Macroinvertebrates and Fishes as Bioindicators of Stream Water Pollution

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http://dx.doi.org/10.5772/65084

Abstract

Freshwater ecosystems worldwide have been progressively deteriorated during the past decades due to an increasing human pressure that has lead to a decrease in aquatic biodiversity. Among the human activities of high impact on freshwater ecosystems is the land-use change, principally from native forests to agriculture. To evaluate the impacts of human activities on water quality, a traditional approach has considered the use of single physical-chemical parameters. However, this approach may be insufficient to fully assess the impact of these human activities on freshwaters. Therefore, there is a need for alternative tools such as the indices of biotic integrity that may provide a complement to traditional approaches. In the literature, there are several examples of biotic indicators that have shown promising results in evaluating water quality including the use of macroinvertebrates and fish diets. Here, we provide a review of the indicators of biotic integrity that included fish assemblages as well as macroinvertebrates as bioindicators. We identify pros and cons of using aquatic communities as indicators of water quality. Finally, we develop a procedure that combines fish and macroinvertebrate assemblages as bioindicators and discuss their effectiveness using illustrative examples from streams under several agricultural uses in the Mediterranean region of Chile.

Keywords: biological monitoring, biotic index, macroinvertebrates, fishes, Mediterranean, Chile
1. Introduction

Of all the water on earth, freshwater accounts for just 0.01% and covers only 0.8% of the planet’s surface [1]. Freshwaters are among the most threatened ecosystems of the world, and thus, understanding their health statuses is of special relevance. Indeed, the physical, chemical, and biological integrities of water are highly important for successfully implementing conservation and management strategies before ecosystem health or biotic integrity are affected [2–4]. This chapter provides a review of known biotic integrity indicators, including benthic macroinvertebrate and fish communities that have been proposed to serve as water quality indicators. In addition, the pros and cons of using aquatic communities as water quality indicators are discussed. Finally, we present a research case study in which benthic macroinvertebrate and fish communities are used as bioindicators, in addition to discussing the effectiveness of using illustrative examples for streams subject to several agriculture uses in a region of Chile dominated by agricultural activities.

Worldwide, a primary threat to freshwater ecosystems is the rapid changes occurring in land uses (Figure 1), a situation that has intensified over the past decade [5, 6]. Most recent land use conversion has been for crop production, which notably impacts proximal ecosystems due to changes over extensive crop areas [7]. In particular, the fertilizers and pesticides used in agriculture negatively affect freshwater ecosystems by draining into rivers, where eutrophication and other negative effects, such as high sediment deposits and postsedimentation, subsequently occur. Furthermore, the extensive land use of farming many times results in landscape deforestation, which often arrives to the riverbank itself. This deforestation can increase the temperature of and quantity of light in river water. When coupled with eutrophication, the trophic changes within the aquatic ecosystem can be disturbed, causing, for example, a decreased quantity of aquatic taxa as compared to rivers with fewer alterations [8, 9].

Figure 1. Examples of land use in the central-south of Chile. Left: Stream nearby corn crops, right: Stream borderer by native forest of the Maule Region watershed (photographs by P. Fierro).
2. Indicators of aquatic ecosystem health

The definition of a healthy ecosystem has been widely debated in the literature. Nevertheless, the definition proposed by Rapport is one of the most widely accepted [10]. This definition states that a healthy ecosystem is defined by the “absence of danger signals in the ecosystem, the ability of the ecosystem to quickly and completely recover (resilience), and/or the lack of risks or threats that push the ecosystem composition, structure, and/or function.” The purpose of monitoring aquatic ecosystem health is to identify physicochemical and biological changes arising from anthropogenic impacts [11]. This information is crucial for managers and policy makers to make informed decisions towards improving the environment and, consequently, human health [12].

Traditional techniques for measuring water quality and to establish aquatic health assess a number of physical and chemical parameters of the water. However, these measurements do not accurately account for the real impacts that physicochemical activities have on freshwater ecosystems [13]. Indeed, these parameters interact and evidence accumulative effects over time, the impacts of which can finally affect aquatic biota [14]. Due to this, other measurements that consider non-natural disturbing effects on ecological integrity should be used to calculate the quality of aquatic resources [15]. Indices based on aquatic biota have been widely successful in determining the integrity of aquatic ecosystems [16].

