

Response of the fish community to human-induced changes in the Biobío River in Chile

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SUMMARY

1. The Biobío River basin of south-central Chile exhibits the greatest species richness of all rivers in Chile, where it is one of the most important rivers for human use. Use for provision of drinking water, irrigation, sewage effluents, hydropower generation and industry has increased dramatically during the last decade. To help understand the effects of human activities on the Biobío River, we document recent changes in the fish community.

2. In this study, current patterns of distribution and abundance of fishes were compared with the expected longitudinal pattern, and to historical data from studies conducted before the rapid development of the last decade. Fish distribution, biomass, abundance and diversity were studied at eight sampling stations in the middle and lower zones of the river in both high and low flow seasons.

3. Contrary to the pattern observed in less impacted river systems, species richness (S), diversity (H') and abundance [calculated catch per unit effort (CPUE)] all tended to decrease downstream from the uppermost sampling locations. Mean S decreased from 7.9 to 5.4, mean H' decreased from 0.7 to 0.4, and mean CPUE decreased from 111 to 43 from hyporhithral to potamal locations.

4. Comparison with previous records indicates loss or reduction in distribution of native species, and a concurrent expansion in distribution and abundance of tolerant introduced species (e.g. *Gambusia holbrooki*, and *Cyprinus carpio*) over the last 10–15 years. These comparisons suggest a large-scale and long-term effect of recent human impacts on the river.

Keywords: alien species, Chile, fish community, long-term effects, zonation pattern

Introduction

The freshwater fishes of Chile are heterogeneously distributed with respect to latitude, with three areas of freshwater ichthyofaunal endemism in the Chilean Province (Central, South-Central, Southern; Dyer, 2000). Among these areas, the South-Central area

(Mataquito to the Tolten River) exhibits the highest level of endemism and species richness. The Biobío River basin is located in the middle of the South-Central area of endemism, and exhibits the greatest species richness of all river basins in Chile (Vila, Fuentes & Contreras, 1999), with 17 native and four introduced species (Campos *et al.*, 1993a,b; Ruiz & Berra, 1994). Native species found in the Biobío River basin are a unique assemblage, with 65% of them endemic to the Chilean Province (11 species) and 80% considered vulnerable (seven species) or endangered (seven species; Campos *et al.*, 1998).

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In addition to the biological uniqueness of the Biobío River, it is also one of the most important rivers in Chile for human use. The river and its tributaries are integral to multiple human activities ranging from the provision of drinking water and irrigation to hydroelectric power generation and industrial uses. Use of this fluvial system has increased dramatically during the last decade, which has been characterised by the construction of five new hydropower plants (Pangué, Ralco, Mampil, Peuchen and Rucúe), ongoing and additional irrigation diversions, development of two new forest products industries (i.e. pulp mills, one with primary effluent treatment and the other with secondary treatments), and new sewage treatment plants (Parra *et al.*, 2004). This rapid increase in human uses would change the Biobío classification from 'moderately affected' by flow regulation and channel fragmentation, to 'strongly affected' (Nilsson *et al.*, 2005).

This increase in human activities presents potentially significant threats to the unique ichthyofauna of the Biobío basin. Major threats to conservation of native fishes occur in all three physical zones of the river. In the upper part of the river, conservation concerns include the interruption of movement patterns by hydropower dams and the potentially negative effects of introduced salmonids. In the middle and lower regions of the river, the major threat is pollution from sewage and industrial effluents. In particular, the segment of the Biobío River located between the confluences of the Duqueco and the Laja Rivers (Fig. 1) receives sewage from the city of Los Angeles, for which a sewage treatment plant with secondary treatment of activated sludge has been in operation only since 2003, and receives effluent from three pulp mill plants with a combined production >1 million tons year⁻¹ (Parra *et al.*, 2004). The lower segment of the Biobío River, downstream of the confluence of the Laja River to the city of Concepción, receives sewage from Santa Juana and the Concepción-Talcahuano-Chiguayante area, where a sewage treatment plant with secondary treatment has been in operation since 2003. These effluents create a pattern of decreasing water quality (higher levels of faecal coliforms, nutrients, COD and hydrocarbons) from upper to lower regions of the river (Parra *et al.*, 2004). Understanding how human impacts on the Biobío River alter natural patterns of abundance and distribution of native fishes through time is important for the conservation of this unique fauna.

Typically, distribution and abundance of native fishes in Chilean rivers follow a pattern of longitudinal zonation from upper to lower areas (Campos, 1985). Because of the proximity of the Andean mountain range to the coast, and the resulting high gradient mountainous terrain, all rivers in Chile flow from east to west, and they are characterised by three distinct physical zones from upstream to downstream, namely the rithral, transitional, and potamal zones. Species richness and abundance change from relatively low levels in the upstream, rithral zone to relatively high in the lower, potamal zone (Duarte *et al.*, 1971; Campos, 1985; Campos *et al.*, 1993b; Ruiz, 1996) (Fig. 2). This pattern is created by overlaying the distribution of widespread species that occur in all three zones, with those species of more restricted distribution that occur in the lower zones only (Habit, Victoriano & Rodríguez-Ruiz, 2003). Immediate and long-term effects of human activities on distribution and abundance of native fishes in Chilean rivers are poorly known.

To avoid loss of native populations and species in the Biobío River basin, it is critical that we understand the effects of human-induced changes. In this paper, we address effects of recent changes along the river on the distribution and abundance of native fishes in two ways. First, we compared current patterns of distribution and abundance of fishes with the expected pattern of longitudinal zonation observed in other Chilean river systems, including both high flow and low flow seasons (winter and summer) to allow comparison of effects by season. Second, we compared current patterns of distribution and abundance to data from studies conducted before the rapid development of the last decade (e.g. two large hydropower dams in the upper region, and two pulp production plants in the middle region). Understanding how fish populations have changed in recent years in the Biobío River is critical to conservation efforts in this area (Pegg & McClelland, 2004; Quist, Rahel & Hubert, 2005).

