are much degraded, therefore many times was not possible identify to level genera o specie, as if was possible in benthos sample. Anyway, these results show that the prey selectiveness of the *O. mykiss* is conditioned by the seasonal availability of the prey, and by their size and accessibility.

## 5. Conclusion

This study shows that exotic vegetation may produce an impact on environmental variables and benthic macroinvertebrates communities, leading to changes in stream food webs. Our findings show that land use changes in southern Chile, mainly due to monoculture plantations of forestry species replacing native forest, seem to affect the dissolved solids, suspended solids, nitrates, chlorides and sulphates on streams. Furthermore, these land use changes appear to affect the composition of aquatic macroinvertebrate assemblages. However, these effects are not seen at the functional feeding group level due to the fact that collectors-gatherers are still the most abundant group under both land use types. Lastly, the diet of Rainbow Trout is based mainly on the most abundant taxa and FFG in the benthos in both watersheds all year round. Therefore, the use of diets of Rainbow Trout may serve as a good tool for stream ecosystem assessment. Lastly, if the deforestation of native forest in the watersheds of southern Chile continues to increase, accompanied by an increase in plantations of exotic species, we may expect the disappearance of certain species of benthic macroinvertebrates, especially in the most vulnerable systems.

## Acknowledgements

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## Appendix.

**Appendix I.** Summary of abundances (in  $\text{dm}^{-2}$ ) of benthic macroinvertebrates by taxa sampled from two watersheds in southern Chile. The Functional Feeding Group (FFG) for each taxon is also indicated.

	FFG	Summer		Autumn		Winter		Spring	
		Native	Exotic	Native	Exotic	Native	Exotic	Native	Exotic
<i>Andesiops torrens</i>	CG	59.43	5.89	36.40	9.39	5.38	10.78	21.29	9.73
<i>Andesiops peruvianus</i>	CG	17.76	29.78	15.70	15.78	4.04	5.22	12.13	7.13
<i>Chilopteryx eatoni</i>	P	0.83	0.00	0.27	0.00	0.33	0.00	1.04	0.13
<i>Chaquihua bullocki</i>	P	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00