The use of indices that evaluate water quality through biological parameters, such as freshwater ecosystem structure and performance, has considerably increased in recent years and has gained recognition as an important measure for calculating the global integrity of freshwater ecosystems [17–19]. Biological monitoring is advantageous in that it can integrate and reflect accumulative changes over time, which is in contrast to a number of other methods, such as flow regimen, energetic resources, and biotic interactions [20, 21]. Another benefit is that the high fauna diversity found in aquatic ecosystems, which include microorganisms, algae, periphyton, phytoplankton, zooplankton, macroinvertebrates, fish, and mammals, can be included in evaluations of river health [4].

Among fauna, fish and macroinvertebrate assemblages have been highlighted as good bioindicators for monitoring ecosystem degradation related to farming and forestry, as well as to urban and industrial effluents [9, 22]. Diverse proxies are used to measure ecosystem condition, such as species density and the presence/absence of several species in assemblage structures [23]. A notable advantage of using these aquatic biota is the relative simplicity of their capture and sampling [24, 25]. In particular, the sampling of fish assemblages can be performed through electrofishing, a highly common tool, whereas macroinvertebrate sampling is facilitated and simplified by Surber, D-frame dip, and kick nets (Figure 2).

Furthermore, recent studies report that the stomach contents of salmonids (i.e., *Oncorhynchus mykiss* and *Salmo trutta*) contain a diversity of invertebrate prey present in the benthos of nonintervened (hereafter termed “native”) basins, thereby reflecting anthropogenic impacts to the basin [26]. Related to this, Fierro et al. [6] reported similarities in stomach contents and prey diversity of the benthos in river sections with land use different than in the basin.
Likewise, similarities have been found between rivers with more local perturbation, such as through the effects of dams [27, 28]. Therefore, the *O. mykiss* diet might represent an effective bioindicator for evaluating environmental disturbances within the entire basin [6].

![Figure 2](image_url)

**Figure 2.** Left: Fish communities sampled using electrofishing. Right: Aquatic macroinvertebrates sampled using a Surber Net (photograph by P. Fierro).

Among the ecological indices commonly used to evaluate river health, three primary groups exist – biotic indices, multivariate methods, and multimetric indices [15, 19]. Of these, multimetric indices are the most recommended since a large quantity of data can be considered and since these indices may also identify the cause(s) of degradation. This information can then be applied to obtain better understandings of ecosystem status [4]. In turn, biotic indices evaluate river health based only on organism tolerance to organic pollution. One of the most well-known biotic indices is the Hilsenhoff Biotic Index [29], which has been widely used and adapted around the world (e.g., [30–32]). Continuing, multivariate methods require the use of models that relate physicochemical properties of rivers with observed organisms, which are represented under reference (relatively pristine) conditions. These models then compare the observed organisms with those that were “expected.” This comparative method can ultimately detect potentially degraded areas. The most widely used multivariate index is the River Invertebrate Prediction and Classification System [33], which was first implemented in the UK and then adapted to other countries, including Australia [34]. Finally, multimetric indices capture broad characteristic of community structure and function (metric), thus providing a broader understanding of the events occurring in the river [35]. Multimetric indices are powerful tools for establishing the consequences of human activities. These effects may include a high amount of specific and blurred disturbances (nonpoint pollutant discharge), which encompass impacts arising from agriculture, grazing, deforestation, physical alterations of river or bank habitats, damps, sewage discharges, urban areas, and mining [36, 37]. These indices can be applied in several animal assemblages, plant communities, and ecosystems, including terrestrial, marine, and freshwater environments [35]. Corresponding indices of integrity are frequently performed and applied in fish [38] and macroinvertebrates [39]. A summary that contrasts among the three types of indices is presented in Table 1.
3. Assessing the ecological integrity of streams

Ecological integrity, which is also referred to as river health or ecological status, is a measure of the global condition of an aquatic ecosystem. This measurement integrates physical, chemical, and biological integrity elements [15, 17]. Importantly, biological integrity is defined as the ability of aquatic ecosystems to support and maintain a balanced and integrated community with adapted organisms and a composition, diversity, and functional organization comparable to natural habitats within the same region [40–42]. Therefore, a loss of integrity indicates any human-induced positive or negative divergence of the system from a natural, model condition [43].