Methods

Species composition, distribution, biomass, abundance, and fish diversity of the Biobío River, were studied at eight sampling stations located in the middle and lower zones (Negrete to Concepción; Fig. 1). To determine temporal variability of the fish

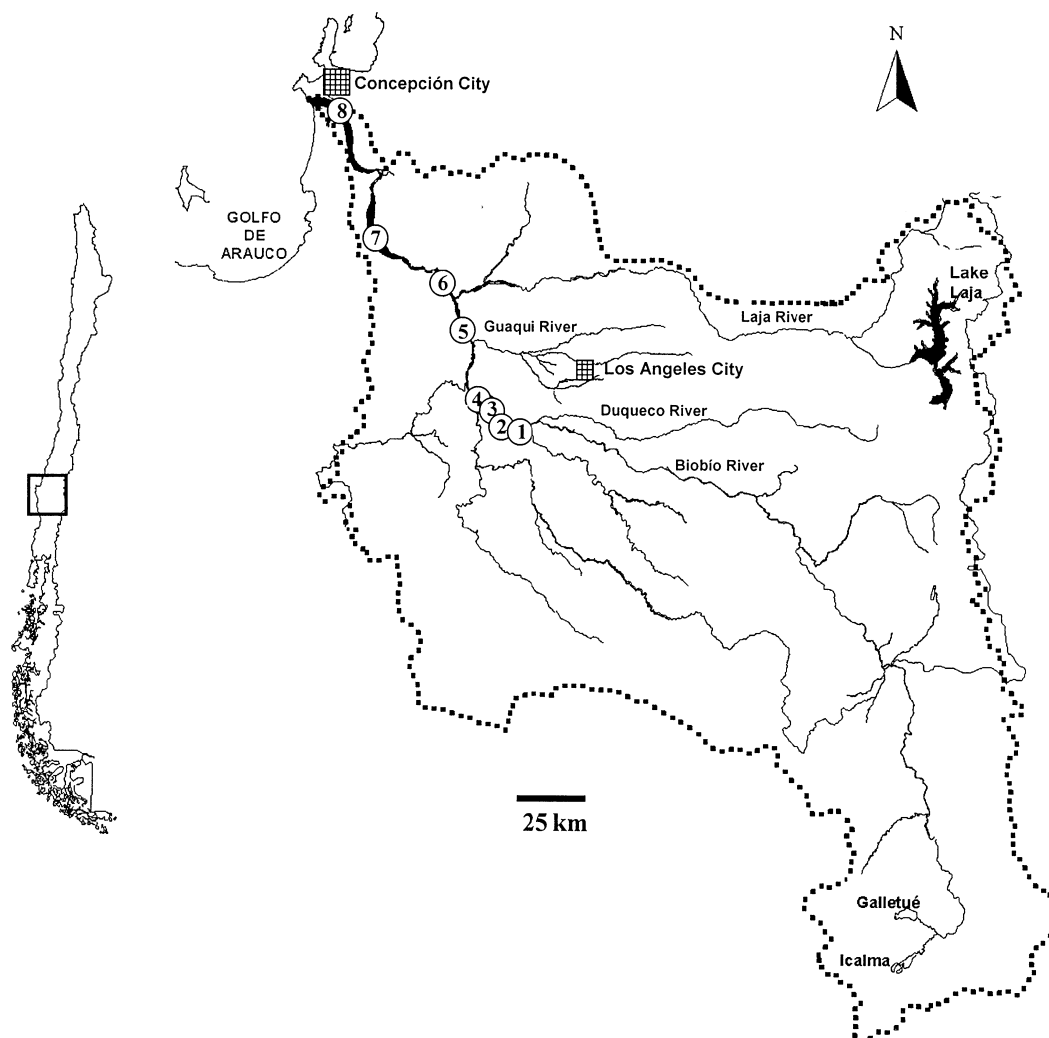


Fig. 1 Biobío River basin and location of sampling stations. Station 1: Negrete; station 2: Puente Coigüe; station 3: Santa Fé; station 4: Nacimiento; station 5: downstream the confluence of Vergara River; station 6: downstream the confluence of the Laja River; station 7: Santa Juana; station 8: Concepción. An outline of Chile, showing the position of the Biobío River, is shown as an inset.

community, two sampling surveys were carried out: one in the low flow season (March 2003, 132 m s^{-1} mean discharge in the mouth of the river), and the second during the high flow season (August 2003, 668 m s^{-1} mean discharge in the mouth of the river). Sampling was stratified by microhabitats (e.g. zones of varying substrates, water depths and current velocities within the river channel, generally on a scale of tens of meters in extent; Maddock, 1999). Fishing effort was concentrated in shallow and deep microhabitats with low current velocity, which includes the majority of fishes in fluvial systems (Arunachalam, 2000). At the macrohabitat scale (e.g. pool/riffle sequence within the bank full floodplain,

generally on a scale of hundreds of meters in extent; Maddock, 1999), stations 1, 2, 3, and 4 exhibit rithral-hyporithral characteristics (*sensu* Welcomme, 1985), whereas stations 5, 6, 7 and 8 exhibit potamal characteristics. At station 1 the channel is braided and exhibits a succession of pool and riffle habitats. In the other stations there is only one channel. For the purposes of parametric statistical comparison we combined stations into 4 zones according to both natural characteristics and human influences as follows. Zone 1, comprised of localities 1 and 2, was mainly rithral in character and had a low level of human influence. Zone 2, comprised of localities 3 and 4, was hyporithral and was strongly influenced

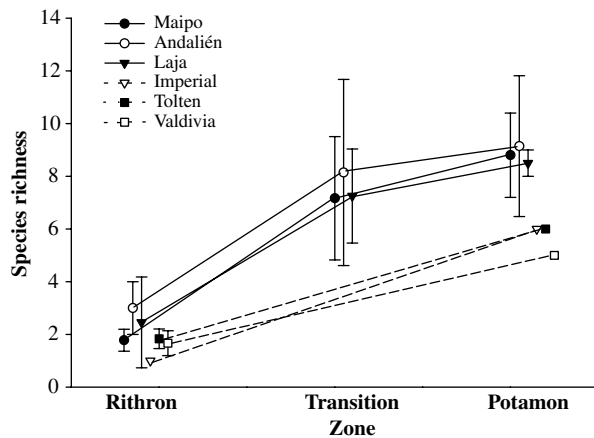


Fig. 2 Species richness (± 1 SE) trends in Chilean rivers ordered from north to south. Lines are drawn to indicate estimates from the same river and not to suggest values of species richness between sampling sites. Broken lines are used for rivers with no information from transitional zones. Data from: Maipo River, Duarte *et al.*, 1971; Andalién River, Ruiz, 1993; Laja River, Ruiz, 1996; Imperial, Tolten and Valdivia Rivers, Campos, 1985.

by pulp mill effluent. Zone 3, comprised of localities 5 and 6 was potamal, and was influenced by effluents from sewage treatment plants, pulp mills from upstream, and manipulation of the river channel. Zone 4, comprised of locations 7 and 8 was potamal, and was located considerably downstream from the other locations, near effluents from the Concepción city area.

