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Application of the Lake Biotic Index (LBI) in the ecological characterization of a North Patagonian lake in Chile

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ABSTRACT

Increased pollution and degradation of water resources and their associated ecosystems has stimulated the development of tools and methodologies to characterize, estimate, predict, and reverse the environmental impact of anthropic effects on water bodies. The Secondary Water Quality Standards (NSCA) adopted in Chile have incorporated the use of bioindicators complementary to physicochemical analyses, in order to determine the ecological condition of lotic and lentic environments. Our research used the "Lake Biotic Index" (LBI) to establish the ecological condition of Lake Rupanco using benthic macroinvertebrates. The results indicated an Oligo-Eubiotic condition for this lake given the high concentration of oxygen and low organic matter content in sediments, in addition to low biogenic potential and good taxa preservation in both the autumn and spring surveys. Features of the ecological condition obtained through the application of the LBI (benthic subsystem) conform to the results of physicochemical and microalgae analyses undertaken previously in Lake Rupanco (pelagic subsystem). Based on these results, we support application of the LBI index as a complementary tool for the integrated management of lentic ecosystems.

1. Introduction

The biophysical dimensions of ecosystems, such as nutrient cycling, energy flow and biodiversity are inextricably related (Rapport et al., 2009). As a result, any modification of these dimensions threatens the system's resilience capacity (FAO, 2011). Demographic growth together with a concept of development that does not consider, or underestimates, the associated environmental and social costs of economic development and productivity (Gladwin et al., 1995), have produced drastic change and environmental degradation of continental aquatic ecosystems over the last few decades (Revenga et al., 2005). In response to this situation, different models and methods have been created to implement action plans with a local or regional focus (Directiva Marco de Agua, 2000; Poikane et al., 2016) aimed at reducing or detaining the environmental degradation of natural freshwater resources.

Lakes are ecosystems that support a specialized community of organisms according to their physical and chemical characteristics (Wetzel, 1986). To determine the biological integrity, trophic level and recreational potential of these aquatic ecosystems, it is necessary to determine physicochemical and biological indicators that impact, or indicate changes in biological quality (Zhao et al., 2019). The ecological integrity of a water body has been defined as the capacity to support and maintain a balanced, integrated, adaptive biological system, considering all the elements and processes in the natural habitat (Schallenberg et al., 2011). The concept of ecological integrity requires a methodological framework that can quantify spatial and temporal changes of ecosystems and take effective measures for the conservation and management of these ecosystems (Bridgewater et al., 2015; Wurtzebach and Schultz, 2016).

Numerous methods and indices are used to interpret the degree of anthropogenic disturbance in aquatic systems, based exclusively on analysis of the physicochemical conditions (trophic level). However, these factors only indicate water conditions (pelagic subsystem) at the point in time when measurements are taken and do not reveal the evolution of pollutant load and the resilience and buffering capacity of aquatic ecosystems in general (Vadeboncoeur et al., 2002). In this context, consideration of bioindicators is useful (Revenga et al., 2005) given their capacity to detect impacts and evaluate environmental conditions over a broader time scale (Campbell, 2004). Bioindicators of environmental impact based on benthic macroinvertebrates gained special relevance in Europe (Dermott and Pachkevitch, 2012). However, in

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Chile and South America they are mostly used to assess the ecological condition of rivers (Brand and Miserendino, 2015; Cardoso and Novaes, 2013; Correa-Araneda, 2016; Correa-Araneda et al., 2010; Dedieu et al., 2016; Dos Santos, 2017; Fierro et al., 2016; Figueroa et al., 2003; Kay et al., 2001; Lallement et al., 2014; Luo et al., 2018; Meza-S et al., 2012; Miserendino et al., 2012; Nascimento et al., 2018).

Macroinvertebrate communities are indicators of medium and long-term disturbances because the life cycle of these species ranges from less than one month to over a year. Thus, their indicative value spans an intermediate time interval between that of biological elements with either shorter (phytobenthic, phytoplankton, zooplankton) or longer (fish) response times (Segnini, 2003).

Macroinvertebrates are a dominant group in rivers and have a significant presence in the littoral zone and bottom of lakes and wetlands (Bazzanti et al., 2012). They play an important ecological role in the littoral zone of the lacustrine ecosystem, as a link between primary producers and fish. In deep lakes a strong correlation has been observed between macroinvertebrate assemblages and eutrophication, as they are considered to be indicators of oxygen levels and trophic conditions (Jyväsjärvi et al., 2014; Schallenberg et al., 2011).

