



**The inbuilt long-term
unfeasibility of
environmental flows**

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The inbuilt long-term unfeasibility of environmental flows when disregarding riparian vegetation requirements

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Abstract

Environmental flows remain biased towards the traditional fish biological group and ignore the inter-annual flow variability that rules longer species life cycles, thus disregarding the long-term perspective of the riverine ecosystem. Incorporating riparian vegetation requirements into environmental flows could bring an important contribute to fill in this gap. The long-term after-effects of this shortcoming on the biological communities downstream of dams were never estimated before. We address this concern by evaluating the effects of environmental flow regimes disregarding riparian vegetation in the long-term perspective of the fluvial ecosystem. To achieve that purpose, the riparian vegetation evolution was modeled considering its structural response to a decade of different environmental flows, and the fish habitat availability was assessed for each of the resulting riparian habitat scenarios. We demonstrate that fish habitat availability changes accordingly to the long-term structural adjustments that riparian habitat endure following river regulation. Environmental flow regimes considering only aquatic biota become obsolete in few years due to the change of the habitat premises in which they were based on and, therefore, are unsustainable in the long run. Therefore, considering riparian vegetation requirements on environmental flows is mandatory to assure the effectiveness of those in the long-term perspective of the fluvial ecosystem.

1 Introduction

Freshwater ecosystems provide vital services for human existence that greatly exceed the commodities to which we commonly associate them (Daily, 1997) but are on the top world's most threatened (Dudgeon et al., 2006; Revenga et al., 2000) mainly due to river damming (Allan and Castillo, 2007). The capacity of freshwater ecosystems to provide goods and services is sustained by water-dependent ecological processes, whereby providing enough water to ensure its functioning is an important ethical concern (Acreman, 2001) which needs a firm commitment from science and policy to guar-

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antee that its provided life-support processes carry on unimpaired (Arthington et al., 2010). Such concern compelled the scientific community to appeal to all governments and water-related institutions across the globe to engage environmental flow restoration and maintenance in every river (Brisbane Declaration, 2007). In accordance, environmental flows have been acknowledged in major international programs addressing water issues (e.g. UNESCO-IHP, the IUCN, the DIVERSITAS freshwater BIODIVERSITY, Conservation International (CI), the World Wide Fund for Nature (WWF), the Ramsar Convention and the European Water Framework Directive).

Environmental flows can be defined as “the quantity, timing and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and wellbeing that depend upon these ecosystems” (Brisbane Declaration, 2007) and play an essential role on the conservation of freshwater ecosystems (Arthington et al., 2006; Hughes and Rood, 2003). However, for a successful environmental flow management it is essential to possess a great understanding of the existing hydroecological interactions in order to properly model those and thereby efficiently achieve the protection and rehabilitation of the riverine ecosystems (Arthington et al., 2010).

Environmental flow management has been an ongoing debate among the scientific community for the last couple decades (e.g. M. Acreman et al., 2014; Acreman and Dunbar, 1999; M. C. Acreman et al., 2014; Arthington et al., 1992; Arthington and Zaslucki, 1998; Dyson et al., 2003; Gillespie et al., 2014; Hughes and Rood, 2003; King and Brown, 2006; King et al., 2003; Poff et al., 2010; Poff and Zimmerman, 2010; Rood et al., 2003) in which the last ten years were prolific in methodological development (Davies et al., 2013; Dyson et al., 2003; Richter et al., 2006; Tharme, 2003) and application of environmental flows (M. C. Acreman et al., 2014). As a result, it is now consensual that the flow regime is the key driver of river ecosystems (Power et al., 1995; Walker et al., 1995) and its natural flow characteristics are critical to maintain ecological integrity (Poff et al., 1997) due to their direct influence on the local structure of both aquatic and riparian communities (Johnson and Waller, 2012; Kuiper et al., 2014; Nilsson and Svedmark, 2002; Poff and Allan, 1995; Shilla and Shilla, 2012).

Accordingly, environmental flows must be based on the ecological requirements of different biological communities (Acreman et al., 2009) and should present a dynamic and variable water regime to maintain the native biodiversity and the ecological processes that portray every river (Bunn and Arthington, 2002; Lytle and Poff, 2004; Poff et al., 1997; Postel and Richter, 2003). For instance, environmental flows must incorporate the temporal variability in habitat preferences of stream fish in order to represent critical habitat requirements and therefore lead to the setting of appropriate flow targets (Davey et al., 2011). Additionally, increased attention has been given to better mimic the historical flood levels downstream of dams (Arthington and Pusey, 2003) and to the importance of the role of physical habitat heterogeneity (Holl et al., 2013).

However, there is still a need for more comprehensive process-based restoration approaches (Beechie et al., 2010) and environmental flows persist generally based on the requirements of a single biological group, mostly fish (Acreman et al., 2009; Tharme, 2003). Consequently, environmental flows remain biased towards this traditional biological group and clearly need for less typically monitored taxa in future studies (Gillespie et al., 2014). In addition, these biased approaches typically determine environmental flow requirements on a hydrological year basis, considering different times of the year and at most different water-year types (wet, dry and average years). As a result, environmental flows still disregard the inter-annual flow variability that rules longer species lifecycles and therefore lack the long-term perspective of the riverine ecosystem (Stromberg et al., 2010). The feedbacks of these shortcomings on the biological communities downstream of dams were seldom estimated before and for that reason the assessment of the efficiency of such biased approaches along with its long-term after-effects remain practically unknown.

