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Importance of considering riparian vegetation requirements for the long-term efficiency of environmental flows

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Abstract. Environmental flows remain biased toward the traditional biological group of fish species. Accordingly, these flows ignore the inter-annual flow variability that rules species with longer life cycles, thereby disregarding the long-term perspective of the riverine ecosystem. We analyzed the influence of considering riparian requirements for the long-term efficiency of environmental flows. For that analysis, we modeled the riparian vegetation development for a decade facing different environmental flows in two case studies. Next, we assessed the corresponding fish habitat availability of three common fish species in each of the resulting riparian habitat scenarios. Modeling results demonstrated that the environmental flows disregarding riparian vegetation requirements promoted riparian degradation, particularly vegetation encroachment. Such circumstance altered the hydraulic characteristics of the river channel where flow depths and velocities underwent changes up to 10 cm and 40 cm s^{-1}, respectively. Accordingly, after a decade of this flow regime, the available habitat area for the considered fish species experienced modifications in absolute from 18.16% to 109.75% compared to the natural habitat. In turn, environmental flows regarding riparian vegetation requirements were able to maintain riparian vegetation near natural standards, thereby preserving the hydraulic characteristics of the river channel and sustaining the fish habitat close to the natural condition. As a result, fish habitat availability never changed more than 16.17% from the natural habitat.

1 Introduction

Freshwater ecosystems provide vital services for human existence but are on top of the world's most threatened ecosystems (Dudgeon et al., 2006; Revenga et al., 2000), primarily due to river damming (Allan and Castillo, 2007). The ability to provide sufficient water to ensure the functioning of freshwater ecosystems is an important concern as its capacity to provide goods and services is sustained by water-dependent ecological processes (Acreman, 2001). The relevance of this subject compelled the scientific community to appeal to all governments and water-related institutions across the globe to engage in environmental flow restoration and maintenance in every river (Brisbane Declaration, 2007). Truly, this issue is a global reach topic, as all dams, weirs and levees change the magnitude of peak flood flows of rivers to a certain extent (e.g., Nilsson and Berggren, 2000; FitzHugh and Vogel, 2010; Maheshwari et al., 1995; Miller et al., 2013; Uddin et al., 2014a, b). Therefore, there are still opportunities for the implementation of environmental flow restoration at hundreds of thousands of these structures worldwide (Richter and Thomas, 2007).

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 they also play an essential role in the conservation of freshwater ecosystems (Arthington et al., 2006; Hughes and Rood, 2003). It is now in agreement that environmental flows must ideally be based on the ecological requirements of different biological communities (Acreman et al., 2009) and should present a dynamic and variable hydrological regime to maintain the native biodiversity and the ecological processes that portray every river (Bunn and Arthington, 2002; Lytle and Poff, 2004; Postel and Richter, 2003). In this sense, holistic methodologies are clearly being increasingly applied out of Australia and South Africa (Hirji and Davis, 2009), but the most commonly applied methods throughout the world are still hydrologically based methods (Dyson et al., 2003; Tharme, 2003; Linnansaari et al., 2012). Conversely, environmental flows ascertained through habitat simulation methods still persist generally based on the requirements of a single biological group, mostly fish (Acreman et al., 2009; Tharme, 2003; Arthington, 2012), and clearly require an input from less typically monitored taxa (Gillespie et al., 2014). Accordingly, these biased approaches still disregard the inter-annual flow variability that rules species with longer lifecycles, therefore lacking the long-term perspective of the riverine ecosystem (Stromberg et al., 2010). The feedbacks of these shortcomings on the riparian and aquatic communities were seldom estimated before and so, the efficiency of such approaches along with its long-term after-effects remains practically unknown.

Riparian restoration is an indispensable implementation measure to recover the natural river processes and is the most promising restoration action in many degraded rivers (Palmer et al., 2014). Moreover, riparian vegetation is a suitable environmental change indicator (Benjankar et al., 2012; Nilsson and Berggren, 2000) that responds directly to flow regime in an inter-annual timeframe (Capon and Dowe, 2007; Naiman et al., 2005; Poff et al., 1997) and has a clear significance in the habitat improvement of aquatic systems (e.g., Broadmeadow and Nisbet, 2004; Pusey and Arthington, 2003; Van Looy et al., 2013). Hence, incorporating riparian vegetation requirements (the need for specific flows to preserve the naturalness of recruitment and meta-stability facing fluvial processes) into environmental flows could be an important contribution to fill in these gaps.

The purpose of this study is to evaluate the effect of disregarding riparian vegetation requirements in the efficiency of environmental flow regimes regarding fish habitat availability in the long-term perspective of the fluvial ecosystem. We used an approach from an ecohydraulic point of view to evaluate the effects of riparian degradation on fish species. We were particularly interested in answering the following questions: are environmental flows exclusively addressing fish requirements capable of preserving the habitat availability of these 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?