Because the Biobío is a large river we used multiple gears to increase the capture efficiency in all microhabitats. In shallow microhabitats (<1 m) we used a backpack electroshocker (EFKO-Elektrofisch-Fanggeräte GmbH, Leutkirch, Germany; motor JLO gasoline, 50–400 V, DC, one anode) to sample fishes. Differences in sampling efficiency of gears may arise in different habitats and could confound estimates of abundance. However, three-pass depletion electrofishing in the same types of habitats in another river basin next to the Biobío River yielded similar sampling efficiency among all habitats (Andalién River, E. Habit & M. Belk, unpublished data). We think it is unlikely that variation in sampling efficiency could account for the pattern of species composition and abundance observed in this study. In deeper habitats (>1 m) we used monofilament gill nets of 55 and 130 mm mesh, 25 × 3 m, set overnight when open-water species are most active (Scasso, 1996). Pools in hyporithral and potamal zones had similar characteristics. None were >3 m in depth and all had low water

velocity and fine depositional substrates, so sampling efficiency of gill nets was unlikely to differ by zone. In deeper habitats of higher current velocity (>0.8 m s⁻¹) we used set lines of 30 m length set overnight, each with 15 hooks (1, 1.5 and 2 cm length) and baited with lepidopteran larvae and oligochaetes. No fish were captured on the set lines.

Captured fishes were identified to species, measured for total length and wet mass (0.01 mm, and 0.01 g precision) and almost all were returned alive to the location of capture. Individuals that died were preserved in ethanol (96%) and retained as voucher specimens. Species richness and diversity (Shannon–Wiener index, Magurran, 1988), and abundance and biomass by species were determined for each station and each sampling period. To standardise estimates of abundance and biomass we incorporated effort expended, by fishing gear, and calculated catch per unit effort (CPUE), and biomass per unit effort (BPUE) by sampling station (standardised per 50 m² for 20 min for the electrofishing, and 40 m² of gill nets for 10 h).

Differences in species richness, diversity, total CPUE and BPUE (from electrofishing) were compared by a two-way ANOVA with zone (group of sampling stations) and season (high or low flow) as main effects. Data for species richness and diversity were log-transformed, and means for main effects were compared with Bonferroni-adjusted *t*-tests (SAS Institute Inc., 1997). Data for CPUE and BPUE were transformed with a Box–Cox function (Box & Cox, 1964). Preliminary analysis revealed no significant interaction between zone and season in any test, so the interaction term was dropped from the model to provide a more powerful test of main effects. In addition, for the three most abundant native species we performed the same analysis as above (two-way ANOVA on CPUE and BPUE) for each species separately. As with to the analysis on all species combined, the interaction between zone and season was not significant for any species, and thus was not included in the final analyses. Planned linear contrasts were used in comparisons of CPUE and BPUE to determine if zone 1 differed from all other zones (SAS Institute Inc., 1997).

To understand how sampling locations were related and to visualise changes among locations by season (not grouped by zones) we used ordination methods based on pair-wise similarity matrices of fish abun-

dances. First, CPUE of fish was fourth root transformed; we then generated similarity matrices based on a Bray–Curtis similarity index (according to this index pair-wise similarities are higher if samples are more similar and lower if they are dissimilar; Bray & Curtis, 1957). To visualise the relationship of fish assemblages among river zones we used nonparametric multidimensional scaling (NMDS; Primer v.5, Clarke & Gorley, 2001) to produce two-dimensional plots of relationships among locations. We used the unweighted pair group method with arithmetic mean (UPGMA) to define the boundaries of similarity among locations in the NMDS plots. To determine which species were most important in generating the resulting patterns and which species contributed to the differences among zones (as identified above) we used the similarity percentage procedure (SIMPER; Primer v.5, Clarke & Gorley, 2001) on transformed variables.

Finally, to analyse the temporal variation in species composition and distribution in the Biobío River over the past decade we compared data gathered for this study with data from studies conducted at least 10 years earlier. Given the lack of standardised estimates of abundance from previous data for most study locations, we restricted our comparisons to measures based on presence/absence data, such as species composition and distribution except for station 4 where we were able to compare relative abundance of several species. Results of our 2003 data were compared with data described by Campos *et al.* (1993a,b) for the Negrete reach (equivalent to sampling station 1 of the present study) and for the Nacimiento to Concepción reach (stations 4–8 of the present study). Campos *et al.* (1993a,b) include data from March, September and December 1990, and January and April 1992. For the Santa Fé reach (equivalent to sampling stations 2 and 3 of the present study), we compared data from this study to data reported by Campos (1991) from samples taken in November 1989, and April, August and November 1990. In addition to the studies by Campos *et al.* (1993a,b), for comparison to stations 7 and 8 of our work we used data provided by Ruiz & Berra (1994) from sampling carried out during October 1988, February 1989 and December 1989. Only station 6 from the present study did not have any comparable historical data. To compare differences in relative abundance by number of fishes at station 4 between

current and historic samples we used chi-squared test on a 2×2 contingency table (relative abundance of native and non-native open-water fishes crossed with previous data and 2003 sampling periods; Ramsey & Schafer, 2002).