	FFG	Summer		Autumn		Winter		Spring	
		Native	Exotic	Native	Exotic	Native	Exotic	Native	Exotic
<i>Caenis chilensis</i>	CG	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
<i>Siphonella guttata</i>	CG	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00
<i>Murphyella needhami</i>	CF	0.24	0.11	0.23	0.17	0.33	0.11	1.38	0.00
<i>Nousia maculata</i>	CG	0.00	7.56	0.67	7.83	0.29	0.11	0.50	0.53
<i>Nousia delicata</i>	CG	0.67	0.00	2.77	0.22	0.17	0.00	0.08	0.00
<i>Nousia</i> sp.	CG	0.00	0.11	0.00	0.11	0.00	0.00	0.00	0.00
<i>Meridialaris diguillina</i>	CG	53.24	12.33	46.07	20.17	13.58	17.22	18.96	14.07
<i>Meridialaris chiloense</i>	CG	6.14	0.00	0.20	0.00	0.00	0.00	0.04	0.00
<i>Hapsiphlebia anastomosis</i>	CG	5.74	0.11	1.03	0.44	0.13	0.11	0.29	6.07
<i>Massarttelopsis irrazavali</i>	CG	0.05	0.00	5.47	0.00	0.33	3.56	0.00	0.00
<i>Penaphlebia chilensis</i>	CG	1.40	1.67	9.67	8.39	1.13	2.11	0.63	1.87
<i>Penaphlebia vinosa</i>	CG	0.00	0.00	0.00	0.00	0.00	1.11	0.00	0.13
<i>Penaphlebia</i> sp.	CG	0.00	0.33	0.00	0.00	0.13	0.44	0.00	0.00
<i>Diamphipnopsis samali</i>	S	6.67	2.56	0.80	0.56	0.63	0.00	9.21	0.13
<i>Diamphipnoa helgae</i>	S	1.14	0.00	0.17	0.17	0.83	0.33	0.21	0.20
<i>Diamphipnoidae</i>	S	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00
<i>Kempnyella genualis</i>	P	0.05	0.33	0.33	0.11	0.08	0.00	0.29	0.00
<i>Inconeuria porteri</i>	P	0.36	0.22	0.23	0.56	0.00	0.00	0.13	0.20
<i>Pictoperla gayi</i>	P	0.00	0.00	0.00	0.06	0.04	0.00	0.00	0.00
<i>Perlidae</i>	P	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00
<i>Neuroperlopsis patris</i>	S	0.19	0.44	0.27	0.83	0.00	0.56	0.25	0.33
<i>Penturoperla barbata</i>	S	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00
<i>Klapopteryx armillata</i>	S	10.24	1.22	3.80	0.83	0.58	0.00	11.54	0.73
<i>Udamocercia</i> sp.	SC	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.27
<i>Astronemoura chilena</i>	S	0.69	0.78	0.13	1.61	0.00	0.00	0.04	0.27
<i>Pelurgoperla personata</i>	S	0.43	6.11	0.00	0.94	0.08	0.00	0.17	1.53
<i>Limnoperla jaffueli</i>	S	15.45	8.78	41.43	33.00	22.54	19.67	17.54	9.67
<i>Notoperlopsis femina</i>	S	8.57	1.67	0.53	0.50	4.21	13.89	0.00	0.00
<i>Antarctoperla michaelsoni</i>	S	0.05	0.00	0.37	12.11	0.04	0.00	0.00	0.00
<i>Ceratoperla schwabei</i>	S	0.00	0.44	0.00	0.11	0.00	0.00	0.04	0.00
<i>Ecnomidae</i>	CG	0.79	0.33	1.13	1.56	0.04	0.00	0.08	0.20
<i>Hydrobiosidae</i>	P	0.76	1.44	0.00	5.89	0.58	1.00	0.58	0.60
<i>Leptoceridae</i>	SC	0.05	0.11	0.00	0.06	0.08	0.11	0.04	0.00
<i>Hydroptilidae</i>	SC	0.19	0.00	0.00	0.06	0.00	0.00	0.04	0.00
<i>Hydroptila</i> sp.	CG	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00
<i>Smicridea annulicornis</i>	CF	0.57	2.56	6.47	17.61	0.00	0.00	0.00	0.00
<i>Smicridea</i> sp.	CF	1.21	21.11	0.70	2.56	1.38	0.67	2.46	1.20
<i>Triplectides</i> sp.	S	0.00	0.67	0.00	0.00	0.00	0.00	0.00	0.00
<i>Metrichia</i> sp.	CG	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.20
<i>Neotrichia</i> sp.	CG	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00
<i>Neotrichia chilensis</i>	CG	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00
<i>Austrotinodes</i> sp.	CG	0.00	0.11	0.00	0.00	0.00	0.00	0.00	0.00
<i>Dolophilodes</i> sp.	S	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00
<i>Parasericostoma</i> sp.	S	0.10	0.00	0.20	0.11	0.04	0.00	0.00	0.00
<i>Polycentropus</i> sp.	P	0.00	0.00	0.07	0.06	0.00	0.11	0.08	0.20
<i>Brachysetodes</i> sp.	SC	1.81	0.00	0.00	0.00	0.08	0.00	0.13	0.07
<i>Rheocorema</i> sp.	P	0.00	0.00	2.00	0.00	0.00	0.00	0.00	0.00
<i>Psychodidae</i>	CG	0.00	0.00	0.13	0.00	0.04	0.00	0.04	0.00
<i>Ephydriidae</i>	P	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
<i>Empididae</i>	P	0.14	0.00	0.00	0.22	0.00	0.00	0.00	0.00
<i>Hemerodromia</i> sp.	P	0.36	0.11	0.00	0.06	0.04	0.11	1.13	0.20
<i>Simulium</i> sp.	CF	1.52	5.78	5.63	20.33	2.25	1.22	0.67	0.47
<i>Arachnephioidea</i> sp.	CF	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07
<i>Gigantodax</i> sp.	CF	0.52	0.56	0.13	0.06	0.21	0.11	0.04	0.00
<i>Blephariceridae</i>	SC	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00
<i>Tipula</i> sp.	P	1.29	0.00	0.00	0.00	0.00	0.11	1.42	0.13
<i>Atherix</i> sp.	P	1.48	0.11	3.37	2.83	0.50	0.11	0.83	0.40
<i>Hexatoma</i> sp.	P	2.12	0.11	9.37	7.61	8.88	12.89	18.83	9.27
<i>Limonia</i> sp.	P	2.24	1.22	0.33	1.78	0.25	0.56	1.42	6.20
<i>Tipulidae</i>	P	0.57	0.00	0.00	0.17	0.00	0.00	0.00	0.00
<i>Stilobezzia</i> sp.	P	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00
<i>Alluaudomyia</i> sp.	P	1.14	0.22	0.40	0.17	0.33	0.11	0.46	0.27
<i>Corynoneura</i> sp.	CG	5.52	1.89	7.30	0.56	0.92	0.89	1.25	0.40
<i>Eukiefferella</i> sp.	CG	8.55	24.44	1.90	24.44	0.17	0.11	10.50	3.40
<i>Dicretendipes</i> sp.	CG	0.24	0.00	0.07	0.44	0.08	0.00	0.00	0.00
<i>Coelotanypus mendax</i>	CG	0.10	0.11	0.07	0.00	0.00	0.00	0.08	0.13
<i>Lopescladius</i> sp.	CG	1.38	1.89	1.80	0.56	0.42	1.00	3.00	5.07
<i>Orthocladius</i> sp.	CG	28.55	12.11	3.33	11.56	4.96	1.11	6.54	6.00
<i>Paratrichocladius</i> sp.	CG	0.00	0.11	0.00	0.11	0.04	0.00	0.00	0.00
<i>Pentaneura</i> sp.	CG	9.02	1.78	0.20	0.17	0.04	0.11	1.00	0.40
<i>Rheotanytarsus</i> sp.	CG	0.00	0.00	0.00	0.00	0.08	0.00	2.25	0.00
<i>Tanytarsus</i> sp.	CG	0.14	0.33	0.00	1.28	0.00	0.00	0.88	0.00
<i>Thienemaniella</i> sp.	CG	5.05	0.11	12.70	2.06	3.08	2.67	1.67	0.33
<i>Symbiocladius wygodzinskyi</i>	PA	0.43	0.00	0.13	0.06	0.00	0.00	0.25	0.00
<i>Austrolimnius</i> sp.	CG	22.55	9.00	8.37	7.89	1.58	0.67	3.75	5.93
<i>Austrelmis</i> sp.	CG	0.05	0.00	0.00	0.00	0.00	0.00	0.04	0.40
<i>Luchoelmis</i> sp.	CG	0.05	0.00	0.20	0.28	0.00	0.00	0.13	0.00



