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DOWNSTREAM ENVIRONMENTAL EFFECTS OF DAM OPERATIONS: CHANGES IN HABITAT QUALITY FOR NATIVE FISH SPECIES

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ABSTRACT

Hydropeaking dam operation and water extractions for irrigation have been broadly stated as alterations to natural flow regimes, which have also been noticed in the Biobío Watershed, in Central Chile, since 1996. In the Biobío River, most of native fish species are endemic and very little is known about them. Their ecological and social values have never been estimated, and there is lack of information about their habitat preferences. Furthermore, changes on fish habitat availability due to natural and/or man-made causes have not been evaluated. In this study, eight native fish species, in a representative reach of the Biobío River, were studied and their preferred habitats were surveyed and characterized. A hydrodynamic model was built and linked to the fish habitat simulation model CASiMiR. Fuzzy rules and fuzzy sets were developed for describing habitat preference of the native fish species. CASiMiR was then used to simulate how physical habitat conditions vary due to flow control (i.e. upstream dams). Results show how overall habitat quality, expressed as weighted usable area (WUA) and hydraulic habitat suitability (HHS), changes and fluctuates due to the dam operation and how the daily hydropeaking is influencing quantity, quality and location of different habitats. The study suggests that the analysed fish are highly susceptible to flow control, as dams are currently operated, and fish habitat improvement suggestions are proposed. Copyright © 2010 John Wiley & Sons, Ltd.

KEY WORDS: fish habitat modelling; CASiMiR; hydropeaking; native fish species; multi-use river; Biobío; Chile

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INTRODUCTION

Hydropeaking dam operation and water extractions for irrigation have been broadly stated as alterations to natural flow regimes (Ward and Stanford, 1983; Petts, 1984; Poof and Allan, 1997; Magilligan and Nislow, 2005; Jorde *et al*., 2008), i.e. changes in flow magnitude, frequency, duration, timing and rate of change (Junk *et al*., 1989). These alterations change physical and chemical habitat conditions for fish species, and therefore could decrease fish and other animals' abundance (Travnichek and Maceina, 1994; Lamouroux *et al*., 2006). They also could alter some ecosystem services like regulation of the trophic web (Rooney *et al*., 2005), 'mosquito' control (Howard *et al*., 2007), biodiversity (de Mérona *et al*., 2005), pollination (Knight *et al*., 2005) and biomass either for gaming or as a food source, provided directly or indirectly by water bodies.

The Biobío River basin, located in central Chile (Figure 1), with 24 371 km^2, is one of the most important centres for economical developments in the country (Parra, 1996; TWINBAS, 2007), and classified as one of the world's largest river systems strongly affected by fragmentation and change in flow regime (Nilsson *et al*., 2005). Pangue and Ralco dams (Table I), located on the upper Biobío River (Figure 1), are fully operational since 1996 and 2004, respectively. Together, they generate 1.15 GW, about 24% of the national hydroelectric power and 13% of the total national power (by July 2007, source: Comisión Nacional de Energía at www.cne.cl). The projected impacts of these dams, in their Environmental Impact Assessments (EIAs), were described only for the impounded area, while the Biobío River downstream of Pangue was not considered as an impacted region (Meier, 1995; Goodwin *et al*., 2006). Nevertheless, the operation of these dams and reservoirs has dramatically altered the hydrological regime downstream Pangue in two ways. First, the annual regime has been altered and flattened because of the annual storage capacity of Ralco, and second, strong daily fluctuations are overlapping the altered mean flows since both dams are used for hydropeaking, where the flow variability shifted from monthly to the daily level after the operation of Pangue dam.

Furthermore, 158.5 m^{3}s^{-1} are taken from the Biobío River (and tributaries) to irrigate 220 000 ha in and outside the watershed during the dry season, increasing the

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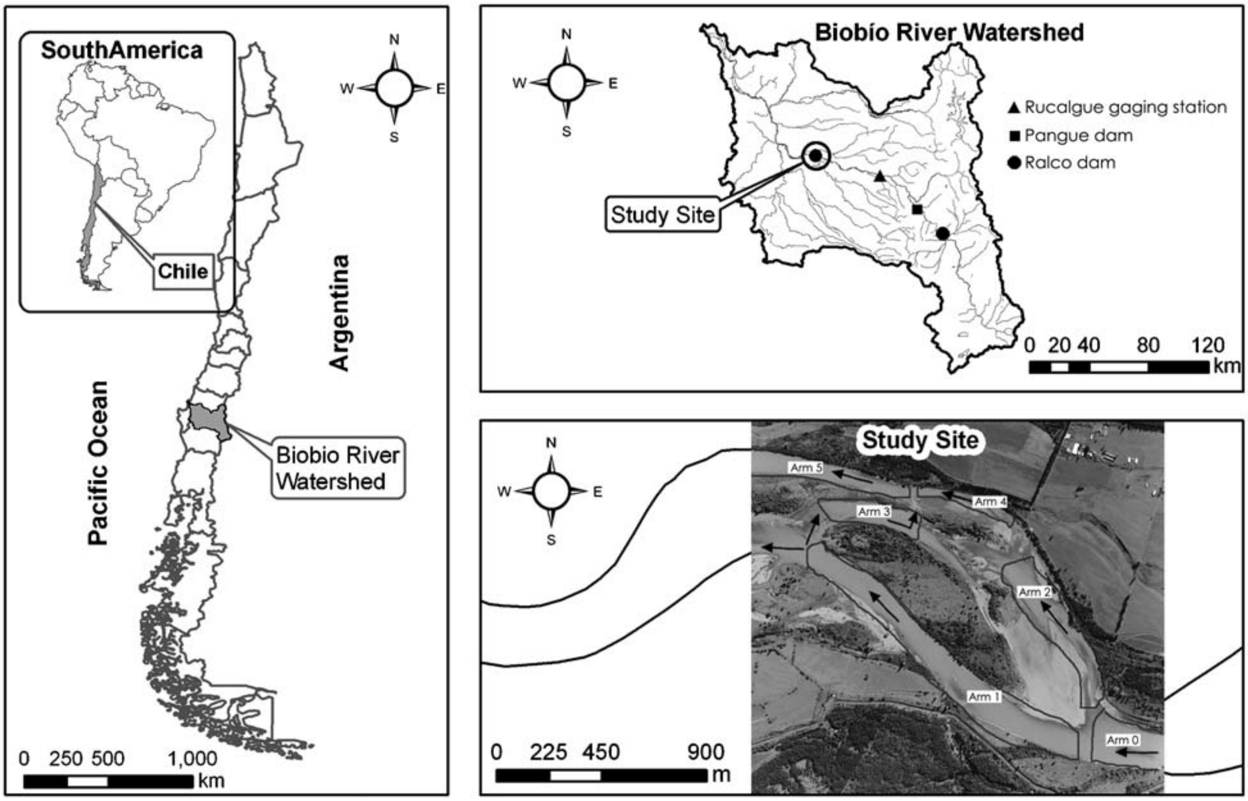

Figure 1. Study site, in the Biobío River, Chile. Left: Biobío watershed in the Central Chile. Top right: The study site within the Biobío Basin network. Bottom right: Study site with six arms (numbers) and flow directions (arrows)

seasonal flow alteration generated by the reservoir operation. These are major human disturbances in the middle and upper Biobío basin, while the effect of industries and city waste waters are the main concern in the middle and lower basin (Parra *et al*., 1993; Goodwin *et al*., 2006; Habit *et al*., 2006; Parra *et al*., 2009). Although the Biobío is the most studied river in Chile (Campos *et al*., 1993a; Parra, 1996; Parra and Meier, 2003; Habit *et al*., 2006), there is still a considerable knowledge gap regarding the basic biology, life stages and habitat selection of native freshwater fish fauna probably because most of these species are very small and not considered either economically or were socially relevant.

The effects of flow alteration are exacerbated in summer, when the water availability is lower and small to medium disturbances in flow availability may have important consequences to the environment and over delicate life stages (i.e. fry, juvenile, adults) (Lamouroux *et al*., 2006). Thus, this situation combined with the growing economic pressure on Chile's un-utilized hydropower potential puts a strong pressure on the endemic biota, and particularly on fish species, threatening their existence.

Table I. Pangue and Ralco dams characteristics

Attribute	Pangue dam	Ralco dam
Geographic location	37°54′42″ S 71°36′44″ W	37°59′40″ S 71°31′14″ W
Net head (m)	99	175
Reservoir area ($10^6\,m^2$)	52	347
Design discharge (m^3s^{-1})	500	368
Mean annual flow (m^3s^{-1})	296	270
Total reservoir volume ($10^6\,m^3$)	175	1222
Regulation volume ($10^6\,m^3$)	50	800
Installed capacity (MW)	467	690
Mean annual generation (GWh)	2345	3100

Source: www.endesa.cl

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Modelling tools that assess fish habitat availability and suitability to predict physical conditions under different scenarios have been applied for river management in the United States since early 1980s with the methodology developed by Bovee (1982). Habitat models assess habitat availability and suitability for freshwater species based on species selection of physical conditions such as flow velocity, water depth and substrate (Bovee, 1982). A rather new approach to evaluate habitat quality is CASiMiR (Computer Aided Simulation Model for Instream Flow Regulations) (Jorde, 1997; Schneider, 2001). This model has been developed since the early nineties as a simulation tool with components for benthic habitat quality (Jorde, 1997) and fish habitat quality (Schneider, 2001). CASiMiR has been widely used throughout Europe to address alterations of riverine habitats due to hydropower operation or other engineering structures like irrigation channels and diversions (Bloech *et al*., 2005; Mouton *et al*., 2007).

CASiMiR allows working with approximated or 'fuzzy' information (i.e. fuzzy rules and fuzzy sets), and has the significant advantage that expert knowledge, readily available from experienced fish biologists, can easily be transferred into preference data sets by setting up check-lists with possible combinations of relevant physical criteria and let experts define habitat quality by using natural language such as good, medium or low (Jorde *et al*., 2001a; Jorde *et al*., 2001b). This modelling tool has proven to perform better at the validation step than the method based on previous approaches like preference curves (Leclerc, 2005).