The Index of Biotic Integrity (IBI), which was initially developed for western USA rivers by [28], is the most used index based on fish assemblages. Consequently, the IBI has been adapted for use to numerous rivers on all continents to evaluate stream health [4, 28]. Indeed, since the creation of the IBI, over 2374 researchers, as of 2014, have used, modified, or mentioned the importance of the IBI (Google Scholar). Furthermore, the number of citations for the IBI grew exponential until 2005, at which point citations “stagnated” near 140 studies per year (Figure 3).

Worth highlighting, of the studies presented in this review, the most important milestone occurred from 1986 to 1990. During this period, researchers first began adapting and making modifications to indices based on fish, in addition to these indices being applied in reports to the US government. Between 1991 and 1995, integrity indices were developed for several groups, including macroinvertebrates, birds, and zooplankton. Furthermore, this period was witness to index adaptations to marine and estuary environments. Even terrestrial environments were assessed by the IBI to measure the environmental quality of forests. Between 1996 and 2000, the IBI continued to expand to other groups and environments, such as periphyton.
communities, macrophytes, corals, and wetlands. Corresponding adaptations of the IBI to other continents, including Africa, Europe, and South America (Brazil), also occurred [44, 45]. Since 2001, this index is in use on almost all continents and has been adapted several times to different ecoregions within the same countries.

![Figure 3. Accumulative number of worldwide publications on the index of biotic integrity around the world, starting with the first related publication by [28] (Source: own elaboration).](image)

The advantage of establishing the biotic integrity of rivers based on fish arises as these organisms are present in all, or almost all, rivers, even those that are polluted. Additionally, extensive life history information is available for many species, and fish assemblages generally represent a variety of trophic levels. Indeed, fish are located within the top of the aquatic food chain and can thus help to provide an integrated view of basin environments. Other benefits of the IBI using fish are that fish populations are relatively stable in the summer, when most monitoring occurs; fish are easily identifiable; and the general public can relate to statements about the conditions of fish assemblages. On the other hand, a noted disadvantage of the IBI is that fish are highly mobile, making sampling difficult. Indeed, large groups of personnel, various tools, and an extended period in the field are needed to record daily and seasonal variations [31].

Although less used, the Benthic Index of Biotic Integrity (B-IBI) was developed by Kerans and Karr [46] for rivers of the Tennessee Valley (USA), using the IBI as an initial base [28]. The advantages of using macroinvertebrates as bioindicators are a great biodiversity and an extreme sensitivity and fast response of many taxa to pollution. This quick response is likely due to many macroinvertebrates being sessile and having aquatic life cycles, thus any alterations in environmental limits could lead to death [14]. One significant disadvantage of the B-IBI is that a taxonomic specialist is needed to identify the macroinvertebrate species, which takes a long time. To address this limitation, Rolls et al. [27] used higher levels of taxonomic identification (e.g., genus, family, or both) as a method for adequately describing taxa traits for B-IBI use. Through this technique, a greater cost-benefit might be obtained as less time will be required to taxonomically identify species. Indeed, in countries with few taxonomists and without access to species-level identification keys, application of the B-IBI is very important, as is the case in Chile. Other disadvantages include widespread ignorance about the life histories of many species. Furthermore, it is more difficult for the general public to feel
4. Chile: a case study

Mediterranean-climate ecosystems are priority areas of conservation efforts; however, these ecosystems remain threatened globally due to environment degradation [48, 49]. Of the five regions worldwide that present this climate, Chile is the least studied in regards to aquatic ecology [50]. This is despite reporting high national endemism and being considered among the 34 biodiversity hotspots in the world [48, 51].

The Mediterranean-climate ecosystem basins of Chile are host to significant industrial activities. This constitutes an increasing problem for aquatic ecosystems due to severe site degradations. Of the various human activities that threaten this region, land use and land cover conversion are highly ranked [52]. Indeed, while many activities directly or indirectly influence aquatic ecosystems, land use is the principal determinant of water quality and of water quantity entering aquatic ecosystems [53]. Furthermore, land cover conversions for crop production or monoculture plantations directly affect freshwater fauna, decreasing, for example, aquatic insect densities and possibly inducing local extinction [9].