Results

A total of 17 species (13 native, four introduced) was captured during sampling in 2003. Common or widespread native species (present at least in four sampling locations, and total CPUE > 30) were *Trichomycterus areolatus* (Valenciennes), *Percichthys trucha* (Valenciennes), *Percilia irwini* Eigenmann, *Bullockia maldonadoi* (Eigenmann), *Cheirodon galusdae* Eigenmann, and *Galaxias maculatus* (Jenyns). Less common native species were *Percichthys melanops* Girard, *Basilichthys australis* Eigenmann, *Diplomystes nahuelbutensis* Arratia, *Geotria australis* Gray, *Mordacia lapicida* Gray, *Nematogenys inermis* (Guichenot), and *Mugil cephalus* Linnaeus. Introduced species were *Gambusia holbrooki* Girard, *Cyprinus carpio* Linnaeus, *Salmo trutta* Linnaeus, and *Oncorhynchus mykiss* (Walbaum). *Nematogenys inermis*, and *C. carpio* were captured only during low flows, whereas *S. trutta* was found only during high flows. *Bullockia maldonadoi* and *P. melanops* were found exclusively in the hyporithral area (locations 1–4), whereas *M. cephalus*, *N. inermis*, *S. trutta* and *O. mykiss* were found only in the potamal zone. Species richness (S) and diversity (H') varied significantly among zones but not by season (zone 1: S = 7.8 ± 2.1, H' = 0.6 ± 0.1; zone 2: S = 8.0 ± 2.0, H' = 0.8 ± 0.1; zone 3: S = 3.3 ± 1.0, H' = 0.3 ± 0.2; zone 4: S = 7.8 ± 2.1, H' = 0.4 ± 0.2; Table 1). Species richness was lower in zone 3 compared with all other zones ($P < 0.05$), and diversity was higher in zones 1 and 2 compared with zone 3, whereas diversity in zone 4 was intermediate ($P < 0.05$). Lower diversity in zone 3 is a result of both lower species richness and inequality in abundance among species. *Percichthys trucha* in station 5 and *P. irwini* in station 6 exhibited relatively high abundance compared with other species. Lower diversity in zone 4 is primarily because of the dominance of *P. irwini* in station 7 and the presence of at least three rare species in the reach (*P. trucha*, *G. australis*, and *N. inermis*).

Total CPUE (all species combined) differed by zone, but not by season (Table 1). Zone 1 and 2 CPUE did not differ ($P = 0.47$), but CPUE in zone 1 was higher

Table 1 Analysis of variance table for diversity, species richness, combined CPUE (catch per unit effort), combined BPUE (biomass per unit effort), and CPUE and BPUE of the three most abundant species. Main effects are zone (group of sampling stations) and season (high or low flow). Interactions were not significant for any of the tests, so they were not included in the final analysis. Significant results are given in bold.

Parameter	Source	d.f. (num,den)	F-value	P-value
Diversity	Zone	3,14	5.54	0.016
	Season	1,14	0.58	0.464
Species richness	Zone	3,14	8.35	0.004
	Season	1,14	0.04	0.853
CPUE (all species)	Zone	3,114	3.78	0.012
	Season	1114	0.00	0.980
BPUE (all species)	Zone	3,114	1.80	0.150
	Season	1,114	1.08	0.301
CPUE <i>T. areolatus</i>	Zone	3, 10	4.69	0.027
	Season	1, 10	2.13	0.174
BPUE <i>T. areolatus</i>	Zone	3, 10	4.69	0.027
	Season	1, 10	6.87	0.025
CPUE <i>P. trucha</i>	Zone	3, 10	5.21	0.020
	Season	1, 10	5.32	0.043
BPUE <i>P. trucha</i>	Zone	3, 10	5.58	0.016
	Season	1, 10	7.52	0.020
CPUE <i>P. irwini</i>	Zone	3, 10	1.29	0.331
	Season	1, 10	0.11	0.746
BPUE <i>P. irwini</i>	Zone	3, 10	0.85	0.496
	Season	1, 10	0.79	0.395

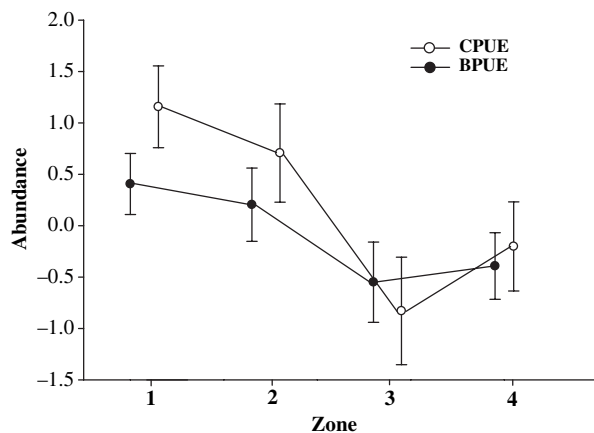


Fig. 3 Mean (adjusted least squares) CPUE and BPUE (± 2 SE) for all species combined by river zone in the Biobío River basin.

than zones 3 and 4 (both $P < 0.02$; Fig. 3). Total BPUE (all species combined) did not differ by season, or by zone (Table 1). For *T. areolatus*, both CPUE and BPUE differed by zone, and BPUE differed by season (Table 1). Zone 1 had higher BPUE and CPUE for *T. areolatus* than all other zones, and the low flow