Thus, the goal of this work is to determine the habitat use of Chilean native freshwater fish in a reach of the Biobío River, and predict changes in habitat availability and suitability due to dam operation and water extractions during the summer season by developing fuzzy sets, fuzzy rules and using CASiMiR.

METHODS AND MATERIALS

Study site

The study was conducted in a 2 km long braided reach that represents well the hyporithron or middle zone of the Biobío River. It is located 98 km downstream Pangue dam (Figure 1). Mean summer flow, mean annual flow and bankfull flow for the study site are 160, 466 and 3880 m^3s^{-1}, respectively. Currently, the flow regime is controlled by hydropeaking operation (Pangue and Ralco dams), but it is also affected by water withdrawals for irrigation purposes. Peak floods or flow peaks (Junk *et al*., 1989) have been observed and estimated by the authors. During the summer time water surface elevation rises about 0.9 m in a period close to an hour, decreasing its level back to the original stage 4–12 h later.

Rucalhue gaging station (BNA code: 08317001–8, Dirección General de Aguas, www.dga.cl), located 42 km downstream Pangue (Figure 1), has registered mean daily data since 1937 and hourly readings since March 1999. In the pre dam scenario (before 1996), February's monthly flow was $142.8 \pm 40.5\,m^3s^{-1}$ (period 1970–1995); while, in the post dam scenario (after 1996) the mean monthly flow for February was $155.6 \pm 40.6\,m^3s^{-1}$ (period 1999–2008). This shows an 8.8% increase in mean monthly flow in the post dam scenario, and a similar flow variability for both scenarios. Moreover, in the post dam scenario, 30% of the days of February show a daily variability (standard deviation of hourly data) at least equal to the variability during the pre dam scenario at a monthly level.

Water quality is described as in natural conditions from the upper Biobío to 15 km downstream the study site (Parra *et al*., 2008), and water releases from reservoir do not produce any effect on the river's water temperature at the study area (Link *et al*., 2008).

The Biobío River has been identified as the Chilean River with the highest fish species richness with 17 (Campos *et al*., 1993a; Campos *et al*., 1993b; Ruiz and Berra, 1994) of 44 known native species. Nine of these native Chilean freshwater fish species have been described for this particular area (Campos *et al*., 1993b). Introduced species in this area are rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*) and common carp (*Cyprinus carpio*), but there is non quantitative reported effect of these species over native fish species (Dyer, 2000). Thus, the effects of changes in flow regime over native fish species during summer can be isolated from other disturbances at the study site.

Ralco dam has an annual flow regulation capacity with a storing volume equivalent to 7% of the mean annual discharge of the Biobio River (at Ralco). Pangue dam has a daily or weekly storing capacity (Table I). Both dams produce about 5450 GWh-year; they operate as hydropeaking power plants, i.e. increasing their production when demand and the price of electricity is higher (Vehanen *et al*., 2005). Three irrigation canals (Bío–Bío Norte, Bío–Bío Sur and Bío–Bío Negrete) are located between Rucalhue gaging station and the study site, which have consumptive water rights of 70 m^3s^{-1} (Dirección General de Aguas, www.dga.cl). There are not tributaries flowing into the Biobío River between Rucalhue gaging station and the study site.

Fish sampling

Five fish sampling techniques were used to sample 64 sites over three field trips done in May (fall), August (winter) and December (summer) 2004. (1) Sets of gill nets (80 m^2, 4.5 cm mesh) and (2) minnow traps (30 cm diameter and 45 cm long) were set at sites deeper than 1 m and water flow velocity lesser than 0.1 ms^{-1}, (3) hook lines were set close to

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water edges in riffle areas and checked early morning and before the sunset. (4) Electrofishing (Elektrofishfangerat motor JLO gasoline, 50–400 V, DC, one anode) was done at wadeable sites of about 15 m^2 ($3\,\text{m} \times 5\,\text{m}$) by doing three passes of 10 min each, walking from upstream to downstream with a blocking net set at the downstream end. (5) Underwater observation with a black and white monitor, connected to a video camera installed inside a sealed-transparent container was done from water edges and from boats at sites with slow water flows. Direct visual observations were complementary to the fishing techniques explained.

Mean flows during the campaigns were 90, 440 and 190 m^3s^{-1}, respectively. All sampling was performed under good weather conditions, in clear water with conductivity about 80 $\mu\text{S}\,\text{cm}^{-1}$, during steady flow conditions within a regime of daily hydropeaking fluctuations. Each sampling site was representative of one habitat type (or habitat patch), with homogeneous characteristics of flow velocity, water depth and substratum size.

The environmental variables, flow velocity, water depth and substratum size were measured at each site where fishing was done. Mean flow velocity and water depth were measured with a Pygmy current meter (model 625) at three points within each fishing site and then averaged to characterize it. Substratum size of each site was recorded within ranges based on the Wentworth scale (Wentworth, 1922) as clay and silt (<0.063 mm), organic matter, mud, sand (0.063–2 mm), fine gravel (2 mm–5 cm), gravel (5–10 cm), cobbles (10–20 cm), boulders (20–40 cm), big boulders and rocks (>40 cm) (Figure 3) and determined by eye or obtaining the mean diameter (d_m) with the Wolman method (Wolman, 1954) when needed. Occurrence of clay and silt, organic matter and mud were negligible in the study site and as fish respond to them in a similar way, these three categories were treated evenly (Figure 3) and referred as fine material for all practical applications.

Fish collected were first grouped by species and then sorted into adult or juvenile based on their size and other morphological characteristics explained in literature (Manríquez *et al*., 1988; Campos *et al*., 1993b; Vila *et al*., 1996; Ruiz and Marchant, 2004; Habit *et al*., 2005; Habit *et al*., 2006). Then, most of the individuals were returned to the sites where they were caught and a few were kept in alcohol for further studies.

Hydraulic modelling

Topographic and bathymetric surveys were carried out over the study reach during the summer of 2004 and 2005. A Leica differential GPS system 500 model SR530 with accuracy of ±2 cm in the vertical and ±1 cm in the horizontal was used to capture data on land and in wadeable areas. The GPS was coupled to an echosounder Lawrance (model X-16) in 2004 and to a depth sounder Innerspace Technology (model 455) in 2005 to survey the river bathymetry at deeper areas. A total of 149 cross sections were surveyed with a density of 3–5 cross sections per unit of river width, spaced at distances that captured the bathymetric features and allowed good representation of the channel bed.

Discharge measurements from Rucalhue gaging station were used to assess the study site flow conditions. The river flows through confine sections between Rucalhue and the study site, precluding lateral interaction with the floodplain and therefore having insignificant flow storage. Figure 2 shows the similarity on the water surface elevation wavelengths at both points. As there are not tributaries along this 42 km of river, the area factor and water extractions for irrigation are sufficient to describe the hydrograph at the interest site (Equation 1). Hourly flow data from 2005 and mean daily data from 1979 were obtained from *Dirección General de Aguas* (General Water Directive), and water surface elevation data at the upstream end of the study site was obtained from the *Asociación de Canalistas del Canal Bio-Bio* Negrete (Bio-Bio Negrete Irrigation Canal Association). Thus, Equation 1 shows the mass balance of water from Rucalhue gaging station to the study site as follows:

$$Q_{\text{Site}} = (Q_{\text{Rucalhue}} - Q_{\text{Base}}) + Q_{\text{Base}} \cdot \text{Area\_factor} - Q_{\text{e}} \quad (\text{m}^3\text{s}^{-1}) \tag{1}$$

where Q_{Site} is the discharge at site, Q_{Rucalhue} is the discharge recorded in Rucalhue gaging station, Q_{Base} is the base flow for February at Rucalhue, Area_factor is the rate between both the Study site and Rucalhue contributing area calculated as 1.089 and Q_{e} is the discharge extracted from the Biobío River for irrigation purposes.

For the pre-dam scenario (1979), Q_{Rucalhue} and Q_{Base} are the same since the river is under a natural base flow regime;

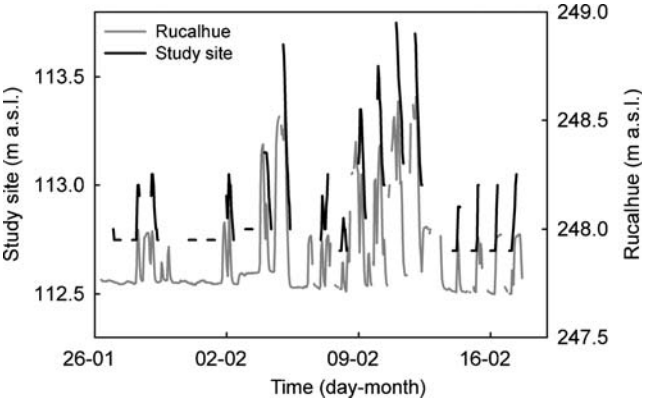

Figure 2. Water surface elevation at Rucalhue gaging station (in grey) and at the study site (in black). Rucalhue's data are delayed 8.15 h to compare both records vertically

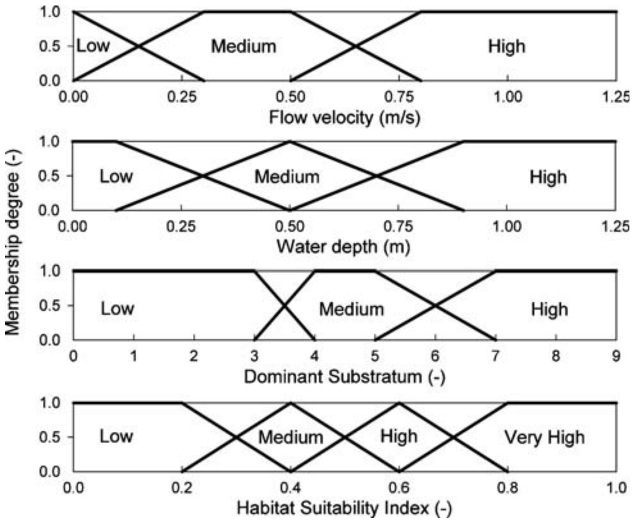

Figure 3. Membership functions for the input variables flow velocity, water depth and dominant substratum, and for the output variable habitat SI, for species occurring at the study reach. Substratum classes are: 0 = clay and silt (<0.063 mm), 1 = organic matter, 2 = mud, 3 = sand (0.063–2 mm), 4 = fine gravel (2 mm–5 cm), 5 = gravel (5–10 cm), 6 = cobbles (10–20 cm), 7 = boulders (20–40 cm), 8 = bigger boulders or rocks (>40 cm)

and for the post-dam scenario, Q_{Base} was estimated as the minimum flow occurring in February 2005.