In Chile, the use of bioindicators to assess water quality is limited, with applications focused on benthic macroinvertebrate assemblages through a modified Hilsenhoff Biotic Index (e.g., [31, 32, 54]). Notably, these studies were conducted only as a part of basic scientific research as no regulations or laws in Chile stipulate the use of biological criteria for measuring water quality. In contrast, bioindicators are widely used in other countries for assessing and monitoring water quality, often times to meet governmental regulations. In the United States, for example, the Environmental Protection Agency established the “Use of Biological Assessments and Criteria in the Water Quality Program” [55], whereas the European Environment Agency has used biomarker-based monitoring in a number of countries (e.g., Austria 1968 and United Kingdom 1970 [56]).

5. Effects of agricultural land use on aquatic ecosystems

Agricultural land use can increase the delivery of several compounds, such as phosphorous and nitrogen, to fluvial ecosystems. In turn, this can produce eutrophication and, consequently, limit the presence of some macroinvertebrate and fish species. For example, when 22 streams were sampled across five Mediterranean-climate watersheds in the farming, central-south region of Chile, agricultural land use was found to be an important predictor of both macroinvertebrate and fish assemblages. Specifically, significant differences in the composition of macroinvertebrate (Figure 4; ANOSIM: \( r = 0.203, P = 0.01 \)) and fish (Figure 5; ANOSIM: \( r = 0.563, P = 0.01 \)) assemblages between land use types were found. In addition,
taxonomic diversity of macroinvertebrates was higher in native streams than agricultural streams (average Shannon-Wiener index in native streams: 1.5, agricultural streams: 1.1).

**Figure 4.** nMDS plot based on the composition of macroinvertebrates in 11 native streams and 11 agriculture streams in Mediterranean-climate ecosystems in the farming, central-south region of Chile. The data matrix was constructed using the Bray-Curtis Similarity Index with the square-root transformation of data (9999 restarts). Axes are relative scales and therefore appear without legends (personal data P. Fierro).

**Figure 5.** nMDS plot based on the composition of fish in seven native streams and seven agriculture streams in Mediterranean-climate ecosystems in the farming, central-south region of Chile. The data matrix was constructed using the Bray-Curtis Similarity Index with the square-root transformation of data (9999 restarts). Axes are relative scales and therefore appear without legends (personal data P. Fierro).
The principal difference in both assemblages was community heterogeneity, where native streams were constituted by greater abundances of Ephemeroptera larvae and presented Plecoptera larvae, while in agriculture streams, Diptera larvae and gastropods were more abundant (Figure 6). Regarding fish assemblages, a higher amount of taxa were recorded in native streams, and included exotic trout (e.g., *O. mykiss* and *S. trutta*; Table 2). These species are unique to environments with low temperatures and high oxygen content, indicators of good water quality. In contrast, the catfish *Trichomycterus areolatus* (Figure 7) was recorded at all native and agriculture sites, supporting the broad environmental tolerance of catfish species in general [57].

![Figure 6. Macroinvertebrate classes found in agricultural dominated and reference streams (N = 22) (unpublished data P. Fierro).](image)

| Species                          | Agriculture | Native  |
|----------------------------------|-------------|---------|
| Diplomystes nahueltuetaensis     | 0%          | 4.4%    |
| Trichomycterus areolatus         | 20.9%       | 34.1%   |
| Brachygalaxias bullocki         | 0.2%        | 0%      |
| Cheirodon galusdae               | 3.5%        | 0.6%    |
| Percilia gillisi                | 20.4%       | 28.7%   |
| Basilichthys microlepidotus      | 0%          | 1.6%    |
| Percichthys trucha              | 3.2%        | 0.8%    |
| Gambusia holbrooki*              | 50.3%       | 0%      |
| Cnesterodon decemmaculatus*      | 0.1%        | 0%      |
| Oncorhynchus mykiss*             | 1.8%        | 26.7%   |
| *Salmo trutta*                   | 0%          | 3.2%    |
| Cyprinus carpio*                 | 0.5%        | 0%      |

*Exotic species (unpublished data P. Fierro).