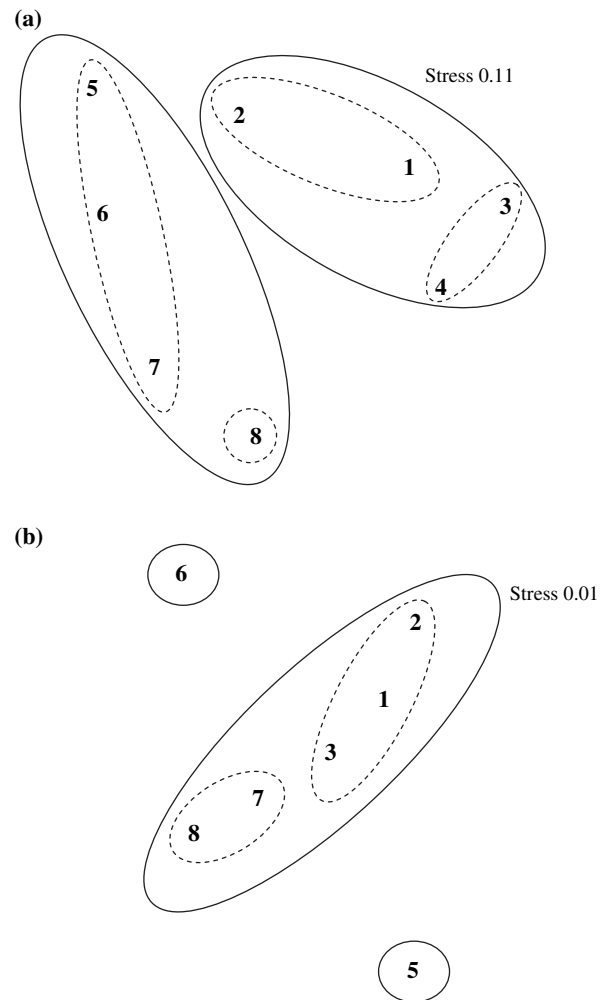


Fig. 4 Relationship of sampling locations based on relative abundance of fishes in the Biobío River. (a) Nonparametric multidimensional scaling diagram during the low flow season of the Bray–Curtis similarity matrix. Boundaries at Bray–Curtis similarity levels of 50% (solid) and 55% (dashed). (b) Nonparametric multidimensional scaling diagram during the high flow season of the same similarity matrix. Boundaries at Bray–Curtis similarity levels of 45% (solid) and 55% (dashed).

season had higher BPUE than the high flow season. Similarly, for *P. trucha*, both CPUE and BPUE differed by zone and season (Table 1). Both CPUE and BPUE of *P. trucha* were higher in zone 1 compared with zones 3 and 4 (CPUE and BPUE of zone 1 did not differ from zone 2), and CPUE and BPUE were higher during the low flow season. In contrast, for *P. irwini*, neither BPUE nor CPUE differed by zone or season (Table 1).

Non-metric multidimensional scaling of CPUE of fishes by sampling stations reveals two major groups during the low flow season (Fig. 4a). The four hypor-

ithral locations form the first group (comprising zones 1 and 2). The other group is composed of the four potamal locations (zones 3 and 4), and it is connected to the previous group at about 50% similarity. Among hyporithral localities there are two discernable groups (localities 1–2, and 3–4), and among potamal localities there are three groups (localities 5, 6–7, and 8). Mean similarity between locations 1 and 2 was 63%, and was determined mostly by abundance of *T. areolatus* (27%) and *P. trucha* (25%). Mean similarity between locations 3 and 4 was 56%, and was determined mostly by *B. maldonadoi* (26%) and *G. australis* (23%). Mean similarity among potamal locations 5, and 6–7 was 61%, and was determined mainly by *P. irwini* (46%).

A different spatial grouping of localities emerged from the NMDS for the high flow season (Fig. 4b). The two main groups are similar to those observed from the low flow data, with one group comprised of hyporithral locations 1, 2, and 3 (data were unavailable from station 4 during high flows), and the other main group comprised of potamal locations 7 and 8 (connected at about 50% similarity). However, stations 5 and 6 represent two independent groups connected to the others at 38% and 21% similarity, respectively. Mean similarity among rithral locations (1, 2 and 3) was 67% and was determined by *P. irwini* (25%) and *T. areolatus* (19%). Mean similarity between potamal locations (7 and 8) was 63% and was determined by *P. irwini* (32%) and *G. maculatus* (26%).

For both seasons combined, the CPUE of eight species contributed over 90% of the differences observed among the four *a priori* identified zones (Table 2). Zone 3 (localities 5 and 6) showed the

greatest differences in relation to the other three zones. Differences resulted mainly from lower abundance of *T. areolatus* and *B. maldonadoi* in relation to zones 1 and 2, and the diminished abundance of *P. irwini*, *G. holbrooki*, and *G. maculatus* in relation to zone 4 (Table 2).

Comparisons of distribution and abundance from this study to those reported from previous studies reveals several large changes in the fish community of the Biobío River over the last decade. First, *Odontheistes mauleanum*, was previously recorded in both hyporithral and potamal zones of the Biobío River (Campos *et al.*, 1993b; Ruiz & Berra, 1994), but in the current study it was not found at any of the sampling stations and appears to have become extremely rare or extinct. Second, previous studies recorded *O. mykiss*, *S. trutta*, and *N. inermis* in the hyporithral zone (locations 1–4 present study; Campos, 1991; Campos *et al.*, 1993b; Ruiz & Berra, 1994). None of these three species were recorded above station 7 in the current study, indicating a reduction in upstream distribution along nearly 70 km of river. Third, *P. melanops* and *B. maldonadoi* were previously recorded in the potamal zone (locations 7 and 8, Ruiz & Berra, 1994) but neither was found below station 4 in the current study, indicating a reduction of at least 53 km in downstream distribution. Fourth, the distribution of introduced *G. holbrooki* has increased upstream from near Nacimiento (station 4, Campos, 1991) to sampling station 1, about 16 km upstream. Finally, comparison of the composition and abundance of open-water species of the Nacimiento reach (station 4) between current and previous studies revealed a dramatic shift from a fish assemblage dominated by three native species (90% combined

Table 2 Species contributions to differences (dissimilarity) among zones based on SIMPER analyses for CPUE including both sampling seasons (high and low flow)

Species	Mean CPUE				Mean dissimilarity and contribution (%)					
	Zone 1 (1–2)	Zone 2 (3–4)	Zone 3 (5–6)	Zone 4 (7–8)	1 versus 2 (40.9%)	1 versus 3 (64.8%)	1 versus 4 (55.5%)	2 versus 3 (67.8%)	2 versus 4 (51.9%)	3 versus 4 (62.6%)
<i>T. areolatus</i>	56.1	3.5	1.5	6.5	5.1 (12)	10.5 (16)	7.0 (13)	4.2 (6)	3.9 (7)	7.4 (12)
<i>P. irwini</i>	36.1	12.5	5.8	39.8	4.8 (12)	7.4 (11)	3.7 (7)	7.0 (10)	6.5 (12)	10.4 (17)
<i>P. trucha</i>	18.8	10.9	2.5	3.1	2.8 (7)	8.9 (14)	7.2 (13)	8.0 (12)	6.3 (12)	6.7 (11)
<i>B. maldonadoi</i>	30.2	4.2	0	0	2.8 (7)	12.6 (19)	9.9 (18)	9.9 (15)	7.6 (15)	–
<i>G. maculatus</i>	4.5	8.4	0.6	11.6	4.2 (10)	4.6 (7)	5.2 (9)	7.8 (11)	6.0 (12)	9.8 (16)
<i>G. holbrooki</i>	0.2	3.5	0	14.2	4.4 (11)	–	4.8 (9)	9.0 (14)	4.3 (8)	9.6 (15)
<i>D. nahuelbutaensis</i>	4.9	0	0.1	0	4.7 (11)	6.4 (10)	5.5 (10)	–	–	–
<i>C. galusdae</i>	2.1	5.1	0	1.2	3.4 (8)	5.0 (8)	3.0 (5)	6.1 (9)	4.1 (8)	7.1 (11)