	FFG	Summer		Autumn		Winter		Spring	
		Native	Exotic	Native	Exotic	Native	Exotic	Native	Exotic
<i>Tychepephenus felix</i>	SC	0.69	0.22	1.17	0.67	0.58	0.00	1.13	0.27
Halipilidae	SC	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
<i>Halipilus</i> sp.	SC	0.00	0.00	0.20	0.00	0.00	0.00	0.00	0.00
Hydrophilidae	P	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coleoptera	P	0.00	0.00	0.20	0.11	0.00	0.11	0.04	0.00
<i>Protochauliodes</i> sp.	P	0.10	0.00	0.17	0.00	0.00	0.00	0.00	0.13
<i>Neogomphus</i> sp.	P	0.05	0.00	0.43	0.00	0.00	0.00	0.13	0.00
Hydracarina	P	0.43	0.11	0.07	0.11	0.08	0.00	0.00	0.07
<i>Littoridina cumingi</i>	SC	0.19	0.11	0.00	0.00	0.00	0.00	0.08	0.00
<i>Aegla araucaniensis</i>	P	1.40	0.11	0.93	0.50	0.46	0.11	1.46	1.00
<i>Aegla abtao</i>	P	0.14	0.00	0.00	0.00	0.04	0.00	0.04	0.00
<i>Aegla</i> sp.	P	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.07
<i>Chilina dombeyana</i>	SC	0.14	0.00	0.20	0.00	0.04	0.00	0.00	0.07
<i>Dugesia</i> sp.	D	0.00	0.11	0.00	0.00	0.04	0.00	0.00	0.00
<i>Hyalella costera</i>	CG	0.05	0.00	0.00	0.00	0.04	0.00	0.04	0.00
<i>Hyalella</i> sp.	CG	0.29	0.00	0.00	0.17	0.00	0.00	0.00	0.00
<i>Tubifex</i> sp.	D	3.45	1.22	3.83	4.50	1.38	0.33	0.04	0.13
<i>Chaetogaster</i> sp.	D	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00
Lumbriculidae	D	0.00	0.00	0.07	0.17	0.13	0.00	0.00	0.00
Naididae	D	0.10	0.22	0.00	0.00	0.04	0.00	0.00	0.00
<i>Heterias exul</i>	CG	0.10	0.00	0.00	0.00	0.00	0.00	0.08	0.40
<i>Temnocephala chilensis</i>	PA	0.00	0.00	1.87	0.33	0.00	0.00	0.00	0.00