Incorporating riparian vegetation requirements into environmental flows could bring an important contribute to fill in these gaps. A recent comprehensive study on restoration outcomes has considered riparian restoration as an indispensable implementation measure to recover the natural processes and the most promising restoration action in many degraded rivers (Palmer et al., 2014). Besides, riparian vegetation is a suit-

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able environmental change indicator (Benjankar et al., 2012; Nilsson and Berggren, 2000; Rodríguez-González et al., 2014) that responds directly to flow regime in an inter-annual timeframe (Capon and Dowe, 2007; Junk et al., 1989; Mallik and Richardson, 2009; Naiman et al., 2005; Poff et al., 1997; Richter et al., 1997; Toner and Keddy, 1997) and has a clear significance in the habitat improvement of aquatic systems (e.g. Aguiar et al., 2002; Broadmeadow and Nisbet, 2004; Davies-Colley and Quinn, 1998; Ghermandi et al., 2009; Gregory et al., 1991; Naiman and Décamps, 1997; Naiman et al., 2005; Pusey and Arthington, 2003; Tabacchi et al., 2000; Van Looy et al., 2013).

The purpose of this study is to evaluate the efficiency of environmental flow regimes when disregarding riparian vegetation in the long-term perspective of the fluvial ecosystem. We were particularly interested in answering the following questions: are the fish only-addressed environmental flows capable of preserving the habitat availability of aquatic species in the long-term? In what extent could this overlook derail the goals of environmental flows addressing only aquatic species as a result of the riparian habitat degradation? Are environmental flows regarding riparian requirements able to maintain the habitat availability of fish species?

We approached these questions using riparian vegetation modeling to forecast its structural response to a decade of different environmental flows followed by fish habitat availability assessment in each of the resulting riparian habitat scenarios. Such modeling approach was never used before in the validation of the long-term efficiency of environmental flow regimes and can provide in advance an extremely valuable insight of the expected long-term effects of environmental flows in river ecosystems.

2 Methods

2.1 Study site

The study site area is situated in the Ocreza River, East Portugal, immediately downstream of the projected location of the Alvito hydroelectric power plant (39°44'07.05" N,

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7°44'16.51" W; Fig. 1) and 30 km upstream from the river mouth. The Ocreza River is a medium-sized stream that runs on schistose rocks for 94 km and drains a 1429 km² watershed with a mean annual flow of 16.5 m³ s⁻¹. Its flow regime is typically Mediterranean with a low flow period interrupted by flash floods in winter and a very low flow, even null, during summer (Gasith and Resh, 1999). At the study site, river is free flowing on a boulder substrate and is considered to be representative of the overall river course (Boavida et al., 2014). The study site area encompasses a river length of approximately 500 m, laterally limited by the 100 year flooded zone and totaling approximately 4 ha. Herein, fish community is characterized by native cyprinid species, mainly *Luciobarbus bocagei* (Iberian barbel, hereafter barbel), *Pseudochondrostoma polylepis* (Iberian straight-mouth nase, hereafter nase) and *Squalius alburnoides* (calandino) whereas the local riparian vegetation is composed mostly by willows (*Salix salviifolia* Brot. and *Salix atrocinerea* Brot.) and ashes (*Fraxinus angustifolia* Vahl).

2.2 Data collection

2.2.1 Hydraulic data

The riverbed topography was surveyed in 2013 using a combination of a Nikon DTM330 total station and a Global Positioning System (GPS) (Ashtech, model Pro Mark2). Altogether, 7707 points were surveyed at the studied site. Trees, boulders and large objects emerging from the water were defined by marking the object intersection with the riverbed and by surveying the points necessary to approximately define its shape.

Hydraulic data – i.e., water velocities and depths – were measured at a series of points along 9 cross-sections. Depths were measured with a ruler and water velocities were measured with a flow probe (model 002, Valeport) positioned at 60 % of the local depth below the surface (Bovee and Milhous, 1978). These data were used to calculate discharge and to calibrate the model. Additionally, the substrate composition was visually assessed (supplementary information on hydraulic data and channel bed characteristics is presented in the Supplement, Sect. S1).

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2.2.2 Riparian vegetation data

The riparian vegetation assessment was accomplished simultaneously with the river bed topography and intended to characterize the existing riparian landscape in order to calibrate the riparian vegetation model. This task consisted in recording the location and shape of all homogeneous vegetation patches with a sub-meter precision handheld GPS (Ashtech, Mobile Mapper 10) while dendrochronological methods were used to determine the approximate age of the patches. The homogeneous vegetation patches were later classified by succession phase according to its corresponding development stage (patch georeferencing, patch aging and succession phase classification followed Rivaes et al., 2013).