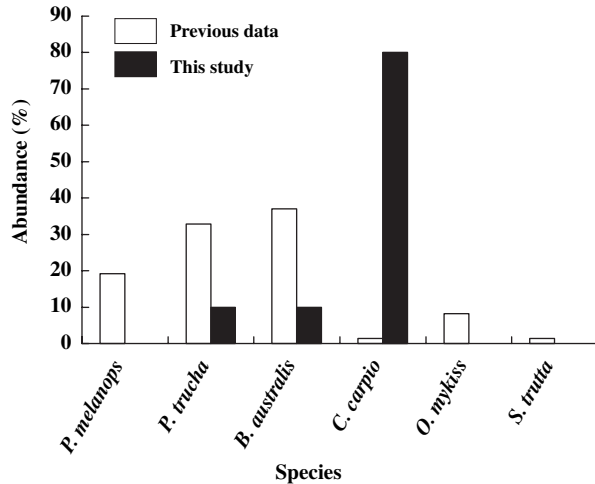


Fig. 5 Comparison of the relative abundance of open-water species in the Nacimiento zone (station 4) in 2003 (black bars) and previous data (white bars).

relative abundance by number) and a small proportion of introduced salmonids, to a fish assemblage dominated by *C. carpio* (80% relative abundance by number; $\chi^2 = 95.99$, $P < 0.01$; Fig. 5). Overall, comparison with previous records indicates loss or reduction in range of native species and environmentally sensitive introduced species (e.g. salmonids), and a concurrent expansion in range and abundance of environmentally tolerant introduced species (e.g. *G. holbrooki*, and *C. carpio*) in the middle to lower reaches of the Biobío River.

Discussion

Fish communities in riverine systems typically follow a pattern of increasing species richness, diversity and abundance from upstream to downstream (Welcomme, 1985; Bayley & Li, 1994; Granado, 2000). Chilean rivers, in particular, exhibit a fairly homogenous pattern of river zonation comprised of upstream, rithral areas with low species diversity, followed by intermediate, transitional areas with comparatively higher species diversity, and finally, downstream, potamal areas that exhibit the highest diversity and abundance of fishes. Historically, fish communities of the Biobío River probably followed such a pattern (Campos *et al.*, 1993a; Ruiz & Berra, 1994). However, the current pattern of species richness, diversity, and abundance of fishes contrasts sharply with the typical pattern. Species richness,

diversity, and abundance (measured as CPUE) were lower in the lower areas in this study compared with the upper area. This pattern is unlikely to result from sampling variation, because the same sampling gears were used in studies shown in Fig. 2, resulting in the typical increase in richness pattern downstream. Fisheries can impact species distribution and abundance over relatively short time scales (Welcomme, 1985). But, in this section of the Biobío River there are no commercial fisheries and recreational fishing is unimportant because larger salmonids are rare and native fishes are small and of little interest. Rather, the pattern found in the Biobío River suggests cumulative temporal and spatial effects of habitat loss or environmental degradation in the lower zone (Scrimgeour & Chambers, 2000; Wolter *et al.*, 2000).

Fluvial systems are dynamic and changes in fish species composition and distribution occur on a seasonal and annual basis (Ostrand & Wilde, 2002; Pegg & McClelland, 2004). Populations may increase or decrease through time, and fluctuations may be correlated among similar species (Rahel, Lyons & Cochran, 1984). However, in the absence of environmental change, we would not expect large-scale and long-term directional changes in species composition or distributional patterns (Quist *et al.*, 2005). The distributional and abundance changes between historical and current fish communities in the Biobío River are of a larger scale and longer term compared with the seasonal variation observed in the current study, variation reported in less impacted Chilean rivers (Duarte *et al.*, 1971; Campos, 1985; Ruiz, 1993, 1996) and annual variation in other rivers of the South-Central area of endemism (E. Habit & M. Belk, unpublished data).

Loss and reduction in distribution of native species noted in the lower study areas are consistent with an effect of declining water quality and loss of habitat resulting from human activities (Patton, Rahel & Hubert, 1998; Scrimgeour & Chambers, 2000; Wolter *et al.*, 2000), especially in the last decade. Water quality of the middle to lower Biobío River over the last decade has exhibited a consistent pattern of variation between upper and lower sampling stations – water quality is higher at upstream stations compared with downstream stations (Parra *et al.*, 2004). In addition, in a study of three different fish species in the Biobío River comparing enzymatic activity above and below

effluent areas, Orrego *et al.* (2005) found a clear increase in the enzyme ethoxyresorufin-O-deethylase (EROD) activity in fish liver in areas affected by effluents, which is used as a biomarker for water quality studies (Whyte *et al.*, 2000). Their results suggest a potential physiological mechanism that could account for decreased abundances observed in highly polluted segments in our study. Furthermore, the only species that have expanded their distribution in the Biobío River in the last decade are broadly tolerant introduced species typical of degraded systems (e.g. *G. holbrooki* and *C. carpio*; Prochelle & Campos, 1985; Rincon *et al.*, 2002; Parra *et al.*, 2003). Increased abundance and distribution of species like *G. holbrooki* are good indicators of habitat degradation (Kennard *et al.*, 2005). However, this change in species composition and distribution has not been accompanied by an overall change in temperature. Mean temperatures have remained relatively unchanged during the period covered by this comparison (Parra *et al.*, 2004). Rather, this change follows the pattern of homogenisation noted in aquatic systems worldwide as a consequence of habitat loss and degradation and introduction of invasive species (Rahel, 2002).