To approach these questions, we first modeled the structural response of riparian vegetation facing a decade of different environmental flows in two different case studies. Next, we performed an assessment of habitat availability for fish species in each of the resulting riparian habitat scenarios. We are not aware of such a modeling approach ever being used in the appraisal of the long-term efficiency of environmental flow regimes, which can provide an extremely valuable insight of the expected long-term effects of environmental flows in river ecosystems in advance.

2 Methods

2.1 Study sites

The two study sites were selected in the Ocreza River, East Portugal (Figure 1). This is a medium-sized stream that runs on schistose rocks for 94 km and drains a 1429 km^2 watershed with a mean annual flow of 16.5 m^3 s^{-1}. The flow regime is typically Mediterranean, with a low flow period interrupted by flash floods in winter and a very low flow, even null at times, during summer (Gasith and Resh, 1999). Two study sites were considered (OCBA and OCPR) to provide a broader analysis of the aquatic habitat modifications in different hydrogeomorphological contexts. The OCBA study site (39° 44’ 07.05” N, 7° 44’ 16.51” W) is located 30 km upstream from the river mouth and OCPR (39° 43’ 16.88” N, 7° 46’ 01.05” W) is approximately 5 km downstream of OCBA. Despite the relatively small distance between them, several characteristics differentiate the two study sites. While in OCBA, the river flows freely on a boulder substrate and is confined to steep valley hillsides, in OCPR, the river flows on a coarser boulder substrate with sparse bedrock presence and is in a relatively wider valley section. OCBA and OCPR also contrast in watershed areas, representing 54 and 72% of the entire river basin, respectively. This feature further differentiates the two case studies, as the intermediate watershed of OCPR collects water from a much rainier zone, therefore conferring an increased flow regime in this study site. The surveyed areas in the OCBA and OCPR study sites encompass a river length of approximately 500 and 300 m, respectively, laterally limited by the 100-year flooded zone, thus totaling approximately 4 and 3 ha for OCBA and OCPR study sites, respectively. In both cases, the 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 of 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 OCBA and 25132 at OCPR. 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 as a series of points along several cross-sections in the study sites. Depths were measured with a ruler and water velocities with a flow probe (model 002, Valeport) positioned at 60% of the local depth below the surface (Bovee and Milhous, 1978). Additionally, the substrate composition was visually assessed and mapped to determine posteriorly the effective roughness heights of the riverbed. These data were used to calculate river discharge in each study site and to calibrate the model. Additional information about hydraulic data and channel bed characteristics is provided as supplementary material.

2.2.2 Riparian vegetation data

The riparian vegetation was assessed in 2013 to support the calibration and validation of 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 patches were later classified by succession phase according to its corresponding development stage. Patch georeferencing, patch aging and succession phase classification followed the methodology used by Rivaes et al. (2013).

Five succession phases were identified in the study sites: 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 gravel bars, sometimes covered by herbaceous vegetation but without woody arboreal species. The patches dominated by the recruitment of woody arboreal species 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, such as willows and alders (*Alnus glutinosa*). Older patches dominated by macrophanerophytes, such as ash-trees, were considered to be 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 the characterization of succession phases is provided as supplementary material.

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 preferences of stream fishes in order to reflect their optimal habitat (Gorman and Karr, 1978). Sampling occurred 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 (the gudgeon *Gobio lozanoi*) occurred in the study area, but in very low density. Field procedures followed those by Boavida et al. (2011, 2013a). Ensuing fish sampling, microhabitat measurements of flow depth (cm), mean water velocity (cm s^{-1}) and dominant substrate composition were taken in 0.8 x 0.8 m quadrats at the location where each fish was captured. 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. To develop Habitat Suitability Curves (HSC) for target fish size classes, microhabitat variables (flow depth, water velocity, dominant substrate and cover) were divided into classes, and histograms of frequencies of use and availability were constructed (Boavida et al., 2011). A summary on collected fish

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data, as well as data analysis to determine habitat use, availability and preference of fish species regarding the analyzed variables, is provided as supplementary material.

2.3 Flow regime definition

Three flow regimes were considered for the modeling of riparian vegetation: 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 (Ferreira et al., 2014), with a projected location immediately upstream of OCBA. This environmental flow regime considers both fish and riparian requirements, which are presented in a multiannual fashion for a decadal time frame while 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 the requirements of riparian vegetation. The flow regime addressing fish requirements is built on a monthly basis to embody the intra-annual variability ruling the main life cycle events of this biological group (Encina et al., 2006; Gasith and Resh, 1999). Likewise, the flushing flows of the riparian flow regime intend to characterize the inter-annual flow variability to which the arrangement of riparian vegetation communities respond (Hughes, 1997).