Studies on the bathymetric distribution of benthic communities in deep lakes of the northern hemisphere established that the presence and biomass of these communities is mainly influenced by oxygen concentration, quality and quantity of food and water temperature (Hirabayashi et al., 2015). According to the biotic indices of the Biological Monitoring Working Party (BMWP) and Average Score Per Taxon (ASPT), total number of taxa among lakes with different trophic categories only differs in the deep sub-littoral zone (Bazzanti et al., 2012).

Verneaux et al. (2004a) proposed the use of the ecological integrity index "Lake Biotic Index" (LBI), which is based on two combined indices: the first determines the Littoral Biotic Index (Bl) the littoral macroinvertebrate community related to the biogenic potential of the lake (quantitative); the second, Taxonomic deficit index (Df) is related with quality of the water-sediment interface in the deep zone (qualitative data). These two indices are combined in the LBI within a range of 0–20, which provides information on the biogenic capacity of the lake for the development of macroinvertebrate communities.

In Chile, limnological studies have characterized the North Patagonian Lakes (NPL) providing valuable biological information based on determination of primary productivity and the concentration and diversity of phytoplankton and zooplankton (De Los Ríos et al., 2007; DGA, 2014; Woelfl et al., 2003). Specific studies have been conducted on load capacity in some of the lentic bodies exposed to significant anthropic pressure, such as intensive aquaculture activities (León-Múñoz et al., 2013; Soto, 2002). However, the use of bioindicators associated with the benthic subsystem is very limited, both with regard to application of indices already in existence, or their integration in the development of monitoring tools to determine the quality of lacustrine ecosystems (DGA, 2010).

In this research, the deep Lake Rupanco is an interesting case study given that biological communities may be impaired by nutrient loads, mainly nitrogen and phosphorus, from diffuse sources in watersheds, as well as intensive salmon farming in the water column (León-Múñoz et al., 2013). These factors can cause environmental imbalances (Sterner et al., 2007) even though this lake is classified as oligotrophic (DGA, 2014; Campos et al., 1992; Soto, 2002; Woelfl et al., 2003). Studies conducted by Fuentes et al. (2018) recorded blooms of the cyanobacteria *Microcystis aeruginosa* during periods of mixing in the water column.

The aim of this study was to determine the ecological condition of Lake Rupanco, using the Lake Biotic Index (LBI) and physicochemical variables of the benthic subsystem and their response to seasonal change. The abundance and richness of the benthic communities vary seasonally, registering greater abundances in spring-summer (Gabriels et al., 2010). In view of the bathymetric characteristics and macrofaunistic composition of Lake Rupanco, the variables of the deep zone index of the littoral (Z*f*) and the tolerance of the organisms present were adapted in order to apply the LBI.

2. Materials and methods

2.1. Study site and experimental design

Lake Rupanco (40°50′ S, 72°26' W) (Fig. 1 A, B), is located 118 m above sea level, with a basin of 9994 km^2, a surface area of 235 km^2, a maximum depth of 274 m and an average depth of 163 m. It is a warm monomictic lake with a mixing period in winter (Soto, 2002). In spring, the euphotic zone (Z_{EUF}) generally reaches a depth of up to 28.5 m, although it can extend up to 45 m; the ratio between depth of the euphotic zone and maximum depth (Z_{EUF}/Z_{MAX}) is 14% (DGA, 2014). Land use in the Lake basin has increased substantially: 12 % corresponds to shrubland patches; 23% a crops and grasslands in the lowest zones and 65 % to native forest. Salmon culture activities have also increased and the concessioned area corresponds to 32.7% (160.8 ha) of the total area of these water bodies. It has been shown that the combined effect of changes in land use together with salmon culture activities is the principal source of phosphorous and nitrogen in the water column and in the sediment (León-Múñoz et al., 2013).

Two sampling campaigns were conducted during 2013, first in April (autumn) and the second in November (spring). Six stations were sampled during each campaign: LRC1, LRC2, LRC3, LRC4, LRC5 y LRSUBVI located throughout the lake (Fig. 1 B, Table 2). In each season we established one Littoral (L) substation at a depth of 5 m and one sub-littoral substation (Z*f*), located at 25 m depth, taking into consideration the depth of the epilimnion during the sampling periods (18–26 m). In each station, three replicates were obtained in both L and in Z*f*. The LRSUBVI represents a zone that has been exposed to the direct impact of salmon fish farming installations for over 20 years.