Five succession phases were identified in the study site: Initial phase (IP), Pioneer phase (PP), Early Successional Woodland phase (ES), Established Forest phase (EF), and Mature Forest phase (MF). Initial phase was attributed to all patches dominated by open sand or gravel bars, sometimes covered by herbaceous vegetation but without woody arboreal species. The patches dominated by woody arboreal species recruitment were considered as Pioneer phase. The Early Successional Woodland phase classification was attributed to all patches with a high standing biomass and well established individuals, dominated by pioneer watertable-dependent species like willows and alders. Older patches presenting moderate to high canopy cover and dominated by macrophanerophytes like ash-trees were considered as Established Forest phase. The Mature Forest phase was considered at patches where terrestrial vegetation was also present, determining the transition phase to the upland vegetation communities. Further information on riparian vegetation patch characterization of succession phases is provided in the Supplement (Sect. S2).

2.2.3 Fish data

Fish populations were sampled during 2012 and 2013 at undisturbed or minimally disturbed sites in the Ocreza basin, an essential requisite when studying habitat prefer-

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ences in stream fish to reflect their optimal habitat (Gorman and Karr, 1978). Sampling took place in autumn (November 2012), spring (May 2013) and early summer (June 2013) when there is full connectivity among instream habitats. Overall, four native species (cyprinids) were found – barbel, nase, calandino and the Southern Iberian chub *Squalius pyrenaicus*. The latter was however excluded from the present study, as an insufficient number of individuals were collected to draw unbiased conclusions. Non-native fish (represented by a single species, the gudgeon *Gobio lozanoi*) occurred in the study area but in very low density. Field procedures followed Boavida et al. (2011, 2013a, b). Sampling was performed during daylight using pulsed DC electrofishing (SAREL model WFC7-HV; Electracatch International, Wolverhampton, UK), with low voltage (250 V) and a 30 cm diameter anode to reduce the effect of positive galvanotaxis. A 200 m long reach at each site was surveyed by wading upstream in a zigzag pattern to ensure full coverage of available habitats. To avoid displacements of individuals from their original positions, a modified point electrofishing procedure was employed (Copp, 1989). Sampling points were approached discreetly, and the activated anode was swiftly immersed in the water for 5 s. Upon sighting a fish or a shoal of fishes, a numbered location marker was anchored to the stream bed for subsequent microhabitat use measurements. Fish were immediately collected by means of a separate dip net held by another operator, quickly measured for total length (TL), and then placed in buckets with portable ELITE aerators to avoid continuous shocking and repeated counting, before being returned alive to the river.

Following fish sampling, flow depth (cm), mean water velocity (cm s^{-1}) and dominant substrate composition were measured in $0.8 \text{ m} \times 0.8 \text{ m}$ quadrates directly below the fish. Depth was measured with a meter rule to the nearest centimeter. Water velocities were measured with a water flow probe (model FP101, Global Water Instrumentation, Inc., USA). For depths of less than 0.8 m, mean water velocity was measured at 60 % of the distance from the surface to the substrate; otherwise, water velocity was considered to be the mean of measurements taken at 20 and 80 % of total depth (Bovee and Milhous, 1978). Substrate was measured according to the modified Wentworth

scale (Bovee, 1986) and assigned as (1) organic detritus, (2) silt (1–2 mm), (3) sand (2–5 mm), (4) gravel (5–25 mm), (5) cobble (25–50 mm), (6) pebble (50–150 mm) and (7) boulders (> 150 mm). Microhabitat availability measurements were made using the same variables by quantifying randomly selected points along 15–25 m equidistant transects perpendicular to the flow at each sampling site. In order to develop Habitat Suitability Curves (HSC) (Boavida et al., 2013b; Bovee, 1986; Vismara et al., 2001) for target fish size classes, microhabitat variables (flow depth, water velocity and dominant substrate) were divided into classes, and histograms of frequencies of use and availability were built (Boavida et al., 2011). A summary on collected fish data as well as data analysis to determine habitat fish use, habitat availability and fish habitat preference is provided in the Supplement (Sect. S2).

2.3 Flow regime definition

The evolution of riparian vegetation was modeled according to three flow regimes: (i) the natural flow regime (hereafter named Natural flow regime), (ii) an environmental flow regime considering only fish requirements (hereafter named Eflow regime) and (iii) an environmental flow regime considering both fish and riparian requirements (hereafter named Eflow+flush regime). The considered environmental flow regimes were adapted from the environmental flow regime proposal for the future Alvito dam. This environmental flow regime considers both fish species and riparian vegetation requirements and is presented in a multiannual fashion for a 10 year period incorporating two discharge components: a mean monthly discharge intended to address fish species requirements and a series of flushing flows with different recurrence intervals to fulfill riparian vegetation requirements. Fish-addressed environmental flow regime was determined according to the Instream Flow Incremental Methodology (Bovee, 1982) and aimed the following goals: (i) maximize the habitat of the target species while attributing the same weight for each species, (ii) privilege the spawning months (spring) and promote the younger life stages during summer, (iii) maintain the characteristic intra-annual variability of the river flow; and (iv) preserve the natural regime whenever

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the environmental flows suggest higher discharges. The riparian requirements were defined based on the need of riparian communities for the minimum necessary flushing flow regime to maintain the viability and sustainability of riparian vegetation, particularly, avoiding vegetation encroachment and conserving the ecological succession equilibrium of the riparian ecosystem. The method consists in the assessment of the deviation from the natural reference riparian landscape of the forecasted riparian landscapes driven by diverse flushing flow regimes considering various combinations of floods with different recurrence intervals (Rivaes et al., 2015). Thus, the Eflow regime only acknowledges the mean monthly discharges addressing fish species requirements while the Eflow+flush regime incorporates both mean monthly discharges and flushing flows (Fig. 2).