The study reach was analysed hydraulically as six arms with strong hydraulic controls at their downstream ends (Figure 1). Flow measurements were performed during three different days in the arms 1, 2, 3 and 5 (Figure 1). Discharge from arms 1 and 2 indicate the upstream flow entering the study site, and individual measurements give the distribution of flow within the arms.

Hydraulic profiles were calibrated for each flow measurement with the water surface elevations recorded at the Bio-Bio Negrete canal intake and the water elevations at all cross sections surveyed at low flow. Two extra flows (2* and 3*) were interpolated linearly between flow 2 and 3 (Table II), and between 3 and 4 (Table II) to smoother the outcomes of habitat modelling. Another extra flow (1* in Table II) was modelled and calibrated with fewer sections to generate a lower boundary to be used for habitat modelling. Longitudinal water surface elevations profiles at site, associated to the flow measured (1, 2 and 3) and the flows interpolated (1*, 2* and 3*) were modelled with HEC-RAS (USACE, 2008). Then, flows at site were related with flows in Rucalhue gaging station with Equation 1, as shown in Table II.

Physical habitat modelling

The fish module of the habitat simulation model CASiMiR model was used to predict habitat availability and suitability for the native fish species. Introduced species were not considered because they are out of the scope of this study. In this study, the model was used to simulate pseudo two dimensional river hydraulics based on riverbed topography and roughness (XYZ coordinates plus substratum type), flow discharge and water surface elevation (either measured or modelled with HEC-RAS) at each cross section. In a second stage, habitat suitability was simulated based on hydraulic conditions, fuzzy sets and fuzzy rules (Jorde, 1997; Schneider, 2001).

For the habitat modelling, the values assigned to the input physical variables (i.e. water depth, flow velocity and dominant substratum size) are defined by fuzzy sets, instead of preference curves. Briefly, a fuzzy set is defined by its membership function that gives a certain degree of membership for a variable value. The mathematical function that defines each set describes the edge of individual membership functions, which overlap, so a value for a physical variable can belong to two or more sets, having a membership degree between zero and one. Fuzzy sets are named with linguistic expressions that are commonly used by fish experts for the description of habitat preferences: 'low', 'medium' or 'high' (Figure 3). Thus, a linguistic statement like 'the flow velocity is medium (or moderate) tending to high' would be processed as a velocity with a membership degree of 0.7 for medium and 0.3 for high. During the field campaigns, the expert biologist was asked about her perception regarding flow velocity, water depth

Table II. Flows recorded at Rucalhue gaging station and measured at the study site

Flow	Q_{Rucalhue} (m^3s^{-1})	Q_{Site} (m^3s^{-1})	Arm 1 (m^3s^{-1})	Arm 2 (m^3s^{-1})	Arm 3 (m^3s^{-1})	Arm 4 (m^3s^{-1})	Arm 5 (m^3s^{-1})
1*	90	50	50	0.4**	0.4**	8	8.4
2	196	154	148	6	6	26	32
2*	250	218	194	24	21	36	60
3	382	365	300	65	56	60	125
3*	520	504	436	74	63	82	156
4	685	650	579	84	71	106	190

*Flows modeled with HEC-RAS.
**Runoff from hyporheic flow (measured).

and substratum size as high, medium or low at 53 spots within the study site, meanwhile a different crew measured them to generate the information to develop the fuzzy sets shown in Figure 3 (for more details refer to García-Lancaster, 2006).

The fuzzy rules define the relation between input variables and habitat suitability for certain species/life stage. For instance, one rule (see Table V) for *Trichomycterus areolatus* would be: ‘If flow velocity is high, and water depth is high, and substratum size is high, then suitability index (SI) is low’. In the same way, all combinations have to be analysed based on literature reviews for each species life stage, data collected at the study site and the experience of the biologist in the river (Mouton *et al*., 2007), thus generating a total of 27 (3 variables with 3 linguistic expressions) rules per species life stage. Since the amount of fish data available for this study were limited and the development of fuzzy rules for the description of habitat suitability was based on expert knowledge and the limited database, a quantitative validation of the model could not be performed within this study.

The inference processor of CASiMiR calculates the degree of fulfillment (DOF) of each rule for a given combination of input values (e.g. flow velocity of 0.5 $\mathrm{ms^{-1}}$, water depth of 0.75 cm and substratum of gravel) for each cell of each river reach. This DOF indicates to what extent a rule has an impact on the resulting habitat suitability of the cell. Then, the fuzzy sets of the habitat suitability (lower set in Figure 3) are weighted with these DOF and combined to a final function describing the total DOF of all rules, as a crisp number representing the habitat SI, between 0 and 1 (Jorde *et al*., 2001a; Mouton *et al*., 2007).

Three outputs of CASiMiR were used to assess changes in fish habitat availability and suitability. First, habitat suitability maps for fish species at each arm were generated to visualize the areas within the river that provide certain physical conditions for fish under a steady flow situation. The habitat available is divided in ten suitability classes, which ranges from 0 to 1. Second, weighted usable area (WUA) (Bovee, 1982), is the area weighted summation of the available habitat in each cell, which shows the area available for each species as a function of the discharge and gives an integral view of the habitat quality over the reach. Third, hydraulic habitat suitability (HHS) (Jorde, 1997) is the equivalent of the WUA divided by the inundated area. It explains if overall habitat suitability changes because the velocity and depth patterns change or due to a change in inundated area, and represents the suitability or appropriateness of the physical variables for the species considered. One value of WUA and HHS is derived for each river arm at each discharge.

The WUAs for each arm were added up for each discharge considered (Table II) to obtain the WUA for the study reach. HHS for the study reach was obtained dividing the WUA for the study reach by the total inundated area of the reach for every discharge.

Habitat maps were generated for the native catfish *Trichomycterus areolatus* in all arms, as this is the most common species in this area of the Biobío River, and set over an aerial geo-referenced photograph to visualize the distribution of the good habitats. This was repeated for three discharges that represented the typical hydropeaking event that occurred on 18 February 2005 to assess the shift and change in habitat suitability under the current situation. WUA and HHS time series (Milhous *et al*., 1990) were derived from WUA and HHS curves coupled with the hydrograph at site for February 2005 (post dams) and the condition of summer 1979 (pre dams) to compare habitat quality at both scenarios. Values of WUA and HHS were calculated for each flow of both hydrographs (pre and post dams) and mean and standard deviation values were obtained for comparison.

RESULTS

Eight native fish species were caught and their habitats characterized over the surveys: (1) silverside: *Basilichthys australis* Eigenmann, 1927 (BA); (2) catfishes: *Bullockia maldonadoi* (Eigenmann, 1927) (BM); (3) *Trichomycterus areolatus* Valenciennes, 1848 (TA); (4) darters: *Percilia irwini* Eigenmann, 1927 (PI); (5) *Percichthys trucha* (Valenciennes, 1833) (PT); (6) galaxid: *Galaxias maculatus* (Jenyns, 1842) (GM); (7) lamprey: *Geotria australis* Gray, 1851 (GA); and (8) tetra: *Cheirodon galusdae* Eigenmann, 1927 (CG) (Table III). They were classified as ‘adult’ (a) or ‘juvenile’ (j), but *Geotria australis* was caught only in juvenile stages named ‘ammocoetes’ (c) and ‘macrophthalmia’ (m), as shown in Table IV. The gap of knowledge about *D. nahuelbutaensis*, and the very few individuals caught during two campaigns (May and August) did not allow considering this species on this work.

The outputs of habitat suitability maps for each species were contrasted with the fish data collected during the campaigns to refine fuzzy rules (and/or sets). Thus, the integration of expert knowledge and data collected supported the construction of the fuzzy rules for eight species in two life stages, as shown in Table V.

Habitat suitability maps were generated for each species at all arms of the river reach. As an example, Figure 4 shows the habitat suitability map for *T. areolatus* adult in arm 5 at low flow condition. There, a cascade (Montgomery and Buffington, 1997) in the upstream end and a riffle in the downstream end are hydraulic controls of this arm for low flows. Flow velocity, water depth and substratum size maps are shown next to the habitat suitability map. It can be seen

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Table III. Species collected at site with their habits and distribution along the Biobío River

Species name	Common name	Order	Habitat	Distribution
Basilichthys australis	Silverside	Atheriniformes	Mid-water	Hyporithron
Bullockia maldonadoi	Catfish	Siluriformes	Benthic	Hyporithron, potamon
Trichomycterus areolatus	Catfish	Siluriformes	Benthic	Rithron to potamon
Percilia irwini	Darter	Perciformes	Mid-water-Benthic	Rithron to potamon
Percichthys trucha	Darter	Perciformes	Mid-water	Rithron to potamon
Galaxias maculatus	Galaxid	Osmeriformes	Mid-water	Hyporithron, Potamon
Geotria australis	Lamprey	Petromyzontiformes	Benthic (juvenile)	Hyporithron, potamon
Cheirodon galusdae	Tetra	Characiformes	Mid-water-Benthic	Hyporithron, potamon

Table IV. Summary of the fish sampling

Species	BA a	BA j	BM a	BM j	TA a	TA j	PI a	PI j	PT a	PT j	GM j	GA m	GA c	CG a	CG j
N ind.	3	18	173	233	129	251	83	266	13	55	8	12	19	106	33
N sites	3	3	15	20	16	16	19	24	5	12	4	7	9	11	13

N ind.: Total number of individuals caught.
N sites: Total number of sites where the species was caught. Species with less than three individuals caught are not included in the table. Species names are abbreviated as: BA: *B. australis*, BM: *B. maldonadoi*, TA: *T. areolatus*, PI: *P. irwini*, PT: *P. trucha*, GM: *G. maculatus*, GA: *G. australis*, CG: *C. galusdae*. a: adult, j: juvenile, m: macrophthalmia and c: ammocoetes.

that the most suitable habitats for *T. areolatus* occur mainly in rather shallow and fast flowing waters over a coarse substrate, and close to the water edges, avoiding deep slow-flowing areas and sandy substrates, as stated in the fuzzy rules (Table V). Also, the other seven species' habitat preferences were well represented in the habitat maps in all arms.