**Appendix II.** Index of Relative Importance (IRI) of the prey items in the stomach of *O. mykiss* in the two watersheds in southern Chile during the study period.

	Summer		Autumn		Winter		Spring	
	Native	Exotic	Native	Exotic	Native	Exotic	Native	Exotic
<i>Andesiops peruvianus</i>	1.45	2.57	1.61	1.53	0.89	1.00	0.46	0.77
<i>Andesiops torrens</i>	0.64	1.14	0.86	0.94	0.59	0.27	0.15	0.32
Baetidae	0.00	0.00	0.22	0.13	0.00	0.00	0.23	0.89
<i>Meridialaris</i> sp.	0.19	0.29	0.11	0.28	0.74	0.45	0.13	0.00
<i>Meridialaris diguillina</i>	0.60	0.29	0.00	0.00	0.00	0.00	0.00	0.00
<i>Nousia</i> sp.	0.40	0.00	0.00	0.16	0.55	0.00	0.12	0.00
Leptophlebiidae	0.24	0.00	0.27	0.31	0.18	0.00	0.18	0.44
<i>Hapsiphlebia anastomosis</i>	0.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00
<i>Penaphlebia</i> sp.	0.32	0.00	0.00	0.00	0.37	0.00	0.00	0.00
Ephemeroptera	1.72	0.29	0.57	2.00	0.40	0.00	0.24	0.68
<i>Notoperlopsis femina</i>	0.00	0.00	0.13	0.63	0.27	0.45	0.58	0.55
<i>Limnoperla jaffuelli</i>	0.40	0.00	0.11	0.31	0.33	1.64	0.63	0.66
<i>Antactoperla michaelsoni</i>	0.00	0.00	0.11	0.31	1.64	0.55	0.00	0.00
Gripopterygidae sp1.	0.40	0.00	0.45	0.19	0.00	0.00	0.00	0.00
Gripopterygidae sp2.	0.40	0.00	0.00	0.00	0.00	0.00	0.00	0.00
<i>Pelurgoperla personata</i>	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.00
<i>Diamphipnopsis samali</i>	0.00	0.00	0.00	0.00	0.15	0.00	0.12	0.00
<i>Diamphipnoa helgae</i>	0.00	0.00	0.00	0.00	0.15	0.00	0.76	0.00
<i>Neuroperlopsis patris</i>	0.00	0.00	0.00	0.00	0.00	0.27	0.13	0.00
Plecoptera	0.16	0.00	0.53	1.67	0.27	0.14	0.13	0.22
<i>Smicridea</i> sp.	0.56	0.00	0.92	0.51	0.59	1.00	0.23	0.33
Anomalopsychidae	0.00	0.00	0.00	0.00	0.00	0.00	0.58	0.00
Leptoceridae	0.40	0.00	0.00	0.31	0.00	0.00	0.76	0.00
Limnephilidae	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00
Helicopsychidae	0.40	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Helicophidae	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
Hydrobiosidae	0.40	0.00	0.00	0.00	0.37	0.45	0.25	0.00
<i>Polycentropus</i> sp.	0.00	0.00	0.00	0.13	0.00	0.00	0.58	0.00
Hydroptilidae	0.00	0.00	0.11	0.00	0.74	0.00	0.13	0.00
Glossosomatidae	0.40	0.00	0.00	0.00	0.00	0.00	0.13	0.55
Ecnomidae	0.40	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Trichoptera	0.79	0.00	0.75	0.13	0.30	0.00	0.11	0.22
Psychodidae	0.28	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Ptychopteridae	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.38
<i>Tipula</i> sp.	0.00	0.00	0.00	0.00	0.37	0.00	0.00	0.55
<i>Limonia</i> sp.	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.38
Tipulidae	0.36	0.00	0.18	0.00	0.00	0.32	0.76	0.33
<i>Simulium</i> sp.	0.40	0.00	0.45	0.45	0.57	0.00	0.13	0.00
Simuliidae sp1.	0.00	0.00	0.00	0.00	0.44	0.00	1.30	0.39
Simuliidae sp2.	0.00	0.00	0.00	0.00	0.26	0.45	0.00	0.00
<i>Alluaudomya</i> sp.	0.36	0.00	0.11	0.00	0.00	0.00	0.13	0.00
Ceratopogonidae	0.00	0.00	0.11	0.00	0.00	0.00	0.00	0.00
<i>Rheotanytarsus</i> sp.	0.40	0.00	0.00	0.00	0.00	0.00	0.13	0.00
<i>Tanytarsus</i> sp.	0.00	0.29	0.00	0.31	0.00	0.00	0.13	0.44
<i>Paratrichocladius</i> sp.	0.40	0.00	0.00	0.31	0.00	0.00	0.00	0.00