The Eflow was determined according to the Instream Flow Incremental Methodology (Bovee, 1982) and aimed for 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; Santos et al., 2005) 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 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 (Rivaes et al., 2015). Thus, the Eflow regime only acknowledges the mean monthly discharges, while the Eflow&Flush regime incorporates both mean monthly discharges and flushing flows (Figure 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 succession dynamics of riparian vegetation, based on the existing relationships of the ecological relevant hydrological elements (Poff et al., 1997) and the vegetation metrics that reflect riparian communities to such hydrological alterations (Merritt et al., 2010). The rational of this model is based on the fact that riparian communities respond to the hydrological and habitat variations on a time scale between the year and the decade (Frissell et al., 1986; Thorp et al., 2008), being that the flood pulse is the predominant factor on these population dynamics (Thoms and Parsons, 2002). For these reasons, the hydrological 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. Notwithstanding, the model also predicts the annual riparian

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adjustments according to its vital rates in relation to groundwater depth, as well as the annual recruitment areas, based on the annual minimum mean daily discharges.

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 to be in good agreement with the observed riparian landscape (Altman, 1991; Viera and Garrett, 2005). The riparian vegetation model was further validated in this specific watershed (Ferreira et al., 2014). After calibration (calibrated parameters provided as supplementary material), the riparian vegetation was modeled for periods of ten years according to the corresponding 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 in each study site.

2.5 Hydrodynamic modeling of fish habitat

The hydrodynamic modeling was performed using a calibrated version of the River2D model (Steffler et al., 2002). This 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 2D 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 calibration procedure followed the methodology proposed by Boavida et al. (2013b, 2015) and calibrated parameters are provided in supplementary material. The hydrodynamic modeling comprised the Eflow discharge ranges in the study sites (0 – 2 $m^3 s^{-1}$ and 0 – 5.5 $m^3 s^{-1}$ for OCBA and OCPR, respectively) and was accomplished for each riparian habitat. The riparian habitats were represented in the hydrodynamic model by changing the channel roughness according to the spatial extent of the riparian succession phases. Roughness is a critical feature influencing the physical variables of flow hydraulics (Curran and Hession, 2013; Chow, 1959), whose distinct combinations typify diverse functional habitats, which are selected by fish according to its preference. The roughness classification of riparian vegetation succession phases was determined based on roughness measurement literature on similar vegetation types (Chow, 1959; Wu and Mao, 2007) and expert judgment during model calibration. The hydraulic characteristics of each habitat (roughness, flow depth and velocity) were compared using a t-test (confidence level of 99%) 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) with preference curves information for the considered target species. The riverbed characteristics of substrate and cover were kept unchanged during the hydrodynamic modeling. Changing the substrate according to the modifications in succession phase disposal seemed to be an incorrect practice in this case because during data treatment, no significant differences were detected in riverbed substrate between succession phases. Cover modification was also disregarded because the CASiMiR-vegetation model only reproduces the riparian area, not the aquatic zone (part of the river channel that is permanently submerged) and therefore, this feature cannot be correctly modeled by the riparian vegetation model. Notwithstanding, the most important variables determining fish habitat availability influenced by riparian vegetation degradation were considered, namely, depth, velocity and substrate (Parasiewicz, 2007).

The Habitat Suitability Index (HSI) was determined for each species and 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):

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

The product of the HSI by the influencing area (A) of the corresponding model ith 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):

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

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, while deviations were measured using the most commonly used measures of forecast accuracy, namely, Root Mean Square Deviation (RMSD), Mean Absolute Deviation (MAD) and Mean Absolute Percentage Deviation (MAPD). In all cases, smaller values of these measures indicate better performance in parameter estimation.

3 Results

Different configurations of riparian habitat resulted from the riparian vegetation modeling according to the considered flow regimes in both case studies (Figure 3). Nonetheless, the modeled response of riparian vegetation to each flow regime is similar in the two study sites. The riparian habitat, driven by the natural flow regime, presents a river channel that is largely devegetated, where Initial (IP) and Pioneer (PP) phases together represent approximately 43% and 35% of the study site areas in OCBA and OCPR, respectively. In this habitat, Early Succession Woodland phase (ES) can only settle in approximately 8% of OCBA and 1% of OCPR areas. The floodplain succession phases, namely, Established Forest phase (EF) and Mature Forest phase (MF), represent nearly 40 and 10% of the study area for OCBA and, close to 42% and 23% for OCPR, 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 toward mature phases due to the lack of the river flood disturbance. IP is now reduced to approximately 3% in OCBA and 6% in OCPR, while PP is inexistent in both cases. ES covers up to approximately 48% and 26% of the corresponding study areas, whereas EF and MF maintain about the same area in both case studies. The riparian habitat driven by the Eflow&Flush regime shows the capacity of this flow regime in hold back vegetation encroachment in both cases. In this habitat, IP and PP are maintained at approximately 30% of the study site area in both case studies, whereas ES is kept under 21% in OCBA and only 2% in OCPR. Once again, EF and MF preserve their areas in both case studies.