The general trend of decline in the fish community reflects several species-specific conservation threats. In particular, three endangered species are found in this section of the Biobío River, namely *N. inermis*, *D. nahuelbutaensis* and *P. melanops*, and changes to their status represent important conservation issues. *Nematogenys inermis* has been described as a typical inhabitant of potamal areas (Arratia, 1983), however, it has been captured in both rithral and potamal habitats of the Andalién River Basin (Ruiz, 1993). In the Biobío River, Campos *et al.* (1993a) recorded its presence in the Negrete hyporithral zone, and Ruiz & Berra (1994) recorded it in the upper Biobío River. Based on these accounts, we expected to find *N. inermis* in the middle river sections in the current study. However, it appears that *N. inermis* may be extinct in the middle zone of the Biobío River possibly as a result of environmental changes over the last decade (e.g. water quality; Parra *et al.*, 2004). *Diplomystes nahuelbutaensis* has been described in the upper and middle zones of the Biobío River (Galletué Lake, and Biobío River from Callaqui to Nacimiento; Campos *et al.*, 1993b), and we found it in the

hyporithral zone (stations 1 and 2) and downstream of the confluence of the Laja River (station 5). This suggests a continuous distribution from the headwaters of the Biobío and Laja River through to their confluence. However, the fact that *D. nahuelbutaensis* was not found in stations 3 and 4 (directly influenced by industrial discharges), and given its sensitivity to environmental changes (Vila, Contreras & Fuentes, 1996; Habit, 2005), its distribution may have become fragmented by degradation of habitat in the middle Biobío River. *Percichthys melanops* was reported by Campos *et al.* (1993a,b) as an abundant species in the Nacimiento reach (over 10% of the total abundance), but in recent sampling we found only two individuals: one in the Nacimiento reach (station 4) and the other in the Puente Coigüe (station 2). The biology of this species is poorly known, but a large-scale decline as noted here, may indicate high sensitivity to a decline in water or habitat quality.

Conservation implications of this study are clear. Unless changes are made in water quality and consequent habitat suitability, native species are likely to continue to decline, with several species in danger of extinction (Campos *et al.*, 1998), and the river may become increasingly dominated by introduced species. Recent efforts to improve water quality have had an immediate effect. New sewage treatment plants (operational in late 2003) and stricter requirements for effluent content of new pulp mills, as a consequence of recent implementation of legislation, have led to better water quality at some of our lower sampling stations. We applaud these efforts and expect that as this trend continues, decline of native species will lessen. However, recovery of abundant populations of native species may also depend on reduction of highly tolerant introduced species, such as *C. carpio* and *G. holbrooki*. Because both species are environmentally tolerant, and have been shown to have negative impacts on native species in many locations (Koehn, Brumley & Gehrke, 2000; Rincon *et al.*, 2002; Mills, Rader & Belk, 2004) gains in water quality are likely to favour introduced as well as native species, making it difficult to recover a native fish community. Although degradation of water quality and habitat seems to have allowed the range and population expansion of tolerant introduced species, improvement of water quality alone will probably not reverse the trend (Kennard *et al.*, 2005).

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References

- Arratia G. (1983) Preferencias de hábitat de peces siluriformes de aguas continentales de Chile (Fam. Diplomystidae y Trichomycteridae). *Studies on Neotropical Fauna and Environment*, **18**, 217–237.
- Arunachalam M. (2000) Assemblage structure of stream fishes in the Western Ghats (India). *Hydrobiologia*, **430**, 1–31.
- Bayley P. & Li H. (1994) Riverine fishes. In *The River Handbook: Hydrological and Ecological Principles*. (Eds Calow P. & G. Petts), pp. 251–281. Vol I. Blackwell Science, Boston, MA.
- Box G.E. & Cox D.R. (1964) An analysis of transformations. *Journal of the Royal Statistical Society. Series B*, **26**, 211–252.
- Bray J.R. & Curtis J.T. (1957) An ordination of the upland forest of the southern Wisconsin. *Ecological Monographs*, **27**, 325–349.
- Campos H. (1985) Distribution of the fishes in the Andean rivers in the South of Chile. *Archives Hydrobiology*, **104**, 169–191.
- Campos H. (1991) Peces. In: *Estado Actual del sistema acuático y predicción de impacto ambiental de la planta de Celulosa de Industrial y Forestal Santa Fé*. (Eds Parra O. & H. Campos), pp. 106–121. Informe Final Fase Final, Universidad de Concepción, Chile.
- Campos H., Ruiz V.H., Gavilán J.F. & Alay F. (1993a) Comunidad íctica de la hoya hidrográfica del río Biobío. In: *Evaluación de la calidad del agua y ecología del sistema limnético y fluvial del río Biobío*. (Eds Faranda F. & O. Parra), pp. 249–278. *Monografías Científicas EULA*, Vol. 12. Universidad De Concepción, Chile.
- Campos H., Gavilán J.F., Alay F. & Ruiz V.H. (1993b) Peces del Río Biobío. Serie. *Publicaciones de Divulgación EULA*, **5**, 1–100.
- Campos H., Dazarola G., Dyer B. et al. (1998) Categorías de Conservación de peces nativos de aguas continentales de Chile. *Boletín del Museo Nacional de Historia Natural*, **47**, 101–122.
- Clarke KR & Gorley RN. (2001) *PRIMER v5: User manual/Tutorial*. PRIMER-E, Plymouth, U.K.
- Duarte W., Feito R., Jara C., Moreno C. & Orellana A. (1971) Ictiofauna del sistema hidrográfico del río Maipo. *Boletín Museo Nacional Historia Natural Chile*, **32**, 227–268.
- Dyer B. (2000) Systematic review and biogeography of the freshwater fishes of Chile. *Estudios Oceanológicos (Chile)*, **19**, 77–98.
- Granado C. (2000) *Ecología de comunidades el paradigma de los peces de agua dulce*. Universidad de Sevilla Secretariado de Publicaciones, Sevilla, Spain.
- Habit E. (2005) Aspectos de la biología y hábitat de un pez endémico de Chile en peligro de extinción (*Diplomystes nahuelbutaensis* Arratia, 1987). *Interciencia*, **30**, 8–11.
- Habit E, Victoriano P. & Rodriguez-Ruiz A. (2003) Variaciones espacio-temporales del ensamble de peces de un sistema fluvial de bajo orden del centro sur de Chile. *Revista Chilena de Historia Natural*, **76**, 3–14.
- Kennard M.J., Arthington A.H., Pusey B.J. & Harch B.D. (2005) Are alien fish a reliable indicator of river health? *Freshwater Biology*, **50**, 174–193.
- Koehn J., Brumley A. & Gehrke P. (2000) *Managing the Impacts of Carp*. Bureau of Rural Sciences, Canberra.
- Maddock I. (1999) The importance of physical habitat assessment for evaluating river health. *Freshwater Biology*, **41**, 373–391.
- Magurran A. (1988) *Ecological Diversity and Its Measurement*. Princeton University, New Jersey.
- Mills M.D., Rader R.B. & Belk M.C. (2004) Complex interactions between native and invasive fish: the simultaneous effects of multiple negative interactions. *Oecologia*, **141**, 713–721.
- Nilsson C., Reidy C.A., Dynesius M & Revenga C. (2005) Fragmentation and flow regulation of the World's large river systems. *Science*, **308**, 405–408.
- Orrego R., Jiménez B., Bordajandi L.R., Gavilán J.F., Inzunza B., Abad E., González M.J., Rivera J. & Barra R. (2005) EROD induction and PCD/F levels in fish liver from the Biobío river in Chile. *Chemosphere*, **60**, 829–835.
- Ostrand K. & Wilde G. (2002) Seasonal and spatial variation in prairie stream-fish assemblage. *Ecology of Freshwater Fish*, **11**, 137–149.
- Parra O., Valdovinos C., Habit E. & Figueroa R. (2004) *Programa de monitoreo de la calidad del agua del sistema río Biobío*. Informe Técnico, Centro de Ciencias Ambientales Eula-Chile, Universidad de Concepcion, Chile.
- Parra O., Valdovinos C., Urrutia R., Cisternas M., Habit E. & Mardones M. (2003) Caracterización y tendencias tróficas de cinco lagos costeros de Chile central. *Limnetica*, (1–2), 51–83.
- Patton T., Rahel F. & Hubert W. (1998) Using historical data to assess changes in Wyoming's fish fauna. *Conservation Biology*, **12**, 1120–1128.