2.2. Obtaining macroinvertebrates

A Van-Veen type grab sampler, 0.1m^2 surface size was used to collect macroinvertebrate (Dermott and Pachkevitch, 2012) and sediment samples and a Garmin GPS Model 421S with Echo sounder, to establish the position and depth of the stations. Samples were sifted through a 0.5 mm sieve then fixed in 70% alcohol and taxonomic determination was obtained using a stereoscopic microscope and taxonomic keys (Domínguez and Fernández, 2009; Jara, 1989; Pessacq, 2009; Peters and Campbell, 1991; Rojas, 2006; Thorp and Covich, 1992; Vera and Camousseight, 2006) in the Limnology Laboratory of the Universidad de Los Lagos. The tolerance value of each taxon was determined using the Figueroa et al. (2007) table for freshwater benthic macroinvertebrates in Chilean Mediterranean rivers. According to Mandaville (2002), these indices are fully applicable in the case of lakes, especially in larger and deeper lakes. In addition, Gabriels et al. (2010) developed a biotic index for aquatic ecosystems in Belgium, based on the tolerance values of both river and lake macroinvertebrates, given that these are the most abundant organisms in freshwater ecosystems (rivers, streams, lakes and ponds). They noted that benthic macroinvertebrates maintain their bioindication capacity and only present different adaptive strategies according to habitat (Roldán, 1999; Kim et al., 2013).

2.3. Sediment samples

The pH and Redox Potential of each sample were measured in situ with a Thermo Orion A121 multi-parameter sensor. The samples were transported to the Limnology Laboratory to determine the % of particulate organic matter in the sediment (POM). All measurements were conducted according to the procedures and techniques established by Resolution 3612/2009, and subsequent modifications, issued by the Undersecretariat for Fisheries and Aquaculture.

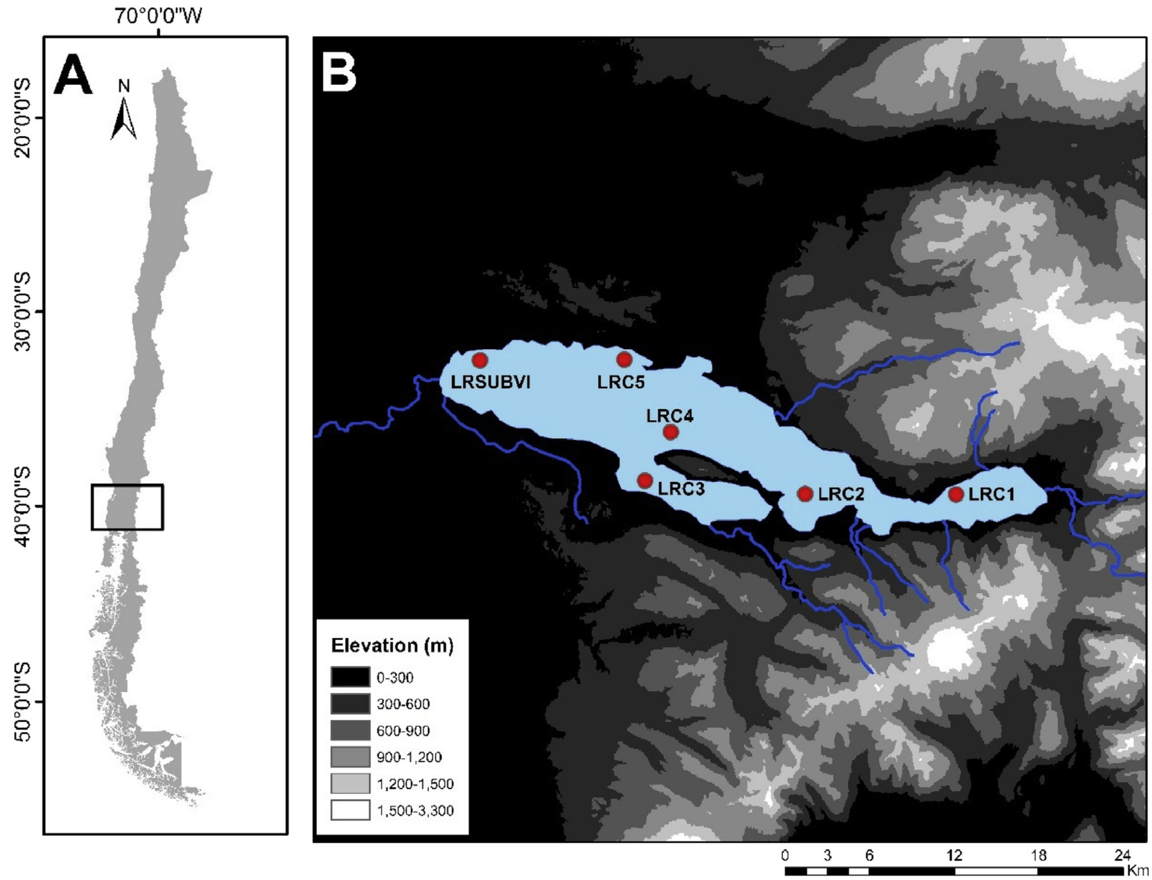

Fig. 1. A) Geographic location of Chile (in black box location Lake Rupanco, region of Los Lagos); B: Location of monitoring stations in the Lake Rupanco (points in red).