The habitat maps for *T. areolatus* adult under the hydropeaking events occurring on 18 February 2005 are shown in Figure 5. Change in size and shift in the location of the good habitats is shown in 4 h time steps. All reaches show a reduction of the area with optimal conditions to mostly very bad conditions as flow increases. Also, the better places for fish occur at different locations under different discharges. Thus, location of good habitats for *T. areolatus* may move from a few meters (when good habitat remains at the river margins) to about 200 m (when good habitat disappears in some part of the reach) under this peak flow event, depending on the river arm observed.

WUA and HHS curves are almost parallel lines (Figure 6), which are expected to occur for species that use common habitats occurring close to water edges, especially when the streambed is wide and has a smooth slope across the river (Schneider and Jorde, 2003). It can also be noticed that, for the range of flows modelled (50–650 m^3s^{-1}), WUA and HHS curves do not present a rising limb and the values become smaller as the discharge increases.

WUA and HHS time series, during summer 1979 and February 2005 for *T. areolatus* adult and juvenile in the river arm 5, are shown in Figure 7 and summarized in Table VI. The variability of flows in the post dam scenario is evident, as well as the smooth decrease in flow over summer in the pre dam scenario, mostly driven by snowmelt with rare precipitation events. The values of WUA and HHS during 1979 are rather stable and higher than the mean values occurring during 2005. The ratio between the flow variability (i.e. standard deviation) in February 2005 and 1979 is 6.8.

For all species, Table VI shows that habitat availability and suitability is higher and more stable in 1979 than 2005. The ratio in habitat variability for each species over the whole study site is close to 4, and the highest value is 5.2 for *B. australis* juvenile. It can also be seen that variability in habitat availability and suitability are smaller than variability in water flows, which is also higher in 2005 than 1979.

DISCUSSION AND CONCLUSIONS

As seen in the field and in our results, the alterations of the natural flow regime during the summer season have been increased with the operation of Pangue and Ralco dams in the middle Biobío River. Irrigation water withdraws reduce the river flow, and for the range of flows modelled (50–650 m^3s^{-1} at site), these water reductions increase the fish habitat availability for the eight species occurring in this part of the Biobío. Flood pulses (Junk *et al*., 1989), on the other hand, are more frequent, with shorter lengths and larger

Table V. Fuzzy rules for the native fish species captured at site based on flow velocity (V), water depth (D) and substratum size (S)

Parameter			Suitability index for each species															
V	D	S	BA a	BA j	BM a	BM j	TA a	TA j	PI a	PI j	PT a	PT j	GM a	GM j	GA m	GA c	CG a	CG j
H	H	H	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
H	H	M	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
H	H	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
H	M	H	L	L	L	L	M	L	M	L	L	L	L	L	L	L	L	L
H	M	M	L	L	L	L	M	L	L	L	L	L	L	L	L	L	L	L
H	M	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
H	L	H	L	L	L	L	H	VH	L	L	L	L	L	L	L	L	L	L
H	L	M	L	L	M	M	L	M	L	L	L	L	L	L	L	L	M	L
H	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
M	H	H	L	M	L	L	L	L	M	L	L	M	L	L	L	L	L	L
M	H	M	L	M	L	L	L	L	M	L	L	M	L	L	L	L	L	L
M	H	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
M	M	H	L	M	M	H	VH	M	M	M	L	M	L	L	L	L	H	L
M	M	M	L	M	M	M	H	M	M	M	L	M	L	L	L	L	M	L
M	M	L	L	L	M	M	L	L	L	L	L	L	L	L	M	M	M	L
M	L	H	L	M	M	M	VH	VH	H	H	L	M	L	L	L	L	M	L
M	L	M	L	H	H	H	H	H	H	H	L	M	L	L	L	L	H	L
M	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
L	H	H	VH	H	M	M	L	L	M	M	VH	H	M	M	L	L	M	M
L	H	M	VH	M	H	H	L	L	M	M	VH	M	M	M	M	M	H	H
L	H	L	VH	L	M	M	L	L	L	L	VH	L	M	M	M	M	M	H
L	M	H	M	VH	H	H	M	M	VH	VH	M	H	M	M	L	L	H	M
L	M	M	M	H	VH	VH	M	H	VH	VH	M	H	H	H	M	M	VH	H
L	M	L	M	L	M	M	L	M	L	L	M	L	VH	VH	VH	VH	M	H
L	L	H	L	H	H	H	L	M	VH	H	L	VH	M	M	L	L	H	H
L	L	M	L	M	VH	VH	L	M	H	H	L	H	H	V	M	M	VH	H
L	L	L	L	L	M	M	L	L	L	L	L	L	VH	VH	VH	VH	M	VH
			2, 7		1, 2, 4		1, 2, 4		8		4, 3		5, 6		2, 4		4	

VH: Very High, H: High, M: Medium, L: Low. Species names are abbreviated as: BA: *B. australis*, BM: *B. maldonadoi*, TA: *T. areolatus*, PI: *P. irwini*, PT: *P. trucha*, GM: *G. maculatus*, GA: *G. australis*, CG: *C. galusdae*. a: adult, j: juvenile, m: macrophthalmia, and c: ammocoetes. 1: Arratia, 1983; 2: Campos *et al*., 1993b; 3: Ruiz *et al*., 1993a; 4: Ruiz, 1993b; 5: Vila *et al*., 1996; 6: Barriga *et al*., 2002; 7: Ruiz and Marchant, 2004; 8: Habit and Belk, 2007.

amplitudes than the natural flow conditions and thus, are less predictable for fish. Although, pre and post dams mean monthly flows are similar, fish do not live in 'averages', and the drastic hourly changes in flow are strong disturbances that the fish populations are not necessarily prepared for.

The three selected variables were chosen (i.e. water depth, flow velocity and substrate) because they are known to be the most important ones to explain physical habitat selection of native fish at small scales based on previous studies (García and Habit, 2007). These variables are referred to as the habitat template (Bornette *et al*., 1994) and are commonly used in physical habitat studies (Leclerc, 2005; Mouton *et al*., 2007). In addition to water depth, flow velocity and substratum, the existence of shelter provided by big boulders, logs, tree branches or submerged vegetation in shallow waters has been noticed to influence native fish species like *P. trucha* (Ruiz *et al*., 1993a) *C. galusdae*, *B. maldonadoi* (Ruiz, 1993b) and *T. areolatus* juvenile (Arratia, 1983), but it was not a consistent observation in the current study. Nevertheless, shelter might be also considered in future works as a physical variable to characterize native fish habitat availability.

The pre dam scenario, with its rather smooth flow variations during the summer season has stable conditions that provide the highest habitat availability for the native species (Table VI, Figure 7). This stability fits with the findings of species like *B. australis* (Campos *et al*., 1993a), *C. galusdae* (Ruiz and Marchant, 2004), *D. nahuelbutaensis* (Vila *et al*., 1996; Lundberg *et al*., 2004), *G. maculatus* (Barriga *et al*., 2002) and *G. australis* (Neira, 1984; Ruiz and Marchant, 2004) that spawn during late spring and early summer season in the Biobío River, increasing their possibilities of success. Incubation and rearing life stages would also be expected to have higher possibilities of survival under steady flow conditions (Welcomme, 1985; García de Jalón *et al*., 1993).

Larger values of WUA and HHS occur at lower flows, while smaller values of WUA and HHS occur at higher flows

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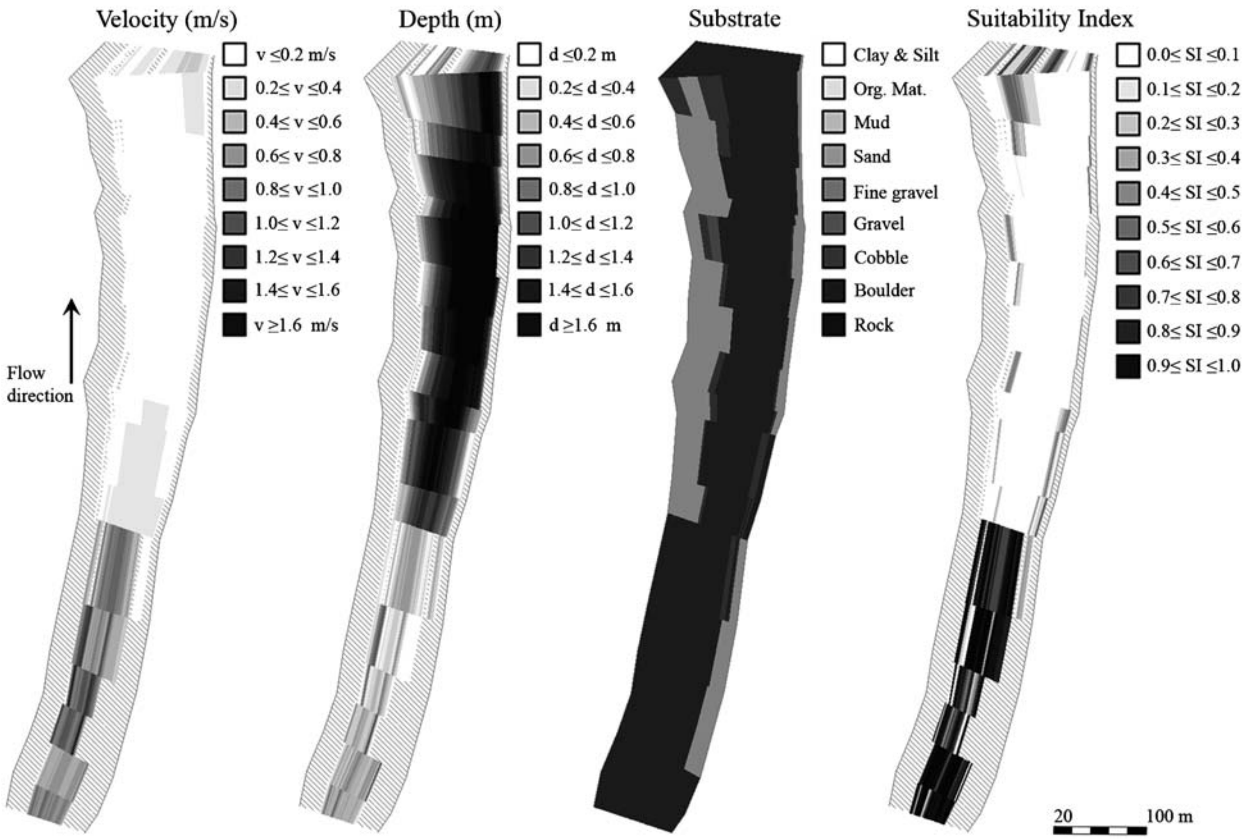