- Pegg M. & McClelland M.A. (2004) Spatial and temporal patterns in fish communities along the Illinois River. *Ecology of Freshwater Fish*, **13**, 125–135.
- Prochelle O. & Campos H. (1985) The biology of the introduced carp *Cyprinus carpio* L., in the river Cayumapu, Valdivia, Chile. *Studies on Neotropical Fauna and Environment*, **20**, 65–82.
- Quist M.C., Rahel F.J. & Hubert W.A. (2005) Hierarchical faunal filters: an approach to assessing effects of habitat and nonnative species on native fishes. *Ecology of Freshwater Fish*, **14**, 24–39.
- Rahel F.J. (2002) Homogenization of Freshwater Faunas. *Annual Review of Ecology and Systematics*, **33**, 291–315.
- Rahel F.J., Lyons J.D. & Cochran P.A. (1984) Stochastic or deterministic regulation of assemblage structure? It may depend on how the assemblage is defined. *American Naturalist*, **124**, 583–589.
- Ramsey F.L. & Schafer D.W. (2002) *The Statistical Sleuth: a Course in Methods of Data Analysis*, 2nd edn. Duxbury Press, Pacific Grove, California.
- Rincon P.A., Correas A.M., Morcillo F., Risueño P. & Lobón-Cervia J. (2002) Interaction between the introduced eastern mosquitofish and two autochthonous Spanish toothcarps. *Journal Fish Biology*, **61**, 1560–1585.
- Ruiz V.H. (1993) Ictiofauna del río Andalién (Concepción, Chile). *Gayana Zoología*, **57**, 109–278.
- Ruiz V.H. (1996) Ictiofauna del río Laja (VIII Región, Chile): una evaluación preliminar. *Boletín Sociedad Biología de Concepción*, **67**, 15–21.
- Ruiz V.H. & Berra T. (1994) Fishes of the high Biobío river of south-central Chile with notes on diet and speculations on the origin of the ichthyofauna. *Ichthyology Exploration Freshwaters*, **5**, 5–18.
- SAS Institute Inc. (1997) *SAS/STAT Software: Changes and Enhancements Through Release 6.12*. SAS Institute, Cary, NC, U.S.A.
- Scasso F. (1996) Productividad íctica en Lagos de diferente Estado de Trofico: recomendaciones de conservación para la pesca deportiva. Post Graduate Thesis, Centro EULA-Chile, Universidad de Concepción, Concepción, Chile.
- Scrimgeour G. & Chambers P. (2000) Cumulative effects of pulp mill and municipal effluents on epilithic biomass and nutrient limitation in a large northern river ecosystem. *Canadian Journal Fisheries Aquatic Science*, **57**, 1342–1354.
- Vila I., Contreras M. & Fuentes L. (1996) Reproducción de *Diplomystes nahuelbutaensis* Arratia 1987 (Pises: Diplomystidae). *Gayana Oceanología*, **4**, 129–137.
- Vila I., Fuentes L. & Contreras M. (1999) Peces Límnicos de Chile. *Boletín Museo Historia Natural, Chile*, **48**, 61–75.
- Welcomme R.L. (1985) River fisheries. *FAO Fisheries Technical Paper*, **262**, 1–318.
- Whyte J.J., Jung R.E., Schmitt C.J. & Tillitt D.E. (2000) Ethoxyresorufin-O-deethylase (EROD) activity in fish as a biomarker of chemical exposure. *Critical Reviews in Toxicology*, **30**, 347–570.
- Wolter C., Minow J., Vilcinskas A. & Grosch U. (2000) Long-term effects of human influence on fish community structure and fisheries in Berlin waters: an urban water system. *Fisheries Management and Ecology*, **7**, 97–104.

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