2.4. LBI index

After the qualitative and quantitative identification of each sample, the following descriptors were calculated: littoral taxonomic richness at family level *Zf* (v*l*); littoral density (d*l*) expressed in number of individuals/m^2 collected in Z*l;* taxonomic richness in Z*f*; descriptor of littoral fauna quality (q*l*) and correction coefficient k of taxonomic loss from Z*l* to Z*f*. k= (0.033 vl) + 1. The symbols *l* and *f* were used to designate the littoral (Z*l* 5 m) and deep sub-littoral (Z*f* 25 m) zones respectively.

The following were determined:

- Littoral biogenic index $(\mathrm{B}l) = (\sqrt{\mathrm{vl}})(\log_e \mathrm{d}l)$ based on the taxon diversity and abundance.
- Taxonomic deficit index $(\mathrm{D}f) = \sqrt{\mathrm{k}}{*}\mathrm{vf}{*}\mathrm{ql}\ \ \mathrm{vl}$ associated with the degree of conservation of taxa found between the littoral zone (Z*l* = 5m) and the sub-littoral zone (Z*f* = 25 m).
- LBI index: Lake Biotic Index, obtained by applying the formula $LBI = 2.5\sqrt{(Bl{*}Df)}$ with $0 \leq \mathrm{LBI} \leq 20$.

In the D*f* index the vf/vl ratio was inflected by the k coefficient in order to obtain relative, comparable D*f* values (Verneaux et al., 1993), because the bathymetric loss of taxa depends on the littoral taxonomic richness. A littoral fauna quality index (ql) has been included in D*f* because lakes with little loss of taxa from littoral to deep zones may be either of high biological quality or, on the contrary, highly polluted (Verneaux et al., 2004a,b).

The B*l* index is related to the Lake trophic potential (conductivity, calcium and nutrients), while the D*f* index is associated with the dissolved oxygen at the bottom and the organic matter content of the sediment. Based on the B*l* (quantitative) and D*f* (qualitative) indices, the LBI of each monitoring station was calculated in both autumn (April) and spring (November) 2013.

2.5. Statistical analyses

A nonparametric Kruskal-Wallis and Wilcoxon statistical analysis was performed on the physical chemical response variables in the sediment (pH, REDOX and POM) and LBI results. This enabled us to detect differences between the autumn and spring seasons using the R-Statistics (v. 3.2.2. 2015) program.

3. Results

3.1. Taxa abundance and diversity

A total of 978 individuals were obtained, corresponding to 405 individuals in the littoral zone (5 m) and 325 individuals in the sub-littoral zone (25 m) in autumn and 182 individuals in Z*l* (5 m) and 66 individuals in Z*f* (25 m) in spring.

A total of 9 taxa were recorded in autumn and 8 in spring. Of these, representatives of the families Hyallelidae (*Hyallela sp.*), Hyriidae (*Diplodon chilensis* and Tubificidae were the most abundant in autumn and Chironomidae, Hyallelidae and Hyriidae, in spring. Furthermore, Aeglidae (*Aegla denticulata, A. lacustris*), Chilinidae (*Chilina sp.*)*,* Sphaeridae (*Pisidium sp.*), Tubificidae (Glossiphoniidae) and Hirudinea were recorded in both autumn and spring. A specimen of the family Gomphidae (Insecta Odonata) was found in the littoral zone (Z*l*) in autumn. The highest average densities in the littoral were obtained during the autumn season, with Hyallelidae being the most represented taxon in autumn and

Hyriidae in spring (Table 1).

The taxon percentages of occurrence show that the *ql* (value established by the least tolerant taxon represented in at least half of the samples collected in the littoral zone) corresponded to the family Hyriidae (represented by *D. chilensis*), with 100% occurrence in autumn and 76% in spring (Table 1).

The tolerance value for this family is 0.6 according to the table (Figueroa et al., 2007) used as a reference in this study; we transformed the scale of values from 1 to 10 to 0.1 to 1 (according to Verneaux et al., 2004a).

3.2. Lake Biotic Index (LBI)

The Biogenic Potential of the littoral (B*l*), relative to the linear gradient of diversity in the different seasons, fluctuated between 8.25 to 16.37 in autumn (April) and from 8.03 to 10.93 in spring (November). While the taxonomic Deficit (D*f*) presented values of 0.56–0.86 in autumn and 0.53 to 1.05 in spring.

Based on these descriptors, the LBI value was obtained for each macrozoobenthos collection season. The LBI fluctuated from 5.78 to 8.73 in autumn and from 5.81 to 7.47 in spring (Table 2).