The changes undertaken by the riparian vegetation facing different flow regimes are able to modify the hydraulic characteristics of the river stretches (Figure 4). Channel effective roughness heights (k_s) change dramatically according to the considered riparian habitats, increasing proportionally to the encroachment level of vegetation in the study sites. In both case studies, the k_s values of the Eflow habitats are clearly distinct and higher compared to the other two habitats. The k_s in the Eflow&Flush habitats were found to be between the values of Eflow and natural habitats, and were very similar with the natural habitat in the case of OCPR. Notwithstanding, in both case studies, the k_s mean values are significantly different between all three habitats (test results in supplementary material). The mean k_s of the Eflow, Eflow&Flush and natural habitats are 0.999, 0.709 and 0.462 m, respectively, in OCBA, and 1.034, 0.742 and 0.7178 m, respectively, in OCPR.

Changes also occur in flow depth and flow velocity for the considered discharge range of the proposed environmental flows. In OCBA, the Eflow habitat creates a circumstance with significantly higher depths (mean depth is 0.402 m) and lower flow velocities (mean flow velocity is 0.128 m s^{-1}) than the natural and Eflow&Flush habitats (test results in supplementary material). In contrast, depth and flow velocity are not significantly distinguishable between the natural and Eflow&Flush habitats, where mean depth and flow velocity are 0.397 m and 0.136 m s^{-1}, respectively, in the former, and 0.399 m and 0.135 m s^{-1}, respectively, in the latter. For the OCPR study site, flow depths are not significantly different (mean values of flow depth for Eflow, Eflow&Flush and natural habitats are 0.420, 0.417, 0.418, respectively) but flow velocities are; with the Eflow habitat creating significantly lower flow velocities (0.271 m s^{-1}) compared to the significantly indistinct Eflow&Flush (0.277 m s^{-1}) and natural (0.278 m s^{-1}) habitats (test results in supplementary material).

During a hydrological year, each riparian habitat provides different WUAs for the target fish species, with the same environmental flow regime addressing fish species (Figure 5). Differences from the natural habitat suitability are greater in the Eflow habitat for both case studies. In OCBA, 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. Compared to the natural habitat, the WUA modifications instilled by the Eflow habitat are on average approximately 12%, and are higher than 17% in a quarter of the cases and can reach 80% in an extreme situation. Particularly, the Eflow habitat provides less habitat suitability during autumn and winter months for the barbel and nase juveniles, c. 17% and 14%, respectively. Likewise, in this habitat, the habitat suitability during spring months increases approximately 23% for the barbel juveniles and approximately 20 and 27% for the calandino juveniles and adults, respectively. On the other hand, throughout the year, the Eflow&Flush habitat provides a WUA very similar to the natural habitat. The habitat changes created by the Eflow&Flush habitat are on average approximately 2% and never reach 8% for all species and life stages.

As for OCPR, major differences in WUA are seen almost all year round for calandino and nase, and exist particularly in spring months for barbel. WUA modifications due to the Eflow habitat are on average near 29%, being a quarter more than 50% and reaching up to more than 100% different in the most extreme case. The Eflow habitat consistently provides less habitat suitability during autumn and winter months for the barbel and nase, c. 50% and 38%, respectively, while the habitat suitability increases in approximately 46% of calandino. Moreover, the Eflow habitat provides an increased WUA during spring months in approximately 18% of the barbel adults and 71% of the calandino adults, while it decreases the habitat on average for approximately 7% of the remaining species and life stages. Also in this case study, the Eflow&Flush habitat provides a WUA very similar to the natural habitat throughout the year. The habitat changes created by the Eflow&Flush habitat are on average near 3% and always less than 17% for all species and life stages. Accordingly, in both case studies, the WUA differences evidenced in the Eflow habitat revealed to be significant in several months by the χ^2 test whereas this were never the case for the Eflow&Flush habitats (test results provided in supplementary material).

The riparian-induced modifications on the WUAs are also confirmed by all the employed deviation measures (Table 2). According to RMSD, MAD and MAPD, the Eflow habitat is always farther apart from the natural habitat for all species and life stages. In OCBA, the larger deviations occur for the barbel juveniles and nase adults, whereas in OCPR, the calandino adults and the barbel juveniles are the ones enduring greater habitat deviations from the natural circumstance.

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 suitability. To this end, the riparian vegetation was modeled for 10-year periods according to three different simulated flow regimes and results were inputted as the habitat basis for the hydrodynamic modeling and subsequent assessment of the fish habitat suitability in those riparian habitats. Such ecological modeling approach, where a joint analysis is performed while embracing a suitable time response for the ecosystems involved, pushes through realistic biological-response modeling and substantiates the long-term research that is required in environmental flow science (Arthington, 2015; Petts, 2009). Furthermore, this approach allows one to foresee and assess the outcome of recommended flow regimes, which is an essential topic but has been poorly considered in environmental flow science (Davies et al., 2013; Gippel, 2001). This research provides an insight of the expected long-term effects of environmental flows in river ecosystems, therefore unveiling the potential remarkable role of riparian vegetation on the support of environmental flows efficiency, which can transform the actual paradigm in environmental flow science.