2.4 Riparian vegetation modeling

The riparian vegetation modeling was performed using the *CASiMiR-vegetation* model (Benjankar et al., 2009). This tool simulates the riparian vegetation succession dynamics based on the existing relationships of the ecological relevant hydrologic elements (Poff et al., 1997) and the vegetation metrics that reflect riparian communities to such hydrologic alterations (Merritt et al., 2010). The rationale of this model is based on the fact that riparian communities respond to the hydrologic and habitat variations on a time scale between the year and the decade (Frissell et al., 1986; Thorp et al., 2008) being the flood pulse the predominant factor on these population dynamics (Thoms and Parsons, 2002). For these reasons, the hydrologic regime is inputted into the model in terms of maximum annual discharges as these discharges are considered as the annual threshold for riparian morphodynamic disturbance that determine the succession or retrogression of vegetation.

Model calibration was carried out in accordance with the methodology described in previous studies (García-Arias et al., 2013; Rivaes et al., 2013). During calibration, the riparian vegetation model achieved an agreement evaluation of 0.61 by the quadratic weighted kappa (Cohen, 1960), which is considered as a good (Altman, 1991; Viera

and Garrett, 2005) agreement with the observed riparian landscape. After calibration (calibrated parameters in the Supplement, Sect. S1), the riparian vegetation was modeled for periods of ten years according to the flow regimes (Table 1). Such modeling period was considered to be long enough to avoid the influence of the initial vegetation conditions while river morphological changes still do not assume importance in vegetation development (Politti et al., 2014). The resulting riparian vegetation maps were then used as the respective riparian habitats (hereafter named Natural, Eflow and Eflow+flush habitats) in the hydrodynamic modeling of fish habitat.

2.5 Hydrodynamic modeling of fish habitat

The hydrodynamic modeling was performed using a calibrated version of the River2D model (Steffler and Blackburn, 2002). Calibration procedure followed Boavida et al. (2013b, 2014) and calibrated parameters are provided in the Supplement (Sect. S1). River2D is a finite element model widely used in fluvial modeling studies for the assessment of habitat availability (Boavida et al., 2011; Jalón and Gortázar, 2007) that brings together a 2-D hydrodynamic model and a habitat model to simulate the flow conditions of the river stretch and estimate its potential habitat value according to the fish habitat preferences. The hydrodynamic modeling comprised the Eflow discharge range ($0-2 \text{ m}^3 \text{ s}^{-1}$) and was accomplished for each riparian habitat. The riparian habitats were represented in the hydrodynamic model by changing the channel roughness accordingly to the spatial extent of the riparian succession phases. Roughness classification of riparian vegetation succession phases was determined based on roughness measurement literature on similar vegetation types (Chow, 1959; Wu and Mao, 2007). The hydraulic characteristics of each habitat (roughness, flow depth and velocity) were compared using a *t* test in R environment (R Development Core Team, 2011) in order to determine the existence of mean significant differences between habitats. Habitat simulation was achieved by the combination of the hydraulic modeling (flow depth and velocity) and substrate with preference curve information for the considered target species. The Habitat Suitability Index (HSI) was determined for each specie and

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life stage regarding the product of the velocity (Velocity Suitability Index – VSI), depth (Depth Suitability Index – DSI) and substrate (Substrate Suitability Index – SSI) variables, according with Eq. (1).

$$HSI = VSI \times DSI \times SSI \quad (1)$$

- 5 The product of the HSI by the influencing area (A) of the corresponding model i th node defines the Weighted Usable Area (WUA) of that node. The sum of the WUA's result in the total amount of habitat suitability for the study site, as described by Eq. (2).

$$WUA = \sum_{n=1}^i A_i \times HSI_i = f(Q) \quad (2)$$

10 Considering that the BACI approach (Before-After Control-Impact) is generally the best way of detecting impacts or beneficial outcomes in river systems (Downes et al., 2002) the resulting WUA's were then compared to the natural habitat in a census-based benchmark. The equality of proportions between habitat availabilities was tested using the χ^2 test for proportions in R environment (R Development Core Team, 2011) while deviations were measured using the most commonly used measures of forecast
15 accuracy, namely, Root Mean Square Deviation (RMSD), Mean Absolute Deviation (MAD) and Mean Absolute Percentage Deviation (MAPD). The RMSD and MAD are scale-dependent measures in which the former penalizes variance giving more weight to larger error values while the latter is unambiguous giving the same weight to all errors. MAPD is a non-scale dependent measure adjusting for population size using
20 a percentage error to allow for a forecast comparison between datasets. In all cases, smaller values of these measures indicate better performance in parameter estimation.