The low B*l* values recorded in all the stations over the two sampling periods, indicate an oligobiosis condition (low biogenic potential) arising from low species richness and density in the littoral. While D*f* that are elevated above the axis average, indicate good taxon preservation at a depth of 25 m. The variation observed in this descriptor among different stations is due to taxon presence-absence in Z*f* recorded in each station. These results categorize Lake Rupanco as oligo-eubiotic, a condition indicative of lakes with low mineral and high oxygen content, associated with poor organic matter content for the deep sediment.

Both the LBI index and the physicochemical pH and Redox (NHE) variables present significant differences ($p < 0.05$) between the seasonal samples (Table 3, Figs. 2, 3a, 3b). The % particulate of organic matter (POM) was the only variable that did not present such differences. The quantitative component of the LBI (B*l*) was the cause of the temporal differences, associated with variations in macroinvertebrate abundance values (Fig. 3).

4. Discussion and conclusions

For the first time in Chile, this study contrasts the results of studies on the water column in Lake Rupanco using a biological index based on benthic communities. According to the results, the oligo-eubiosis condition that characterizes the lake represents good ecological integrity and is consistent with the trophic level reported in previous studies based on physicochemical parameters and chlorophyll concentrations (DGA, 2014; Soto, 2002; Woelfl et al., 2003). The results obtained from applying the LBI in this study coincide with those recorded in Lake Llanquihue during the spring of 2010. In the latter case, results indicated an oligo-eubiosis condition defined by the typology graph of LBI, with values of 0.77 for the taxonomic deficit index (Df), 6.1 for the littoral biogenic index (Bl) and 5.4 for the LBI (Ecosistemas, 2011). Similarities between the results in Lake Rupanco and Lake Llanquihue are mainly due to the fact that they share a paleoclimatic origin of 18,000 BP (Batist et al., 2008) and to the trophic characterization of both lakes, which makes them suitable for applying bioindicators associated with the benthic subsystem.

The littoral benthic community provides the quantitative data of the LBI based on information regarding taxonomic richness and density. In this study, both descriptors were low, determining a low biogenic potential (B*l)* which, when accompanied by an above average taxonomic deficit (D*f*), defined LBI values within the oligo-eubiosis range. D*f* dispersion would be due to differences in number of taxa collected in Z*l* for each station. Seasonal differences (autumn-spring) in the LBI were significant. This would be associated with depth of the euphotic zone which varies according to season, as well as the equally significant variation in sediment pH and Redox potential that occurs between seasons.

On comparing the results of our study with those obtained in other continental water bodies located in the northern hemisphere, we found that our values are inferior to those established by Verneaux et al. (2004a) for 10 French lakes (LBI of 9–18) and to those obtained by Dermott and Pachkevitch (2012): LBI of 12.8–21.3 in Bay of Quinte, Batchawana Bay (Lake Superior) and Hamilton Harbour in Canada. Soto (2002) suggests that, in despite the fact that they are of a similar size, differences in the behavioral patterns of the North Patagonian Lakes (NPL) and those of the temperate lakes in the northern hemisphere could be due to nitrogen limitation affecting the NPL as a result of minimum levels of industrial development, as well as mixing depth, which is greater in the NPL where the epilimnion is deeper. Other factors can also be highlighted, such as wind exposure and oceanic influence, which are linked to the smaller land mass surrounding the NPL in comparison to the lakes in large continents such as North America, Europe or Asia. In this context, Soto (2002) indicates that lakes with mixing depths below 25 m, will be more sensitive to nutrient uptake produced by agricultural activity or salmon fish farming installations.

Regarding the physicochemical variables measured in the sediment, the pH levels recorded in autumn and spring fall below the tolerance level determined by Resolution 2612/2009, modified by Resolution 1508/2014, which establish anaerobiosis conditions in marine and continental waters. It should be noted that organic matter dissolved in the bottom is specific to each environment and sample period and that Lake Rupanco presented an Oligotrophic condition in 100% of the cases studied. These low lake trophy levels are present throughout the entire average annual cycle and were maintained between the years 2012 and 2013 (DGA, 2014). It has been established that an increase in organic matter and a low Redox potential are associated with eutrophication, and that high OM concentrations have correlated with benthic macroinvertebrate presence (DGA, 2010).

The NPL present moderate taxonomic richness and a high degree of endemism in freshwater invertebrates (Jara et al., 2006). The highest taxon density for limnetic crustaceans is found between the VIII and X

Table 1
Relative abundance (%), density in the littoral zone (d*l*) of the taxa collected in autumn and spring. For Z*f* (25 m) (*) indicates presence. Percentage of occurrence and tolerance value of taxa collected in autumn and spring from the littoral zone (in bold, taxon used to calculate the q*l*).