Figure 4. Flow velocity, water depth and substratum size combined with fuzzy sets and fuzzy rules produce the Habitat suitability map for *Trichomycterus areolatus* adult in arm 5 of the study site in the Biobío River. Higher Suitability Index (SI), in dark, indicates better physical conditions for this species. Borders with diagonal lines represent zones with no superficial runoff for this flow

(Figure 6), and the locations with best habitat suitability is different for each flow scenario (Figure 5). For instance, a discharge of 318 m^3s^{-1} (see Figure 5) provides a WUA of 45 000 m^2 (HHS = 0.08) to *T. areolatus*, while 91 m^3s^{-1} provides a WUA of 90 000 m^2 (HHS = 0.2). The higher flow, with less suitable habitat, provides half WUA of the lower flow, which provides more suitable habitat with a higher proportion of highly suitable area, i.e. more area with high SI (blue shades in Figure 5) (Mouton *et al*., 2007). Thus, as habitat availability is reduced, habitat use will be affected (Mäki-Petäys *et al*., 2002). During summer flood pulses, benthic fish species like *T. areolatus*, *G. australis* and *B. maldonadoi* are not likely to swim back and forth during a flood pulse to follow suitable habitat conditions due to their natural behaviour and the energy consumption that these movements imply. Therefore, flow variability can be understood as a connectivity problem from the fish physiology point of view. Instead, this fish may stay buried in rather fixed spots during these events even when the physical conditions, water depth and flow velocity, are not adequate for them. For instance, the shift and reduction of good habitats in Figure 5 means that *T. areolatus* should swim from 70 to 7000 times its length to get to another good spot in 4 h, or even pass across the deeper parts of the river to get to the other water edge. Then, after 4 h it would have to swim back close to the original good spots to remain in its preferable habitat conditions, increasing also the risk of predation during these movements.

Mid-water species (e.g. *P. irwini* and *P. trucha*) also have less suitable area available when flow increases, but it is easier for them to swim to a more suitable spot than for benthic ones. Nonetheless, good spots occurring in smaller areas not necessarily can support all fish that occur over much larger areas, and territoriality issues that have been observed in captivity of native fish can occur (Habit pers. com.), as well as food limited scenarios (Wei-wei *et al*., 2008). On the other hand, stranding can affect mid-water and benthic fish when water descends (Cushman, 1985; Wei-wei *et al*., 2008), especially when there are gently sloping shores or bars (Cushman, 1985), and has been noticed in amphibians and insects in the study site (personal observations).

Fish feeding behaviour is also affected by flood pulses and is not likely to be equivalent to natural conditions (Leclerc,

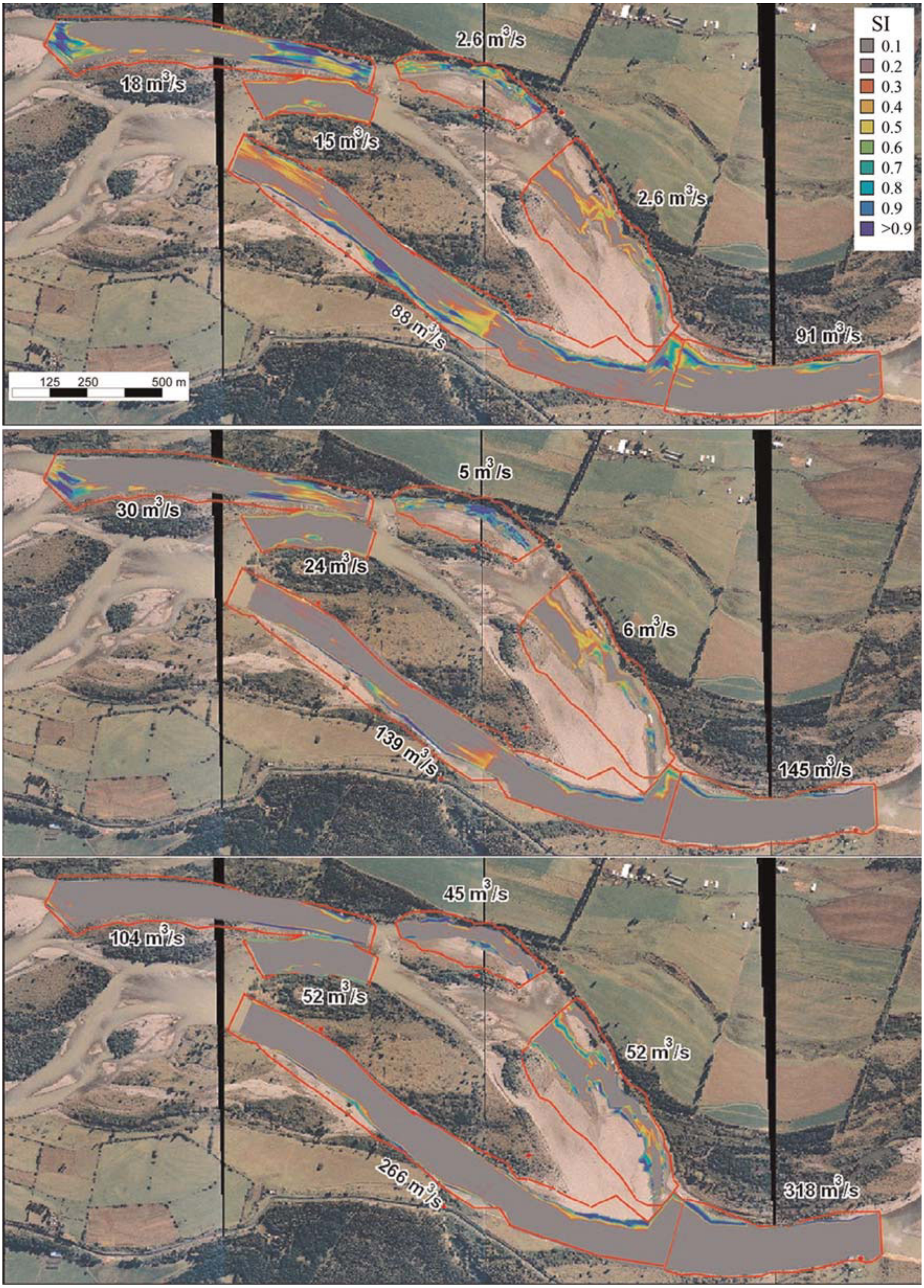

Figure 5. Sequence of habitat suitability maps for *Trichomycterus areolatus* adult under a hydropeaking event on 18 February 2005. Optimal conditions in blue and very bad conditions in grey. This figure is available in colour online at wileyonlinelibrary.com/journal/rra

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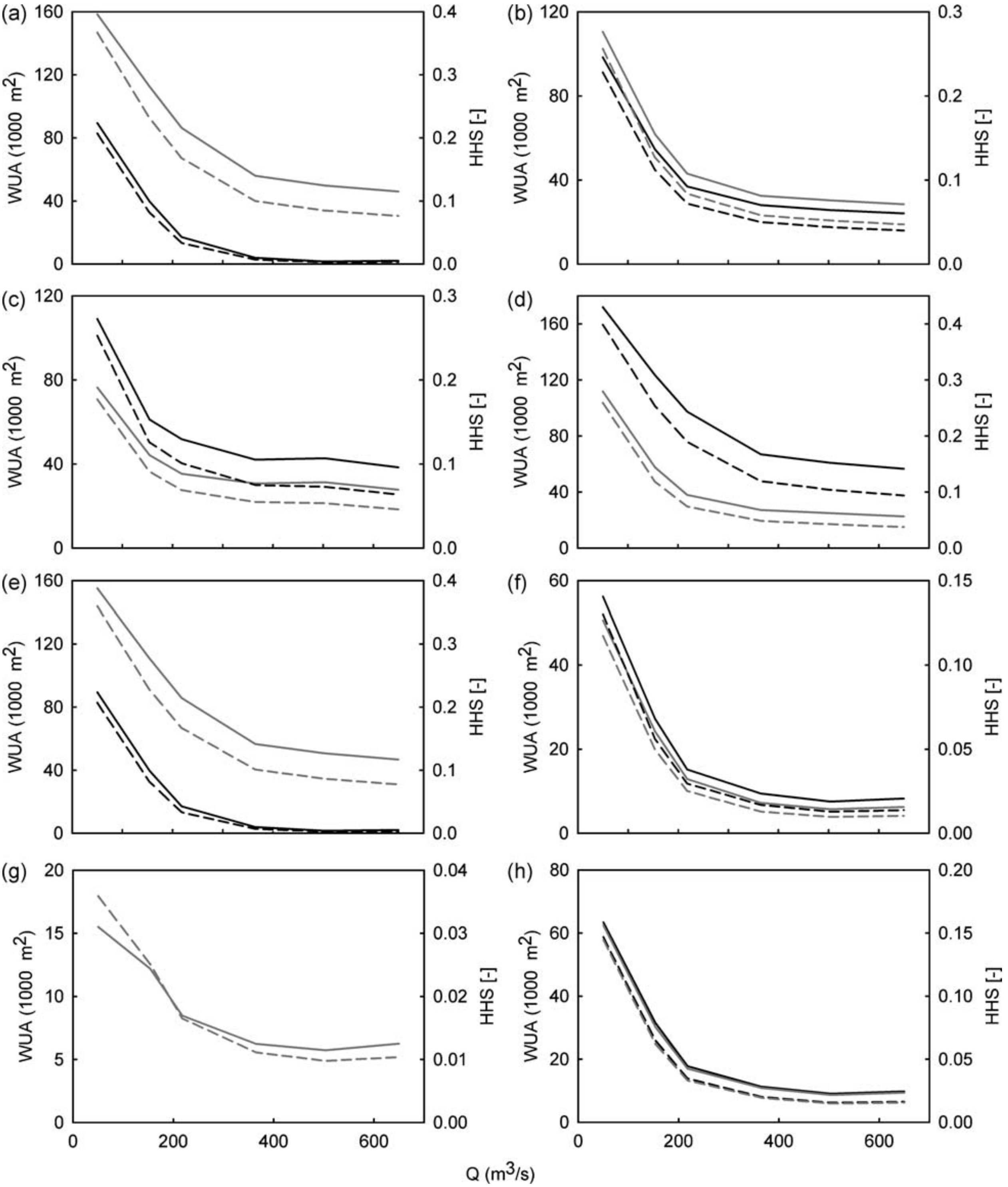