	Summer		Autumn		Winter		Spring	
	Native	Exotic	Native	Exotic	Native	Exotic	Native	Exotic
<i>Orthocladius</i> sp.	0.67	0.00	0.95	0.00	0.22	0.00	0.00	0.00
<i>Coryneura</i> sp.	0.00	0.00	0.15	0.00	0.00	0.00	0.00	0.00
<i>Pentaneura</i> sp.	0.00	0.00	0.45	0.00	0.37	0.00	0.00	0.00
Chironomidae	0.60	0.29	0.19	0.85	0.00	0.00	0.14	0.22
Blephariceridae	0.00	0.00	0.11	0.00	0.37	2.45	0.00	0.00
Diptera	0.24	0.00	1.22	0.26	0.46	0.45	0.46	0.55
<i>Tychepephenus felix</i>	0.00	0.00	0.18	0.00	0.00	0.00	0.13	0.19
<i>Luchoelmis</i> sp.	0.16	0.00	0.00	0.31	0.00	0.00	0.76	0.22
<i>Austrolimnius</i> sp.	0.16	0.00	0.35	0.88	0.00	0.00	0.13	0.00
<i>Austrelmis</i> sp.	0.00	0.00	0.00	0.00	0.00	0.00	0.46	0.00
<i>Phanocerus</i> sp.	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.55
<i>Macrelmis</i> sp.	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.16
Elmidae	0.00	0.00	0.11	0.31	0.00	0.00	0.11	0.22
Dryopidae	0.00	0.00	0.00	0.00	0.00	0.00	0.35	0.55
Dystiscidae	0.40	0.00	0.45	0.31	0.00	0.00	0.36	0.55
Haliplidae	0.00	0.00	0.00	0.00	0.00	0.00	0.58	0.00
Hydrophilidae	0.00	0.00	0.11	0.31	0.00	0.00	0.13	0.00
Staphylinidae	0.00	0.00	0.11	0.00	0.00	0.00	0.13	0.22
Coleoptera	0.12	0.00	0.00	0.00	0.37	0.00	0.46	0.77
Belostomatidae	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.55
Corixidae	0.48	0.00	0.63	0.26	0.00	0.00	0.69	0.00
Mesoveliidae	0.00	0.00	0.11	0.00	0.00	0.00	0.00	0.00
Hemiptera	0.00	0.00	0.18	0.31	0.37	0.00	0.76	0.33
Lepidoptera	0.14	0.57	0.00	0.00	0.00	0.00	0.00	0.00
Formicidae	0.79	0.00	0.00	0.00	0.00	0.00	0.26	0.70
<i>Littoridina cumingi</i>	0.00	0.00	0.45	0.00	0.37	0.00	0.58	0.00
<i>Chilina</i> sp.	0.40	0.00	0.45	0.13	0.00	0.00	0.00	0.00
<i>Chilina dombeyana</i>	0.00	0.00	0.00	0.00	0.00	1.00	0.00	0.55
<i>Aegla</i> sp.	0.36	0.00	0.18	0.28	0.37	0.00	0.36	0.00
Crustacea	0.40	2.86	0.13	0.31	0.22	0.00	0.13	0.33
Araneae	0.00	0.00	0.00	0.00	0.15	0.00	0.35	0.22
<i>Heterias exul</i>	0.00	0.00	0.00	0.00	0.12	0.00	0.00	0.77
Oligochaeta	0.00	0.00	0.11	0.00	0.00	0.00	0.00	0.00
Unknown	0.00	0.00	0.87	0.63	0.00	0.00	0.00	0.55