Table 2. Species richness and relative abundances of fish species in agriculture and native streams in the farming, central-south region of Chile.
6. Conclusion

Macroinvertebrates and fish are used to evaluate the health of streams worldwide. The case results presented in this chapter evidence the importance of using one or more taxonomic groups in bioassessments, where both evaluated assemblages efficiently responded to pressures of human agricultural activities. These results suggest that macroinvertebrates and fish can be used as indicators of water pollution in monitoring programs. Using both assemblages as bioindicators presents several methodological advantages as compared to only assessing physicochemical parameters. These include low costs, easily identifiable fish, and, principally, the sensitivity of both assemblages to different stressors. For example, macroinvertebrates responded differently to substrate compositions than fish, which, in turn, responded to variables such as stream morphometry.

Rivers are increasingly affected by multiple physicochemical and biological stressors. Considering the ongoing rise in environmental management programs for aquatic communities, one related future goal is to develop appropriate indices, such as multimetric or biotic integrity indices, to differentiate between taxonomic groups, thereby facilitating assessments of stream health. However, the effectiveness of these indices will be highly dependent on applicability in different ecoregions.

Acknowledgements

This work was supported by Becas Doctorado Nacional CONICYT and funded in part by DAND Codelco-Andina, Fondecyt 1151375 and Fondap-Ideal 15150003 to L.V-C. We are grateful to Loretto Arriagada, Alfonso Jara, Gustavo Diaz, Jorge González, Cindy Cerna, and Aliro Manosalva for technical assistance in the field and laboratory.
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References

[1] Dudgeon D, Arthington A, Gessner M, Kawabata Z, Knowler D, Lévêque C, Naiman R, Prieur-Richard A, Soto D, Stiassny M, Sullivan A. Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews 2006; 81: 163–182. DOI: 10.1017/S1464793105006950.

[2] Lyons J, Navarro-Pérez S, Cochran P, Santos E, Guzmán-Arroyo M. Index of biotic integrity based on fish assemblages for the conservation of streams and rivers in West-Central Mexico. Conservation Biology 1995; 9: 569–584. DOI: 10.1046/j.1523-1739.1995.09030569.x.

[3] Butcher J, Stewart P, Simon T. A Benthic community index for streams in the Northern Lakes and forests ecoregion. Ecological Indicators 2003; 3: 181–193. DOI: 10.1016/S1470-160X(03)00042-6.

[4] Herman M, Nejadhashemi A. A review of macroinvertebrate- and fish-based stream health indices. Ecohydrology & Hydrobiology 2015; 15: 53–67. DOI: 10.1016/j.ecohyd.2015.04.001.

[5] Barletta M, Jaureguizar AJ, Baigun C, Fontoura NF, Agostinho AA, Almeida-Val VMF, Val AL, Torres RA, Jimenes-Segura LF, Giarrizzo T, Fabré NN, Batista VS, Lasso C, Taphorn DC, Costa MF, Chaves PT, Vieira JP, Correa MFM. Fish and aquatic habitat conservation in South America: a continental overview with emphasis on neotropical systems. Journal of Fish Biology 2010; 76: 2118–2176. DOI: 10.1111/j.1095-8649.2010.02684.x.

[6] Fierro P, Quilodrán L, Bertrán C, Arismendi I, Tapia J, Peña-Cortés F, Hauenstein E, Arriagada R, Fernández E, Vargas-Chacoff L. Rainbow Trout diets and macroinver-
tebrates assemblages responses from watersheds dominated by native and exotic plantations. Ecological Indicators 2016; 60: 655–667. DOI: 10.1016/j.ecolind.2015.08.018.

[7] Allan JD. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics 2004; 35: 257–284. DOI: 10.1146/annurev.ecolsys.35.120202.110122.

[8] Wang L, Robertson DM, Garrison PJ. Linkages between nutrients and assemblages of macroinvertebrates and fish in Wadeable streams: implication to Nutrient Criteria Development. Environmental Management 2007; 39: 194–212. DOI: 10.1007/s00267-006-0135-8.

[9] Fierro P, Bertrán C, Mercado M, Peña-Cortés F, Tapia J, Hauenstein E, Caputo L, Vargas-Chacoff L. Landscape composition as a determinant of diversity and functional feeding groups of aquatic macroinvertebrates in southern rivers of the Araucania, Chile. Latin American Journal of Aquatic Research 2015; 43: 186–200. DOI: 10.3856/vol43-issue1-fulltext-16.