Figure 6. WUA (black lines) and HHS (grey lines) curves for eight fish species caught at the study site: (a) *B. australis*, (b) *B. maldonadoi*, (c) *T. areolatus*, (d) *P. irwini*, (e) *P. trucha*, (f) *G. maculatus*, (g) *G. australis*, (h) *C. galusdae*. Adult fish in solid lines, juvenile in segmented lines

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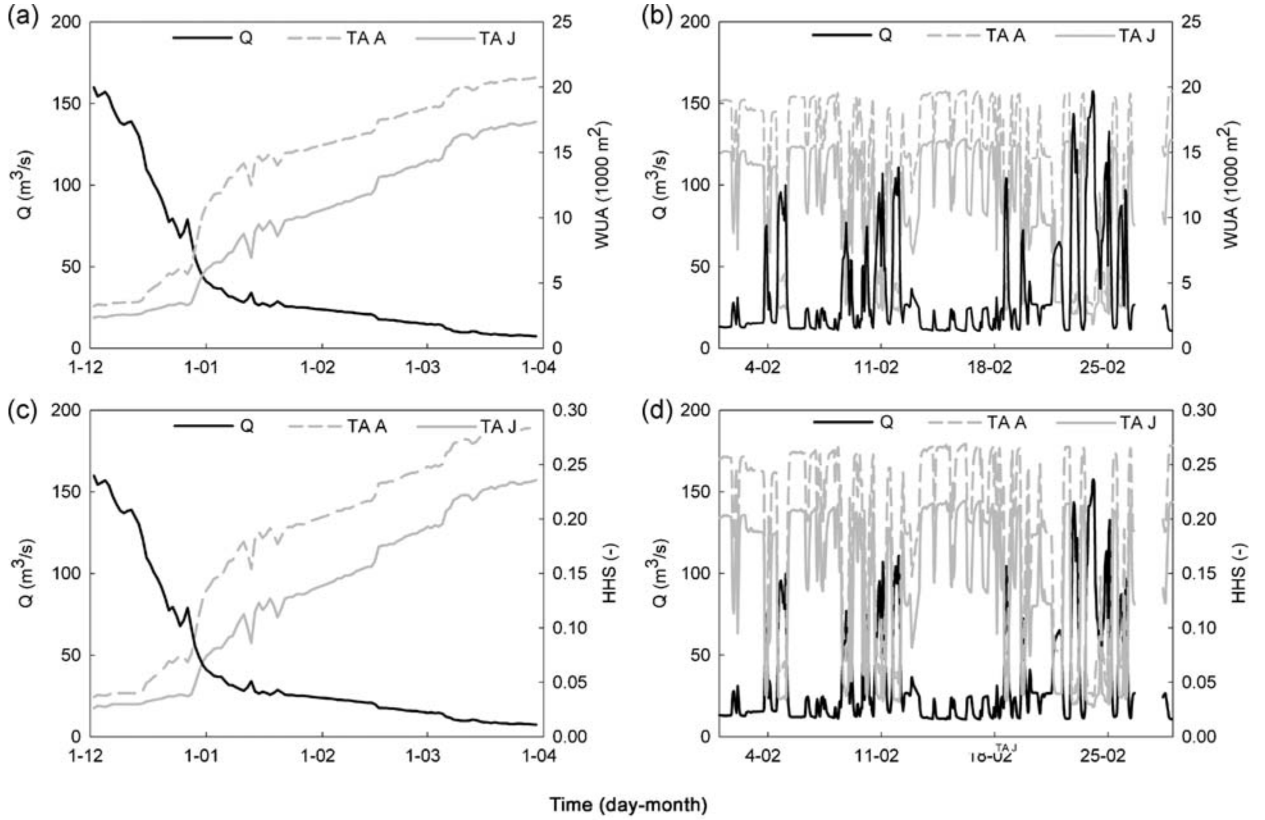

Figure 7. WUA and HHS time series for *Trichomycterus areolatus* adult (TA A, grey-segmented lines) and juvenile (TA J, grey-solid lines), and flows (Q, black-solid lines) in arm 5. (a): pre dam flow regime from December 1978 to March 1979; (b): post dam flow regime for February 2005

2005). Invertebrate drift occurs mainly during the night in natural systems (Waters, 1972) and during the rising limb of floods in altered systems (Lauters *et al*., 1996), in search of food and substratum, the higher chance of survival to predators or competitive interactions, to escape from unfavorable physical or chemical conditions and to colonize new habitats (Ciborowski, 1983; Smock, 1996). After drift, most insects had colonized new locations before the sunrise (in natural systems) or during the falling limb of the flood (in altered systems). Thus, flood pulses occurring during day or night can affect insects' drift and settling timing and duration, and consequently change food availability for fish feeding in the water column or in the benthos.

A more friendly design and operation of hydroelectric facilities have been encouraged in last decades in the legislation of developed countries with policies like the European Water Framework Directive in the European Union or the Clean Water Act in the United States. However, this is still a future issue for most developing countries, where hydroelectricity is still a short-term cost-effective source of energy at the expense of natural resources (Goodwin *et al*., 2006). Cushman (1985) proposed three major areas of management to minimize the impacts of hydropeaking production (rapidly varying flow): (i) structural changes with re-regulating dams, i.e. utilization of a small dam located downstream a big one (operating with peaking flows), to stabilize flows further downstream; (ii) Habitat modification, i.e. manipulating river section to increase habitat availability (although this may reduce habitat diversity) and (iii) operational changes, i.e. specify upper limits to the amount of variability of one or more characteristics of the flow released, like the change in discharge per unit time as a function of preexisting discharge or ramping. Ždankus & Sabas (2006) proposed to limit the water level fluctuation (specifically receding) to the maximum rate occurring under natural conditions (around $20\,\mathrm{cm\,h^{-1}}$), which may limit riverbed scouring and direct harmful effects over biota, Olson (1990, *in* Irvine *et al*., 2009) suggested a ramping rate lesser than $2.5\,\mathrm{cm\,h^{-1}}$ as adequate for biota, while Flodmark (2004, *in* Irvine *et al*., 2009) suggests that the ramping rate might be site specific and should be obtained for each river. Becker *et al.* (1981) suggested avoiding water descends during the night to reduce stranding chance, although Irvine *et al.* (2009) did

Table VI. Mean ± standard deviation of WUA and HHS, in February, over the study site for the eight species sampled. Flow in February 1979: $98.9 \pm 13.2\,\mathrm{m^3s^{-1}}$, in February 2005: $137 \pm 90.6\,\mathrm{m^3s^{-1}}$

Species	WUA [1000 m^2] A		WUA [1000 m^2] J		HHS [-] A		HHS [-] J	
	2005	1979	2005	1979	2005	1979	2005	1979
BA	55.5 ± 27.0	66.0 ± 6.3	127.3 ± 30.5	136.8 ± 5.8	0.12 ± 0.06	0.15 ± 0.02	0.27 ± 0.08	0.30 ± 0.02
BM	69.3 ± 22.4	77.8 ± 5.6	79.9 ± 24.1	87.6 ± 6.2	0.15 ± 0.06	0.17 ± 0.01	0.17 ± 0.06	0.20 ± 0.02
TA	79.5 ± 20.9	86.5 ± 6.1	57.3 ± 14.4	61.3 ± 4.1	0.17 ± 0.06	0.19 ± 0.02	0.12 ± 0.04	0.14 ± 0.01
PI	136.4 ± 30.6	149.1 ± 6.2	77.9 ± 26.2	86.4 ± 6.9	0.30 ± 0.08	0.33 ± 0.02	0.17 ± 0.07	0.19 ± 0.02
PT	55.5 ± 26.8	66.0 ± 6.3	125.2 ± 29.5	134.3 ± 5.6	0.12 ± 0.06	0.15 ± 0.02	0.27 ± 0.08	0.30 ± 0.02
GM	36.9 ± 14.9	42.6 ± 3.7	34.1 ± 13.0	38.2 ± 3.3	0.08 ± 0.04	0.10 ± 0.01	0.07 ± 0.03	0.09 ± 0.01
CG	42.0 ± 16.6	48.4 ± 4.0	41.8 ± 16.0	47.3 ± 4.1	0.09 ± 0.04	0.11 ± 0.01	0.09 ± 0.04	0.11 ± 0.01
GA	—	—	12.6 ± 2.8	13.9 ± 0.4	—	—	0.03 ± 0.01	0.03 ± 0.00
TA*	14.6 ± 5.3	16.9 ± 0.9	10.8 ± 4.6	12.4 ± 1.2	0.19 ± 0.08	0.22 ± 0.02	0.14 ± 0.07	0.16 ± 0.02

Species names are abbreviated as: BA: *B. australis*, BM: *B. maldonadoi*, TA: *T. areolatus*, PI: *P. irwini*, PT: *P. trucha*, GM: *G. maculatus*, GA: *G. australis*, CG: *C. galusdae*. A: adult, J: juvenile. TA*: Values correspond to arm 5.

not find difference in time of day for stranding and discussed on different results of other authors. In the other side, Freeman *et al.* (2001) describes that persistent (steady) flows can produce significant higher fish productivities.