**Appendix III.** Taxa macroinvertebrates found in the analysed samples from benthos and stomach *Oncorhynchus mykiss* from the watershed dominated by native vegetation (black circles) and the watershed dominated by exotic vegetation (grey circles).

Land use/Taxa	Macroinvertebrates		Diet <i>O. mykiss</i>	
	Native	Exotic	Native	Exotic
Ephemeroptera				
<i>Andesiops torrens</i>	●	●	●	●
<i>Andesiops peruvianus</i>	●	●	●	●
Baetidae			●	●
<i>Chiloporter eatoni</i>	●	●		
<i>Chaquihua bullocki</i>	●			
<i>Caenis chilensis</i>				
<i>Siphonella guttata</i>	●			
<i>Murphyella needhami</i>	●	●		
<i>Nousia maculata</i>	●	●		
<i>Nousia delicata</i>	●	●		
<i>Nousia</i> sp.		●	●	●
<i>Meridialaris diguillina</i>	●	●	●	●
<i>Meridialaris chiloense</i>	●			
<i>Meridialaris</i> sp.			●	●
<i>Hapsiphlebia anastomosis</i>	●	●	●	
<i>Massarttelopsis irrazavali</i>	●	●		
<i>Penaphlebia chilensis</i>	●	●		
<i>Penaphlebia vinosa</i>		●		
<i>Penaphlebia</i> sp.	●	●	●	
Leptophlebiidae			●	●
Ephemeroptera n/i			●	●
Plecoptera				
<i>Diamphipnopsis samali</i>	●	●	●	
<i>Diamphipnoa helgae</i>	●	●	●	
Diamphipnoidae		●		
<i>Kempnyella genualis</i>	●	●		
<i>Inconeuria porteri</i>	●	●		
<i>Pictoperla gayi</i>	●	●		
Perlidae	●			
<i>Neuroperlopsis patris</i>	●	●	●	●