[10] Fu-Liu X, Shu T. On the study of ecosystem health: state of the art. Journal of Environmental Sciences 2000; 12: 33–38.

[11] Hughes R, Whittier T, Thiele S, Pollard J, Peck D, Paulse S, Mcmullen D, Lazorchak J, Larsen D, Kinney W, Kaufmann P, Hedtke S, Dixit S, Collins G, Baker J. Lake and stream indicators for the United States Environmental Protection Agency’s environmental monitoring and assessment program. In: McKenzie D, Hyatt D, McDonald V, editors. Ecological Indicator, I. Elsevier Applied Science, London, UK; 1992. p. 305–335. DOI: 10.1007/978-1-4615-4659-7_20.

[12] Weigel B, Henne L, Martínez-Rivera L. Macroinvertebrate-based index of biotic integrity for protection of streams in west-central Mexico. Journal of the North American Benthological Society 2002; 21: 686–700. DOI: 10.2307/1468439.

[13] Oberdoff T, Hughes RM. Modification of an index of biotic integrity based on fish assemblages to characterize rivers of the Seine Basin, France. Hydrobiologia 1992; 228: 117–130. DOI: 10.1007/BF00006200.

[14] Roldan G. Macroinvertebrates and value as water quality indicators - Los Macroinvertebrados y su valor como indicadores de la calidad del agua. Revista Academica Colombiana de Ciencias 1999; 23: 375–387.

[15] Oliveira SV, Cortes RMV. Environmental indicators of ecological integrity and their development for running waters in northern Portugal. Limnetica 2006; 25: 479–498.

[16] Karr J. Biological monitoring and environmental assessment: a conceptual framework. Environmental Management 1987; 11: 249–256. DOI: 10.1007/BF01867203.

[17] Barbour MT, Gerritsen J, Snyder BD, Stirling JB. Rapid bioassessment protocols for use in streams and Wadeable Rivers: periphyton, benthic macroinvertebrates and fish, second edition. EPA 841-B-99-002 U.S. Environmental Protection Agency; Office of Water; Washington, D.C.; 1999. 339 p.
[18] Karr J, Chu E. Sustaining living rivers. Hydrobiologia. 2000; 422/423: 1–14. DOI: 10.1023/A:1017097611303.

[19] Ollis D, Dallas H, Esler K, Boucher Ch. Bioassessment of the ecological integrity of river ecosystems using aquatic macroinvertebrates: an overview with a focus on South Africa. African Journal of Aquatic Science 2006; 31: 205–227. DOI: 10.2989/16085910609503892.

[20] Cairns J, Pratt JR. A history of biological monitoring using benthic macroinvertebrates. In: Rosenberg DM, Resh VH, editors. Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman & Hall, New York; 1993. p. 10–27. DOI: 10.1002/aqc.3270040110.

[21] Alba-Tercedor J. Aquatic Macroinvertebrates and water quality of streams - Macroinvertebrados acuáticos y calidad de las aguas de los ríos. IV Simposio del Agua en Andalucía, Almería 1996; 2: 203–213.

[22] Dos Santos DA, Molineri C, Reynaga MC, Basualdo C. Which index is the best to assess stream health? Ecological Indicators 2011; 11: 582–589. DOI: 10.1016/j.ecolind.2010.08.004.

[23] Hilty J, Merenlender A. Faunal indicator taxa selection for monitoring ecosystem health. Biological Conservation 2000; 92: 185 197. DOI: 10.1016/S0006-3207(99)00052-X.

[24] Li H, Li J. Role of fish assemblages in stream communities. In: Hauer FR, Lamberti GA, editors. Methods in Stream Ecology, second edition. Academic Press, San Diego, CA; 2007. p. 489–514.

[25] Merrit RW, Cummins KW. Trophic relationships of macroinvertebrates. In: Hauer FR, Lamberti GA, editors. Methods in Stream Ecology, second edition. Academic Press, San Diego, CA; 2007. p. 585–610.

[26] Vargas-Chacoff L, Quilodrán L, Bertrán C, Arismendi I, Fierro P, Tapia J, Peña-Cortés F, Hauenstein E. Food of rainbow trout and changes in soil use: the Chilean Example. In: Polakof S, Moon TW, editors. Trout: From Physiology to Conservation. Nova Science Publishers, New York; 2013. p. 65–80.