Currently, in Chile it is not possible to establish or impose any management action on dam operations for protecting fish habitat availability or any other water use because of the lack of regulation on this matter. It is only possible to propose and (sometimes) introduce management actions in the EIA of new projects. In this context, legal tools like re-licensing of hydroelectric projects (US Federal Energy Regulatory Commission, www.ferc.gov) might be desirable for Chile, which will account for shift in societal values and will allow the participation of more stakeholders in decisions about what is the best use for water in the watershed (Goodwin *et al.*, 2006). For instance, a shift towards environmental preservation and enhancement, when social-economical welfare increased, has been seen in the Western USA in the last couple decades (Goodwin *et al.*, 2006). In Europe, the opening of the electric market has risen the 'green' or 'eco' labeling, where private households and customers in the service industry are willing to pay a surcharge for green electricity, supporting a more sustainable electricity production (Truffer *et al.*, 2001; Bratrich *et al.*, 2004).

In the Biobío Basin, although Pangue dam was first constructed, the hydroelectric complex Pangue-Ralco was supposed to operate jointly (as a structural solution for peaking), where Ralco dam would store some water from winter and early spring and operate producing hydropeaking events along the year to satisfy electric demand, and Pangue dam would store the flood pulses and release the water at a rather constant rate (Meier, 1995), which is the most desirable dam operation to reduce flow fluctuations from Pangue dam to the estuary (220 km). Thus, the quality of the physical habitat, the ecological conditions of the river, and more stable conditions for other water uses (water withdraws, recreation, etc.) would be provided. Habitat modification would be expensive even if done in a few reaches in a big river like the Biobío (mean slope of 3.1%, mean flow of $1000\,\mathrm{m^3\,s^{-1}}$ at the mouth) that modifies its streambed during winter. On the other hand, assessing natural water level drops after precipitation events and proposing upper limits to Pangue dam operation would require work beyond the scope of this paper, since official hourly flow data are only available since 1999 (3 years after the starting of Pangue dam operation) and the response of native fish to the hydrologic alterations has not been assessed. Thus, proposing biologically meaningful limits to Pangue dam operation, specially in summer season, would require to understand the effects of peak flows over native species, and not only adjust the daily dam operation to the most extreme hydraulic conditions occurring naturally in rare opportunities.

Effects of climate change have been predicted to change precipitation patterns and river discharges worldwide (IPCC, 2007). Likewise, an increase in the precipitation/snow ratio during winter has been predicted for the Biobío watershed for the near future, which will reduce the snow packages in the Andes and consequently will reduce discharges in the Biobío during the following spring and summer seasons (Stehr *et al.*, 2008). Thus, water uses from different stakeholders (hydropower companies, municipalities, irrigation associations, forestry industries, tourism companies, etc.) will be stressed and more pressure will fall on the river ecosystem. The influence of physical habitat on river biology is likely to decrease further downstream the river as other factors such as water quality become more

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dominant (Vannote *et al*., 1980). In fact, aquatic pollution is a mayor issue in the middle and lower Biobío River (Karrasch *et al*., 2006; Parra *et al*., 2008; Parra *et al*., 2009), which combined with less intensive effects of dam operation has been shown to decrease fish richness (Habit *et al*., 2005) and therefore, different management strategies should be applied along the Biobío River to improve its ecological status.

The assessment of different disturbing sources and a comparison with a natural regime can show how altered the quality of the water body is. Such simulations may be tools to describe the river health or ecosystem status and support managers in decision making. Although this idea has been around for some time in the scientific world, it will still take time to put it in practice in developing countries like Chile.

Once the shift in the location of good habitats and the amount of habitat availability for the species has been assessed, new questions that arise for further research are: how do individual fish respond to these disturbances? Does this situation impose some energetic limitations that affect the fish either in their heath, weight or reproductive rates with consequences to the ecosystem and to human beings? How can a more environmentally friendly water management be achieved with the participation of most of the stakeholders in the watershed? These dams have been operating for a short period and therefore most of the effects that they produce to the Biobío River may still be unknown.

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REFERENCES

Arratia G. 1983. Preferencias de hábitat de peces siluriformes de aguas continentales de Chile (Fam Diplomystidae y Trichomycteridae). *Studies of Neotropical Fauna and Environment* **18**: 217–237.

Barriga JP, Battini MA, Macchi P, Milano D, Cussac VE. 2002. Spatial and temporal distribution of landlocked *Galaxias maculatus* and *Galaxias platei* (Pisces: Galaxiidae) in a lake in the South American Andes. *New Zealand Journal of Marine and Freshwater Research* **36**: 345–359.

Becker CD, Fickeisen DH, Montgomery JC. 1981. *Assessment of Impacts from Water Level Fluctuations on Fish in the Hanford Reach, Columbia River*. Pacific Northwest Laboratory: Richland, Washington, USA. PNL-3813.

Bloech J, Schneider M, Ortlepp J. 2005. An application of physical habitat modelling to quantify ecological flow for the Rheinau hydropower plant, river Rhine. *Large Rivers* **16**(1–2): 305–328.

Bornette G, Henry C, Barrat M, Amoros C. 1994. Theoretical habitat templets, species traits and species richness: macrophytes in the Upper Rhône River and its floodplain. *Freshwater Biology* **31**: 487–505.

Bovee KD. 1982. A guide to stream habitat analysis using the instream flow incremental methodology, *Instream Flow Paper* 12, U.S. Fish and Wildlife Service, FWS/OBS-82/26, Fort Collins, Colorado, United States.

Bratrich C, Truffer B, Jorde K, Markard J, Meier W, Peter A, Schneider M, Wehrli B. 2004. Green hydropower: a new assessment procedure for river management. *River Research and Applications* **20**: 865–882. DOI: 10.1002/rra.788

Campos H, Gavilán JF, Alay F, Ruiz VH. 1993a. Comunidad íctica de la hoya hidrográfica del Río Biobío. In *Gestión de Recursos Hídricos de la Cuenca del Río Biobío y del Área Marina Costera Adyacente*, Faranda F, Parra O (eds). Centro EULA-Chile, Universidad de Concepción. Concepción, Chile. Serie: Monografias científicas Vol. 12: 249–278.

Campos H, Ruiz VH, Gavilán JF, Alay F. 1993b. *Peces del Río Biobío*, Centro EULA-Chile, Universidad de Concepción, Concepción, Chile.

Cushman RM. 1985. Review of ecological effects of rapidly varying flows downstream from hydroelectric facilities. *North American Journal of Fisheries Management* **5**: 330–339.

Ciborowski JJH. 1983. Downstream and lateral transport of nymphs of two mayfly species (Ephemeroptera). *Canadian Journal of Fisheries and Aquatic Sciences* **40**: 2025–2029.

de Mérona B, Vigouroux R, Tejerina-Garro FL. 2005. Alteration of fish diversity downstream from Petit-Saut Dam in French Guiana. Implication of ecological strategies of fish species. *Hydrobiologia* **551**: 1–33.

Dyer BS. 2000. Systematic review and biogeography of the freshwater fishes of Chile. *Estudios Oceanológicos* **19**: 77–98.

Freeman MC, Bowen ZH, Bobee KD, Irwin ER. 2001. Flow and habitat effects of juvenile fish abundance in natural and altered flow regimes. *Ecological Applications* **11**: 179–190.

García A, Habit E. 2007. Comparison of Qualitative and Pseudo-quantitative Approaches of Habitat Classification to Characterize Chilean Native Fish Species. *Proceedings of the 6th International Symposium on Ecohydraulics*, Christchurch, New Zealand.

García-Lancaster A. 2006. Aquatic habitat modeling of chilean native fish in a reach of the biobío river. *MS Thesis*, University of Idaho, Boise, ID.

García de Jalón D, Mayo M, Hervella F, Barceló E, Fernández T. 1993. *Principios y Técnicas de Gestión de la Pesca en Aguas Continentales*. Mundi-Prensa: Madrid.

Goodwin P, Jorde K, Meier C, Parra O. 2006. Minimizing environmental impacts of hydropower development: transferring lessons from past projects to a proposed strategy for Chile. *Journal of Hydroinformatics* **8**(4): 253–270. DOI: 10.2166/hydro.2006.005

Habit E, Victoriano P, Campos H. 2005. Ecología trófica y aspectos reproductivos de *Trichomycterus areolatus* (Pisces, Trichimycteridae) en ambientes lóticos artificiales. *Revista de Biología Tropical* **53**(1–2): 195–210.

Habit E, Belk MC, Tuckfield RC, Parra O. 2006. Response of the fish community to human-induced changes in the Biobío River in Chile. *Freshwater Biology* **51**(1): 1–11.

Habit E, Belk M. 2007. Threatened fishes of the world: *Percilia irwini* Eigenmann, 1927 (Perciliidae). *Environmental Biology of Fishes* **78**: 213–214.

Howard AFV, Zhou G, Omlin FX. 2007. Malaria mosquito control using edible fish in western Kenya: preliminary findings of a controlled study. *BMC Public Health* **7**: 199–214. DOI:10.1186/1471-2458-7-199

IPCC. 2007. Intergovernmental Panel on Climate Change report. Available at: http://ipcc-wgl.ucar.edu/wgl/wgl-report.html

Irvine RL, Oussoren T, Baxter JS, Schmidt DC. 2009. The effects of flow reduction rates on fish stranding in British Columbia, Canada. *River Research and Applications* **25**: 405–415. DOI: 10.1002/rra.1172

Jorde K. 1997. *Ökologisch begründete, dynamische Mindestwasserregelungen bei Ausleitungskraftwerken*, Dissertation. Universität Stuttgart, Stuttgart, Germany.

Jorde K, Schneider M, Peter A, Zöellner F. 2001a. Fuzzy based Models for the evaluation of fish habitat quality and instream flow assessment. *Proceedings of the 3rd International Symposium on Environmental Hydraulics*, Tempe, AZ.

Jorde K, Scheidt N, Peter A, Zöellner F. 2001b. Models for the evaluation of fish habitat quality and instream flow assessment. *Proceedings of the 3rd International Symposium on Environmental Hydraulics*, Tempe, AZ.