3 Results

Field survey resulted in the assessment of 56 vegetation patches and in the capture of 2091 cyprinid fishes. The succession phases of riparian vegetation were arranged by

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gradients of height to mean water level, patch age and woody species richness. During fish sampling, four different species were captured, namely, barbel, nase, calandino and the Southern Iberian chub (*Squalius pyrenaicus*). The barbel and calandino were the most abundant species representing near 51 and 37 % of the captures, respectively (further information on collected data is provided in the Supplement (Sect. S2)).

Different configurations of the riparian habitat resulted from the riparian vegetation modeling according to the flow regimes (Fig. 3). The riparian habitat driven by the natural flow regime presents a river channel largely devegetated where Initial phase (IP) and Pioneer phase (PP) represent approximately 43 % of the study site. In this habitat, Early Succession Woodland phase (ES) can only settle in about 8 % of the study area. The floodplain succession phases, namely Established Forest phase (EF) and Mature Forest phase (MF), represent near 40 and 10 % of the study area, respectively. In contrast, the riparian habitat created by the Eflow regime is where the riparian vegetation encroachment is more prominent. Herein, riparian vegetation settles in the channel and evolves towards mature phases due to the lack of the river flood disturbance. IP is now reduced to about 3 % of the study area and PP is inexistent. ES covers up to about 48 % of the study area while EF and MF maintain about the same covered area. The riparian habitat driven by the Eflow+flush regime shows the capacity of this flow regime in hold back vegetation encroachment. In this case IP and PP is maintained at approximately 30 % coverage while ES is kept under 21 % of the study site area. Once again, EF and MF maintain their covered areas.

The changes undertaken by the riparian vegetation are able to modify the hydraulic characteristics of the river stretch (Fig. 4). Channel roughness height changes dramatically accordingly with the considered riparian habitats. The natural habitat presents the lowest roughness height values with an average roughness of 0.46 m. Channel roughness height increases accordingly to the encroachment level of vegetation reaching a mean value of 0.71 m on the Eflow+flush habitat and 1 m on the Eflow habitat. Those mean values are significantly different between all three habitats. Changes also occur in flow depth and flow velocity for the considered flow range of the proposed envi-

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ronmental flows. The Eflow habitat creates a circumstance with higher depths (mean depth is 0.402 m) and lower flow velocities (flow velocity is 0.128 m s^{-1}) which are significantly different from the Natural and Eflow+flush habitats. In contrast, depth and flow velocity are not significantly distinguishable between the Natural and Eflow+flush habitats, where mean depth and flow velocity are respectively 0.397 m and 0.136 m s^{-1} in the former, and 0.399 m and 0.135 m s^{-1} in the latter.

During a hydrological year, each riparian habitat provides different weighted usable areas for the target fish species considering the same fish-addressed environmental flow regime (Fig. 5). Differences from the natural habitat availability are greater in the Eflow habitat. In this case, major differences in the WUA can be found almost all year round for the barbel juveniles, throughout autumn and winter months for the nase juveniles and during spring months for the calandino. Comparatively to the natural habitat, the habitat modifications created by the Eflow habitat are on average about 12 % where in a third of the cases is higher than 15 % and reaching 80 % in an extreme situation. Particularly, the Eflow habitat provides less habitat availability during autumn and winter months for the barbel and nase juveniles, ~ 17 and 14 %, respectively. Likewise, for this habitat the habitat availability during spring months is increased approximately 23 % for the barbel juveniles and increased about 20 and 27 %, respectively for the calandino juveniles and adults. On the other hand, throughout the year, the Eflow and Flush habitat provides a WUA very similar to the natural habitat. The habitat changes created by the Eflow+flush habitat are on average around 2 % and never reach 8 % for all species and life stages. Accordingly, the WUA differences evidenced in the Eflow habitat revealed to be significant by the χ^2 test while this were never the case for the matter of the Eflow and Flush habitat (Supplement, Sect. S3).

The riparian-induced modifications on the fish habitat availability are also confirmed by all the employed deviation measures (Table 2). According to RMSD, MAD and MAPD, the Eflow habitat is farther apart from the natural habitat for all species and life stages, being the barbel and nase juveniles the most penalized.

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4 Discussion

This study evaluated the benefits of incorporating riparian requirements into environmental flows by estimating the expected repercussions of riparian long-term changes driven by regulated flow regimes on the fish habitat availability. In order to do so, the riparian vegetation was modeled for 10 year periods according to the different flow regimes and results were inputted as the habitat basis for the hydrodynamic modeling and following fish habitat availability assessment. Such ecological modeling approach over a decadal time-scale pushes through realistic biological-response modeling and substantiates the long-term research that is required in a 21st Century environmental flow science (Arthington, 2015; Petts, 2009). Furthermore, this approach allows foreseeing and assessing the outcome of recommended flow regimes which is an essential topic, although poorly considered in environmental flow science (Arthington and Zalucki, 1998; Davies et al., 2013; Gippel, 2001). In this case study, the capability of providing in advance an extremely valuable insight of the expected long-term effects of environmental flows in river ecosystems enabled us to demonstrate the remarkable role of riparian vegetation on the support of environmental flows, which revolutionizes the actual paradigm in environmental flow science.