The results of the vegetation modeling illustrate how the natural flow regime generates morphodynamic disturbances, without which the riparian vegetation is able to settle and age in the river channel. Consequently, of the latter, the microhabitat analysis demonstrated that changes in the riparian habitat induce modifications in the hydraulic characteristics of the river stretches. The differences in mean values of these parameters are subtle between habitats but are statistically significant. A detailed analysis using a pairwise comparison of flow depths and velocities between scenarios show that modifications can reach 10 cm in water depth and more than 40 cm s^{-1} in flow velocity. Such change can shift the habitat preference of fishes in one or two classes of the corresponding habitat preference curves. These changes are particularly important considering that an alteration of one class regarding these parameters is sufficient to change fish preferences from near null to maximum and vice-versa in many cases, as it can be seen in the preference curves provided in the supplementary material.

The hydrodynamic modeling also indicated changes directly affecting the habitat suitability of the existing fish species according to the riparian habitat. Through time, the habitat shaped by the Eflow regime diverged substantially in habitat suitability from the natural and Eflow&Flush habitats, and there were cases where the habitat suitability was modified by more than double.

The relationship between fish assemblages and habitat has long been acknowledged (e.g., Matthews, 1998; Clark et al., 2008; Pusey et al., 1993) and can have a significant impact on the ecological status and function of the existing fish communities (Freeman et al., 2001; Jones et al., 1996; Randall and Minns, 2000). Effectively, habitat loss is the major threat concerning fish population dynamics and biodiversity (Bunn and Arthington, 2002), thereby promoting population changes with a proportional response to the enforced habitat change (Cowley, 2008). This is particularly true for the fish species considered in this study (Cabral et al., 2006). The habitat decrease for barbel and nase 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 river and Mediterranean rivers in general. On the other hand, the habitat change during spring months undermines the spawning activity and consequently the sustainability of future population stocks (Lobón-Cerviá and Fernandez-Delgado, 1984). The habitat increase of calandino during this period can be ecologically tricky due to the habitat plasticity of this species (Doadrio, 2011; Gomes-Ferreira et al., 2005), as well as its characteristic adoption for an r-selection strategy as an evolutionary response to frequently disturbed environments (Bernardo et al., 2003). Above all, one should not ignore that the relationship 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).

Our results indicate that environmental flows taking into account riparian vegetation requirements are able to preserve the naturalness of the riparian habitat and consequently, the maintenance of the fish habitat suitability. Accordingly, the implementation of such measure can provide significant positive ecological effects in downstream reaches (Lorenz et al., 2013; Pusey and Arthington, 2003) and results in additional ecosystem services (Berges, 2009; Blackwell and Maltby, 2006) while imposing minor revenue losses to dam managers (Rivaes et al., 2015). The implementation of such environmental flows could provide an additional way to attain the "good ecological status" required by the Water Framework Directive (WFD). In addition, taking up a procedure such as this one 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).

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, the outcomes of this study are a product of long-term simulations by models that were calibrated and validated for the corresponding watershed with local data in natural river flow conditions. This standard procedure in modeling strengthens confidence in our predictions as the models proved to correctly replicate the response of the riparian and fish communities when paralleled with simultaneous observational data.

In conclusion, we predict a change in fish habitat suitability according to the long-term structural adjustments that riparian habitats endure following river regulation. These changes can be attributed to the effects that altered riparian habitats have on the hydraulic characteristics of the river stretches. In our view, environmental flow regimes considering only the aquatic biota are expected to become obsolete in few years due to the alteration of the habitat premises in which they were based. This situation points to the unsustainability of these environmental flows in the long-term perspective of the fluvial ecosystem, failing to achieve the desired effects on aquatic communities to which those were proposed in the first place. An environmental flow regime that simultaneously considers riparian vegetation requirements contributes to the preservation of the hydraulic characteristics of the river channel at the natural riverine habitat standards, therefore maintaining the habitat assumptions that support the environmental flow regimes regarding aquatic communities. Consequently, accounting for riparian vegetation requirements poses as an essential measure to assure the effectiveness of environmental flow regimes in the long-term perspective of the fluvial ecosystem.

Data availability

Riverbed topography, hydraulic measurements, riparian vegetation and fish sampling were collected by the authors and are available at http://home.isa.utl.pt/~ruirivaes/Data_availability/HESSD/. Both River2D and CASiMiR-vegetation models are freeware available at http://www.river2d.ualberta.ca/download.htm and http://www.casimir-software.de/ENG/download_eng.html, respectively.