Jorde K, Burke M, Scheidt N, Welcker C, King S, Borden C. 2008. Reservoir operations, physical processes and ecosystem losses. In *Gravel-Bed Rivers VI: From Process Understanding to River Restoration*, Habersack H, Piégay H, Rinaldi M (eds). Elsevier, Amsterdam, The Netherlands: 607–636.

Junk WJ, Bayley PB, Sparks RE. 1989. The flood pulse concept in river-floodplain systems. *Canadian Journal of Fisheries and Aquatic Sciences* **106**: 110–127 (Special publication).

Karrasch B, Parra O, Cid H, Mehrens M, Pacheco P, Urrutia R, Valdovinos C, Zaror C. 2006. Effects of pulp and paper mill effluents on the microplankton and microbial self-purification capabilities of the Biobio River, Chile. *Science of the Total Environment* **359**(1–3): 194–208.

Knight TM, McCoy MW, Chase JM, McCoy KA, Holt RD. 2005. Trophic cascades across ecosystems. *Nature* **437**: 880–883. DOI: 10.1038/nature03962

Lamouroux N, Olivier JM, Capra H, Zylberblat M, Chandesris A, Roger P. 2006. Fish community changes after minimum flow increase: testing quantitative predictions in the Rhone River at Pierre-Benite, France. *Freshwater Biology* **51**: 1730–1743.

Lauters F, Lavandier P, Lim P, Sabaton C, Belaud A. 1996. Influence of hydropeaking on invertebrates and their relationship with fish feeding habits in a pyrenean river. *Regulated Rivers: Research & Management* **12**: 563–573.

Leclerc M. 2005. Ecohydraulics: a new interdisciplinary frontier for CDF. In *Computational Fluid Dynamics: Applications in Environmental Hydraulics*, Bates P, Lane S, Ferguson R (eds). John Wiley & Sons Ltd: Chichester, UK.

Link O, Espinoza A, Stehr A, García A. 2008. Development and verification of JAZZ1D: a stream temperature model. *Proceedings of the International Conference on Watershed Technology: Improving Water Quality and Environment*, American Society of Agricultural and Biological Engineers, Concepción, Chile.

Lundberg J, Berra T, Friel J. 2004. First description of small juvenile of the primitive catfish *Diplomystes* (Siluriformes: Diplomystidae). *Ichthyological Exploration of Freshwaters* **15**: 71–82.

Magilligan FJ, Nislow KH. 2005. Changes in hydrology regime by dams. *Geomorphology* **71**(1–2): 61–78.

Mäki-Petäys A, Huusko A, Erkinaro J, Muotka T. 2002. Transferability of habitat suitability criteria of juvenile Atlantic salmon (*Salmo salar*). *Canadian Journal of Fisheries and Aquatic Sciences* **59**: 218–228.

Manríquez A, Huaquin L, Arellano M, Arratia G. 1988. Aspectos Reproductivos de *Trichomycterus areolatus* Valenciennes, 1846 (Pisces: Teleostei: Siluriformes) en Río Angostura Chile. *Studies on Neotropical Fauna and Environment* **23**(2): 89–102.

Meier CI. 1995. A critical review of the Pangue Dam project EIA (Biobío River, Chile). In *Proceedings of the 1st Annual Conference of the Water Resources Engineering Division*, San Antonio, TX. American Society of Civil Engineering, Reston, Virginia.

Milhous RT, Bartholow JM, Updike MA, Moos AR. 1990. Reference manual for generation and analysis of habitat time series—version II, *Instream Flow Information Paper* No. 27. US Fish and Wildlife Service Biological Report, Vol. 90 (16), p. 249.

Montgomery DR, Buffington JM. 1997. Channel-reach morphology in mountain drainage basins. *Bulletin of the Geological Society of America* **109**(5): 596–611.

Mouton AM, Schneider M, Depestele J, Goethals PLM, De Pauw N. 2007. Fish habitat modelling as a tool for river management. *Ecological Engineering* **29**: 305–315.

Neira FJ. 1984. Biomorfología de la lampreas parásitas *Geotria australis* Gray, 1851y *Mordacia lapicida* (Gray, 1851) (Petromyzoniformes). *Gayana Zoología* **48**(1–2): 3–40.

Nilsson C, Reidy CA, Dynesius M, Revenga C. 2005. Fragmentation and flow regulation of world's large river systems. *Science* **308**: 405–408. DOI: 10.1126/science.1107887

Parra O, Vighi M, Chuecas L, Vismara R, Campos H. 1993. Caracterización físico y química, y evaluación de la calidad para uso múltiple del agua del río Biobío (Chile Central). In *Evaluación de la Calidad del Agua y Ecología del Sistema Limnético y Fluvial del río Biobío*, Faranda F, Parra O (eds). Serie Monografías Científicas EULA, Concepción, Chile Vol. 12: 15–159.

Parra O. 1996. *El río Biobío: Elemento base del desarrollo de la Región*, Cuadernos del Biobío, Ediciones Universidad de Concepción.

Parra O, Meier C. 2003. The Biobío and Laja River: fluvial systems under strong human pressure. Available at: www.cirf.org/kioto/biobio1.pdf

Parra O, Valdovinos C, Figueroa R, Habit E. 2008. *Programa de monitoreo de la calidad del agua del sistema río Biobío (Chile Central)*, August 2008. Centro EULA-Chile, Universidad de Concepción, Concepción, Chile.

Parra O, Valdovinos C, Habit E, Figueroa R. 2009. Log Term Study of the Biobío River: A Complex Multiuse Fluvial System in Chile. *Proceedings of The International Conference of Science and Technology for Sustainable Management of Aquatic Ecosystems*, Concepción, Chile.

Petts GE. 1984. *Impounded Rivers. Perspectives for Ecological Management*. John Wiley & Sons: Chichester.

Poof NL, Allan JD. 1997. The natural flow regime. *BioScience* **47**(11): 769–784.

Rooney N, McCann K, Gellner G, Moore JC. 2005. Structural asymmetry and the stability of diverse foodwebs. *Nature* **442**: 265–269. DOI: 10.1038/nature04887

Ruiz VH, López MT, Moyano HI, Marchant M. 1993a. Ichthyology of the high Biobío River system: some taxonomical, alimentary, reproductive and ecological aspects and a discussion on the river basin. *Gayana Zoología* **57**(1): 77–88.

Ruiz VH. 1993b. Ictiofauna del Río Andalién (Concepción, Chile). *Gayana Zoología* **57**(2): 109–278.

Ruiz VH, Berra T. 1994. Fishes of the high Biobio river of South-Central Chile with notes on diet and speculations on the origin of the ichthyofauna. *Icthyological Explorations to Freshwaters* **5**(1): 5–8.

Ruiz VH, Marchant M. 2004. *Ictiofauna de aguas continentales*, Facultad de Ciencias Naturales y Oceanográficas, Departamento de Zoología, Universidad de Concepción. Concepción, Chile.

Schneider M. 2001. *Habitat und Abflussmodellierung für Fließgewässer mit unscharfen Berechnungsansätzen,* Dissertation. Institut für Wasserbau, Universität Stuttgart, Stuttgart, Germany.

Schneider M, Jorde K. 2003. Fuzzy-ruled based models for the evaluation of fish habitat quality and instream flow assessment. *Proceedings of the IFIM User's Workshop*, Fort Collins, CO.

Smock L. 1996. Macroinvertebrate movements: drift, colonization and emergence. In *Methods in Stream Ecology*, Hauer FR, Lamberti GA (Eds). Academic Press: London.

Stehr A, Debels P, Arumi JL. 2008. Modelling hydrological response to climate change; experiences from two south-central Chilean Watersheds. *Proceedings of the International Conference on Watershed Technology: Improving Water Quality and Environmen*t, American Society of Agricultural and Biological Engineers, Concepción, Chile.

TWINBAS. 2007. Twinning European and third countries river basins for development of integrated water resources management methods: *Final Report*, June 2007. Available at: www.twinbas.org

Travnichek VH, Maceina MJ. 1994. Comparison of flow regulation effects on fish assemblages in shallow and deep water habitats in the Tallaposa River, Alabama. *Journal of Freshwater Ecology* **9**(3): 207–216.

Truffer B, Markard J, Bratrich C, Wehrli B. 2001. Green electricity from alpine hydropower plants. *Mountain Research and Development* **21**: 19–24.

United States Army Corps of Engineers (USACE). 2008. United States Army Corps of Engineers. Avaiable at: www.usace.army.mil/Pages/Default.aspx

Vannote RL, Minshall GW, Cummins KW, Sedell JR, Cushing CE. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* **37**: 130–137.

Vehanen T, Jurvelius J, Lahti M. 2005. Habitat utilisation by fish community in a short-term regulated river reservoir. *Hydrobiologia* **545**: 257–270.

Vila I, Contreras M, Fuentes L. 1996. Reproduction of *Diplomystes nahuelbutaensis* Arratia 1987 (Pisces: Diplomystidae). *Gayana Oceanología* **4**(2): 129–137.

Waters TF. 1972. The drift of stream insects. *Annual Review of Entomology* **17**: 253–272.

Ward JV, Stanford JA. 1983. The serial discontinuity concept of lotic ecosystems. In *Dynamics of Lotic Ecosystems*, Fontaine TD, Bartell SM (eds). Ann Arbor Scientific Publishers: Ann Arbor, MI; 29–42.

Wei-wei H, Wang G-x, Deng W, Li S-n. 2008. The influence of dams on ecohydrological conditions in the Huaihe River basin, China. *Ecological Engineering* **33**: 233–241.

Welcomme RL. 1985. River Fisheries. *FAO Fisheries Technical Paper* **262**: 1–318.

Wentworth CK. 1922. A scale of grade and class terms for classifying sediments. *The Journal of Geology* **30**: 377–392.

Wolman MG. 1954. A method of sampling coarse river-bed material. *Transactions of the American Geophysical Union (EOS)* **3**: 951–956.

Ždankus N, Sabas G. 2006. The impact of hydropower plant on downstream river reach. *Environment Research, Engineering and Management* **4**: 24–31.

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