Land use/Taxa	Macroinvertebrates		Diet <i>O. mykiss</i>	
	Native	Exotic	Native	Exotic
<i>Penturoperla barbata</i>		●		
<i>Klapopteryx armillata</i>	●	●		
<i>Udamocercia</i> sp.		●		
<i>Astronemoura chilena</i>	●	●		
<i>Pelurgoperla personata</i>	●	●	●	
<i>Limnoperla jaffueli</i>	●	●	●	●
<i>Notoperlopsis femina</i>	●	●	●	●
<i>Antarctoperla michaelsoni</i>	●	●	●	●
<i>Ceratoperla schwabei</i>	●	●		
Gripopterygidae sp1			●	●
Gripopterygidae sp2			●	
Plecoptera n/i			●	●
Trichoptera				
Anomalopsychidae			●	
Ecnomidae	●	●	●	
Glossosomatidae			●	●
Helicopsychidae			●	
Helicophidae				●
Hydrobiosidae	●	●	●	●
Leptoceridae	●	●	●	●
Limnephilidae			●	
Hydroptilidae	●	●	●	
<i>Hydroptila</i> sp.	●			
<i>Smicridea annulicornis</i>	●	●		
<i>Smicridea</i> sp.	●	●	●	●
<i>Triplectides</i> sp.		●		
<i>Metrichia</i> sp.	●	●		
<i>Neotrichia</i> sp.	●			
<i>Neotrichia chilensis</i>	●			
<i>Austrotinodes</i> sp.		●		
<i>Dolophilodes</i> sp.	●			
<i>Parasericostoma</i> sp.	●	●		
<i>Polycentropus</i> sp.	●	●	●	●
<i>Brachysetodes</i> sp.	●	●		
<i>Rheocorema</i> sp.	●			
Trichoptera n/i			●	●
Diptera				
Psychodidae	●		●	
Ptychopteridae			●	●
Ephydriidae				
Empididae	●	●		
<i>Hemerodromia</i> sp.	●	●		
<i>Simulium</i> sp.	●	●	●	●
<i>Arachnephioides</i> sp.		●		
<i>Gigantodax</i> sp.	●	●		
Simuliidae sp1			●	●
Simuliidae sp2			●	●
Blephariceridae	●		●	●
<i>Tipula</i> sp.	●	●	●	●
<i>Atherix</i> sp.	●	●		
<i>Hexatoma</i> sp.	●	●		
<i>Limonia</i> sp.	●	●		●
Tipulidae	●	●		●
<i>Stilobezzia</i> sp.	●		●	
<i>Alluaudomyia</i> sp.	●	●	●	
Ceratopogonidae			●	
<i>Corynoneura</i> sp.	●	●	●	
<i>Eukiefferella</i> sp.	●	●		
<i>Dicrotendipes</i> sp.	●	●		
<i>Coelotanypus mendax</i>	●	●		
<i>Lopescladius</i> sp.	●	●		
<i>Orthocladius</i> sp.	●	●	●	
<i>Paratrichocladius</i> sp.	●	●	●	●
<i>Pentaneura</i> sp.	●	●	●	
<i>Rheotanytarsus</i> sp.	●	●	●	
<i>Tanytarsus</i> sp.	●	●	●	●
<i>Thienemaniella</i> sp.	●	●		
<i>Symbiocladius wygodzinskyi</i>	●	●		
Chironomidae			●	●
Diptera n/i			●	●
Coleoptera				
<i>Austrolimnius</i> sp.	●	●	●	●
<i>Austrelmis</i> sp.	●	●	●	●
<i>Luchoelmis</i> sp.	●	●	●	●
Elmidae			●	●

Land use/Taxa	Macroinvertebrates		Diet <i>O. mykiss</i>	
	Native	Exotic	Native	Exotic
<i>Phanocerus</i> sp.			●	●
<i>Macrelmis</i> sp.			●	●
<i>Tychepephenus felix</i>	●	●	●	●
Dryopidae			●	●
Dystiscidae			●	●
Haliplidae	●		●	
<i>Haliphus</i> sp.	●			
Hydrophilidae	●		●	●
Staphylinidae			●	●
Coleoptera n/i	●	●	●	●
Hemiptera				
Belostomatidae			●	●
Corixidae			●	●
Mesoveliidae			●	
Hemiptera n/i			●	●
Megaloptera				
<i>Protochauiodes</i> sp.	●	●		
Odonata				
<i>Neogomphus</i> sp.	●			
Lepidoptera			●	●
Hymenoptera				
Formicidae			●	●
Non Insecta				
Hydracarina	●	●		
Araneae			●	●
<i>Littoridina cumingi</i>	●	●	●	
<i>Aegla araucaniensis</i>	●	●		
<i>Aegla abtao</i>	●			
<i>Aegla</i> sp.		●	●	●
Crustacea			●	●
<i>Chilina dombeyana</i>	●	●		●
<i>Chilina</i> sp.			●	●
<i>Dugesia</i> sp.	●	●		
<i>Hyaella costera</i>	●			
<i>Hyaella</i> sp.	●	●		
<i>Tubifex</i> sp.	●	●		
<i>Chaetogaster</i> sp.		●		
Lumbriculidae	●	●		
Naididae	●	●		
Oligochaeta			●	
<i>Heterias exul</i>	●	●	●	●
<i>Temnocephala chilensis</i>	●			
Unknown			●	●
N taxa	91	81	76	56

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