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

	OCBA			OCPR		
Year	natural	Eflow	Eflow&Flush	natural	Eflow	Eflow&Flush
1	671	0.99	0.99	951	5.51	5.51
2	203	0.99	167	287	5.51	237
3	327	0.99	0.99	464	5.51	5.51
4	217	0.99	167	308	5.51	237
5	316	0.99	0.99	449	5.51	5.51
6	371	0.99	167	526	5.51	237
7	702	0.99	0.99	995	5.51	5.51
8	202	0.99	167	286	5.51	237
9	195	0.99	0.99	276	5.51	5.51
10	440	0.99	371	624	5.51	527

Hydrol. Earth Syst. Sci. Discuss., doi:10.5194/hess-2017-57, 2017
Manuscript under review for journal Hydrol. Earth Syst. Sci.
Discussion started: 21 March 2017

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). Values stand for the habitat availability deviation, in area and percentage, of the environmental flow regimes compared to the natural habitat availability of each species and life stage.

	OCBA study site						OCPR study site					
	Eflow			Eflow&Flush			Eflow			Eflow&Flush		
	RMSD (m^2)	MAD (m^2)	MAPD (%)	RMSD (m^2)	MAD (m^2)	MAPD (%)	RMSD (m^2)	MAD (m^2)	MAPD (%)	RMSD (m^2)	MAD (m^2)	MAPD (%)
Luciobarbus bocagei (juv.)	86.00	72.10	15.40	12.17	7.24	2.52	26.23	17.37	35.55	2.51	1.50	0.63
Luciobarbus bocagei (adult)	29.46	20.55	5.83	2.87	2.12	1.55	12.94	7.73	23.15	3.44	1.79	3.01
Pseudochondrostoma polypepis (juv.)	128.21	86.14	11.58	9.42	5.72	2.26	45.42	32.71	34.43	1.55	0.92	2.51
Pseudochondrostoma polypepis (adult)	7.32	5.85	18.70	2.17	1.37	2.10	9.00	7.00	10.34	0.51	0.35	2.42
Squalius alburnoides (juv.)	44.05	28.16	8.46	6.20	4.06	2.10	33.10	27.78	28.37	2.44	1.35	2.18
Squalius alburnoides (adult)	92.41	52.47	10.23	7.49	5.31	2.37	61.76	47.83	40.54	0.96	0.63	2.90

Hydrol. Earth Syst. Sci. Discuss., doi:10.5194/hess-2017-57, 2017
Manuscript under review for journal Hydrol. Earth Syst. Sci.
Discussion started: 21 March 2017

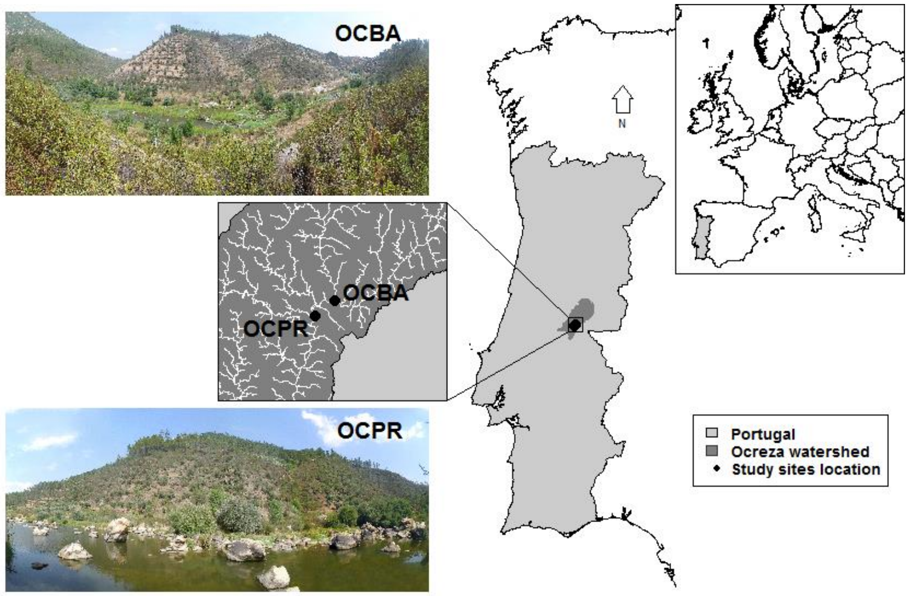

Figure 1. Location and characterization of the study sites OCBA and OCPR.