Taxon	Autumn			Spring			Tolerance value	Sample ocurrence (%)	
	Abundance (%)	d*l* (ind/m2)	Z*f*	Abundance (%)	d*l* (ind/m2)	Z*f*	q*l*	Autumn	Spring
Chironomidae	12	700	*	21	390	*	0.2	67	47
Aeglidae	3	110	*	1	20		0.6	33	12
Hyalellidae	32	1300	*	13	240	*	0.6	72	29
Chilinidae	3	130	*	8	150	*	0.6	50	41
Hyriidae	20	800	*	48	880	*	**0.6**	**100**	**76**
Sphaeridae	2	100	*	0	0		0.3	22	0
Tubificidae	21	840	*	8	140	*	0.1	89	35
Hirudinea	1	60	*	0	0		0.3	22	0
Gomphidae	0.2	10		0	0		0.7	6	0

Table 2
Littoral biogenic index (B*I*) (quantitative), taxonomic deficit (D*f*) and LBI during the monitoring stations used to calculate the LBI in Lake Rupanco during autumn and spring (year 2013).

Station	Litoral substation (5m)		Sublittoral substation (25m)		BI		Df		LBI	
	Latitude	Longitude	Latitude	Longitude	Autumn	Spring	Autumn	Spring	Autumn	Spring
LRC1	40°52′08.3″	72°15′49.9″	40°52′08.3″	72°15′40.8″	8.25	10.22	0.65	0.53	5.78	5.81
LRC2	40°52′41.1″	72°22′41.1″	40°50′40.7″	72°22′27.7″	16.37	8.52	0.69	1.05	8.39	7.47
LRC3	40°51′53.5″	72°31′26.5″	40°51′52.2″	72°31′23.7″	15.29	8.03	0.73	1.05	8.33	7.25
LRC4	40°50′18.6″	72°29′43.8″	40°50′18.5″	72°29′50.4″	15.26	9.39	0.8	0.84	8.71	7.01
LRC5	40°46′36.2″	72°31′17.7″	40°46′36.2″	72°31′23.7″	14.97	10.93	0.56	0.69	7.26	6.88
LRSUBVI	40°46′06.0″	72°37′44.0″	40°46′01.0″	72°37′40.7″	14.19	9.52	0.86	0.58	8.73	5.89

Table 3
Results of the non-parametric statistics test, for comparing LBI and physico-chemical variables of sediment between the sampling seasons of autumn (April) and spring (November) 2013, in Lake Rupanco.

	Environmental indicators		Kruskal-Wallis	Wilcoxon	Significant level
	Autumn (mean)	Spring (mean)	α = 0.05	α = 0.05	
LBI	7.83	6.72	0.00000177	0.00000183	(***)
pH	6.97	6.67	0.003	0.003	(**)
REDOX (NHE)	180.77	123.39	0.0046	0.0047	(**)
POM	1.15	1.57	0.3087	0.3115	not sig.

regions of Chile, with the X region hosting the greatest number of species (15 taxa). Sixteen of the 18 species of *Aegla* and three of the seven species of *Hyalella* are found exclusively in Chile and numerous species of *Aegla,* a genus of limnetic malacostraceans (fresh water crabs), are under threat. This latter group is very important and has been used as the basis on which to prioritize conservation in 18 eco-regions of southern South America (Jara et al., 2006). *D. chilensis* was the taxon indicator with the highest sampling occurrence. A high relative abundance of this species was observed in spring and of *Hyalella sp.*, in autumn. The Chironomidae, Chilinidae and Tubificidae were well represented in the samples, while Aeglidae and Sphaeridae were scarce.

The importance of this lake ecological index (LBI) is evidenced by its contribution to knowledge about the composition, structure and bathymetric distribution of the macrozoobenthos. This is particularly so in systems where studies of this nature are lacking and which host species under threat or in danger of extinction which, in some cases, have been mentioned as potential impact indicators, for example: *Aegla sp.*, *Hyalella sp.* y *Diplodon sp.* (Bazzanti et al., 2012). The operationality of biological quality indices needs to be standardized, or at least well documented, with reproducible field and laboratory protocols; the main component supporting the system's capacity to sustain biodiversity is that of benthic invertebrate assemblages (Lafont, 2007). Sites with conditions similar to those found in the natural environment must also be located, since these “reference sites” are required to undertake comparisons with sites that may be experiencing impacts (Revenga et al., 2005).