The vegetation modeling results confirm that the natural flow regime generates the major morphodynamic disturbance of the three considered flow regimes where riparian vegetation is forced to a metastable oscillation state (Formann et al., 2013). Such disturbance must be preserved artificially in regulated rivers in order to maintain the viability and sustainability of the riparian communities. To do so, the riparian landscape management downstream of dams towards its restoration should be performed by releasing environmental flows considering riparian requirements (Greet et al., 2011a, b; Miller et al., 2013; Rivaes et al., 2015). The riparian modeling results obtained in the present research are consistent with this. Without adequate environmental flows, the flood frequency reduction determines the riparian vegetation settlement and aging in the river channel.

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Our microhabitat analysis demonstrated that the changes in the riparian habitat induce modifications in the hydraulic characteristics of the river stretch. Furthermore, the hydrodynamic modeling determined that those changes affect directly the habitat availability of the existing fish species. The relation between fish assemblages and habitat has long been acknowledged (Ayllón et al., 2009; Matthews, 1998; Schlosser, 1987, 1988) with changes in fish habitat availability posing profound effects on fish distribution and assemblages (Clark et al., 2008; Pusey et al., 1993; Vadas and Orth, 2001). Moreover, the habitat loss has been considered a main threat to the concerned fish species, particularly for the nase and the calandino (Cabral et al., 2006). With regard to this particular case study, the habitat shaped by the Eflow regime diverges substantially from the Natural habitat availability for the species life stages, throughout the different seasons. The habitat decrease of barbel and nase juvenile during autumn and winter months jeopardizes those species survival by refuge loss which is particularly important in flashy rivers (Hershkovitz and Gasith, 2013), such as the Ocreza. During spring months the adult barbel's habitat is also decreased, undermining the spawning activity and consequently the sustainability of future population stocks (Lobón-Cervía and Fernandez-Delgado, 1984). In contrast, during this season the juvenile barbel's and calandino habitats are substantially increased compared to the habitat provided by the Natural flow regime. This may seem a positive effect, but one may not ignore that the relations between fish assemblages and habitat are extremely complex (e.g. Diana et al., 2006; Hubert and Rahel, 1989; Santos et al., 2011), being a consequence of the actual natural conditions (Poff et al., 1997; Poff and Allan, 1995) that when disrupted may allow the expansion of more generalist and opportunistic fauna (Poff and Ward, 1989). In fact, this is the particular case of the calandino which has a great plasticity regarding the habitat use and has an opportunistic behavior in terms of the habitat exploitation (Doadrio, 2011; Gomes-Ferreira et al., 2005). In addition, the revealed habitat changes and expected impacts on fish habitat availability driven by environmental flows disregarding riparian vegetation requirements are predicted for a decade of flow regulation. Considering that a dam life span has been considered to be up to 100 years

(Cooper et al., 2005; Wieland, 2010; Workman, 2007), but in fact, will exceed this limit in many cases, further changes are likely to continue occurring with possible dramatic consequences to downstream reaches.

Environmental flows regarding riparian vegetation requirements are able to preserve the naturalness of the riparian habitat and consequently the maintenance of the fish habitat availability. Furthermore, these environmental flows impose minor revenue losses to dam managers (Rivaes et al., 2015) with significant positive ecological effects in downstream reaches (Beschta et al., 1987; Connell, 1978; Lorenz et al., 2013; Pusey and Arthington, 2003; Shilla and Shilla, 2012) and ecosystem services provision (Berges, 2009; Blackwell and Maltby, 2006; Gumiero et al., 2013; Hey and Philippi, 1995; Sweeney et al., 2004). The implementation of such environmental flows could also provide an additional way to attain the “good ecological status” required by the Water Framework Directive (WFD). Besides, taking up a procedure like this can act both as “win-win” and “no-regret” adaptation measures during the second phase of the WFD, because it potentiates the improvement of other ecological indicators and mitigates the impacts of flow regulation while being robust enough to account for different scenarios of climate change (EEA, 2005).

Little empirical knowledge exists to support our modeling results, as water science still lacks strong links between flow restoration and its ecological benefits (Miller et al., 2012), particularly regarding long-term monitoring of environmental flow performance (King et al., 2015 and citations herein). Nevertheless, there are some observational evidence that can somehow provide support to our conclusions regarding the ecological response of riparian vegetation to environmental flows (Bond et al., 2014; Little et al., 2012; Miller et al., 2013; Shafroth et al., 2010; Sims and Colloff, 2012; Stromberg and Patten, 1990; Webb et al., 2013), as well as the effects of the riparian quality and physical habitat condition on fish fauna (Arthington et al., 2015; Curran and Hession, 2013; dos Santos et al., 2015; Efird and Konar, 2014; Jowett et al., 2009; Mouton et al., 2012; Rowe et al., 2009).