[27] Rolls R, Boulton A, Gowans I, Maxwell S, Ryder D, Westhorpe D. Effects of an experimental environmental flow release on the diet of fish in a regulated coastal Australian river. Hydrobiologia 2012; 686: 195–212. DOI: 10.1007/s10750-012-1012-5.

[28] Veloso C, Costa F, Santos P. Hydropeaking effects of on the diet of a Neotropical fish community. Neotropical Ichthyology. 2014; 12: 795–802. DOI: 10.1590/1982-0224-20130151.

[29] Hilsenhoff W. Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society. 1988; 71: 65–68. DOI: 10.2307/1467832.
[30] Lenat D. A biotic index for the southeastern United States: derivation and list of tolerance values, with criteria for assigning water-quality ratings. Journal of the North American Benthological Society. 1993; 12: 279–290. DOI: 10.2307/1467463.

[31] Figueroa R, Valdovinos C, Araya E, Parra O. Benthic Macroinvertebrates as indicators of water quality of southern Chile rivers - Macroinvertebrados bentónicos como indicadores de calidad de agua de ríos del sur de Chile. Revista Chilena de Historia Natural. 2003; 76: 275–285. DOI: 10.4067/S0716-078X2003000200012.

[32] Fierro P, Bertrán C, Mercado M, Peña-Cortés F, Tapia J, Hauenstein E, Vargas-Chacoff L. Benthic macroinvertebrate assemblages as indicators of water quality applying a modified biotic index in a spatio-seasonal context in a coastal basin of Southern Chile. Revista de Biología Marina y Oceanografía. 2012; 47: 23–33. DOI: 10.4067/S0718-19572012000100003.

[33] Wright JF, Furse MT, Moss D. River classification using invertebrates: RIVPACS applications. Aquatic Conservation: Marine and Freshwater Ecosystems. 1998; 8: 617–631. DOI: 10.1002/(SICI)1099-0755(199807/08)8:4<617::AID-AQC255>3.0.CO;2-#.

[34] Davies PE. Development of a National River Bioassessment System (AUSRIVAS) in Australia. In: Wright JF, Sutcliffe DW, Furse MT, editors. Assessing the Biological Quality of Fresh Waters: RIVPACS and Other Techniques. Freshwater Biological Association, Ambleside, UK, 2000. p. 113–124.

[35] Reynoldson TB, Norris RH, Resh VH, Day KE, Rosenberg DM. The reference condition: a comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. Journal of the North American Benthological Society. 1997; 16: 833–852. DOI: 10.2307/1468175.

[36] Barbour MT, Gerritsen J, Griffith GE, Frydenborg R, McCarron E, White JS, Bastian ML. A framework for biological criteria for Florida streams using benthic macroinvertebrates. Journal of the North American Benthological Society. 1996; 15: 185–211. DOI: 10.2307/1467948.

[37] Varandas S, Vitor R. Evaluating macroinvertebrate biological metrics for ecological assessment of streams in northern Portugal. Environmental Monitoring Assessment. 2010; 166: 201–221. DOI: 10.1007/s10661-009-0996-4.

[38] Karr J. Assessment of biotic integrity using fish communities. Fisheries. 1981; 6: 21–27. DOI: 10.1577/1548-8446(1981)006<0021:AOBIF>2.0.CO;2.

[39] Griffith M, Hill B, McCormick F, Kaufmann P, Herlihy A, Selle A. Comparative application of indices of biotic integrity based on periphyton, macroinvertebrates, and fish to southern Rocky Mountain streams. Ecological Indicators. 2005; 5: 117–136. DOI: 10.1016/j.ecolind.2004.11.001.

[40] Karr J, Dudley D. Ecological perspectives in water quality goals. Environmental Management. 1981; 5: 55–68. DOI: 10.1007/BF01866609.
[41] Karr J. Biological integrity: a long neglected aspect of water resource management. Ecological Applications. 1991; 1: 66–84. DOI: 10.2307/1941848.

[42] Angermeier PL, Karr J. Applying an index of biotic integrity based on stream-fish communities: considerations in sampling and interpretation. North American Journal of Fisheries Management. 1986; 6: 418–429. DOI: 10.1577/1548-8659(1986)6<418:AAIO-BI>2.0.CO;2.