Verneaux et al. (2004b) state that there is no simple relationship between taxon richness and community density; no factor can be considered in isolation given the complexity of interactions among the biotic and abiotic factors that affect macrobenthos distribution. We determined that dissolved oxygen and organic matter parameters are variables that are connected with macrobenthos bathymetric distribution used recently in lake typology. They also found that the limit values specified can provide biotic potentials that can be modified by other factors. Given that different environmental stressors affect the structure and functioning of the ecosystems to various degrees, integrative and multi-metric approaches give a more holistic representation of the state of the ecosystem, combining physical, chemical and biological measurements in one synthetic index (Revenga et al., 2005; Roley et al., 2014).

Hirabayashi et al. (2015) studied the bathymetric distribution of benthic macroinvertebrates in a deep oligotrophic lake in Japan (Lake

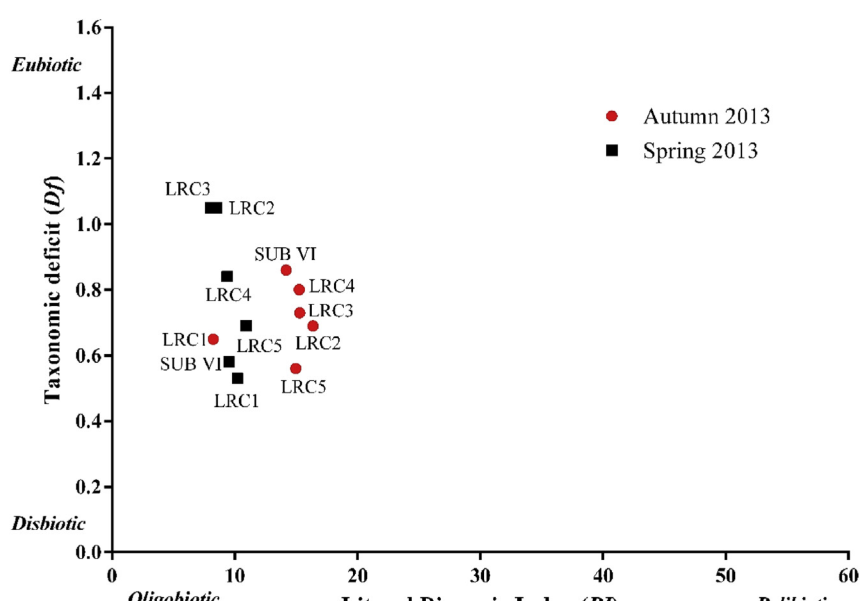

Fig. 2. Typology graph of LBI construed with a littoral biogenic index (B*I*) (axis X) and a taxonomic deficit index (D*f*) (axis y) with seasonal results from Lake Rupanco. The value in brackets corresponds to LBI calculated for each monitoring station based on the B*I* and D*f* according to the methodology of Verneaux et al. (2004a,b).