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5 Conclusions

In conclusion, we demonstrated that fish habitat availability changes accordingly to the long-term structural adjustments that riparian habitat endure following river regulation. These changes can be assigned to the modifications that altered riparian habitats induce on the hydraulic characteristics of the river stretches. Environmental flow regimes considering only aquatic biota will become obsolete in a few years due to the alteration of the habitat premises in which they were based on and, therefore, are unsustainable in the long-term perspective of the fluvial ecosystem, failing to achieve the desired effects on aquatic communities to which those environmental flows were proposed in the first place. An environmental flow regime that also considers riparian vegetation requirements contributes to preserve the hydraulic characteristics of the river channel at the natural riverine habitat standards, therefore maintaining the habitat assumptions which support the environmental flow regimes regarding aquatic communities. Accounting for riparian vegetation requirements is thus mandatory to assure the effectiveness of environmental flow regimes in the long-term perspective of the fluvial ecosystem.

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Table 1. Maximum annual discharges ($\text{m}^3 \text{s}^{-1}$) considered in the CASiMiR-vegetation model.

Year	Natural	Eflow	Eflow+flush
1	671	0.99	0.99
2	203	0.99	167
3	327	0.99	0.99
4	217	0.99	167
5	316	0.99	0.99
6	371	0.99	167
7	702	0.99	0.99
8	202	0.99	167
9	195	0.99	0.99
10	440	0.99	371

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Table 2. Deviation analysis of the weighted usable areas for the considered regulated flow regimes benchmarked by the natural flow regime (RMSD – Root Mean Square Deviation, MAD – Mean Absolute Deviation, MAPD – Mean Absolute Percentage Deviation).

	Eflow			Eflow+Flush		
	RMSD (m ²)	MAD (m ²)	MAPD (%)	RMSD (m ²)	MAD (m ²)	MAPD (%)
<i>Luciobarbus bocagei</i> (juv.)	86.00	72.10	15.40	12.17	7.24	2.52
<i>Luciobarbus bocagei</i> (adult)	29.46	20.55	5.83	2.87	2.12	1.55
<i>Pseudochondrostoma polypepis</i> (juv.)	128.21	86.14	11.58	9.42	5.72	2.26
<i>Pseudochondrostoma polypepis</i> (adult)	7.32	5.85	18.70	2.17	1.37	2.10
<i>Squalius alburnoides</i> (juv.)	44.05	28.16	8.46	6.20	4.06	2.10
<i>Squalius alburnoides</i> (adult)	92.41	52.47	10.23	7.49	5.31	2.37

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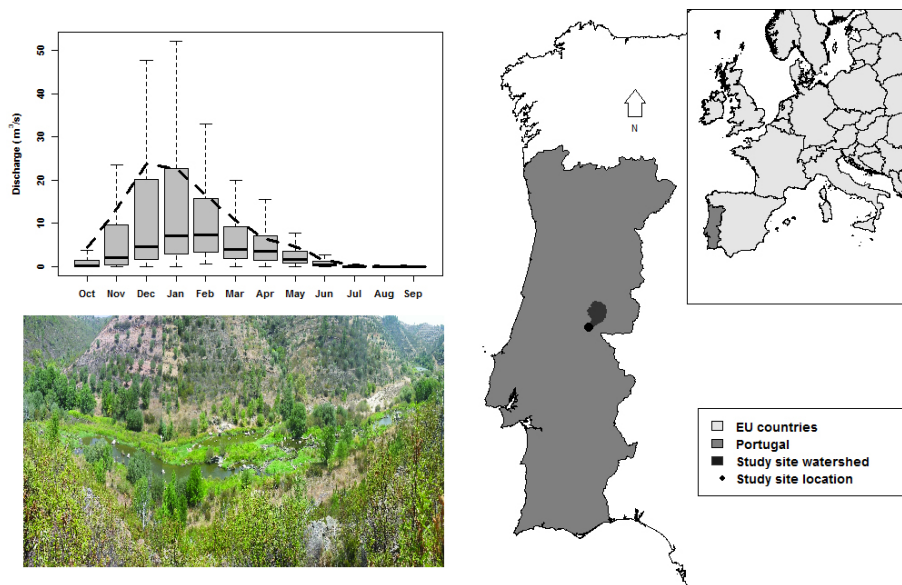


Figure 1. Location of the study site. Box plots stand for the study site mean daily discharge and dashed bold line for the mean monthly discharge.