[43] Westra L, Miller P, Karr J, Rees W, Ulanowicz E. Ecological Integrity and the aims of the global integrity project. In: Pimentel D, Westra L, Noss RF, editors. Ecological Integrity: Integrating Environment, Conservation, and Health. Island Press, Washington, DC. p. 19–41. DOI: 10.1093/ije/31.3.704.

[44] De Freitas Terra B, Hughes R, Rocha Francelino M, Gerson Araújo F. Assessment of biotic condition of Atlantic Rain Forest streams: a fish-based multimetric approach. Ecological Indicators 2013; 34: 136–148. DOI: 10.1016/j.ecolind.2013.05.001.

[45] Tiku Mereta S, Boets P, De Meester L, Goethals P. Development of a multimetric index based on benthic macroinvertebrates for the assessment of natural wetlands in Southwest Ethiopia. Ecological Indicators 2013; 29: 510–521. DOI: 10.1016/j.ecolind.2013.01.026.

[46] Kerans BL, Karr J. A benthic index of biotic integrity (B-IBI) for rivers of the Tennesse Valley. Ecological Applications 1994; 4: 768–785. DOI: 10.2307/1942007.

[47] Karr J, Chu E. Biological Monitoring and Assessment: Using Multimetric Indexes Effectively. EPA 235-R97-001. University of Washington, Seattle, WA; 1997. 149 p.

[48] Myers N, Mittermeier R, Mittermeier C, Da Fonseca G, Kent J. Biodiversity hotspots for conservation priorities. Nature 2000; 403: 853–858. DOI: 10.1038/35002501.

[49] García N, Cuttelod A. Biodiversity loss in the Mediterranean: causes and conservation proposals - Pérdida de biodiversidad en el Mediterráneo: causas y propuestas de conservación. Memorias Real Sociedad Española de Historia Natural 2013; 2: 41–54.

[50] Gasith A, Resh V. Streams in Mediterranean climate regions: abiotic influences and biotic responses to predictable seasonal events. Annual Review of Ecology, Evolution, and Systematics 1999; 30: 51–81. DOI: 10.1146/annurev.ecolsys.30.1.51.

[51] Myers N. Biodiversity Hotspot Revisited. Bioscience 2003; 53: 916–917. DOI: 10.1641/0006-3568(2003)053[0916:BHR]2.0.CO;2.

[52] Aguayo M, Pauchard A, Azocar G, Parra O. Land use change in the south central Chile at the end of the 20th century. Understanding the spatio-temporal dynamics of the landscape - Cambio del uso del suelo en el centro sur de Chile a fines del siglo XX. Entendiendo la dinámica espacial y temporal del paisaje. Revista Chilena de Historia Natural. 2009; 82: 361–374. DOI: 10.4067/S0716-078X2009000300004.
[53] Cuevas J, Huertas J, Leiva C, Paulino L, Dörner J, Arumi J. Nutrient retention in a microcatchment with low levels of anthropogenic pollution. Bosque 2014; 35: 75–88. DOI: 10.4067/S0717-92002014000100008.

[54] Figueroa R, Ruiz VH, Encina-Montoya F, Palma A. Simplification in the use of macroinvertebrates in the evaluation of water quality in fluvial systems - Simplificación en el uso de macroinvertebrados en la evaluación de la calidad de las aguas en sistemas fluviales. Interciencia 2005; 30: 770–774.

[55] EPA. U.S. Environmental Protection Agency. Policy on the USE of Biological Assessments and Criteria in the water quality program. Office of Science and Technology, Washington, D.C.; 1991. 19 p.

[56] EEA Report. Surface water quality monitoring: 4.2 Biological assessments of river quality [Internet]. 2016. Available from: http://www.eea.europa.eu/publications/92-9167-001-4/page021.html [accessed: 2016-04-04].

[57] Habit E, Victoriano P, Campos H. Trophic ecology and reproductive aspects of Trichomycterus areolatus (Pisces, Trichomycteridae) in irrigation canal environments - Ecología trófica y aspectos reproductivos de Trichomycterus areolatus (Pisces, Trychomyceteridae) en ambientes loticos artificiales. Revista Biología Tropical. 2005; 53: 195–210. DOI: 10.15517/rbt.v53i1-2.14416.