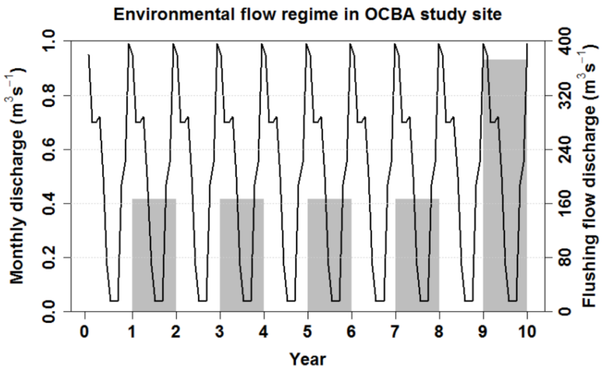

Figure 2. Environmental flow regime addressing fish (black line, left axis) and riparian (grey bars, right axis) requirements considered for the habitat modeling in OCBA study site. Fish requirements are addressed by a constant monthly discharge and riparian requirements by a flushing flow in the years in which are planned 5 (duration of the flushing flow is similar to a natural flood with equal recurrence interval). The hydrograph for the Eflow&Flush flow regime is similar in the OCPR study site.

Hydrol. Earth Syst. Sci. Discuss., doi:10.5194/hess-2017-57, 2017
Manuscript under review for journal Hydrol. Earth Syst. Sci.
Discussion started: 21 March 2017

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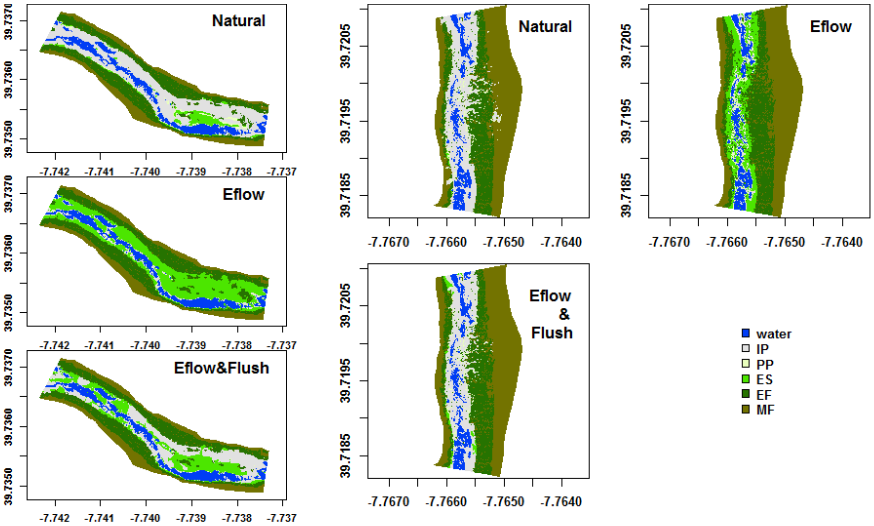


Figure 3. Expected patch mosaic of the riparian vegetation habitats shaped by the natural, Eflow and Eflow&Flush flow regimes (detailed by succession phase, namely, initial phase – IP, pioneer phase – PP, early succession woodland phase – ES, established forest phase – EF and mature forest phase – MF) in the OCBA study site (on the left) and in the OCPR study site (on the right).