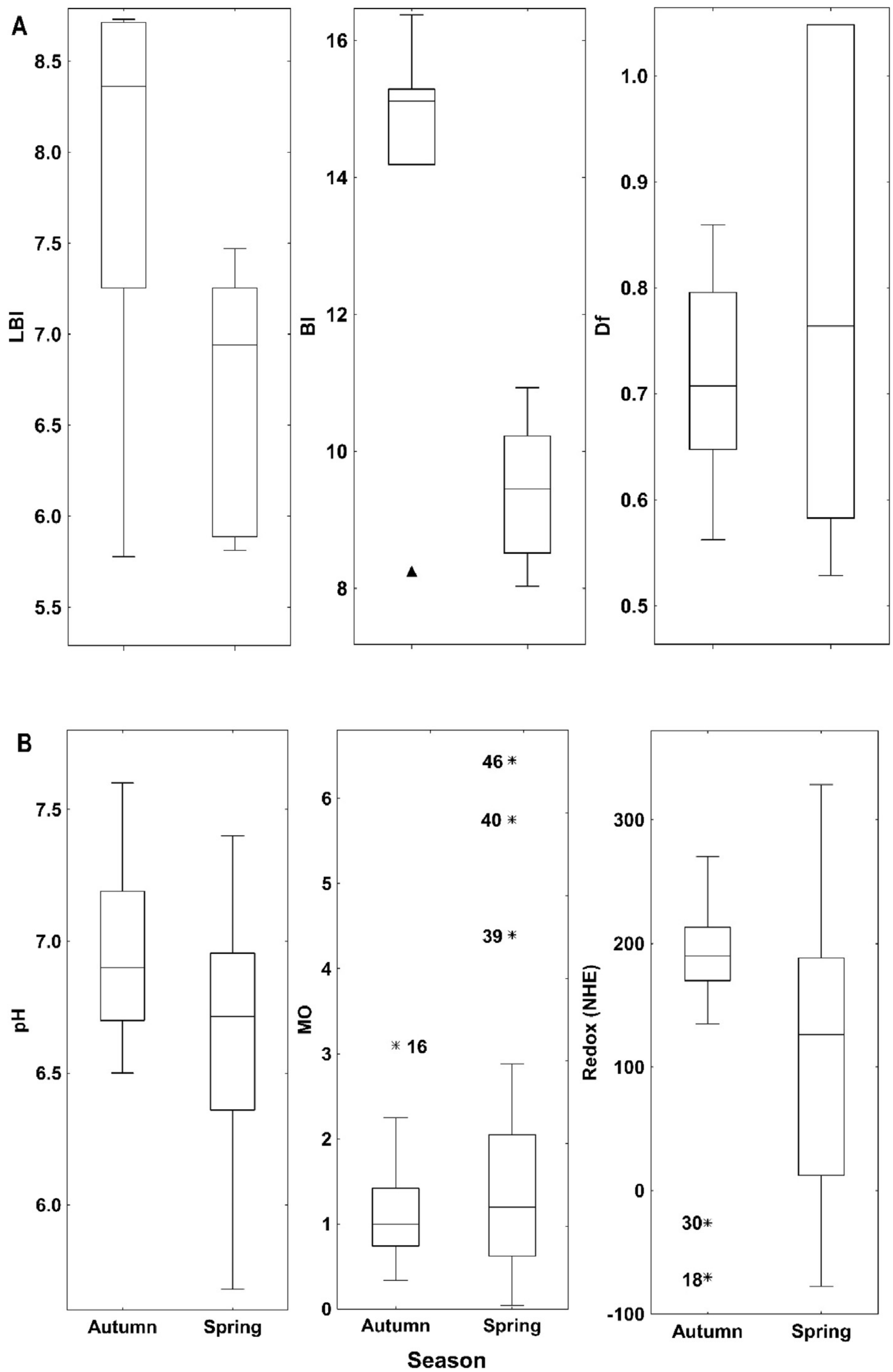

Fig. 3. Boxplot of the seasonal variation, autumn and spring 2013; A: Lake Biotic Index (LBI), the Littoral Biogenic Index (B*I*) and the Taxonomic Deficit Index (D*f*) in Lake Rupanco; B: physicochemical variables measured in sediments from Lake Rupanco.

Motosu) and found that benthic macroinvertebrate density decreases as depth increases, although the density of one species in particular increased in the deepest lake zone. In Lake Llanquihue macroinvertebrate richness and abundance decreases in depths over 30 m (Ecosistemas, 2011). This highlights the importance of taxonomic determination at the genus or species level when biological indices are proposed as indicators. Similarly, it underlines the difficulties encountered when defining applicable threshold values.

In Chile, bioindicators have been used to determine the environmental quality of rivers (Figueroa et al., 2007), but they have not yet been used in the case of lakes. Studies in this respect are most advanced in the Bío-Bío river basin, where the use of bioindicators is considered as part of the vigilance program. At present the Comisión Nacional de Medio Ambiente (Chilean National Environmental Commission) has formulated different studies that include the objective of applying bio-indicators to evaluate the quality of water bodies. In addition, the Departamento de Ciencias Ambientales y Recursos Naturales (Department of Environmental Studies and Natural Resources) of the Universidad de Chile (2010) elaborated a preliminary proposal for the typology and classification of water bodies in Chile. This has proved to be the first step towards elaborating a typology, up to now inexistent, that, according to the authors, will make it possible to establish water quality standards associated with biological or bio-indicators in the future.

Finally, studies on a global and local level have concentrated on indicators of environmental quality, both physicochemical and biological, present in the water column. We suggest that an integrated pelagic-benthic approach to the management of the lentic environments would not only aid and improve determination of the environmental condition of these ecosystems, but also stimulate the use and development of more effective management tools for the control and mitigation of pollution in these ecosystems (Vadeboncoeur et al., 2002). We believe that the LBI index is a good option for incorporating both macroinvertebrate bio-indicators and analysis of physicochemical variables of the benthic dimension as part of the evaluation and management procedures related to lacustrine bodies. The application of the LBI would also benefit environmental evaluation associated with integrated basin management, a criterion already adopted by more than two thirds of the nations of the world (Aguirre-Núñez, 2011).

Declarations

Author contribution statement

German Leiva, Norka Fuentes: Conceived and designed the experiments; Performed the experiments; Analyzed and interpreted the data; Contributed reagents, materials, analysis tools or data; Wrote the paper.

Sara Zelada: Conceived and designed the experiments; Analyzed and interpreted the data; Wrote the paper.

Catalina F Ríos-Henríquez: Contributed reagents, materials, analysis tools or data; Wrote the paper.

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Competing interest statement

The authors declare no conflict of interest.

Additional information

No additional information is available for this paper.

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