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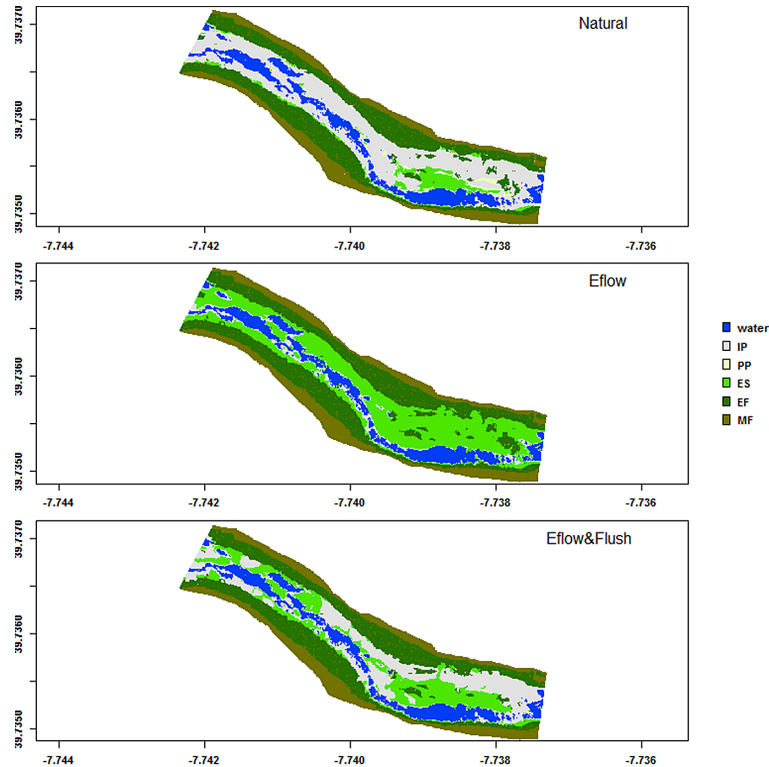


Figure 3. Expected patch mosaic of the riparian vegetation habitats shaped by the considered flow regimes (detailed by succession phase, namely, initial phase – IP, pioneer phase – PP, early succession woodland phase – ES, established forest phase – EF and mature forest phase – MF).

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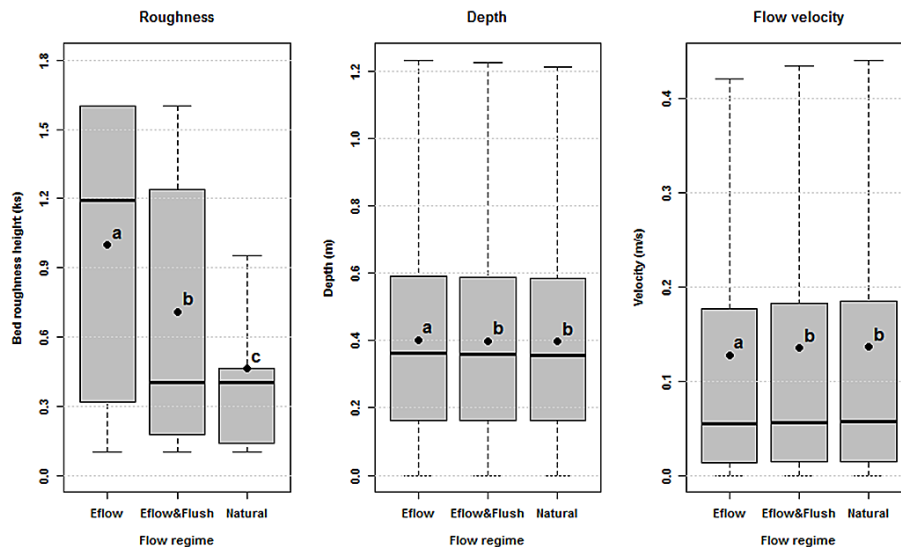


Figure 4. Hydraulic characterization of the study site according to the different expected riparian vegetation habitats driven by the considered flow regimes (data obtained from 2-D hydrodynamic modeling). Letters stand for the significant different mean groups (t test). Boxplots portray value range, thick black lines the median value and black dots the mean values.

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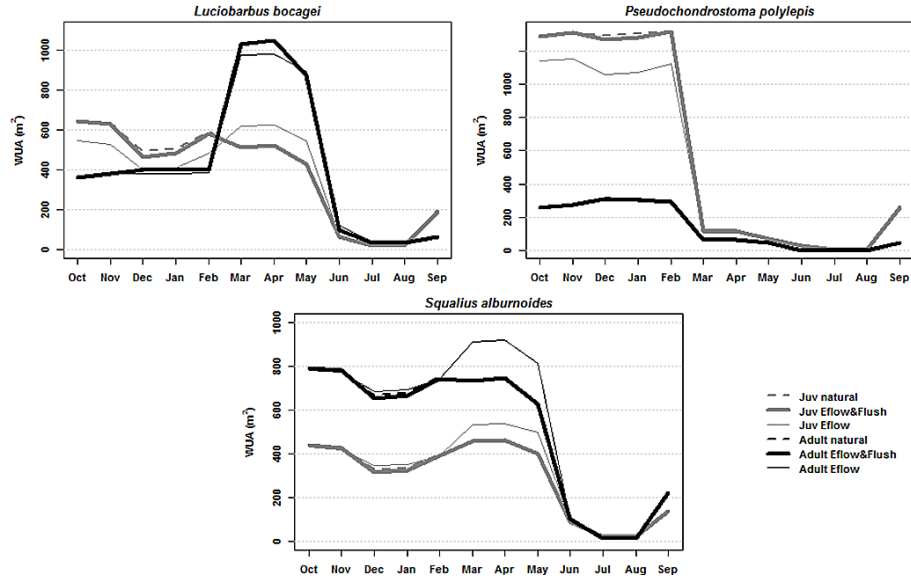


Figure 5. Fish weighted usable areas provided by the different riparian habitats during the release of the environmental flow regime addressing only fish requirements (Eflow).