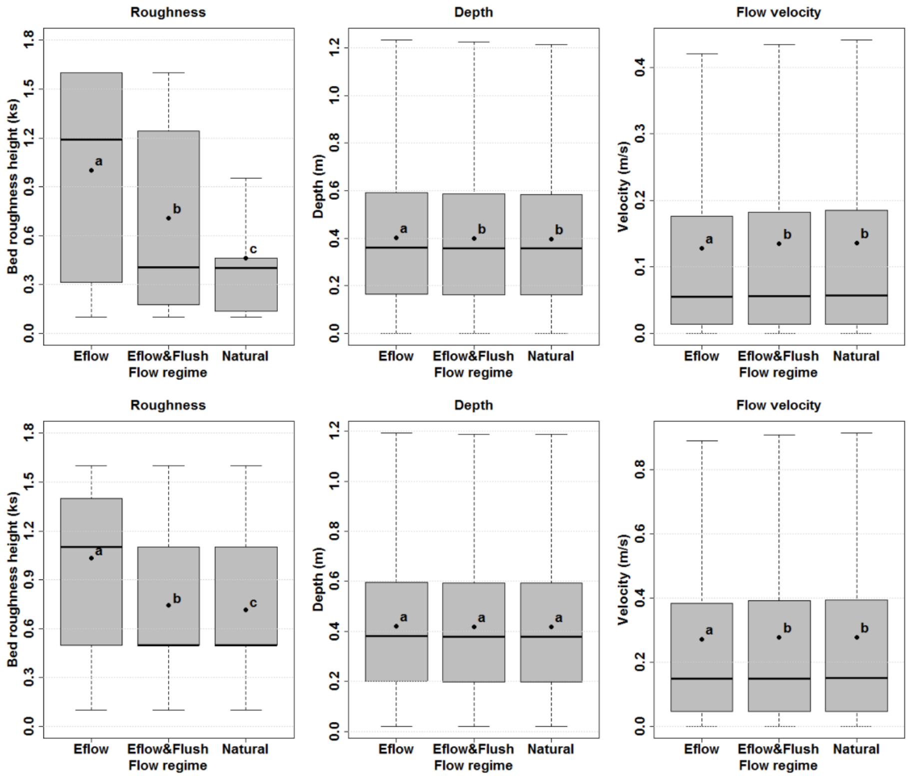
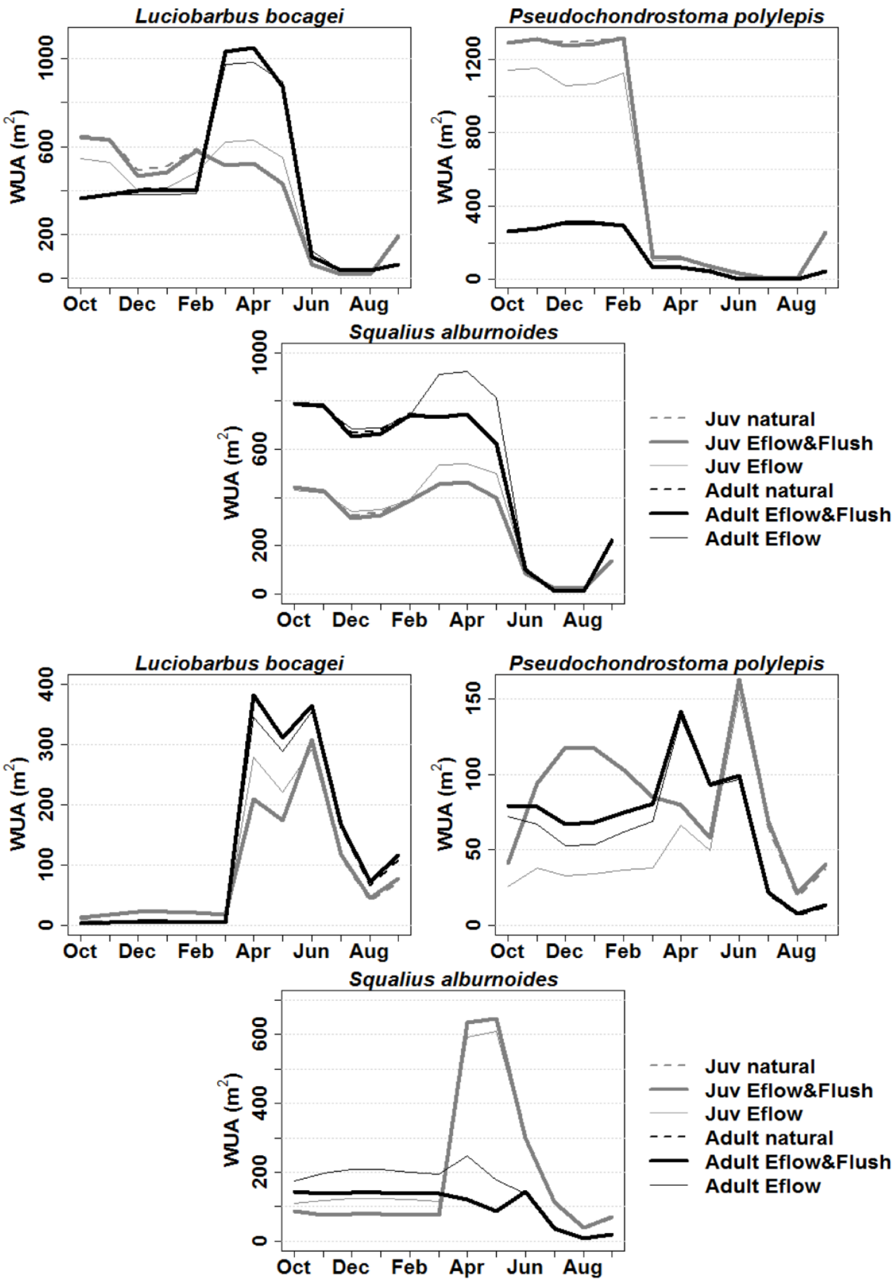


Figure 4. Hydraulic characterization of OCBA (top) and OCPR (bottom) according to the different expected riparian vegetation habitats driven by the Eflow, Eflow&Flush and natural flow regimes (data obtained from 2D hydrodynamic modeling). Different letters stand for significant differences between groups (t-test). Boxplots portray non-outlier value range, thick black lines the median value and black dots the mean values.



5 **Figure 5. Fish weighted usable areas provided by the fish-addressed environmental flow regime (Eflow) flowing through the different riparian habitat scenarios originated by a decade of three different flow regimes (natural, Eflow&Flush and Eflow).**